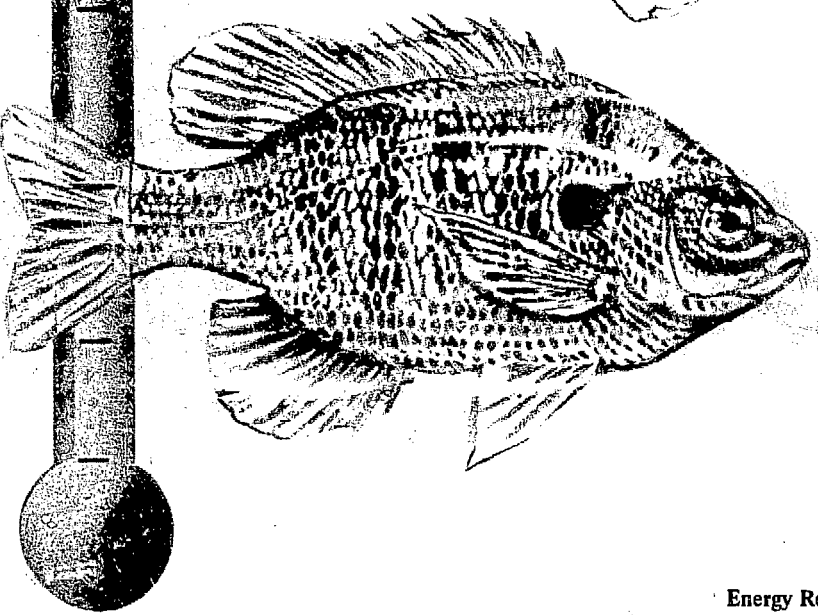
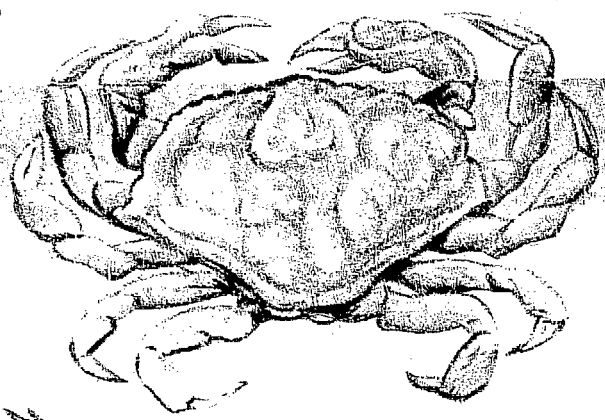


# Thermal Ecology II

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# Thermal Ecology II

Proceedings of a symposium held at Augusta, Georgia  
April 2<sup>nd</sup> - 5, 1975

*Sponsored by*

Savannah River Ecology Laboratory  
Institute of Ecology, University of Georgia

Division of Biomedical and Environmental Research  
Energy Research and Development Administration

*Edited by*

Gerald W. Esch  
Wake Forest University

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Savannah River Ecology Laboratory

1976

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# Preface

The escalating demands for electrical power and the concomitant requirement to minimize the environmental impact of generator-cooling systems has assured continuing interest in thermal ecology. In response to the need for more rapid dissemination of the knowledge being assembled, the second Thermal Ecology Symposium was held at Augusta, Ga., Apr. 2-5, 1975. The symposium was jointly sponsored by the Savannah River Ecology Laboratory; the Institute of Ecology, University of Georgia; and the United States Energy Research and Development Administration. The extent of the burgeoning interest in this field is reflected in comparisons of this symposium with the first Thermal Ecology Symposium held in 1973. The number of participants increased 20%, to more than 300, and the number of scientific presentations increased 42%. Eighty of the 105 papers presented were submitted to the editors, and 57 were selected for inclusion in this volume.

This volume commences with the four invited papers at the symposium. Eugene P. Odum's contribution is based on his banquet address proposing the adoption of a unique power-park concept. Victor H. Hutchison, J. Whitfield Gibbons, and John Cairns, Jr., who were requested to provide a review of thermal ecology at the individual, population, and ecosystem levels, respectively, responded with outstanding and provocative essays. The bulk of the volume is comprised of the contributed papers of the symposium. These have been grouped into related topics to provide continuity for the reader. As with any task of this nature, however, a certain degree of subjectivity is required, and numerous papers could have been placed into several categories with

equal justification. We urge the reader not to limit his initial perusal to a narrow selection for we are certain there are contributions of interest in all sections.

The value of any symposium proceedings depends on the quality of the contributed papers. In the interest of "quality control," each manuscript was reviewed by two scientists in the appropriate field of specialization. Insofar as possible, the reviewer was unaware of the author's identity, a situation we feel contributes to more objective and critical evaluation. We came to value an excellent review (whether positive or negative) with fervor usually reserved for an excellent manuscript. The final judgment of and the associated responsibility for all manuscripts, of course, lies with the editors.

It goes without saying that temperature is one of the most significant regulating factors in any aquatic ecosystem. Wherever excessive thermal effluent is dispersed into an ecosystem, there will be an impact. In some instances the impact will be minor, virtually unnoticeable, or perhaps even beneficial. In other situations, however, the conditions created will be intolerable, and the system will be irrevocably altered. We hope that the contributions herein will be of interest and significance to scientists pursuing the basic questions involved in thermal studies and to administrators directly responsible for managing our nation's natural resources. It is to these two groups of citizen-scholars that we dedicate this volume.

Gerald W. Esch  
Robert W. McFarlane  
*Editors*



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Preparing a good manuscript review is very time consuming, and the anonymity of the task does not provide acknowledgment. The diligence of our reviewers has rescued numerous authors from embarrassing oversight, or absence of insight, and perhaps salvaged manuscripts that otherwise may have been jettisoned. Thus it is with particular pleasure that we thank the following individuals for their assistance in reviewing manuscripts:

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# The Power Park Concept: Ameliorating Man's Disorder with Nature's Order

Eugene P. Odum  
Roger L. Kroodsma

As the age of petroleum draws to a close, and with it the era of cheap energy, nearly all optimistic projections for technological growth must be reassessed. Without an abundant source of cheap energy, such grandiose schemes as planned cities, extraction of minerals and freshwater from the sea, energy-intensive agriculture as a means of feeding the world, etc., are going to be difficult to carry out and in some cases impossible at an economic and environmental price we can afford to pay. We are clearly entering an era, perhaps a fairly short one, of expensive energy which will dictate a profound change in our life-styles and will require a reduced growth rate in the production of hard goods and a corresponding increase in maintenance and consumer-oriented businesses. On a global scale man is face to face with the problem of developing a survival strategy based on a more symbiotic and less disorderly linkage between the fuel-powered systems of man and the solar-powered system of nature, which provide the essential life-support module for spaceship earth. Now these are not doomsday words! Much good for the human spirit will come from a period of history devoted to the consolidation of gains already made by applying science and technology. Although there is a rough correlation between standard of living and energy consumption, a high standard of living can certainly be maintained at a somewhat lower per capita energy consumption than is now current in the United States. In fact, we can make a strong case for the proposition that a modest reduction in per capita energy consumption, if accompanied by an increase in efficiency of conversion and use, would actually raise the quality of human life. Saving human lives on the highways and reducing pollution-related cancer would alone make such a strategy worthwhile. Certainly, there is no scientific evidence that a doubling of per capita energy consumption, as once was confidently predicted to occur every 10 years or so, would automatically raise the standard of living for the average person.

We might well think of this period of consolidation as one of rest and relaxation following a period of almost warlike exploitation of natural resources in hectic pursuit of materialistic wealth. It is almost a natural law that growth booms must be followed by periods of consolida-

tion, in which negative feedback control is developed to prevent excesses and time is taken to relieve pressures created during the expansion. We must continue to dream dreams of brave new technological worlds and to make plans for the day when cheap energy might again be available. But we must also remove a huge backlog of disorder accumulated during the growth boom of the past half-century. By disorder, I do not mean only waste heat from energy conversion, the subject of this conference, but even more serious disorder associated with extensive contamination of air, water, and soil with toxic and carcinogenic chemicals; with crime and terrorism; with urban decay; and so on—all of which are by-products of the cultural lag that usually accompanies rapid technological growth. Putting the well-being of the individual and his life-support environment back into the technological formula is a goal well worth striving for.

## THE POWER PARK CONCEPT

The need for a thorough reassessment of our technological options for energy procurement is particularly urgent, especially in regard to atomic energy, which has failed to come up to expectations as a major alternative to fossil fuels—as was optimistically predicted at the first Atoms-for-Peace Conference in 1955—chiefly because of environmental considerations. Avoiding deleterious effects on the environment stemming from energy conversion becomes a major problem as the power density of urban-industrial development increases. I want to review with you the role that the Savannah River project can play in developing an essentially new technology devoted to ways and means of coupling man's energy-conversion processes with the natural and seminatural ecosystems so as to ameliorate the inherent disorder associated with any type of energy production. Not only is this an exciting new challenge but, in my opinion, it is also one of the things we

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must do to survive. For a while at least, we will not have the cheap energy necessary to carry out artificial tertiary treatment of wastes on a large scale. If energy-conversion and industrial processes can be redesigned to reduce poisonous and nondegradable wastes, natural systems can do much of the work of tertiary treatment for us free. This is, in fact, one of the most logical ways to use solar power since natural ecosystems are already constructed (i.e., no capital costs are required), are highly evolved, and are capable of assimilating and diluting large volumes of degradable waste. Using solar power in this manner becomes especially attractive in times of inflation, which, it is generally agreed, will be with us for a long time to come. As the cost of fuel increases because of the difficulty of obtaining it, we get less energy per dollar; more dollars must be spent to do the same work. Such a trend makes the natural environment worth more dollars since its proportionate role in creating a dollar's worth of gross national product is greater. Thus, if the cost of the water in a city doubles or the cost of tertiary treatment of waste doubles, the natural watershed that supplies the water or the swamp that receives the waste is worth twice as many dollars as before the inflation.

In theory, then, natural environments can function as efficient and inexpensive waste-treatment systems, if: (1) the industrial and energy-conversion processes are specifically designed so that donor wastes can be assimilated by recipient natural systems; (2) poisons that might damage the "living filter" are removed from the waste stream; and (3) there is sufficient diversity and stability in the natural systems to allow for adaptation or self-design processes to occur so that input perturbation becomes an integral part of the metabolism of the recipient systems. In other words, coupling man's fuel-powered system with nature's solar-powered system is a two-way street, an exercise in reciprocal design. In the past man has used natural systems, especially large rivers, estuaries, and the sea, as free sewers for dumping whatever wastes happened to be produced, with little or no regard for the consequences. The sheer volume and increasingly poisonous nature of man-made pollutants ruins out such a haphazard procedure for the future. We can no longer be preoccupied with one-at-a-time studies of the impact on the environment of a long list of by-products. In the jargon of the engineer, we can no longer afford to use nature as a "tail-end Charlie," like something that might be attached to the tail pipe of an automobile, as a kind of afterthought, to remove products of internal combustion of fuel. Rather the donor and recipient components must be considered together as an integrated system, with a redesign of the donor if necessary.

With this general theory in mind, we can define a power park as a designated area of the landscape or a complex of natural ecosystems set aside and managed for the assimilation of large volumes of dilute and degradable end products of energy conversion and urban-industrial activity. By adopting a design-with-nature management philosophy, which enhances the capacity of existing natural systems to

self-design, and by continuous monitoring to ensure that assimilative capacity is not exceeded, we could achieve essentially zero discharge into human living space. Such a park could even be a money-maker in terms of auxiliary products and services, which might, e.g., be associated with agricultural, horticultural, and forest products; recreation; water supply for cities; etc. The Savannah River reservation provides a wonderful laboratory for testing out such a plan on a realistic scale. In fact, the 300-sq mile reservation has been unofficially designated a National Environmental Research Park, whose mission includes research along these lines. Public announcement and dedication of the research park has been delayed by the reorganization of the Atomic Energy Commission and the formation of the Energy Research and Development Administration, but this need not delay our thinking about how the reservation could function as an experiment station for waste management. (After this paper was written, ERDA formally announced the designation of the Savannah River Plant as the nation's first National Environmental Research Park.)

## INVENTORY OF ECOLOGICAL SYSTEMS ON THE SAVANNAH RIVER PLANT RESERVATION

In 1974 we undertook an "Ecological Systems Inventory" of the Savannah River Plant (SRP) area for the specific purpose of evaluating the diversity of natural systems from the viewpoint of planning land use in the context of a National Environmental Research Park. This survey was intended to extend the inventories made by the Forest Service (Aydelott, no date; Beavers et al., no date; Forest Service, no date) to include considerations other than wood and fiber production.

The first phase of the ecosystems inventory consisted of determining the aerial extent, location, and diversity of easily identifiable macrohabitats with the use of a grid overlay on large-scale aerial photographs. A 6-mile-deep transect extending east-west across the SRP reservation, as shown in Fig. 1, was selected for detailed study. As shown in Fig. 2, the transect cuts across three of the four physiographic regions that represent successively lower (from northwest to southeast) coastal-plain terraces. The F and H separation areas, where studies on the fate of accidental releases of trace amounts of plutonium will begin in the near future, lie almost exactly in the middle of the transect. Blocks of land 2.5' latitude and longitude on a side (rectangular in shape) were chosen as the mapping units. The blocks are roughly 2.4 miles wide and 3 miles deep (19 by 23 in. in map dimensions, scale: 8 in. = 1 mile), covering an area of 4436 acres. The transect contains a total of 16 blocks, but four of them lie only partially within SRP. Figure 1 shows the blocks and their assigned numbers; they lie in two tiers, north (numbers 15 to 22) and south (numbers 24 to 31). The total area mapped within the SRP property (not including the Allendale corridor) is 102 sq miles, about one-third of the research park.

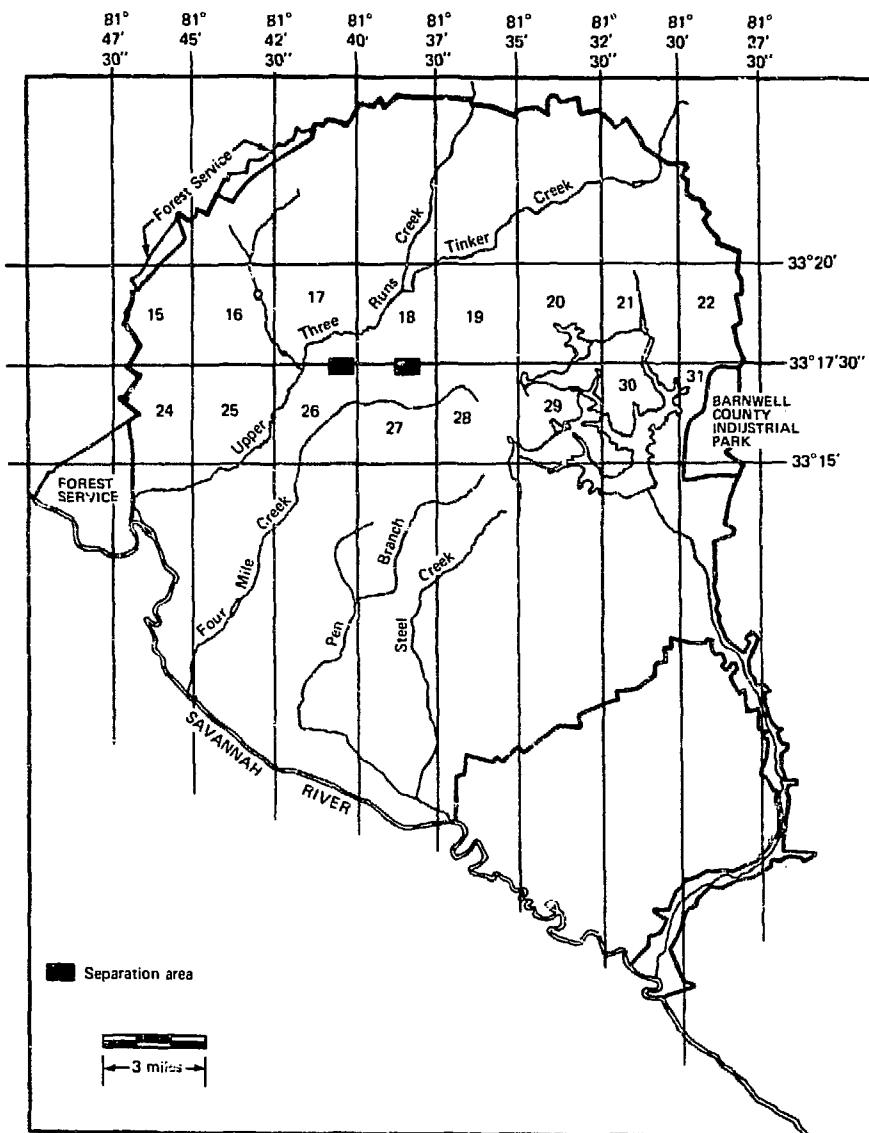


FIG. 1 The 65,447-acre transect across the Savannah River Plant. Numbered rectangles are the 2.5-square blocks in which habitat types were mapped.

In the first stage of data acquisition, maps (available at the Savannah River Ecology Laboratory) of habitats were prepared by tracing habitat boundaries from aerial photographs (scale: 8 in. = 1 mile). The photos, taken during May 1966, were the latest set available. An attempt was made to draw habitat boundaries in such a way that each upland habitat delineated was homogenous as far as could be determined from the aerial photos. Contiguous small pine plantations and contiguous bottomland mixed hardwood-pine forests differing in species, age, or density were often mapped as one stand, however, because the differences

were difficult to identify on aerial photos. In general, each mapped upland habitat other than pine is relatively homogenous in vegetation structure, but each deciduous bottomland habitat mapped may be quite heterogeneous.

After maps delineating the boundaries had been prepared, the habitat units were numbered consecutively, beginning at the upper left and ending at the lower right of each block (following the path taken by the eyes while reading a page). Each numbered habitat was then classified according to the list of habitat types and their respective numerical codes. Estimations of percent vegetational compo-



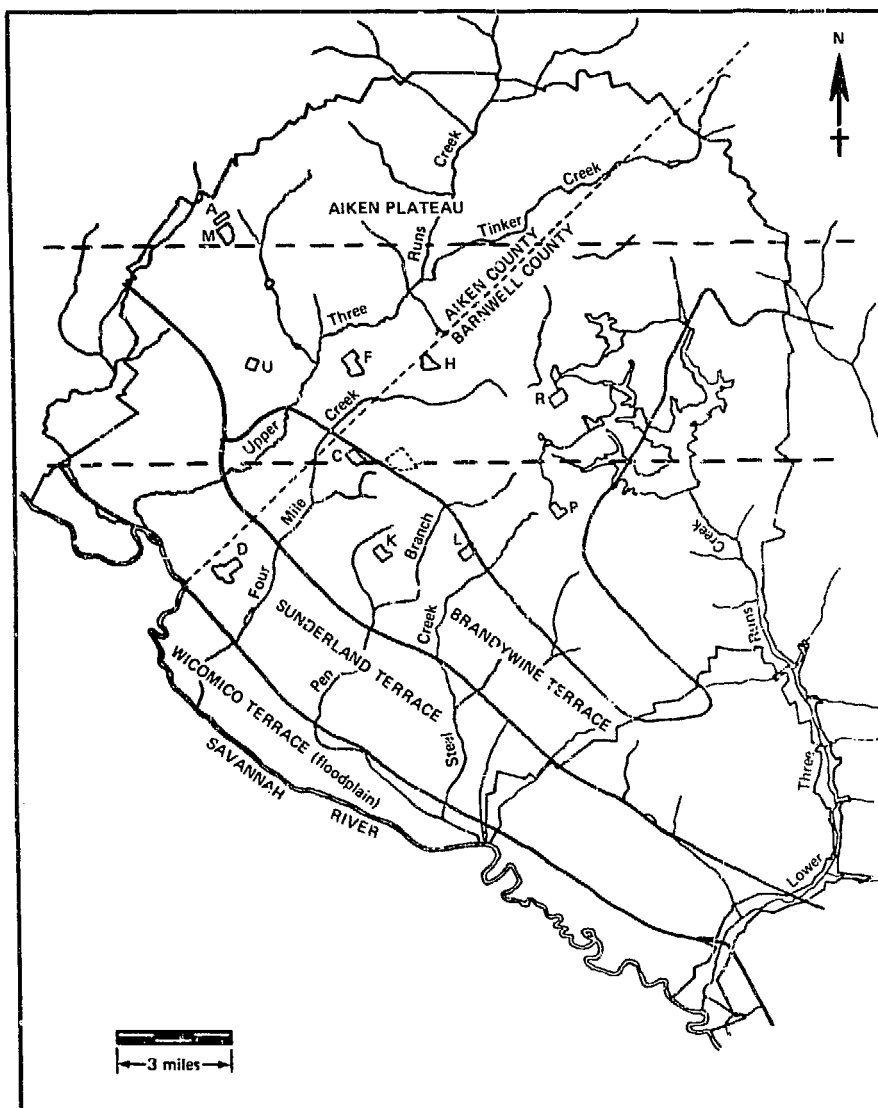


FIG. 2 Location of the transect with respect to the two main physiographic subregions, the Aiken Plateau and the Pleistocene coastal terraces (including the three terraces indicated). Lettered polygons are plant facilities, F and H are separation areas.

sition necessarily refer only to the vegetative cover that can be seen on aerial photos; e.g., a dense pine stand in which the pine canopy completely blocks from aerial view a dense hardwood understory would be classified as a pure pine stand. A second phase of the study will assess the understory and ground-level properties of the ecosystem types and in-habitat and microhabitat diversity within each of the major macrohabitats.

Then a grid of 625 dots was laid over each of the 16 maps representing the 16 2.5'-square blocks. The number of dots falling within each numbered unit (the terms unit, stand, and habitat are synonymous here) were recorded. The

following data were calculated by computer for each habitat type in each block:

1. Total number of units.
2. Total number of acres (each block is 4436 acres, and each dot represents 7.1 acres).
3. Percent of the total acreage (4436) covered.
4. Average size (total number of unit-type acres divided by the number of units).
5. Minimum size (usually zero since there were usually several units for each type which were so small that no dots occurred within their boundaries).

6. Maximum size (this could be zero for small units that occurred only a few times in a block and had no dots falling within their boundaries).

7. The sum of squares ( $\Sigma y^2$ ), the sum of each observation squared, for use in statistical comparisons.

8. Variance.

9. Standard deviation.

10. Two standard errors of the mean.

11. Diversity indexes for each of the 16 blocks, calculated on the basis of ratios  $N_i/N$  or  $P_i$ , where  $N_i$  is number of acres in each habitat category, as shown in the tables in the Appendix, and  $N$  is the total number of acres. The Shannon index,  $H' = -\Sigma P_i \ln P_i$ ; the Pielou evenness index,  $e = H'/\ln S$  (where  $S$  is number of species), and several other indexes on an existing program were calculated.

Computer printouts for all data are contained in the full report, "Ecological Systems Inventory of the Savannah River Plant Reservation," copies of which are on file at the Savannah River Ecology Laboratory and the Institute of Ecology at the University of Georgia, Athens.

The percent occurrence of major habitat types is summarized in Table 1. Pine plantations now cover all, or

TABLE 1 Distribution of 27 Habitat Types\* on an East-West Transect of the SRP Reservation

| Habitat             | Distribution, % |
|---------------------|-----------------|
| Pine plantation     | 36              |
| Other pine (5)      | 12              |
| Pine-scrub oak      | 15              |
| Other mixed (4)     | 13              |
| Bottomland hardwood | 6               |
| Other hardwood (4)  | 3               |
| Impoundments (3)    | 3               |
| Facilities (3)      | 7               |
| Open (5)            | 4               |

\*Numbers in parentheses indicate the number of distinct habitat types in the broad category.

practically all, land that was cultivated in 1952 when the plant was established. In our first land-use survey in 1954, we estimated that 33% of the entire reservation was at that time in old fields; this percentage agrees closely with the 36% pine plantation in the sample transect. If the mixed habitats (listed as pine-scrub oak and other in the table) are considered 50% pine and 50% hardwood, approximately 62% of the entire transect is pine, 23% hardwood, 10% man-made, and 4% open. The open habitats include herbaceous and shrub vegetation and Carolina Bays, shallow sinks with aquatic or prairie-like vegetation. The man-made category includes Par Pond, other thermal ponds and canals, reactor sites, construction, and road and power-line rights-of-way. The 1-to-10 ratio of man-made to natural environment provides an interesting starting point for

modeling various mixes of high-powered donor and lower-powered recipient systems. The question to be researched is, How many acres of natural environment are required to assimilate without stress the tertiary waste products of a given fuel-conversion process?

The percent coverage of major vegetation-types in the various blocks of the transect is shown in Fig. 3. Pine plantations average 36% but make up as much as 50% of the coverage in some blocks and as little as 19% in block 18, where an undisturbed stream floodplain dominates the landscape. Bottomland timber decreases from

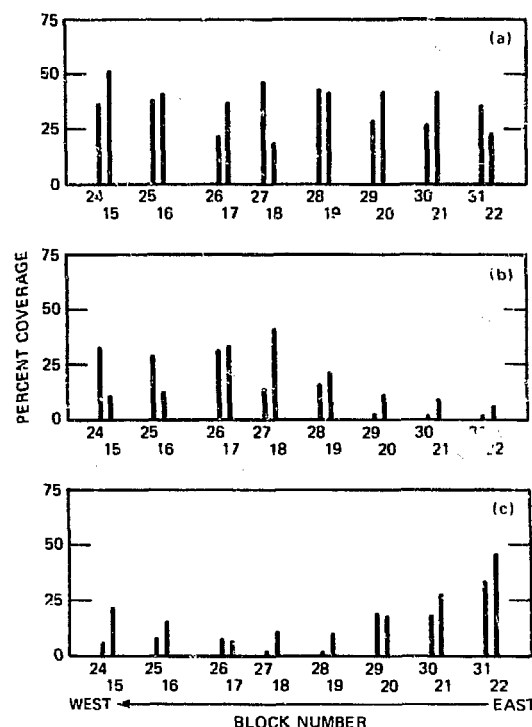


FIG. 3 Percent of total area covered by the three major habitat types for each block. The blocks are arranged so that west is to the left. The right and left lines in the eight pairs of lines for each habitat type represent the north and south blocks, respectively. (a) Pine plantation, average 36%. (b) Bottomland pine-hardwood and bottomland hardwood, average 10%. (c) Pine-hardwood scrub and hardwood scrub, average 15%.

west to east, and, conversely, upland pine-scrub coverage increases in the higher sand hills of the Aiken Plateau to the east. Much of what has been classified as pine-hardwood scrub is a unique association of long-leaf pine (*Pinus palustris*), turkey oak (*Quercus catesbaei*), and other dwarf oaks which is endemic to the sand-hills region extending as an intermittent strip just below the fall line (the junction between Piedmont and Coastal Plain physiographic provinces) from North Carolina through Georgia.

## LANDSCAPE PATTERN DIVERSITY

The diversity of major habitat types, estimated by calculating indexes based on ratios between number of acres in each habitat and total acreage sampled, is shown in Table 2. We can think of such calculations as indexes of

TABLE 2 Landscape Pattern Diversity on the SRP Transect

| Management plan                                              | Diversity index* |       |
|--------------------------------------------------------------|------------------|-------|
|                                                              | $\bar{H}$        | e     |
| Present situation, 36% pine plantation                       | 2.337            | 0.709 |
| Intensive forest management†<br>level 2: 50% pine plantation | 1.916            | 0.611 |
| Intensive forest management†<br>level 3: 75% pine plantation | 0.999            | 0.379 |

\*See text for discussion.

†Proposed by the Forest Service as possible alternatives for the future, as outlined in a report to the Atomic Energy Commission (Beavers et al., no date).

diversity of the landscape as a whole, in contrast to diversity of species within a major habitat. Two indexes are listed, the Shannon index ( $\bar{H}$ ) and the Pielou evenness index (e). The evenness index can be considered a scaled index since values fall within the range of 0 and 1 (0 is no diversity, i.e., only one kind of habitat or monoculture, and 1 is the maximum value if all habitats are of equal importance). Present habitat diversity (or diversity as of 1966) is compared with estimated diversity that would exist should the level 2 and level 3 management plans proposed by the Forest Service (Beavers et al., no date) be put into effect.

At the present time, the SRP reservation is by no means a vast monoculture of pines, as might be assumed from superficial examination of aerial photos. In fact, there is a relatively high diversity in the landscape because of the way numerous blocks and strips of different vegetative types are intermixed. Figure 4 shows diversity levels for the different blocks across the transect. As would be expected, the habitat diversity is greater in the blocks containing ponds

and streams than in those which are largely upland in elevation. A recent paper (Odum, 1975) suggested, on the basis of an analysis of diversity levels in a wide range of ecosystems, that a diversity index with an e scale of between 0.7 and 0.8 is optimum for stable self-maintaining ecosystems that are not subject to either stress or subsidy (auxiliary energy or nutrients). Thus, from the theoretical ecological viewpoint, the pattern diversity of SRP is quite good. So far, forest management on the area has concentrated on planting pines on the abandoned agricultural land, which eventually would have been invaded by pines in the course of natural ecological succession. Thus, although the in-habitat diversity of pine plantations is less than that in natural pine successional stages would be, the landscape pattern diversity of the area is now essentially that of a natural area.

Diversity would be substantially reduced, however, if intensively managed pine culture were extended into the two-thirds of the area not under cultivation in 1950. Increasing pine plantation coverage to 50% for the whole area, as proposed in the Forest Service's level 2 management plan, would necessitate converting the scrub oak and other upland hardwood and mixed stands to pure pine. We estimate that this would reduce the number of major habitats from 27 to 20 and would also reduce the overall landscape diversity (see Table 2). Increasing pine coverage to 75%, as proposed in the level 3 management plan, would necessitate extending pines into the bottomlands and other soils not naturally suited for commercial species of pines. The number of major habitats would be reduced to 14, and overall diversity would be drastically reduced (Table 2). An evenness index of 0.38 represents the kind of low diversity found in highly stressed ecosystems or highly subsidized ecosystems that require large inputs of energy and materials to maintain.

Increasing the coverage of pine monoculture would also greatly reduce the in-habitat abundance and diversity of most if not all animal populations, including game species. Table 3 compares density and diversity of breeding bird populations in pure pine and mixed bottomland forests to illustrate the reduction in animal populations expected

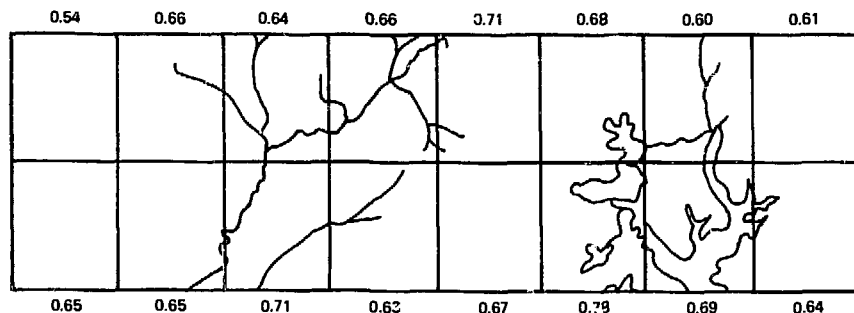


FIG. 4 Landscape pattern diversity on the evenness (e) scale for each of the 16 blocks across the sample transect. See text for explanation.

TABLE 3 Bird Populations in Pure Pine Forests and Mixed Bottomland Hardwood Forests on the SRP Reservation\*

| Habitat                                  | Number of species | Density, pairs/100 acres | Diversity index |       |
|------------------------------------------|-------------------|--------------------------|-----------------|-------|
|                                          |                   |                          | $\bar{H}$       | $e$   |
| Pure pine<br>(188 acres censused)        | 9                 | 31                       | 1.509           | 0.687 |
| Mixed bottomland<br>(145 acres censused) | 19                | 268                      | 2.600           | 0.883 |

\*Based on unpublished census data of Robert Pearson gathered during two successive years and recorded in the 1954 Annual Report submitted to AEC by the University of Georgia.

when natural forests are converted to even-aged monocultures. As already indicated, this trend can be counteracted insofar as the landscape as a whole is concerned by preserving strips and blocks of other habitat between and within plantations.

Since the interaction of climate, topography, and hydrology is reflected in the physical characteristics of soils, we next looked at the diversity of soil types on the SRP reservation. Percent coverage of 21 soil types is shown in Table 4, together with calculations of diversity indexes  $\bar{H}$  and  $e$ . Interestingly enough, diversity indexes for soil types

TABLE 4 Overall Diversity of Soil Types in the SRP Area Exclusive of the Savannah River Swamp\*

| Soil type                 | Coverage, % |
|---------------------------|-------------|
| Furquay—Wagram            | 30.7        |
| Dothan—Norfolk            | 10.7        |
| Troup loamy sand          | 9.4         |
| Gunter sand               | 8.4         |
| Vaughan—Blaney            | 7.3         |
| Johnson—Okenee            | 6.7         |
| Troup sand                | 5.8         |
| Variable bench soils†     | 4.9         |
| Kalmia—Johns              | 3.4         |
| Orangeburg—Red Bay        | 3.4         |
| Lakeland—Kershaw          | 2.4         |
| Grady—Bayboro             | 2.1         |
| Izagora sandy loam        | 2.1         |
| Leaf—Wahce                | 0.25        |
| Americus sand             | 0.64        |
| Mascotte sand             | 0.21        |
| Johnson—Terra Ceia        | 0.21        |
| Ocilla—Albany             | 0.21        |
| Orangeburg, terrace phase | 0.21        |
| Lucy—Wagram               | 0.21        |
| Oiler sand                | 0.21        |
| Total (21 types)          | 100.05      |
| Diversity index           |             |
| $\bar{H} = 2.349$         |             |
| $e = 0.772$               |             |

\*Data from Aydelott (no date), table on p. 32.

†Narrow strips of varying types along sharp breaks and benches adjacent to waterways.

are almost identical to those for habitats (see Table 2, Present situation); this supports our contention that the present vegetation of the SRP area is in tune, so to speak, with the natural diversity of the physical habitat.

The justification for increasing pine coverage (proposed management levels 2 and 3), and thus reducing diversity, is based on an economic projection that the demand for paper will continue to increase at exponential rates and the price will rise sufficiently to justify the heavy expenditures involved in channelizing streams, draining wet soil, and suppressing hardwoods and other vegetation more naturally adapted to many of the soil types present. Since, for reasons already outlined, energy conservation and recycling programs are almost certain to become major national policies, such straight-line projections for the growth in paper demand are unrealistic. Furthermore, putting all the eggs in the pine basket, so to speak, increases the future risk of epidemic disease. To design "against the grain" of the natural topography and soil diversity is questionable, even on a private tract owned by a paper mill, and certainly it is inappropriate at a National Environmental Research Park, which is to provide a number of different functions. To further emphasize this point, Table 5 compares values under different management options or missions of uplands, which are largely populated by pine stands, and bottomlands, which currently support well-developed hardwood stands. If the reservation is to be managed strictly as a timber- and pulp-producing area, pine-covered uplands would be more valuable in the present-day economy than bottomlands and wetlands. Just the reverse would be true if the main mission is to be waste management. The water and sponge-like vegetation in the valleys have a much greater capacity for waste assimilation, but spray irrigation of waste water on the uplands has possibilities for the future. As evidence for the high value of wetlands for waste disposal, we can call attention to use of lagoons planted with bulrushes in the purification of industrial waste water

TABLE 5 Comparative Values of Possible Major Missions for Management of the Savannah River Reservation

| Management mission                   | Comparative value                                                                                                            |          |
|--------------------------------------|------------------------------------------------------------------------------------------------------------------------------|----------|
|                                      | Uplands                                                                                                                      | Lowlands |
| Pump and saw-timber production area  | High                                                                                                                         | Low      |
| Waste assimilator (power park)       | Low                                                                                                                          | High     |
| Fuel forest                          | High                                                                                                                         | High     |
| National environmental research park | Greatest value is high natural diversity (habitat diversity in optimum range of 0.7–0.8, paralleling natural soil diversity) |          |

in West Germany. Seidel (1975) of the Max Planck Institute reports that this seminatural system is quite effective and costs one-third as much as artificial tertiary treatment. If the use of solar energy via burning wood to produce electricity is a feasible energy option, any kind of tree would be valuable, with hardwoods perhaps being more valuable because of their ability to resprout after cutting, thus making possible a coppice-type management. Finally, as already emphasized, the present natural diversity is the most valuable attribute of the area in terms of research.

Theoretically, the capacity of different kinds of ecosystems to assimilate heated water, nutrients, dilute organic matter, and other degradable wastes can be monitored by measuring two major ecosystem properties: (1) productivity, rate of production or gross metabolic rate, and (2) diversity of one or more major functional components. Performance curves, illustrated in Fig. 5, can be used to

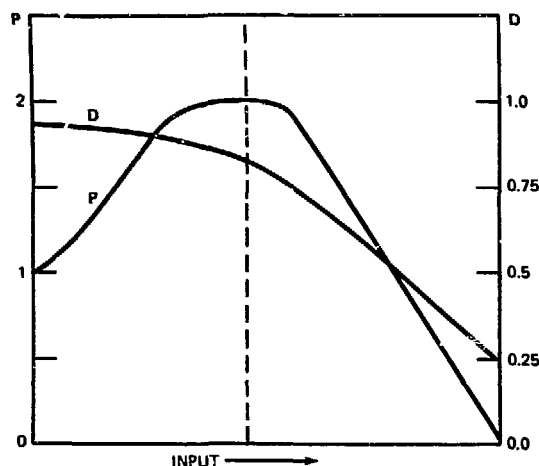


FIG. 5 Hypothetical performance curves for productivity (P) and diversity (D) as function of increasing inputs of usable inorganic and organic nutrients. The vertical dashed line indicates the level beyond which additional input becomes undesirable.

determine when inputs begin to act as stresses to the recipient environment. Assimilable wastes tend to act as subsidies at low rates of input, in that the productivity of the system as a whole will be increased. If the recipient system has a high diversity to begin with, an increase in metabolic rate or eutrophication generally is accompanied by a decrease in diversity. If the natural diversity is low to begin with, diversity may actually increase during the initial stages of eutrophication. The point where productivity levels off and diversity begins to be strongly reduced marks the level beyond which a further increase in input will become stressful to the system as a whole and, therefore, undesirable. As already indicated, if natural diversity is of the order of 0.7 to 0.8 on the *e* scale, which we used to assess habitat diversity, then a drop below about 0.6 indicates increasing stress and instability. Since, in practice,

it is often easier to measure diversity than productivity, and since such indexes are more independent of daily and seasonal cycles, a well-chosen measure of diversity may be the key to monitoring.

Using surface cooling from natural or seminatural ecosystems instead of powered cooling with artificial towers reduces the cost of cooling water at electricity-generating plants by more than half, as indicated by Table 6, but very large areas are required if the heat is to be dissipated without harm to the biotic component of aquatic

TABLE 6 Cost of Cooling Water at Electricity-Generating Power Plants (in Addition to Generation Cost)\*

| Types of cooling                    | Mills/kw-hr |
|-------------------------------------|-------------|
| Seawater (sea coast site)           | 0.03        |
| Freshwater cooling ponds            | 0.09        |
| Wet cooling tower, natural draft    | 0.20        |
| Wet cooling tower, mechanical draft | 0.21        |
| Dry cooling tower                   | 0.82        |

\*From data of Hauser (1971).

or semiaquatic ecosystems. In the past, the release of too much hot water into too small an area (small streams) in the SRP area resulted in great stress on floodplain ecosystems. Determining the capacity of ponds and floodplains to dissipate waste heat within the adaptive limits of the systems is one objective of the thermal ecology research program at the Savannah River Ecology Laboratory. The possibility of spray irrigation of waste water on the 40,000 acres of sand hills and other well-drained soils offers possibilities for experimentation.

Although burning wood to produce electricity may appear to be an unlikely option to solve the country's energy crisis, it is interesting to assess the SRP reservation as a fuel forest. Use of wood as fuel, as well as for paper and other products, represents another way to tap solar energy through photosynthesis, which is, from the standpoint of the second law of thermodynamics, an efficient conversion of low-utility sunlight into high-utility organic fuel. The current harvest of wood from the SRP area is  $3.5 \times 10^6$  ft<sup>3</sup>/year (Beavers et al., no date); this has an estimated energy content of  $5.7 \times 10^{11}$  BTU, assuming 165,000 BTU/cubic foot of wood. At a 38% conversion efficiency, a large power plant requires 9000 BTU/kw-hr of electricity produced (National Power Survey, 1970). Accordingly, the SRP harvest would produce  $6.4 \times 10^7$  kw-hr, enough electricity to supply 8000 people at per capita consumption of  $7.3 \times 10^3$  kw-hr/year. If all the annual growth of the pine uplands, estimated to be  $12 \times 10^6$  ft<sup>3</sup>/year, were harvested and converted to electricity,  $2.2 \times 10^8$  kw-hr could be produced; this would supply 30,000 people.

Szego and Kemp (1973), and Kemp and Szego (1974) made a detailed assessment of wood as fuel for the generation of electricity, basing their models on different

sun-to-wood conversion efficiencies and different rates of insolation. At a 0.5% conversion level and an insolation rate of  $1300 \text{ BTU ft}^{-2} \text{ day}^{-1}$ , both reasonable for the SRP area, the annual growth from 500 acres would be required per megawatt plant capacity. According to this assessment, the 200,000 acres of SRP could support a 400-Mw plant that would supply 120,000 people with electricity. Incidentally, this is about the same amount of wood yield needed to support a 1000 ton/day paper mill. It is ironic that for all the sophisticated technology involved in the development of atomic energy at SRP, the reactors at the site are not capable of supplying a single kilowatt of energy for peaceful uses, but the forests are capable of supplying appreciable amounts!

In conclusion, the concept of the power park is not really new. Man has always depended on natural areas adjacent to his cities and factories to assimilate and recycle the end products of civilization's metabolism. What is new perhaps is the idea of setting aside natural areas to remain free from human habitation for this purpose and establishing a reciprocal design coupling the waste maker and the waste assimilator into an integrated or symbiotic system. Such a strategy creates an economically attractive way to use the solar-powered free work of nature to ameliorate the disorder inherent in man's energy-conversion processes.

Although the Savannah River Plant reservation was originally established solely to provide security for the production of weapon materials, it turns out that the large size and high natural diversity of the area makes it an ideal location for basic research and experimentation needed to determine how the coupling can be accomplished without

undue stress to the biotic components of the living filter system. I would urge ERDA to establish waste-management research, emphasizing ecological carrying-capacity research, as a major mission for the National Environmental Research Park. Such an expanded operation would not interfere with security needs nor with a level of timber production compatible with the natural site capabilities of the land.

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# Factors Influencing Thermal Tolerances of Individual Organisms

## ABSTRACT

The diversity of experimental methods and terminology employed by investigators to measure the effects of high temperatures on individual organisms, plus the often overlooked complexities of the holocoenotic environment, has often led to disconcerting conclusions. A plea is made for standardization of testing methods and for a wider appreciation of factors that may alter thermal tolerances. The influence of elevated temperature is grouped into three categories, lethal effects, controlling effects, and directive effects, all of which should be considered in assessing the impact of thermal effluent on organisms. In addition, the terminology (acclimation, acclimatization, adaptation, habituation, lethal temperature, critical thermal maximum, etc.) needs standardized definitions. The important factors influencing thermal effects on organisms include photoperiod, seasonal and daily cycles, geographic variation, diet, sex, breeding condition, age, life-cycle stage, salinity, chemicals, body water content and partitioning, oxygen supply, pH, innate and learned behavior, history of thermal exposure, sublethal exposure to limiting factors, and experimental methods. Examples of most of these are given to illustrate the role of temperature in the holocoenotic environmental complex of individual organisms.

Measurement of the influence of temperature on organisms has a long and voluminous history. Even the more limited topic of this paper is much too broad to be dealt with in detail here. The diversity of experimental methods and terminology employed by investigators, plus the often overlooked complexities of the holocoenotic environment, has led to disconcerting conclusions. My purpose here is to plead for standardization of testing methods and definitions and an appreciation of often overlooked factors that may alter thermal tolerances in subtle ways. More detailed reviews of many of the topics discussed here can be found in Brett (1956), Fry (1967; 1971), Krenkel and Parker (1969), Whitrow (1970), Coutant (1973), Coutant and Goodyear (1972), Sylvester (1972), and Precht et al. (1974). Raney and Menzel (1969), Beltz et al. (1974), and Kennedy and Mihursky (1967) have compiled useful bibliographies of the literature on thermal effects on aquatic organisms. For the sake of brevity, only tolerance to high temperature is considered here.

Of all the physical factors of the environment, temperature is perhaps the most pervasive; it mediates the effects of most other environmental factors and is influenced reciprocally. Temperature is only one of many factors in the environmental complex of an organism. The holocoenotic nature of the environment (Allee and Park, 1939) requires that the complexity of the interrelationships between an organism and all the components of its environment be known if we are to understand fully the ecological niche. The intricacy of these interrelations is so involved that any attempt at complete analysis and synthesis may be far in the future. The synergistic actions of environmental factors are now widely recognized, however, and multifactorial experiments are improving our predictive assessments of the alteration of environments (Brett, 1969).

Some factors that influence the thermal tolerance of organisms, as well as the experimental measurement of thermal effects, are summarized in Fig. 1. The biological effects of each of the factors may interact synergistically with temperature as well as with any other factor or combination of factors. There is little evidence that some of the factors listed directly influence the thermal tolerance of animals (e.g., gravity, rotational forces, or geomagnetism), but influences may be found in the future. Examples of the interactions of some of the factors are discussed later. The quality, quantity, duration, and rate of change of a factor often determine its influence on an organism. For example, wave length, intensity and photoperiod of light, and the rate of change of simulated twilight periods have all been shown to be important considerations governing the behavior, reproduction, and physiology of organisms. The methods used for pretreatment (acclimation, acclimatization) of test animals, and the techniques used to determine thermal tolerance will significantly influence the data obtained. Exposure to sublethal levels of other factors may alter thermal tolerance, and exposure to sublethal temperatures may produce increased thermal tolerance in later exposures to high temperature. These are a few of the multitudinous interactions within the environmental complex of an organism which may influence thermal adjustment or tolerance.

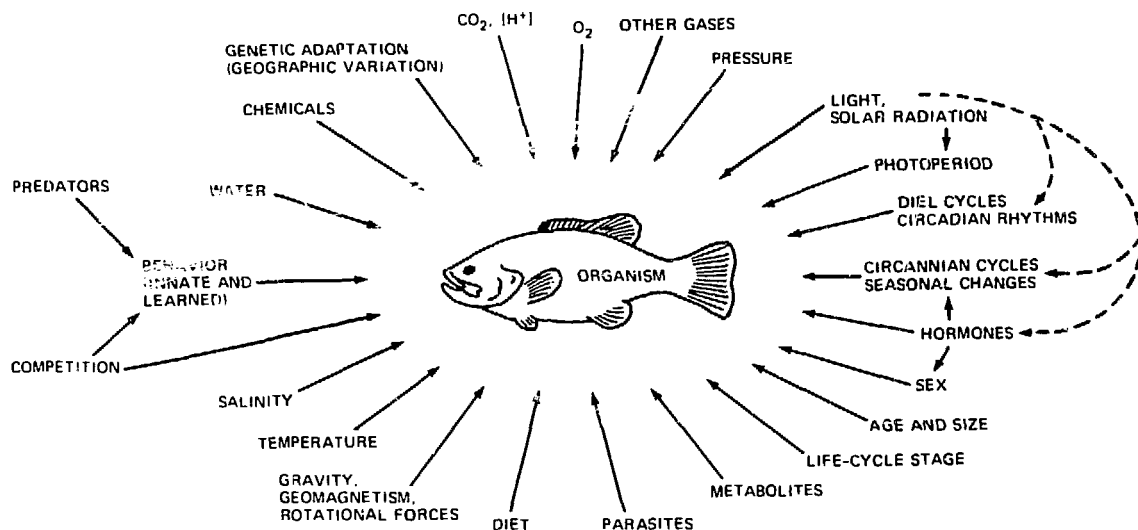


FIG. 1 Some factors that may influence the thermal tolerance of organisms. Other influences are quality, quantity, duration, and rate of change of environmental factors; methods of testing; methods for acclimation and acclimatization; sublethal exposure to limiting factors; and interrelations between controlling, directing, and lethal forces.

## TERMINOLOGY

Fry (1971) grouped the effects of environmental forces on individual organisms under five headings (lethal, controlling, limiting, masking, and directive factors). In considering the effects of temperature, I have included limiting and masking factors under controlling factors since all three operate through similar mechanisms and are not always clearly separable. *Lethal* effects result from temperatures that kill in a finite time within what would otherwise be a normal life-span. *Controlling* effects result from the influence of temperature on metabolic rates and the underlying biochemical reactions, such as alterations in enzyme-substrate affinities and changes in activation energy (Somero, 1969) and control of isoenzyme systems uniquely fashioned to operate over the environmental temperature ranges of ectotherms (Hochachka and Somero, 1973). These basic changes may give highly diverse results, and controlling effects include almost any activity of an organism. A discussion of controlling effects is beyond the scope of this paper. *Directive* effects result from the influence of temperature on spontaneous movements. The response of animals within a temperature gradient is a directive effect.

The history of an organism can determine its response to changes in environmental factors. This is particularly true of temperature; organisms must be "prepared" before temperature effects are measured in the laboratory. On the basis of priority, frequency, and authoritative uses, the following terms should be used to describe the history of the thermal environment of an organism (Folk, 1974): *Acclimation* is the compensation made to a single environmental factor. Thus animals held at a given constant

temperature before measurement or observation of experimentally produced changes have been acclimated to the given temperature. The time required for acclimation to a new temperature may range from a few hours to a few weeks. *Acclimatization* is the adjustment made to two or more environmental factors, e.g., in controlled experiments in the laboratory, animals may be acclimatized to a given temperature and photoperiod, with other major environmental factors held as constant as is feasible. In a general sense, an organism in its natural environment has been acclimatized to the complex of its holocoenotic environment, but the term is more frequently applied to the controlled conditions of the laboratory. The time courses for acclimation and acclimatization are similar. *Adaptation* applies only to genetic change that results in phylogenetic adjustment over long periods and usually through many generations. *Habituation* is "the process of forming into a habit or accustoming"; this implies that it "depends on the mind, that it is reversible and that it may involve the diminution of normal responses or sensations" (Glaser, Hall, and Whittow, 1959). Although it has been applied to both whole animals and specific organ-system components, this definition is more commonly used in behavioral terms in reference to the waning response of repeated stimuli not followed by reinforcement. The term habituation should replace the older usage of adaptation. The change in the "psychological set" of an animal as a result of its being placed in a new laboratory environment, often evidenced by increased activity and some degree of stress, and the subsequent diminution of the behavior and stress to more "normal" levels have been described as habituation. This use is termed general habituation, as opposed to the



adjustment made to a specific stimulus and specific organ system component, which is specific habituation. Application of the term habituation is, however, limited to the nervous system or to behavior.

## METHODS OF DETERMINING THERMAL TOLERANCE

There are basically two methods for determining lethal temperatures in ectothermic animals, and each has its advantages and disadvantages. Both methods yield valuable information, but the various techniques, definitions, and interpretations have created some confusion in the literature. The method for lethal temperatures is based on measuring resistance times at various constant test temperatures of animals previously acclimated to a wide range of temperatures in the laboratory. In this method the animals are abruptly exposed to the test temperature. Papers by Fry, Brett, and Clawson (1942), Fry, Hart, and Walker (1946), Fry (1947), Hart (1952), and Brett (1944; 1952) describe this method. The statistical treatment of the data is based on pharmacological procedures used to determine LD<sub>50</sub> from dosage-mortality experiments (Bliss, 1937).

In the lethal-temperature method, the geometric mean resistance times are calculated for each group of animals tested at several test temperatures for each acclimation temperature. The results are plotted on probit X logarithmic axes to obtain the median resistance times for each sample (Fig. 2). The median resistance times obtained can then be plotted on arithmetic X logarithmic axes for each acclimation temperature. At lower test temperatures there is a point where further exposure does not result in additional deaths and the median resistance times level off

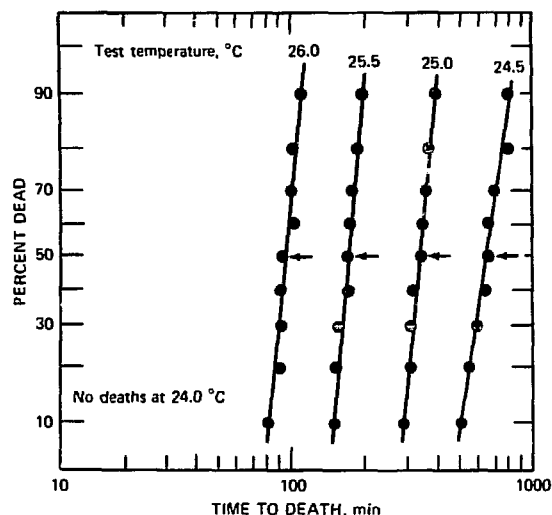


FIG. 2 Times to death at different lethal test temperatures in young spring salmon (*Oncorhynchus tshawytscha*) acclimated to 10°C, plotted on probit X logarithmic axes. Calculated geometric mean resistance times coincide with median resistance times at probit 5.0, indicated by the arrows. (Adapted from Brett, 1952.)

to parallel the time axis (Fig. 3, dashed lines). The points at which this break occurs mark the boundary between the zone of thermal tolerance and the zone of thermal resistance. The boundary is the upper incipient lethal temperature. A similar method is used to determine resistance to low temperatures and the lower incipient lethal temperature. The upper incipient lethal usually increases with increased acclimation temperature only to a point; at acclimation temperatures beyond this point,

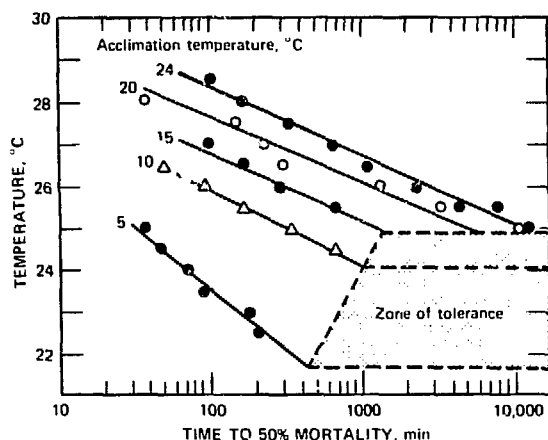


FIG. 3 Median resistance times to high temperatures among young spring salmon acclimated to various temperatures, plotted on arithmetic X logarithmic axes. Median resistance times were calculated as in Fig. 2. (From J. R. Brett, 1952, Temperature Tolerance in Young Pacific Salmon, Genus *Oncorhynchus*, *Journal of the Fisheries Research Board of Canada*, 9: 279).

thermal resistance is not increased, and a plateau occurs in the upper incipient lethal line (Fig. 4). Constructing a line at an angle of 45° (where acclimation temperatures equal lethal temperatures) allows us to determine the ultimate upper lethal temperature. The area bounded by the lower incipient lethal, by a vertical line drawn from the 45° diagonal to the lower incipient lethal, by the upper incipient lethal, and by the appropriate portions of the x and y axes of the graph is the zone of thermal tolerance, which can be expressed in degrees centigrade squared, (°C<sup>2</sup>). The area outside the same boundaries constitutes the zone of thermal resistance, where continued survival is not possible (Fig. 4). The area of the zone of thermal tolerance and its position relative to the x and y axes are measures of the degree of "thermy." Oligothermal animals, for example, have small zones of thermal tolerance toward the low temperatures, and eurythermal species have large zones of thermal tolerance extending toward high temperature values. Brett (1956) summarized such data for various species of fish and showed how the area of the zone of thermal tolerance correlates with habitat and distribution.

The critical thermal maximum (CTM) was originally defined by Cowles and Bogert (1944) as "the thermal point at which locomotory activity becomes disorganized and the

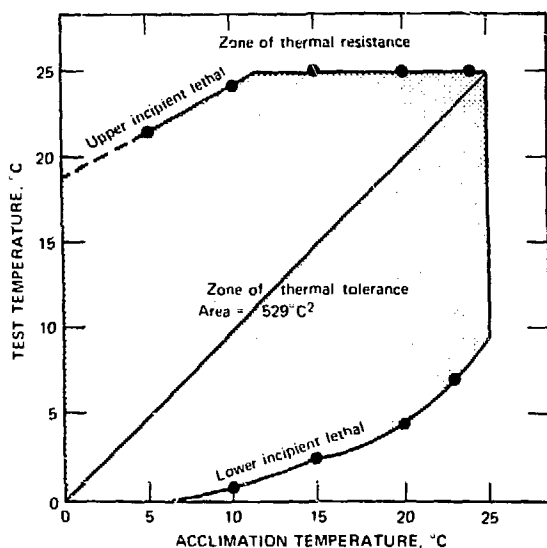


FIG. 4 Thermal tolerance for young spring salmon. (Adapted from Brett, 1952.)

animal loses its ability to escape from conditions that will promptly lead to its death." This definition has been modified to include statistical variation (Lowe and Vance, 1955) and standardized methods (Hutchison, 1961). The CTM was first used largely by herpetologists but has found wide use for many ectothermic vertebrates as well as invertebrates. The CTM is determined by heating animals from the acclimation temperature at a constant rate that is just fast enough to allow deep body temperature to follow the ambient test temperatures without a significant time lag. A slower rate may allow partial thermal acclimation to occur (Cox, 1974). A rate of 0.5 to 1.5°C per minute is often used for smaller animals. As the temperature is increased, the animal will usually increase its activity in attempts to escape, will lose its righting response, and at some point will go into violent muscular spasms. The onset of spasms is usually used as the end point, but some workers have used the temperature at which the righting response is lost. The loss of righting response usually occurs a few degrees below the onset of spasms and is often more variable (Mahoney and Hutchison, 1969; Hutchison, Vinegar, and Kosh, 1966). If the animal is removed to a lower temperature immediately after the CTM is reached, it will recover, but long-term survival may be impaired in some individuals receiving such a sublethal heat shock. If test animals are allowed to remain at the CTM or higher temperatures, death will follow almost immediately. In a sense, the CTM is the ecological lethal index since animals in nature may encounter such temperatures as acute fluctuations above their tolerance limits.

The two methods have been compared by Fry (1967). Because of the acute nature of exposure to high tempera-

tures, the CTM produces values somewhat higher than the lethal temperature, especially if the CTM is defined as the point at which death occurs (Fig. 5). The lethal-temperature method may be more significant for aquatic organisms, which are subject to a much slower environmental temperature change. This method provides more detailed information on the overall thermal tolerance of an organism, but it requires a large number of test animals and a fairly large quantity of acclimation and constant-temperature test chambers. The CTM method is much more economical in the use of animals and equipment, and results can be obtained much faster. When the two methods are compared in the same species, the slope of the CTM value to acclimation may differ from the response of the incipient lethal temperature to acclimation (Fig. 5), at least when death is used as the CTM end point. The loss of muscular coordination, usually occurring in the form of spasms, seems to be the best CTM end point since it fits the original definition and the accepted modifications to the method. The CTM method can be a useful tool in thermal ecology provided its limitations are recognized.

## EXAMPLES OF FACTORS INFLUENCING THERMAL TOLERANCE

### Seasonal Changes

Seasonal differences in thermal tolerance are well documented but are often overlooked. Such seasonal changes may be most critical at certain ages or stages of a life cycle. Temperature may influence reproduction, growth, and survival differently according to season (Fig. 6). Seasonal variation is often related to changing photoperiod but may be due to circannian rhythms with unknown zeitgebers. For example, Tyler (1966) was not able to reverse completely the seasonal effect on temperature tolerance in the red-bellied dace (*Chrosomus eos*) by reversing the annual cycle of photoperiod. In a study of Gulf of California shore fishes, Heath (1967) found that in winter fishes "have CTM lower than their maximum habitat temperatures in summer." Heath concluded that "seasonal acclimation on the order of 4 to 6°C of temperature tolerance is an ecological necessity." The CTM of newts (*Notophthalmus viridescens*), all collected in the same locality, are significantly different at different times of the year (Fig. 7, Orange County, North Carolina).

### Photoperiod

The thermal resistance of animals may be altered significantly by the light regime to which they are exposed. Hoar (1956) showed that goldfish maintained on a long-day photoperiod were more heat resistant than short-day fish. Similar effects have been described in carp (Roberts, 1961), salamanders (Hutchison, 1961), turtles (Hutchison and Kosh, 1965), and other animals.

The pattern of increased heat resistance under a long photophase and increased cold resistance under a short

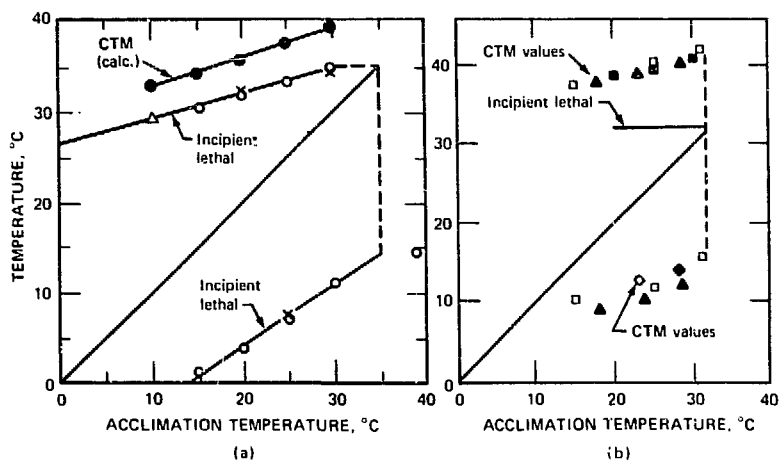


FIG. 5 Effect of temperature on mortality. (a) The minnow, *Notemigonus crysoleucas*; ●, Hart (1952); △, Algonquin Park; x, Put-In Bay; ○, Welaka (Arai et al., 1963, *Can. J. Zool.*, 41: 1011). (b) The guppy, *Lebistes reticulatus*; □, Auerbach, 1957, *Z. Fishcheri. u. d. Hilfswiss. N. E.*, 6: 605; ■, Gibson, 1954, *Can. J. Zool.*, 32: 393; ○, ●, Pitkow, 1960, *Biol. Bull.*, 119: 231; ▲, Tsukuda, 1961, *J. Biol. Osaka City Univ.*, 12: 15. The graphic method of Fry et al. (1946) was used to calculate CTM values, and the figure was constructed by Fry. (From F. E. J. Fry, 1967, *Responses of Vertebrate Poikilotherms to Temperature*, in *Thermobiology*, A. H. Rose (Ed.), p. 383, Academic Press, Inc., New York.)

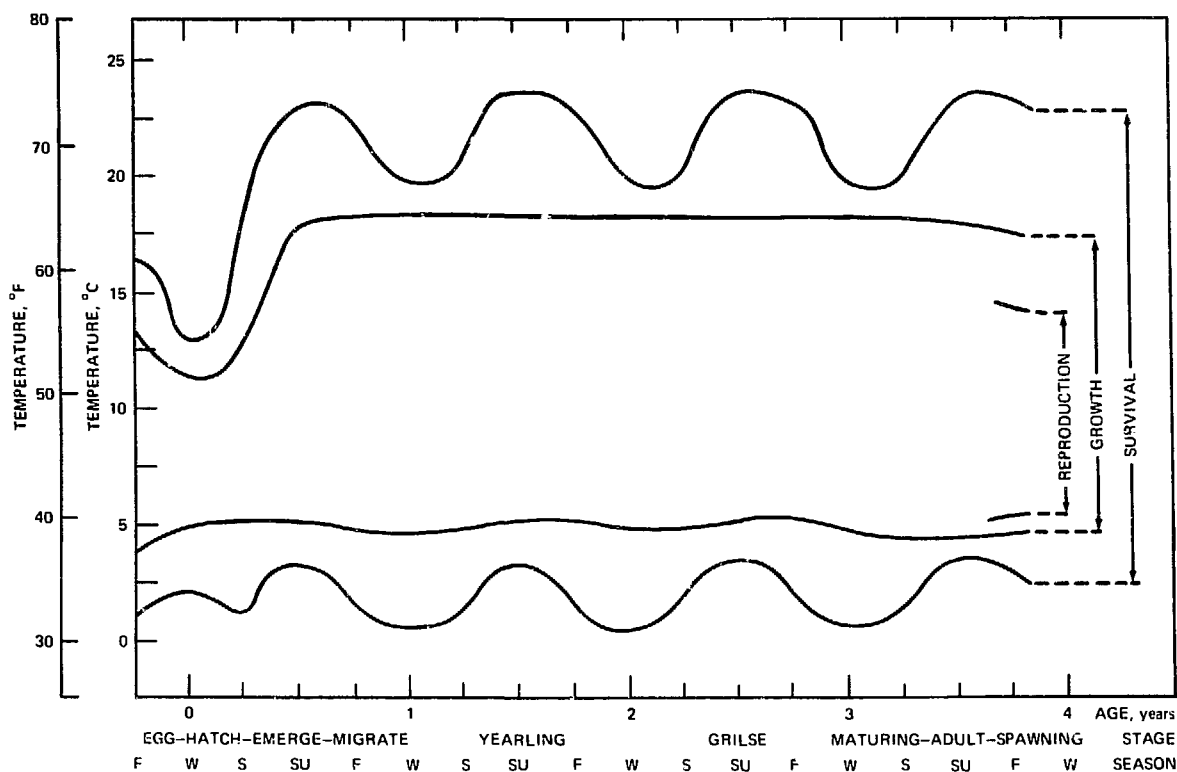


FIG. 6 Schematic representation for minima and maxima of different life processes in Pacific salmon from data compiled by Brett (1969). (From O. Kinne, 1963, *The Effects of Temperature and Salinity on Marine and Brackish Water Animals. I. Temperature*, *Oceanography and Marine Biology, Annual Review*, 1: 309.)

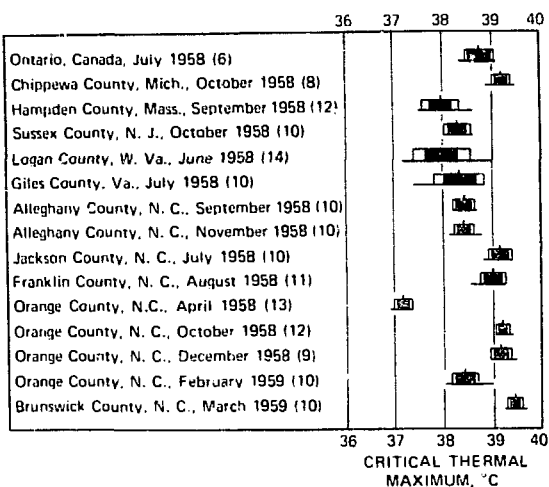


FIG. 7 Critical thermal maxima of *Notophthalmus viridescens* collected at different localities and different times of the year and acclimated to 15°C. Sample size is given in parentheses. The short vertical line shows the mean for each sample; the horizontal line, the range; the black rectangle, two standard errors on each side of the mean; the white rectangle plus black rectangle on each side of the mean, one standard deviation. (From V. H. Hutchison, 1961, Critical Thermal Maxima in Salamanders, *Physiological Zoology*, 34: 110.)

photophase would be of adaptive value in temperate and polar regions (Hoar and Robertson, 1959).

## Diel Changes

Despite the daily cycles known for lethality of drugs, bacterial toxins, radiation, noise convulsions, etc., little attention has been paid to the possibility that thermal tolerance may vary with the time of testing (which is seldom reported), especially when we use the CTM method, in which short exposures to the test conditions may be greatly influenced by the time of day. In the lethal-temperature method, exposure times to test temperatures are much longer and daily cycles in temperature sensitivity may be masked. Significant daily cycles have been described in painted turtles (*Chrysemys picta*) (Hutchison and Kosh, 1965). In this diurnal basking species, the CTM is highest at 1400 hr EST when the animals are acclimatized and tested under a photoperiod of L-D 16 : 8 at 20°C and L-D 8 : 16 at 10°C. "Abnormal" combinations of high temperature and short-day photoperiod (L-D 6 : 18, 20°C) or low temperature and long-day photoperiod (L-D 16 : 8, 10°C) resulted in phase shifts of the time of maximum tolerance in the daily cycle of +4 and -8 hr, respectively (Fig. 8). Similar daily cycles of thermal tolerance and phase shifting have been reported in other reptiles (Spellerberg and Hoffman, 1972) and in amphibians (Mahoney and Hutchison, 1969; Dunlap, 1969; Seibel, 1970; Johnson, 1971; 1972a; 1972b). Because of the methods of testing,

we cannot be certain that the diel cycles are circadian rhythms (endogenous cycles that persist under "free-running" conditions of constant light or darkness), but the ubiquity of circadian rhythms in organisms suggest that such an endogenous mechanism is operative here. Certainly these findings suggest that the time of day at which animals are tested and the photoperiod to which they have been exposed are important factors to be considered in assays of thermal sensitivity. Unusually rapid acclimation rates in response to natural daily temperature variations may complicate the assessment of diel cycles of temperature sensitivity (Pough, 1974).

## Geographic Variation

Possible genetic differences in populations of the same species from different areas of the range may cause significant differences in thermal tolerance. In the salamander *Notophthalmus viridescens*, samples of populations from different parts of the range collected during the same season (to avoid differences due to seasonal variation) have significantly different CTM (Fig. 7). Similar geographic variation in lethal temperatures is known in some fishes (Hart, 1952; McCauley, 1958) and turtles (Hutchison, Vinegar, and Kosh, 1966). The absence of significant differences in thermal tolerance between latitudinally separated populations in some species has led some investigators (e.g., Ushakov, 1964) to argue that geographical variation is slight. Some species do not show significant shifts in thermal resistance over very wide ranges. Fiddler crabs of the genus *Uca*, which occur from Massachusetts to Brazil, do not vary significantly in upper lethal temperatures but do in lower lethal temperatures (Vernberg and Vernberg, 1967). Similarly, intertidal snails of the family Littorinidae, which have a worldwide distribution, apparently have similar tolerances to high temperature (Fraenkel, 1968).

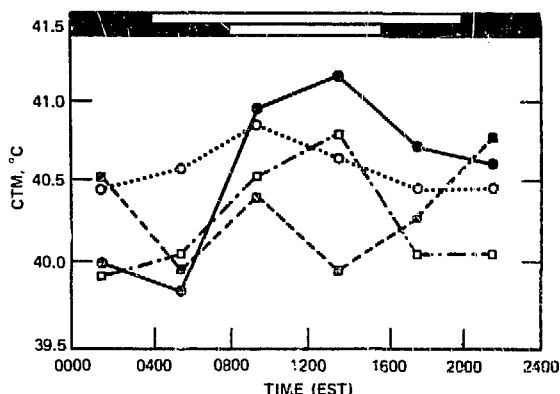


FIG. 8 Daily cycles in the critical thermal maximum of painted turtles (*Chrysemys picta*) acclimatized to four combinations of constant temperature and photoperiod. Photoperiod of L-D 16 : 8, —●—, at 20°C; —■—, at 10°C. Photoperiod of L-D 8 : 16, .....○, at 20°C; —□—, at 10°C. (Data from Kosh and Hutchison, 1968.)

Some species show geographic variation and some do not, but the possibility that such genetic differences can influence thermal tolerance should not be overlooked.

## State of Development

One of the subsidiary principles of Shelford's law of tolerance is that the period of reproduction is usually a critical period when environmental factors are most likely to be limiting (Decola, 1970; Frank, 1974). This is certainly true for sensitivity to temperature (examples are shown in Figs. 6 and 9), except for the resistant egg or diapause stage

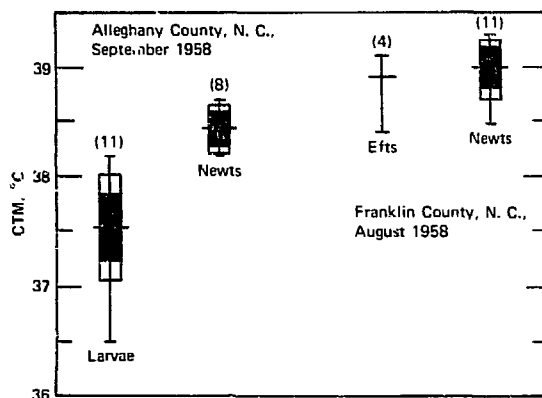


FIG. 9 Critical thermal maxima at different stages in the life history of *Notophthalmus viridescens* from two locations. Means, ranges, and errors represented as in Fig. 7 except that the scale of CTM is vertical rather than horizontal. Numbers in parentheses are sample size. (From V. H. Hutchison, 1961, Critical Thermal Maxima in Salamanders, *Physiological Zoology*, 34: 113.)

of some animals, but much of the literature on thermal tolerance deals with adult forms. Clearly, more attention needs to be given to determining the thermal tolerance of developmental stages.

## Salinity

The salinity of water often has a pronounced, but sometimes unpredictable, influence on the thermal tolerance of aquatic organisms. Often increased salinity will increase the resistance of poikilosmotic animals to both low and high temperature, e.g., in the oligochaete *Enchytraeus albidus* (Kähler, 1970). In cyprinodontid fishes, Garside and Jordan (1968) and Garside and Chin-Yuen-Kee (1972) found the highest upper lethal temperatures in an isosmotic environment, with a decline in the lethal in freshwater and in seawater. Survival of salmon (*Salmo salar*) and sea trout (*Salmo trutta*) at high temperatures is dependent on salinity, especially when the fish are suddenly changed to freshwater from saltwater; parr and smolt differed in their sensitivity (Alabaster, 1967). In the lobster (*Homarus americanus*), McLeese (1956) found that increased salinity has less effect on thermal tolerance at higher acclimation temperatures (Fig. 10).

## Chemicals

The synergistic action of chemicals and temperature has been the subject of numerous papers, especially on fishes. The recent increased concern with the effects of industrial and agricultural waste water and thermal effluent is largely responsible for this interest. Exposure to sublethal concentrations of chemicals in the environment often results in a decrease of the upper lethal limits of an organism (Doudoroff, 1957). Some examples are phenol (Brown, Jordan, and Tiller, 1967), hydrogen sulfide (Adelman and Smith, 1972), sodium pentachlorophenate (Alderdice, 1963), toxaphene (Mahdi, 1966), DDT (Anderson, 1971) and other pesticides (Macek, Hutchison, and Cope, 1969), fluorides (Angelovic, Sigler, and Neuhold, 1960), pyridyl-mercuric acetate (Clemens and Sneed, 1958), copper and zinc (Sprague, 1964), mercury (Thatcher, 1974), cadmium (McKenney and Dean, 1974), and cyanide (Wuhrmann and Woker, 1953). Natural hardness (due primarily to carbonates) in water can also modify thermal resistance. Rainbow trout (*Salmo gairdnerii*) reared in hard water are less resistant to thermal stress than those reared in soft water, but increased salinity can result in increased thermal tolerance (Craigie, 1963).

## Diet and Nutritional Status

Significant changes in tolerance to both high and low temperatures in goldfish fed on fats of different degrees of hydrogenation were described by Hoar and Cottle (1952a) and Hoar and Dorchester (1949). Those fed on fats with high melting points were most resistant to high temperature, but no direct relation between resistance and the degree of hydrogenation of the lipids was found. The addition of cholesterol and phospholipids to diets of

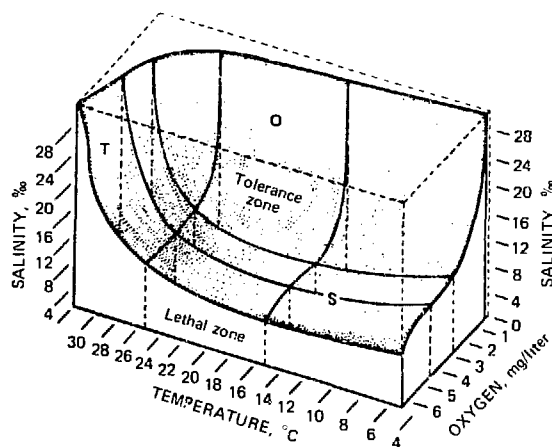


FIG. 10 Diagram of the boundary of lethal conditions for lobsters for various combinations of temperature, salinity, and oxygen. T is the region in which temperature alone acts as a lethal factor, S is the region in which salinity alone acts as a lethal factor, and O is the region in which oxygen alone acts as a lethal factor. (Adapted from McLeese, 1956.)

goldfish increased thermal tolerance, and the relative effectiveness of the treatment varied seasonally and with the duration of feeding (Irvine, Newman, and Hoar, 1958). Sarcophagid dipteran larvae (*Pseudosarcophaga affinis*) fed on diets containing higher saturated fatty acids had a greater degree of saturation of body lipids and an increased tolerance to high temperatures (House, Riordan, and Barlow, 1958). The effect of fats in the diet on thermal resistance may be related to the integrity of plasma membranes, where the degree of unsaturated lipids in the membrane structure may change its permeability (Hoar and Cottle, 1952b). The percentage of body fat may be an index of short-term heat-induced environmental stress in ectotherms (Falke and Smith, 1974). Also, the malnutrition that may result from exposure to thermal effluents because of increased metabolic rate and shortage of food organisms (Graham, 1974) could result in a lowered resistance to thermal stress (Allen and Strawn, 1967).

### Body Water Content

Considerable evidence exists that thermal tolerance to high temperature is related to the reduction of free body water (Fry, 1958). Conditions leading to desiccation in insects are associated with elevated thermal resistance (Baldwin, 1954; Baldwin and Riordan, 1956). In the newt, *Notophthalmus viridescens*, an average dehydration of 18.7% of body weight resulted in a significant increase in the CTM (Hutchison, 1961). Although the highest daily tolerance to temperature of juvenile spotted salamanders (*Ambystoma maculatum*) in the field was associated with the period of greatest dehydration, Pough and Wilson (1970) found that in the laboratory desiccation reduced the CTM. Reductions in body water content during acclimation to increased temperature have been described in goldfish (Hoar and Cottle, 1952b), amphibians (Jørgensen, 1950; Schmidt-Nielsen and Forster, 1954; Hutchison, 1961), and a variety of other metazoans (Precht et al., 1974). Such a change in an amphibian is shown in Fig. 11. Changes in body water content are closely associated with changes in salinity; the increase in tolerance at elevated salinities may be due to the concomitant and necessary reduction in free water. Thus, in ectotherms, acclimation and tolerance to temperature is closely associated with factors governing osmoregulation.

### Oxygen

The physiological adjustment to environmental oxygen content can influence thermal acclimation and tolerance. The upper lethal temperature of lobsters (McLeese, 1956), rainbow trout, the roach, *Rutilus rutilus* (Alabaster and Welcomme, 1962), and other animals (Precht et al., 1974) can be lowered significantly by a decrease in dissolved oxygen. Similarly, low oxygen content can prevent acclimation. For example, Brett (1946) found that insufficient oxygen prevented acclimation to high temperature in the goldfish (Fig. 12), and Sprague (1963) found similar results in four freshwater crustaceans.

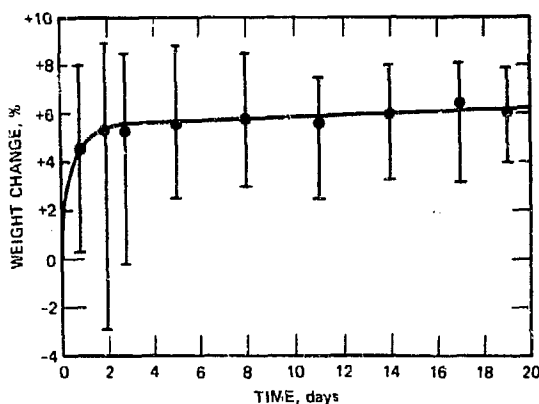


FIG. 11 Change in weight due to increased body water content in newts (*Notophthalmus viridescens*) transferred to 4.0°C after acclimation at 20°C. •, mean of each sample. †, range of observations. (Adapted from Hutchison, 1961.)

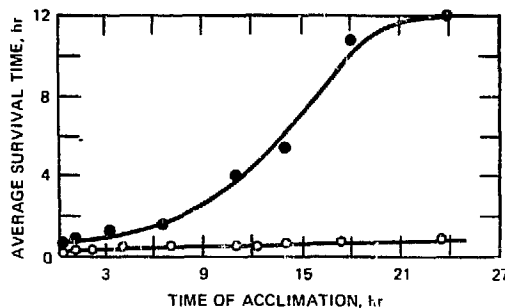


FIG. 12 Influence of oxygen tension on the rate of thermal acclimation in the bullhead, *Ameiurus nebulosus*. The sufficient-oxygen curve (●—●) indicates the average survival time when fish are moved from 20 to 28°C, and the insufficient-oxygen curve (○—○), the average survival time when they are moved from 17 to 27°C. (From J. R. Brett, 1946, *Rate of Gain of Heat-Tolerance in Goldfish (Carassius auratus)*, University of Toronto Studies, Biological Series 53, Ontario Fisheries Research Laboratory Publication 63, p. 18.)

### Carbon Dioxide and pH

The effects of dissolved carbon dioxide cannot be easily separated from the effects of hydrogen-ion concentrations due to the formation and dissociation of carbonic acid. A decrease in water temperature produces an increase in the ionization constant of water. Ectotherms regulate the concentration of  $H^+$  and  $OH^-$  ions at a relatively constant distance from the neutral point of water (which increases as temperature decreases) at all temperatures, and changes in blood pH result from changes in temperature (Robin, 1962; Rahn, 1967; Howell, 1970). This temperature-dependent pH change may have significant influence on oxygen transport to tissue, on metabolic control where enzymes are affected by pH (Behrisch, 1973), and, thus, on thermal tolerance.

## Temperature

Perhaps the most important and direct influence of all environmental factors on thermal tolerance is the thermal regimen to which an organism has been exposed before testing. When has an animal become fully acclimated to a new constant temperature? To be sure, we must determine the rate of acclimation to constant high and low temperatures for periods known or suspected to be more than adequate for complete acclimation; this usually takes at least several days or a few weeks for ectotherms. Animals are then transferred from high to low and from low to high temperatures, and the thermal tolerance is tested periodically (with due regard for possible diel cycles of sensitivity) until a new constant tolerance obtains (Fig. 13b, dashed curve). The rate of acclimation to high temperatures is

resulted in higher tolerance than did laboratory acclimation to constant temperature (Lowe and Heath, 1969). Leopard frogs (*Rana pipiens*) exposed to a daily cycle of 15 to 25°C after acclimation to 15°C had a faster rate of acclimation to the highest value of the cycle (25°C) than did animals transferred from acclimation at a constant 15°C to constant 25°C (Hutchison and Ferrance, 1970). After animals were exposed to only one 24-hr thermoperiod, the CTM was significantly higher than in those exposed to constant acclimation temperatures (Fig. 13). Similar increased scopes for thermal tolerance after acclimation to cyclic temperatures have been demonstrated in terrestrial isopods (Edney, 1964), the Mojave Desert pupfish (*Cyprinodon amargosae*) (Feldmeth, Stoue, and Brown, 1974), and the green-throated darter (*Etheostoma lepidum*) (Hubbs, 1964).

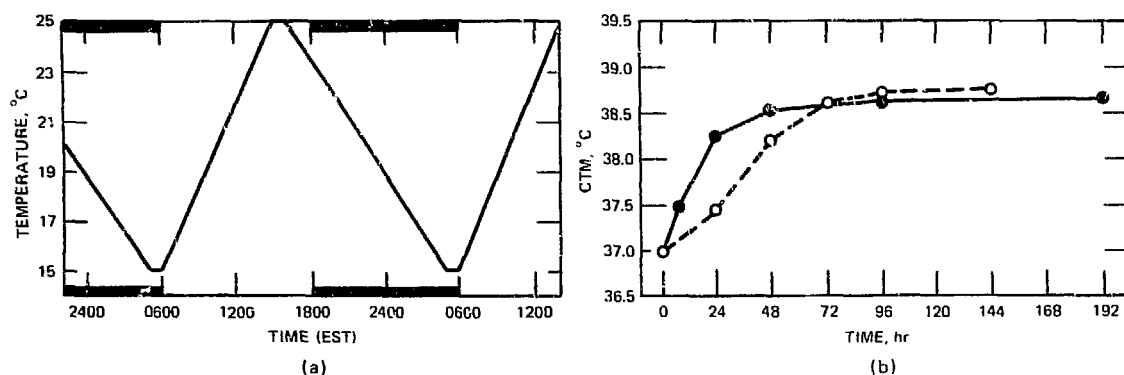


FIG. 13 (a) Pattern of programmed fluctuating temperature employed for 15 to 25°C cycle. Black bars show the scotophase of the L-D 12 : 12 photoperiod employed. (b) Rate of acclimation of the leopard frog (*Rana pipiens*) transferred from a constant acclimation at 15°C to constant 25°C (o---o) and from a constant 15°C to the daily cycle of 15 to 25°C (●—●). (From V. H. Hutchison and M. R. Ferrance, 1970, Thermal Tolerances of *Rana pipiens* Acclimated to Daily Temperature Cycles, *Herpetologica*, 26: 2, 3.)

usually significantly faster than that to low temperatures (Fig. 14). In fishes the acclimation rate to high temperatures is rapid; less than 23 hr is required at temperatures above 20°C. The gain in resistance to low temperature may require as much as 20 days for complete acclimation (Brett, 1944; 1956).

Most investigators have employed constant acclimation temperatures to determine acclimation rates, but such conditions seldom prevail in many environments. Except for certain deep-water benthic, troglodytic, and other forms in fairly isothermic environments, daily temperatures fluctuate, sometimes widely. There is now ample evidence that increased thermal resistance may result from short-term temperature fluctuations. Heath (1963) exposed cutthroat salmon to square-waved temperature cycles (thermoperiods) from 0.25 to 2.0 times the 24-hr time cycle and found the highest tolerance at the 24-hr thermoperiod (Fig. 15). This suggests a physiological adaptation to the more natural cycle. In the Sonoran desert pupfish (*Cyprinodon macularius*), the thermally cycled natural environment

A brief exposure to sublethal high temperatures may increase heat resistance; this "heat hardening" differs from "resistance adaptation" in being quick and usually transitory. Resistance adaptation usually refers to the more slowly developed and more persistent change in thermal tolerance to temperatures within the normal range. The difficulty is to distinguish hardening from the fast development of resistance adaptation. It has been suggested that transitory heat hardening may be a result of physiological stress occurring as a consequence of exposure to a high sublethal temperature. Thus heat hardening may be reasonable or paradoxical (Precht, 1967); i.e., increased tolerance may result from the animal's ability to resist further damage, or decreased tolerance may result from the stress (Precht, et al., 1974). There are few clear-cut examples of heat hardening, as defined by Precht. However, significant effects of sublethal exposures to thermal shock have been described in newts (*Notophthalmus viridescens*) by Hutchison (1961) and in a cyprinid fish (*Idus idus*), a cyprinodontid fish (*Xiphophorus helleri*), and a salamandrid

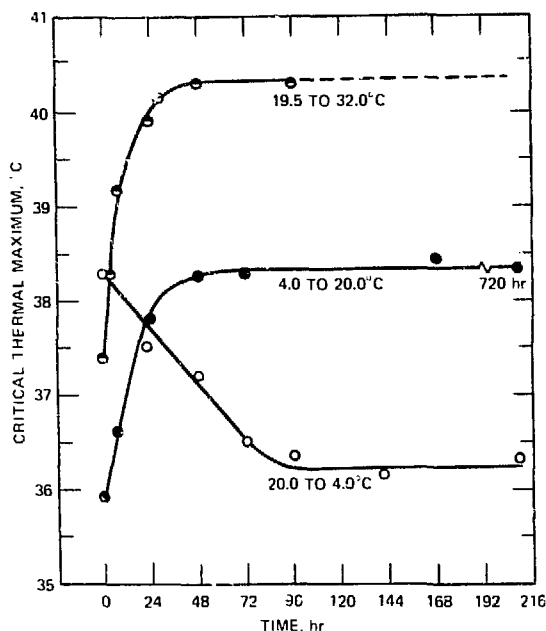
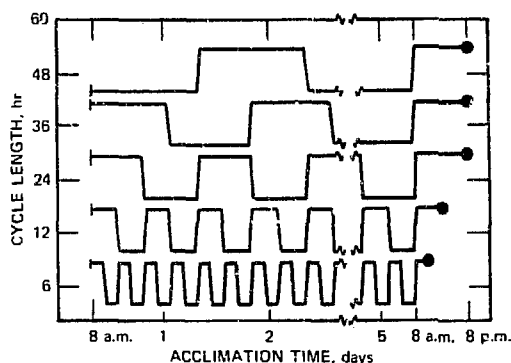


FIG. 14 Rates of acclimation in *Notophthalmus viridescens* transferred between different acclimation temperatures. Each point represents the mean of 6 to 10 animals. (From V. H. Hutchison, 1961, Critical Thermal Maxima in Salamanders, *Physiological Zoology*, 34: 99.)

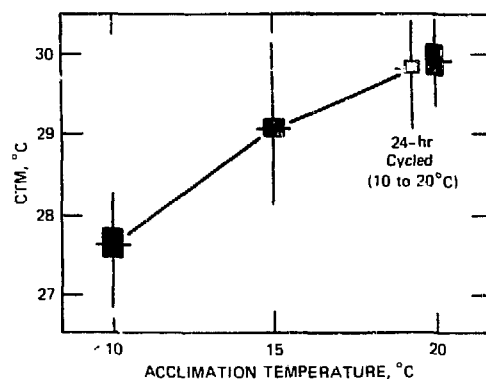
amphibian by Basedow (1969). In *Notophthalmus* an initial exposure to the CTM followed by a repeated exposure at various times after the initial shock resulted in an increase in the CTM on the second exposure. The increased tolerance reached a peak within 30 min after the first exposure and, after 65 to 71.5 hr, declined to a point where the CTM of the first and second determinations were not different (Fig. 16). Some of the invertebrates tested by Basedow (1969) showed decreased heat resistance for short periods after a shock from supraoptimal temperatures, but one species showed increased tolerance. Basedow was unable to distinguish between heat hardening and resistance adaptation, but his results do not rule out heat hardening.

### Behavior

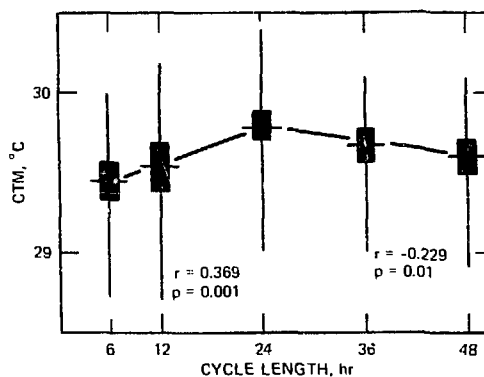
Stress caused by predation and competition or other interactions may well alter the limits of thermal tolerance in an organism. In the goldfish (*Carassius auratus*), the degree of stress from handling or sham injections is influenced by the acclimation temperature (Umminger and Gist, 1973), and death induced by sham injection procedures is more prevalent at low than at high temperature (Umminger, 1973). Reciprocally, exposures to a sublethal temperature shock may increase susceptibility to predation (Coutant, 1969). Little is known about such behavioral interactions and the resultant effects on the limits of thermal tolerance; this is a fruitful area for future investigations.



(a)



(b)



(c)

FIG. 15 Acclimation to square-waved temperature cycles (thermoperiods) and critical thermal maxima (CTM) in sea-run cutthroat salmon (*Salmo clarki*). (a) Patterns of cycled (10 to 20°C) acclimation. (b) Regression of CTM with constant-temperature acclimation (■) with CTM for 24-hr cycled acclimation (□). (c) Relation of CTM to length of acclimation cycles shown in (a). Short horizontal line shows the mean; (■) two standard errors on each side of the mean; vertical line, range;  $r$  and  $p$  are regression and probability of the slopes on each side of the 24-hr cycle length. (From W. G. Heath, 1963, Thermoperiodism in Sea-Run Cutthroat Trout (*Salmo clarki clarki*), *Science*, 142: 487.)



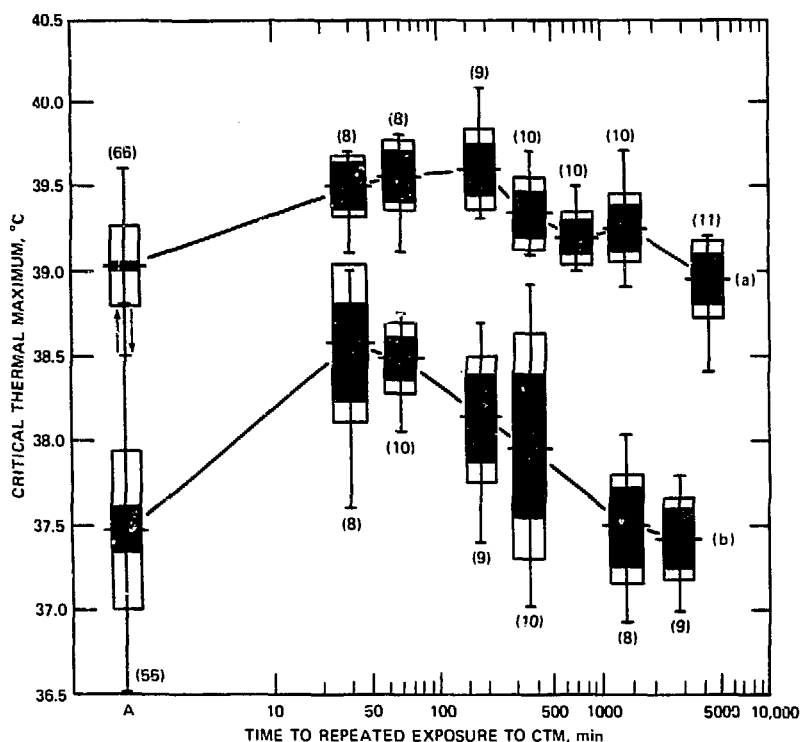


FIG. 16 Effect of a single repeated exposure to the critical thermal maxima (CTM) after different time intervals in the newt (*Notophthalmus viridescens*) acclimated at (a) 20.0°C and (b) 4.0°C. Range, mean, etc., presented as in Fig. 7 except the scale of the CTM is vertical (note that the time axis is logarithmic). (From V. H. Hutchison, 1961, Critical Thermal Maxima in Salamanders, *Physiological Zoology*, 34: 101.)

Normal well-coordinated behavior usually occurs in an area between temperature limits which is narrower than the area of thermal tolerance. Although many of the components of behavior are influenced by controlling factors (which are beyond the scope of this paper), the behavioral aspects of thermal selection and locomotor ability (as directive factors) are closely associated with tolerance to high temperatures in the natural environment. The ability of an ectotherm to detect temperatures below the lethal limit and to orient itself and move into areas with more optimal thermal conditions is obviously an important adaptation. Behavioral responses to thermal stimuli may be the result of learning and memory. In ectotherms the rate of learning is often slower at low temperatures, but the retention of learned material (memory) is better at low temperatures (Roots and Prosser, 1962; Prosser and Fahri, 1965; Gleitman and Rozin, 1971).

Certain activity is stimulated by a recent temperature change and is a response to a directive factor, but activity at a given constant temperature is part of the response to temperature as a controlling factor (Sullivan, 1954). Thus goldfish tested at the same temperature to which they were acclimated had cruising speeds different from those recorded for individuals acclimated at one temperature but

tested at another (Fry and Hart, 1948). Some examples of cruising speeds of fishes in relation to temperature are shown in Fig. 17. In animals acclimated to each test temperature, the maximum spontaneous activity occurs in the area of the preferred temperature (Sullivan, 1954; Fry, 1967).

The observation of thermal selection is very difficult in the field, and various gradients have been constructed to determine the preferred temperature of animals under a variety of conditions. The behavioral response of an animal to the gradient itself may influence thermal selection, and this possibility should be recognized. [See Fry (1971) and Precht et al. (1974) for discussions of the use of temperature-selection devices.] When presented with an environment containing a gradient of temperatures, most animals tend to concentrate their activity or spend more time in a distinct zone, the thermal preferendum (also referred to as eccentric temperature, thermal optimum, and preferred temperature). Since the distribution around the preferred temperature is often skewed toward higher temperatures, data should be reported to show the variation about the mean (Precht et al., 1974).

Most of the factors influencing thermal tolerances can also be expected to alter thermal selection. Preferred

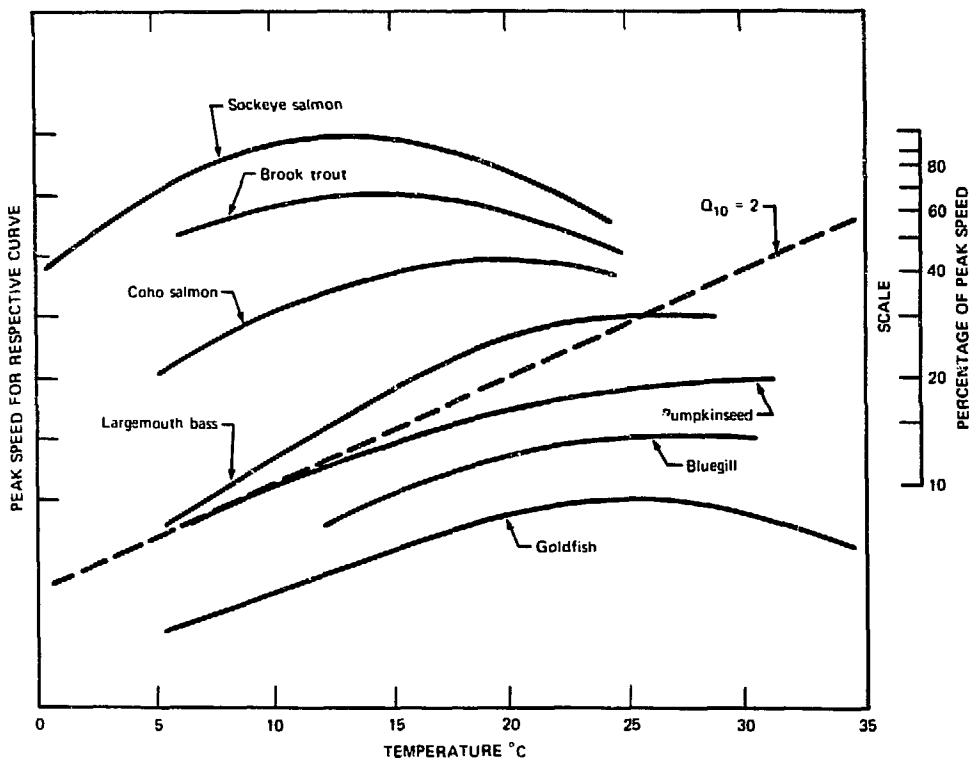


FIG. 17 Relative cruising speeds of various fish in relation to temperature when they are acclimated to the test temperature. The scale for percentage of peak speed is set to the peak of the topmost curve. Data were compiled from the literature, and the figure was constructed by Fry. (From F. E. J. Fry, 1967, Responses of Vertebrate poikilotherms to Temperature, in *Thermobiology*, A. H. Rose (Ed.), p. 391, Academic Press, Inc., New York.)

temperature is often related to age (Norris, 1963) or state of development. The thermal selections made by bullfrog tadpoles (*Rana catesbeiana*) at different stages of development and four acclimation temperatures are shown in Fig. 18. Precht et al. (1974) reviewed the alterations in preferred temperature by external and internal factors, including age, state of development, daily and seasonal changes, previous thermal history, moisture, starvation, the presence of other individuals of the same species, and illumination. In young Atlantic salmon, low concentrations of DDT caused a downward shift in preferred temperatures, and high doses caused an increase; the effect was greater for warm-water-acclimated than for cold-water-acclimated fish. These results suggested that DDT may interfere with the mechanism for thermal acclimation (Ogilvie and Anderson, 1965).

The heliothermic habits of numerous reptiles have led to several studies of the role of illumination and photoperiod as well as acclimation temperature and other factors in thermal selection (reviewed by Templeton, 1970). Water snakes (*Natrix erythrogaster*) subjected to a photoperiod of L-D 12 : 12 showed a daily rhythm of thermal selection with a maximum at 2100 hr and a minimum at 1200 hr. The rhythm was abolished under constant illumination (L-L), but the mean preferred temperature increased (Gehrman, 1971). Both photoperiod and light intensity influenced thermal selection in lacertid lizards (Spellerberg, 1974). In the lizard *Anolis carolinensis*, significant daily cycles of thermal tolerance occur under some combinations of photoperiod and acclimation temperature and can be phase-shifted by changing the photoperiod (Kosh and Hutchison, 1972). In the same species Hutchison and Kosh

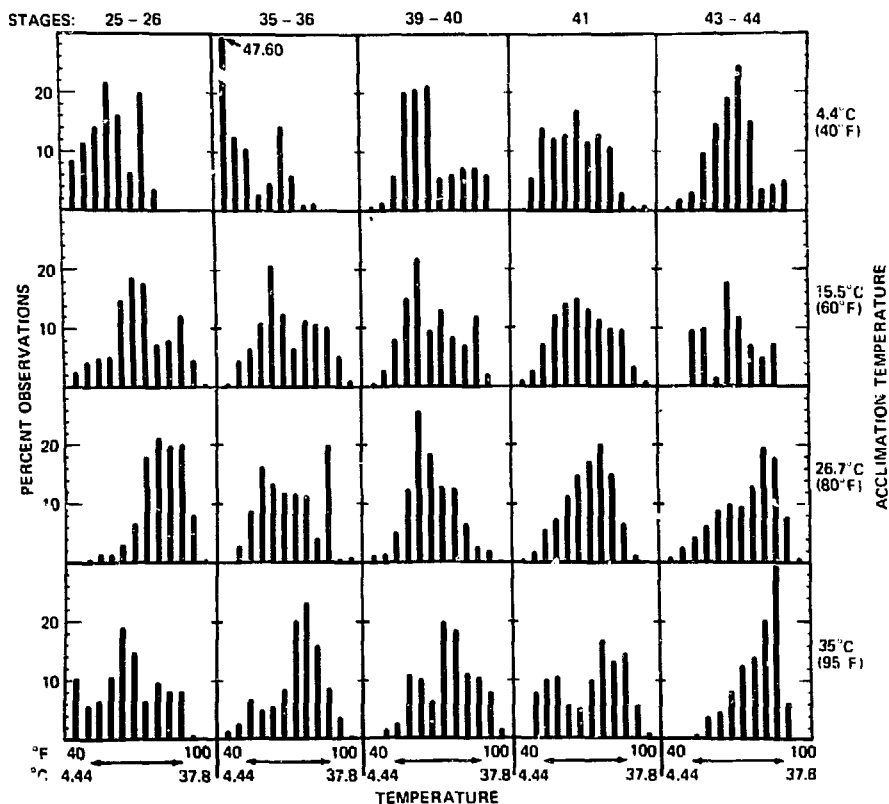


FIG. 18 Thermal selection of bullfrog (*Rana catesbeiana*) tadpoles at five different stages of development and four acclimation temperatures. The temperature range of the thermal gradient used in the study is shown at the bottom of each column. Stage numbers follow Gosner (1960). (From unpublished data of V. H. Hutchison and L. G. Hill.)

(1974) found a daily rhythm in preferred temperature. Parietalectomized animals selected significantly higher temperatures than controls except from 0800 to 1200 hr (Fig. 19). These studies suggest that the lizard parpineal plays a direct role in thermoregulation in addition to the indirect function as an illuminometer suggested by Stebbins and Wilhoft (1966). Similar studies on the role of the epiphyseal complex in thermal selection have apparently not been made in fishes.

## CONCLUSION

A plea is made for a better appreciation for the factors that influence the thermal tolerance of ectotherms and the determination of thermal resistance under experimental conditions. In nature animals continually adjust to fluctuations of environmental factors. Although many of these factors may be stabilized in the laboratory by maintenance of constant conditions for acclimation or acclimatization, many cannot. Since organisms have adapted to cyclic environmental conditions through the evolution of endogenous clocks, the exposure to constant conditions in the laboratory may produce results that have no meaning for an

animal in its normal environment. Although it is much more difficult to do, normal environmental cycles of the most important factors should be reproduced in the laboratory, and thermal selection and tolerance should be determined at different times during the cycle.

Since it is not possible to reproduce the entire natural holocoenotic environment and its variations in the laboratory, we should recognize that many environmental factors may modify thermal tolerance and its measurement. Adopting more standardized methods and terminology will greatly improve communications and will allow valid comparisons of data obtained by independent investigators.

The complexity of interactions of environmental factors suggests that many more multifactorial experiments incorporating both physical and biological variables are needed to "improve predictive powers of assessing ecological consequences of altered thermal environments" (Brett, 1969).

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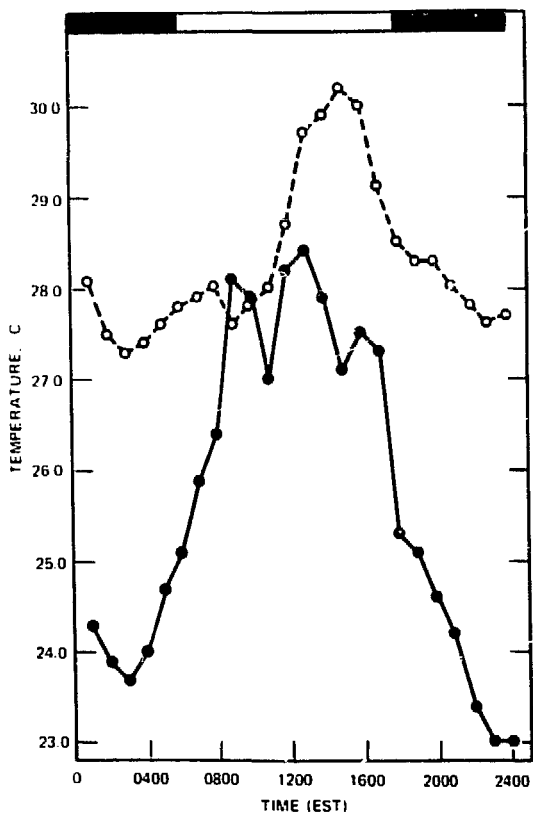


FIG. 19 Diel cycle of temperature selection in a thermal gradient in parietectomized *Anolis carolinensis* (○---○) compared with sham-operated controls (●—●). Photoperiod was L-D 12:12. The dark bars at the top of the figure show hours of scotophase, and the open bar shows hours of photophase. All animals were acclimated to a daily temperature of 15 to 25°C [as in Fig. 13(a)]. For parietectomized lizards each data point represents the mean of seven animals tested for 15 to 20 days and for controls, that for six animals for 9 to 11 days. (From V. H. Hutchison and R. J. Kosh, 1974, Thermoregulatory Function of the Parietal Eye in the Lizard *Anolis carolinensis*, *Oecologia (Berlin)*, 16: 175.)

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# Thermal Alteration and the Enhancement of Species Populations

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## ABSTRACT

The environmental energy supplement resulting from thermal loading is examined as a potential enhancement to plant and animal populations. Examples are given to show that the energy surplus is often converted into positive returns for populations. Establishing the conditions under which populations respond positively to artificial thermal elevation should be given equal importance with considering negative effects.

One approach to understanding how critical population parameters are influenced by thermal phenomena is to seek principles (i.e., common denominators) which apply generally across all or most populations. Principles of thermal ecology at the population level are often only hinted at in information presently available (Gibbons and Sharitz, 1974a; 1974b). Such information has often resulted in impressions that need to be defined as hypotheses and tested. This paper concentrates on one such impression, or major idea, which potentially can be framed as a testable hypothesis and supported by appropriate exemplary material.

At some point above the normal temperature range for the species, a population will begin to show negative responses to elevated temperature. The population's success will deteriorate progressively beyond some point, the shape of the negative-response curve and its point of initiation depending on complex environmental and species qualities. A thermal regime will ultimately be reached at which the species can no longer exist.

Despite this, an elevation in temperature is an energy supplement that seemingly could serve as an enrichment to a biological system. With few exceptions, plants and animals live in environments with fluctuating temperatures. Selection has eliminated or reduced the frequency of genetic sets that are enzymatically or physiologically incapable of adjusting to at least minor temperature shifts under natural conditions. Thus individuals of almost all species are preadapted to tolerate slight temperature increases, whether they result from natural variation or from artificially created thermal loading. As indicated, thermal loading is an added energy input to an ecosystem, and, as such, should be reflected positively in some parameters at the population level.

A general idea involving thermal alteration and species populations, then, is that an artificial increase in environmental temperature is, at some level, an enhancement to the population (Fig. 1). Since additional increase ultimately will be negative and the most positive responses may occur only in the initial stages of temperature increase, the shape of the curve can vary considerably. Also, the temperature units will vary with species, environment, and the particular parameter. The point of most positive response may be several degrees or only a fraction of a degree above ambient, depending on the situation. At some point, however, the thermal energy surplus will be converted into positive returns for any population.

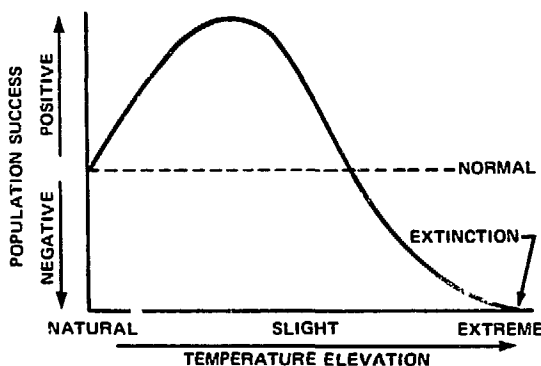


FIG. 1 Theoretical responses of plant or animal populations exposed to environmental temperatures above those naturally encountered. The ultimate negative response is preceded by an enhancement to the species population. The shape of the curve and the absolute temperatures associated with it depend on a complex of species and environmental qualities. The dashed horizontal line indicates the normal population condition for the parameter chosen to represent population success.

If we can establish that the energy supplement from thermal waste is positively reflected in most populations at some level, then identifying thermal conditions that result in the enhancement of critical population parameters in the economically or ecologically important species will become an issue in thermal ecology. This approach would complement the numerous studies currently establishing condi-



tions that adversely affect species. Basic information of both types is essential if biologically rational decisions are to be made on thermal manipulation of the environment. To support the population-enhancement idea, we must examine population responses to the initial, subtle temperature rises as well as to the more extreme situations resulting in obvious population reductions or eliminations. In comparison with the physiological literature, information on thermal effects on species populations is limited. However, several critical studies have been completed which lend credence to the enhancement idea; these are discussed in the following examples.

### THERMAL ELEVATION AND INCREASED PRIMARY AND SECONDARY PRODUCTIVITY

Positive population effects of elevated temperature occur through a variety of channels, but the precise mechanisms are often difficult to identify. In general, moderate temperature elevation in an aquatic ecosystem is accompanied by an increase in primary productivity. This increase provides a potential for more rapid turnover rates or higher standing crops in responsive producer species. A further potential is also created for an increase in primary and, subsequently, in secondary consumers and in secondary productivity at all levels.

Thermally induced physiological alterations that convert thermal energy into an enhancement for certain plant and animal populations are often poorly defined. In some instances, however, it is possible to identify the mechanism. The increase in metabolic rate that accompanies elevated temperatures provides an opportunity for the extension of seasonal or daily activity and increased feeding rates. This can be advantageous to a population by increasing the amount of time the species is effectively utilizing its environment. Higher food intake can result in enhanced growth and body size, which, in turn, may lead to earlier maturity and a higher reproductive output. An extension of daily and seasonal activity can result in more offspring through reproduction during normally nonreproductive periods. The following examples indicate that population enhancement often accompanies elevated temperatures although the mechanisms leading to the changes are more clearly understood in some instances than in others.

Tilly (1975) indicated that a major increase in primary productivity occurred in the thermally elevated region of the Par Pond Reservoir in South Carolina. The mechanism was not identified, but he noted that the increase in productivity corresponded to greater nutrient availability. Tilly's explanation for the increased availability of nutrients in the reservoir relies on circumstances peculiar to the particular system of Par Pond; however, a general application may exist in one component of his explanation. He suggested that observed increases in nutrient levels in Par Pond are partially a function of "progressive evaporative concentration of recycled cooling water." Thus the heat energy can be responsible for a greater concentration of

critical elements and permit greater exposure of the microbiota to these nutrients. Interestingly, plankton productivity was higher, but standing crop was not; this suggests an increase in turnover rate rather than biomass accumulation at the primary-producer level in this instance.

Another example of enhancement by temperature increase of primary productivity in algae through greater nutrient availability is given by Foerster, Trainor, and Buck (1974). They indicate that nutrient uptake and utilization were enhanced by an elevation in temperature through an increase in the rate of nutrient diffusion across the depletion zone. This resulted in more rapid algal growth in the thermally elevated sections of the Connecticut River. This study identified temperature as the promoter of a more efficient use of nutrients already available, whereas Tilly showed how temperature could increase the nutrient levels in the total aquatic system.

An example of increasing primary consumer population density as a result of thermal input, with an apparent direct effect on abundance of secondary consumers, is given by Storr and Schlenker (1974). The densities of amphipods (*Gammarus* sp.) increased an order of magnitude following temperature elevation in Lake Ontario. This is in accordance with Cooper (1965), who indicated that the amphipod *Hyaella azteca* grew significantly more rapidly and matured earlier as a function of temperature elevation. Concurrent with the increase in amphipod numbers in Lake Ontario was an increase in the number of white and yellow perch (*Roccus americanus* and *Perca flavescens*). *Gammarus* was found to be a major food component of both species of fish. This study does not document, or even suggest, the mechanism by which the numbers of primary consumers were increased. Nonetheless, secondary consumers in the particular area were directly enhanced in response to a greater availability of a critical species of primary consumers, which was a result, in some way, of the input of heated effluent.

Another primary consumer, *Scapholeberis kingi*, a zooplankton, has been shown to have a more rapid turnover because of an increased reproductive rate under heated conditions (Carlson, 1974). At a 5°C temperature elevation, the number of young produced per adult per day was 18% higher than at natural temperatures. The rate was 102% higher at 7°C above ambient but dropped to 32% lower than normal when water temperatures were elevated to 14°C above ambient. The enhancement model (Fig. 1) can readily be adapted to fit these particular data.

Some work with slider turtles, *Pseudemys scripta*, (Gibbons, 1970; Christy et al., 1974) suggests that thermal alteration may actually result in a shift in trophic level of this species from primary to secondary consumer. Once again, the precise mechanism by which the turtles incur a more rapid growth rate and larger size in thermal regions is unknown, but an apparent change from a predominantly herbivorous diet in natural populations to a more carnivorous diet in environmentally affected areas has been observed. If these interpretations are correct, they indicate

that greater nutrient levels and higher primary productivity in a system such as Par Pond can be reflected in the upper trophic levels through enhancement of certain population characteristics in some species.

An excellent example of an extension of seasonal activity leading to an overall increase in feeding rate is given by Tinsman and Maurer (1974). American oysters (*Crassostrea virginica*) placed in a heated effluent in Indian River Bay, Delaware, showed winter increases in growth and glycogen storage, whereas controls in natural-temperature areas ceased growing and declined in glycogen reserves. The differences were attributed to an artificial extension of the growing season. Although the oysters in the effluent during the summer showed greater loss of glycogen reserves and less rapid growth, the winter advantage resulted in an annual net advantage for animals living in the heated region.

Young and Frame (1975) likewise found that barnacles (*Balanus* sp.) in a discharge canal with elevated temperatures experienced optimal growth conditions during the winter. Growth rates of individuals in natural-temperature areas ceased during this period. Growth throughout the remainder of the year was similar at both sites.

Another example of seasonal extension of a critical population parameter is seen in mosquito fish (*Gambusia affinis*), which normally do not reproduce in winter (Krumholz, 1948). In a comparison of reproductive parameters between populations in heated and unheated environments, Ferens and Murphy (1974) found that a large proportion of female *Gambusia* from thermal areas contained eyed embryos, an advanced stage of embryonic development, during winter. No eyed embryos were found in the fish from natural-temperature sites. This, coupled with the observation of greater numbers of young fish in the heated areas, led Ferens and Murphy to conclude that this species produced more generations per year in thermal environments than in natural areas. This conclusion is further supported by the work of Goldner (unpublished data), in which habitat temperature was positively correlated with *Gambusia* brood size during summer.

Christy and Sharitz (1975) give an outstanding example of population enhancement in a plant species through an artificial extension of season. They compared growth and reproductive responses of *Ludwigia leptocarpa* and *Ammanium coccineum*, two semiaquatic shoreline species, growing at different water and sediment temperatures along the delta of thermal streams on the Savannah River Plant. The stand density and standing crop of both species were highest throughout the growing season in populations occurring at water temperatures of 35 to 40°C and lowest at normal water temperatures. At temperatures greater than 42°C, standing crop was less than that at intermediate temperatures but was higher than that under normal conditions. Also, the reproductive responses of the populations of both species were temporally different in the three areas. Both species flowered and fruited 4 to 6 weeks earlier at intermediate temperatures than they did at natural or at highly elevated temperatures.

These examples indicate that increases in primary or secondary productivity do occur as a consequence of elevated temperature. The challenges are to establish the level of generality at which this phenomenon holds true and to identify more precisely the mechanisms at the population level whereby the thermal energy is transformed into an increase in biological material at different trophic levels.

## SPECIES INTERACTIONS

Species interactions, although highly difficult to quantify, undoubtedly have an overwhelming influence on the dynamics of a particular species population. Although details of the mechanisms involved are not understood, some species are known to have inhibitory, controlling, or dampening effects on other species. Some of the more important interactions are competitive, predator-prey, or host-parasite relationships. Numerous examples exist of interspecific relationships in which a given pattern of dominance is altered by different environmental conditions. We can expect that in many instances the thermal regime will be identified as the primary environmental determinant of species success. One response to eliminating or reducing the controlling species could be dominance shifts or population increases in the host, prey, or competitor species.

Populations of certain species may also be enhanced by alteration of intraspecific interactions. Elevated temperatures may result in selection for genetically suited individuals of a population and subsequent elimination of those genetically unfit for the particular thermal conditions. This effectively enhances success of one genetic segment of the species population by eliminating intraspecific competitors.

Few studies have documented these phenomena in natural environments receiving thermal input, but the implication is frequent. A field example of possible competitive displacement as a result of a change in thermal regimes can be seen in the work of Sharitz (1975) with cattails (genus *Typha*). Two species occur throughout a reservoir (Par Pond) receiving heated effluent, but, in the ambient, thermally unaltered regions, one (*T. domingensis*) dominates in number and size of stands. In the heated region of the lake, another, *T. latifolia*, becomes more prevalent, eventually becoming the sole *Typha* survivor in the warmest sections. How much of the enhancement to *T. latifolia* populations is a function of increased temperature and how much to the removal of *T. domingensis* as a competitor is presently under investigation through reciprocal transplant studies, but the implication is that removal of *T. domingensis* results in a positive response from *T. latifolia*.

In laboratory experiments, Saks et al. (1974) combined populations of the ciliate *Euplotes vannus* with those of a foraminiferan (*Allogromia laticollaris*) and a nematode (*Chromadorina germanica*). At 15°C populations of *A. laticollaris* and *C. germanica* reproduced slowly when cultured alone and were rapidly eliminated in the presence

of *Euplotes*. At 25°C they did far better. Pure, control populations of *A. laticollaris* and *C. germanica* reproduced more rapidly at 25°C than at 15°C, and both were successful when cultured with *E. vannus*. Presumably *Euplotes* held a competitive edge over the other two species at the low temperature, but the advantage was lost at the higher temperature because of its own reduced success and the thermal enhancement of the other species.

No solid example of a species population being enhanced by the thermal elimination of a parasite species presently exists; however, two studies indicate the possibility of such a phenomenon. Bourque and Esch (1974) found that the numbers of trematodes (*Telorchis* sp. and *Macraevestibulum* sp.) were significantly reduced in slider turtles (*Pseudemys scripta*) taken from heated areas, regardless of the habitat type. Another example of a host-parasite relationship that is altered by thermal conditions is given by Cho, Gibbons, and Esch (this volume). A strigeid parasite (Trematoda) was abundant in mosquito fish (*G. affinis*) from natural-temperature waters but was totally absent in fish taken from the waters of a highly heated reservoir. Although there is no evidence to show that natural levels of these particular trematodes affect these species of fish or turtles, numerous examples can be found of the Trematoda causing harmful effects within natural host species (Spall and Summerfelt, 1970; Larson, 1965). Thus, if these were parasites that kept the numbers of the host species in check, the elevated temperatures would be an enhancement to the *P. scripta* or *G. affinis* populations.

As indicated earlier one mechanism of species enhancement is genetic selection for individuals tolerant of a particular thermal regime and environmental situation. Pinpointing population adjustment through genetic alteration is a challenging task and ultimately must be approached through documentation of changes in gene frequency. This may be accomplished implicitly by use of characteristics, such as morphological or physiological differences, for which the genetic basis of control is totally undefined. Sometimes, however, a more explicit approach is possible through use of a single-gene-controlled morphological character or an enzyme system known to serve as a genetic marker.

An example of possible selection for physiological thermal tolerance is given by Holland, et al. (1974). Bluegills (*Lepomis macrochirus*) taken from a population inhabiting high-temperature areas of a heated reservoir exhibited significantly higher critical thermal maxima than individuals from lower temperature habitats. Fish from the heated area also had dramatically lower mortality rates during acclimation. One explanation for the observed phenomenon was that selection had in some way favored individuals capable of tolerating the abnormal temperature conditions. This genetically redefined population, if indeed that was the case, would thus be spared from intraspecific competition with individuals of the physiologically intolerant segment of the population, which ultimately would be eliminated or significantly reduced in numbers.

An attempt to locate a thermally sensitive genetic marker by electrophoretic analysis of enzyme systems was largely unsuccessful in one study of several selected species of fish (Yardley et al., 1974). However, one system, malate dehydrogenase (MDH), in largemouth bass was implicated as a possible reflector of thermal regime. Frequency of one MDH allele in three populations was 0.44, 0.58, and 0.66. The habitat in which the 0.44 frequency occurred had a panmictic population that rarely encountered elevated water temperatures. The habitat in which the 0.66 frequency occurred had consistently higher temperatures and a much greater proportion of thermally influenced area. A slight selective advantage for a particular MDH allele was inferred as a result of the differences in thermal regime between the areas. The intermediate (0.58 frequency) population inhabited a normal-temperature lake that was several years post-thermal. The implication was that a reversal in selection pressure accompanied the shift from thermal back to natural-temperature environment; so allelic frequencies in the post-thermal lake were intermediate.

Perhaps the best example of genetic selection resulting from an artificial temperature elevation has been reported for the mummichog (*Fundulus heteroclitus*) along the northeast coast of New York and New Jersey (Mitton and Koehn, 1975). Allelic frequencies in several polymorphic loci were found to differ significantly in the fish populations taken from the outfall region of a power plant, where mean temperatures were as much as 16°C higher than in eight control habitats in the vicinity. Among the observed changes was an increase in a malate dehydrogenase (MDH) allele in the heated population. Comparison with a more southerly population revealed a near fixation of this allele, whereas the control populations had low frequencies. The inference is that the MDH locus in this species, as in the bass in the study of Yardley et al., was responsive to changes in environmental temperatures in some manner sufficient to alter allelic frequencies. Further indications of an adaptive response to temperature elevation were seen in allelic frequency changes in the lactate dehydrogenase and esterase-4 loci. Again, populations from the more southern, warmer habitat were similar to those from the artificially warmer environment when contrasted with populations of the eight control areas. Mitton (1973) also noted changes in certain morphological features of this species living in the thermal environment which were characteristic of those found in populations from the habitats further south. Genetic change at the population level could be a major consequence of thermal alteration and is deserving of an intensive research effort.

## CONCLUSIONS

Selected examples indicate a variety of mechanisms whereby species populations may be enhanced by an artificial elevation of environmental temperatures. The temperatures invoking positive responses in different populations may vary both absolutely and in their level above ambient. An explicit feature of the population-

enhancement idea is that different species will respond positively to temperature increases at different points along the spectrum of thermal change from a subtle rise to a major one. Attempts to establish the thermal condition most advantageous for a particular species will be confronted with typical biological variability, but it is essential that thermal models be designed to identify positive thermal regimes for each species, as well as negative ones.

In many instances an artificial elevation in temperature will negatively alter the total environment. On the way toward a negative response, however, most populations will pass through a positive-response phase. Identification of this phase is as critical as identification of the conditions resulting in negative response or death. A compromise must be reached between economics and environment. If the environment must be altered, our best approach is through an understanding of both the positive and the negative outcomes that are possible. Thus, by appropriate manipulation of a given set of conditions, we can, in some instances, convert an environmental liability into an asset.

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# Heated Waste-Water Effects on Aquatic Ecosystems

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## ABSTRACT

Subtle perturbations of aquatic ecosystems caused by heated waste-water discharges are difficult to document because: (1) They are difficult to distinguish from normal successional changes. (2) Normal variability or "noise" masks them. (3) Adequate background data on either ecosystem structure or function are rarely available. (4) Heated waste-water discharges are often associated with other stressful conditions, such as chlorination, addition of heavy metals, etc. (5) Since in the past the burden of proof of "no harmful effect" has been more on regulatory agencies than on dischargers, there was little incentive for gathering appropriate ecological data. (6) The academic community was not prepared for this assignment and frequently gathered inappropriate data.

The prospects for determining acute gross changes in aquatic community structure are excellent. Three case histories are discussed, including an example where corrective action "reversed" a structural change. Use of cluster analysis and other techniques should allow the detection of long-term chronic changes in structure despite natural variability and successional processes. Although functional assessments are available (e.g., nutrient and energy transfer), more studies of the effects of pollution on function are needed to increase their utility. Examination of biological and ecological information redundancy is also essential to increase the cost effectiveness of the data-gathering process. Three criteria important in determining whether a system can assimilate heated waste-water discharges without being degraded are: (1) elasticity, the ability to recover from damage, (2) inertia, the ability to resist displacement, and (3) resiliency, the number of times a system can snap back after displacement.

The ability to predict the effects of heated waste-water discharges on a particular aquatic ecosystem would undoubtedly be enormously useful in this era of escalating power generation. The inherent complexity of ecosystem analysis is readily apparent, and very few studies with an adequate data base can be found to document effects on a given ecosystem. Attempts to extrapolate from the scanty information available are generally unsuccessful.

This paper reviews three studies that do, however, indicate the potential predictability of ecosystem studies, discuss current ecosystem assessment, outline the major

features of ecosystem vulnerability, and propose a rating system for the critical factors.

## CASE HISTORIES

### Savannah River

In 1951 the Limnology Department of the Academy of Natural Sciences, under a contract with E. I. duPont de Nemours and Company, began a series of ecosystem studies at the Savannah River Plant, Aiken, S. C., which have continued intermittently throughout the years until the present time. The major publication resulting from these studies was by Patrick, Cairns, and Roback (1967).

In addition to the ecological surveys, a further line of evidence was provided by diatometers (Patrick, Hohn, and Wallace, 1954), which were positioned in areas close to or identical to those in which the main surveys were carried out. These furnished information on a continuing basis (every few weeks) throughout the entire period of the studies. Never during the course of the river study were any significant ecosystem effects noted which could be attributed to the waste-water discharge from the nuclear production reactors of the Savannah River Plant, but a number of scientists have reported effects within the project site itself (some reports are in this volume). Succession of species occurred, populations varied, and the kinds of species inhabiting the river changed, but these changes were in no identifiable way different from those which occurred at the upstream control or reference station. It is also important to note that the overlap in species, whether for a particular time between stations or at a single station through time, was ~25% and substantially larger for other groups but was never 90 to 100% (Patrick, Cairns, and Roback 1967). Thus comparisons of the species present at one time or place with those present at another were not useful.

### Potomac River

The Potomac River studies were designed to assess the biological impact of heated waste-water discharges from the Potomac Steam Electric Power Plant at Frederick, Md. Two preoperational surveys were carried out in 1955 and 1956, one during high-water conditions and one during low-water

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conditions. Postoperational surveys were carried out in 1959, 1960 (two), 1961, 1962 (two), 1963, 1964, 1965, and 1966 (see Kaesler and Cairns, 1972). The results for these surveys were essentially similar to those for the Savannah River in that significant deleterious effects were not noted. Since the heated waste-water discharges at this site changed the river temperature more markedly over a larger area than those from the Savannah River Plant changed the main Savannah River, degradational effects were more probable. This potential was enhanced by the fact that there were substantial habitat differences (the Savannah is a moderately deep river with a sandy bed, whereas the Potomac is a relatively shallow river with a rocky bed). In addition, population densities and species diversity were quite different in the two rivers. Nevertheless, deleterious effects were not noted in either. In 1967 the possibility of disclosing hitherto undetected effects by the use of cluster analysis appeared worth exploring, and an extensive series of papers resulted (e.g., Cairns and Kaesler, 1971). These studies verified the usefulness of cluster analysis and confirmed the absence of pollution effects.

The Potomac River studies posed the same general problems as the Savannah River studies, namely, when is there "no effect"? The evaluation was taken one step further, however, on the basis of a hindsight assumption that the analytical techniques originally used were not sufficiently discriminating to detect subtle changes. Accordingly, a substantial additional series of analyses was carried out in an attempt to be more precise and discriminating. All this effort merely substantiated the original determination. Since retroactive questioning of an assessment design is relatively simple, however, we can always find many measurements that were not made. The basic question is, How do we determine when to cease examining a problem if the original assessment design gives results that do not show degrading or deleterious change?

### Green River Survey

A preoperational and a postoperational survey for the Tennessee Valley Authority (TVA) Paradise Steam Plant on the Green River in Kentucky, was carried out by the Academy of Natural Sciences Limnology Survey Team in areas above and below the proposed heated waste-water discharge. The results of these studies have been discussed by Churchill and Wojtalik (1969). The academy study concluded that degradational effects on certain components of the aquatic community did result from the heated waste-water discharges, and, though not severe, these effects were considered significant. After cooling towers were installed, Churchill and Wojtalik (1969) carried out another evaluation not involving the academy team. They reported that the deleterious effects were eliminated and that the community was then quite similar in diversity and other characteristics to the preoperational aquatic community. Their study was very important because it showed that the assessment methods used in the Savannah and Potomac studies could detect ecosystem damage caused by

heated waste-water discharge. It also indicated that, when the damage was not severe, the effects were apparently reversible.

### OPTIMIZING ECOSYSTEM ASSESSMENT

Clearly, with the projected proliferation of steam electric-power-generating plants and the increased public recognition of environmental problems, pressures for ecological evaluations will increase enormously, even during periods of financial uncertainty. Although there are many sites where a substantial amount of environmental data have been gathered and extensive biological monitoring has been carried out both before and after plants began operating, a significant segment of the industry has, until recently, successfully resisted pressure for these investigations. As a partial result of this resistance, the body of information available for estimating and predicting ecosystem effects from heated waste-water discharges is totally inadequate for our present needs. In addition, the manpower and trained scientific personnel necessary to carry out the sophisticated assessments required to identify ecosystem effects is also inadequate. To optimize assessment of ecosystem effects, we must consider the national scope of the problem, the availability of trained personnel, the major categories of ecosystems receiving heated waste-water discharges, and a variety of other factors, including governmental legislation and the cost involved. A body of information is presently available which would help to determine priorities for this assessment process. A condensed list of considerations might include:

1. Number of plants presently operating, under construction, or projected for location on various types of ecosystems, e.g., oceans, estuaries, lakes, and streams.\*
2. Generating capacity and energy source (e.g., oil, coal, gas, or nuclear energy) of the various plants in each type of ecosystem.
3. Ecological vulnerability of the various sites.
4. Background information available for the various sites in question.
5. Availability of trained personnel for carrying out appropriate studies.
6. Number of plants for which the possibility of obtaining a variance from existing legislation is good, not likely, or highly unlikely.

This type of information should be expanded beyond this list, but these few criteria indicate that it is possible to organize the information available and to establish priorities for gathering information necessary to evaluate ecosystem effects. From this list urgent needs could be identified and situations selected which are highly likely to produce evidence from which broad generalizations could be made and from which assessment protocols could be established for other situations in each of the general categories. Thus

\*Some information related to number of steam electric power plants, etc., is given in Dickson et al., this volume.

ecosystem assessments could be programed in the most efficient way to obtain information that would be generic. It would increase the probability of using the limited numbers of trained personnel efficiently and would enhance proper coverage of the high-priority problems and vulnerable ecosystems. To carry out this program, we must have ecosystem evaluations and biological monitoring for a substantial number of plants. The evaluations will probably be less numerous than we would wish because sufficient trained personnel to carry them out will not be available in the near future. A series of decision matrices patterned after those given in *Principles for Evaluating Chemicals in the Environment* (National Academy of Sciences, 1975) could be developed and put into use. This would, for the first time, systematize data gathering on ecosystem effects resulting from heated waste-water discharges.

## ESTIMATING ECOSYSTEM VULNERABILITY

Although generalizations about ecosystem effects are useful, it would clearly be disastrous to use the same assessment methods to determine the effects of heated waste-water discharges on all types of water ecosystems. The mix of methods used should be determined on a site-specific basis using ecological criteria rather than those of administrative convenience. In addition to methodology employed, we also need to know how much effort should be expended and how frequently assessments are needed. Assessment efforts will largely be determined by the qualities or characteristics of the particular ecosystem involved. Such characteristics are:

1. Vulnerability to irreversible damage.
2. Degree of elasticity, or the ability to recover from damage.
3. Inertia, or the ability to resist displacement of structural and functional characteristics (i.e., ability to resist being placed into disequilibrium).
4. Resiliency, or the number of times a system can recover after displacement.

### Vulnerability to Irreversible Damage

Vulnerability is an exceedingly difficult characteristic to define because we may well hypothesize that, in time, all damage is reversible. It seems reasonable to be anthropocentric, however, and to define irreversible damage as that requiring a recovery period longer than the human life-span. All water ecosystems are not equally vulnerable to irreversible damage, nor are they subjected to the same intensity or frequency of perturbations of human origin (irreversible changes that are part of the normal evolutionary or successional processes are excluded from this discussion). Thus a committee of scientists with experience in studying a variety of water ecosystems should be charged with producing a rank ordering of vulnerability. Regulatory agencies should be able to estimate the frequency of societal perturbations. We should emphasize again that the degree of ecological vulnerability will depend to a large extent on an ecosystem's elasticity (its ability to snap back

after displacement) and inertia (its ability to resist displacement).

### Elasticity\*

Even if adequate preoperational studies and ecological monitoring programs are carried out, some deleterious effects are likely to occur because of accidental spills, such as those reported by Cairns et al. (1971). Some means of estimating the elasticity of an ecosystem is badly needed. This would be useful not only in determining accidental damage to an ecosystem but also in estimating the ecological vulnerability of various sites proposed for development. If an ecologist was included on a site-selection team, he could make some approximation of the relative vulnerability of the proposed sites and incorporate this into the series of factors considered in site selection. In addition, if severe ecological damage did occur, the financial penalty to the offending discharger might be based on not only the value of the organisms killed but also the length of time required for recovery.

The speculations that follow are based on studies of damaged rivers previously cited and on a report on recovery and restoration of damaged ecosystems (Cairns et al., in press). During the course of the recovery studies, it became evident that, although they are based on information derived entirely from freshwater systems, the factors involved are sufficiently general to apply fairly well to terrestrial, marine, and freshwater ecosystems. We should note, however, that the influence of residual toxicity or altered conditions would be markedly different for standing water, such as lakes, than for flowing water, such as rivers. Despite the obvious ecological disparity between various aquatic ecosystems, the factors in the following discussion are undoubtedly broadly encompassing.

The ordering of the factors important to the recovery process was deliberate, and the sequence in the simple equation that follows the list should not be altered. The most important factors are listed first and thus most strongly influence the final number. In view of our present state of substantive knowledge regarding the recovery process, it seemed ludicrous to attempt a more elaborate equation. On the other hand, it is quite clear that there is a "multiplier" effect between the factors involved and that some of the factors, such as management or organizational capabilities, are of little or no importance if the other components are weak. These then are the factors I consider important in the development of a recovery index, with a brief discussion of each.

a. *Existence of nearby biologic epicenters (e.g., for rivers these might be tributaries) to provide organisms to invade a damaged system.* In a series of studies carried out

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\*Elasticity is used instead of resilience as used in the superb paper by Holling (1973) to differentiate between societal displacement and natural perturbations. Similarly, inertia is used instead of stability to indicate resistance to displacement as a result of societal or technological perturbations.

on the Clinch River (see Kaesler, Cairns, and Crossman, 1974), in the initial stages of the recovery process some tributaries clearly contributed more colonizing organisms than others, and some of the organisms recolonizing damaged areas came from the headwaters of the main stream. The presence of comparatively healthy tributaries and headwaters was a key factor in the very rapid recovery that occurred. These areas might be considered biologic epicenters from which organisms departed to invade and subsequently colonize the recovery areas. If there had been fewer sources of potential recolonizing organisms, the recovery pattern of the Clinch would have undoubtedly been altered. Therefore, the existence of epicenters is a major factor in the recovery pattern of a damaged ecosystem, for without new potential colonizers the recovery process obviously cannot occur. (See Table 1 for rating system.)

*b. Transportability or mobility of dissemules (dissemules might be spores, eggs, larvae, flying adults that may lay eggs, or another stage in an organism's life history which permits it, either voluntarily or involuntarily, to move to a new area).* The Clinch River studies showed quite clearly that some groups of organisms have a greater potential for being reestablished in a damaged area than others. Fish, for example, moved into the damaged areas relatively soon after the occurrence of a fly-ash pond spill that destroyed the biota. The same was true for an acid spill that occurred sometime later but affected a more limited area. Aquatic insect larvae that "drift" downstream and are, therefore, good recolonizers, also were reestablished rather quickly. On the other hand, the mollusks were slow to reinvade damaged areas, and some species of mollusks had not returned to the damaged areas a few years after the last spill. If the damaged community consisted almost entirely of organisms with a high degree of dissemule transportability, the prospects for rapid recovery would be high, but, if it consisted primarily of those not easily transported, and thus less likely to reinvade, the prospects of rapid recovery would be poor. (See Table 1 for rating system.)

*c. Condition of the habitat following pollutional stress.* The fly-ash pond spill and the acid spill in the Clinch River

had marked effects on the indigenous biota but only small short-term effects on the physical habitat of the river and the chemical characteristics of the water. On the other hand, extensive siltation blanketing the riffle areas would have resulted in marked alteration of the habitat which might have persisted for a substantial period of time. (See Table 1 for rating system.)

*d. Presence of residual toxicants following pollutional stress.* The two Clinch River spills changed the pH of the river, the first raising it and the second lowering it, but in both cases the "slug" of water differing markedly from the ambient pH passed through the river system leaving no residual effects. On the other hand, the intrusion of biocides (e.g., dieldrin, aldrin, mercury, or lead) would almost certainly have left residuals that would probably have persisted in the system for a considerable length of time. The presence of residual toxicants might well impair the recovery of a damaged ecosystem by maintaining toxic conditions unsuitable for potential colonizing organisms and thus diminish the recoverability potential of a system. (See Table 1 for rating system.)

*e. Chemical-physical environmental quality following pollutional stress.* In some cases pollutional stress will place the chemical-physical environment of an ecosystem into severe disequilibrium either by altering the substratum or other portions of the system or by eliminating certain biota that could affect chemical-physical environmental quality. For example, a reservoir or lake with a substantial algal growth may normally have a dissolved oxygen concentration at saturation during daylight hours and well above 2 or 3 ppm even during the longest periods of darkness. If these plants were eliminated, however, the additional organic load that would be added from decay, together with the absence of the plants as a source of oxygen, could alter certain parts of the system from aerobic to anaerobic. This change might be of considerable duration if recovery were left entirely to natural processes. If we assume that recovery involves a return to approximately the original condition, the system could be considered either displaced or in disequilibrium. In systems like the Clinch River, where the flow-through rate was quite high, the restoration process required only a few hours because the offending materials were rapidly removed from the original spill site. On the other hand, a substantial portion of the river biota was damaged during the passage of the slug. The return to approximately the original chemical-physical conditions is an important prerequisite for the reestablishment of a community characteristic of that particular locality. (See Table 1 for rating system.)

*f. Management, or organizational capabilities for immediate and direct control of the damaged area.* In some cases river drainage authorities or other management groups capable of aiding the recovery process exist. For example, if an oxygen disequilibrium exists in a reservoir or lake, an approximation of the normal oxygen regime can be achieved by artificial aeration (see Fast, in press). This would presumably enhance conditions for reestablishing

TABLE 1 Rating the Critical Factors in Elasticity

| Factor                           | Rating                |                      |               |
|----------------------------------|-----------------------|----------------------|---------------|
|                                  | 1                     | 2                    | 3             |
| Existence of nearby epicenters   | Poor                  | Moderate             | Good          |
| Transportability of dissemules   | Poor                  | Moderate             | Good          |
| Condition of habitat             | Poor                  | Moderate             | Good          |
| Presence of residual toxicants   | Large amounts         | Intermediate amounts | Small amounts |
| Chemical-physical water quality  | Severe disequilibrium | Partially restored   | Normal        |
| Regional management capabilities | None                  | Intermediate         | Strong        |



populations of organisms characteristic of the system and thereby enable the natural balance to be restored more quickly than it might be otherwise. The cleanup of oil spills (excluding ecologically damaging cleanup methods, e.g., Nelson-Smith, in press) by organizations charged with the management of a specific ecosystem is another example of an activity that aids the recovery process. Reintroduction of certain types of species not likely to return to the area on their own is still another rather simple example of management intervention. Probably the most valuable contribution of a management organization is to establish baseline or "normal" conditions so that the degree of disequilibrium can be documented when an accident occurs. When the displacement from normal is known, the necessary corrective steps are usually clear and the resources available to aid the recovery process can be more efficiently directed to achieve the desired goals. As organizations charged with ecosystem management become politically and operationally stronger, their role in the recovery process will become increasingly important.

These corrective actions are relatively simple and straightforward. Knowing which corrective action is appropriate, however, requires a substantial knowledge of the system and a pool of background information regarding normal conditions. This is probably where most management groups fail. (See Table 1 for rating system.)

Using the characteristics discussed (which must be placed into the equation in exactly the sequence in which they are given), we arrive at a rather crude approximation of the probability of relatively rapid recovery. This would mean that under optimal conditions somewhere between 40 and 60% of the species might be reestablished in the first year after a severe stress, between 60 and 80% the second year, and perhaps as many as 95% of the species by the third year. Natural processes, with essentially no assistance from a management or a river basin group, accomplished this after the Clinch River spills (Kaesler, Cairns, and Crossman, 1974). The usefulness of this estimate has also been confirmed with data provided by studies of acid mine drainage (Herricks and Cairns, 1974), and the estimate seems adequate in the present context as well.

$$\text{Recovery index} = a \times b \times c \times d \times e \times f \quad (1)$$

A product of 400+ indicates that the chances of rapid recovery are excellent; 55 to 399, fair to good; and less than 55, poor.

Considerably more complicated equations were considered and rejected because the refinements seemed meaningless in view of our present state of knowledge. On this basis we might reject even the simple effort made, but there seems to be a definite need to formalize the estimation of recovery. I hope that more precise equations, properly weighted, will evolve from this modest beginning.

### Inertia

For the purposes of this discussion, inertia is defined as the ability of an aquatic community or ecosystem to resist

displacement or disequilibrium in terms of either structural or functional characteristics. As was the case for the recovery index, the factors are listed in order of importance, although this rank ordering is substantially less scientifically justifiable than that for the recovery index. It seems quite clear, however, that the factors are interacting and the justification for multiplying the ratings is considerably stronger than it was for the rank ordering. The time element is exceedingly difficult to address when we evaluate inertia because there may be a substantial lag between the onset of stress and the subsequent symptoms of displacement recognizable to ecologists. Therefore we should view this index with considerably more caution than the recovery index, which is only used after recognizable displacement.

Although time lag and displacements much removed from the onset of stress are a serious problem, they are not as hopeless as they might at first appear to be. We can make estimates of the probability of effects that do not surface initially by going through the protocol outlined in: *Principles for Evaluating Chemicals in the Environment* (National Academy of Sciences, 1975). Such procedures, with some modification of the methodologies used, enable us to categorize other stresses and to estimate the probability of a lag response. A brief discussion of the primary factors producing inertia follows.

*a. Indigenous organisms accustomed to highly variable environmental conditions.* We intuitively expect organisms in an estuary, in certain deserts, and in other environments where temperature and other environmental conditions may shift rapidly to have developed various physiological, behavioral, and structural capabilities for resisting the deleterious effects of the stresses. Some but not all these mechanisms (e.g., the ability of a clam to isolate itself from the environment for brief intervals) provide protection against both naturally occurring stresses and those induced by human activities. Heilbrum (1943) noted that coral reef organisms inhabiting a comparatively stable environment may be killed at temperatures only a few degrees above those to which they are accustomed. Although it is not likely that we will be able to categorize all organisms in a system according to tolerance to variable environmental conditions, if we are knowledgeable about the system, we can make sound estimates of the degree of resistance to changing conditions the majority of the indigenous organisms might be expected to have. (See Table 2 for rating system.)

*b. High structural and functional redundancy.* As Odum (1969) has shown, in early stages of evolution, ecosystems are relatively simple and, as a consequence, have relatively low functional redundancy. Thus the loss of a particular species may well mean the loss of a particular trophic level or function because no other species capable of fulfilling this role is present. On the other hand, a complex, mature system in which the fractionation of activities and roles is substantial is likely to have a high functional redundancy that minimizes the loss of a single species. Although its

exact role or function may be lost and the species may not be entirely replaced or its role taken over by other species present, a substantial portion may well be. A system with a high functional redundancy should, therefore, be less vulnerable to the loss of a single component than one with low functional redundancy and, consequently, be better able to resist displacement of both structure and function. (See Table 2 for rating system.)

*e. Proximity to a major ecological transitional threshold (e.g., from a cold-water to a warm-water fishery).* With regard to thermal loading, we may say with reasonable confidence that the Columbia River is nearer a major transitional temperature threshold (which may cause the loss of salmonoid fisheries) than is the Savannah River (where thermal effluent would cause the loss of the channel catfish, bass and other organisms characteristic of that

TABLE 2 Rating the Critical Factors in Inertia

| Factor                                                  | Rating |                           |                              |
|---------------------------------------------------------|--------|---------------------------|------------------------------|
|                                                         | 1      | 2                         | 3                            |
| Indigenous organisms accustomed to variable environment | Poor   | Moderate                  | Good                         |
| High structural and functional redundancy               | Poor   | Moderate                  | Good                         |
| Mixing capacity                                         | Poor   | Moderate                  | Good                         |
| Chemical characteristics                                | Poor   | Moderate                  | Good                         |
| Proximity to ecological threshold                       | Close  | Moderate margin of safety | Substantial margin of safety |
| Regional management capabilities                        | Poor   | Moderate                  | Good                         |

*c. Mixing capacity.* Stream order, flow dependability, turbulent diffusibility, and flushing capacity, among other characteristics, all have to do essentially with the volume of water available for dilution, the rapidity with which a waste or other form of stress could be dissipated, and the rate at which it would be removed from the system. A system in which these characteristics are very dependable is less vulnerable to structural or functional displacement than one that periodically has substantial losses or reductions in one or more of these characteristics. In considering these characteristics relative to the inertial stability of a system, we should not lose sight of the fact that wastes not degraded or transformed but merely mixed and carried away will have an impact elsewhere (those that are degraded and transformed may also have an impact but it will be a lesser one). (See Table 2 for rating system.)

*d. Chemical characteristics.* The literature on water pollution is replete with illustrations of variations in the impact of toxic chemicals because of differences in water quality (hardness, pH, temperature, etc.). Some hard, well-buffered waters may substantially reduce the impact of many toxic materials, whereas very soft, well-buffered waters may not. If we have some knowledge of the causative factors of pollutional stress, we can estimate whether water quality will significantly affect the toxicity or stress dose-response curve. In addition to using the literature to prepare estimates, we should carry out site-specific tests with indigenous organisms. These will enable us to define the relationships among the organisms, water quality, and dose-response curves to a particular pollutional stress. (See Table 2 for rating system.)

water system). Long-term continuing studies are necessary to determine when a system is beginning to approach a major transitional threshold, and, for those systems where such studies do not exist, determinations will be extraordinarily difficult. Even when data do exist, the task will not be easy. It is clear that the inertial stability of a system approaching a major ecological transitional threshold will be less than one significantly removed from this point. (See Table 2 for rating system.)

*f. Regional management capabilities.* A drainage-basin management group having a water-quality monitoring program can protect against displacement or disequilibrium in two primary ways: (1) by on-going studies that enable the investigators and decision makers to know when a major ecological transitional threshold is being approached, and (2) by developing and maintaining an environmental-quality-control system with rapid information feedback to detect the onset of a pollutional stress that might cause displacement or disequilibrium (Cairns et al., 1970; Cairns, 1972).

To be effective, of course, this group must have the capability of taking immediate remedial action before the waste enters the receiving system—or, if it has already entered, of taking appropriate measures to reduce the impact on the system. For thermal loading the long-term studies would provide information about the system that would permit the identification of ecological transitional thresholds. Since power plants frequently use chlorine, slimicides, and other toxic materials, and since heavy metals may be found in the waste discharges, a biological—chemical—physical monitoring system that would enable

the group to detect the appearance of toxic materials and take swift remedial action is also important. (See Table 2 for rating system.)

$$\text{Inertial index} = a \times b \times c \times d \times e \times f \quad (2)$$

A product of 400+ indicates high inertial stability, 55 to 399, fair to good inertial stability, less than 55, poor inertial stability.

## Resiliency

Resiliency is defined as the number of times a system can snap back after displacement. An informative illustration of resiliency was provided by Dicks (in press) in his investigation of the effect of oil spills on the plants of a salt marsh in Great Britain. Simulating a series of rather small oil spills restricted to a limited area, Dicks found that the salt-marsh organisms could recover rapidly after two or three spills in reasonably close succession. Even four or five exposures in close sequence were tolerated, but, when 16 or 17 insults were given at about the same frequency and volume to a comparable area, there was no recovery—at least within a time span considerably longer than that in which recovery had been previously demonstrated. This suggests that the salt-marsh ecosystem had a reserve to carry it through repeated stresses. The number of stresses that could be tolerated was finite, however, even though the area exposed and the volume of oil spilled was kept constant for each exposure. Thus the resiliency of the system could be determined by a field bioassay of repeated exposure in which the exposure intervals and volumes of oil used were controlled, and no other extraneous materials were placed in the natural system. Perhaps there are other systems for which resiliency could also be easily determined, but it is quite evident that, for the Great Lakes, oceans, and other large and complex ecosystems, determining resiliency would be difficult. Recognizing the problem and making crude estimates of the degree of resiliency may be enormously useful in developing management plans.

## CONCLUSIONS

Present evidence on the effects of heated waste-water discharges on ecosystems are totally inadequate for both the present rate of discharge into aquatic ecosystems and the projected future development of steam electric-generating plants. We know from a comparatively few instances, such as the Green River study cited earlier, that acute ecosystem effects can be discerned with already established methods. It is probably also true that gross displacement of biological integrity can be detected by available methods, but it is not at all clear that the more subtle indications preceding gross disruption will be adequately detected by these methods. Clearly each site-specific case will require a different set of methods and assessments, but all these should be selected so that both the structural and functional characteristics of aquatic ecosystems are assessed. Also, the data should be generated in such a way that the response patterns of similar systems in different locations can be compared. Finally, the process of evaluation should

be somewhat standardized and, more importantly, should get more rigorous academic review than is now the case. As far as I know, there is not now in effect a national screening effort of preoperational and postoperational surveys of plans for evaluating the effects of heated waste-water discharge into water ecosystems.

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# Potential Influence of Thermal Effluents on the Production and Water-Use Efficiency of Mangrove Species in South Florida

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## ABSTRACT

Data on mangrove-water relations and photosynthesis and energy-balance relationships between solar and infrared radiation, wind, air and surface temperature, and humidity profiles in mangrove canopies in south Florida were synthesized in a simulation model of leaf energy exchange, water relations, and photosynthesis. The model calculates the diurnal courses of these processes for sunlit and shaded leaves at different levels in the canopy. The model includes only canopy microclimate interaction and does not include the possible effects of changing root temperatures. The data on which model parameters are based are reviewed, and results of simulated changes in air and ground-surface temperature are discussed. Changes in ground-surface temperature showed little effect. Several trends and patterns appear in the simulations. Daily net photosynthesis is suppressed throughout the year by air temperatures over 5°C above ambient. Peak production occurs in spring when solar irradiance is high and air temperatures are moderate. Increased air temperatures shift the diurnal pattern of photosynthesis in winter, when maximum photosynthesis occurs at midday, to the pattern of summer, when maximum photosynthesis occurs in early morning and late afternoon. Increased temperatures are expected to affect *Rhizophora mangle* most adversely, but photosynthesis in *Laguncularia racemosa* showed the greatest sensitivity to temperature changes.

Mangrove forests, which occupy ~75% of the shoreline in the tropical and subtropical areas of the world (Golley, Odum, and Wilson, 1962), are rapidly being destroyed as shorelines are developed for housing, agriculture, and recreation. Mangroves are also vulnerable to thermal effluents because they are commonly located in estuaries, which are desirable locations for power plants. The ecology and biology of mangroves has been the subject of recent

reviews by Macnae (1968), Lugo and Snedaker (1974), and Walsh (1974). Kolehmainen, Martin, and Schroeder (1975) discussed some of the changes in the mangrove ecosystem, and Miller (1975a; 1975b) considered some of the short-term physiological responses of mangroves to increased environmental temperatures from thermal effluents.

Interpreting the effects of increased temperature on various plant processes is difficult because the processes respond differently to temperature. Most plant physiological processes are temperature dependent and are suppressed by temperatures above or below an optimum. Optimum temperatures and high and low temperatures at which the process stops vary with the species and with the process. Increasing plant temperatures when temperatures are below optimum increases plant processes, but increasing plant temperatures when temperatures are above optimum suppresses the processes. When adequate water is available, net photosynthesis can increase with leaf temperature to the overall temperature optimum of its biochemical and physical processes; it decreases as leaf temperature increases above this optimum. Environmental temperatures vary sinusoidally, diurnally, and seasonally so that temperatures of different plant organs may be above or below the temperature optimum for the major process of that organ (i.e., for the leaf, photosynthesis and, for roots, water or nutrient absorption) at different times. Transpiration will continue to increase with leaf temperature unless water stress develops and stomates close. However, interactions between plant-water relations, photosynthesis, and air temperature can be such that, even though leaf temperatures are initially below optimum for photosynthesis, increasing air temperatures can increase water stress so that photosynthesis does not increase with increasing leaf

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temperature. The effects of thermal additions vary depending on the relation of environmental temperatures to the temperature dependencies of the processes. A simulation model can approximate these interactions and facilitate making predictions of plant response to changing environmental conditions. Simulation models can be used to perform experiments that would be difficult or impossible to complete in the field and to focus attention on processes or expectations requiring further study.

The simulation model used here is a synthesis of available physiological data (Miller, 1972; Moore et al., 1972; 1973; Miller 1975b; Miller, Hom, and Poole, 1975) such that information on different processes can be juxtaposed and interactions explored. The simulations discussed here are from a model using parameters developed from data for a scrub red mangrove ecosystem receiving thermal additions. Other versions of this model, using data from Mediterranean shrub and Arctic tundra ecosystems, have proven equally useful, however (Miller and Mooney, 1974; Miller and Tieszen, 1972; Miller, Stoner, and Tieszen, 1976). Because acclimation and ecotypic variations may occur in water relations and photosynthesis physiology, these can be represented only as simulations of our study area in south Florida. Using the same methods to measure certain relationships on mangroves of another area, however, we can make a reasonable simulation of that area. Our research site is located on South Biscayne Bay, ~10 km east of Homestead, Fla., and 35 km south of Miami. Radiation, wind, air-temperature, and humidity profiles; plant-water relations; and photosynthesis for three mangrove species (*Avicennia germinans*, *Laguncularia racemosa*, and *Rhizophora mangle*) were measured in the canopies at different times of the year during a 5-year period. The mangrove growing season at this location is almost 365 days per year. Water is available throughout the year, and temperatures are slightly below optimum for photosynthesis in winter and slightly above optimum in summer. At Miami incoming solar irradiance through the year varied from 325 to 575 cal cm<sup>-2</sup> day<sup>-1</sup>, with the highest rate occurring in April (Fig. 1). Mean daily air temperatures varied between 19 and 28°C, with the highest temperatures occurring in August. The optimum temperature for photosynthesis is 25°C (Moore et al., 1972; 1973), and leaf temperatures in spring, summer, and fall were commonly higher than this.

## MODEL BACKGROUND AND REVIEW OF DATA BASE

The primary production system is conceived in terms of a set of submodels, each describing the processes of different aspects of the vegetation-climate interactions. The regional climate is comprised of solar and infrared radiation, wind velocity, air temperature, and vapor density. The solar and infrared radiation interact with the canopy structure to produce profiles through the canopy of direct and diffuse solar radiation and infrared radiation. The air temperature, wind velocity, and vapor density interact with the canopy structure and with the radiation

profiles to produce profiles of air temperature and vapor density within the canopy. The solar and infrared radiation, air temperatures, and vapor densities at the bottom of the canopy affect the temperatures of the soil, depending on such soil thermal properties as thermal conductance and volumetric heat capacity. Solar radiation, air temperature, wind speed, and vapor density affect the leaf temperature and the rate of water loss from leaves. The soil moisture content and temperature affect the rate of water uptake through the roots. The leaf water status, absorbed solar radiation, and leaf temperature affect the rate of photosynthesis. The results discussed here include only plant-canopy microclimate interactions and indicate short-term responses to thermal effects. A summary of parameter values for the three species is given in Table 1.

The radiation model calculates the downward flux of total solar radiation, the direct beam of solar radiation, the diffuse solar radiation comprised of both diffuse radiation from the sky and that reflected from leaves and stems, the upward reflected solar radiation, the downward flux of infrared radiation received from both the sky and from leaves and stems in the canopy, and the upward flux of infrared radiation received from the ground surface and from leaves and stems in the canopy. Radiation measurements in the canopy were made in July 1972. Two solar

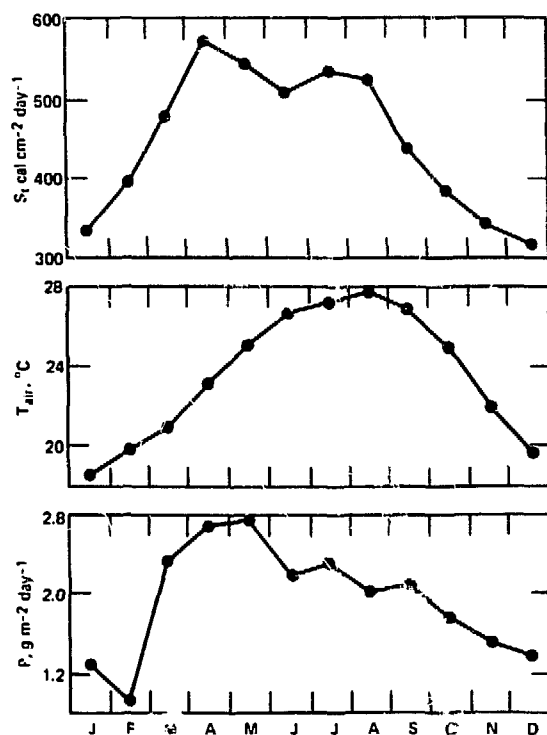


FIG. 1 (a) Seasonal progression of incoming solar irradiance ( $S_i$ ). (b) Mean monthly air temperatures ( $T_{air}$ ). (c) Total production ( $P$ ) of organic matter (grams per square meter of ground per day).

TABLE 1 Summary of Parameter Values Used to Define the Species in the Simulations

| Parameter                                                                                       | Species          |                     |                   |
|-------------------------------------------------------------------------------------------------|------------------|---------------------|-------------------|
|                                                                                                 | <i>Avicennia</i> | <i>Laguncularia</i> | <i>Rhizophora</i> |
| Leaf-area index, $\text{cm}^2/\text{cm}^2$                                                      | 2.0              | 2.0                 | 2.0               |
| Minimum leaf resistance, sec/cm                                                                 | 2.4              | 1.8                 | 2.4               |
| $\psi_l$ at $3 \times R_{\text{min}}$ , bars                                                    | -26.0            | -18.0               | -18.0             |
| $\psi_l$ at maximum leaf resistance, bars                                                       | -34.0            | -30.0               | -42.0             |
| Coefficients of leaf resistance vs. solar-radiation curve <sup>*</sup> :                        |                  |                     |                   |
| $a_1$                                                                                           | 1.02             | 1.02                | 1.02              |
| $a_2$                                                                                           | 0.0083           | 0.0083              | 0.0083            |
| $a_3$                                                                                           | 16.7             | 16.7                | 16.7              |
| Coefficient of leaf resistance vs. temperature curve <sup>†</sup>                               | 0.06             | 0.06                | 0.06              |
| Temperature at which stomatal openings are not inhibited ( $T_{\text{opt}}$ ), °C               | 25.0             | 25.0                | 25.0              |
| Coefficients of mesophyll resistance vs. light curve <sup>‡</sup> :                             |                  |                     |                   |
| $a_0$                                                                                           | 900.0            | 900.0               | 450.0             |
| $a_{10}$                                                                                        | -15.0            | -16.0               | -15.0             |
| Minimum mesophyll resistance, sec/cm                                                            | 48.0             | 22.8                | 45.0              |
| Temperature at maximum photosynthesis, °C                                                       | 25.0             | 25.0                | 20 to 25          |
| Resistance of root mass to water flow, $10^6$ sec/cm                                            | 142.2            | 144.0               | 84.0              |
| Coefficients relating leaf water potential to leaf relative saturation deficit: intercept, bars |                  |                     |                   |
| slope, bars/%                                                                                   | 0.0              | 0.0                 | 0.0               |
| Resistance of soil to water flow to root surface, $10^6$ sec/cm                                 | -2.0             | -1.0                | -0.7              |
| Extinction coefficient of eddy diffusivity, $(\text{leaf-area index})^{-1}$                     | 0.1              | 0.1                 | 0.1               |
| Air resistances at air-soil interface, sec/cm                                                   | 1.8              | 1.8                 | 1.8               |
| Relative humidity of air at air-soil interface, %                                               | 0.05             | 0.05                | 0.05              |
| Reflectance of the soil surface, fraction                                                       | 100.0            | 100.0               | 100.0             |
| Average soil water potential at root depth, bars                                                | 0.2              | 0.2                 | 0.2               |
|                                                                                                 | -12.0            | -12.0               | -12.0             |

\*The equation has the form  $\Delta r_l = a_1 (a_2 + a_3 \cdot S_{\text{abs}})^{-1}$

†When leaf temperature is less than optimum, the equation has the form  $\Delta r_l = a_0 (T_{\text{opt}} - T_l)^2$ ; otherwise  $\Delta r_l = 0$ .

‡The equation has the form  $r_{\text{int}} = a_0 + a_{10} (S_{\text{abs}})^{-1}$  for tundra and  $r_{\text{int}} = r_{\text{int}} + a_{10} e^{-a_{10} S_{\text{abs}}}$  for mangroves and chaparral.  $r_{\text{int}}$  is the minimum mesophyll resistance.

radiometers, one with a shading ring, and a total hemispherical radiometer were mounted facing upwards on a small cart to measure the downward fluxes of total solar, diffuse solar, and infrared radiation. A solar radiometer and a total hemispherical radiometer were also mounted facing downwards on the same cart to measure the upward fluxes of reflected solar and infrared radiation. The carts were drawn on a track through a 3-m traverse above the canopy and at four levels within the canopy at two locations. The radiation components were calculated and related to the leaf-area index above the level, the altitude of the sun, and the angle of the leaves. The model predicted the measured values within 20% (Miller, 1975b).

The model for canopy microclimate is an adaptation of the model of turbulent transfer in vegetation canopies proposed by Waggoner and Reifsnyder (1968) and Waggoner, Furnival, and Reifsnyder (1969). Wind, air-temperature, and humidity profile measurements were made in August 1970. Wind profiles were summarized in a negative exponential equation that calculates wind speed as a function of the vertical profile of leaf-area index. Although the assumptions of the model are not wholly fulfilled because of the variability in natural vegetation, the model predicted vapor-density profiles within  $0.5 \text{ g/m}^3$  and air-temperature profiles within  $0.5^\circ\text{C}$ . Predictions deviated more during the day than at night. The predictions were

poorer near the top of the canopy than within it. All the processes of energy exchange in the canopy (i.e., radiation, convection, and evaporation) affect plant processes by affecting the temperatures of sunlit and shaded leaves at different levels in the canopy. Leaf temperatures calculated in the model were within  $1^{\circ}\text{C}$  of measured values over a range of 12 to  $35^{\circ}\text{C}$  (Miller 1975b).

### Plant-Water Relations

The plant-water relations model characterizes the flux of water from the substrate through the roots to the leaves and from the leaves to the air. Control of the flux is by leaf resistance to water loss and root resistance to water uptake (Ehleringer and Miller, 1975; Stoner and Miller, 1975). Leaf resistance to water loss is related to either leaf water content or leaf water potential. The root resistance was constant in these simulations but will be made variable as a function of water temperature on the basis of experiments carried out by Hom (1976). To include leaf water stresses caused by unequal rates of water uptake and loss, we measured the relation between leaf water content and leaf water potential.

The water relations of the three mangrove species differ. *Avicennia* takes up salts from the substrate through the roots and excretes them at the leaves; *Laguncularia* and *Rhizophora* exclude salts at the roots (Scholander et al., 1966; Walsh, 1974). Total xylem and leaf water potentials and osmotic potentials (measured with a thermocouple psychrometer and a pressure bomb) of *Avicennia* are lower than those of the other two species (Miller, Hom, and Poole, 1975). Turgor is lost readily in *Avicennia* as leaf water content decreases, but it decreases slowly over a wide range of water contents in *Laguncularia*. In *Rhizophora* turgor is maintained as water content decreases until a threshold is reached. Zero turgor occurs at relative water contents of about 0.92 in *Avicennia*, 0.85 in *Laguncularia*, and 0.89 in *Rhizophora* (Miller, Hom, and Poole, 1975). The leaf water potential decreases as leaf water content decreases in all species. With the change in leaf water content, the change in leaf water potential is least in *Rhizophora* and greatest in *Avicennia* until leaf water potentials reach -40 bars (Miller, Hom, and Poole, 1975).

The leaf resistance increases as the water content decreases in both *Avicennia* and *Laguncularia*, but the increase is slower in *Laguncularia*. In *Rhizophora*, however, leaf resistance tends to decrease as water content decreases until relative water content is about 0.95. At lower water contents leaf resistance tends to increase.

### Photosynthesis

Measurements of photosynthesis on branches of the three mangrove species were made in February, May, and October 1972 (Moore et al., 1972; 1973). Measurements were made with a controlled-environment cuvette system, a gas analyzer, and a dew-point hygrometer. Measurements are most complete on *Rhizophora*. Because of the limited base for *Avicennia* and *Laguncularia* (measurements in May

were limited by small sample size), the seasonal progression of photosynthesis was simulated for *Rhizophora* only. Physiological changes in the photosynthesis responses of *Rhizophora* through the season were interpolated from the data available. The responses of the three species to thermal changes were compared using the October data for simulation with July conditions and using February data for simulation with January conditions. In the submodel photosynthesis is related to solar radiation, temperature, leaf resistance, and internal mesophyll resistance.

### Driving Variables

The annual progression of climatic variables to be used as driving variables was synthesized from weather data for Miami (National Oceanic and Atmospheric Administration, 1974). In the simulation experiments air temperatures at the top of the canopy and ground-surface temperatures were changed by +10, +5, +2.5, and  $-2.5^{\circ}\text{C}$  from the ambient conditions. Air and ground-surface temperatures were varied separately and together in the simulations.

## RESULTS AND DISCUSSION OF SIMULATION EXPERIMENTS

### Photosynthesis and Water-Use Efficiency Through the Season

Simulated production by *Rhizophora* was highest in April and May, at high solar irradiance and moderate air temperatures (Fig. 1). Production decreased through the summer as solar irradiance declined and temperature increased. In the fall, when temperatures were moderate, irradiance was low. The lowest production occurred in winter. The low in February was caused partly by seasonal shifts in photosynthesis responses.

The seasonal progression of vertical distribution of photosynthesis and water-use efficiency for *Rhizophora* indicates that photosynthesis is highest at the top of the canopy in the spring and fall, when temperatures are near optimum and solar irradiance is adequate (Fig. 2). Toward

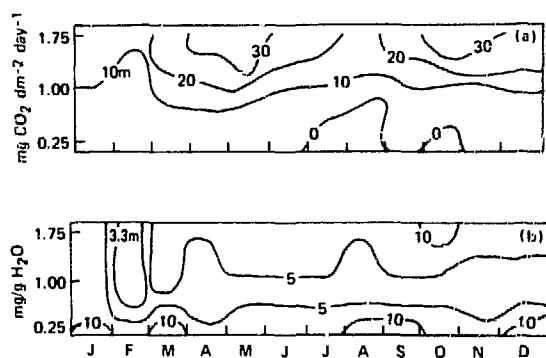


FIG. 2 Seasonal progression for a *Rhizophora* canopy in south Florida. (a) Photosynthesis (milligrams  $\text{CO}_2$  per  $\text{dm}^2$  of leaf per day). (b) Water-use efficiency (milligrams organic matter per gram  $\text{H}_2\text{O}$ ) through the profile. Vertical axis is height in meters.

the bottom of the canopy solar irradiance and photosynthesis rates decrease. During the summer temperatures throughout the canopy are above optimum and photosynthesis is suppressed at all levels, especially at the top, where solar irradiances are highest. The high solar irradiance increases the leaf temperatures, which are already above optimum. Photosynthesis within the canopy during the summer is almost zero because of the low solar irradiance and high temperatures. During the winter photosynthesis is depressed throughout the canopy by low solar irradiance and suboptimal temperatures. The water-use efficiency was highest in October and lowest in February and was higher within the canopy than near the top throughout the year.

### Photosynthesis Through the Season with Thermal Changes

The seasonal pattern of photosynthesis is changed with the addition of heated air or water (Fig. 3). At air temperatures 10°C above ambient, leaf temperatures during the day were above optimum for photosynthesis and net photosynthesis was negative during most of the year throughout the canopy; it was slightly positive only at the bottom of the canopy during spring. High solar irradiances

at the top of the canopy increased the leaf temperatures further. At the bottom of the canopy, however, solar irradiance was lower and temperature increases were less. Thus net photosynthesis is less negative at the bottom of the canopy. In addition to suppression of photosynthesis during the day by increased temperatures, respiration during the night is also increased; this further depresses the daily net CO<sub>2</sub> balance. When air temperatures were 5°C above ambient, photosynthesis was still suppressed in relation to the ambient conditions. Photosynthesis was positive at the top of the canopy in midwinter and throughout the canopy in spring. Total annual net photosynthesis was negative, however. With a 2.5°C rise in air temperature, photosynthesis was lower than for ambient conditions through most of the year but was increased in the winter. Photosynthesis was negative in February and during the summer. Air temperatures 2.5°C below ambient resulted in increased photosynthesis rates, particularly in spring, summer, and fall. Net photosynthesis was increased within the canopy in the summer because of the decreased nighttime respiration.

Increased soil-surface temperatures, which are not shown, had less effect on photosynthesis than did the increase in air temperature. Changes in soil-surface temperature affected leaf temperatures only in the lower levels in the canopy. Because the lower levels are less productive than the upper levels, total canopy photosynthesis is less affected by changes in soil-surface temperature than by changes in air temperature. Simulations in which both air and soil-surface temperatures were changed gave results only slightly different from those in which only air temperature was varied, and the results were almost additive (Miller, 1975a).

The simulations show that, because of the 365-day growing season, photosynthesis in *Rhizophora* is affected throughout the year by thermal additions. When temperatures are below optimum, photosynthesis is enhanced, and, when temperatures are above optimum, photosynthesis is suppressed. The net effect is to shift the active photosynthetic period slightly earlier in the season by increasing photosynthesis rates in March and April and depressing them in May (Miller, 1975a).

### Diurnal Courses of Photosynthesis

The diurnal progression of photosynthesis with ambient conditions and with 5°C increases in air and soil-surface temperatures show the enhancing or suppressing effects of increased temperatures during both winter and summer (Fig. 4). Under the ambient conditions in January, the highest photosynthesis rates occurred at midday. Throughout the day photosynthesis was highest at the top of the canopy and lowest within it. Under ambient conditions in July, photosynthesis was highest in early morning and late afternoon and was suppressed in the middle of the day by the higher temperatures. The effect of increasing air temperature by 5°C in January is to shift the January pattern to the July pattern, and the effect in July is to

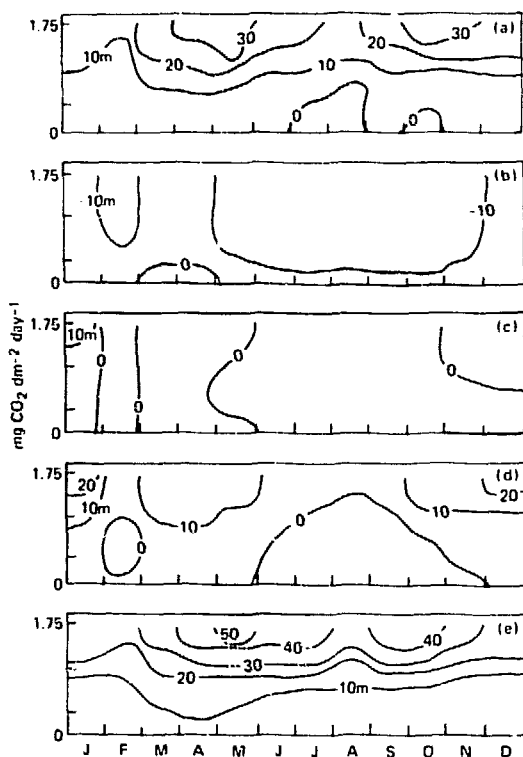


FIG. 3 Seasonal progression of photosynthesis (milligrams CO<sub>2</sub> per dm<sup>2</sup> of leaf per day) through the profile with air temperature (a) ambient; (b) ambient + 10°C; (c) ambient + 5°C; (d) ambient + 2.5°C; and (e) ambient - 2.5°C. Vertical axis is height in meters.



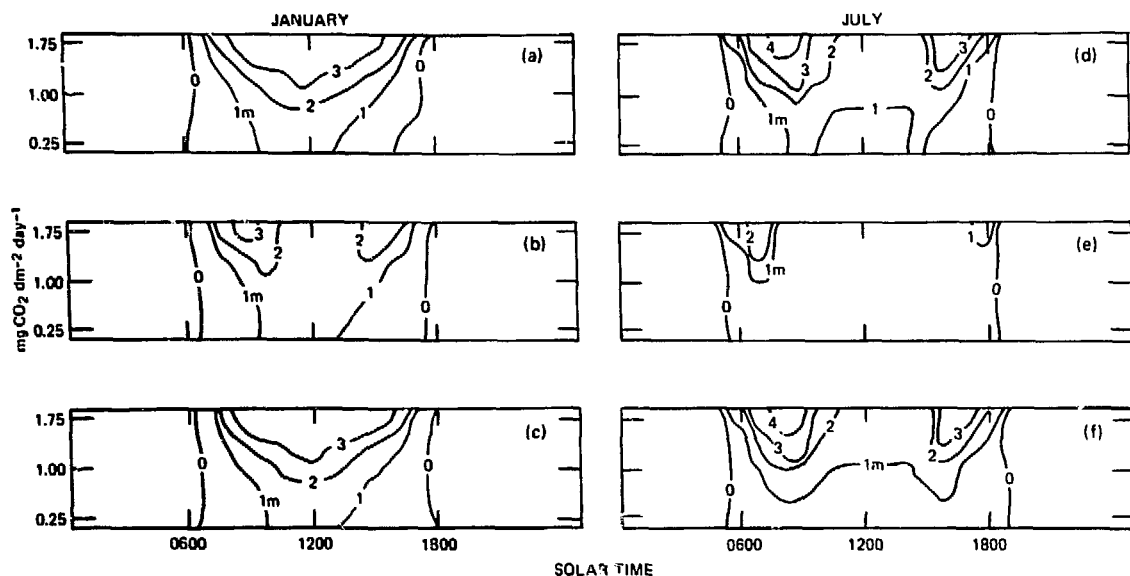


FIG. 4 Progression through the day in January and July of photosynthesis (milligrams  $\text{CO}_2$  per  $\text{dm}^2$  of leaf per hr) at different levels in the canopy at ambient air and soil-surface temperatures (a and d); air temperature ambient +  $5^\circ\text{C}$  (b and e); and soil-surface temperature, ambient +  $5^\circ\text{C}$  (c and f). Vertical axis is height in meters.

restrict positive rates to the early morning and late afternoon. Increasing the soil-surface temperature had little effect on daily photosynthesis patterns, causing slightly decreased photosynthesis rates within the canopy in the summer.

#### Thermal Responses of *Avicennia*, *Laguncularia*, and *Rhizophora*

The simulated responses of *Avicennia*, *Laguncularia*, and *Rhizophora* to thermal changes varied (Fig. 5). Any of the three mangrove species in south Florida may be affected by thermal addition, but *Rhizophora* forms the continuous canopy in most of the mangrove areas. The seasonal and vertical patterns of photosynthesis and water-use efficiency will be qualitatively similar in all species but will differ in quantitative detail.

All species showed decreasing photosynthesis with increasing air temperature and little change in photosynthesis with increasing ground-surface temperature in both January and July. The effect of increased temperature was less in January than in July. Under ambient conditions, *Avicennia* and *Laguncularia* showed higher photosynthesis in July than in January. *Rhizophora*, however, showed higher photosynthesis in January. Photosynthesis rates for *Rhizophora* are lower than those for the other two species and, although its sensitivity to temperature was usually less, photosynthesis in *Rhizophora* was the lowest of the species

at high temperatures. *Laguncularia* was the most temperature sensitive. Water-use efficiency clearly decreased with increased air temperature but only slightly with ground-surface temperature. The water-use efficiency was highest in *Avicennia* and lowest in *Rhizophora*.

#### ACKNOWLEDGMENTS

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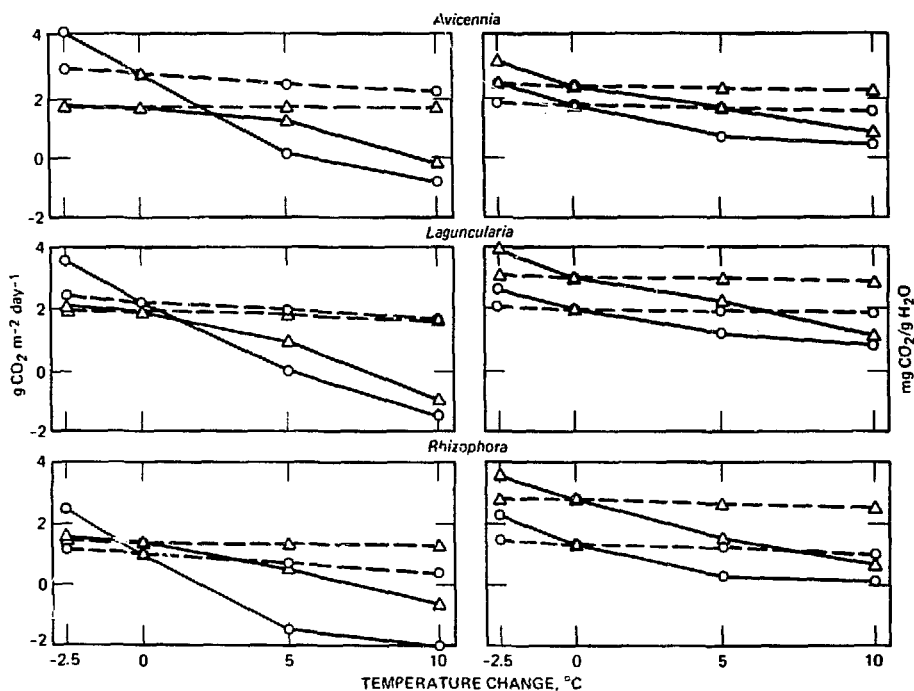


FIG. 5 Effect of changing air (—) and ground-surface (---) temperatures on production (milligrams  $\text{CO}_2$  per square meter of ground per day) and water-use efficiency (milligrams  $\text{CO}_2$  per gram  $\text{H}_2\text{O}$ ) of *Avicennia*, *Laguncularia*, and *Rhizophora* in January ( $\Delta$ ) and July ( $\circ$ ).

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# Rooting and Growth of Red Mangrove Seedlings from Thermally Stressed Trees

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## ABSTRACT

At Guayanilla, on the south coast of Puerto Rico, a fossil-fueled electric generating station of 1100 Mw(e) discharges its cooling water into a nearly enclosed lagoon of about 25-ha area. The plume and lagoon typically have water temperatures 10 and 8°C above ambient; so the winter and summer lagoon temperatures are 34 and 39°C, respectively. The shores of this lagoon have extensive stands of red mangrove trees which are visibly stressed by the elevated temperatures. The critical temperature is between 37 and 38°C. Ripe red mangrove seedlings from the bearing trees are significantly smaller than those from nonthermally stressed trees in Guayanilla Bay and from trees in unpolluted bays from western Puerto Rico. Seedlings from thermally stressed trees developed negative buoyancy and initial roots faster, but the first pair of leaves developed slower than seedlings from control areas. Compared to seedlings from nonstressed areas, these smaller seedlings have a very low probability for survival and growth. No young rooted and growing seedlings have been found in the thermal lagoon.

At Guayanilla, on the south coast of Puerto Rico, a fossil-fuel electric generating station of about 1100 Mw(e) has a once-through cooling system that uses coastal seawater. The discharge canal enters a nearly enclosed lagoon of about 900-m length. The major portion of the shoreline of this lagoon and adjacent shore is a mangrove swamp of about 19 ha dominated by red mangroves (*Rhizophora mangle* L.) with a minor population of black mangroves (*Aveicennia nitida* Jacq.) and white mangroves (*Laguncularia racemosa* Gaertn.). The waters from this lagoon exit via a channel about 40 m wide and 5 m deep (Fig. 1).

The topography of the lagoon and the behavior of the thermal plume in 1971–1972 have been described by Kolehmainen, Morgan, and Castro (1974). At that time the power plant had a capacity of 710 Mw(e). The increased thermal load has been handled by increasing the cooling-water flow; so the temperature at the discharge has not risen above the 40°C observed in 1972. However, the

increased volume of heated water has altered temperature gradients in the lagoon, water currents, and stress on mangroves. In 1972 only mangroves near the discharge canal were subject to water temperatures as high as 38 to 39°C and showed evidence of serious stress (Kolehmainen, Morgan, and Castro, 1974). The major portion of the surface waters in the cove were 4 to 5°C below the discharge temperature (Kolehmainen, Morgan, and Castro, 1974, Fig. 3).

Early in 1974 it was clear that red mangroves along most of the shoreline of the lagoon were under serious stress and many were dying (Fig. 2). Signs of stress included large numbers of aerial roots; loss of leaves so that only the terminal leaves remain; smaller leaves; leaves with greyish-green coloration instead of clear, dark green; and small, scarce seedlings.

Since red mangroves expand and replace themselves by dropping viviparous seedlings into the water, it is important to know if the seedlings from these thermally stressed trees will float, rotate, form roots, and grow in a manner that will permit replanting and regeneration of the mangroves. The floating, rotation, rooting, and growth of red mangrove seedlings from an unstressed area have been described in detail by Banus and Kolehmainen (1975), and this behavior has been related to the propagation of these mangroves in southern and western Puerto Rico.

When mangroves are stressed to the point of dying, there are two possibilities for seedlings to replace the dying trees. Seedlings from stressed trees can remain in the stressed area and grow. Seedlings from nonstressed trees can enter the stressed area and grow. Since the water entering the thermal lagoon comes in either through the power-plant cooling system or as a deep countercurrent through the mouth of the lagoon, it is unlikely that unstressed seedlings enter the lagoon. Furthermore, the prevailing northeast and southeast winds blow seedlings out of the lagoon rather than into it against the current. If the stressed mangrove swamp is to survive, replacement trees must grow from

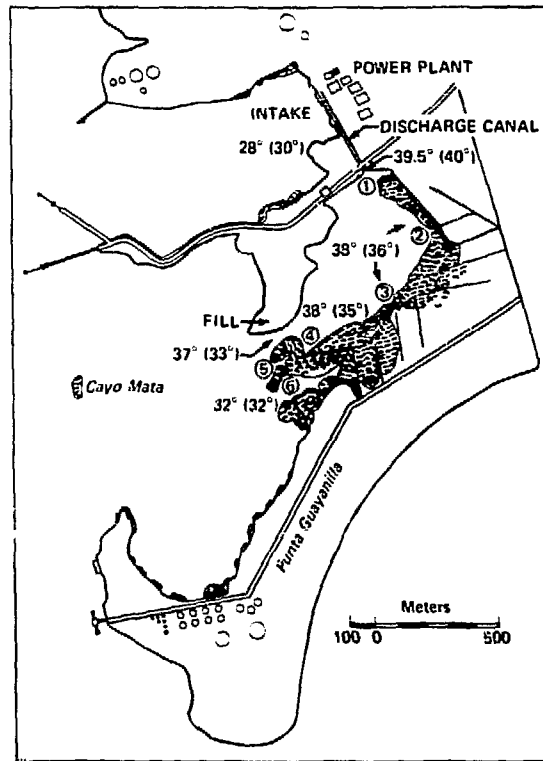


FIG. 1 Plan of Guayanilla Bay thermal cove. Site locations are circled numbers. Water temperatures for April 1974 are compared with temperatures for October 1971 (in parentheses).

seedlings from the stressed trees. Therefore we are studying the behavior and survival of these seedlings.

As part of the study, we placed large batches of seedlings from stressed trees in a nonstressed location to compare their behavior with nonstressed seedlings under the same conditions. This report describes preliminary results and relates the behavior of seedlings to their ability to grow and replace the dying trees in Guayanilla Bay.

The fossil-fuel electric plant is part of a major refinery-petrochemical complex located in this area. These industries are polluting the air and water with hydrocarbons, heavy metals, and organic chemicals. Stresses on the mangroves and seedlings therefore may be chemical as well as thermal. For this reason we are using seedlings from mangroves close to the thermal lagoon for controls and are determining heavy metals and hydrocarbons in seedlings.

## METHODS

The temperature of the top 15 cm of the waters in the locations shown in Fig. 1 were measured several times with a precision thermometer during a year's observation. At the mangrove stations the temperatures were measured close to the roots. Salinities were measured with a portable refractometer (American Optical Corp.).

Ripe red mangrove seedlings were collected twice from trees between sites 2 and 3 (Fig. 1) in the lagoon; at site 5 outside the thermal cove, which has water temperatures above ambient; at the river mouth of Rio Yauco in western Guayanilla Bay; from Boquerón Bay (relatively unpolluted); and from the control embayment south of Punta Ostiones (unpolluted and undeveloped). The ripeness of the seedlings was judged according to criteria described previously (Banus and Kolehmainen, 1975). All seedlings up to about 5 m above the water which met the ripeness criteria were harvested so that no selection bias would enter the size classification. Almost no ripe seedlings were found between sites 1 and 2.

The fruit was removed from each seedling and the seedlings were measured and weighed. Seedlings from the heated lagoon were split into two batches, short and long, to see if their behavior was size dependent. One hundred seedlings from each batch were placed in separate net cages in the ambient control site at Punta Ostiones. The floating angles, buoyancy, and formation of roots and leaves were observed at daily intervals as previously described (Banus and Kolehmainen, 1975). The diurnal temperature changes in the cages were recorded with a thermograph. The temperature and salinity at the time of observation were also recorded.



(a)



(b)

**FIG. 2** Red mangroves in Guayanilla Bay. (a) Stressed trees in thermal lagoon between sites 2 and 3, October 1974, water temperature 38°C. (b) Trees without visible stress at site 5, January 1975, water temperature 29°C.

Thirty seedlings from most batches were promptly frozen at about  $-10^{\circ}\text{C}$  and kept for analysis. Later the top and bottom 4 cm were cut off, dried at  $100^{\circ}\text{C}$ , dissolved in inverse aqua regia, and analyzed for heavy metals by atomic absorption spectrophotometry (Montgomery, Kolehmainen, and Banus, 1975).

## RESULTS

The change in the thermal regime of the lagoon due to the increased flow of hot water between October 1971 and October 1974 is shown in Table 1 and Fig. 1. The entire lagoon is now subjected to temperatures 1 to  $2^{\circ}\text{C}$  below the discharge temperature. Trees along the shoreline are clearly stressed and many are dying. The mortality is probably due to the recent increase in temperature since trees in a small

inner lagoon appear quite healthy. Here the water temperatures are consistently about  $2^{\circ}\text{C}$  below those at site 3. Similarly, trees at site 5 with water temperatures about  $5^{\circ}\text{C}$  below those in the thermal lagoon appear healthy and have abundant seedlings (Fig. 2). The trees at sites 3 and 5 are subjected to the same hydrocarbons and heavy metals as those in the thermal lagoon.

The mean lengths and weights of seedlings collected in the thermal lagoon and from other areas are listed in Table 2. Histograms of the length distributions for these collections are given in Fig. 3. These data show that the seedlings from the heated lagoon were significantly smaller. Collections Guay. H1 and Guay. H2 had many small, ripe seedlings 13 to 19 cm in length, a maximum number between 20 and 25 cm, and few above 30 cm. Seedlings from nonheated areas had few lengths below 20 cm, most lengths between 25 and 32 cm, and some lengths up to 45 cm. Batch Guay. S5 from site 5, with temperatures about  $3^{\circ}\text{C}$  above ambient, showed a more even size distribution with few seedlings below 17 cm or above 30 cm.

The floating, rotating, and rooting behavior of red mangrove seedlings is light dependent (Banus and Kolehmainen, 1975). Since the intensity and duration of sunlight varies seasonally, a control batch was observed simultaneously with batches from thermally stressed trees. The average monthly day and night water temperatures at the experimental site are given in Table 3. The daily maximum temperatures of  $32^{\circ}\text{C}$  in August did not appear to stress the surrounding site.

TABLE 1 Daytime Water Temperatures ( $^{\circ}\text{C}$ ) at Guayanilla Study Areas\*

| Site      | October |      | January |      |
|-----------|---------|------|---------|------|
|           | 1971    | 1974 | 1972    | 1975 |
| Intake    | 30      | 28   | 25.5    | 26   |
| Discharge | 40      | 39.5 | 35      | 36   |
| 1         | 37      | 38.5 | 33      | 36   |
| 2         | 36      | 38   | 32      | 34.5 |
| 3         | 36      | 37.5 | 32      | 34   |
| 4         | 35      | 37.5 | 32      | 34   |
| 5         | 33      |      | 29      | 29   |
| 6         | 32      |      | 27      | 28   |

\*Refer to Fig. 1 for site location.

TABLE 2 Description of Seedling Batches Used in Cage Experiments

| Batch*            | Source                                | Approx. tree height, m | No. | Length, cm (mean $\pm$ SE) | Weight, g (mean $\pm$ SE) | Ratio (length/weight) |
|-------------------|---------------------------------------|------------------------|-----|----------------------------|---------------------------|-----------------------|
| Guay. H1-S        | Heated lagoon, $38^{\circ}\text{C}$   | 5-7                    | 143 | $18.3 \pm 2.5$             | $12.4 \pm 8.5$            | 1.47                  |
| Guay. H1-L        | Heated lagoon, $38^{\circ}\text{C}$   | 5-7                    | 137 | $24.8 \pm 2.4$             | $16.8 \pm 3.2$            | 1.48                  |
| Guay. H2-S        | Heated lagoon, $34.5^{\circ}\text{C}$ | 5-7                    | 103 | $17.0 \pm 2.4$             | $10.3 \pm 2.4$            | 1.65                  |
| Guay. H2-L        | Heated lagoon, $34.5^{\circ}\text{C}$ | 5-7                    | 105 | $23.3 \pm 2.9$             | $15.4 \pm 4.0$            | 1.51                  |
| Guay. S5          | Station No. 5, $29^{\circ}\text{C}$   | 5-6                    | 134 | $23.7 \pm 4.4$             | $15.8 \pm 5.7$            | 1.50                  |
| Guay. Amb.        | Guayanilla-west, $25^{\circ}\text{C}$ | 10-15                  | 137 | $29.4 \pm 5.9$             | $21.4 \pm 7.7$            | 1.37                  |
| Pta. O. control 1 | Punta Ostiones, $28^{\circ}\text{C}$  | 6-8                    | 100 | $25.2 \pm 4.4$             | $16.0 \pm 4.5$            | 1.58                  |
| Pta. O. control 2 | Punta Ostiones, $28^{\circ}\text{C}$  | 6-8                    | 110 | $34.7 \pm 3.8$             | $24.7 \pm 5.0$            | 1.40                  |
| Pta. O. control 3 | Punta Ostiones, $28^{\circ}\text{C}$  | 6-8                    | 120 | $34.8 \pm 3.7$             | $25.9 \pm 4.8$            | 1.34                  |
| Boq. control      | Boquerón Bay, $26^{\circ}\text{C}$    | 6-8                    | 135 | $30.7 \pm 4.8$             | $20.6 \pm 5.0$            | 1.49                  |

\*Collections Guay. H1 and Guay. H2 were split into large (L) and small (S) batches at 21 and 20 cm lengths, respectively. Pta. O. controls 2 and 3 were used in experiments that were not part of this study, but they are included since they were part of a single collection.

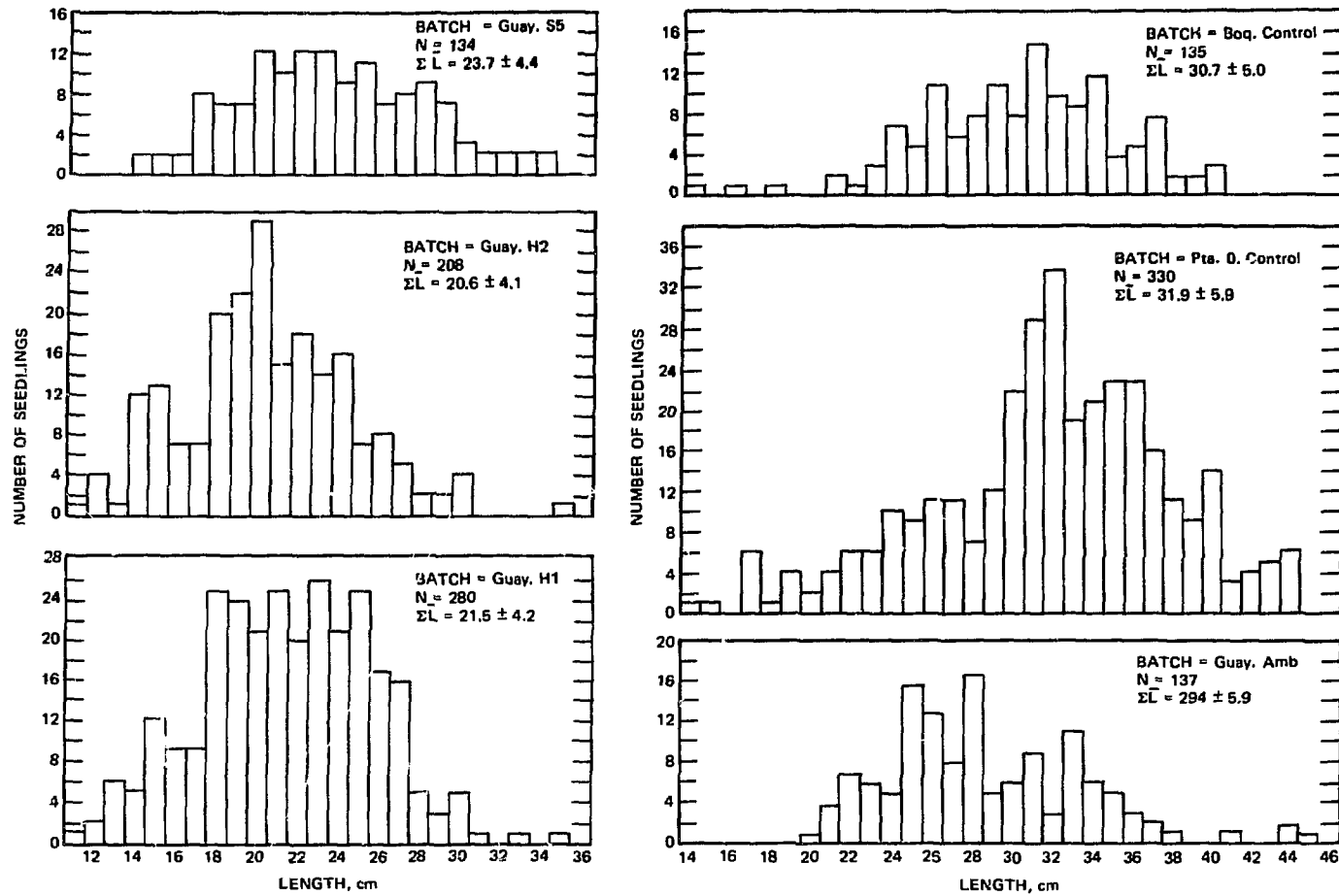


FIG. 3 Numbers of seedlings of each length for the seedling collections. At the 99% confidence level, using Student's t-test,  $\Sigma L$  for seedling collections Pta. O. control, Boq. control, and Guay. Amb. are not different from each other but are larger than Guay. S5. The  $\Sigma L$  of Guay. H1 and Guay. H2 are not significantly different from each other and are smaller than Guay. S5.

TABLE 3 Average Monthly Water Temperatures at Experimental Cages at Punta Ostiones

|                                       | Temperature, °C |      |      |      |      |      |      |
|---------------------------------------|-----------------|------|------|------|------|------|------|
|                                       | Feb.            | June | July | Aug. | Oct. | Nov. | Dec. |
| Daytime*<br>(maximum in afternoon)    | 27              | 29   | 30   | 32   | 30   | 29   | 28   |
| Nighttime*<br>(minimum near midnight) | 24              | 27   | 29   | 30   | 28   | 26   | 25   |

\*Average of daytime high temperatures and nighttime low temperatures that were constant for several hours.

The salinities at the net cages (Punta Ostiones) varied from 26 to 29‰ in the fall (rainy) season to 34 to 36‰ in the late spring (dry) season. The salinities in Guayanilla Bay are 34 to 36‰ all year because the bay is not affected by large freshwater inputs.

Initially, most seedlings from all batches floated horizontally as had been found previously (Banus and Kolehmainen, 1975). They rotated normally, and over 90% were vertical in 10 to 20 days, depending on the month. However, a larger percentage from stressed trees developed negative buoyancy than did so from control trees (Fig. 4). The length of seedlings does not affect this behavior. In one experiment the shorter seedlings of a divided batch had more individuals sinking, whereas, in the second experiment, the longer seedlings had more sinking. Seedlings Guay. S5 showed a floating behavior similar to one control batch. All the data became erratic after about 50 days because of the large quantities of firmly attached epiphytes on the seedlings. Control batch Guay. Amb. had a substantially lower percentage of negative buoyant seedlings than any other batch.

The relationship between root development and time is shown in Fig. 5. Only batches from Guayanilla are shown since they all have seedlings that started roots after 1 to 3 days in seawater. Batches from Punta Ostiones and Boquerón Bay had no seedlings with roots until after 10 days. This is consistent with previous observations (Banus and Kolehmainen, 1975, Fig. 5). The seedlings from the two collections in the heated lagoon initially formed roots more quickly than the Guay. Amb. batch or the batch Guay. S5. However, about 10% of the seedlings from each group did not root until 60 to 75 days. For the second collection, Guay. H2, both large and small seedlings rooted at the same rate. The longer seedlings from the first sampling (Guay. H1L), however, were very erratic and generally slow to root.

The rate of initial leaf formation for the first sampling of stressed seedlings was much slower than was previously observed for unstressed seedlings (Fig. 6). The stressed seedlings are compared with this earlier data because Guay. H1S was in the same experimental cage that held batch 3 and Guay. H1L was in the cage that held batch 7. The

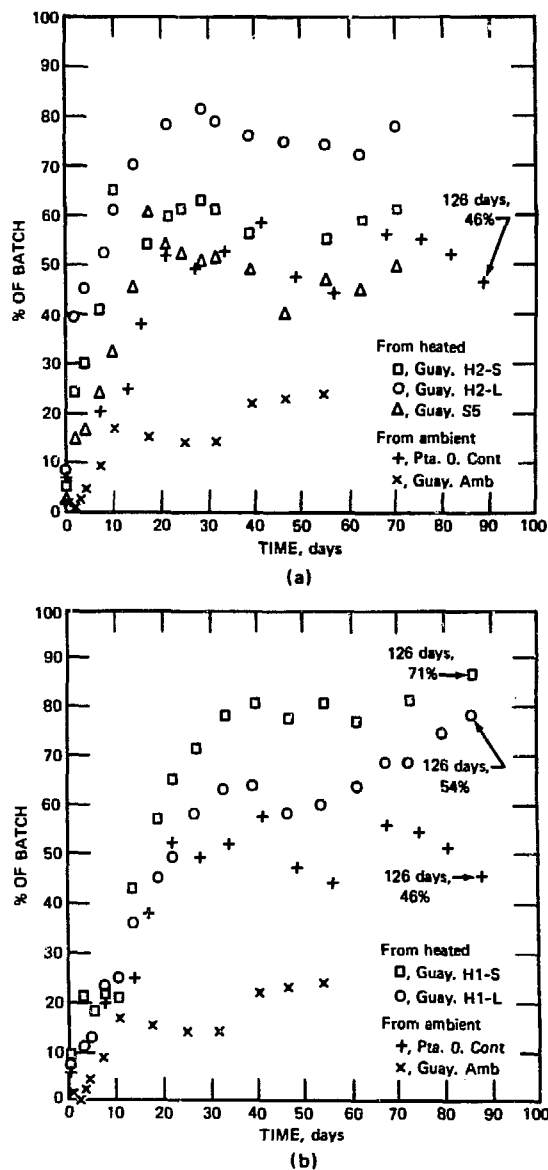


FIG. 4 Percent of seedlings in batch with negative buoyancy. (a) Batches Guay. H2-S, Guay. H2-L, and Guay. S5 compared with batches Pta. O. control and Guay. Amb. (b) Batches Guay. H1-S and Guay. H1-L compared with Pta. O. control and Guay. Amb. Negative buoyancy is reversed by tightly adhering epiphytes. When this occurs, no further buoyancy data is valid.

second sampling of the stressed seedlings (Guay. H2S and Guay. H2L) had leaves at the same rate for the first 120 days.

A number of trees in the heated lagoon produced small seedlings that were bright orange or red instead of green. The seedlings grow from fruit of the same color, have been found in all stages of development, and are from trees that



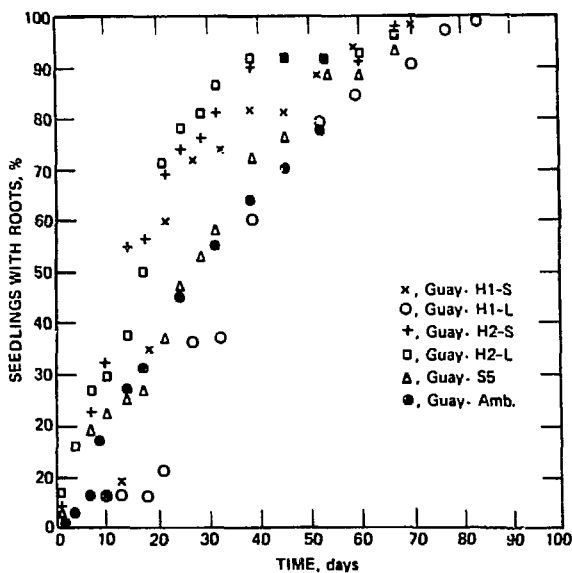


FIG. 5 Percent of seedlings with roots related to time.

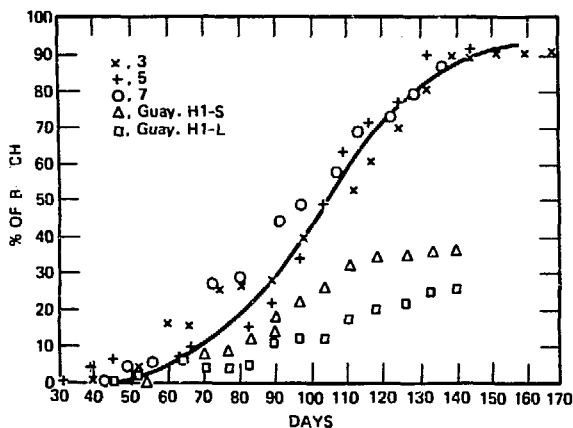


FIG. 6 Percent of seedlings with the first pair of leaves related to time. The line is the best fit for the data from earlier base-line experiments.

also produce green seedlings. Several of these odd mature seedlings have been observed in seawater under the same conditions as the usual batches. The red seedlings have cotyledons of the same color as the seedling, and they have a waxy surface that makes them hydrophobic for several days in seawater. The seedlings we have observed floated horizontally for 2 months before partially rotating to vertical floating. After 80 days none had roots, there was no evidence of growth, and their red coloring had faded to yellow orange. Therefore we have not been able to determine if these unusual seeds are viable or not.

Chemical analysis on the tops and bottoms of 30 seedlings from Guay. H1S, Guay. H1L, and Boq. Amb. showed cadmium, <0.4 ppm (dry weight); copper, 4.5 to 5.2 ppm; and manganese, 27.9 to 32.5 ppm in tops and 7.4 to 8.2 ppm in bottoms. The seedlings from Guayanilla had the higher values, but the differences were not significant within the variation of the batches.

## DISCUSSION

The large increase in heated effluents from the fossil-fuel power plant has raised the temperature of the entire lagoon since 1973 to close to the discharge temperature. This temperature has visibly stressed these mangroves, many are dying and seedling production is low and erratic. Stressed seedlings ripen when they are smaller, and the mean lengths and weights are substantially lower than from control areas.

The size of the seedlings might be related to the size of the trees that produce them, i.e., larger trees may produce larger seedlings. Therefore the height of trees, estimated using a 4-m pole, are included in Table 2. The seedlings of the two large samples from the trees between site 2 and 3 which were 5 to 7 m tall (Fig. 2a) were significantly smaller (mean lengths 21.5 and 20.6 cm) than those from somewhat shorter trees, 5 to 6 m (Fig. 2b), of site 5, (mean length, 23.7 cm) but which did not show stress. Similarly, the seedlings from the mouth of Río Yauco came from trees that are up to 15 m tall (some of the tallest and best appearing mangroves in southern Puerto Rico). They had about the same mean length as seedlings from Boquerón Bay and Punta Ostiones, where the trees are substantially shorter. These tall trees did have a few exceptionally long seedlings but so did the trees at Punta Ostiones. Therefore thermal stress is apparently the principal cause of the smaller size.

The ratio of length to weight of the seedlings has been shown by Banus and Kolehmainen (1975) to be related to their ripeness. At first seedlings rapidly grow lengthwise from the fruit. Then they slowly increase in girth and weight as they ripen and the cotyledonary collar projects. Ratios of 1.40 to 1.55 (cm/g) were found for ripe seedlings, while those whose cotyledonary collars were barely visible, or not visible at all, had ratios from 1.70 to 2.0 (cm/g). The smaller seedlings from the heated lagoon had ratios (Table 2) of the same range as did the control batches. Therefore they were believed to be equally mature.

A possible cause for mature small seedlings other than thermal stress to the trees could be the presence of a chemical in the air or water, which would affect the growth and ripening behavior of the fruit. Ethylene is a chemical that has been implicated in the premature ripening of fruit (Burg and Burg, 1962) and has been shown to be present in the air downwind from this area at levels above those for the nonpolluted countryside (Tsai, 1975). However, if ethylene or other similar chemicals are the cause, they would also affect the trees at sites 5 and 6. Since the seedlings collected from trees at site 5 were larger and

heavier, thermal stress is probably more important than the presence of a chemical as a cause for the seedlings to ripen at a smaller size.

Mangrove seedlings have been shown (Banus and Kolehmainen, 1975) to establish themselves in mud-bottom and turtle-grass areas where the water is the right depth so that vertically floating seedlings can make contact with the bottom and take root. The normal tidal range for the south coast of Puerto Rico is about 20 cm. Therefore seedlings of this length or smaller have little chance to establish themselves, because the seedlings will be completely exposed at low tide if the rooting tips are just in contact with the bottom. In fact, no seedlings of 20-cm length have been found growing naturally in the shallow areas of the thermal lagoon or at other mangrove areas. Thus the smaller seedlings have low probability of survival.

Negative buoyancy, which was more prominent in the seedlings from the thermal lagoon, can aid in establishing new plants if the seedlings have been carried to suitable shallow water before they sink. In the field we have not observed "apple-boring" by seedlings, described by Egler (1948) as a method by which seedlings plant themselves. In net cages with mud bottoms, however, we have observed many negatively buoyant small seedlings that have bored themselves into the mud and have taken root. The water in the cages varied from 20 to 45 cm above mud according to tide and was protected from waves and some wind by the net sides. Three out of four batches of seedlings from the thermal lagoon had most of the seedlings rooting earlier than the controls. These seedlings may have compensated for their smaller size by this negative buoyancy and prompt rooting to increase their chance to survive and grow. However, since there are no suitably shallow areas inside the thermal lagoon for small seedlings to grow, they cannot replace the dying trees.

There is a small area in the thermal lagoon at site 2 where small trees judged to be 3 to 5 years old are living. However, no seedlings in their first year of growth have been found. The small trees started growing before the temperature at this site became 38°C during part of the year. In locations outside the thermal lagoon, where there

are similar young mangrove trees, many seedlings in their first year of development are also present (Banus and Kolehmainen, 1975, Figs. 10, 11, and 13). Therefore seedlings may not be able to survive, root, and grow in water at 38°C. The absence of growing seedlings is additional evidence of the stress to which this mangrove is subjected.

The data and observations so far have related elevated water temperatures to this stress and not chemical pollution. The critical temperature is between 37 and 38°C. Natural water temperatures as high as 32°C are found in flourishing mangroves. At sites 5 and 6, the mangroves show no stress even though a water temperature of 37°C was recorded at site 5. The major deterioration of the trees was between sites 1 and 4, where the summer water temperatures had increased from 35 to 36°C up to 38°C. Additional studies over the next several years will be needed to confirm the lethal temperature and the effects of air pollutants and of hydrocarbons and toxic metals in the water on mangrove seedlings and mature trees.

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# The Effects of Thermal Loading on the Growth and Ecology of a Northern Population of *Spartina alterniflora*

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B. R. Larson

## ABSTRACT

Thermal effluents from the Maine Yankee Atomic Power Co. increased the plant densities of *Spartina alterniflora* during the first year of operation (1973) but reduced the densities substantially the second year (1974). Growth and flowering were initiated earlier near the heated effluent. Subsequent growth was reduced in early summer but was more normal during late summer and fall of the first year. During the second year of operation, growth was greatly reduced, and large plants never developed. Biomass and net-production estimates at the control site compare favorably with other marshes along the east coast of North America. In the marsh affected by the thermal discharge, biomass decreased by 48% in 1973 and 40% in 1974. The rhizome system of affected plants collapsed in 1974–1975. These plants, in contrast to southern forms, appear to have lost their tolerance of high temperatures.

Salt marshes and shallow estuaries of the east coast of North America rank among the most productive areas in the world (Odum, 1971). Numerous studies emphasize the importance of *Spartina alterniflora* Loisel (marsh grass) in these ecosystems (Cooper, 1974; Nixon and Oviatt, 1973b). Not only does *S. alterniflora* supply nutrients to filter feeders and other organisms through bacterial food webs (Teal, 1962), but it also supplies phosphorus directly to estuarine waters (Reimold, 1972).

New England marshes tend to be small and are characterized by peat or semipeat substrates composed of silt and plant remains (Cooper, 1974; Redfield, 1972). Maine marshes are limited by the preeminence of rocky headlands and recent glaciation. Ideal locations for marsh development occur in the glaciated, drowned, and sediment-filled valleys of the Sheepscot River drainage. Numerous coves and backwater areas contain small *Spartina* marshes. Several of these marshes occur in Montsweag Bay, the site of Maine Yankee Atomic Power Co. (MYAPCO).

Thermal effluents from MYAPCO are discharged into Bailey Cove, which contains an extensive population of *Spartina*. Since *S. alterniflora* is highly productive in warm bays in southern marshes (Odum and Fanning, 1973), our objective was to determine the response of a northern population of *S. alterniflora* to thermal stress by measuring density, growth, reproduction, and biomass. Odum (1971) suggests that thermal discharges from atomic power plants may reduce stress in arctic estuaries, which may lead to increased productivity and diversity. Our data provide a test of the productivity portion of this concept in a boreal estuary.

## MATERIALS AND METHODS

We conducted this study from 1972 to 1975. The stressed site (Bailey Marsh) was located 250 m from the discharge, and control site (Causeway Marsh) was located at the northern end of the bay (Fig. 1). In both marshes *S. alterniflora* was the dominant taxon. However, small populations of *Spartina patens* and *Salicornia europaea* occurred at higher elevations. Both marshes are between 1.35 and 1.85 m above mean low tide; thus rhizomes are submerged for 4 to 6 hr per tide cycle. Rhizomes penetrate the sediment to depths of 50 and 70 cm at Causeway and Bailey marshes, respectively. Operation of MYAPCO on a limited basis began in November 1972. Power output was highly variable throughout the study, with several complete shutdowns, especially during summer. The approximate plant operating schedule is shown in Fig. 2 (see differential temperature scale). Since the power plant did not become operational until November of the first year, growth data from Bailey Marsh during 1972 serves as an additional control.

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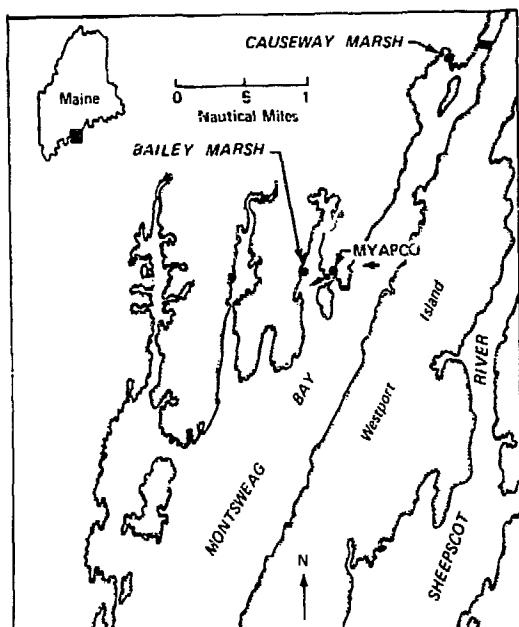


FIG. 1 Location of the Maine Yankee Atomic Power Co. (MYAPCO) at Wiscasset, Maine and stressed (Bailey) and control (Causeway) marshes.

Portions of the two marshes (31 by 19 m in Bailey and 35 by 21 m in Causeway) were selected for quadrat analysis. Areas were divided into square-meter quadrats and assigned numbers randomly. Quadrats ( $\frac{1}{16} \text{ m}^2$ ) were sampled randomly from within the larger quadrat. Quadrats located on small water-filled depressions were excluded from the sampling scheme and replaced randomly. Ten quadrats were sampled monthly, and no unit was sampled more than once during the study. This eliminated the effects of cutting on subsequent growth. Plants in each unit were cut at sediment level with hedge-trimming shears and placed in individual bags. In the laboratory, plants were washed, their lengths were measured, and they were dried at  $70^\circ\text{C}$  to a constant weight and weighed to the nearest 0.1 g. Salinities and temperatures were measured monthly during high tide as 0.15 m and 3.0 m with a salinometer-thermometer. Light penetration was measured with a Secchi disk (Holmes, 1970). Light-energy measurements were made at the MYAPCO site with a pyrheliometer. Sediment temperatures were measured at the surface and 2 cm and 24 cm below the sediment surface by Archambault and Johnson (1972) and by Mazurkiewicz (1973, 1974).

## RESULTS

Salinity patterns at both sites are similar throughout the year, averaging 20 to 25‰ and ranging from 10 to 30‰ (Fig. 2). Monthly and yearly fluctuations are due mainly to precipitation and snow melt. Salinities are usually low in

spring and high in late fall and winter. The increase in salinity from September through November 1974 was due to the removal of a causeway northeast of Causeway Marsh between Westport island and the mainland (Fig. 1), which resulted in better mixing in Montsweag Bay. Water temperatures are more predictable than salinities and range from a winter low of  $-1^\circ\text{C}$  to a summer high of  $22^\circ\text{C}$  in the control marsh and in the preoperation year at Bailey Marsh. Variations in temperature in the control marsh were mainly due to annual variations in insolation. Water temperatures in the stressed marsh were 7 to  $15^\circ\text{C}$  higher during power-plant operation, the exact differences depending on the amount of power production. The differences in temperature between the stressed and control marshes are given in Fig. 2. The maximum differential was less than  $2^\circ\text{C}$  before plant start-up and more than  $10^\circ\text{C}$  after plant start-up.

Sediment temperatures in Bailey Cove during pre-operational phases and during MYAPCO shutdowns show diurnal fluctuations. Similar fluctuations occur at control sites throughout Montsweag Bay (Archambault and Johnson, 1972; Mazurkiewicz, 1973, 1974). Fluctuations are greatest at the surface and decline with depth, with little or no fluctuation occurring at 24 cm. The magnitude of these fluctuations depends on air temperatures and sediment exposure during low tide. The thermal discharge results in abnormally broad temperature fluctuations in the upper 2 cm of the sediment and in elevated temperatures in deeper (24 cm) sediments.

Incoming light energy was similar from year to year. However, extinction coefficients ( $K$ ) were high and indicative of turbid estuarine waters. Irregular increases in turbidity during the study were caused by resuspended sediments resulting from increased effluent volume, construction of a diffuser system, and removal of a causeway. Generally, extinction coefficients were higher in Bailey Marsh. The effects of sediments and turbidity on the growth rates of *S. alterniflora*, however, are negligible. For instance, best growth, observed at Causeway Marsh in 1974, coincided with construction activities and the highest turbidities throughout the growing season. In 1972, 7500  $\text{m}^3$  of sediments were removed from the Bailey Cove channel by pumping water through the plant. These sediments were distributed throughout Montsweag Bay (Takaes and Schnitker, 1973).

## Density

Plant densities in the control and stressed marshes varied seasonally and yearly (Fig. 3). At the control marsh two distinct peaks were apparent each year; these occurred in spring and late summer. Densities during spring increased each year and ranged from 1541 shoots/ $\text{m}^2$  in 1972 to 2035 shoots/ $\text{m}^2$  in 1974. Conversely, densities during the late summer peak declined successively from 1972 to 1974. Plant densities were consistently lower at Bailey Marsh except in August 1972 and September 1973. Two peaks were evident in the preoperational year but thereafter were

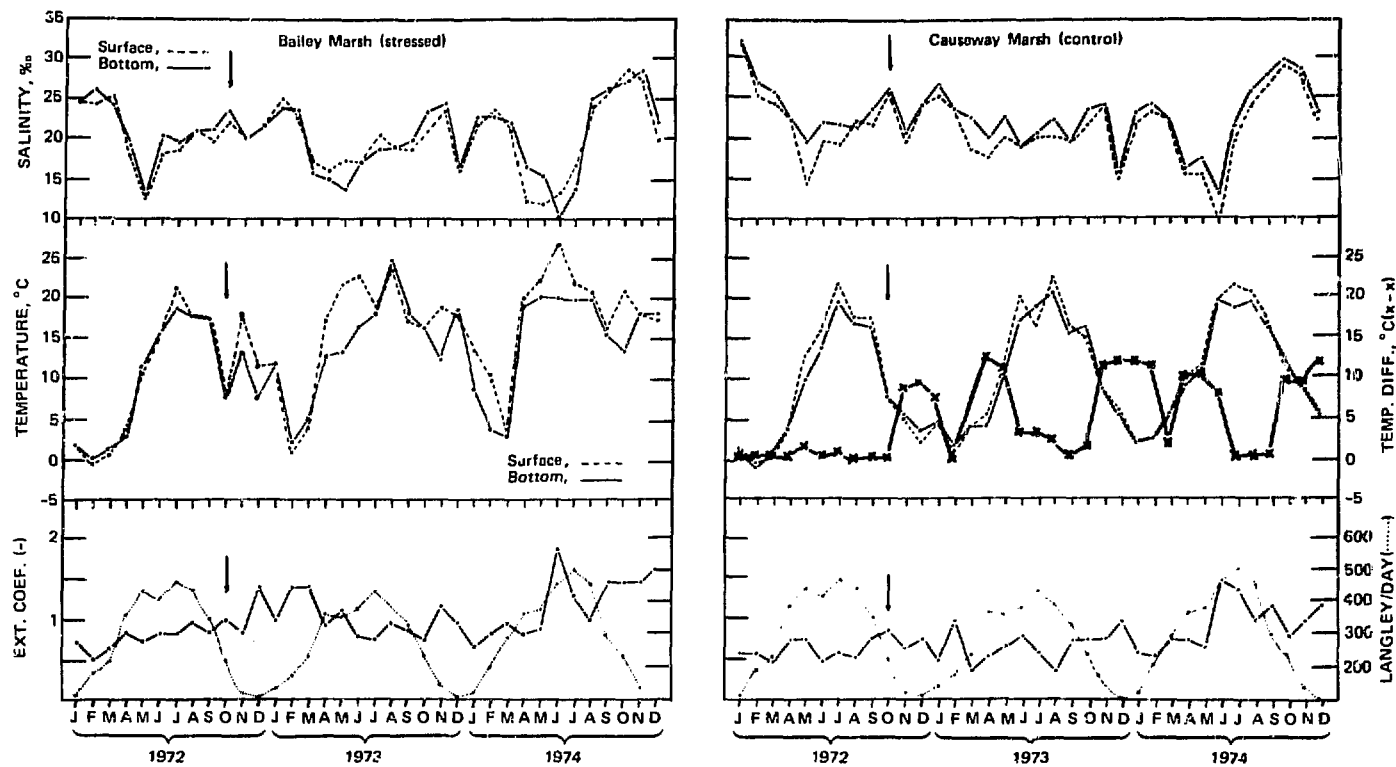


FIG. 2 Hydrographic data for Bailey and Causeway marshes before and during plant operation. Surface (0.15 m) and bottom (3 m) temperature and salinity values are represented by dashed and solid lines, respectively. "Temp. diff." represents the temperature differential between stressed and control sites. Arrows indicate the beginning of power-plant operation.

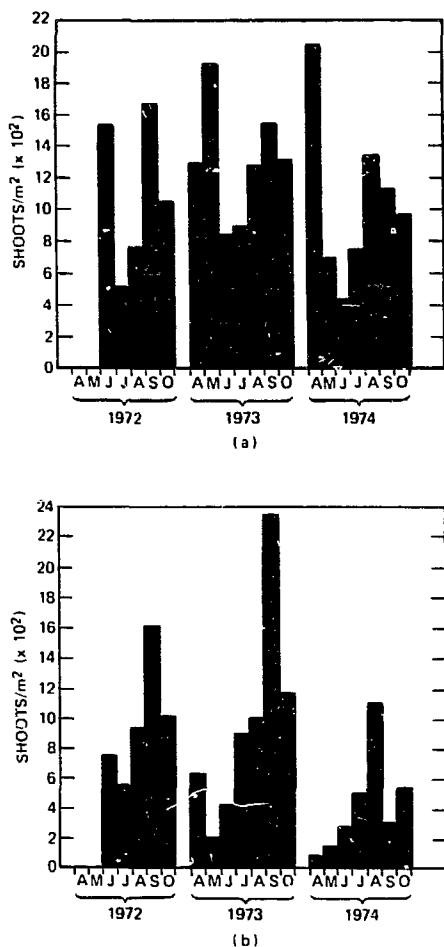


FIG. 3 Density of *Spartina alterniflora* before and during plant operation at (a) Causeway Marsh (control) and (b) Bailey Marsh (stressed).

less pronounced. Usually the highest densities in spring occurred at the time plants first appeared. In 1972, plants appeared in May but were not sampled until June. In the two following years, plants at both sites initiated growth in March and were sampled in April. The variation in the initiation of growth appears to be related to differences in spring temperatures. Water temperatures in 1972 did not exceed 1°C until April, whereas waters in both marshes were warmer in successive springs.

The decline in numbers in late spring and early summer is likely caused by intraspecific competition for nutrients, light, and space. Several studies have shown that light is markedly reduced as the canopy of *S. alterniflora* develops (Blum, 1968; Gallagher and Daiber, 1974). Furthermore, seedlings that germinate in spring may be at a competitive disadvantage with plants that can also draw upon rhizome food reserves.

The second peak in density occurred in August or September. At the Causeway Marsh, values ranged from 1676 shoots/m<sup>2</sup> in 1972 to 1306 shoots/m<sup>2</sup> in 1974. The increase during late summer was due to a new crop of small plants produced from tillers or rhizomes and less commonly from seeds. A similar increase was also observed with *S. townsendii* in England (Taylor and Burrows, 1968) and with *S. alterniflora* in New Jersey (Squiers and Good, 1974). The latter suggested that the new plants increased photosynthetic area before storing nutrients for winter dormancy.

Initially the stressed marsh (Bailey) also showed two distinct peaks. Densities in the spring of 1972 (before discharge) and 1973 (during discharge) were lower than the control marsh and averaged 753 and 620 shoots/m<sup>2</sup>, respectively. The late summer peaks were more pronounced with 1596 and 2361 shoots/m<sup>2</sup>. During the second operational year, density remained low from spring until August when a small peak developed. This was followed by a sharp decline in September. Numerous small plants appeared in October but were present only until a hard frost occurred. Plants are sheared off by ice and storms throughout winter and removed by tides. Little detrital material accumulated in either marsh.

The thermal effluent partially suppressed the initiation of new plants in the spring and early summer of 1973. In July and August MYAPCO was shut down, the numbers of new plants increased rapidly, and by September maximum density was achieved. This rapid increase and high density may have been caused by the growth of previously suppressed plants and the normal appearance of new plants in late summer. A similar explanation is also plausible for the density patterns observed in 1974, except that the early period of suppression was more complete and the subsequent increase during late summer less vigorous.

## Growth

Data on monthly size classes of plants from June to October for the 3 years are given (Figs. 4 and 5). Changes in size classes with time reflect phenological and growth patterns and support the trends observed with density measurements. The changes in size classes at Causeway Marsh (control) are quite similar for the 3 years. However, unlike previous years, plants by June 1974 were taller than in previous years. The flattened histogram for this period is attributed to an earlier onset of growth and higher spring growth rates. Generally, by July most plants were in the largest size classes. Small size classes were evident 1 month later, which supports the trends observed for density. This bimodal curve persisted into fall. The interesting feature here is that most of the second crop does not grow beyond 30 cm, and many rarely grow more than 15 cm.

The distribution of size classes in the stressed marsh in 1972 was similar to the size-class distributions in the control marsh, strengthening the value of both controls. Plants started growth earlier in the stressed marsh during plant operation. Plants were well-developed by mid-April in

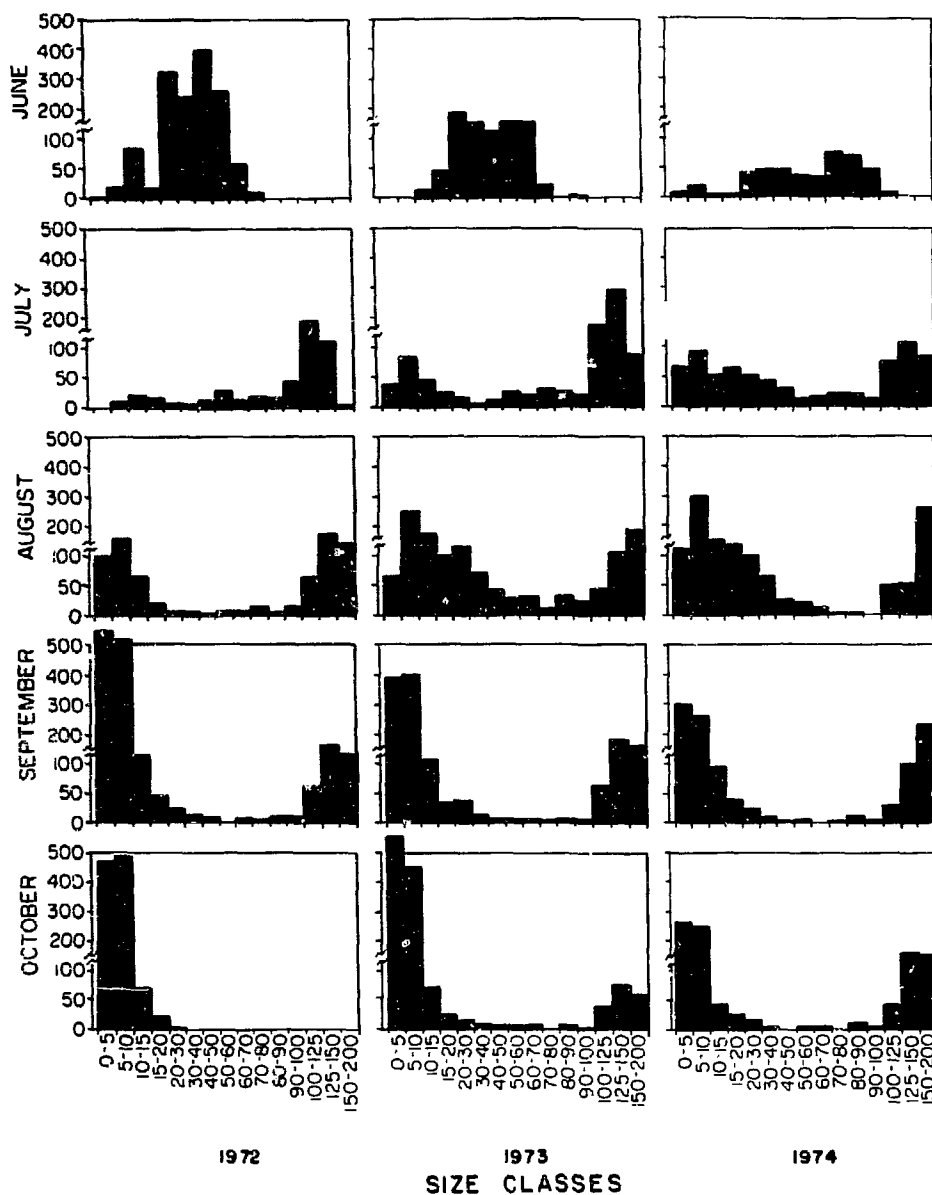


FIG. 4 Monthly size frequency distributions for *Spartina alterniflora* at Causeway Marsh (control) before and during plant operation.

1973 and 1974 but were not initiated until May in 1972. This was evident by the presence of larger size classes in June. Size-frequency distribution remained atypical through July, even though MYAPCO was shut down in late June and remained closed through July of 1973 and September of 1974. Overrepresentation of smaller plants indicates retardation of younger plants and destruction of older plants. In August new plants appeared both before and during discharge. However, a distinct bimodal distribution of size frequencies did not develop in the operational phase until September 1973 and never mate-

rialized in 1974. Plants increased in numbers in September 1972 and 1973. During 1974 this did not occur, except for a small peak in October. In addition, larger size classes (above 80 cm) were absent.

The similarities in the size-class structure of *S. alterniflora* at the control site during 3 years suggest that the control populations are responding to or controlled by the same environmental parameters as the stressed site before power-plant operation. The presence of large size classes in June 1974 seems to be a direct response to the earlier occurrence of warmer waters in spring. This is in agreement

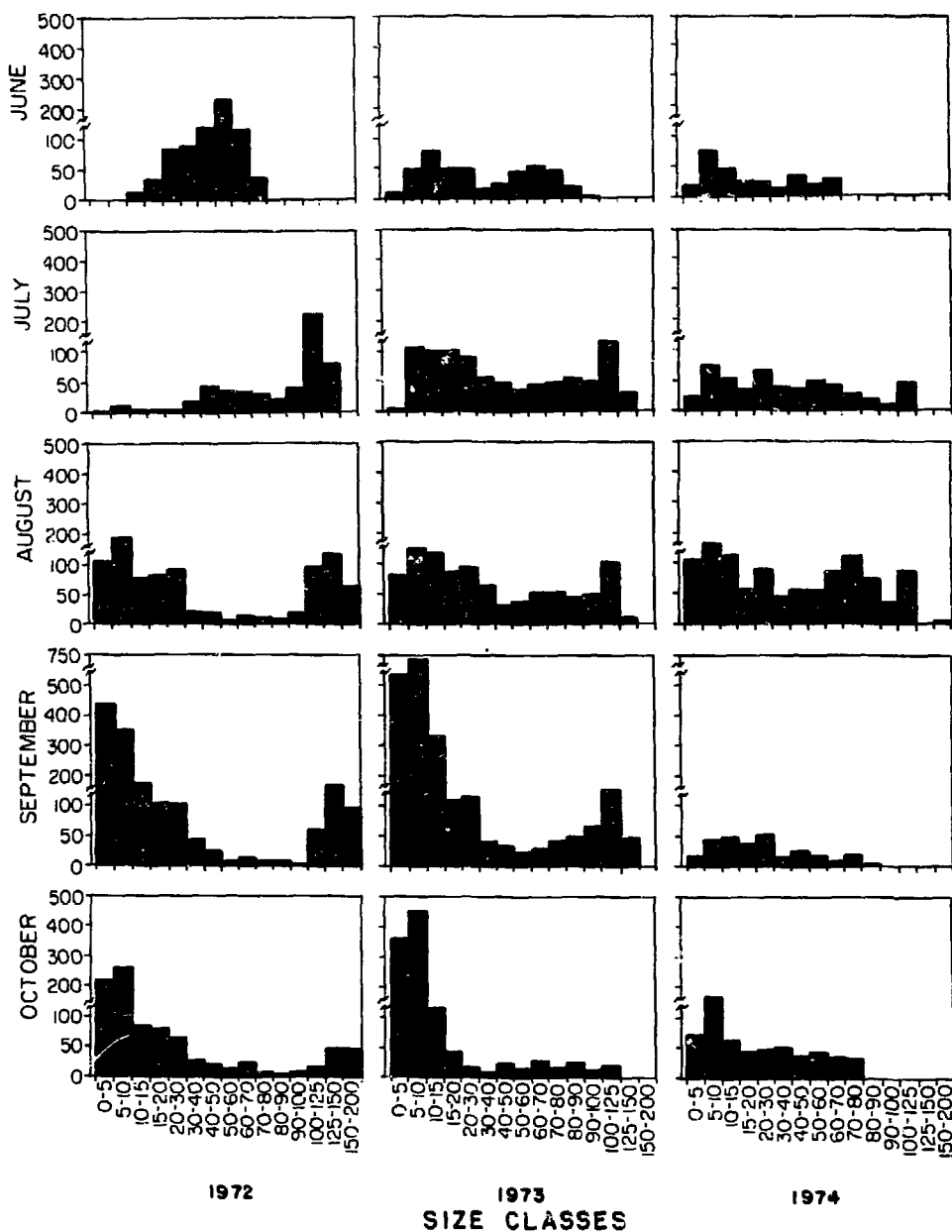


FIG. 5 Monthly size frequency distribution for *Spartina alterniflora* at Bailey Marsh (stressed) before and during plant operation.

with the size-class distribution at Bailey Marsh in June 1973 but not in 1974.

Analysis of age structure thus might provide additional insights into the processes influencing these marshes. Although we are unaware of detailed analysis of this sort, Nixon and Oviatt (1973a) used 1-month (November) size frequencies to help analyse variations in standing crop of *S. alterniflora*.

### Reproduction

The percent of plants flowering in the two marshes in 1973 and 1974 is presented as a function of plant size and the mean height of all flowering plants are given in Table 1. Flowering begins in July at Bailey (stressed) and in August at Causeway (control) and occurs primarily in plants over 100 cm tall. The mean ranges of shoot-height values at the two sites were 154 to 163 cm at Causeway and 78 to



117 cm at Bailey. The smallest plants producing flowers (50 to 75 cm) were observed in the control marsh. At Bailey Marsh, plants in this size range did not flower. By contrast, plants in the 75- to 100-cm category did not flower at Causeway Marsh but did in Bailey, indicating that flowering in plants less than a meter tall is an uncertain event.

size-class data (Fig. 5). An analogous situation occurred in *S. alterniflora* in Florida where numbers increased but biomass remained constant (Young, 1974). The percent decline in biomass during power-plant operation at Bailey Cove, based on the highest values (peaks) obtained in each year, is significant and amounted to 48% in 1973 and 40%

TABLE 1 Size Frequency of Reproductively Mature *Spartina Alterniflora*

| Date                     | Size classes, cm |       |        |       |         |       | Mean height ( $\pm$ SD) of reproductive plants |
|--------------------------|------------------|-------|--------|-------|---------|-------|------------------------------------------------|
|                          | 50-75            |       | 75-100 |       | 100-200 |       |                                                |
|                          | N*               | %FL.† | N*     | %FL.† | N*      | %FL.† |                                                |
| Causeway Marsh (Control) |                  |       |        |       |         |       |                                                |
| Aug. 73                  | 67               | 9.52  | 59     | 0     | 337     | 62.56 | 154.71 $\pm$ 16.76                             |
| Sept. 73                 | 11               | 0     | 16     | 0     | 404     | 23.32 | 155.26 $\pm$ 8.26                              |
| Aug. 74                  | 38               | 8.33  | 3      | 0     | 352     | 63.64 | 163.22 $\pm$ 10.41                             |
| Sept. 74                 | 10               | 0     | 19     | 0     | 361     | 52.21 | 159.90 $\pm$ 5.74                              |
| Oct. 74                  | 6                | 0     | 13     | 0     | 358     | 29.46 | 162.72 $\pm$ 5.94                              |
| Bailey Marsh (Stressed)  |                  |       |        |       |         |       |                                                |
| July 73                  | 99               | 0     | 126    | 2.53  | 155     | 48.45 | 117.33 $\pm$ 7.84                              |
| Aug. 73                  | 113              | 0     | 121    | 0     | 115     | 56.94 | 116.27 $\pm$ 5.79                              |
| Sept. 73                 | 75               | 0     | 141    | 1.14  | 203     | 32.28 | 115.83 $\pm$ 13.87                             |
| July 74                  | 109              | 0     | 35     | 0     | 48      | 60.00 | 117.22 $\pm$ 3.02                              |
| Aug. 74                  | 195              | 0     | 173    | 35.19 | 99      | 74.20 | 101.42 $\pm$ 14.83                             |
| Sept. 74                 | 42               | 0     | 10     | 33.33 | 0       | 0     | 78.00 $\pm$ 0.00                               |

\*N = shoots per square meter.

†%FL. = percent flowering.

The causes for the variability and differences in flowering between the two marshes are unknown. Plants at Causeway appear more vigorous, but the variability in flowering is not entirely related to size. This is emphasized by the increase in flowering of smaller plants at Bailey in 1974 when the marsh was under considerable thermal stress.

### Biomass

Biomass estimates for the two marshes are given in Fig. 6. The increase in weight follows an exponential growth curve. This pattern is evident for both marshes. At Causeway, plants appeared in April, and standing crop values accelerated quickly and became logarithmic between June and July, except in 1972. Although samples were not taken in May 1972, observations in late April indicated an absence of new plants. Using April as zero on the abscissa and approximating the slope, reveals that the maximum period of growth occurred in May. The rate of biomass accumulation in June and July was also high.

The increase in biomass at Bailey in 1972 was almost identical with the pattern at Causeway, intensifying the value of 1972 Bailey data as a control. Total biomass decreased in both operation years at Bailey despite an increase in plant numbers in 1973. This agrees well with the absence of growth in smaller plants inferred from the

in 1974. During these same periods biomass values increased by 20 and 40%, respectively, in the control marsh.

An interesting feature of these data is the unpredictable and precipitous decline in standing crop following the maximum values. Within 1 month maximum biomass often decreased by 40 to 50%. Attempts to measure the productivity of marsh grasses via the harvest method should include adequate controls to ensure that peak values are obtained. Our studies show that maximum values occurred from July to September. Similarly, our data from the Causeway Marsh show considerable variability from year to year. Nixon and Oviatt (1973a) demonstrated that local variability could be a significant factor in the standing crop estimate of *S. alterniflora* in Rhode Island.

It has been suggested that annual net production in marshes decreases with increasing latitude (Cooper, 1974; Keefe and Boynton, 1973). More recent studies in New Jersey (Squiers and Good, 1974), New York (Udell et al., 1969), and Rhode Island (Nixon and Oviatt, 1973b) show a range of 827 to 1592 g dry weight/m<sup>2</sup>. Our highest net estimate from Causeway, without accounting for losses throughout the growing season, is 1419 g dry weight/m<sup>2</sup> in 1974. The mean annual net production for the 3 years at Causeway is 1216 g dry weight/m<sup>2</sup>. Although it is difficult to compare the production values of different marshes because of differences in methodology, generally, most

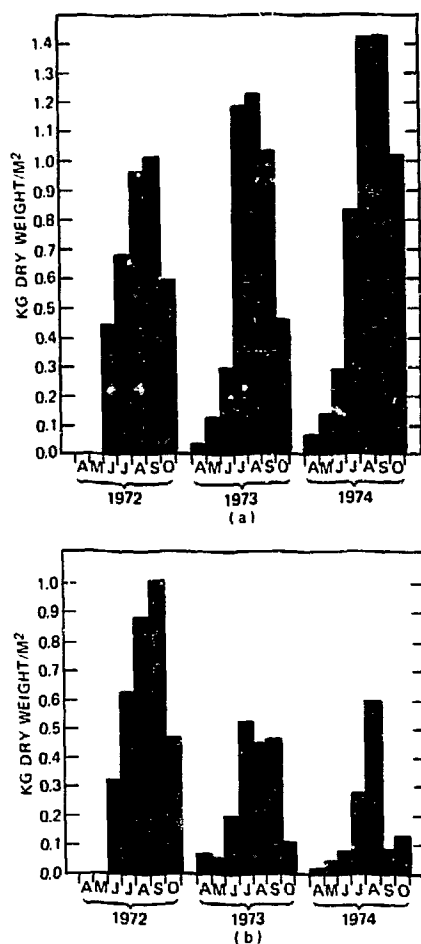


FIG. 6 Biomass of *Spartina alterniflora* before and during plant operation at (a) Causeway Marsh (control) and (b) Bailey Marsh (stressed).

estimates of net production are low (Gosselink, Odum, and Pope, 1973). Our maximum and mean values compare favorably with other northern and most southern populations for the tall form of *S. alterniflora*.

### The Thermal Effect

Our studies show that *S. alterniflora* is adversely affected by the thermal effluent (Fig. 7). Complete collapse of the population in Bailey Cove took 2 years. Probably, without the extensive summer shutdowns by MYAPCO, the plants would have been destroyed sooner. However, in contrast to our findings, Anderson (1969) showed that *S. alterniflora* in Maryland doubled its length in a heated effluent and suggested that it could tolerate temperatures of 35°C. Several studies have shown that the rhizomes and root systems of the sea grass *Thalassia testudinum* were

adversely affected by thermal discharges (Wood and Ziemann, 1969; Bader, 1969).

In view of the latter studies and to determine more closely the impact of the heated effluent on the Bailey Cove population, we collected rhizomes and tillers in March 1975 from both marshes. The rhizomes from Bailey Marsh were gray to cream colored, flaccid, devoid of storage material, and decaying. The outer walls were intact but the rhizome interiors were colorless. In contrast, rhizomes from Causeway Marsh were white, turgid, and filled with stored materials. These observations were quantified by cutting rhizomes into 10-cm sections. Sections were randomized, dried individually to a constant weight at 70°C, and weighed. The difference between the two marshes is significant ( $P \leq 0.05$ ) (Table 2). In effect, the collapse of

TABLE 2 Comparison of Rhizome Weights in Stressed and Control Marshes

| Marsh              | N  | Mean dry weight, mg/10 cm | Significance at 0.05† |
|--------------------|----|---------------------------|-----------------------|
| Bailey* (stressed) | 20 | 43.5 ± 9.3                | a                     |
| Bailey‡ (stressed) | 20 | 71.0 ± 26.3               | a                     |
| Causeway (control) | 20 | 224.5 ± 77.2              | b                     |

\*Rhizomes collected from an area of the stressed marsh devoid of live plants.

†Means sharing the same letter are not significantly different (Student's t-test).

‡Rhizomes collected from an area of the stressed marsh with viable plants.

the upright portion had occurred in the late summer of 1974, and the rhizome system was very near that point in the spring of 1975. Some viable tillers were present, however, and it is likely that a small sparse population could develop in 1975.

### SUMMARY

The introduction of thermal effluents into a boreal salt marsh altered the phenology, abundance, growth, and production of the dominant grass *Spartina alterniflora*. Plant densities increased slightly the first year, but all other measures of vigor were reduced. With the deterioration of rhizomes, collapse of the population is imminent. This train of events is curious and was not predictable at the outset. The marked southern distribution and abundance of *S. alterniflora* and the anticipated amelioration of cold stress suggests that this northern population may have lost the ability to tolerate high temperatures. McNaughton (1966) suggested that the range or adaptability in physiological systems of northern communities of *Typha* was smaller than for southern forms. Perhaps a similar type of ecotypic differentiation occurs in *S. alterniflora*.



(a)



(b)

FIG. 7 Photographs taken in September 1974 of (a) Causeway Marsh (control) and (b) Bailey Marsh (stressed), illustrating the effects of thermal discharge on *Spartina alterniflora*. (A meter stick is held by one of the authors to indicate size relationships in the lower photograph. The arrow shows the location of the heated effluent.)

## ACKNOWLEDGMENTS

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Contribution 86, Ira C. Darling Center, Department of Oceanography, University of Maine, Walpole.

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# Effects of a Thermal Effluent on the Reproduction of the American Oyster

Jeff C. Tinsman  
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Don Maurer

## ABSTRACT

Oysters were sampled from September 1970 to September 1972 from two stations in Indian River Bay, Delaware, for histological examination. Station 1 (control) was located near the intake of the Delmarva Power and Light Company's fossil-fuel plant in Millsboro, Del. Station 2 (effluent) was 2.5 km east of the plant at the mouth of Island Creek. A combination of classical and quantitative methods was used to document changes in the developmental cycle of oysters resulting from the influence of a thermal effluent. Oysters in the effluent showed precocious gonadal development during the spring, but the time of maximum ripeness and spawning activity was the same at both stations. An extension of the spawning season for several weeks into September was noted for oysters in the effluent. During 1972, oysters in the thermal effluent showed a significantly lower level of gonadal development than did controls. Since this season of decreased reproductive amplitude corresponds with a period of high mortalities among effluent oysters, it seems likely that these oysters were thermally stressed, requiring more energy for maintenance and leaving less available for reproduction.

As the volume of the nation's thermal effluents approaches the annual freshwater runoff, there has been a trend toward locating power plants on bays where flushing with cold ocean water aids in removing waste heat. By 1980 one-third of all power plants in this country will be located adjacent to estuaries (Picton, 1960). Trends toward construction of larger plants with larger cooling-water needs make long-term environmental studies necessary to safeguard these highly productive areas.

By affecting the timing and level of spawning activity or the success of the larvae produced, a thermal effluent may affect the ultimate success or failure of a population. The present study was designed to determine the effects of a thermal discharge on several aspects of reproduction in the American oyster, *Crassostrea virginica* (Gmelin), in Indian River Bay, Delaware (Fig. 1). It was part of a larger study which included an analysis of the immediate effects of the effluent on survival, growth, and condition of the oyster (Tinsman and Maurer, 1974).

A dominant, sessile, benthic invertebrate with a wide range throughout the estuaries of the Atlantic coast, the oyster is an important assay organism in thermal-addition studies. The oyster is dioecious, and its reproductive cycle is temperature related. Increase in volume and density of the gonad occurs as temperatures rise in the spring. The gonad, consisting of follicles that swell with gametes, is unencapsulated with indistinct borders. Spawning, often triggered by a sharp temperature rise, occurs during midsummer, and an individual may spawn several times over a 4- to 6-week period. Resorption of the gonad occurs in late summer, and sexual differences may disappear. Late fall and winter are periods of sexual inactivity.

## MATERIALS AND METHODS

The gonadal development of oysters was monitored at two stations in Indian River Bay, Delaware (Fig. 1). The control station (Station 1), 100 m upstream from the cooling-water intake of the Delmarva Power and Light Company's plant, received ambient water. The test site was Station 2 at the mouth of Island Creek, which serves as the effluent canal. The fossil-fuel facility has a generating capacity of 350 Mw and discharges up to 265,000 gal (900 m<sup>3</sup>) of effluent per minute into Island Creek.

Adult oysters, 4 to 9 cm in shell height, were collected from intertidal oyster bars near the mouth of the Murderkill River, Delaware, in June 1970. Fouling organisms and oyster spat were removed from the shells, and the oysters were placed in off-bottom trays at both stations in approximately 1 m of water. Silt and fouling organisms were removed periodically throughout the study. These oysters were sampled monthly from September 1970 to September 1972 when the supply was exhausted. Temperature was monitored continuously at both stations with Foxboro temperature recorders.

Ten oysters per station were sampled monthly for histological examination. These were returned to the laboratory, placed in cold-air storage, and sacrificed within 24 hr. Shucked oysters were preserved whole in Bouin's

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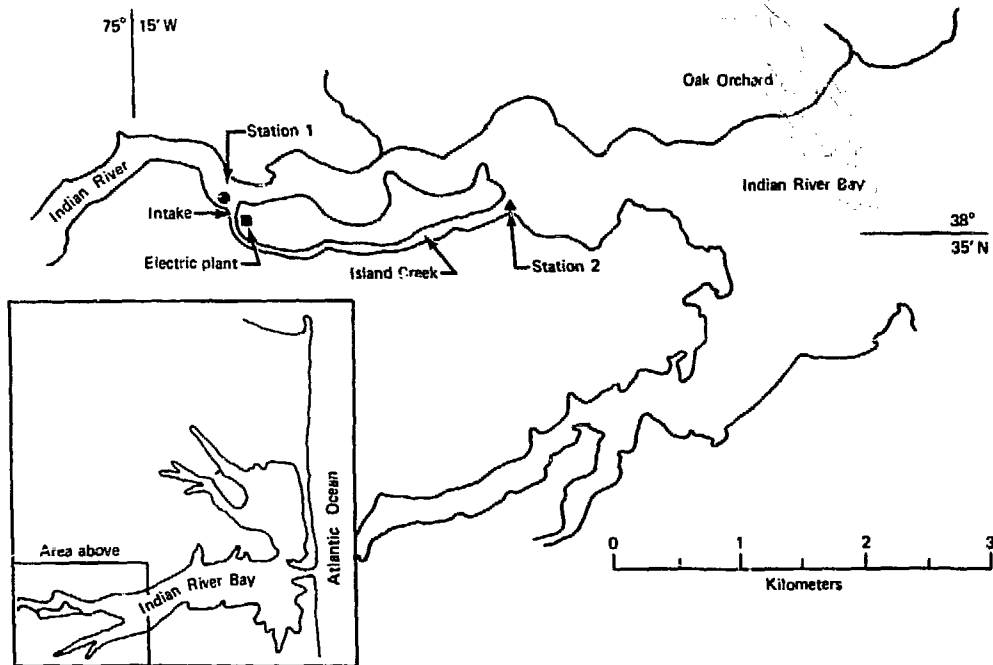


FIG. 1 Map of Indian River Bay, Delaware, showing the location of the Delmarva Power and Light Company's electric generating plant and the sampling stations.

fixative for at least 72 hr. A 5-mm cross section was cut perpendicular to the anterior-posterior axis, midway between the anterior edge of the adductor muscle and the anterior end of the oyster, passing through the midbody region. Prepared cross sections were dehydrated in methanol, cleared in xylene, and embedded in Tissue Prep (melting point, 61°C). Sections (10  $\mu$  thick) of the embedded tissues were cut with a microtome, affixed to microscope slides with albumin, air dried, stained with haematoxylin and eosin, and mounted under Permount.

With development the gonad of the oyster thickens, increasing in volume. A photographic enlarger was used to quantify this change in volume with time. Using the enlarger, each oyster section was projected (magnification = 10X) onto a grid of points. The point-count method of determining area, described by Gladman and Woodhead (1960), was used since this method is comparable to the polar planimetry method of determining area if 200 or more points are counted (Iwantsch, 1970). The number of points falling within the gonad and the number falling on the entire visceral mass were counted for each oyster. The ratio of these measurements was expressed as percent gonad for each oyster.

$$\text{Percent gonad} = \frac{\text{area of gonad}}{\text{area of visceral mass}} \times 100\%$$

The size of the gonadal region (percent gonad) alone was not an adequate measure of developmental progress, because fullness (density) of follicles within the gonad also increases during development. Therefore the density of each gonad was also determined. We used a microslide projector (magnification = 25X) to examine four standard representative regions within the gonad in each oyster, and the area covered by follicles within each region was determined by the point-count method. The ratio of these measurements is the gonad density.

$$\text{Gonad density} = \frac{\text{area in follicle}}{\text{standard sample area within gonad}}$$

The gonad density is expressed as a decimal between 0.0 (no follicle) and 1.0 (all follicle). The density was observed to change rapidly and predictably with the spawning stage so as to give a quantitative measure of changes in the developmental stage of the oyster. Generally, resorbed (indifferent) oysters showed densities of 0.01 to 0.15; developing oysters, 0.15 to 0.80. Between 0.60 and 0.80 some ripe gametes are present, and attempts to induce spawning suggest that low-level spawning can occur at a density of 0.60. Natural spawning probably occurs most commonly at a density above 0.80.

A follicle index (percent gonad  $\times$  gonad density) was calculated for each oyster. This index is a measure of the

area devoted to germinal tissue in each oyster cross section, which is in turn a measure of the degree of gonadal development of the oysters.

In addition, each slide was examined under an AO compound microscope at 100X and 430X to determine sex and probable spawning stage. The following stage classification was used:

1. Indifferent. Sexually indistinct with no recognizable gametes in the follicle. Follicles are generally small and gonad density low. Follicle lumina if present are filled with amoebocytes.

2. Developing. Some gametes and intermediate germinal cells are present in low numbers. Eggs may be small, and mature sperm may not yet be present. Gonad density and gonad thickness are increasing. Amoebocytes are not present within follicles.

3. Ripe—spawning. Eggs are large and fairly uniform in size; mature sperm fill the lumen of the follicle. Germinal cells in the follicle wall are few, although some development continues. Gonad density is very high. Some amoebocyte activity may occur in follicles after spawning begins.

4. Spawning out—resorbing. Few gametes remain in each follicle; lumina are invaded by large numbers of amoebocytes; there is a decline in gonad density, approaching the indifferent stage.

During the spring of 1971 and 1972, periodic attempts were made to artificially induce spawning. Fifteen oysters per station were returned to the laboratory, and the

thermal and chemical stimuli described by Loosanoff and Davis (1963) were applied. Oysters were tested on two successive days, receiving thermal stimulation for 8 hr on the first day and 4 hr on the second day. If spawning was not induced in this period, an oyster sperm suspension was introduced periodically over an additional 4-hr period.

The *t*-test (0.05), with modified critical *t* values where appropriate (in cases where an *f*-test showed sample variances to be significantly different at the 0.05 level) was used to test the significance of differences in means of the follicle index and the gonad density on a monthly basis (Sokal and Rohlf, 1969).

With this combination of classical methods of studying gonadal development (such as spawning-stage determinations), and more quantitative and objective methods (such as follicle index and gonad density), an attempt was made to document changes in the developmental cycle of oysters resulting from the influence of a thermal effluent.

## RESULTS

### Temperature

Mean weekly water temperatures for both stations for the period Sept. 1, 1970, through Sept. 30, 1972, are shown in Fig. 2. Temperature maxima occurred at the control station in September 1970 (27°C), June 1971 (29°C), and June 1972 (32°C); and minima occurred in December 1970 (−1.5°C), December 1971 (3.0°C), and February 1972 (3.0°C). Temperatures at the effluent

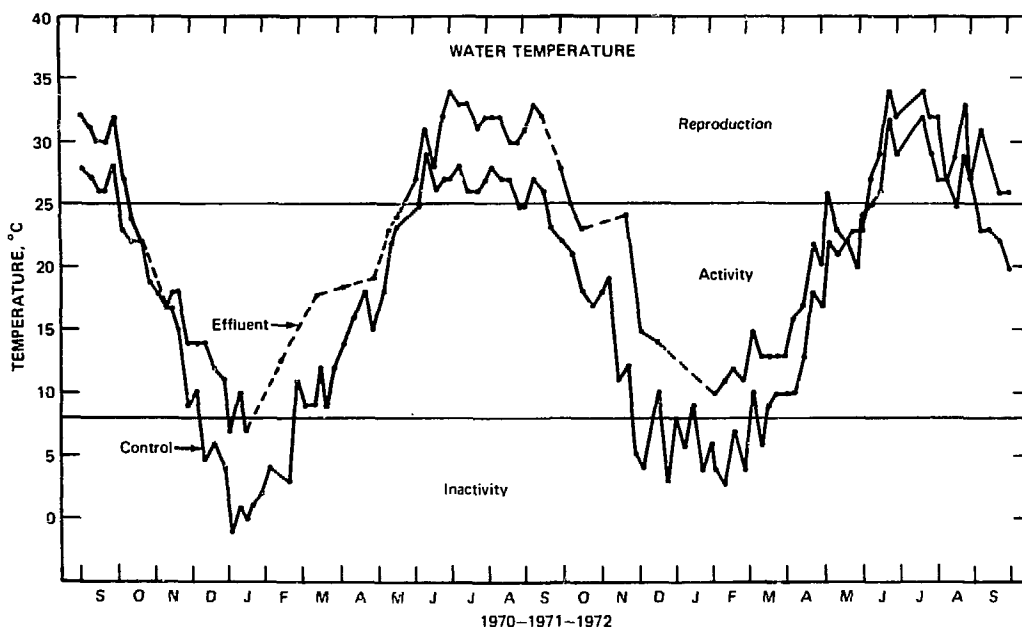


FIG. 2. Annual water temperatures at Station 1 (intake canal) and Station 2 (effluent canal). Each point is the weekly mean of 42 temperature readings from a continuous temperature recorder. Approximate temperature ranges for oyster inactivity, activity, and reproduction are indicated. Dotted lines represent areas where one or more weekly means are missing.

station paralleled those at the control station but were higher. Maxima were recorded in September 1970 (34°C), June 1971 (34°C), and June and July 1972 (34°C); and minima occurred in January 1971 (7.0°C) and February 1972 (10.0°C). The greatest mean difference between stations (12.0°C) occurred in November 1971.

### Mortality

Monthly mortalities of oysters at both stations are presented in Fig. 3. Mortalities were seasonal in pattern, the highest occurring at the control station in September 1971 (20.4%) and July 1972 (3.0%) and the lowest occurring in January 1971 (0.7%), in December 1971, and in January, February, April, and May 1972 (0.0%). Mortality maxima at the effluent station occurred in September 1971 (13.6%) and September 1972 (29.3%), and minima occurred in January 1971 (0.2%) and February through April 1972 (0.0%). The greatest differences between stations occurred in September 1971 and July through September 1972. A difference in mortality pattern was noted between the 2 years of the study. During the summer of 1971, mortalities were higher at the control station, whereas during 1972 summer mortalities were higher in the effluent.

### Follicle Index

The follicle index, a measure of the level of gonadal development, is shown in Fig. 4. The follicle index of oysters from the control station declined from September

through December 1970, remaining low during the winter before increasing in April 1971. A peak was reached in July and August and was followed by a dramatic decline in September 1971. The index remained low through April 1972, rising dramatically in May and reaching the highest level measured at this station in June 1972. Small declines in July and August were followed by a large decline in September 1972. During 1971, effluent oysters were higher in follicle index, although this trend reversed itself during 1972. Significant differences were noted in March and May 1971 and in April and June through September 1972.

### Gonad Density

Gonad-density data are shown in Fig. 5 as a quantitative measure of changes in the spawning stage. Seasonal variations in gonad density were similar in general pattern to fluctuations in the follicle index. For Station 1, gonad density showed minima in March and November 1971 and maxima in July 1971 and June 1972. At Station 2, minima were recorded in January 1971 and January 1972 with maxima in July 1971 and June 1972. Significant differences between the two stations occurred in September 1970, in March, May, and July through September 1971, and in April, June, July, and September 1972.

### Spawning Stage

A summary of spawning stages is shown in Fig. 6. Oysters from Station 1 were predominantly indifferent

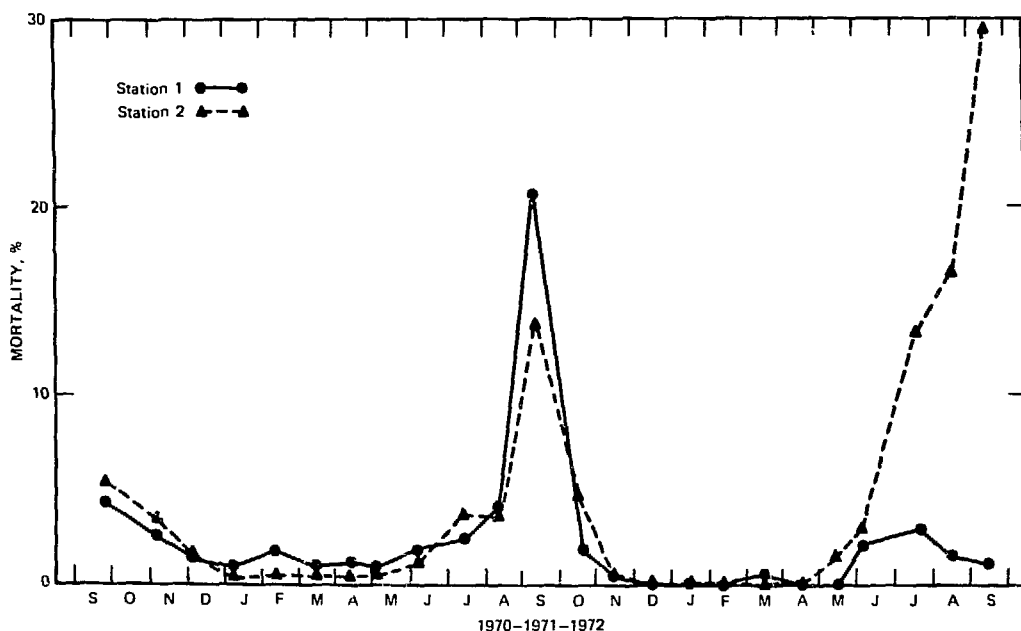


FIG. 3 Mortality expressed as the percentage of oysters that died at each station during the month preceding the sampling date.



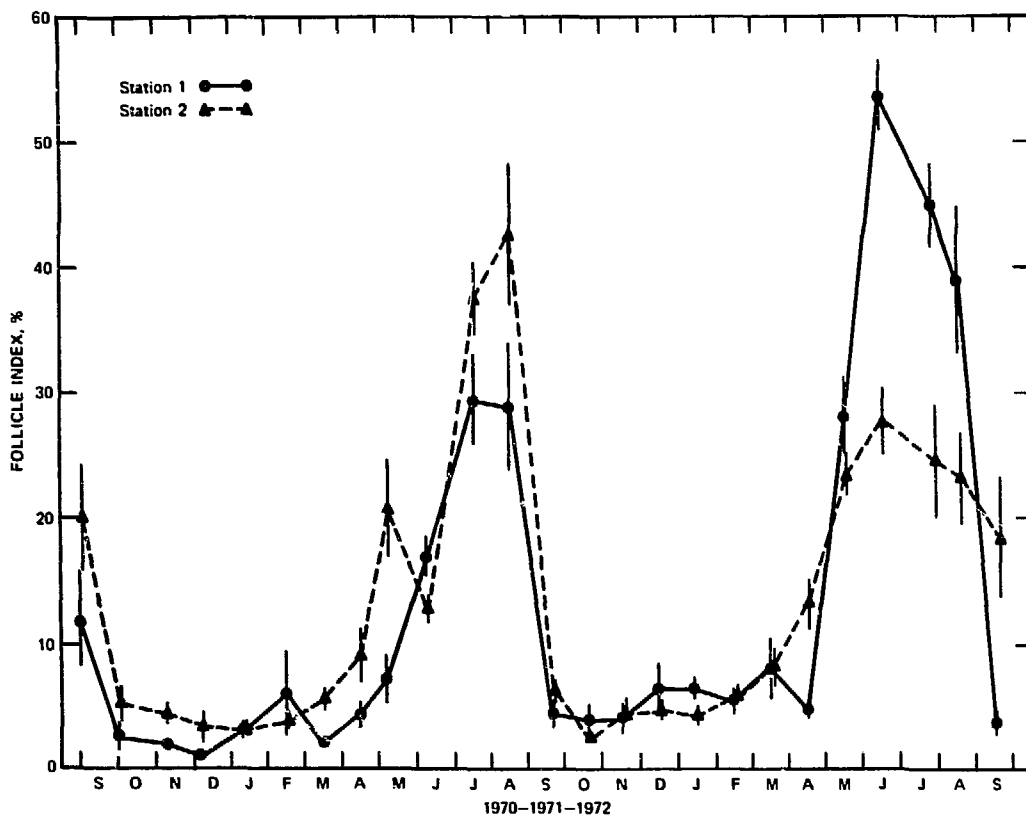


FIG. 4 Follicle index (percent gonad  $\times$  gonad density). Each point is the mean of 10 oysters. Vertical bars represent one standard error of the mean on each side of the mean.

from September 1970 through March 1971. In April through June most oysters were developing. Ripe-spawning individuals predominated in July and August 1971. In September most oysters were resorbing, and from October through January 1972 most were again indifferent. In February through May 1972, developing oysters predominated, and ripe individuals were present from June through August. By September 1972 most control oysters were indifferent.

Oysters from Station 2 were largely indifferent from October 1970 through February 1971. In March through June most oysters were developing, whereas ripe individuals predominated in the July and August samples. In September 1971 most oysters were resorbing, and the indifferent stage predominated in October 1971 through February 1972. Development occurred during March through May, and ripe individuals were most abundant in June through September 1972.

A difference in the developmental pattern between the 2 years of the study was noted at both stations. During the second year, low-level development occurred in October and November and progressed very slowly throughout the winter. This pattern was not obvious in the first year, but a

similar maturation pattern was mentioned by Loosanoff (1942). Differences in the pattern of gonad maturation between stations also occurred. Earlier development of the gonad was evident during the spring of 1971 at the effluent station, and ripe individuals appeared first at this station. During 1972, differences in early development between stations were more difficult to assess because spring gonadal development was superimposed on the low-level development that had been initiated in October 1971. Oysters held in the effluent were generally indifferent for a shorter period due to earlier spring development and an extension of the spawning season into September 1970 and September 1972.

### Induced Spawning

Attempts to induce spawning during the spring months in 1971 and 1972 provided additional evidence of spawning condition. In 1971, only one attempt was successful. Two oysters collected from Station 2 were successfully induced to spawn on May 5. Water temperature at the time of collection was 23°C, and the computed mean gonad density of this group of oysters was 0.62. Oysters sampled from Station 1, where the ambient temperature was 17°C, failed

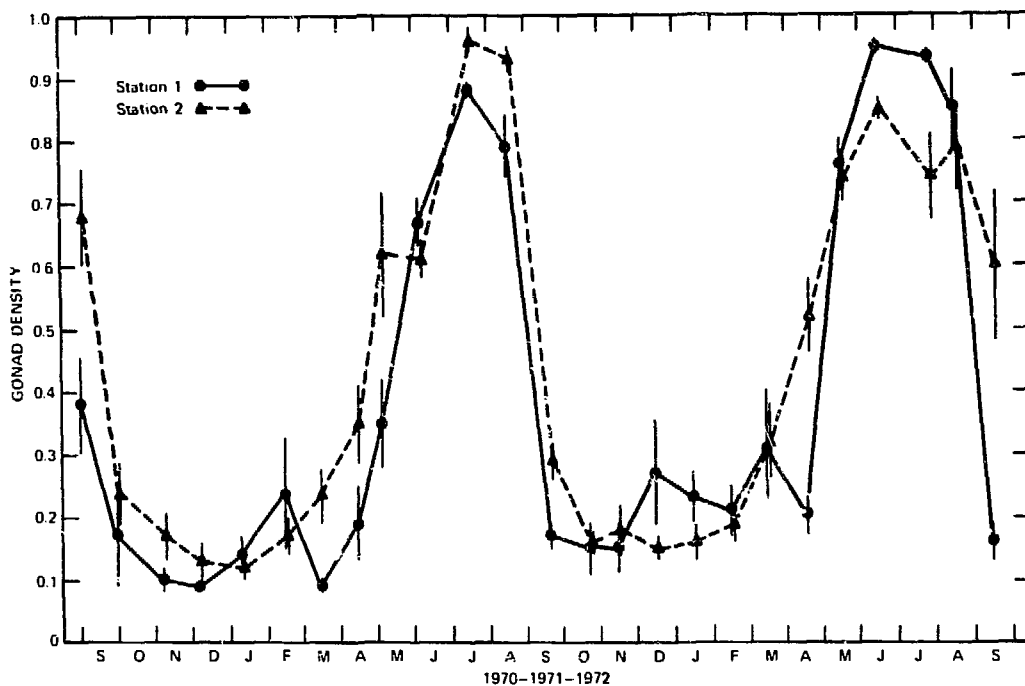


FIG. 5 Gonad density (area in follicle/area of gonad) expressed as a decimal. Each point is the mean of 10 oysters, and vertical bars represent one standard error of the mean on each side of the mean.

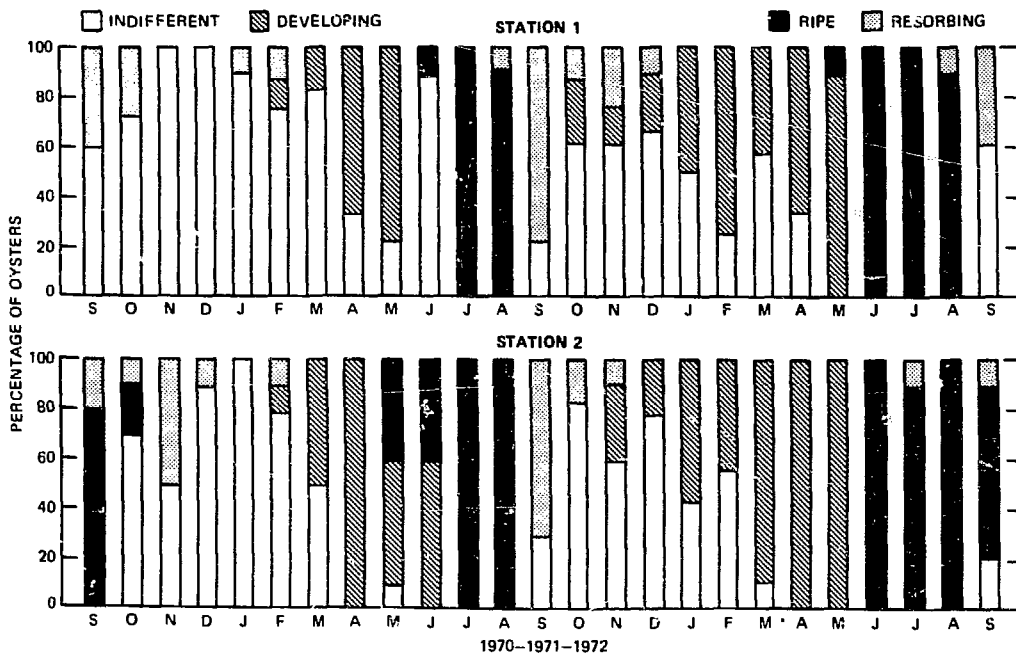


FIG. 6 Spawning stage expressed as the percentage of oysters at each station in each of the four spawning stages. Determinations were generally based on 10 oysters per month.

to spawn. The gonad density of this group proved to be 0.35. During 1972, two successful spawning attempts were made. On May 2 one oyster from the effluent station was induced to spawn, whereas control oysters failed to spawn. At the time of collection, the ambient water temperature was 20°C and the temperature in the effluent canal was 26°C. Computed gonad densities for these groups of oysters were 0.50 for Station 1 and 0.62 for Station 2. On May 30, 1972, 9 of 15 control oysters and 6 of 15 effluent oysters were successfully spawned. In both cases, temperatures exceeded 25°C and gonad densities were above 0.80.

## DISCUSSION

Temperature is the major environmental factor influencing poikilotherms (Kinne, 1963). Oysters experience ambient water temperatures of 0 to 32°C over their geographic range, but little is known about the prolonged effects of higher temperatures (Galtsoff, 1964). Stauber (1950) proposed the existence of physiological races of oysters on the basis of differences in spawning temperatures of the populations along the east coast. Loosanoff and Nomejko (1951) concurred, noting that southern oysters failed to develop gonad and spawn at ambient summer temperatures in Connecticut. Galtsoff (1964) summarized the abundant literature on the normal spawning patterns of northern (Kennedy and Battle, 1964; Loosanoff, 1942; and Loosanoff and Davis, 1952) and southern (Ingle, 1951; Hopkins, Makin, and Menzel, 1954; and Butler, 1955) races of oysters. Generally, southern oysters initiate development and are triggered to spawn at higher temperatures than northern races. The dramatic thickening of the gonad during development of northern oysters occurs to a lesser degree among Gulf Coast oysters. The spawning season which may cover only 1 or 2 months in the north may go on at a lower level for 4 to 5 months in the south.

During the summer months, when reproductive activity occurred, oysters held at Station 2 were exposed to temperatures in excess of 30°C for weeks at a time. On some occasions temperatures exceeded 35°C for several days at a time, and temperatures as high as 41°C were measured. There is abundant literature to show that the optimum temperature range of the oyster is below this level. Collier (1951) and Loosanoff (1958) concurred that the optimum temperature for pumping is near 25°C. Federighi (1929) found that the heartbeat rate of the oyster declined beyond a maximum at 30°C. Similarly, Galtsoff (1927) found that the activity of oyster gill cilia declined at temperatures in excess of 31°C. This was interpreted as a response to thermal stress.

Brooks, Smith, and Jensen (1974) reported the results of studies on primary production by unicellular algae, the primary food of the oyster at the Indian River site. Production was enhanced in the heated effluent when ambient temperatures were below 22°C. Heating above 27°C reduced the primary production rate. These data are in support of those of Warriner and Brehmer (1966). In midsummer, when production was reduced at the outfall, it

slowly returned to levels similar to the intake as the water mass moved down Island Creek (Brooks, Smith, and Jensen 1974). Oysters at Station 2 received about the same food concentrations as oysters at Station 1. Since effluent temperatures were higher, however, more energy was necessary for maintenance of oysters at the effluent station. In addition, the studies of Ukeles (1961) and Yanase and Imai (1969) suggest that, at temperatures in excess of 30°C, many species of algae which are most acceptable for shellfish culture are killed. In addition, Dame (1972) found that the assimilation rate of food consumed was lower at high ambient temperatures, owing to increased metabolic demands. Thus, by affecting the species composition of unicellular algae and the rate of assimilation by the oyster, a thermal effluent may adversely affect the feeding capabilities of the oyster during midsummer.

Results indicate several significant deviations from control patterns for oysters exposed to the thermal effluent. Spawning-stage data indicate that the effluent has the effect of lengthening the reproductive season and shortening the midwinter indifferent period. In the spring of 1971, gonadal development and the occurrence of ripe individuals was advanced by a month at the effluent station. In September 1970 and 1972, ripe-spawning individuals were sampled at the effluent station, probably several weeks after control oysters had started resorption.

The follicle index provides partial confirmation of these data, regarding whole-gonad response. Significant differences in gonad level in March and May 1971 and April 1972 document the precocious development of oysters held in the thermal effluent. A significant difference in September 1972 shows that late seasonal spawning probably occurred at the effluent station. Gonad-density data confirm this, showing significant differences between stations during the spring (March and May 1971 and April 1972) and during the fall (September 1970, 1971, and 1972).

Spawning-stage data, artificial-spawning attempts, and gonad-density data show that oysters held in the effluent in 1971 became ripe enough to spawn earlier than oysters held in ambient water. Some natural premature spawning may have occurred, but gonad-density data indicate that the time of maximum ripeness occurred during July at both stations. In 1972, initial development was earlier in the effluent (follicle index, gonad density), but ripe levels of development were reached at about the same time at both stations, with a maximum in June at both stations. In support of these data are those of Ruddy, Feng, and Campbell (1973), who found precocious development for oysters held in a thermal effluent and extension of the spawning season into early fall. Despite precocious development, they noted that the major spawning occurred at the same time at effluent and control stations. Roosenburg (1965), Gilmore (1973), and Oja (1974) found no differences in developmental level or timing between effluent and control stations.

Another type of difference between the gonadal development of oysters at the two stations can be seen in the

follicle-index data. In addition to the significant differences in spring and fall due to the altered time of development and resorption at the effluent station, the level of gonadal development was significantly different during June through August 1972. This level was also appreciably lower than that noted for effluent oysters during the summer of 1971. Since follicle-index patterns are the reverse of mortality patterns during the summers of 1971 and 1972, it seems likely that the lowered reproductive amplitude of effluent oysters during the summer of 1972 is a stress response, reflected also in the high mortality rate. The condition of the effluent population was apparently poor during the summer of 1972, and the energy available for reproduction was small, resulting in a significantly reduced level of gonadal development. Butler (1955) found a similar pattern for Chesapeake Bay oysters transplanted to Florida waters. These oysters assumed the spawning pattern of the southern race of *C. virginica*, spawning over an extended season at higher temperatures and presumably at a lower level. In the present study the oysters responded to the thermal-stress situation in a manner similar to a southerly shift in latitude. Increased metabolic demands left less energy for the reproduction effort.

There are several implications of these data. Precocious development of oysters at the effluent station could result in the loss of a significant portion of the larvae produced. Oysters exposed to this artificial thermal environment might be triggered to spawn early by a sharp rise in the temperature. This occurred in laboratory spawning attempts during early May 1971 and 1972. At this time, larvae would be washed into ambient waters too cold to support normal development. This would be detrimental to the long-term success of a natural population by reducing larval recruitment.

Since the level of gonadal development of effluent oysters was significantly lower throughout the summer (June to August 1972), it seems probable that the genetic potential of effluent oysters was far less than that of control oysters during some spawning seasons. If this decreased level of reproductive effort continued or showed further decline, there could be long-term effects on the recruitment of oysters inhabiting such effluents. As the portion of our estuaries influenced by thermal effluents increases, care must be taken to avoid detrimental effects on the life processes of many species.

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# Effect of Temperature on Growth, Mortality, Reproduction, and Production of Adult Snails

## ABSTRACT

Shell lengths and egg production were measured weekly under constant ( $K$ : 10, 15, 20, 25°C) and varying temperature regimes during the reproductive period. Varying regimes included field temperature in a pond ( $F$ ; diurnal and seasonal), mean daily field temperature ( $\bar{F}$ ; seasonal), and 5 and 10°C above  $\bar{F}$ . Groups of snails at regime  $F$  were exposed to both diurnal and seasonal temperature variation, and groups at  $\bar{F}$ ,  $\bar{F} + 5$ , and  $\bar{F} + 10$  were exposed to seasonal variation alone (i.e., temperatures were changed once every 24 hr). Growth rate of large snails (>10 mm) was unaffected by temperature, but small snails (6 to 10 mm) grew fastest at 15°C( $K$ ). Growth and reproductive periods were longest, production was highest, and mortality rate was lowest at 15°C( $K$ ). Rate (per snail) of egg production increased with temperature. At equal mean temperature, regime affected growth rate only at  $\bar{F}$ . Regime affected the following values as shown: mortality rate,  $F > \bar{F} = K$ ; rate of reproduction,  $F > K > \bar{F}$ ; and total production,  $K > \bar{F} = F$ . The validity of extrapolation of energetic data from laboratory to field is discussed. Data relating production and temperature are valuable in thermal-impact analysis.

Studies involving field observation of natural populations and concomitant studies in the laboratory where conditions can be controlled are vital to the understanding of the factors affecting production (Kinne, 1963; Barfield, 1968; Kajak, 1970; Klekowski et al., 1970; Winberg, 1971). One of the most important abiotic environmental factors is temperature. Much work has been done measuring physiological rates at different temperatures. Temperature in the field varies seasonally and diurnally, however, and most laboratory work has been carried out at constant temperature. Recent work has shown that cycled temperature regimes do not always affect physiological rates that are important in production measurement in the same way as do constant temperatures (Van Winkle, 1969; Costlow and Bookhout, 1971; Kelso, 1972; Mattice, 1975; Pattee, 1975; Roux, 1975); but no comparisons of production in constant or cycled temperatures have been made, even though results of these comparisons are important for prediction of

the environmental impact of thermal effluents. The present work was done to compare the production of the snail, *Lymnaea obovata* Say, exposed to natural temperature variation (diurnal and seasonal), with that of snails exposed to seasonal and constant temperature regimes in the laboratory.

## MATERIALS AND METHODS

Specimens of the pulmonate gastropod mollusk, *L. obovata* Say, were collected at 12°C on Apr. 5, 1973, from a pond on the Energy Research and Development Administration Reservation near Oak Ridge, Tenn. The pond has a surface area of about 8800 m<sup>2</sup> and a maximum depth of about 3 m. Annual temperature variation is relatively small. Pond temperatures varied over a range of about 12 to 23°C for the years 1972 to 1973. The population of *L. obovata* is limited to depths less than 50 cm where the pond plants *Ludwigia* sp. and *Elodea* sp. and their associated periphyton dominate the flora. The water temperature of this area is more variable than the rest of the pond. Maximum daily variation for the period from April through August 1973 was 5.5°C, and the mean was 2.1°C.

Snails were acclimated in the laboratory at a rate of less than 1°C per day to a series of constant or varying test temperatures. The constant temperatures were 10, 15, 20, 25, and 30°C. The varying regimes included the field temperature in the pond ( $F$ ), mean daily field temperature ( $\bar{F}$ ), and temperatures exceeding  $\bar{F}$  by 5 ( $\bar{F} + 5$ ) and 10°C ( $\bar{F} + 10$ ). The latter three regimes were maintained in the laboratory. Field temperature was monitored using temperature recorders in cages placed beside the experimental cages in the pond. Each noon the mean field temperature for the previous 24-hr period was estimated from the temperature recording. Regime  $\bar{F}$  was then set at the estimated mean field temperature for the next 24-hr period, and  $\bar{F} + 5$  and  $\bar{F} + 10$  were set appropriately. Mean temperatures for the total period of observation were obtained by averaging the daily temperatures. Mean tem-

peratures for the varying regimes were  $\bar{F}$ , 17°C;  $\bar{F} + 5$ , 21.5°C; and  $\bar{F} + 10$ , 26°C. The intervals were not exactly 5°C because of differences in the length of the reproductive period. Groups of snails at regime  $\bar{F}$  were exposed to both diurnal and seasonal temperature variation, whereas groups at  $\bar{F} + 5$ , and  $\bar{F} + 10$  were exposed to seasonal variation alone (i.e., temperatures were changed once every 24 hr).

Flow-through cages with a capacity of about 3 liters were used to hold the snails at constant or varying temperature regimes (Fig. 1). Each cage was separated into halves by a divider and covered with commercial clear-plastic wrap. Each side contained about 3 cm of mud and gravel and two strands each of *Ludwigia* sp. and *Elodea* sp. from the pond. Four cages, each initially containing 10 snails, were maintained at each different temperature. The cages were kept in 190-liter aquaria through which ran a constant supply of well water maintained at the proper temperature ( $\pm 0.3^\circ\text{C}$ ). Natural day length was approximated by photocell control of fluorescent lights above the aquaria. The lights were either fully off or fully on. Initial shell lengths (SL) of snails were less than 10.0 mm in two of the cages at each temperature and greater than 10.0 mm in the other two cages. Two sets of 36 cages each were used. During the week that one set of cages was being used, the second set (without snails) was held in the pond to allow accumulation of aufwuchs and settled seston. A similar food supply was thus provided for snails at all

regimes. During weekly exchange of cages, shell lengths (the distance from the apex of the shell to the farthest point on the aperture) of live snails were measured to the nearest 0.1 mm with a dial caliper. Eggs and dead snails were also collected. The eggs were preserved in 12% neutral Formalin for later counting. The live snails were transferred to the second set of cages which had been held in the pond, and the new cages were placed in the aquaria. The old cages were then placed in the pond. Observation of snails at a regime was ended arbitrarily whenever either (1) only 5 of the original 40 snails were left alive or (2) when no eggs had been laid at that regime for two consecutive weeks. Live snails collected at the end were fixed in 12% neutral Formalin.

Growth and egg production were converted to equivalent units to estimate production. Total organic carbon of both snails and eggs was estimated using the wet oxidation method of Russell-Hunter et al. (1968). Eggs collected at the beginning, at about the middle, and near the end of the reproduction period at 10, 15, and 20°C were analyzed. Snails held in cages like the experimental ones were sampled about 2 weeks after initial oviposition and at about the middle of the reproductive period. These snails as well as those collected about 2 weeks postoviposition were also analyzed for total organic carbon.

## CALCULATIONS

Mortality rates were calculated from least-squares fits to the relationships between cumulative mortality (percent of the snails present at the beginning of observation) and time. Although some escape occurred, most was during the first 2 to 3 weeks. Therefore these snails were not counted as part of the original number. Mortality rates for each cage were included in the calculation of production. All but 6 of the 31 coefficients of determination exceeded 0.70. Differences in mortality rate with size and temperature were tested using analysis-of-variance (ANOVA) procedures. For comparison of mortality rates at the different temperatures, all groups at a temperature were pooled to calculate one rate.

Because weekly measurements were of SL, growth data were initially computed on that basis. The dividers in the cages were not perfect, and occasionally snails would move from one side to another. However, with only five snails on each side of the cage, it was possible to follow individual snails in each group. Data were converted to the average change in SL for all the snails in the group. Growth was thus

$$G = \frac{\sum_{i=1}^{i=n} (SL_{t+1} - SL_t)}{n}$$

where SL is the shell length at the beginning ( $t$ ) and end ( $t + 1$ ) of the week, and  $n$  is the number of snails living at the end of the week. The growth increments were then added to the original mean SL of the group to give the estimated mean snail size at the end of each weekly interval.

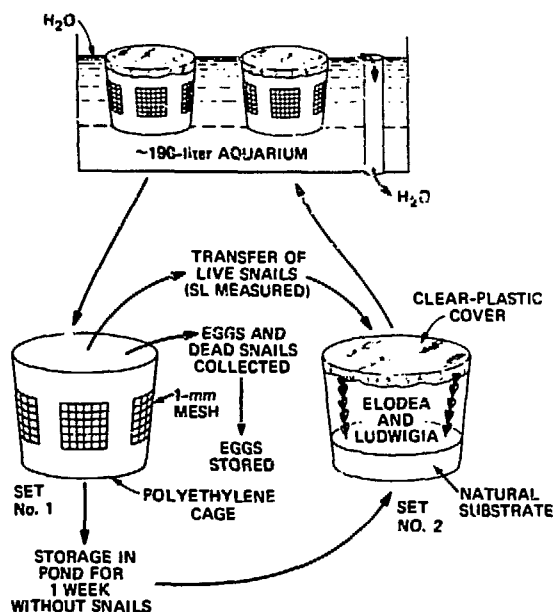


FIG. 1 Protocol for the experiment involved four cages at each temperature regime and two sets of cages. Measurements of SL, count of eggs laid, and snails dead were made each week. Each set of cages was stored in the pond on alternate weeks to collect natural food. For further discussion, see text.

These weekly mean SL values were converted to units of total organic carbon, using an empirical fit to data obtained from snails sampled about 2 weeks before initiation of egg laying. Least-squares fits to a third-order polynomial were found for snails before, during, and after oviposition. The polynomial was

$$y = \alpha x^3 + \beta$$

where  $y$  is total organic carbon ( $\mu\text{g}$ ),  $x$  is shell length (mm), and  $\alpha$  and  $\beta$  are constants.

Following conversion of mean SL to micrograms of carbon ( $\mu\text{g C}$ ), the growth rate was determined by calculating the best least-squares fit to cumulative carbon content over time. In most cages, growth continued to the end of the experiment. In cases where growth stopped during the reproduction period, the end of the growth curve was truncated until the maximum coefficient of determination was obtained and the growth rate was taken as the slope of that line. Factors affecting growth rate were then determined by ANOVA.

Egg production per snail per week was expressed in terms of total organic carbon, using the carbon value obtained per egg. Data in terms of numbers of eggs produced were presented earlier (Mattice, 1975); reproductive output in terms of carbon is presented here. Confidence limits for egg carbon production used here are based on ANOVA. Means and 95% confidence intervals (CI) for constant temperatures of 17, 21.5, and 26°C were estimated using a least-squares fit to a polynomial

$$E[Y(T)] = \alpha + \beta T + \gamma T^2$$

where  $E[Y(T)]$  is the expected or mean egg-production rate at constant temperature  $T$  and  $\alpha$ ,  $\beta$ , and  $\gamma$  are fitted constants. Analysis of the regression indicated that the polynomial accounted for a large percentage of the variability ( $R^2 = 0.91$ ) and that the lack of fit for the quadratic equation was not significant.

Calculation of production in terms of total organic carbon was made for each individual cage of a regime using the growth, mortality, and egg-production rates and reproductive and growth period lengths observed for snails of that cage. Calculations were based on a starting snail number of 100. Carbon production of eggs and growth were calculated separately for each week. Growth and egg-production rates were each multiplied by the number of live snails existing at the end of the week to estimate total carbon produced in each category for that week. The sum of each of these over the weeks that they occurred gave totals for the reproductive period. Before calculation of total production, total growth carbon was corrected for negative growth that occurred toward the end of the reproductive period. Comparison of "before" and "end" curves relating SL to total-body carbon showed the necessity for this correction. It was assumed that this negative growth applied to all snails dying during the period. The number dying in each weekly period was derived from the mortality rate. Size at death for each snail

was derived from the growth curve. The negative growth component was taken as the difference in carbon value predicted by the "before" and "end" curves relating SL to body carbon for a snail of that size. The sum of these differences for snails dying during the reproductive period, as well as those alive at the end of the period, was then subtracted from the estimate of total carbon growth as calculated above and the result summed with reproductive carbon to give an estimate of total production.

## RESULTS

### Mortality

The mortality rate varied significantly with temperature ( $P < 0.01$ ), but not with size ( $P > 0.75$ ). The minimum mortality rate was found at 15°C, and the rate increased at both higher and lower temperatures (Fig. 2). Mortality rates for groups  $F$  and  $\bar{F} + 5$  appeared somewhat higher than that expected for the same mean constant temperature. Variance of mortality rate increased at higher temperatures.

### Growth

For a given SL, a snail sampled during the reproductive period had a higher carbon content than one sampled before oviposition, whereas a snail sampled after oviposition had a lower carbon content than before oviposition (Fig. 3). The relationships between SL and total organic carbon for snails before, during, and after the reproductive period differ significantly ( $P < 0.05$ ). The slopes ( $\hat{\alpha}$ ) of the relationships for snails before and during reproduction were not significantly different ( $P > 0.5$ ), but both differed ( $P < 0.05$ ) from the slope found for snails sampled after the reproductive period. Calculation of growth as carbon is based on the relationship between SL and carbon for the

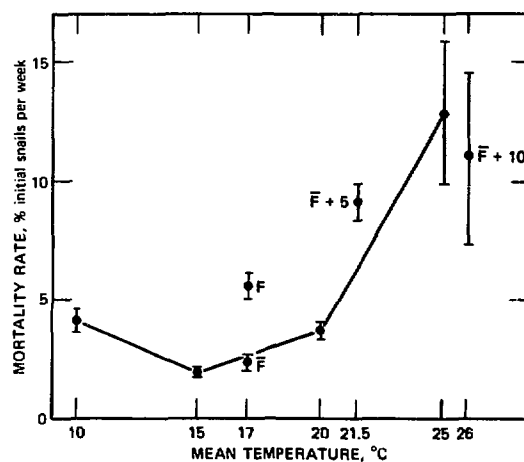


FIG. 2 Mortality rate and 95% CI shown for the various mean temperatures. The CI were calculated from a least-squares fit of percent total snails (of those alive at the start) dead vs. time. The line connects mean mortality rates at constant temperature.



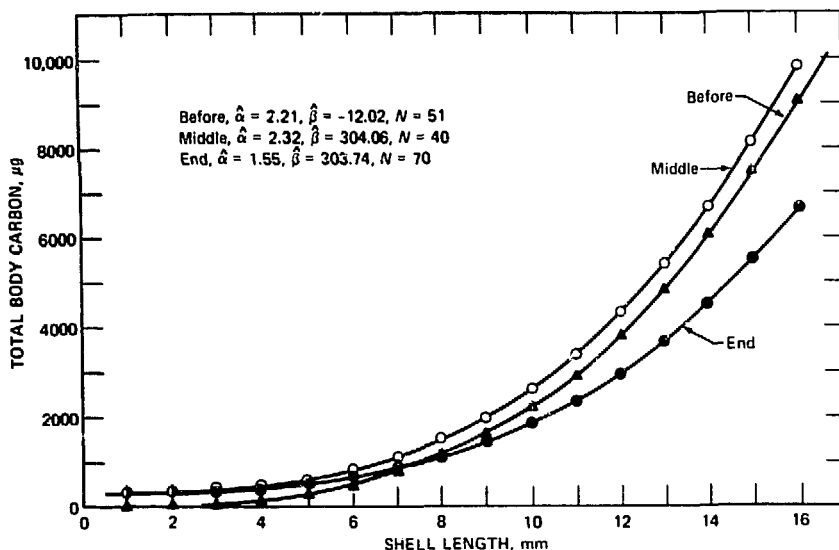


FIG. 3 Relationships between SL and total-body carbon, found by fitting a third-order polynomial  $y = \alpha x^3 + \beta$  ( $y$  = total body carbon and  $x$  = shell length) to data for snails sampled 2 weeks preoviposition ("before"), during oviposition ("middle"), and 2 weeks postoviposition ("end"). Curve parameters are also shown.

preoviposition period. The rationale for this will be discussed in a subsequent section.

Results of ANOVA showed that size affected the relationship between temperature and growth rate. There was a significant relationship ( $P < 0.025$ ) between temperature and growth rate for the small snails, but there was no significant relationship ( $P > 0.75$ ) between temperature and growth rate for the large snails (Fig. 4). Growth rate of the smaller snails was maximum at  $15^\circ\text{C}$  and was significantly lower at 10 and  $20^\circ\text{C}$ . Growth rate appeared to level off at about  $40 \mu\text{g snail}^{-1} \text{ week}^{-1}$  at temperatures from above  $21.5^\circ\text{C}$ . No animals lived longer than 2 weeks at  $30^\circ\text{C}$ . Growth rates for groups F,  $\bar{F} + 5$ , and  $\bar{F} + 10$  appeared to fit the relationship indicated by the results at constant temperature. Growth rate of group  $\bar{F}$  was significantly lower than F and did not fit the general relationship indicated by the curve. Conversely, growth rates for the snails with larger initial SL were not significantly related to temperature ( $P > 0.75$ ). The 95% confidence interval for these larger snails was  $\pm 41 \mu\text{g C}$ .

Temperature also affected the length of the growth period [Fig. 5(a)]. Three groups at  $15^\circ\text{C}$ , one group at F ( $17^\circ\text{C}$ ), and two groups at  $\bar{F}$  were still growing at the end of the observations. Growth period for these groups may thus be somewhat underestimated. There was no difference in length of the growth period for large and small snails.

## Reproduction

Length of reproductive period varied with temperature [Fig. 5(b)] in a fashion similar to that of growth period. Egg production lasted longest at  $15^\circ\text{C}$ . If groups F and  $\bar{F}$

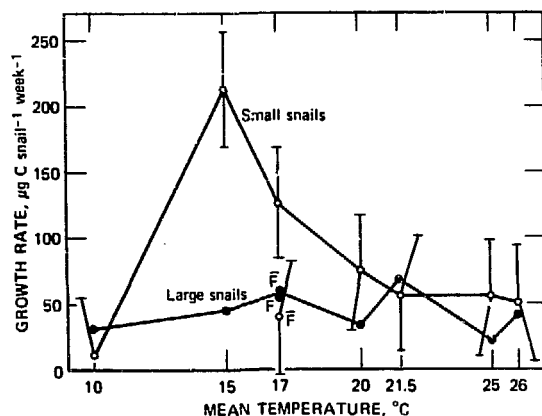


FIG. 4 Mean carbon growth rate and 95% confidence intervals for the various temperatures for two size classes of snails. Results for the varying temperature regimes are shown at the mean temperature over the reproduction period. Lines connect the mean rate obtained for each temperature. The CI were calculated from the pooled error mean square obtained from one-way ANOVA for each size. The CI (not shown) for the large snails was  $\pm 41 \mu\text{g C}$ .

are ignored, a relatively smooth decrease in the egg-laying period is apparent with increasing temperature. The egg-laying periods for groups F and  $\bar{F}$  are shorter than would be expected. Egg laying is also shorter at  $10^\circ\text{C}$ . Three of the four groups of snails at the F regime were still laying eggs at the time of termination. Length of the egg-laying period for F is thus an artificial underestimate of the true value.

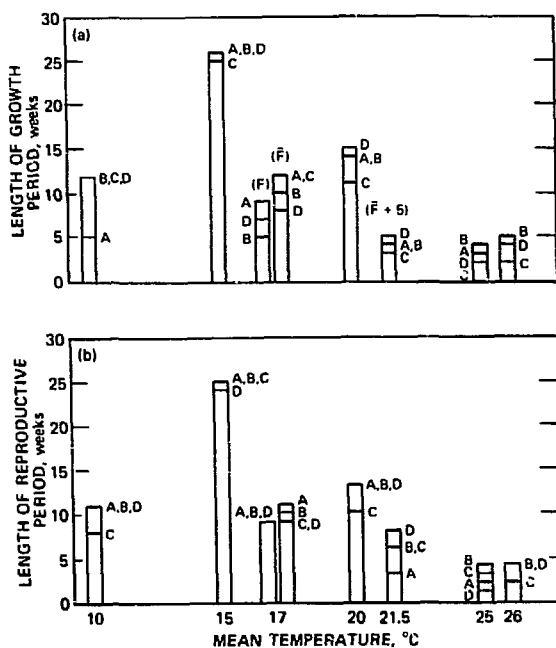


FIG. 5 Mean weeks of (a) growth and (b) reproduction during the reproductive period at the various temperatures. Times for the varying temperatures are shown at the mean temperatures for the reproductive period.

Eggs per snail per week were expressed in terms of total organic carbon (Table 2) following estimation of the mean egg carbon content (Table 1). Carbon per egg did not vary significantly with either temperature or time ( $P > 0.05$ ). Therefore  $22.7 \mu\text{g C/egg}$  was used to convert numbers of eggs to weight of carbon produced. Egg-production rates in terms of carbon increased with increase in temperature, but regime also affected the value of the rate (Table 2). Egg carbon-production rates by groups at  $\bar{F}$  (seasonal and diurnal variation) were significantly higher and by groups at  $\bar{F}$  (seasonal variation) were significantly lower than for the predicted value for  $17^\circ\text{C}$  constant temperature ( $P < 0.05$ ). Carbon-production rates by groups at  $\bar{F} + 5$  and  $\bar{F} + 10$  were lower than, but not significantly different from ( $P > 0.05$ ), that predicted for the corresponding mean constant temperatures ( $21.5$  and  $26^\circ\text{C}$ ), respectively.

### Production

Both size ( $P < 0.05$ ) and temperature ( $P < 0.005$ ) significantly affected total production (Fig. 6). Production for the individual groups ranged from  $-32.4$  to  $889.4 \text{ mg C}$ . There were two negative values, one at a mean temperature of  $25^\circ\text{C}$  and the other at  $26^\circ\text{C}$ . Production was highest at  $15^\circ\text{C}$ . Production was greater for snails initially smaller than  $10 \text{ mm}$  than for snails initially larger than  $10 \text{ mm}$ . Total production for large and small snails at  $\bar{F} + 5$  and  $\bar{F} + 10$  and for the large snails at  $25^\circ\text{C}$  was not significantly

TABLE 1 Mean Carbon ( $\mu\text{g}$ ) per Egg and 95% Confidence Intervals for Eggs Collected at Different Times During the Period of Oviposition at Three Temperatures

| Temperature, °C | Early period*              | Middle period*  | Late period*   |
|-----------------|----------------------------|-----------------|----------------|
| 10              | $22.7 \pm 5.4$             | $28.2 \pm 10.1$ | $22.4 \pm 7.4$ |
| 15              | $22.7 \pm 2.6$             |                 | $19.7 \pm 4.6$ |
| 20              | $24.8 \pm 8.8$             | $18.1 \pm 3.27$ | $19.0 \pm 4.4$ |
|                 | Grand mean $22.7 \pm 1.91$ |                 |                |

\*Early, middle, and late are in relation to the reproductive period at the temperature given; actual dates of sampling differ.

TABLE 2 Rate of Egg Production ( $\mu\text{g C snail}^{-1} \text{ week}^{-1}$ ) at Different Temperatures and Temperature Regimes

| Regime    | At mean temperature of |                    |                    |                    |                      |                    |                    |
|-----------|------------------------|--------------------|--------------------|--------------------|----------------------|--------------------|--------------------|
|           | $10^\circ\text{C}$     | $15^\circ\text{C}$ | $17^\circ\text{C}$ | $20^\circ\text{C}$ | $21.5^\circ\text{C}$ | $25^\circ\text{C}$ | $26^\circ\text{C}$ |
| Diurnal   |                        |                    | 406*               |                    |                      |                    |                    |
| Constant  | 208                    | 259                |                    | 322                |                      | 507                |                    |
| Constant† |                        |                    | 273‡               |                    | 379§                 |                    | 540¶               |
| Seasonal  |                        |                    | 166                |                    | 275                  |                    | 493                |

\*Except where indicated, 95% confidence interval (CI) =  $\pm 86 \mu\text{g C}$ .

†Predicted from polynomial; see text.

‡95% CI =  $\pm 33 \mu\text{g C}$ .

§95% CI =  $\pm 37 \mu\text{g C}$ .

¶95% CI =  $\pm 57 \mu\text{g C}$ .

different from zero. Small snails at  $\bar{F}$  and  $\bar{F}$  appear to fit the relationship for the large snails more closely than that for the smaller snails.

## DISCUSSION

### Shell Length—Carbon Relationship

Use of the previviposition relationship between total-body carbon and SL for calculation of growth rate assumes that no eggs are present in the snails at that time. Evidence that the increase in the relation between total-body carbon and SL from before oviposition ("before") to during the oviposition period ("middle") is due to an increase in reproductive-carbon content of the snails is circumstantial. The "middle" snails analyzed for total organic carbon averaged about  $11 \text{ mm SL}$ . The difference in total carbon predicted from "before" and "middle" curves (Fig. 3) for an  $11\text{-mm}$  snail is  $396.4 \mu\text{g}$ . A weighted mean for weekly production of egg carbon based on the number of snails analyzed from each temperature and the corresponding carbon-per-adult-per-week value from Table 2 gave a mean and 95% confidence interval of  $378.3 \pm 40.8 \mu\text{g}$ . The most logical reason for the lack of significance between these figures is that the "middle" snails contained eggs and the "before" snails did not. Increase in reproductive tissue is not limited to eggs (sperm must comprise some of the

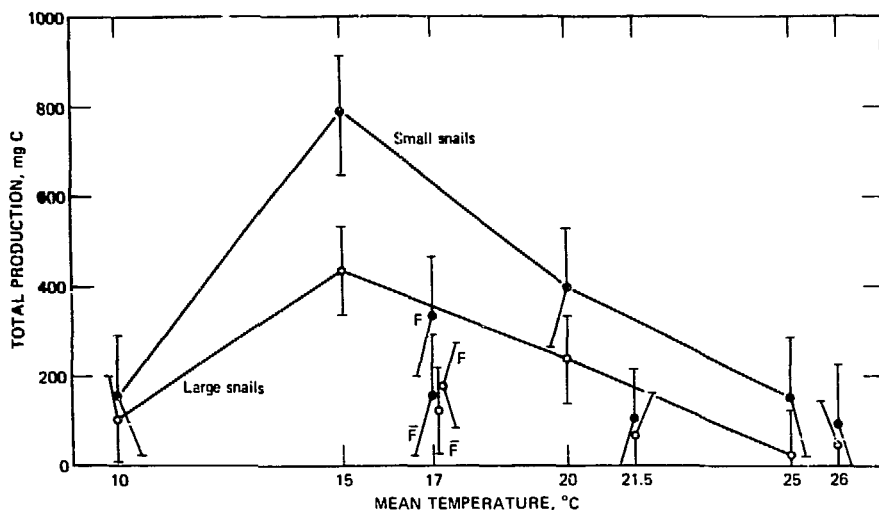


FIG. 6 Mean total production and 95% CI in terms of carbon at the various temperatures for two size classes of snails. Production for the varying temperatures is shown at the mean temperature for the reproductive period. Points for large snails at F and  $\bar{F}$  (17°C) have been displaced slightly for clarity. The CIs for large and small snails were calculated separately from the error mean square found through one-way ANOVA. Mean production values at constant temperatures are connected by the upper line for small snails and by the lower line for large snails.

increase), or even to gametes. Few snails, however, would probably be full of completed egg masses. Egg membranes are added in the oviduct. The closeness of the carbon values (378 and 396) may thus be somewhat fortuitous, but the increase from "before" to "middle" periods probably is due mostly to reproductive carbon.

Depletion of body carbon apparently occurs (1) rapidly and (2) near the end of the reproductive period. Both these assumptions are critical to use of the preoviposition curve in calculating growth rate. The rapidity with which the decrease occurred in snails at 25°C and  $\bar{F} + 10^\circ\text{C}$  (3 weeks or less) indicates the probability of the first assumption. As for the second, there is only one other explanation for the carbon increase from "before" to "middle": reproductive carbon is stored prior to initiation of oviposition, and egg production continues until it is depleted. However, this would require that snails of 8 mm SL store double their body carbon in about 2 weeks. This does not seem very likely. Rapid depletion in the last weeks of oviposition is more reasonable.

The changes in the relation between total carbon and SL have implications for future studies of production in populations that are short-lived and have a relatively prolonged reproductive period. The continuous intake of food and release of eggs indicate that measurement of snail weight or carbon change over the reproductive period underestimates production since only the reproductive carbon contained by the snail at the moment would be measured. Thus a constant weight could be measured over a period of weeks, and the carbon released in the eggs could be over 500  $\mu\text{g C}$  per snail per week (at 25°C). The amount

of error involved in the production estimate would depend on the length of the reproductive period and the difference between the eggs (or sperm) held in the animal at one time and the total released. To avoid error requires that organisms of this type be treated as having continuous reproduction.

### Effect on Production Size

The lower production of large snails in relation to small ones implies a decreased ability to obtain food and/or a decreased efficiency in utilizing it. The difference reflects differences in growth since no relation was found between size and either mortality or egg production. Evidence exists that feeding rate (Newell, Pye, and Ahsanullah, 1971), assimilation efficiency (Kelso, 1972), growth rate (Shelbourn, Brett, and Shirahata, 1973), and conversion efficiency (Kinne, 1960) can decrease with increasing animal size, but data presented here do not allow decision as to the applicability of any or all of these to *Lymnaea*.

### Temperature

Production exhibits the general rule of increase with temperature to a maximum, followed by a decrease at higher temperatures (Precht et al., 1973). Growth rate, survivorship, and lengths of growth and reproductive periods also follow this rule. All are optimal at 15°C. Other *lymnaeid* species also have optima between 12 and 20°C (Vaughn, 1953; van der Schalie and Berry, 1973). These optima are relatively low compared to other snails. This may be due in part to the amphibious nature of *Lymnaea*.

*obrusa*. Conversely, egg-production rate increased from 10 to about 25°C. Since no eggs were laid at 30°C, the decrease following an optimum somewhere near 25°C must have been rapid. This increased egg-production rate at the high temperatures was, however, accompanied by a shortened reproductive period so that total egg production was maximum at 15°C. The effect of temperature on production probably revolves around differential effects on such physiological rates as feeding (Newell, Pye, and Ahsanullah, 1971) and respiration, both of which strongly influence various ecological efficiencies. The concurrence of all the optima at 15°C suggests a rather stenothermal organism that would be highly susceptible to thermal pollution. This may account for its wide range and sporadic occurrence (Loenard, 1959).

#### Temperature Regime

Comparison of the production estimates at the different temperature regimes indicates that some factor connected with exposure to daily and/or seasonal regimes negatively influences production. There is little question that the decrease in production at  $\bar{F}$  is real; growth and reproduction were consistently and significantly lower than at the constant regime. However, results at  $\bar{F} + 5$  and  $\bar{F} + 10$  suggest that increase in temperature decreases the difference between the constant and seasonal regimes. Production at  $F$  is apparently also low. However, the significantly greater egg-production and growth rates found at  $F$ , as compared to  $\bar{F}$ , imply that there are differences in the effect of these two regimes, even though the production values were not significantly different. The decrease in production is thus not simply a result of temperature variation per se, but is influenced by the type of variation as well.

The complexity of the effect of regime on production is emphasized by the differences in response of some of the important parameters of production. Rates of growth and reproduction were affected differently by the different regimes. Regime affected only the rate of growth of small snails at  $\bar{F}$ . On the other hand, regime strongly affected egg-production rate, with snails at  $F$  laying eggs most rapidly and those at  $\bar{F}$  least rapidly. Burky (1971) found that egg-laying intensity of *Ferrissia rivularis*, a pulmonate snail, was directly related to short-term temperature fluctuations. The diurnal regime may be a reproductive stimulus in an area of such relatively small temperature change as the pond. This, however, would not account for the lowered egg-production rate at  $\bar{F}$  or the differences in growth rate between groups at  $F$  and  $\bar{F}$ . The similarity in growth and reproductive-period lengths and the differences in mortality are also difficult to fit into an hypothesis for regime effect.

The complexity, as well as the occurrence, of the above effects indicates need for caution in extrapolating production data obtained under constant-temperature conditions to field situations. Snails at  $\bar{F}$  had lower growth rates, shortened growth and reproductive periods, and lower reproduction rates than those at constant tem-

perature. They also had lower growth and reproductive rates than those at  $F$ . These differences imply that an organism which has evolved under relatively strong diurnal temperature regimes may be dependent on them for certain facets of its physiology.

#### Applicability of Production vs. Temperature Data

Data on production at different temperatures are useful in predicting the impacts resulting from thermal releases to the environment. Accurate predictions are important in decision-making processes (e.g., siting power plants). It is not possible, at present, to accurately predict effects on the ecosystem as a whole, and predictions based on individual tolerance data bog down in the extrapolation of percent mortality. Interpretation of effects on populations, especially of important or representative species (Coutant, 1975), is probably the best available alternative. Population production is of importance for prediction because it gives a first estimate of the importance of the population to the ecosystem. In the case of *L. obrussa* adults, increase in summertime environmental temperature would probably result in a decrease in production since the optimum temperature is so low. The relative effect on the population would vary with the actual temperature change and the range over which it occurred. A small change occurring near 25°C could have a strong effect on the population.

The results presented here apply to *L. obrussa* adults exposed to different temperatures as adults. Prereproductive exposure to the same temperatures could also affect fecundity (Precht et al., 1973). Studies are needed comparing total lifetime population production with both constant and varying temperature regimes to yield information of importance in predicting thermal impact.

#### ACKNOWLEDGMENT

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# Effects of Constant and Cyclic Temperature Regimes on the Pelagic Larval Development of a Brachyuran Crab

A. N. Sastry

## ABSTRACT

Larvae of a brachyuran crab, *Cancer irroratus*, were cultured at constant and cyclic temperatures. Survival of all the larval stages cultured at 10 to 20°C and at 15 to 25°C cycles was better than that of larvae grown at comparable constant temperatures. At the 12.5 to 17.5°C cycle, the survival of larvae was less than that at the comparable 10 to 20°C cycle. Larvae cultured at the 17.5 to 22.5°C cycle developed only to the megalops stage. Duration of the zoeal stages at both 12.5 to 17.5°C and 10 to 20°C cycles decreased and of the megalops stage increased when compared to that at constant 15°C. The duration of the megalops stage increased considerably at the 15 to 25°C cycle compared to that at constant 20°C. In the larval development of *C. irroratus*, megalops is a more sensitive stage than the earlier ones. The amplitude and rate of temperature change have affected differently the duration and survival of the zoeal and megalops stages. Variable temperatures of a favorable amplitude and rate of change within the normal range for development appear to confer a greater capacity for tolerance and survival than comparable constant temperatures.

Pelagic larvae of benthic marine and estuarine invertebrates are a vulnerable link in the life cycle, and their development and survival are important for maintaining adult populations. In higher latitudes the larval release is generally synchronized to a period of the year when conditions in the pelagic environment are optimal for development and growth. While in the pelagic environment, the larvae are exposed to varying combinations of temperature, light, salinity, and other factors, especially in coastal and estuarine regions. Among the physical factors, temperature, singly or in combination with other factors, has been generally recognized as an important parameter influencing the duration of development, survival, and the distribution of the larvae.

The development of reliable laboratory techniques for culturing pelagic larvae of brachyuran crabs within recent years provided an opportunity for describing the successive larval stages and determining the combined effects of temperature and salinity on the larval development of a number of species (Costlow, Bookhout, and Monroe, 1960; 1962; 1966; Sastry, 1970; 1975). To a much lesser extent, the physiological and behavioral responses of developing larval stages to alterations in the environment have also been examined (Kalber and Costlow, 1966; Vernberg and Costlow, 1966; Forward and Costlow, 1974; Sastry and McCarthy, 1973). For all these studies the larvae were cultured under some constant conditions optimal for the species.

In an overwhelming number of cases, pelagic larvae experience diurnal, vertical, and temporal changes of temperature during their pelagic existence. The effects of varying environmental temperatures on larval development and survival have not been examined. The only exception to this is a study by Costlow and Bookhout (1971), who determined the effects of cyclic temperatures on the larval development of an estuarine mud crab, *Rhithropanopeus harrisi* (Gould). It is generally recognized, however, that varying environmental conditions have different effects on the development and physiology of organisms than do constant conditions; thus studies of varying factors are valuable for understanding basic adaptive phenomenon and for practical application (Precht et al., 1973). The wealth of information accumulated on the fitness of *Drosophila* populations has convincingly demonstrated that populations in variable climates have wider homeostatic ranges than do those in more equable conditions (Thoday, 1953; Lewontin, 1957; Beardmore and Levine, 1963). This paper presents initial results of a study on the effects of cyclic temperatures on the larval development of a littoral brachyuran crab, *Cancer irroratus* Say.

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## MATERIALS AND METHODS

Ovigerous female *C. irroratus* were collected from Narragansett Bay, R.I. The egg masses removed from the animals were rinsed several times in filtered seawater at ambient temperatures, and the isolated egg strands were incubated at a constant temperature of 15°C in 30‰ salinity seawater and a 14:10 light-dark cycle until hatching (Sastry, 1970). The resulting larvae were cultured at constant temperatures of 10, 15, 20, and 25°C and at 10 to 20, 15 to 20, 12.5 to 17.5°C, and 17.5 to 22.5°C daily cycles and a 14:10 light-dark cycle in Sherer controlled-environment chambers. The light-dark cycle was programmed so that the day regime coincided with the warm temperatures and the dark period with the cold temperatures in each temperature cycle. Within 24 hr after hatching, the larvae were placed in individual compartments of plastic boxes containing 50 ml of seawater of 30‰ salinity and were maintained at the experimental temperatures. The larvae were transferred to fresh seawater in duplicate boxes held at each temperature on alternate days. Freshly hatched *Artemia salina* nauplii were provided as food. Larvae were examined each day to record molting and survival.

Data gathered on the duration of development and on survival of larvae at constant temperatures from hatches made at different times between December and July for several years have been combined for analysis to eliminate seasonal variation. The larvae cultured under cyclic temperatures were hatched in early January 1975 from ovigerous females collected in November and December 1974.

## RESULTS

### Breeding Period and Larval Development

*Cancer irroratus* occurs over a wide geographical range, with a continuous distribution from Labrador to South Carolina; its populations occur in shallow waters in the north and offshore in the south (MacKay, 1943; Williams, 1965; Musick and McEachran, 1972). The species is an important member of the benthic community in Narragansett Bay, R.I. Ovigerous females occur in the bay from December to July, and their numbers increase between April and July (Jones, 1973). Larvae occur in the plankton from April through July, with a peak in May (Hillman, 1964). Eggs can be hatched in the laboratory to procure larvae throughout the period when ovigerous females are available in the bay. During the period larvae occur in Narragansett Bay, the temperature increases from 5 to 6°C in early April to 15 to 19°C by the end of June (Hicks, 1958). Sastry (1970; 1975; unpublished) has described successive larval stages of *C. irroratus* and has determined the effects of various temperature and salinity combinations on larval development. Larval development includes five zoeal stages and a megalops stage (Fig. 1). The megalops metamorphoses into a crab and assumes the benthic life. Maximum survival of larvae to the crab stage was observed at constant 15°C in 30‰ salinity.

### Survival of Larval Stages at Constant and Cyclic Temperatures

Larval development to the crab stage was observed at constant 15 and 20°C (Fig. 2 and Table 1). Survival of all

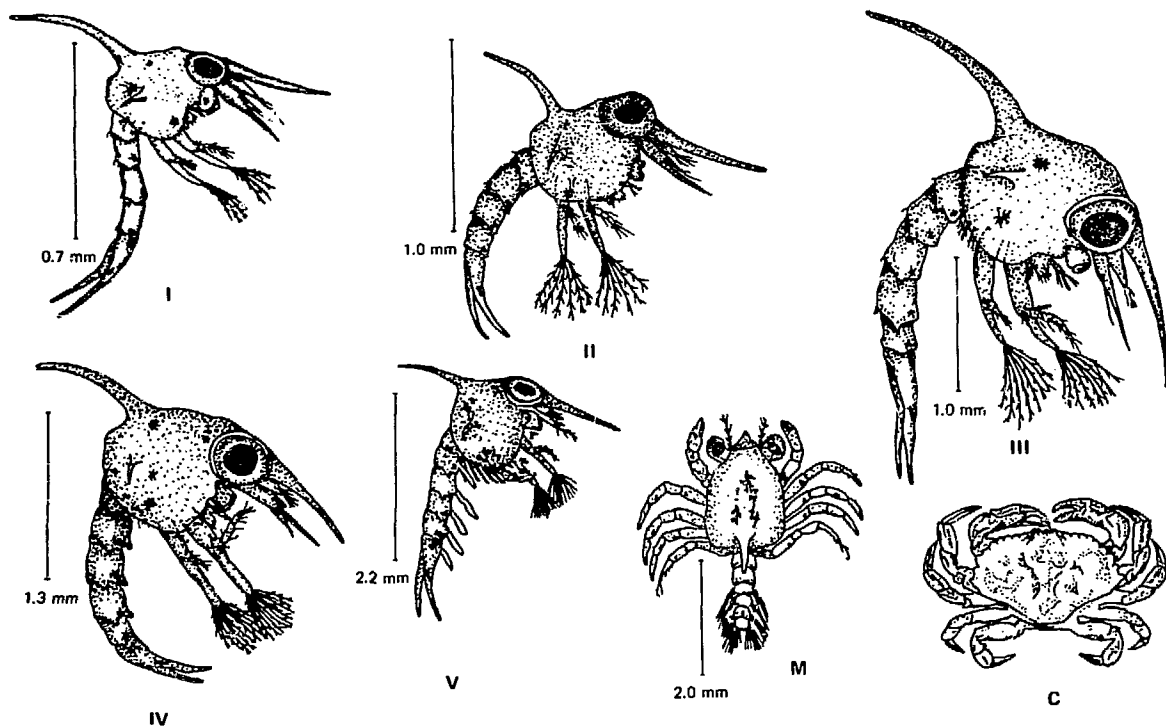


FIG. 1 Life cycle stages of *C. irroratus*. I to V, zoeal stages; M, megalops; C, crab.

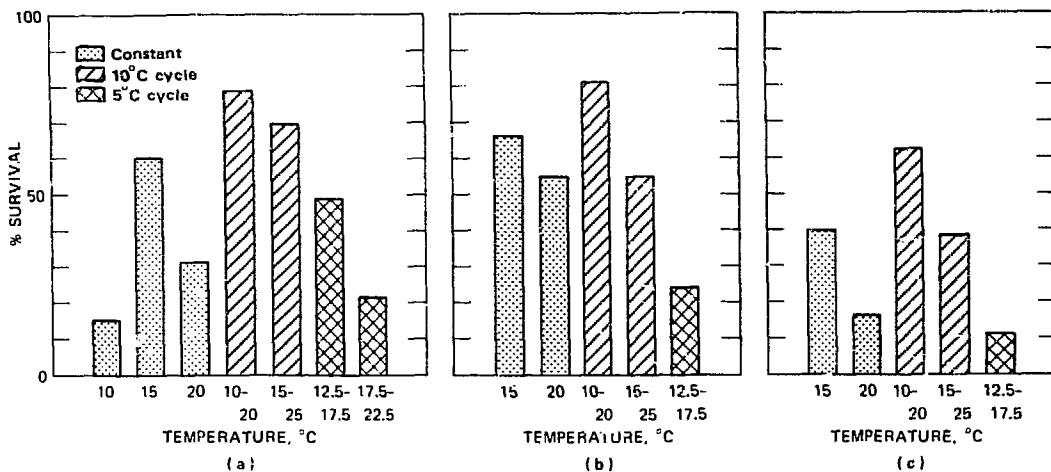


FIG. 2. Percentage survival of *C. irritatus* larvae at constant and cyclic temperatures. (a) Hatch to megalops. (b) Megalops to crab. (c) Hatch to crab.

TABLE 1. Survival of *C. irritatus* Larval Stages at Constant and Cyclic Temperatures

| Larval stage* | Survival at constant temperature, % |      |      |      | Survival at cyclic temperatures, % |         |             |             |
|---------------|-------------------------------------|------|------|------|------------------------------------|---------|-------------|-------------|
|               | 10°C                                | 15°C | 20°C | 25°C | 10-20°C                            | 15-25°C | 12.5-17.5°C | 17.5-22.5°C |
| I             | 54                                  | 85   | 76   | 39   | 88                                 | 87      | 82          | 76          |
| II            | 37                                  | 78   | 68   | 17   | 86                                 | 83      | 72          | 60          |
| III           | 32                                  | 72   | 58   | 6    | 84                                 | 78      | 66          | 40          |
| IV            | 20                                  | 66   | 47   | 0    | 79                                 | 75      | 58          | 27          |
| V             | 15                                  | 60   | 31   | 0    | 78                                 | 71      | 49          | 20          |
| Megalops      | 0                                   | 40   | 17   | 0    | 63                                 | 39      | 12          | 0           |

\*Stages I to V are zoeal stages.

the larval stages was higher at 15°C than at any other constant temperature. Forty percent of the larvae cultured at this temperature developed to the crab stage. Mortalities were highest in the first-stage zoeae and the megalops. Survival of the larval stages at 20°C was higher than that at 10 and 25°C. At 20°C 17% of the larvae developed to the crab stage. Larvae developed only to the megalops stage at 10°C and to the fourth zoeal stage at 25°C. At these two temperatures, the highest mortality of larvae was observed in the first zoeal stage.

Larvae developed to the crab stage at all the temperature cycles except the 17.5 to 22.5°C cycle (Fig. 2 and Table 1). Survival of all the larval stages was highest at the 10 to 20°C cycle when compared with that at the other constant and cyclic temperatures. At this temperature cycle mortalities were highest in the first-stage zoeae and the megalops stage. Sixty-three percent of the larvae developed to the crab stage. The mortality of larvae is significantly lower for fifth-stage zoeae and megalops at the 10 to 20°C cycle than it is at constant 15°C (Table 2). This difference,

TABLE 2. Statistical Analysis\* Testing the Significance of Differences Between Survival of Larval Stages of *C. irritatus* Cultured at Constant and Cyclic Temperatures

| Temperature, °C         | Larval stages† |    |     |    |   | Megalops |
|-------------------------|----------------|----|-----|----|---|----------|
|                         | I              | II | III | IV | V |          |
| 15 vs. 10-20            | -              | -  | -   | -  | + | +        |
| 20 vs. 15-25            | -              | +  | +   | +  | + | +        |
| 15 vs. 12.5-17.5        | -              | -  | -   | -  | - | +        |
| 20 vs. 17.5-22.5        | -              | -  | +   | +  | - | +        |
| 10-20 vs. 15-25         | -              | -  | -   | -  | - | +        |
| 12.5-17.5 vs. 17.5-22.5 | -              | -  | -   | +  | + | +        |
| 10-20 vs. 12.5-17.5     | -              | -  | +   | +  | + | +        |
| 15-25 vs. 17.5-22.5     | -              | +  | +   | +  | + | +        |

\*Minus (-) means not significant; plus (+) means  $p < 0.05$  (Pearson and Hartley, 1954).

†Stages I to V are zoeal stages.



better survival of later larval stages, contributed to the higher survival of larvae to the crab stage in temperature cycles than at constant temperatures.

Survival in each zoeal stage at both 10 to 20°C and 15 to 25°C cycles is not statistically different, but in the megalops stage the mortalities are significantly higher in the warmer temperature cycle. However, survival of each larval stage is significantly higher at the 15 to 25°C cycle than at the comparable constant 20°C. All the larval stages, except the very early stage zoeae, survived better at 10°C cycles than at comparable 5°C cycles. At both the 12.5 to 17.5°C cycle and the comparable constant 15°C, there is no significant difference in survival of each zoeal stage, but the mortalities are higher in the megalops stage at the cyclic temperature. Survival of the third and fourth zoeal stages and the megalops stage at 17.5 to 22.5°C is significantly lower than at the comparable constant 20°C. Comparison between the 5°C cycles shows a lower survival of larvae in the fourth and fifth zoeal stages and the megalops stage at the warmer temperature cycle. In general, survival of zoeal stages, megalops, and hatch to crab is higher at 10°C cycles than at comparable 5°C cycles or constant temperatures.

#### Duration of Larval Development

The experimentally determined duration for each larval stage at different constant temperatures was fitted with Bělehrádek's function (Bělehrádek, 1957; see also McLaren, 1963),  $D = K/(t + \alpha)^b$ , where  $D$  is development time,  $T$  is temperature (°C), and  $K$ ,  $b$ , and  $\alpha$  are constants. The observed data on the development time for each larval stage at different temperatures are adequately described by the curve fitted with Bělehrádek's function (Fig. 3 and Table 3). The development time for all the stages decreased with the increase in temperature. The first zoeal stage developed at a slightly slower rate than the succeeding four zoeal stages. A considerably slower rate of development is predicted from the equation for all the zoeal stages at temperatures below 10°C. A trend for a decrease in the rate of development is also apparent for the first three zoeal stages, which completed development at 25°C. The development time for the megalops stage is considerably longer than that of the earlier zoeal stages at the same culture temperatures. In the 10 to 20°C range, the rate of development of the megalops stage is directly dependent on the temperature. A slower development rate is predicted for this stage at temperatures below 10°C. A trend for the decrease in the rate of development is also apparent above 20°C.

The duration of the zoeal stages decreased and the duration of the megalops stage increased at the 10 to 20 and 12.5 to 17.5°C cycles in comparison with that at constant 15°C (Fig. 4 and Table 4). However, the proportion of total development time required by each of the first four zoeal stages at both constant and cyclic temperatures is approximately the same. The fifth-stage zoeae took less time and the megalops more time at both temperature cycles than at the constant temperature. The cyclic

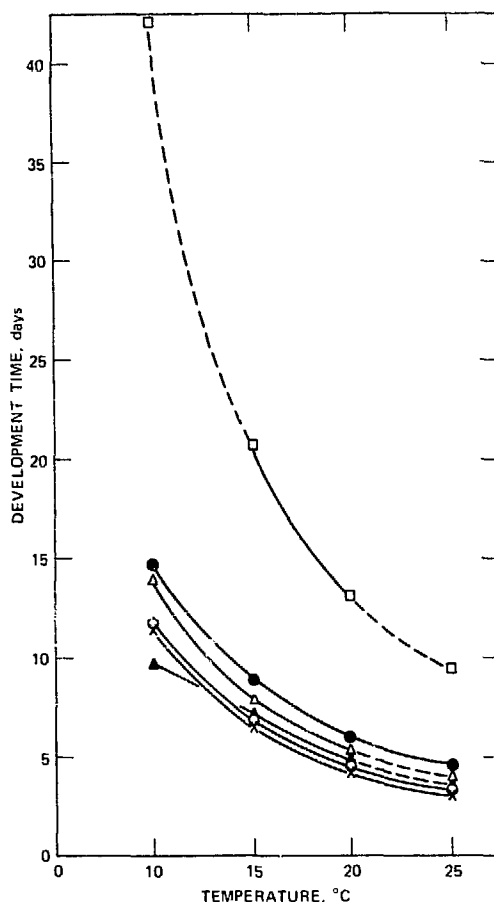


FIG. 3 Effect of temperature on development time for larval stages of *C. irroratus*. ●—●, stage I zoea. x—x, stage II zoea. ○—○, stage III zoea. ▲—▲, stage IV zoea. △—△, stage V zoea. □—□, megalops. Dashed lines on the curves represent extrapolated values.

TABLE 3 Equations Describing the Development Time for Larval Stages at Different Constant Temperatures

| Larval stage* | Equation                    |
|---------------|-----------------------------|
| I             | $D = 500/(T + 1.53)^{1.44}$ |
| II            | $D = 498/(T + 1.72)^{1.53}$ |
| III           | $D = 500/(T + 1.63)^{1.53}$ |
| IV            | $D = 500/(T + 6.6)^{1.40}$  |
| V             | $D = 999/(T + 2.06)^{1.70}$ |
| Megalops      | $D = 500/(T - 3.19)^{1.29}$ |

\*Stages I to V are zoeal stages.

temperature regimes accelerated the zoeal stages and retarded the megalops stage, resulting in approximately the same duration for total development as at a comparable constant temperature.

The duration of zoeal stages is the same at both the 15 to 25°C cycle and a comparable constant temperature of

20°C (Fig. 4 and Table 4). At the 17.5 to 22.5°C cycle, however, the zoeal stages took a longer time than at the comparable constant 20°C. The duration of the megalops stage is also increased considerably at the 15 to 25°C cycle as compared with that at constant 20°C. This delay of megalops development accounted for the longer duration from hatch to crab at the 15 to 25°C cycle. The two warmer temperature cycles have a general tendency to increase the duration of development of the zoeal stages and the megalops stage in comparison with that at constant temperature.

The duration of the zoeal stages decreased at the 15 to 25°C cycle compared with the 10 to 20°C cycle, but this trend was reversed in the megalops stage. The time required for development from hatch to crab is, however, the same at both 10°C cycles despite the difference in the culture temperatures. The duration of zoeal stages at the 17.5 to 22.5°C cycle was slightly longer than at the 12.5 to 17.5°C cycle. The zoeal stages and the megalops stage were differentially sensitive to the amplitude and rate of temperature change; this resulted in a variable duration for their development at different cyclic temperature regimes.

### DISCUSSION

A number of previous studies of the effects of temperature and salinity on the larval development of brachyura have established the optimal conditions and the limits for survival (Costlow, Bookhout, and Monroe, 1960; 1962; 1966; Sastry, 1970). It is evident that larval development, like all physiological processes, occurs within a definite range of environmental conditions. Beyond some upper and lower limit of the environmental range, larval development does not occur. The favorable range, as well as the limits, may vary for successive larval stages, populations, and species in relation to environmental factors acting either singly or in combination. Apparently the variable responses of larvae are a reflection of the environmental flux. Survival of larvae is generally maximum at some combination of the environmental factors within the normal range favoring complete development. Maximum survival of *C. irroratus* larvae has been estimated from

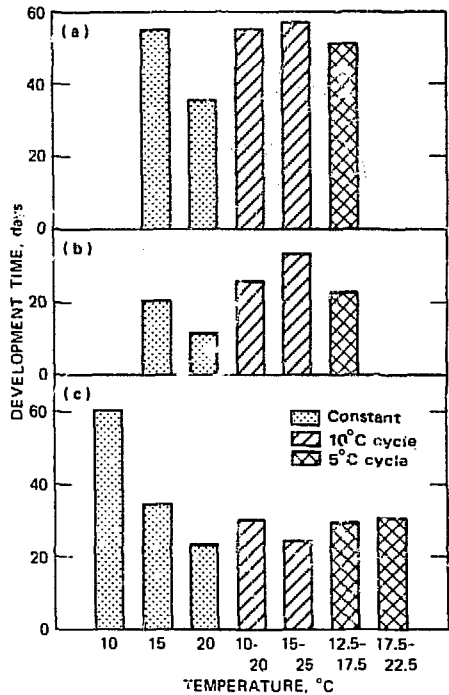


FIG. 4 Duration of larval development at constant and cyclic temperatures. (a) Hatch to crab. (b) Megalops. (c) Hatch to megalops.

multifactorial experiments to occur between 13 to 20°C and 27 to 38‰ salinity (Sastry, unpublished). The limits for 100% mortality have been estimated as temperatures below 9 and above 23°C and salinities below 20‰ and above 38‰. The range of optimal conditions and the limits have been observed to vary, especially when larvae reach the megalops stage. Relationships between environmental factors and survival also seem to vary for larvae hatched at different times during the period ovigerous females are available in Narragansett Bay. Many details of ontogeny of adaptation of larval stages have not been examined, and the variable responses of larvae to environmental alterations

TABLE 4 Mean Days\* and the Proportion of Total Development Time (Hatch to Crabs) for Each Larval Stage at Constant and Cyclic Temperatures

| Larval stage† | Constant temperatures |            |      |            |      |            | Cyclic temperatures |      |             |      |             |      |             |   |
|---------------|-----------------------|------------|------|------------|------|------------|---------------------|------|-------------|------|-------------|------|-------------|---|
|               | 10°C                  |            | 15°C |            | 20°C |            | 10-20°C             |      | 15-25°C     |      | 12.5-17.5°C |      | 17.5-22.5°C |   |
|               | Days                  | %          | Days | %          | Days | %          | Days                | %    | Days        | %    | Days        | %    | Days        | % |
| I             | 15.4 ± 6.8            | 6.5 ± 1.4  | 11.6 | 5.2 ± 1.3  | 14.4 | 4.2 ± 2.4  | 5.1 ± 1.0           | 9.1  | 3.6 ± 1.2   | 6.2  | 5.6 ± 2.7   | 10.8 | 6.9 ± 4.7   |   |
| II            | 26.0 ± 6.6            | 12.9 ± 2.3 | 11.4 | 9.1 ± 2.0  | 10.8 | 8.8 ± 0.41 | 11.4 ± 3.5          | 11.3 | 7.9 ± 2.0   | 7.4  | 11.6 ± 3.6  | 11.7 | 13.3 ± 5.1  |   |
| III           | 38.7 ± 10.3           | 19.4 ± 2.7 | 11.6 | 13.2 ± 2.5 | 11.3 | 13.0       | 17.9 ± 3.5          | 11.6 | 13.6 ± 2.8  | 9.8  | 18.4 ± 3.4  | 13.2 | 18.5 ± 6.4  |   |
| IV            | 41.4 ± 12.6           | 26.0 ± 2.8 | 11.8 | 17.9 ± 1.7 | 12.7 |            | 23.5 ± 3.5          | 10.0 | 19.1 ± 3.5  | 9.5  | 23.9 ± 3.0  | 10.7 | 24.8 ± 3.3  |   |
| V             | 61.3 ± 4.4            | 35.0 ± 3.2 | 16.1 | 32.2 ± 2.0 | 17.7 |            | 36.1 ± 2.5          | 11.8 | 24.3 ± 3.9  | 8.9  | 28.9 ± 2.4  | 9.7  | 30.4 ± 3.7  |   |
| Megalops      |                       | 55.7 ± 4.2 | 37.5 | 36.0 ± 3.8 | 32.8 |            | 56.0 ± 5.5          | 46.3 | 57.9 ± 11.3 | 58.0 | 51.5 ± 2.4  | 43.9 |             |   |

\*Weighted mean.

†Stages I to V are zoeal stages.

remain unexplained. However, these early studies on the effects of environmental factors, determined under constant conditions, have provided some basis for predicting the probable survival and distribution of larvae under natural conditions.

The differences in the survival of larvae of *C. irroratus* cultured at constant and cyclic temperature regimes demonstrate another facet of the complex adaptive phenomenon in larval development. Larvae exposed to daily cycles of 10 to 20, 15 to 25, and 12.5 to 17.5°C appear to tolerate a wider temperature range than those at constant temperatures, and the former complete development to the crab stage. However, the amplitude and rate of temperature change have a different effect on the survival of zoeal stages than on the megalops stage. Costlow and Bookhout (1971) reported better survival of *R. harrisi* larvae cultured at a 30 to 35°C cycle than of those at constant 30 or 35°C. They found no apparent difference in the survival of larvae of this eurythermal and euryhaline mud crab at 10 and 5°C cycles and constant temperatures between 15 and 30°C. In contrast, the survival of *C. irroratus* larvae over 10 and 5°C cycles and comparable constant temperatures is different. The megalops stage of this species is especially sensitive to the amplitude and rate of temperature change in comparison with the earlier zoeal stages. This differential sensitivity of the zoeal and megalops stages to 10 to 5°C cycles is also evident in the duration of larval development under respective culture conditions. Costlow and Bookhout (1971) found no difference in the length of larval life of *R. harrisi* at 5 or 10°C cycles and comparable constant temperatures.

The adaptations of developing larvae to varying environmental conditions are poorly understood. Sastry and McCarthy (1973) determined the metabolic temperature adaptation of successive larval stages of *C. irroratus* cultured at constant 15°C. The larval stages are characterized by major changes in the metabolic temperature-response patterns between first- and second-stage zoeae, third- and fourth-stage zoeae, and fifth-stage zoeae and megalops. The temperature range for metabolic compensation of the larval stages has narrowed from 10 to 20°C in the early stages to 15 to 20°C in the later fourth- and fifth-stage zoeae. The metabolic rate of all the larval stages, especially the third-, fourth-, and fifth-stage zoeae and megalops, is depressed above 20°C. In contrast, larvae cultured at the 10 to 20°C cycle showed an extension of the temperature range for metabolic compensation to 15 to 25°C (Sastry, unpublished). An increase in the tolerance range of the larvae is also indicated by a shift of the temperature range for depression of metabolic rate to 25 to 30°C. The  $Q_{10}$  changes in metabolic rate over the temperature gradient may allow the functioning of different enzyme pathways in a complex multienzyme system for optimal physiological performance. Exposure of larvae to varying temperatures may differentially affect the activity of enzymes and/or induce isozymes, giving the larvae a greater capacity for adaptation than those at constant

temperatures (Somero, 1969; Gilles, 1974). Although the mechanisms in temperature adaptation of developing pelagic larvae have not been examined, properties of enzymes have been demonstrated which allow instantaneous compensation in the metabolism of adult intertidal invertebrates exposed to acute temperature changes, at least over a portion of their natural habitat temperatures (Hazel and Prosser, 1974). Tantawy and Mallah (1961), Beardmore and Levine (1963), and Dawson (1968) hypothesized that populations or species inhabiting variable climates possess greater developmental homeostasis than those at constant temperatures.

The results presented here suggest that both resistance and capacity adaptations of larval stages exposed to varying environmental conditions are different from those of larvae at constant temperatures. Results of studies conducted with organisms maintained under varying conditions appear to be more meaningful for potential application to assessment of the probable impact of thermal alterations in the environment.

## ACKNOWLEDGMENTS

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# Some Effects of Temperature, Chlorine, and Copper on the Survival and Growth of the Coon Stripe Shrimp

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Thomas O. Thatcher  
Charles W. Apts

## ABSTRACT

A series of bioassay and growth-rate experiments were conducted on coon stripe shrimp, *Pandalus danae*, to determine the  $LL_{50}$  value for heat, the  $LC_{50}$  values for chlorine and copper, and the effects of sublethal concentrations of these materials on their growth rate. The critical thermal maxima for three size groups, 1 to 2 g, 4 to 7 g, >9 g, ranged from 26.3 to 31.8°C depending on shrimp size and rate of temperature increase. Bioassays (96 hr) at 10, 15, and 20°C were conducted, using chlorine or copper as the toxicant.

Growth was measured for 1 month at temperatures between 10 and 25°C. The greatest growth occurred at 16°C. Growth of the shrimp held at 16°C while being exposed to sublethal concentrations of both copper and chlorine was studied.

With the accelerated growth in the demand for electrical energy production, it is inevitable that the frequency of selecting coastal or offshore sites for the location of the generating plants will increase. The value of large volumes of coastal water for cooling purposes and for the discharge of effluents is quite attractive to groups making the siting decisions. The impact of these effluents, however, whether they be thermal increments, biocides, trace metals, or radioisotopes, has not been clearly defined. In particular, very little data have been reported on the combined effects of these effluent components, i.e., the effects (either additive, synergistic, or antagonistic) resulting from simultaneous exposure to the several components of the effluents.

In the research program described here, we are investigating the effects of these power-plant effluents, both as individual toxicants and in combination with others. Data are being generated which will assist in predicting both the acute and chronic effects of the discharge of these materials into coastal environments at both the individual species level and the community level.

We are still in the initial phase of the study; so the information reported here is in the form of a progress report. We have begun with the coon stripe shrimp, *Pandalus danae*, which is common in the Pacific Northwest and is a good laboratory experimental animal throughout its life cycle. To date, we have been investigating the effects of three of the most common components of power-plant effluents which are potentially harmful to marine organisms: heated water, chlorine, and copper (Roosenburg, 1969; Forster, 1973; Becker and Thatcher, 1973). The criteria evaluated so far include critical thermal maxima (CTM) tests, 96-hr continuous flow-through lethal concentration ( $LC_{50}$ ) values, and short-term growth.

## METHODS

Shrimp used in the studies were collected from Sequim Bay, Washington, and held in flowing seawater tanks at ambient temperature ( $8^{\circ}\text{C} \pm 0.5$ ) for a 2-week period before acclimation and testing. The shrimp were fed fresh cracked clams, *Protothaca staminea*, and Oregon Moist Fish Pellets during the holding period.

For the CTM tests, individual shrimp were placed in a 2-liter volumetric flask that contained 1 liter of ambient-temperature seawater. The flask was placed in a heating mantle, the temperature of which was controlled by a voltage regulator. The temperature of the water was increased at a given rate by setting the regulator at a predetermined voltage. Water temperature was monitored at 1-min intervals using a calibrated stem thermometer inserted through a rubber stopper. Constant aeration was provided to maintain thorough mixing and an even temperature throughout the flask.

In testing *Pandalus danae* for CTM, there was not a consistent behavior pattern that could be considered a loss of equilibrium; therefore, cessation of gill-bailer movement was used as the end point.

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For these tests the shrimp were divided into three classes: small (1 to 2 g wet weight), medium (4 to 7 g wet weight), and large (>9 g wet weight). A total of 10 shrimp in each class were subjected to four rates of temperature increase: 6.0°C/min, 2.0°C/min, 1.0°C/min, and 0.5°C/min.

The toxicity of chlorine and copper was determined by removing groups of 60 shrimp (2 to 4 g wet weight) from the 8°C acclimation tank and individually distributing them at random among the six exposure aquaria of a bioassay system. Each bioassay system consisted of a 2-liter proportional diluter (Mount and Brungs, 1967) and two sets of six 15-gal glass and fiberglass exposure chambers. The diluters maintained continuous flow-through conditions in the shrimp chambers, and the toxicant exposures extended for 96 hr. The combined effects of temperature and chlorine and/or copper were obtained by altering the toxicant exposure temperature for each new group of animals transferred from the 8°C acclimation chamber; thus groups of shrimp were subjected in successive bioassays to a temperature increase from 8°C to either 10, 15, or 20°C in addition to the toxicant. Several groups were both acclimated and exposed at 15°C. Water samples for toxicant analysis were taken near the bottom of the exposure chambers in close proximity to the experimental animals. The LC<sub>50</sub> values were calculated by probit analysis (Finney, 1952).

Growth experiments were conducted by placing groups of 10 shrimp in 6-liter polyethylene tanks receiving ambient temperature water at a rate of 300 ml/min. The temperature was then increased at the rate of 4°C per day until the desired experimental temperatures were reached. Two groups of shrimp were held at each designated temperature. One group was fed Oregon Moist Fish Pellets at the rate of 10% of their body weight (wet) per day. The other group was starved. It should be noted that the test system was supplied by unfiltered seawater; so some material was available in the tanks not receiving food. However, previous studies have shown that the amount of organic material entering the system is small.

For the thermal growth experiments, three series of tests were run: in two series medium shrimp (4 to 7 g wet weight) and small shrimp (1 to 2 g wet weight) were held at 13, 16, 19, 22, and 25°C for 2 weeks and then returned to 13°C for 2 weeks, and in the third series small shrimp were held at 8.5, 14, 15, 16, 17, and 18°C for 24 days. Wet weight of each shrimp was taken at the beginning of the tests and after 2 week periods.

In the chlorine and copper growth experiments, groups of 10 small shrimp (1 to 2 g) were exposed at 16°C to either chlorine at concentrations of 0.18, 0.08, 0.05, and 0.01 mg/liter or copper at concentrations of 0.041, 0.009, 0.005, and 0.002 mg/liter. Chlorine was metered into the exposure tanks by Mariotte bottle, and copper was metered by a metering pump. Wet weight of each shrimp during these experiments was taken at the beginning of the tests and at the end of each week. The tests were conducted at 16°C for a 1-month period.

In all experiments the chlorine was introduced as a Clorox-distilled water solution (the chlorine being in the form of sodium hypochlorite; the remainder of the solution being water with small amounts of NaCl and NaOH), and the copper was in a CuSO<sub>4</sub>·5H<sub>2</sub>O—distilled water solution. Chlorine concentrations were measured at the infall and the outfall of the growth tanks by the amperometric titration method (American Public Health Association, 1971). Outfall concentrations were 57% (SD ± 10%) of infall concentrations at all test levels except the 0.001-ppm tanks. At this concentration we were working at our detectability limits; therefore, we were unable to distinguish between values >0 and 0.001 ppm. Available chlorine concentrations at the outfall decreased to approximately 14% of infall levels when food was added. The outfall concentration reached 30% of infall within 60 min after addition of food and reached 60% after 210 min. Total copper was measured by atomic absorption spectrophotometer either directly or with ammonium 1-pyrrolidone dithiocarbamate-methyl isobutyl ketone (APDC-MIBK) extraction (Strickland and Parsons, 1968) or with a specific ion electrode. During this phase of the study, the dissolved oxygen ranged between 7.28 and 9.40 mg/liter; the alkalinity was 116 to 121 mg/liter; and the salinity was 28.5 to 31.0 ‰.

## RESULTS AND DISCUSSION

The CTM values for the three sizes of shrimp tested and four test conditions are shown in Table 1. The gradual

TABLE 1 Critical Thermal Maxima (°C) for Three Sizes of Coon Stripe Shrimp, *Pandalus Danae*, at Four Rates of Temperature Increase

| Rate of increase, °C/min | Shrimp, wet weight |                 |                 |
|--------------------------|--------------------|-----------------|-----------------|
|                          | 1 to 2 g           | 3 to 7 g        | >9 g            |
| 0.5                      | 26.3 (SD ± 0.8)    | 27.5 (SD ± 0.4) | 27.0 (SD ± 1.2) |
| 1.0                      | 27.3 (SD ± 0.3)    | 27.8 (SD ± 0.4) | 28.1 (SD ± 0.5) |
| 2.0                      | 27.7 (SD ± 0.5)    | 28.2 (SD ± 0.2) | 28.6 (SD ± 0.8) |
| 6.0                      | 28.4 (SD ± 0.4)    | 29.4 (SD ± 0.6) | 31.8 (SD ± 1.3) |

increase in CTM level with increased size and rate of temperature increase is expected and has been reported for other species by other investigators (Lichatowich, Templeton, and Vanderhorst, 1973; Vanderhorst et al., 1974; Cox, 1974). A significant difference ( $p = 0.05$ ) did not exist between all pairs of conditions, but the CTM levels were significantly different ( $p = 0.05$ ) for the extreme case for both size and rate of temperature increase. The CTM levels for *Pandalus danae* are in the same range as those reported by Vanderhorst et al. (1974) for mysids, Pacific herring larvae, and several other species of Pacific Northwest fishes.

The 96-hr LC<sub>50</sub> values for coon stripe shrimp exposed to the combined effects of temperature change and chlorine are shown in Table 2. The series of tests at 10, 15, and

TABLE 2 The 96-hr LC<sub>50</sub> Values\* of Coon Stripe Shrimp Acclimated at 8°C or 15°C and Exposed to Chlorine at 10, 15, or 20°C

| Acclimation temperature, °C | Exposure temperature, °C | 96-hr LC <sub>50</sub> value, mg/liter total residual chlorine | 95% fiducial limits, mg/liter total residual chlorine |
|-----------------------------|--------------------------|----------------------------------------------------------------|-------------------------------------------------------|
| 8 ± 1.0                     | 10 ± 1.0                 | 0.295                                                          | 0.246 — 0.333                                         |
| 8 ± 0.5                     | 15 ± 0.5                 | 0.178                                                          | 0.159 — 0.199                                         |
| 8 ± 0.5                     | 20 ± 0.5                 | 0.133                                                          | 0.122 — 0.148                                         |
| 15 ± 0.5                    | 15 ± 0.5                 | 0.210                                                          | 0.196 — 0.224                                         |

\*The values represent data composited from either three or four toxicity tests at each set of conditions.

20°C on shrimp acclimated at 8°C resulted in a gradual decrease in tolerance to chlorine. The 95% fiducial limits of these values do not overlap. The tolerance of shrimp acclimated to 15°C and tested at 15°C is higher than for those acclimated at 8°C and exposed at 15°C, but the 95% fiducial limits do overlap. However, the 95% fiducial limits of this value (acclimation and exposure at 15°C) do not overlap with the 8°C acclimation and 10°C test condition. It is interesting that the apparent optimum growing temperature is 16°C (see later in this section), but at this level the shrimp is more susceptible to chlorine.

The 96-hr LC<sub>50</sub> values for the coon stripe shrimp exposed to the combined effects of temperature change and copper are shown in Table 3. There was little difference in

TABLE 3 The 96-hr LC<sub>50</sub> Values\* of Coon Stripe Shrimp Acclimated at 8°C and Exposed to Copper at 10, 15, and 20°C

| Acclimation temperature, °C | Exposure temperature, °C | 96-hr LC <sub>50</sub> value, mg/liter total residual copper |
|-----------------------------|--------------------------|--------------------------------------------------------------|
| 8 ± 0.5                     | 10 ± 0.5                 | 0.037                                                        |
| 8 ± 0.5                     | 10 ± 0.5                 | 0.066                                                        |
| 8 ± 0.5                     | 15 ± 0.5                 | 0.021                                                        |
| 8 ± 0.5                     | 15 ± 0.5                 | 0.049                                                        |
| 8 ± 0.5                     | 20 ± 1.0                 | 0.031                                                        |
| 8 ± 0.5                     | 20 ± 1.0                 | 0.042                                                        |

\*Since these values were estimated from calculated copper concentrations, they are only tentative pending analysis of the preserved water samples.

the values due to temperature change except, perhaps, at 10°C, where one group of shrimp appeared more resistant. Further study will be required before it can be determined whether or not there is a temperature effect on copper toxicity to the shrimp.

During growth experiments, medium-size shrimp (4 to 7 g) were held at ambient (13°C), 16, 19, 22, and 25°C for the first 14 days then returned to ambient for 14 days. For the shrimp being fed, there was an increase in weight in the control group (13°C) throughout both 2-week periods.

At 16 and 19°C there was a weight gain during the period of elevated temperatures but a weight loss during subsequent ambient-temperature exposure. At 22°C the shrimp lost weight during both exposure periods. The only mortalities that were attributed to temperature were at 25°C where all shrimp died within 1 day after reaching that temperature level. The nonfed shrimp showed slight growth at 16 and 19°C during the period of elevated temperature but lost weight at ambient and 22°C. All unfed shrimp died at 25°C also. During the subsequent ambient-temperature exposures, all the unfed shrimp lost weight.

A second group of smaller shrimp (1 to 2 g) were held under similar conditions and exhibited similar growth rates (Fig. 1). The largest weight gain for fed shrimp during thermal elevation occurred at 16°C, with slightly lower gains at ambient (13°C), 19, and 22°C. All shrimp died at 25°C. There was one notable exception between these and the medium shrimp, however. The small shrimp receiving food at 16, 19, and 22°C showed an increase in weight during the 2-week period at ambient temperature, whereas the medium-sized shrimp receiving food showed a decrease. The shrimp receiving no food showed no growth at ambient (13°C), a slight increase at 16 and 19°C, and a decrease at 22°C during thermal elevation. Again, all shrimp died at 25°C. During subsequent ambient exposure, there was little change at ambient (13°C) and 22°C, a slight decrease at 16°C, and a slight increase at 19°C. Because of mortalities and cannibalism in the tanks receiving no food, that portion of the experiment was discontinued after this series of tests.

In the third series, small shrimp (1 to 2 g) were held at ambient (8°C), 14, 15, 16, 17, and 18°C. Unfortunately, a system failure killed the shrimp in the 16°C tank after 17 days. However, the same pattern was again evident with the weight gain at 16°C, double the gain at the other temperatures.

On the basis of these three experiments, it appears that 16°C is the optimal growing temperature for the coon stripe shrimp, at least for short periods of time. There were no noticeable disease problems encountered during the studies.

The 96-hr 50% lethal level temperature, calculated from our growth experiments, is 23.5°C, which is 2.8°C lower than the minimum CTM measured. The water temperatures

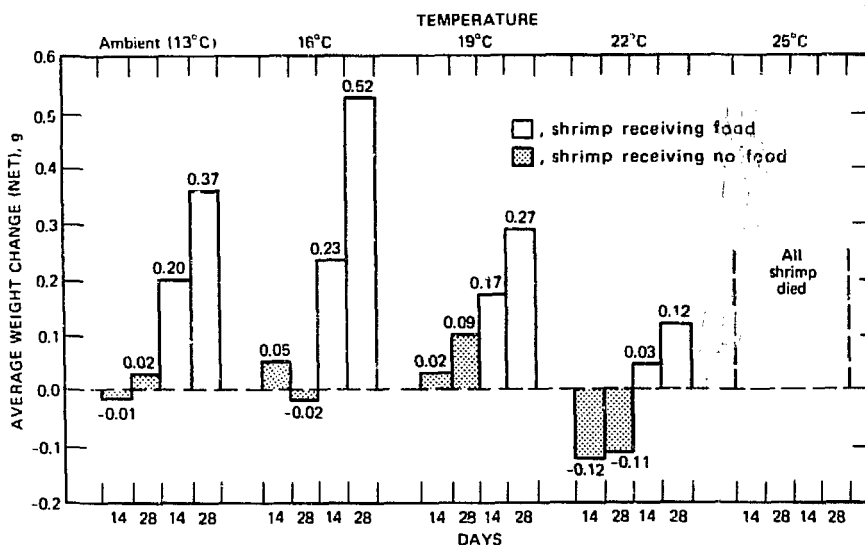


FIG. 1 Net growth for fed and nonfed coon stripe shrimp (1 to 2 g) exposed to 2 weeks of thermal increases followed by 2 weeks of ambient temperature.

of the Straits of Juan de Fuca and northern Puget Sound seldom exceed 13°C; so a power plant operating with a condenser  $\Delta T$  of 10°C or less would have very little effect on the adult stage of *Pandalus danae* in short-term exposures. In addition, the increased growth rate at 16°C, which is above normal summer maximum temperatures found in its natural habitat, makes this species a good candidate for aquaculture using the plant's waste heat.

The results of the copper growth experiment and the chlorine growth experiment are shown in Fig. 2. Small shrimp (1 to 2 g) were exposed at 16°C to chlorine concentrations that averaged 0.18, 0.08, 0.05, and 0.01 mg/liter, or copper that averaged 0.041, 0.009, 0.005, and 0.002 mg/liter. A control was also maintained. The shrimp

exposed to chlorine showed a slight decrease in growth at 0.08 mg/liter and at 0.01 mg/liter, and a slight increase at 0.05 mg/liter compared to the control. All shrimp died at 0.18 mg/liter chlorine. Shrimp exposed to copper showed little difference in growth at 0.009, 0.005, and 0.002 mg/liter and a decrease in growth at 0.041 mg/liter when compared to the control shrimp.

The present Environmental Protection Agency (EPA) recommended allowable effluent concentrations of chlorine (as available chlorine) and copper are 0.01 mg/liter and 0.05 mg/liter, respectively (Environmental Protection Agency, 1973). At the Millstone Power Generating Station in Connecticut, the maximum level of soluble copper found in the effluent between September 1971 and September

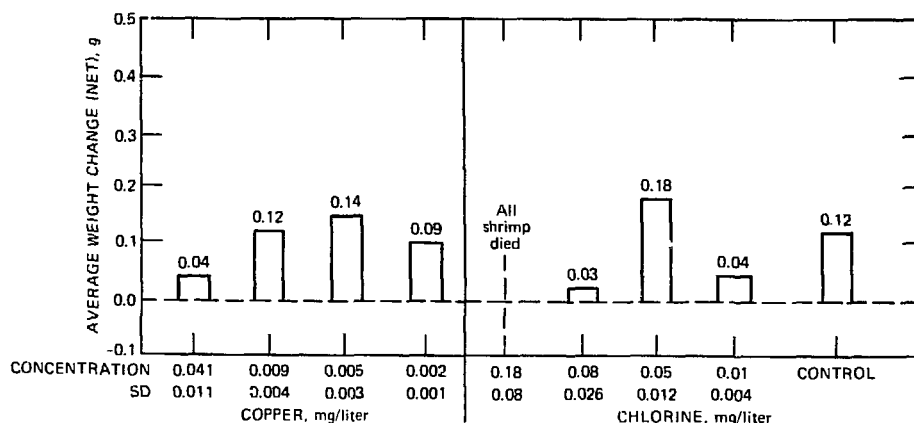


FIG. 2 Net growth for coon stripe shrimp (1 to 2 g) exposed to copper or chlorine for 1 month at 16°C.



1974 was 0.028 ppm, but this was at a time when intake concentrations were also 0.028 ppm (Battelle-Columbus, William F. Clapp Laboratories, 1975). The maximum difference between intake and effluent soluble copper concentrations was 0.015 ppm. Oysters living in the effluent had copper concentrations of up to 640 ppm and were generally three to four times higher in concentration than control oysters (Battelle-Columbus, William F. Clapp Laboratories, 1975). The level proposed by EPA, 0.01 of the 96-hr  $LC_{50}$  value for the most sensitive species (Environmental Protection Agency, 1973), would require the plant effluent levels to be at background (0.003 ppm), which appears to be the case for power plants at least 50% of the time (Battelle-Columbus, William F. Clapp Laboratories, 1975).

The recommended free residual chlorine concentration of 0.71 mg/liter or 0.1 of the 96-hr  $LC_{50}$  value for the most important sensitive species (Environmental Protection Agency, 1973) is below the  $LC_{50}$  concentration for *Pandalus danae*. The normal chlorine demand in coastal waters and an aquaculture situation would generally keep the concentration below the EPA recommended levels. Our growth data are confused and will have to be tested again, but one group of shrimp did grow well at 0.05 mg/liter.

On the basis of the preceding results, the adult stage of the coon stripe shrimp, *Pandalus danae*, is a good candidate for aquaculture using the waste heat from a power plant as a growing medium. Further studies are needed to determine how easily this species can be cultured and if there are any problems with the organisms concentrating effluent contaminants, such as copper or chlorination by-products.

## CONCLUSIONS

1. The CTM values for coon stripe shrimp increase with an increase in shrimp size and rate at which the temperature is elevated.

2. Coon stripe shrimp are more resistant to chlorine when acclimated and exposed at 8 to 10°C than (a) when acclimated at 8°C and exposed at 15 or 20°C or v. when (b) acclimated and exposed at 15°C which is near their optimum short-term growth temperature (16°C).

3. The optimal growing temperature for (1 to 7 g) coon stripe shrimp for periods up to 1 month is 16°C.

4. Copper at a concentration of 0.041 mg/liter effectively retards the growth of (1 to 2 g) coon stripe shrimp at 16°C over a 1-month period.

5. Chlorine at a concentration of 0.18 mg/liter is lethal to (1 to 2 g) coon stripe shrimp at 16°C and reduced their growth at 0.08 mg/liter over a 1-month period.

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# Effects of Power-Plant Entrainment Temperatures on Oxygen Consumption Patterns of Amphipods and Grass Shrimp

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## ABSTRACT

Oxygen-consumption patterns of the amphipod, *Gammarus*, and the grass shrimp, *Palaemonetes*, were used to evaluate the potential effects of exposure to time-temperature changes simulating power-plant condenser entrainment. Base-line metabolism was determined at 15-min intervals for 90 min before thermal exposure. Exposure consisted in (1) a rapid 5°C thermal increase, (2) a 4-min exposure at the elevated temperature, and (3) a 15-min decay back to ambient temperatures. Oxygen-consumption measurements were then continued at 15-min intervals for 90 min to evaluate the organism's response to the thermal exposure. Oxygen-consumption patterns of *Gammarus* were studied at acclimation temperatures of 5, 15, and 25°C and of *Palaemonetes* at 5, 15, 25, and 30°C.

Oxygen consumption ( $Q_{O_2}$ ) of *Gammarus* at all acclimation temperatures was significantly higher than pre-exposure levels for 30 min following thermal exposure but returned to initial levels within 45 min. No changes occurred in  $Q_{O_2}$  patterns of *Palaemonetes* at any acclimation temperature. Although differences in species response patterns occurred, this study shows that both species should be able to tolerate the time-temperature exposure used in the study during entrainment through a once-through low- $\Delta T$  steam electric generating plant.

Two major areas of potential thermal impact to aquatic organisms have been identified for steam electric generating stations with once-through cooling systems. These include exposure of organisms to abrupt temperature increases from intake or pumped entrainment through plants and exposure to increased temperatures from heated effluents discharged from plants into the receiving body of water. Most studies to date have dealt with the effects of heated effluents on aquatic life (e.g., see Raney, Menzel, and Weller, 1974). In the studies that have considered the

thermal effects of entrainment through a plant, most have addressed relatively high temperature increases (8 to 12°C), which are typically produced by present-day plants (Environmental Protection Agency, 1974).

This study was initiated to quantify possible thermal stress that may occur to organisms entrained through low- $\Delta T$  (5°C maximum) steam electric generating stations with once-through cooling systems terminating in high-velocity submerged discharges. Because plants of this design generally use large volumes of water to obtain low temperature increases across the condensers, many are being built on estuarine and marine systems. The estuarine amphipod, *Gammarus*, and grass shrimp, *Palaemonetes*, were chosen for study as representative entrainable macrocrustaceans because of their wide geographical distribution and importance as food-chain organisms for fish.

Preliminary studies revealed that the time-temperature exposure used in this study to simulate power-plant condenser entrainment did not cause any immediate or latent mortality (up to 24 hr after exposure) of either species at the seasonal temperatures studied. Since significant mortality did not occur, the question of possible thermal stress to the organisms was addressed. The effects of sudden rises in temperature with relatively fast decays back to ambient are difficult to assess. However, it is well-established that, depending on the magnitude and duration of a temperature change, subtle responses of poikilothermic animals to a modified thermal environment can occur at the whole-animal, tissue, and cellular level. Since metabolic rate reflects the total energetics of an organism at all levels of organization, changes in whole-animal oxygen-consumption patterns were used to evaluate the possible effects of thermal stress.

Burton, Margrey, and Richardson: Academy of Natural Sciences of Philadelphia, Benedict Estuarine Research Laboratory, Benedict, Maryland. ©

## MATERIALS AND METHODS

### Collection and Maintenance of Test Animals

Gammarid amphipods, *Gammarus* sp., were collected in the Patuxent River near Benedict, Md., with a 390- $\mu$ m-mesh plankton net. *Gammarus* sp. looked similar to *G. daiberi*; however, since there appeared to be an overlap of morphological characteristics with *G. tigrinus* (Bousfield, 1969; 1973), no identification beyond *Gammarus* sp. was attempted. Palaemonid shrimps, *Palaemonetes* spp. (primarily *P. pugio* but also a few *P. vulgaris* and *P. intermedius*) were collected in the Patuxent River near Benedict, Md., and the Calvert Cliffs area of Chesapeake Bay in Calvert County, Maryland, with 0.6-cm-mesh beach seines and plankton bottom trawls.

Adult *Gammarus* were collected at seasonal temperatures approximating the experimental temperatures of 5, 15, and 25°C. *Palaemonetes* were collected and studied at 5, 15, 25, and 30°C. When seasonal temperatures differed slightly from the experimental temperatures, the animals were acclimated by changing the water temperature 1°C per day from ambient to the desired experimental temperature. Once the desired temperature was reached, the animals were held at that temperature ( $\pm 1^\circ\text{C}$ ) for a minimum of 2 weeks before any experimentation began. Each genus, both males and females, was maintained in the laboratory in separate aquaria supplied with a continuous flow of well-aerated Patuxent River water with a basic quality shown in Table 1. A cyclic photoperiod (incandescent light) of 12L : 12D was maintained for animals at 5 and 15°C and 16L : 8D for animals at 25 and 30°C during acclimation and experimental study. A 45-min transition between light periods was used rather than an abrupt on and off light transition. The following served as substrate and food for *Gammarus* and *Palaemonetes* during acclimation in the laboratory: detritus (which consisted of sediment, decaying leaves, and small sticks taken from the Patuxent River),

supplements of finely chopped commercial shrimp added daily, and material introduced via the continuous flow of river water through the holding tanks. The animals were not fed during an experiment.

### Experimental Procedures

Both species were exposed to the hypothetical time-temperature model shown in Fig. 1. All experimental animals were exposed to a rapid 5°C increase above ambient (simulation of the temperature rise across a condenser), held for 4 min at the elevated temperature (simulation of plant passage; time  $T_1$ , Fig. 1), and exposed

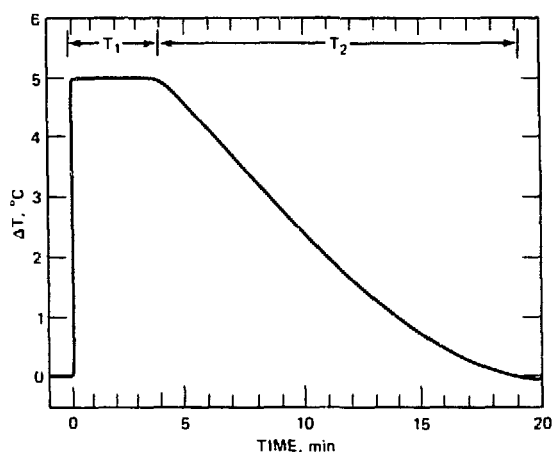


FIG. 1 Time-temperature model to which *Gammarus* and *Palaemonetes* were exposed (see text for details).

to temperature decay back to ambient over a 15-min period (simulation of decay down the center line of a high-velocity discharge plume; time  $T_2$ ). In this study the organisms were exposed to the time-temperature history over a 20-min period; however, under normal plant operating conditions, the temperature decay of the plume at 20 min would be between 0.5 and 1.0°C above ambient (Pritchard and Carter, 1965; Shirazi and Davis, 1972). Several hours would be required for the excess heat to dissipate to ambient temperatures. Because of the infinite number of decay patterns of the plume in the mixing zone after 20 min and the problem of trying to pick a representative decay pattern, the last 0.5 to 1.0°C above ambient was not addressed in this study.

Oxygen consumption was measured in a 20-channel differential respirometer that was modified for oxygen-consumption studies of small aquatic animals exposed to fluctuating time-temperature exposures (Richardson, Burton, and Margrey, 1975). The modified respirometer was used to determine preexposure base-line oxygen consumption, postexposure oxygen consumption, and to simulate the time-temperature exposure. Potential handling stress was avoided by keeping the animals in the reaction vessels throughout the entire experimental operation.

TABLE 1 Basic Water Quality Used for Acclimation and Experimental Runs

| Parameter                    | Mean                                     | Range     |
|------------------------------|------------------------------------------|-----------|
| pH*                          | 7.8                                      | 7.3–8.6   |
| Salinity, † ‰                | 10.0                                     | 6.2–13.6  |
| Chloride, * mg/liter         | 5535                                     | 3400–7500 |
| Sodium, ‡ mg/liter           | 1872                                     | 740–3200  |
| Sulfate, * mg/liter          | 510                                      | 255–1052  |
| Magnesium, * mg/liter        | 385                                      | 241–525   |
| Calcium, * mg/liter          | 119                                      | 83–160    |
| Potassium, ‡ mg/liter        | 81                                       | 32–124    |
| Dissolved oxygen, * mg/liter | >6.0 at all times in acclimation aquaria |           |

\* Analyzed by procedures in American Public Health Association (1971).

† Analyzed by procedures in Strickland and Parsons (1972).

‡ Analyzed on a Perkin-Elmer model 303 atomic absorption spectrophotometer, Perkin-Elmer Corporation, Norwalk, Conn.

One intermolt adult *Palaemonetes* or four to eight intermolt adult *Gammarus* were studied in 15-ml reaction vessels (no center wells) with 7 ml of well-aerated Patuxent River water that had been filtered through a 0.45- $\mu$ m Millipore filter. A 0.5-ml solution of 10% potassium hydroxide was placed in the sidearm of each vessel to absorb  $\text{CO}_2$ . Sterile gauze was placed in the bottom of each reaction vessel to reduce animal activity (Hagerman, 1970) and to minimize sensory deprivation (Florey and Kriebel, 1974). A shaking rate of approximately 80 oscillations per minute was used for oxygen equilibration.

All animals were held in the reaction vessels under preexposure experimental conditions for 2 hr, with dry air pumped through the vessels to allow the animals time to recover from handling and to reach a stabilized, routine oxygen-consumption level. After the equilibration period, preexposure base-line metabolism was determined at 15-min intervals for 90 min before the experimental animals were subjected to the time-temperature exposure. Immediately after the exposure, measurements were resumed at 15-min intervals for 90 min to evaluate the organism's response to the thermal exposure. Three or four experimental runs with each species at each acclimation temperature were conducted. Eight experimental vessels and eight control vessels with animals were usually studied during each experiment. One thermobarometer and four correction flasks containing filtered Patuxent River water with no animals were always present in the system during each experimental run. At the end of each experiment, all animals were removed and dried to a constant weight at 90°C for 24 hr to convert the data to weight specific oxygen consumption [ $\text{QO}_2 = \mu\text{l O}_2 (\text{mg dry weight})^{-1} \text{hr}^{-1}$ ]. Each oxygen reading was corrected for any fluctuation detected by the four correction flasks and then further corrected to standard temperature and pressure before  $\text{QO}_2$  was calculated. In the case of *Gammarus*, the average dry weight of the four to eight animals was used in the  $\text{QO}_2$  calculation.

To minimize possible variations in oxygen consumption due to photoperiod (Newell, 1973), we began all experimental readings 4 hr after the simulated laboratory sunrise reached full daylight. To further reduce variations in oxygen consumption, we used adult organisms of each species in approximately the same size range at each acclimation temperature. Only nonovigerous females were used. When amplexing pairs of amphipods were observed during an experiment, the entire series of readings was thrown out.

### Statistical Treatments

The data were analyzed by multivariate nonparametric repeated measurement statistics taken from Morrison (1967). We tested the hypothesis that the theoretical mean oxygen-consumption values of the 12 readings per vessel were equal (6 preexposure and 6 postexposure readings). If the hypothesis was rejected, we tested between means or contrasts of means to determine where significant responses

occurred. Homogeneity of responses between control and experimental animals for each species at each acclimation temperature was tested for the preexposure values only by Morrison's (1967) test for parallel profiles. The effects of different acclimation temperatures on oxygen consumption were also evaluated by the test of parallel profiles on the preexposure pooled data of the control and experimental animals at each acclimation temperature when the test for homogeneity of responses showed that no significant difference existed between the control and experimental animals of a species at each acclimation temperature.

### RESULTS AND DISCUSSION

The possibility of thermal stress to organisms during pumped entrainment through once-through power plants has been suggested by several investigators (e.g., Clark and Brownell, 1973). Thermal stress is defined in this study as a stimulus that produces a significant deviation from normal changes in whole-animal oxygen-consumption ( $\text{QO}_2$ ) responses, not as a stimulus that necessarily causes a sublethal or prelethal effect as defined by Fry (1971). To adequately evaluate the potential impact of thermal stress to organisms, we should consider three questions: (1) Does a significant response occur? (2) If a significant response occurs, is the response detrimental to the organism? (3) If the response is not detrimental, how long does it take the organism to recover from the stress or to return to preexposure levels once the animal reaches ambient temperatures in the receiving body of water?

Oxygen consumption of *Gammarus* during the first 30 min following thermal exposure was significantly higher ( $P < 0.05$ ) than the mean  $\text{QO}_2$  during preexposure study in all acclimation groups (Table 2). It returned to initial preexposure levels within 45 min in all cases. No significant difference ( $P > 0.05$ ) was found when contrast of means between postexposure values at 15 and 30 min were tested for amphipods acclimated to 5 and 15°C. The mean postexposure  $\text{QO}_2$  at 15 min was significantly higher ( $P < 0.05$ ) than the 30-min mean in 25°C acclimated *Gammarus*. No significant changes ( $P > 0.05$ ) were found in  $\text{QO}_2$  patterns of control amphipods (Table 2) or of control and experimental *Palaemonetes* at any acclimation temperatures as a result of thermal exposure (Table 3).

For the following reasons we think that the  $\text{QO}_2$  changes in *Gammarus* represent normal physiological temperature-compensation responses, not thermal-stress responses. When a poikilothermic animal is exposed to an abrupt change in environmental temperatures, many aspects of its physiology are altered in a manner that often compensate for the temperature change (Prosser, 1973). Many organisms show an initial overshoot reaction in a physiological rate function (e.g.,  $\text{QO}_2$ ) when they are exposed to an abrupt temperature change (Bullock, 1955; Grainger, 1958). This response may last for seconds, minutes, or an hour or longer. The magnitude and possibly the duration of the overshoot has been shown to vary with the degree of temperature change (Grainger, 1958). If the

TABLE 2 Mean ( $\pm$ SD) Oxygen Consumption ( $QO_2$ ) of 5, 15, and 25°C Acclimated *Gammarus* Obtained at Each Preexposure and Postexposure Interval

|                             | $QO_2$ , $\mu$ l $O_2$ (mg dry weight) $^{-1}$ hr $^{-1}$ |                  |                      |                  |                      |                  |
|-----------------------------|-----------------------------------------------------------|------------------|----------------------|------------------|----------------------|------------------|
|                             | 5°C                                                       |                  | 15°C                 |                  | 25°C                 |                  |
|                             | Experimental animals                                      | Control animals  | Experimental animals | Control animals  | Experimental animals | Control animals  |
| Preexposure intervals, min  |                                                           |                  |                      |                  |                      |                  |
| 90                          | 0.3( $\pm$ 0.13)                                          | 0.4( $\pm$ 0.11) | 0.7( $\pm$ 0.19)     | 0.7( $\pm$ 0.14) | 1.7( $\pm$ 0.38)     | 1.5( $\pm$ 0.70) |
| 75                          | 0.3( $\pm$ 0.13)                                          | 0.3( $\pm$ 0.11) | 0.8( $\pm$ 0.15)     | 0.8( $\pm$ 0.16) | 1.6( $\pm$ 0.41)     | 1.8( $\pm$ 0.62) |
| 60                          | 0.3( $\pm$ 0.10)                                          | 0.4( $\pm$ 0.13) | 0.8( $\pm$ 0.13)     | 0.9( $\pm$ 0.15) | 1.7( $\pm$ 0.41)     | 1.5( $\pm$ 0.51) |
| 45                          | 0.4( $\pm$ 0.10)                                          | 0.4( $\pm$ 0.09) | 0.7( $\pm$ 0.17)     | 0.8( $\pm$ 0.14) | 1.7( $\pm$ 0.42)     | 1.8( $\pm$ 0.63) |
| 30                          | 0.4( $\pm$ 0.06)                                          | 0.4( $\pm$ 0.09) | 0.8( $\pm$ 0.15)     | 0.9( $\pm$ 0.14) | 1.4( $\pm$ 0.53)     | 1.5( $\pm$ 0.83) |
| 15                          | 0.3( $\pm$ 0.13)                                          | 0.4( $\pm$ 0.11) | 0.8( $\pm$ 0.16)     | 0.9( $\pm$ 0.23) | 1.8( $\pm$ 0.38)     | 1.8( $\pm$ 0.49) |
| Postexposure intervals, min |                                                           |                  |                      |                  |                      |                  |
| 15                          | 0.8( $\pm$ 0.16)                                          | 0.5( $\pm$ 0.14) | 1.4( $\pm$ 0.23)     | 1.0( $\pm$ 0.19) | 2.7( $\pm$ 0.90)     | 1.5( $\pm$ 0.49) |
| 30                          | 0.6( $\pm$ 0.10)                                          | 0.5( $\pm$ 0.12) | 1.1( $\pm$ 0.15)     | 1.0( $\pm$ 0.16) | 2.3( $\pm$ 0.63)     | 1.8( $\pm$ 0.56) |
| 45                          | 0.4( $\pm$ 0.11)                                          | 0.4( $\pm$ 0.08) | 0.9( $\pm$ 0.17)     | 0.8( $\pm$ 0.22) | 2.0( $\pm$ 0.47)     | 1.7( $\pm$ 0.44) |
| 60                          | 0.5( $\pm$ 0.09)                                          | 0.5( $\pm$ 0.12) | 0.8( $\pm$ 0.15)     | 0.8( $\pm$ 0.14) | 1.9( $\pm$ 0.58)     | 1.7( $\pm$ 0.46) |
| 75                          | 0.4( $\pm$ 0.11)                                          | 0.4( $\pm$ 0.10) | 0.8( $\pm$ 0.15)     | 0.8( $\pm$ 0.14) | 1.8( $\pm$ 0.35)     | 1.7( $\pm$ 0.39) |
| 90                          | 0.5( $\pm$ 0.09)                                          | 0.5( $\pm$ 0.13) | 0.8( $\pm$ 0.20)     | 0.8( $\pm$ 0.21) | 1.7( $\pm$ 0.43)     | 1.8( $\pm$ 0.34) |
| Sample size                 | 22                                                        | 20               | 31                   | 30               | 32                   | 24               |

TABLE 3 Mean ( $\pm$ SD) Oxygen Consumption ( $QO_2$ ) of 5, 15, 25, and 30°C Acclimated *Palaemonetes* Obtained at Each Preexposure and Postexposure Interval

|                             | $QO_2$ , $\mu$ l $O_2$ (mg dry weight) $^{-1}$ hr $^{-1}$ |                  |                      |                  |                      |                  |                      |                  |
|-----------------------------|-----------------------------------------------------------|------------------|----------------------|------------------|----------------------|------------------|----------------------|------------------|
|                             | 5°C                                                       |                  | 15°C                 |                  | 25°C                 |                  | 30°C                 |                  |
|                             | Experimental animals                                      | Control animals  | Experimental animals | Control animals  | Experimental animals | Control animals  | Experimental animals | Control animals  |
| Preexposure intervals, min  |                                                           |                  |                      |                  |                      |                  |                      |                  |
| 90                          | 0.5( $\pm$ 0.23)                                          | 0.5( $\pm$ 0.22) | 0.9( $\pm$ 0.28)     | 0.9( $\pm$ 0.27) | 1.9( $\pm$ 1.04)     | 1.5( $\pm$ 1.10) | 2.7( $\pm$ 1.09)     | 2.5( $\pm$ 0.90) |
| 75                          | 0.5( $\pm$ 0.21)                                          | 0.6( $\pm$ 0.20) | 1.1( $\pm$ 0.26)     | 1.0( $\pm$ 0.33) | 2.3( $\pm$ 0.88)     | 2.4( $\pm$ 1.12) | 2.9( $\pm$ 1.01)     | 2.6( $\pm$ 0.79) |
| 60                          | 0.5( $\pm$ 0.18)                                          | 0.5( $\pm$ 0.17) | 1.1( $\pm$ 0.28)     | 1.1( $\pm$ 0.25) | 2.4( $\pm$ 0.78)     | 2.4( $\pm$ 1.04) | 2.8( $\pm$ 0.94)     | 2.5( $\pm$ 0.71) |
| 45                          | 0.4( $\pm$ 0.22)                                          | 0.5( $\pm$ 0.21) | 1.1( $\pm$ 0.35)     | 1.1( $\pm$ 0.25) | 2.6( $\pm$ 0.81)     | 2.4( $\pm$ 1.11) | 2.7( $\pm$ 0.82)     | 2.5( $\pm$ 0.70) |
| 30                          | 0.5( $\pm$ 0.19)                                          | 0.5( $\pm$ 0.20) | 1.1( $\pm$ 0.29)     | 1.1( $\pm$ 0.25) | 2.6( $\pm$ 0.94)     | 2.6( $\pm$ 1.02) | 2.5( $\pm$ 0.77)     | 2.5( $\pm$ 0.78) |
| 15                          | 0.4( $\pm$ 0.15)                                          | 0.4( $\pm$ 0.15) | 1.1( $\pm$ 0.29)     | 1.1( $\pm$ 0.34) | 2.7( $\pm$ 0.87)     | 2.6( $\pm$ 1.17) | 2.6( $\pm$ 0.85)     | 2.3( $\pm$ 0.89) |
| Postexposure intervals, min |                                                           |                  |                      |                  |                      |                  |                      |                  |
| 15                          | 0.4( $\pm$ 0.34)                                          | 0.4( $\pm$ 0.23) | 0.9( $\pm$ 0.28)     | 0.9( $\pm$ 0.23) | 2.9( $\pm$ 1.25)     | 2.0( $\pm$ 1.69) | 2.7( $\pm$ 1.11)     | 2.1( $\pm$ 0.88) |
| 30                          | 0.5( $\pm$ 0.23)                                          | 0.5( $\pm$ 0.17) | 1.1( $\pm$ 0.28)     | 1.0( $\pm$ 0.21) | 3.0( $\pm$ 1.07)     | 2.5( $\pm$ 1.11) | 2.8( $\pm$ 0.86)     | 2.4( $\pm$ 0.82) |
| 45                          | 0.4( $\pm$ 0.18)                                          | 0.4( $\pm$ 0.18) | 1.1( $\pm$ 0.24)     | 1.0( $\pm$ 0.23) | 2.2( $\pm$ 0.74)     | 2.4( $\pm$ 1.12) | 2.7( $\pm$ 0.82)     | 2.4( $\pm$ 0.76) |
| 60                          | 0.5( $\pm$ 0.17)                                          | 0.5( $\pm$ 0.19) | 1.0( $\pm$ 0.19)     | 0.9( $\pm$ 0.26) | 2.7( $\pm$ 0.94)     | 2.6( $\pm$ 1.17) | 2.7( $\pm$ 0.77)     | 2.4( $\pm$ 0.79) |
| 75                          | 0.4( $\pm$ 0.14)                                          | 0.4( $\pm$ 0.19) | 0.9( $\pm$ 0.26)     | 0.9( $\pm$ 0.24) | 2.4( $\pm$ 0.65)     | 2.6( $\pm$ 1.28) | 2.7( $\pm$ 0.89)     | 2.4( $\pm$ 0.72) |
| 90                          | 0.5( $\pm$ 0.15)                                          | 0.5( $\pm$ 0.18) | 0.9( $\pm$ 0.26)     | 0.9( $\pm$ 0.23) | 2.1( $\pm$ 1.00)     | 2.0( $\pm$ 1.16) | 2.6( $\pm$ 0.84)     | 2.2( $\pm$ 0.92) |
| Sample size                 | 31                                                        | 31               | 32                   | 30               | 21                   | 21               | 21                   | 27               |

change in temperature is small, the initial overshoot reaction may not be seen. The overshoot effect is not confined to any particular acclimation temperature; it may occur at various acclimation temperatures (Grainger, 1958). After the overshoot there is a period in which the rate function stabilizes for several minutes or hours when the animal is held at the new temperature (Prosser, 1958). The

stabilized rate depends on the animal's acclimation state, and, if the animal is returned to the original temperature during this period, the rate returns to the initial, or preexposure, level. According to Prosser (1967), the initial overshoot reaction and the period of stabilized rate are homeostatic in that they provide short-term alterations in a poikilothermic animal which permit it to function at a

different temperature. Grainger (1958) has demonstrated a similar overshoot response in the crustaceans *Artemia salina* and *Neomysis integer* and cited several earlier papers that reported the same response in other invertebrates.

Since no significant changes occurred in  $Q_{O_2}$  patterns of *Palaemonetes* at any acclimation temperature as a result of thermal exposure, it is apparent that no thermal stress occurred. This suggests that the magnitude of the thermal exposure was too small and/or the duration too short to cause an overshoot response (Grainger, 1958). McFarland and Pickens (1965) have shown that *P. vulgaris* possess a high degree of temperature independence for poikilotherms exposed to acute temperature changes, which may explain the no-overshoot response. It may also be possible that *Palaemonetes* fall into a group of poikilotherms that exhibit instantaneous or immediate temperature compensation (Hochachka and Somero, 1973; Hazel and Prosser, 1974). Some organisms, notably intertidal invertebrates that encounter large short-term fluctuations in temperature, exhibit temperature-independent rates of metabolism over a fairly large portion of the range of temperatures that they experience in their natural environments (Newell, 1969; Newell and Northcroft, 1967; Newell and Pye, 1971).

A few studies have dealt with the effects of rapidly increased temperatures on oxygen consumption of amphipods (e.g., Krog, 1954; Armitage, 1962; Ivanova, 1972); however, the animals were held at the increased temperatures for variable periods of time, and in no case were the animals returned to ambient temperatures. Thus it is difficult to relate those studies to the present study insofar as relating the short-term alterations in  $Q_{O_2}$  to those which may occur from thermal changes induced by entrainment. However, in all the studies that, unfortunately from a comparative aspect, were conducted on cold acclimated species, the investigators found that a 5 to 10°C increase above ambient caused significant increases in  $Q_{O_2}$ . From those studies and the present study, it appears that a significant increase in  $Q_{O_2}$  as a result of exposure to rapid increases in temperature may be a general response in certain groups of amphipods.

Significant differences were found between seasonal oxygen-consumption values of each species with the exception of *Palaemonetes* at 25 and 30°C, where no difference was found. Oxygen consumption increased with increasing acclimation temperatures for both species. The same trend has been demonstrated for several gammarid amphipods (e.g., Krog, 1954; Halcrow and Boyd, 1967; Dorgelo, 1973), palaemonid shrimp (e.g., McFarland and Pickens, 1965), and closely related penaeid shrimp (e.g., Venkataramiah, Lakshmi, and Gunter, 1974). The effect of the time-temperature exposure in this study on  $Q_{O_2}$  responses of *Gammarus* and *Palaemonetes* was independent of acclimation temperature, i.e., the response was the same at all acclimation temperatures. Although the responses were different between the species, no thermal stress occurred to the organisms. In contrast to the findings of several investigators studying seasonal survival of zooplankton

entrained at several power plants with relatively high temperature increases (e.g., Davies and Jensen, 1974), this study suggests that a detrimental impact on *Gammarus* and *Palaemonetes* exposed to low temperature increases will not occur as a result of seasonal temperature changes.

It is not too surprising that the time-temperature regime used in this study did not stress *Gammarus* and *Palaemonetes* when we consider the thermal tolerance of amphipods and grass shrimp. For example, Mihursky and Kennedy (1967) and Kennedy et al. (1969) found that the amphipods *G. fasciatus* and *Monoculodes* sp. acclimated from 1 to 3°C had 24-hr  $LT_{50}$  values of 27.9 and 23°C, respectively. *Gammarus fasciatus*, *Monoculodes* sp., and *G. mucronatus* acclimated to 15°C had 24-hr  $LT_{50}$  values of 32, 28, and 30.2°C, respectively. Krog (1954) demonstrated that *G. limnaeus* acclimated from 1 to 3°C and 15 to 22°C could survive shock exposures up to 26°C and 30 to 32°C, respectively, for 1 hr with no mortalities. *Palaemonetes pugio* acclimated to 15, 20, 25, and 30°C were shown by Kennedy et al. (1969) to have 24-hr  $LT_{50}$  values of 32, 35, 35.1, and 36.2°C, respectively. Two species of entrainable juvenile penaeid shrimp (*Penaeus aztecus* and *P. setiferus*) acclimated to 29°C had critical thermal maxima (1°C increase per minute) of 10 to 12°C above ambient (Copeland, Laney, and Pendleton, 1974).

The most relevant thermal-tolerance studies that have dealt with time-temperature exposures similar to the present study are those of Ginn, Waller, and Lauer (1974), Lauer et al. (1974), and Ginn, Waller, and Lauer (1976). These investigators found that the estuarine amphipods *G. tigrinus* and *G. daiberi* acclimated to 25°C could withstand a 10°C increase above ambient for periods ranging from 5 to 60 min with no mortalities occurring in 24 hr. No mortality occurred up to 10 days after 5- to 60-min exposures of 8.3°C above an ambient temperature of 25°C. When amphipods were exposed to temperature increases up to 16.7°C above an ambient temperature of 11.7°C, there were no significant reductions in survival immediately or after 10 days for exposure times up to 180 min. The possibility of long-term impact on the reproduction of *Gammarus* exposed to time-temperature regimes produced by power-plant entrainment was also addressed by Ginn, Waller, Lauer (1976). They demonstrated that exposure of mature *Gammarus* to an 8.3°C increase above ambient temperatures of 26°C for periods up to 30 min did not affect reproductive mating activities or the release of young by ovigerous females.

Although this study has shown that *Gammarus* and *Palaemonetes* are not significantly affected by a low- $\Delta T$  short-duration time-temperature exposure simulating entrainment through a steam electric generating station with a once-through cooling system, we must not overlook the possible impact from pumped entrainment through a commercially operating power plant with similar time-temperature regimes. Such factors as mechanical and pressure changes, exposure to biocides and other cleaning chemicals, and possible reductions in dissolved oxygen are

also potential sources of impact associated with pumped entrainment (Coutant, 1974). The effects of pumped entrainment on *Gammarus* by Consolidated Edison Company of New York's Indian Point Plant was shown by Ginn, Waller, and Lauer (1974) to damage or kill 1.5 to 9.9% of the organisms when the operating  $\Delta T$  was 7.1 to 8.3°C and ambient temperatures were 24.1 to 26.0°C. Entrained *Gammarus* experienced increased mortalities during periods of condenser chlorination. Laboratory temperature bioassays cited by Ginn, Waller, and Lauer (1974) showed that *Gammarus* sp. should be able to tolerate all projected time-temperature exposure combinations encountered during entrainment through the condenser system.

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# Growth of Salmonid Fishes from Heated and Unheated Areas of Lake Michigan-Measured by RNA-DNA Ratios

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## ABSTRACT

Relative growth-rate comparisons were made between tagged fish "resident" in a thermal plume and fish collected from two ambient temperature areas (control). Plume fish were tagged, released, and subsequently recaptured in the thermal discharge area of the Point Beach Nuclear Plant (near Two Rivers, Wis.). Total tag days indicated minimum "residence" time, and temperature-sensitive tags gave estimates of time spent at discharge temperatures. Growth-rate estimates were based on RNA-DNA ratios in epaxial muscle samples taken from brown and rainbow trout and chinook salmon. Mean RNA-DNA ratios of plume rainbow trout and chinook salmon were not significantly different from mean ratios of combined control groups for each species. The mean ratio of plume brown trout was significantly higher than that of combined control fish. Significant differences between mean ratios of control groups for each species suggest considerable natural variability in growth rates among individuals of a population. Although other workers have suggested that residence at elevated temperatures could affect fish growth, our data show no correlation between estimated residence times and RNA-DNA ratios or condition factors in plume fish. Probable reasons for a lack of difference in growth for some species and apparent differences in other species are discussed in terms of results of our fish behavioral research at thermal-discharge sites on Lake Michigan.

Most fishes are capable of sustained or indeterminate growth throughout their life span. Growth rates of fish are primarily dependent on food supply, body size (age), and water temperature. Water temperature can override other limiting factors since temperature affects metabolism and food consumption. Such density-dependent factors as competition and other social interactions can also affect food consumption and growth. These growth-limiting factors are extremely important in terms of the effects of waste-heat discharges on growth of fish that "reside" in heated waters.

A few studies have attempted to document in situ growth of fishes which reside in thermal discharge areas. Bennett (1972) reported length-weight relationships and condition factors of centrarchids collected from warm and cool areas of Par Pond in South Carolina and concluded that the heated environment was beneficial for some species and life stages but detrimental to others. Messengill (1973) observed reduced condition factors and altered feeding habits in brown bullhead that overwintered in the heated discharge area of the Connecticut Yankee Atomic Power Plant at Haddam, Conn. Similar studies were made at various power-plant discharge sites on Lake Michigan to compare size and condition factors of fish collected from heated and unheated waters. In most cases, results were inconclusive owing to very small sample sizes or an obvious lack of thermal history on sampled fish. Also, the techniques used to indicate growth (length-weight relationships and condition factor) are insensitive to short-term changes in metabolic rates or feeding responses. Such considerations are extremely important in the case of migratory fishes since we cannot assume that such fishes remain within the heated waters for extended periods of time. The majority of the Lake Michigan fish populations are migratory, and recent mark and recapture studies by Argonne National Laboratory (ANL) have shown that most salmonids do not remain within the influence of one thermal discharge for long periods of time (Romberg et al., 1975).

Research in protein synthesis and growth of animals has shown that ribonucleic acid (RNA) increases in rapidly growing cells as a result of its role in protein synthesis (Sutcliffe, 1965; Leslie, 1955; Davidson and Leslie, 1950). The potential value of RNA as an indicator of growth is enhanced by the fact that cellular RNA concentrations change almost immediately in response to changes in

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feeding regime (Bulow, 1970). On the other hand, concentrations of deoxyribonucleic acid (DNA) are known to remain relatively constant despite changes in growth rate, feeding regime, or cell number (Davidson and Leslie, 1950; Hotchkiss, 1955). These considerations led Bulow (1970) and Haines (1973) to test the RNA-DNA ratio as an indicator of recent and long-term growth. Both these studies demonstrated that the RNA-DNA ratio is a sensitive measure of growth and can be used to evaluate relative growth rates of fishes under various feeding regimes or under varying environmental conditions. The potential applicability of this technique to studies of growth effects on thermal-discharge-resident fish is apparent.

Our experiments with temperature-sensitive fish tags provided preliminary estimates of discharge residence times and temperature exposures of Lake Michigan salmonids and gave us the opportunity to apply the RNA-DNA technique to compare relative growth in fish with known recent plume exposures and fish collected from unheated areas of the lake.

## STUDY AREA

Lake Michigan is the sixth largest lake on earth and constitutes an invaluable freshwater resource. As a result of state and federal management activities in recent years, the fishery in Lake Michigan has been transformed from a state of total collapse to an economically profitable salmonid fishery. Approximately 20 million pounds of trout and salmon are harvested annually by sport fishermen in Lake Michigan and its major tributaries.

At present there are 25 electric generating stations in operation and under construction on the shores of Lake Michigan, representing a total generating capacity of about 16,000 Mw(e) [18 fossil-fuel stations  $\approx$  8570 Mw(e) and 7 nuclear stations  $\approx$  7430 Mw(e)]. Twenty-three of these power stations use once-through cooling techniques and discharge heated water to the inshore zone of Lake Michigan. These thermal discharges are known to attract some migratory fishes and definitely have an effect on the spatial and temperature distribution patterns of fish, particularly during spawning seasons (Spigarelli, 1975). Significant sport fisheries have developed at many of these warm-water discharges, and fishermen often experience excellent catches of trout and salmon in the heated waters (Spigarelli and Thommes, 1976).

The site of this study was the Point Beach Nuclear Plant on the western shore of the lake, near Two Rivers, Wis. This station consists of two pressurized-water reactors of approximately 500-Mw(e) capacity each. Cooling water is drawn directly from Lake Michigan (offshore crib) at a maximum flow rate of 45 m<sup>3</sup>/sec (690,000 gpm) and is returned to the lake through two surface discharge flumes extending 46 m (150 ft) offshore. Discharge velocities at the ends of the flumes range between 0.33 m/sec (1.1 fps) and 0.93 m/sec (3.0 fps) (Frigo and Frye, 1972). The maximum temperature rise ( $\Delta t$ ) in the cooling water is

10.5°C (19°F) during the majority of the year when there is no recirculation of cooling water.

## MATERIALS AND METHODS

In order to define the migratory and plume-residence behavior of salmonids, ANL initiated a mark and recapture study at Point Beach in 1972. Results of this study are reported by Romberg et al., 1975. Since then we have developed a temperature-sensitive fish tag that integrates temperature exposure and allows us to estimate the maximum time spent at any temperature for a recaptured fish. These tags contain irradiated thermoluminescent materials (CaSO<sub>4</sub>) which emit light at a rate proportional to temperature (Romberg and Prepejchal, 1974).

Between September and October 1973, we collected brown trout (*Salmo trutta*), rainbow trout (*Salmo gairdneri*), and chinook salmon (*Oncorhynchus tshawytscha*) from the discharge area, tagged them with commercial dart tags and temperature-sensitive tags, and returned them to the discharge area. Later some of these fish were recaptured at Point Beach by fishermen or by ANL field crews, and RNA-DNA ratios were calculated for 49 recaptured fish. We anticipated that the fish recaptured in the Point Beach discharge area would have been exposed to heated waters for varying periods of time and that the time-at-temperature histories would be very useful in a study of the effects of elevated temperatures on the growth of these fishes. Consequently we arranged for control fish to be collected in late October 1973 by the Wisconsin Department of Natural Resources (DNR) from unheated areas of the lake. Control brown and rainbow trout were collected from two areas north of Point Beach, Marinette (C<sub>1</sub>) and Bailey's Harbor (C<sub>2</sub>). Control chinook salmon were collected from Strawberry Creek on two separate occasions, Nov. 1, 1973 (1), and Nov. 5, 1973 (2). Although we have no temperature histories on control fish, we are confident that these fish were recent residents of ambient-temperature waters since there are no heated discharges in these control areas.

Because our temperature-sensitive tags only relate to the period between tagging and recapture (no previous temperature history), we used the RNA-DNA technique to indicate recent growths. The RNA and DNA analyses were performed on epaxial muscle samples taken from tagged fish that were recaptured in the plume area and from control fish. Tissue samples were immediately frozen and returned to the laboratory for future analyses. The procedure used to determine RNA and DNA concentrations (Bulow, 1970) involves an extraction of the nucleic acids from dry fat-free muscle samples and colorimetric determinations of phosphorus by means of orcinol (RNA) and diphenylamine (DNA) reactions. Concentrations of RNA and DNA were determined by comparison with standard solutions of RNA and DNA phosphorus. Four replicates were analyzed per fish to determine analytical variability and to reduce the analytical error (mean values were used for each fish).

We calculated condition factors [ $K = \text{weight (g)} \times 10^5 / \text{total length (cm)}^3$ ] for each fish in order to compare the results of two independent indicators of growth. We expected that condition factors would be less sensitive indicators of recent growth (during the time fish were tagged) than RNA-DNA ratios. On the other hand, condition factors could reflect long-term growth effects resulting from extended plume residence before tagging.

Statistical analyses of the data involved the Mann-Whitney U-test for comparisons of group means ( $\alpha = 0.05$ ) and the Spearman rank-correlation test for linear relationships between parameters.

## RESULTS AND DISCUSSION

The mean RNA-DNA ratios of plume resident rainbow trout and chinook salmon were not statistically different from mean ratios of combined controls (1 and 2). The mean ratio of plume-resident brown trout was significantly higher than the combined controls ratio. For each species the ratios of the two controls were statistically different, emphasizing the difficulty in obtaining representative control or reference samples.

Table 1 shows the mean weights, condition factors, and RNA-DNA ratios for each group of fish. A summary of the statistical comparisons between means is given in Table 2. We intentionally collected fish from two control areas (two dates for chinook salmon) to gain a more meaningful comparison with plume fish. Although the differences between control groups seem to confound the interpretations, we initially assumed that the two controls would be lumped for each species. These data show that the RNA-DNA ratios of plume-resident fish were within the range of values observed in populations inhabiting natural waters. The differences between control groups are consis-

tent in that control area 1 (Marinette, Wis.) ratios are low for both brown and rainbow trout, while control area 2 (Bailey's Harbor) ratios are relatively high and not statistically different from plume ratios. Chinook salmon controls were collected from the same area during spawning runs. These samples obviously represent different populations in that weights, condition factors, and RNA-DNA ratios were significantly different between the control groups. Thus RNA-DNA ratios of plume brown and rainbow trout were relatively high but not statistically different from control group 2. Plume chinook salmon had relatively low ratios, but these fish were apparently of a small size class and resembled control group 2.

In order to compare the RNA-DNA results with another indicator of growth, we calculated condition factors for each group of fish (Table 1). The statistical comparisons (Table 2) show that, in general, plume fish condition factors were not statistically different from the combined control groups for each species. Although the relative values of condition-factor means in some comparisons would suggest that plume fish were in relatively poor condition, in no case did the two techniques show opposite statistical differences. The obvious conclusions from both indicators of growth are that (1) plume residence does not cause extensive changes in growth during September and October and (2) plume fish did not remain at elevated temperatures for sufficiently long periods of time to cause changes in growth.

Table 3 shows the results from temperature-sensitive tags that were placed on plume fish to estimate the percentage of time spent at discharge temperature. Although the majority of fish were recaptured within about 20 days, it is apparent that none of these fish resided at discharge temperatures for more than 50% of the time; in fact, the majority spent less than 10% of the time at maximum temperature. The within-species and between-species variability in Table 3 (hours and percent time at discharge temperature) is a reflection of variable tag times (days tagged) and differences in individual fish responses to temperature and current. Figure 1 shows the total tag times and percent times at discharge temperatures for each fish bearing temperature-sensitive tags.

On the basis of these data, it appears that brown trout spent the most time at elevated temperature, followed by rainbow trout and chinook salmon in decreasing order. The fact that these fishes did not reside permanently at discharge temperatures is not surprising when we look at the intake and discharge temperature records over this period (Fig. 2). Discharge temperature varied between 16.5 and 26.5°C and exceeded 22°C approximately 70% of the time. The upper avoidance temperature of brown trout has been reported to be 20°C (Alabaster and Downing, 1966); that of fingerling rainbow trout, 22°C (McCauley and Pond, 1971). No upper avoidance temperature data are available for chinook salmon. Thus it would seem that these fishes avoided the high discharge temperatures and either (1) resided at intermediate temperatures that more nearly

TABLE 1 Mean Weights, Condition Factors, and RNA/DNA Ratios of Fish from Heated Waters (Point Beach Nuclear Plant Plume) and Unheated Waters of Lake Michigan; All Values  $\pm$  Standard Error

| Sample         | N* | Weight, kg       | Condition factor (K) | RNA-DNA ratio  |
|----------------|----|------------------|----------------------|----------------|
| Brown trout    | 40 |                  |                      |                |
| Plume          | 20 | 2.95 $\pm$ 0.25  | 1.41 $\pm$ 0.06      | 15.4 $\pm$ 0.6 |
| Control        | 20 | 2.77 $\pm$ 0.35  | 1.50 $\pm$ 0.04      | 11.1 $\pm$ 0.8 |
| 1              | 9  | 2.18 $\pm$ 0.35  | 1.38 $\pm$ 0.07      | 7.5 $\pm$ 0.4  |
| 2              | 11 | 3.31 $\pm$ 0.48  | 1.59 $\pm$ 0.03      | 14.1 $\pm$ 0.5 |
| Rainbow trout  | 32 |                  |                      |                |
| Plume          | 16 | 2.36 $\pm$ 0.30  | 1.29 $\pm$ 0.06      | 17.0 $\pm$ 1.1 |
| Control        | 16 | 2.72 $\pm$ 0.20  | 1.38 $\pm$ 0.06      | 14.9 $\pm$ 1.5 |
| 1              | 10 | 2.95 $\pm$ 0.27  | 1.44 $\pm$ 0.06      | 11.2 $\pm$ 0.5 |
| 2              | 6  | 2.40 $\pm$ 0.27  | 1.28 $\pm$ 0.11      | 21.2 $\pm$ 2.2 |
| Chinook salmon | 28 |                  |                      |                |
| Plume          | 13 | 7.98 $\pm$ 0.63  | 1.11 $\pm$ 0.04      | 11.8 $\pm$ 1.1 |
| Control        | 15 | 8.98 $\pm$ 0.63  | 1.00 $\pm$ 0.04      | 14.3 $\pm$ 0.9 |
| 1              | 8  | 10.20 $\pm$ 0.70 | 1.07 $\pm$ 0.04      | 16.8 $\pm$ 1.0 |
| 2              | 7  | 7.30 $\pm$ 0.76  | 0.92 $\pm$ 0.05      | 11.3 $\pm$ 0.7 |

\*Number of samples.

TABLE 2 Results of Statistical Comparisons Between Plume and Control Group Means by Mann-Whitney U-Test

| Comparisons             | Weight                         | K-factor                        | RNA/DNA                         |
|-------------------------|--------------------------------|---------------------------------|---------------------------------|
| <b>Brown trout</b>      |                                |                                 |                                 |
| Plume vs. total control | $p > c_t^*$<br>( $P > 0.10$ )† | $c_t > p$<br>( $P > 0.05$ )     | $p > c_t$<br>( $P < 0.002$ )‡   |
| Plume vs. control 1     | $p > c_1$<br>( $P > 0.05$ )    | $p > c_1$<br>( $P > 0.10$ )     | $p > c_1$<br>( $P < 0.002$ )‡   |
| Plume vs. control 2     | $c_2 > p$<br>( $P > 0.10$ )    | $c_2 > p$<br>( $P < 0.02$ )‡    | $p > c_2$<br>( $P > 0.10$ )     |
| Control 1 vs. control 2 | $c_2 > c_1$<br>( $P > 0.10$ )  | $c_2 > c_1$<br>( $P = 0.002$ )‡ | $c_2 > c_1$<br>( $P < 0.002$ )‡ |
| <b>Rainbow trout</b>    |                                |                                 |                                 |
| Plume vs. total control | $c_t > p$<br>( $P > 0.10$ )    | $c_t > p$<br>( $P > 0.10$ )     | $p > c_t$<br>( $P > 0.05$ )     |
| Plume vs. control 1     | $c_1 > p$<br>( $P > 0.10$ )    | $c_1 > p$<br>( $P > 0.05$ )     | $p > c_1$<br>( $P < 0.002$ )‡   |
| Plume vs. control 2     | $c_2 > p$<br>( $P > 0.10$ )    | $p > c_2$<br>( $P > 0.10$ )     | $c_2 > p$<br>( $P > 0.05$ )     |
| Control 1 vs. control 2 | $c_1 > c_2$<br>( $P > 0.10$ )  | $c_1 > c_2$<br>( $P > 0.10$ )   | $c_2 > c_1$<br>( $P < 0.002$ )‡ |
| <b>Chinook salmon</b>   |                                |                                 |                                 |
| Plume vs. total control | $c_t > p$<br>( $P > 0.10$ )    | $p > c_t$<br>( $P > 0.05$ )     | $c_t > p$<br>( $P > 0.10$ )     |
| Plume vs. control 1     | $c_1 > p$<br>( $P > 0.05$ )    | $p > c_1$<br>( $P > 0.10$ )     | $c_1 > p$<br>( $P < 0.002$ )‡   |
| Plume vs. control 2     | $p > c_2$<br>( $P > 0.10$ )    | $p > c_2$<br>( $P < 0.002$ )    | $p > c_2$<br>( $P > 0.10$ )     |
| Control 1 vs. control 2 | $c_1 > c_2$<br>( $P = 0.01$ )‡ | $c_1 > c_2$<br>( $P = 0.02$ )‡  | $c_1 > c_2$<br>( $P < 0.002$ )‡ |

\*Relative mean values (plume > total control).

†Probability of no statistical difference for each comparison.

‡Significant difference.

TABLE 3 Mean Days Between Mark and Recapture and Estimated Times at Discharge Temperature for Plume-Resident Fish Bearing Temperature-Sensitive Tags (Point Beach, Sept. 5–Oct. 24, 1973)

| Species        | Days tagged | Hours at discharge temperature | Percent time at discharge temperature |
|----------------|-------------|--------------------------------|---------------------------------------|
| Brown trout    | 20 (1–83)*  | 66.8 (0–300)                   | 17.0 (0–50.0)                         |
| Rainbow trout  | 8.5 (1–21)  | 16.6 (0–52)                    | 14.9 (0–50.0)                         |
| Chinook salmon | 14.3 (2–28) | 45.0 (0–75)                    | 9.2 (0–19.5)                          |

\*Numbers in parentheses are ranges for individual fish.

approximated their optimum temperatures or (2) moved between warm and cool areas in response to physiological demands.

Unfortunately our temperature-sensitive fish tags integrate temperature exposure and do not allow us to differentiate between long periods at intermediate temperature and short periods at high temperature. Our use of time at discharge temperature is a "worst case" approach and is based on our observations of many of these fishes in the discharge flumes during periods when discharge temperatures were below 22°C. We made measurements of body temperature immediately after capturing brown and rain-

bow trout from the discharge flumes; the maximum body temperatures were approximately 20°C (Spigarelli et al., 1974). Body-temperature measurements also were made on hundreds of brown and rainbow trout and chinook salmon caught by sport fishermen from the near-field plume. The maximum body temperatures measured were: brown trout, 21.3°C; rainbow trout, 23.5°C; and chinook salmon, 23.3°C (Spigarelli, 1975). All evidence to date suggests that these species are avoiding discharge temperatures exceeding 22°C.

Since these fishes were not permanent residents at discharge temperatures and there was some variability in tag

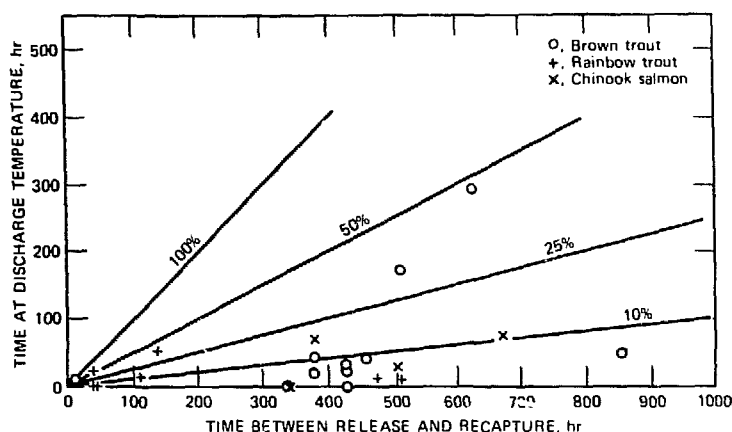


FIG. 1 Estimates of percentage and total time spent at discharge temperature for brown and rainbow trout and chinook salmon marked with temperature-sensitive fish tags at the Point Beach Nuclear Plant (September–October 1973).

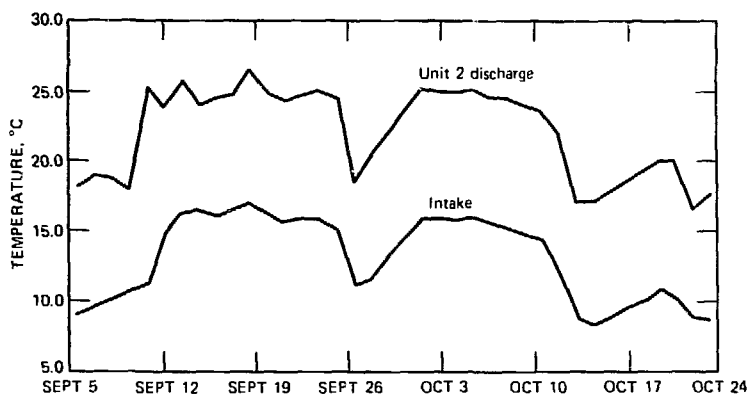


FIG. 2 Daily mean intake and discharge temperatures at the Point Beach Nuclear Plant during temperature-sensitive fish-tag study (September–October 1973).

time and percent times at discharge temperature between individuals of a species, we tested for linear correlations between growth indicators and residence times. There were no significant relationships between (1) RNA–DNA ratios and tag days or percent time at discharge temperature; (2) K-factor and tag days or percent time at discharge temperature; and (3) RNA–DNA ratios and K-factor. The lack of correlation between RNA–DNA ratios and residence times indicates that fish growth rates were not greatly affected by the times spent at elevated temperatures. Similarly, K-factors were not related to times at temperature. Since RNA–DNA ratios were not related to K-factors, we conclude that these two measures of growth are independent and reflect different growth histories (K-factor = long term; RNA–DNA = short term).

Although the sample sizes in this study are limited, we feel that our data on recent temperature exposures amply support the conclusion that these fishes did not experience any severe changes in growth during September and October as a result of their residence in the Point Beach thermal-plume area. The RNA–DNA data show that plume brown trout were growing at a faster rate than average control brown trout (and at a rate similar to the fastest growing control group). Plume rainbow trout were growing at a similar rate to control rainbows, and plume chinook resembled the slowest growing control group. K-factors generally indicated that plume fish were in similar condition to control fish. Temperature-sensitive fish tags showed that plume fish were not remaining at discharge temperatures for long periods of time, suggesting that these fish

# The Effect of Temperature, Salinity, and Food Availability on the Growth and Food-Conversion Efficiency of Postlarval Pinfish

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## ABSTRACT

Growth rate, feeding rate, and food-conversion efficiency of postlarval pinfish, *Lagodon rhomboides*, were measured under various combinations of temperature, salinity, and food availability. Data were analyzed by multiple regression and presented as response surfaces. Temperature accounted for most of the variation in maximum feeding rate. Temperature and feeding rate accounted for over 90% of the observed variation in growth rate. Salinity effects were more important in predicting growth efficiency than in predicting growth rate. Because a feeding-temperature interaction affects growth and because the effect of thermal effluents on food availability is unknown, it is impossible at this time to predict whether thermal alteration of the environment would increase or decrease growth of pinfish.

Environmental factors affect estuarine animals by altering a variety of physiological and behavioral processes. To understand the effects of specific environmental alterations, we must understand the response mechanisms involved. Unfortunately the large number of naturally operating biological processes has precluded an adequate understanding of organismal responses to environmental change. We cannot expect to understand how environmental alterations affect each individual process. We can, however, describe the collective response of these processes in any organism; this integrated response is evident in a growth rate that is easily measured.

As indicators of estuarine environmental quality, we measured growth rates, calculated food-conversion efficiency, and determined maximum feeding rates of postlarval pinfish, *Lagodon rhomboides*. The environmental characteristics that we tested were temperature, salinity, and the amount of available food. The experimental conditions tested include those which are typical for the

species as well as those which are likely to be found in an area receiving thermal effluents.

## MATERIALS AND METHODS

Experiments were conducted using fish held under natural temperature and salinity conditions (6 to 16°C and 25 to 33 ppt) for a short time before experimentation. Fish were caught in the lower Newport River, North Carolina, by dipnetting along the shore in 1973 and in a tidal net in mid-channel during 1974. After capture the fish were held in flowing seawater for 1 to 7 days and fed newly hatched brine shrimp, *Artemia salina*, daily. Mortality, often high at the beginning of the holding period, rapidly declined to near zero as the naturally weak or injured fish died.

Experimental procedure consisted in weighing 70 fish (approximately 15 to 30 mg each), randomly assigning them to a control group or one of six experimental groups of 10 fish, feeding the experimental groups for 4 to 8 days and then measuring their dry weight. The control group was immediately dried, and its dry-weight/wet-weight ratio was used to estimate the original dry weight of the experimental groups. During experiments the 10 fish in each group were held in a single 4-liter 15-cm-diam. white plastic cylinder. After assignment to these containers, the fish were adjusted to a single test temperature and salinity condition. Following this overnight adjustment period, newly hatched live brine shrimp were periodically offered to the fish. One of the six groups was given sufficient food so that a few nauplii were always available. Other containers were offered 80, 60, 40, 20, and 0% of this amount. Each day an aliquot of the *Artemia* suspension which was fed to the fish was strained and the *Artemia* dried and weighed so that the total dry weight of food given each container could be estimated. At low temperatures, weight changes were small

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controlled their exposures by avoiding high discharge temperatures ( $>22^{\circ}\text{C}$ ). The fact that these fish were exposed to elevated temperatures for varying periods of time indicates that salmonids do orient to this discharge under certain conditions during the fall, but that accelerated metabolic demands are compensated by food availability and the ability of motile fish to behaviorally regulate their temperature exposures.

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and the experiments were conducted for 7 or 8 days. With increasing temperature, weight changed more rapidly and the experiments were shortened to as little as 4 days. After the last feeding, sufficient time was allowed for the fish to clear their guts before their dry weight was determined.

In 1973 only the effects of temperature and feeding rate were studied, whereas in 1974 salinity effects were also examined. The first year all fish were maintained at 25-ppt salinity. Each experiment consisted of groups of 10 fish at six different feeding levels. These experiments were conducted at even temperatures from 6 to 34°C and were duplicated at 18 and 24°C. The 1974 experiments were conducted at 15, 25, and 35 ppt salinity and at 8, 10, 16, 24, 30, and 32°C. Fish were tested at all three salinities at each temperature except at 30°C where only 25 ppt was examined. The experiment at 24°C and 25 ppt was repeated three times.

During experimentation we attempted to minimize the effect of extraneous variables. The water was changed daily by gentle siphoning, which did not appear to disturb the fish. A screen over the siphon intake prevented removal of food from the containers. A 12-hr photoperiod was provided at 250 to 300 lux. Temperature was maintained within 0.5°C of intended level, and salinity was within 1.0 ppt.

Growth and maximum feeding rate were expressed as percent of body weight per day. Growth was calculated from Winberg's (1956) formula

$$r = 100 [(W_n/W_o)^{1/n} - 1]$$

where  $r$  = average daily percent change in dry body weight of a group of fish

$n$  = number of days fed

$W_n$  = final dry weight

$W_o$  = original dry weight, estimated by multiplying the original live weight by the dry-weight/live-weight ratio of the control

Feeding-rate calculations required the assumption of constant growth rates. From the original dry weight and calculated growth rates, we determined the dry weight of fish in each container at the beginning of each day. The estimated dry weight of brine shrimp offered to the fish was divided by the estimated daily weight of fish to estimate proportion of body weight ingested. These proportions were then summarized as average daily percent of dry body weight ingested.

Food conversion (i.e., growth) efficiencies were calculated by dividing the growth rate (average daily percent change in dry body weight) by the feeding rate (average percent of dry body weight ingested). Efficiencies were not calculated when growth was negative.

Dry weights obtained in 1973 were adequate for analysis of growth rate but not accurate enough to warrant calculation of food-conversion efficiency. The weights were determined within  $\pm 0.05$  mg but were biased by absorption of moisture during weighing. For our growth-rate

analysis, measurement error was low relative to the total variance of growth rate. However, as true growth rates approached zero, the weighing-error component becomes a significant fraction of estimated growth, which drastically alters estimated efficiencies. The relatively large error at low growth and low feeding rates were further magnified in their quotient (growth efficiency). Warren and Davis (1967) have shown the inappropriateness of calculating a growth efficiency when its component rates are low in relation to error. We judged our error for 1973 to be too large and have not calculated efficiencies.

In 1974 the dry weights were determined more precisely ( $\pm 0.01$  mg) and accurately (absorption of atmospheric water was eliminated by weighing in a low-humidity environment), thus permitting food-conversion-efficiency calculations.

Maximum feeding rates, growth rates, and food-conversion efficiencies were all summarized by multiple regression equations. The equations included linear and quadratic terms for the environmental variables tested and also their first-order interactions. "F" tests were performed for significance of the independent effects of the regression terms. All terms from the regression equation were used in plotting response surfaces for growth rate and efficiency.

## RESULTS AND DISCUSSION

Because of mortality in some containers, not all the test conditions yielded useful data. Experimental fish that died probably ate at different rates than the fish that lived, which made it impossible to calculate true feeding rates of the survivors. Data used in subsequent statistical analysis, i.e., experimental conditions in all containers that experienced no mortality, are found in Table 1.

### Thermal Tolerance

Postlarval pinfish had slightly more thermal tolerance in 1973 than they did in 1974. During 1973, fish at 6°C stopped eating after several days and some fish died, but at 8°C they survived and gained weight. At 32°C they ate and grew, but at 34°C they died. In 1974 our only trial at 6°C resulted in high mortality the first day; at 8°C they ate and survived but lost weight. At 30°C the fish grew well, but several attempts at 32°C failed.

The yearly difference in thermal tolerance cannot be attributed to any particular cause, but it corresponds with our observation that fish in 1974 were not as hardy as those used in 1973. In the later year, more fish had damaged fins, and postcapture mortality was higher and persisted several days longer. Subsequent data have shown that the difference was not a function of location or method of capture. Perhaps a difference in natural acclimation temperature was responsible. During the early months of 1974, mean water temperature was about 4°C warmer than in 1973; thus the larvae were naturally acclimatized to a higher temperature. Acclimation to higher temperatures might explain reduced growth and survival at low tempera-



TABLE 1 Experimental Conditions from Which Data Were Used to Calculate Regression Equations for Growth Rate and Growth Efficiency. Conditions Resulting in Negative Growth Are in Brackets and Were Omitted When Calculating Efficiency

| Year and temperature, °C | Salinity, ppt | Feeding rate, % <i>ad libitum</i>                              |
|--------------------------|---------------|----------------------------------------------------------------|
| 1973                     |               |                                                                |
| 32                       | 25            | 100, 77, 64, 49, [26, 0]                                       |
| 30                       | 25            | 100, 92, 48, [30, 0]                                           |
| 26                       | 25            | 100, 71, 42, 24 [0]                                            |
| 24                       | 25            | 100, 86, 81, 80, 67, 57, 45, 25 [22, 0, 0]                     |
| 22                       | 25            | 100, 82, 61, 48, 24, [0]                                       |
| 18                       | 25            | 100, 79, 68, 64, 61, 44, 29, 23, 16 [0]                        |
| 16                       | 25            | 100, 76, 63, 46, 25 [0]                                        |
| 14                       | 25            | 100, 75, 68, 47, 22 [0]                                        |
| 12                       | 25            | 100, 69, 60, 44, 21                                            |
| 10                       | 25            | 100, 82, 64 [41, 23, 0]                                        |
| 8                        | 25            | 100, 40 [0]                                                    |
| 1974                     |               |                                                                |
| 30                       | 25            | 100, 94, 74, 50 [26, 0]                                        |
| 24                       | 35            | 100, 84, 66, 42 [24, 0]                                        |
|                          | 25            | 100, 85, 82, 75, 67, 66, 61, 54, 58, 37, 35, 25, [21, 0, 0, 0] |
|                          | 15            | 100, 84, 68 [0]                                                |
| 16                       | 35            | 100, 83, 63, 44 [27, 0]                                        |
|                          | 25            | 100, 79 [36, 22, 0]                                            |
|                          | 15            | 100, 77, 56 [42, 23, 0]                                        |
| 10                       | 35            | 100, 73, 52 [37, 0]                                            |
|                          | 25            | 100, 62 [42, 21, 0]                                            |
|                          | 15            | 100, 84, 64 [42, 23, 0]                                        |
| 8                        | 35            | [100, 75, 58, 41, 20, 0]                                       |
|                          | 25            | [0]                                                            |
|                          | 15            | [100, 86, 66, 43, 23, 0]                                       |

tures, which occurred in 1974. But this logic would also predict better survival at high temperatures, which did not occur.

### Maximum Feeding Rate

Maximum feeding rate of postlarval pinfish is highly dependent on temperature but not on salinity. Regression equations with linear and quadratic temperature terms could account for 94% of the observed variation in maximum feeding in 1973 and 88% of the variation in 1974. Independent effects (partial sums of squares, Table 2) of  $T$  and  $T^2$  were significant (1973) and much larger than independent effects of salinity terms (1974). The partial sums of squares for  $S$ ,  $S^2$ , and  $TS$  were far from significant, and inclusion of all three terms in the regression increased the correlation coefficient only negligibly from 0.939 to 0.940. Thus we concluded that salinity has little effect on maximum feeding rate of postlarval pinfish. Strong temperature and weak salinity effects have also been found in postlarval and juvenile estuarine flounders (Peters and Angelovic, 1973; Peters and Boyd, 1972). Because salinity effects are inconsequential, we have ignored them in subsequent feeding-rate analyses.

Effects of temperature on feeding rates were different in 1973 than in 1974 (Fig. 1). Regression analysis indicated that the linear temperature coefficients were positive and different from zero ( $P < 0.05$ ) both years. They were significantly different between years. The quadratic terms were negative and also different between years, and only in 1973 was the term different from zero ( $P < 0.05$ ). Other studies (Brett, Shelbourn, and Schoop, 1969; Peters and Kjelson, 1975) have shown the same type of temperature-feeding-rate curve (i.e., positive linear and negative quadratic temperature effects). From the regression equation we calculated the maximum feeding rate in 1973 to be

TABLE 2 Analysis of Variance of *Ad Libitum* Feeding Rate of Postlarval Pinfish

| Year | Source         | Degrees of freedom | Sum of squares | Sequential sum of squares | Partial sum of squares | Probability of greater partial sum of squares |
|------|----------------|--------------------|----------------|---------------------------|------------------------|-----------------------------------------------|
| 1973 | Regression     | 2                  | 1598           |                           |                        |                                               |
|      | T              | (1)                |                | 1394.86                   | 394.66                 | 0.007                                         |
|      | T <sup>2</sup> | (1)                |                | 203.14                    | 203.14                 | 0.002                                         |
|      | Error          | 9                  | 91             |                           |                        |                                               |
|      | Total          | 11                 | 1689           |                           |                        |                                               |
| 1974 | Regression     | 5                  | 997            |                           |                        |                                               |
|      | T              | (1)                |                | 973.07                    | 66.05                  | 0.081                                         |
|      | T <sup>2</sup> | (1)                |                | 21.67                     | 22.30                  | 0.279                                         |
|      | S              | (1)                |                | 1.50                      | 1.33                   | 0.785                                         |
|      | S <sup>2</sup> | (1)                |                | 1.07                      | 1.07                   | 0.807                                         |
|      | TS             | (1)                |                | 0.01                      | 0.01                   | 0.941                                         |
|      | Error          | 8                  | 132            |                           |                        |                                               |
|      | Total          | 13                 | 1129           |                           |                        |                                               |

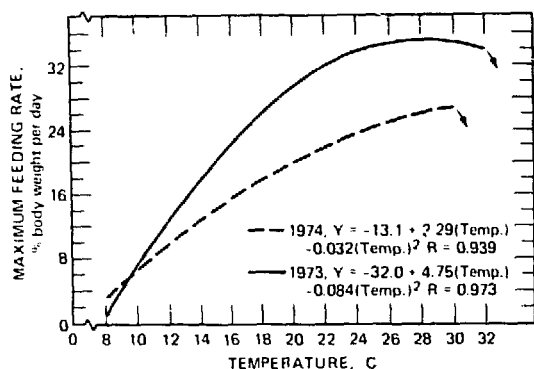


FIG. 1 Estimation of percent body weight ingested per day by postlarval *Lagodon rhomboides* at ad libitum feeding rate.

at 28.3°C, but in 1974 it was calculated to be above the lethal temperature.

The nonsignificant quadratic temperature coefficient and impossibly high temperature for maximum feeding rate in 1974 might be attributed to the experimental temperatures tested. In 1973 one-fourth of the maximally fed fish were held at temperatures above 24°C, and it is primarily at these high temperatures that the quadratic effect becomes important. Since few feeding data were collected at such high temperatures in 1974, the decreased significance of the quadratic temperature effect is not surprising. We believe that a biologically important quadratic temperature effect existed in 1974, even though it was not statistically significant.

### Growth Rate

Feeding rate and temperature were important in predicting growth rate, whereas salinity information was of little value. Regression, while only temperature and feeding-rate information, could account for 96% of the growth-rate variation in 1973 and 92% of the variation in 1974 (Table 3). The independent effects (i.e., partial sums of squares) of  $T + T^2$  were statistically significant (Table 4);

TABLE 3 The  $R^2$  Values Obtained from Regressions Relating Growth Rate and Efficiency to Various Temperature, Salinity, and Feeding-Rate Terms

| Terms in model                       | Growth rate |      | Growth efficiency |
|--------------------------------------|-------------|------|-------------------|
|                                      | 1973        | 1974 | 1974              |
| $T, T^2$                             | 0.07        | 0.11 | 0.26              |
| $F, F^2$                             | 0.72        | 0.63 | 0.29              |
| $T, T^2, F, F^2, TF$                 | 0.96        | 0.92 | 0.71              |
| $T, T^2, S, S^2, TS$                 | *           | 0.12 | 0.46              |
| $F, F^2, S, S^2, FS$                 | *           | 0.31 | 0.65              |
| $T, T^2, F, F^2, TF, S, S^2, TS, FS$ | *           | 0.92 | 0.83              |

\*Salinity data not collected.

however, these terms could explain only a small fraction of the total variation in growth rate (Table 3). The gain in  $R^2$  obtained by including temperature terms ( $T, T^2$ , and  $TF$ ), in addition to feeding terms ( $F$  and  $F^2$ ), was significant ( $P < 0.05$ ) in both years. In 1974, addition of salinity terms ( $S, S^2, TS$ , and  $FS$ ) to the simpler temperature-feeding-rate model also produced a significant ( $P < 0.05$ ) though small increase in  $R$ . However, none of the independent effects of salinity containing terms were significant ( $P < 0.20$ ). Thus the very slight increase in predictive ability following inclusion of salinity terms must be attributed to their joint effects.

Although the full regression models used to describe growth rate differed between years, their corresponding regression coefficients were similar and in some ways resembled comparable terms describing growth of other species. Linear temperature and feeding-rate coefficients were positive, and their quadratic terms were negative (Table 5). The signs of those coefficients indicate the possibility of optimum conditions which could produce a maximum growth rate; however, optima did not occur within the range of conditions tested. The lack of optimal conditions and the signs of the regression terms ( $T, T^2, F, F^2$ , and  $TF$ ) describing growth rate of pinfish are similar to results with summer flounder (Peters and Angelovic, 1973) and southern flounder (Peters and Kjelson, 1975).

The importance of temperature and feeding effects on growth of pinfish, shown here, differs in some respects from data on other species. Temperature has frequently been described as significantly affecting growth rate. However, in studies analyzed by Paloheimo and Dickie (1966), feeding rate was expressed as amount of food ingested which is highly correlated with temperature and results in confounding of the temperature and feeding effects. Herein, with feeding rate expressed as a fraction of feeding capacity, the correlation with temperature is removed, so that it is easier to assess their individual effects. The independent effects of  $T$  and  $T^2$  are real (Table 4) but of very limited predictive value (Table 3). Judging from the  $R^2$  values associated with various regression models (Table 3), the predictive value of temperature is contained primarily in the temperature-feeding interaction term. With the use of similar analysis on growth data from two species of flounder, regressions containing only  $T$  and  $T^2$  terms are of high predictive value,  $R^2 = 0.68$  and  $0.74$  (Peters and Angelovic, 1973; Peters and Kjelson, 1975).

Our results with pinfish indicate that temperature changes may have a significant effect on growth rate. Whether or not the effect is actually important depends on the feeding rate (Figs. 2 and 3). At low feeding rates, increased temperature results in increased weight loss, whereas, at high feeding rates, increased temperature produces faster growth. Little information is available describing feeding rates of postlarval fish in nature. The best ration estimate for postlarval pinfish is 9% of body weight per day at 15°C (Kjelson, personal communication), which corresponds to about 60% of the maximum feeding

**TABLE 4** Probabilities of Larger F for Independent Effects of Variables in Equations Predicting Growth Rate and Efficiency of *L. rhomboides*. Probabilities >0.20 are Associated with the TS and FS Terms in the 1974 Equations and with All Blank Spaces in the Table

| Dependent variable | T     | T <sup>2</sup> | F     | F <sup>2</sup> | TF    | S     | S <sup>2</sup> |
|--------------------|-------|----------------|-------|----------------|-------|-------|----------------|
| Rate, 1973         | <0.01 | <0.01          |       | <0.01          | <0.01 | *     | *              |
| Rate, 1974         | <0.20 | <0.01          |       | <0.01          | <0.01 |       |                |
| Efficiency, 1974   | <0.01 | <0.01          | <0.01 | <0.01          |       | <0.01 | <0.01          |

\*No salinity data collected.

**TABLE 5** Coefficients for Regression Equation Relating Growth Rate (Percent Daily Change) and Growth Efficiency (Percent of Food Deposited as New Tissue) of Postlarval Pinfish to Temperature (°C), Feeding Rate (Percent of *Ad Libitum* Feeding Rate), and Salinity (ppt)

| Coefficients   | Growth rate | Growth rate | Efficiency |
|----------------|-------------|-------------|------------|
|                | 1973        | 1974        | 1974       |
| Intercept      | -7.8674     | -6.6497     | -79.9102   |
| T              | 0.7922      | 0.3298      | 8.2843     |
| T <sup>2</sup> | -0.0303     | -0.0176     | -0.1417    |
| F              | 0.0144      | 0.0322      | 1.9035     |
| F <sup>2</sup> | -0.0008     | -0.0007     | -0.0095    |
| TF             | 0.0125      | 0.0103      | -0.0061    |
| S              |             | -0.0083     | -4.9681    |
| S <sup>2</sup> |             | 0.0016      | 0.1154     |
| TS             |             | -0.0005     | -0.0322    |
| FS             |             | -0.0004     | -0.0016    |

rate in 1974 and 45% of the maximum feeding rate in 1973 (Fig. 1). If these natural feeding rates remained the same, a temperature increase would have relatively little effect on growth. Increased temperature may increase the ability of fish to catch their prey but may also increase the ability of their prey to escape. If the capture and escape capabilities are affected equally by thermal additions to natural environments, then the feeding rate will not change and thermal effects on growth rate can be calculated. Unfortunately there is little information describing the effects of temperature on predation success in nature.

### Growth Efficiency

Temperature, salinity, and feeding rate were all of considerable value in predicting growth efficiency. Statistical significance was attributed to the independent effects of regression terms involving each of the three environmental factors (Table 4). If any one of the factors is eliminated from the regression, the result is a considerable reduction in sum of squares due to regression (Table 3). The importance of temperature and feeding rate was expected since those factors had a significant effect on growth. The importance of salinity terms was unexpected. The salinity terms were of much more value in predicting efficiency than they were in predicting growth rate.

As with growth rate, no one set of conditions existed which would yield a maximum conversion. Growth was efficient at high and low salinities, with lower efficiencies occurring at intermediate salinities. Highest predicted efficiencies were at or near the maximum feeding capacity (Fig. 4), above the 2/3 *ad libitum* rate reported for trout (Warren and Davis, 1967) and flounders (Peters and Angelovic, 1973; Peters and Kjelson, 1975). As with other postlarval fishes we have tested, highest efficiencies occurred below the temperature for maximum *ad libitum* feeding rate but above natural environmental temperatures.

Since zero growth rate and zero growth efficiency are identical physiological conditions, we might expect the zero rate and efficiency isopleths (Figs. 3 and 4) to be the same. They are not identical, because the efficiency isopleths were calculated from a restricted portion of the data used in growth-rate analysis (Table 1). The data used in efficiency analysis were obtained primarily above the 40% feeding level with no data below 25%. Thus the lower efficiency isopleths (Fig. 4) are extrapolations from a portion of the data used in our growth-rate analysis.

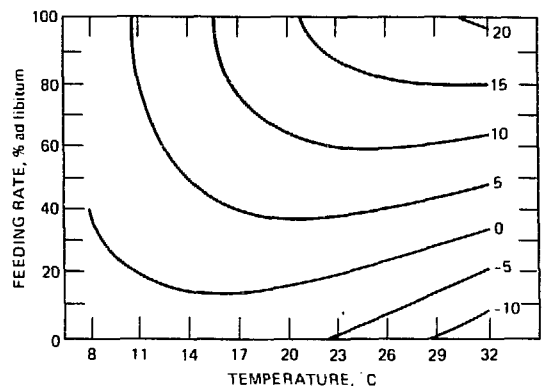


**FIG. 2** Estimation of growth rate (percent daily increase in dry weight) of postlarval *Lagodon rhomboides*, from the regression equation relating growth rate in 1973 to temperature and feeding rate (see Table 5). Each isopleth consists of all the temperature and feeding-rate combinations that result in the growth rate given to the right of each isopleth.

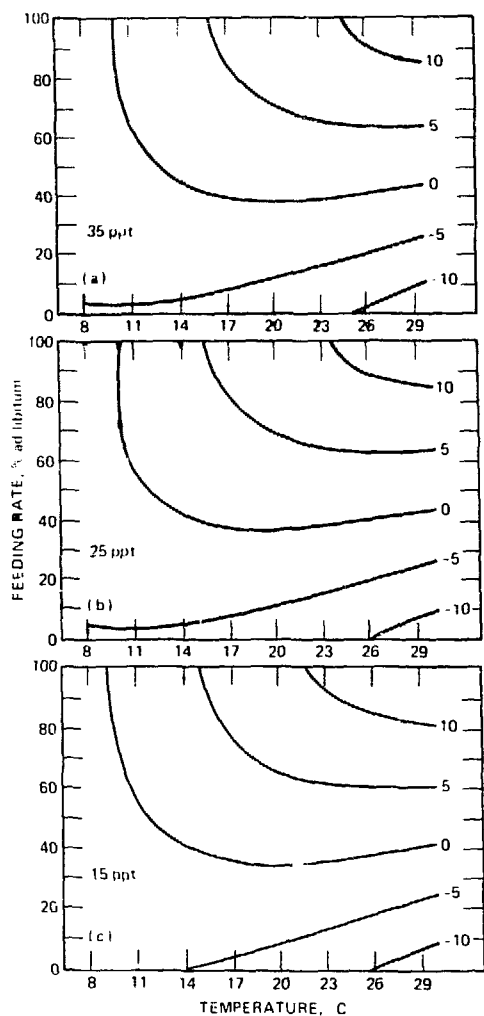


FIG. 3 Estimation of growth rate (percent daily increase in dry weight) of postlarval *Lagodon rhomboides*, from the regression equation relating growth rate in 1974 to temperature, salinity, and feeding rate (see Table 5). Each isopleth consists of all temperature and feeding-rate combinations that result in the growth rate given to the right of each isopleth.

### Conclusions

Maximum feeding capacity of postlarval pinfish is strongly influenced by temperature and relatively unaltered by salinity.

Food-conversion efficiency was significantly affected by temperature, salinity, and food availability. Although we are unsure of the ecological significance, reduced efficiencies were found at intermediate salinities, whereas growth was more efficient at temperatures and feeding levels above those which the postlarvae naturally encounter.

Growth rates were adequately described using only temperature and feeding information. Thus heating natural water would affect pinfish growth rate in a predictable

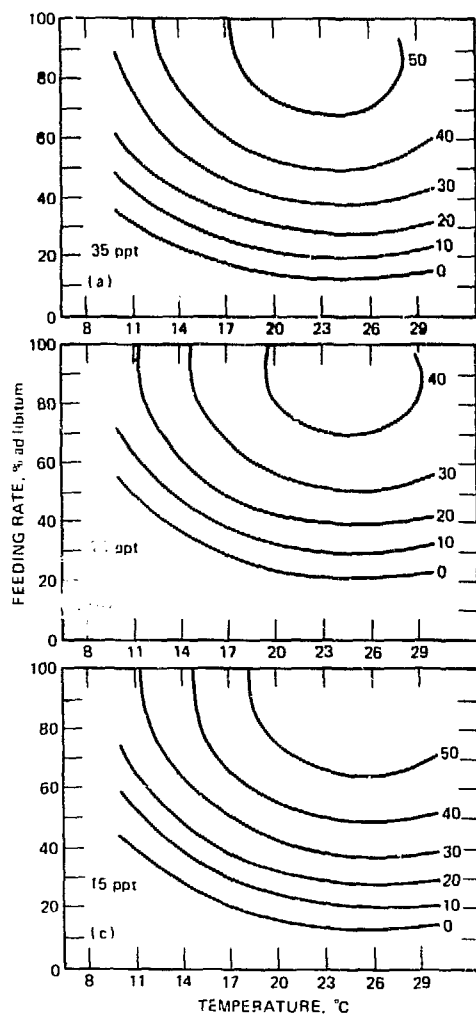


FIG. 4 Estimation of the growth efficiency (percent of food deposited as new tissue) of postlarval *Lagodon rhomboides*, from the regression equation relating efficiency to temperature, salinity, and feeding rate (see Table 5). Each isopleth consists of all the temperature and feeding-rate combinations that result in the growth efficiency given to the right of each isopleth.

fashion, if food availability were known or did not change. However, increased temperature may alter food availability by changing: (1) food abundance, (2) predatory effectiveness of the larvae, or (3) escape capabilities of the prey. Because both food availability and temperature affect growth and because of the unknown effect of thermal effluents on food availability, it is impossible at this time to predict whether thermal effluents will increase or decrease fish growth.

### ACKNOWLEDGMENTS

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# Temperature as a Growth Accelerator in the Spot (Teleost: Sciaenidae)

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## ABSTRACT

Young spot, *Leiostomus xanthurus*,—and 10 other species of fish—were trapped by the installation of a weir across the effluent canal at the Yorktown Power Station of the Virginia Electric and Power Company in December 1973 during a discharge modification. In the subsequent 12 months, the fish were subjected to a wide spectrum of temperature changes and to chlorination associated with the normal operation of two existing units and the start-up of a third unit. Certain individuals of the spot apparently grew at a faster rate and to a larger size than has been heretofore reported in the literature.

Sport fishermen who understand finfish behavior relative to power-station discharges react, as do the fish, in a predictable manner. They congregate during the fall, winter, and spring in areas where their individual production and well-being will attain optimum success. Numerous reports and publications attest to this behavior; some of these have been cited in annual reviews of thermal-effects literature (Coutant and Pfuderer, 1974).

A situation related to this phenomenon which occurred at the Yorktown Power Station of the Virginia Electric and Power Company (Vepco) warrants documentation largely because it happened under field conditions at an operating steam-electric station coincidental with the construction of a new unit and because it concerns a population of fish living in an atypical environment from which they could not escape.

The Commonwealth of Virginia conducts an annual salt-water sport-fishing tournament, awarding certificates to anglers taking certain species weighing more than a specified minimum. In 1974 the minimum weight for spot was set at 15 oz. (425 g). In October reports were received of citation-size spot being taken from the closed-off discharge canal at the Yorktown Power Station, where heated water had been flowing before, during, and after installation of the weir.

At the close of the 1974 fishing tournament, the records showed that 240 spot weighing 425 g or more were caught in the Chesapeake Bay system. Five were taken in

the York estuary. Interviews with the fishermen who caught fish from the York revealed that four actually came from within the canal. One of these four was the largest spot taken in Virginia waters (324 mm in total length (TL) and weighing 595 g). The other three "citation" spot caught in the canal were registered at 279 mm TL, 454 g; 292 mm TL, 496 g; and 311 mm TL, 539 g. All were taken between Oct. 5–11, 1974. The president of the Peninsula Salt Water Fishermen's Association, who was a regular fisherman in the canal, and who caught the largest spot in 1974, in an interview gave a conservative estimate of over 2000 spot removed from the canal between April and November. He also stated that fish were taken from the canal throughout the summer months.

This paper documents the growth and survival of a population of fish inadvertently trapped in the discharge canal for up to 12 months, focusing on the growth of the spot, *Leiostomus xanthurus*.

## STUDY SITE AND METHODS

The Yorktown Power Station is located on the south bank of the York estuary, directly downstream from Yorktown, Va. Before 1974 the station consisted of two fossil-fueled units, 187.5 Mw(e) each, with once-through cooling, taking water from the river via an intake canal and discharging directly to the river via a discharge canal (Fig. 1). On Dec. 1, 1973, the canal was closed at the lower end by a fixed weir during construction of a new 882-Mw(e) unit. Pumps were installed at the lower end of the canal to pump cooling waters from units 1 to 3 through a deep-water multiport diffuser; this eliminated the shore-line discharge. The changeover to the diffuser system was deemed necessary to meet thermal requirements for estuaries.

Unit 3 underwent a period of start-up testing in the summer of 1974, was declared commercial on August 1, and was shut down on August 3 because of a malfunction. Unit 3 is designed to discharge cooling water at a  $\Delta T$  of

White, Hammond, Wooding, and Brehmer: Environmental Services Department, Virginia Electric and Power Company, Richmond, Virginia.

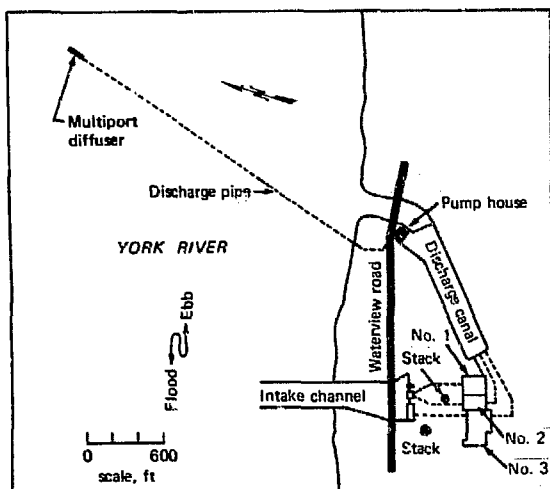


FIG. 1 Yorktown Power Station of the Virginia Electric and Power Company, Yorktown, Va.

5.6°C, which serves to temper the  $\Delta T$  discharge of 8.2°C from units 1 and 2. This, in turn, results in a  $\Delta T$  through the diffuser system of 6.5°C when all three units are operating. Temperatures in the canal were monitored with a temperature recorder installed near the diffuser pumps in August 1974. The film packet was changed in November, giving about three months of continuous hourly readings.

A large open basin, about 305 m long, 60 m wide, and 3.4 m deep, was created by the closing of the discharge canal. The canal sides and bottom are covered with a concrete matting to reduce erosion. Water flows through the canal at 12.6 m<sup>3</sup>/sec when units 1 and 2 are in operation and 58 m<sup>3</sup>/sec when all three units are operating.

Chlorine gas is used as a biocide at the Yorktown Power Station in quantities sufficient to maintain 0.5 mg/liter residual in the condenser discharge during application. Each unit is chlorinated for 27 min every 8 hr shift, with a maximum possible output of 2381.3 kg of chlorine per day collectively for the station.

On Nov. 13 and Dec. 18, 1974, several stationary experimental gill nets (19- to 76.2-mm bar mesh) were set at various locations in the canal. Total fishing time for all nets was about 5 hr.

## RESULTS AND DISCUSSION

Eggs, species of fish were taken by gill netting: 67 spot, 164 to 319 mm TL; 56 striped mullet (*Mugil cephalus*), 150 to 351 mm TL; 13 Atlantic croaker (*Micropogon undulatus*), 237 to 258 mm TL; 4 white perch (*Morone americana*), 205 to 248 mm TL; 1 ladyfish (*Elops saurus*), 302 mm TL; 1 weakfish (*Cynoscion regalis*), 166 mm TL; 1 red drum (*Sciaenops ocellata*), 163 mm TL; and 1 gizzard shad (*Dorosoma cepedianum*), 412 mm TL. All fish appeared to be in excellent condition with no external sign of fungus or parasites.

The fish were living within the canal, but their exact mode of entry is uncertain. There are at least three possibilities. First, most spot, Atlantic croaker, weakfish, white perch, and red drum were probably present in the canal when the outlet weir was installed. Second, jumping species such as the striped mullet, and possibly ladyfish and gizzard shad, may have jumped the weir into the canal since the weir extends ~0.3 m above spring-tide levels. A third possibility is that postlarval fishes survived passage through the station condenser system and remained in the canal.

All the 67 spot taken from the canal by gill net were measured, and 30 were weighed. There were two "citation-size" individuals (319 mm TL, 508 g, and 315 mm TL, 447 g). The 30 fish measured and weighed averaged 233.9 mm TL (range 164 to 319 mm) and 277.2 g (range 134.7 to 508 g). The remaining 37 fish averaged 225.1 mm TL (range 195 to 273 mm).

The length-weight equation for the measured and weighed fish was:

$$\log W = -1.594 + 1.698 \log L$$

The equation was fitted by simple linear regression of logarithmically transformed data using the Regression Procedure of the Statistical Analysis System (Barr and Goodnight, 1972) on an IBM System 370, Model 155 computer.

Gonads were examined to assess the state of sexual maturity of all adult fish. Most spot were in various stages of development from immature to "spent" for females and "running ripe" for males. Some males of striped mullet and Atlantic croaker were also running ripe. Two female spot were spent; this indicates that spawning occurred within the canal. No running ripe females of any species were taken. These data tend to substantiate the current belief that certain fishes, especially some sciaenids, mature on decreasing day length and not temperature (Richards, 1975).

Scales from spot examined by personnel of the Virginia Institute of Marine Science (VIMS) and of Vepco showed individuals with 0 to 2 annual rings. The frequency of occurrence of 0, 1, and 2 ring scales from spot is similar to that found by Pacheco (1962) for samples taken in December from Chesapeake Bay (Table 1) and, therefore, similar to the frequency expected in the estuary when the canal was closed.

TABLE 1 Frequency Comparison (%) of Annual Rings on Spot Scales

| Annual rings | Samples from discharge canal | Samples from Chesapeake Bay* |
|--------------|------------------------------|------------------------------|
| 0            | 87                           | 75                           |
| 1            | 10                           | 15                           |
| 2            | 3                            | 10                           |

\*Pacheco, 1962.

There is general confusion in the literature concerning the year classes of spot after the first year of growth because of their protracted spawning season and behavior patterns. Spot in Chesapeake Bay form an annulus from November through February of each year. January 1 has been established as a year-class "birthday" for spot hatched between November and February of any given year (Pacheco, 1957). Using this date, we examined the growth of spot on a monthly chronological basis from reports throughout its range from New York to Texas (New Jersey: Welsh and Breder, 1923. New York: Nichols and Breder, 1927. Virginia: Hildebrand and Schroeder, 1928: Pacheco, 1957. North Carolina: Hildebrand and Cable, 1930. South Carolina: Dawson, 1958. Florida: Townsend, 1956. Louisiana: Sundararaj, 1960. Texas: Pearson, 1928). Individuals taken from the canal of the Vepco station which displayed no annulus or one annulus on their scales were apparently 23 to 24 months old in November; and December 1974 and belong to the same year class. Spot with two scale annuli were apparently 35 to 36 months of age. Some appeared to have attained a larger size than any reported previously for a given chronological age (Fig. 2). The figures do not include spot taken by sport fishermen from the canal since we did not see them. They were registered and

certified by the Virginia Salt Water Fishing Tournament officials, however.

The average spot, like some other members of the family Sciaenidae, apparently grows larger and faster in northern waters (Hildebrand and Cable, 1930). Plots of total length at the end of each chronological year of life, however, show that Chesapeake Bay spot apparently grow faster during their first 2 years than do similar individuals from more northern or southern waters (Fig. 2). The data indicate that spot in the Vepco discharge canal had a more rapid rate of growth than that normally found in spot populations of Chesapeake Bay.

The supply of food for fish within the canal does not appear limiting. Schools of *Fundulus heteroclitus*, *Anchoa mitchilli*, and *Menidia* sp. and large numbers of grass shrimp, *Palaemonetes* sp., have been observed. These species apparently survived entrainment through the station.

The most interesting and possibly most applicable aspect of this study is the environment in which the fish survived and flourished for about a year. The water temperature in any power-station discharge canal is higher than ambient when the station generators are operating. Also, since heat rejection from condensers is a function of unit load, canal water temperatures usually fluctuate as demands on station generation vary. Records for August through November show some of these fluctuations, which are magnifications of what normally occurs in shallow tidal areas, over 5-day (Fig. 3), 1-day (Fig. 4), and 1-hr (Fig. 5) periods.

At least two investigators have reported a critical thermal maximum (CTM) for spot. Hoss, Coston, and Hettler (1971) found a CTM of 31.1°C for postlarval spot acclimated to 15°C. Bridges (1971) reported a CTM of 37.5°C for juvenile spot acclimated to 30°C. Another report (Meldrim and Gift, 1971) showed that juvenile spot acclimated to 20°C avoided a temperature of 25.6°C and preferred a temperature of 18.4°C. Such data are not applicable to the trapped population discussed here because of different temperature and salinity regimes, light levels, races, size ranges, and physiological stresses involved. Whether the CTM for spot or the other species in the canal was ever exceeded is irrelevant since CTM values are derived from acclimating a fish to a predetermined constant temperature and then heating its environment at a rate of 1°C/min until the CTM is determined. The canal population could exhibit no preference and could not avoid its environment. They were not exposed to a constant acclimation temperature, salinity, or light level or to the stresses of laboratory handling and space limitations. Their environment for 1 year was unique and, except for temperature, chlorine, and an obviously insignificant space limitation, was natural.

That marine fish can exist and apparently thrive in a discharge canal when retreat to a preferred environment is impossible raises serious questions in our minds concerning the applicability and biological validity of using laboratory-

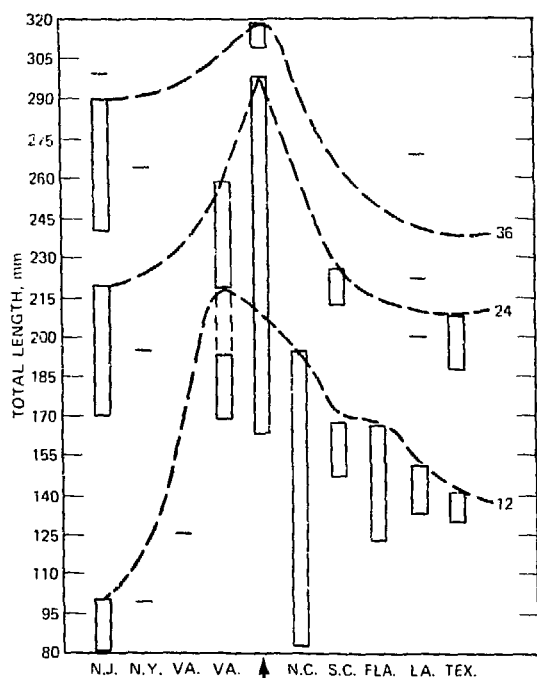


FIG. 2 Maximum total lengths attained by spot, *Leiostomus xanthurus*, populations from New York to Texas at the end of 12, 24, and 36 months of life (dashed curves) as extrapolated from various investigations (see text for references). Vertical bars are length ranges. Horizontal lines are average or modal lengths where no ranges were reported. The arrow denotes Yorktown canal samples, which contained no individuals less than 23 months of age.



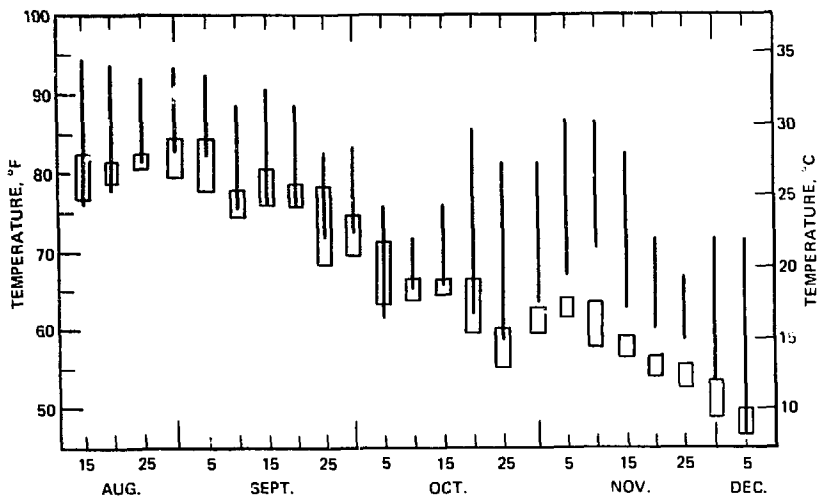


FIG. 3 Five-day maximum and minimum temperatures in the Yorktown discharge canal (vertical lines) and in the York estuary (rectangles, from VIMS records) in 1974.

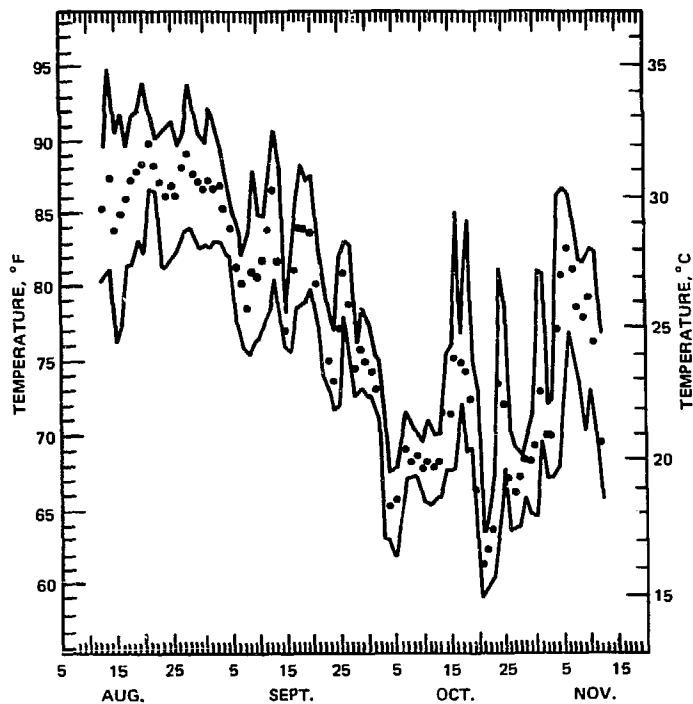


FIG. 4 Daily maximum, minimum, and average temperatures (closed circles, ●) in the Yorktown discharge canal.

derived numbers (e.g., critical thermal maximum and ultimate incipient lethal temperature) to describe the thermal tolerance of individual fish species in nature. These questions become even more significant when numbers are derived not for a specific site but as regulatory

limitations over wide latitudinal and longitudinal ranges. Frequently in laboratory experiments abnormal stresses are imposed, and constant acclimation temperatures (often out of season) and sudden temperature rises (often of prolonged duration) are used.

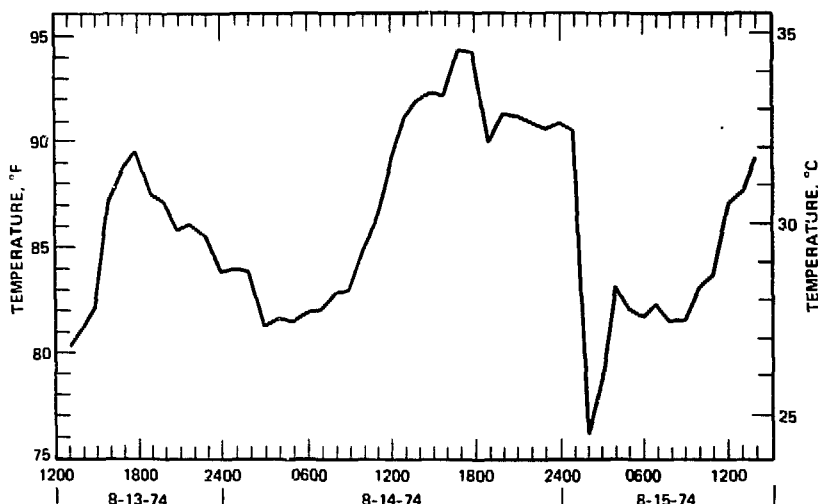


FIG. 5 Representative hourly temperatures in the Yorktown discharge canal.

The fishes in this canal not only survived in an atypical environment but at least one species exhibited accelerated growth characteristics never observed in natural waters.

The data presented here, while probably posing as many questions as are answered, do highlight one important facet of thermal ecology. After several years of intensive study on the effects of waste heat, there is still insufficient knowledge to determine what might constitute a beneficial or detrimental thermal alteration involving the entire ecosystem. Knowledge of the ecosystem is needed before we apply numbers based on fragmentary data and force industry into costly, unproductive, and biologically unjustified decisions at the expense of society.

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# Growth Rates of Subadult Largemouth Bass at 24 to 35.5°C

C. C. Coutant  
D. K. Cox

## ABSTRACT

Subadult largemouth bass, *Micropterus salmoides*, weighing 26 to 243 g were held at 13 temperatures from 24.0 to 35.5°C. Bass were fed live adult fathead minnows, *Pimephales promelas*, *ad libitum* and weighed at weekly intervals to determine growth rates. Growth was fastest at 26 to 28°C, and zero growth was near 35.5°C. The temperature of zero growth was about 1.0°C less than the ultimate incipient lethal temperature reported in the literature.

Studies that relate growth and food-conversion efficiency of some fish species to temperature (e.g., Brett, Shelbourn, Schoop, 1969) show that growth responses may provide useful indicators of stressful high temperatures (Coutant, 1972). Growth occurs when excess energy is available after the immediate needs of temperature-dependent maintenance metabolism have been met. The growth-temperature relationships, however, are highly dependent on the quantity and quality of food (Brett, Shelbourn, Schoop, 1969).

This study defines the pattern of growth responses at various constant temperatures, principally in the upper portion of the tolerance range, by well-fed subadult largemouth bass, *Micropterus salmoides*. We hypothesized that the pattern would resemble that found by Brett, Shelbourn, and Schoop (1969) for sockeye salmon in which there was a broad temperature range of maximal growth rate and a higher temperature somewhat below the lethal level at which growth ceased. We sought especially to determine the optimum temperature range for growth of bass and the temperature of zero growth.

## METHODS

In 1972 small largemouth bass were raised in a quarter-acre pond from adults collected from Watts Bar Reservoir, Tennessee River, Tennessee. Groups of bass brought into the laboratory at four different times between October 1972 and August 1973 were divided among four compartments (one fish each) in each of four rectangular tanks measuring 208 by 55 cm (inside) with water depths of 40 cm. Continuous water flow (~4 liters/min) entered one end of each tank and left via a standpipe at the other

end. Temperatures were controlled by mixing valves that blended heated (37°C) and cool (15°C) well water entering each tank. Tests were conducted at 13 temperatures between 24 and 35.5°C ( $\pm 0.5^\circ\text{C}$ ). Two tests were made at 31°C.

Fathead minnows, *Pimephales promelas*, constituted the principal food. These were occasionally replaced by pond-raised goldfish, *Carassius auratus*, when the supply of minnows became temporarily exhausted. We attempted to keep prey fish available to each test bass at all times. Minnows died rapidly at temperatures above 34°C; consequently bass at high temperatures were fed to satiation twice daily.

Bass were acclimated to the test temperatures for at least 2 weeks. Laboratory lighting was maintained at seasonal photoperiod by an outside photoactivated switch. Test growth periods ranged from 18 to 66 days. Bass were weighed weekly to the nearest gram following anesthesia in MS-222 (Sandoz). Weekly weighings of each bass were combined for the entire growth period. Weights were converted to common units of percentage change in weight per fish per day.

## RESULTS AND DISCUSSION

Smaller bass, tested in the fall of their first year, grew considerably faster at comparable temperatures than did larger fish tested the next spring and summer (Fig. 1, Table 1). The results therefore represent portions of two temperature-growth curves, with the magnitude of specific growth rate showing significant size (age) dependence (student's *t*-test,  $P \leq 0.05$ ). Brett, Shelbourn, and Schoop (1969) and Shelbourn, Brett, and Shirahata (1973) noted that, with increasing age, growth continues to slow down so that optimum temperatures become progressively less apparent.

The pattern of growth rate vs. temperature was similar to the pattern shown for sockeye salmon by Brett, Shelbourn, and Schoop (1969), although the optimum and zero growth for bass occurred at higher temperatures. In

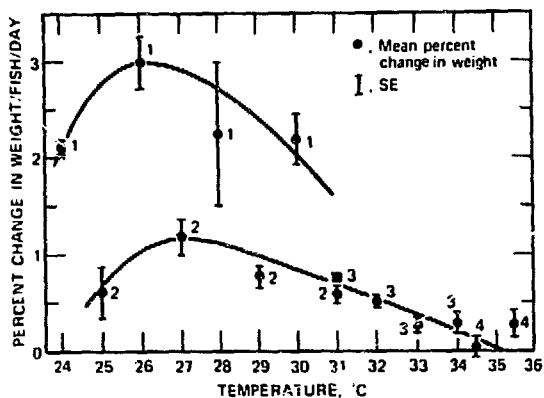


FIG. 1 Growth-temperature relationship for subadult largemouth bass. The numbers 1, 2, 3, and 4 represent data sets for October–November 1972; February–March, April–June, and August–September 1973, respectively. The first data set for younger and smaller bass (~40 g) produces a curve with greater magnitude of specific growth rate when compared to the curve for larger bass (~100 to 200 g) of the other three data sets.

our tests growth was fastest in the 26 to 28°C range. Average growth rate decreased progressively at temperatures above 28°C until growth ceased near 35.5°C. Our results were quite variable at high temperatures. Individuals held at 35.5°C still grew, whereas individuals held at 34 and 34.5°C grew less than at 35.5°C or not at all. Thus our estimate of the temperature of zero growth is not precise. Other authors have also reported difficulty obtaining consistent feeding at temperatures near the boundary of the upper tolerance limit (Brett, 1952; Brett, Shelbourn, and Schoop, 1969). Our estimated 35.5°C for zero growth is 0.9°C less than the ultimate incipient lethal temperature for

largemouth bass (36.4°C) reported by Hart (1952). Brett, Shelbourn, and Schoop (1969) reported zero growth for well-fed sockeye salmon at 23.5°C, 1.5°C below the lethal temperature of 25°C.

Largemouth bass fry studied by Strawn (1961) exhibited a growth-temperature pattern similar to the results shown here. Recalculation of Strawn's data on fish length vs. water temperature for the 11 days over which he grew fish at 17.5, 20.0, 22.5, 25.0, 27.5, and 30.0°C revealed maximum growth rate (millimeters/day) near 27.5°C. Strawn noted some growth of fish held at 32.5°C, but he presented no data beyond 30°C.

A similarity between the optimum temperatures for growth and the temperatures selected by fish in thermal gradients has been observed in some species (Brett, 1971). The optimum thermal range for largemouth bass growth in this study corresponds well with the summertime distribution between 25.5 to 30°C shown by Dendy's (1948) gill-netting studies in Norris Reservoir, Tennessee, and with the 25.5 to 29°C range observed by temperature-sensing ultrasonic transmitters in adult largemouth bass in summer-temperature regimes in small lakes near Oak Ridge, Tenn. (Coutant, 1975). Ferguson (1958), however, cites 30 to 32°C as temperatures preferred by largemouth bass fry on the basis of unpublished data in the files of the Ontario Fisheries Research Laboratory, Toronto, accredited to F. E. J. Fry, McCauley and Read (1973) showed that young yellow perch selected warmer water than adults, and such a size or age difference may account for the especially high preferred temperatures of young bass noted by Ferguson.

The upper limiting temperature for largemouth bass over sustained periods (>1 week) can be estimated from these data by two methods. Both assume that fish populations will become unhealthy when the growth rate of

TABLE 1 Growth Rates of Subadult Largemouth Bass at Constant Temperatures.

| Temperature, °C*  | No. of fish | Initial weight, g† | Growth period, days | Average growth rate‡ |
|-------------------|-------------|--------------------|---------------------|----------------------|
| 24 <sub>1</sub>   | 3           | 42.6 ± 7.85        | 31                  | 2.11 ± 0.0745        |
| 25 <sub>1</sub>   | 4           | 145 ± 51.1         | 42                  | 0.600 ± 0.264        |
| 26 <sub>1</sub>   | 3           | 36.9 ± 13.3        | 31                  | 2.99 ± 0.277         |
| 27 <sub>2</sub>   | 4           | 118 ± 13.0         | 42                  | 1.17 ± 0.176         |
| 28 <sub>1</sub>   | 3           | 44.2 ± 10.6        | 31                  | 2.24 ± 0.745         |
| 29 <sub>1</sub>   | 4           | 127 ± 27.6         | 42                  | 0.756 ± 0.098        |
| 30 <sub>1</sub>   | 3           | 44.2 ± 10.6        | 31                  | 2.18 ± 0.257         |
| 31 <sub>2</sub>   | 4           | 111 ± 21.4         | 42                  | 0.591 ± 0.0995       |
| 31 <sub>3</sub>   | 3           | 168 ± 27.6         | 66                  | 0.704 ± 0.0256       |
| 32 <sub>3</sub>   | 3           | 197 ± 28.6         | 66                  | 0.507 ± 0.0598       |
| 33 <sub>3</sub>   | 3           | 193 ± 23.4         | 66                  | 0.263 ± 0.0655       |
| 34 <sub>3</sub>   | 4           | 146 ± 29.9         | 66                  | 0.123 ± 0.104        |
| 34.5 <sub>4</sub> | 8           | 96 ± 21.1          | 18                  | 0.028 ± 0.109        |
| 35.5 <sub>4</sub> | 7           | 100 ± 8.90         | 18                  | 0.270 ± 0.143        |

\*Temperatures with common subscripts were run simultaneously (1 October–November; 2 February–March; 3 April–June; 4 August–September).

†Mean ± standard deviation.

‡Percent weight increase per fish per day, mean ± standard error of mean.

individuals declines at temperatures much above the optimum. Coutant (1972) suggested that the upper limiting temperature be defined as the temperature midway between the optimum and zero growth. On the basis of the data in this report, that estimate of an upper limiting temperature would be about 31.3°C. The report *Water Quality Criteria 1972* (National Academy of Sciences/National Academy of Engineering, 1973) recommended one-third the temperature range between the optimum and the ultimate incipient lethal temperature. The estimated limiting temperature in that case would be 30.1°C. The exact relationships of such laboratory-determined physiological measurements as growth rates to the productivity of fish populations in nature remain unclear, however, and probably depend on the specific characteristics of a site.

The results of this study suggest that (1) the pattern of largemouth bass growth rate with temperature resembles the general form shown for sockeye salmon, (2) there are characteristic temperatures for optimum and zero growth of largemouth bass, and (3) the temperature for optimum growth coincides with published behavioral preferences in the field.

## ACKNOWLEDGMENT

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# Thermal Effluents, Fish, and Gas-Bubble Disease in Southwestern Lake Michigan

Robert G. Otto

## ABSTRACT

Dissolved oxygen and nitrogen levels determined for intake and discharge waters at the Commonwealth Edison Company's Waukegan Generating Station, Waukegan, Ill., during January–March 1972 indicated that heating of Lake Michigan waters during condenser passage at Waukegan and at the nearby Zion Station will result in gas saturation levels as high as 130%. A review of published literature and observations of fish held in discharge water at the Waukegan Generating Station suggested that such levels may cause gas-bubble disease in local fish populations.

Gas supersaturation tolerance levels were determined in the laboratory for yellow perch and rainbow trout. Yellow perch were unaffected by saturation levels as high as 115%. The 8-day median lethal saturation level was 126%. Trout were unaffected by saturation levels of 110% or below. The estimated 8-day median lethal level was 119%.

Species of fish which frequent the Waukegan and Zion discharge areas during winter months were determined by electroshocking, trawling, and gill netting. Heated effluents from the two power facilities affect different groups of fish. Waukegan Station, with its onshore, canal-type discharge and rapid mixing characteristics, influences primarily such large, salmonid fish as rainbow trout, brown trout, and coho salmon. Zion Station, with an offshore, jet-type discharge, affects the temperature–gas saturation regime for waters occupied primarily by rainbow smelt.

Fish captured in the vicinity of the Waukegan discharge were examined for symptoms of gas-bubble disease. With the exception of carp, resident to the canal proper, no such symptoms were observed. Absence of symptoms indicates that fish are not remaining in the canal long enough to develop the disease.

Gas-bubble disease is an affliction of aquatic organisms, particularly fish, resulting from exposure to waters supersaturated with dissolved gases. The disease is commonly diagnosed by the presence of macroscopic bubbles in the mucous coat, gills, fins, or roof of the mouth. Pathological changes in the liver, spleen, kidney, intestine, and muscle of fishes have also been described (Pauley and Nakatani, 1967). A detailed description of the cause, symptoms, and mechanics of the disease is given by Marsh and Gorham (1905). Briefly, when fishes are exposed to waters super-

saturated with dissolved gases, their body fluids also become supersaturated. As the gases leave solution, bubbles form in the blood and tissues of the fish, accumulating in capillaries or other constrictions in the circulatory system. The end result is a restriction of the flow of blood, and death or incapacitation eventually occurs due to localized anoxia and subsequent necrosis of vital organs or tissues.

Several naturally occurring mortalities of fish due to gas-bubble disease have been recorded. Woodbury (1941) observed mortalities of black crappies (*Pomoxis nigromaculatus*), walleye (*Stizostedion vitreum*), northern pike (*Esox lucius*), and various minnows (family Cyprinidae) in Lake Waubesa, Wisconsin. Renfro (1963) reported the death of large numbers of sea trout (*Cynoscion nebulosus*) and various other marine species in Galveston Bay, Texas. In each of these cases, oxygen supersaturation due to intense photosynthetic activity by algae appeared to be responsible for the disease.

Numerous occurrences of gas-bubble disease in hatchery or aquarium systems have been reported. Most of these occurrences have been traced to faulty design of water supplies whereby leaky water pipes or pumps allowed air to be drawn into the system or to the use of artesian water supplies supersaturated with dissolved nitrogen (Dannevig and Dannevig, 1950; Henly, 1952; Harvey and Smith, 1961; Peterson, 1971; Rucker and Hodgeboom, 1953; Matsue, Egusa, and Saeki, 1953). Similarly gas-bubble disease has been a serious problem in waters below hydroelectric facilities, particularly on the Columbia River system (Bouck et al, 1970; Ebel, 1969). In such cases, gas supersaturation has resulted from water cascading into plunge basins.

There are several documented incidences of gas-bubble disease related to the heated effluent of a steam generating station. The solubility of gases in water is inversely related to temperature. Since ambient dissolved gas levels in many natural lakes and streams frequently approximate air saturation (particularly during winter months), the process of condenser cooling results in supersaturation. DeMont and Miller (1971) observed the symptoms of gas-bubble disease in 13 species of warmwater fish in the discharge

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canal of the (Duke Power Company) Marshall Steam Station, Lake Norman, North Carolina, during the winter of 1970–1971. The primary species affected were bluegill (*Lepomis macrochirus*), threadfin shad (*Dorosoma petenense*), and white bass (*Morone chrysops*). Further discussions of the Lake Norman situation have been provided by Miller and DeMont (1972) and Adair and Hains (1973). Also, Marcello and Strawn (1972) noted that three marine species developed gas-bubble disease when held in the discharge canal of a Galveston Bay power station.

This report considers possible effects of supersaturation of dissolved gases as related to thermal effluents on fish populations in southwestern Lake Michigan. We had been conducting a variety of laboratory studies on thermal effects at the Commonwealth Edison Company's Waukegan Generating Station and had noted in the course of these studies that fish held in the laboratory in water pumped directly from the station discharge canal developed gas-bubble disease during the winter months (Rice, 1972). The present study was intended to assess the possibilities for occurrence of gas-bubble disease in fish populations residing in the region during the winter months, particularly with regard to the operation of the Zion Station, which was then being constructed 4 miles north of the Waukegan site. Our specific objectives were: to determine what levels of gas supersaturation induce gas-bubble disease in representative species for this area, to determine the species of fish which frequent the Waukegan–Zion discharge area during winter months, and to determine if fish using the existing Waukegan discharge waters are afflicted with gas-bubble disease.

## MATERIALS AND METHODS

### Generating-Station Characteristics

Waukegan Generating Station is located on the southwest shore of Lake Michigan, approximately 50 km north of Chicago, Ill. It is a fossil-fueled (coal) station currently operated primarily for peaking purposes. The station has been in operation for more than 20 years and has had as many as eight operating units. However, four units have been permanently retired, and the current maximum power output is approximately 994 Mw(e). Lake water enters the cooling system via an onshore intake and passes through a settling basin before entering the condensers (Fig. 1). Maximum flow across the condensers is about  $44 \text{ m}^3/\text{sec}$ . The cooling water is heated by a maximum of  $8^\circ\text{C}$  during condenser passage and is returned to the lake via a canal and onshore discharge. The discharge canal is about 500 m long, 30 m wide, and 3 to 4 m deep. The lakefront on either side of the intake and discharge is uninterrupted sand–gravel beach. The bottom slopes gradually from the beachline, reaching a depth of 10 m approximately 1500 m offshore. Commonwealth Edison has modified the terrain at the mouth of the discharge canal to facilitate access, and the area receives extensive use by fishermen.

Configuration of the thermal plume in the lake varies greatly with weather conditions as well as seasonally

(O'Neil, 1972). Almost complete mixing occurs at the mouth of the canal during periods of strong onshore winds, and it is difficult to detect any evidence of a plume regardless of season. During spring, summer, or fall, when winds are directed offshore or are lacking, the warmer water rises to the surface. Thermal mixing is rapid, however, and heat dissipation occurs within a short distance of the discharge point. Under winter conditions the discharge plume may either mix from surface to bottom with lake water or sink beneath the receiving waters (Pipes, Pritchard, and Beer, 1973). A generalized temperature decay curve for this discharge is shown in Fig. 2.

Zion Station is also located on the southwest shore of Lake Michigan and is about 6.5 km north of the Waukegan Generating Station. Zion is a nuclear-fueled station having two 1050-Mw(e) units, and plant operation began in July 1973. Cooling water enters the plant from an intake structure located about 800 m offshore at a depth of approximately 7 m (Fig. 1). A velocity cap on top of the intake structure is submerged about 5 m.

Condenser cooling-water flow across each unit at Zion is approximately  $47 \text{ m}^3/\text{sec}$  with a maximum temperature rise ( $\Delta T$ ) of  $11^\circ\text{C}$ . Cooling water for each unit is discharged to the lake through a separate discharge structure located about 230 m offshore. The discharge structures are separated by a distance of 100 m and are equidistant from the intake structure that passes between them. Cooling water is discharged through a series of rectangular ports and is intended to leave the discharge structure as a 1-m-thick sheet, 22 m wide from each side of the structure and 10 m wide from the offshore end. A generalized temperature decay curve for the Zion discharges is shown in Fig. 2.

### Physical–Chemical Considerations

The solubilities of gases in water have specific values defined by physical laws and are commonly expressed as absorption coefficients: gas volume at standard temperature and pressure dissolved by a unit volume of solvent at the temperature of the experiment and under a partial pressure of the gases of 1 atm. The partial pressure of a gas in a mixture is that pressure which each gas would exert if it alone occupied the volume of a mixture of gases at equal temperature (Dalton's law). For dry air at a pressure of 1 atm (760 mm Hg), the approximate percent compositions and partial pressures are: oxygen, 21% at 160 mm Hg; nitrogen, 78% at 593 mm Hg. Since gases dissolve according to their absorption coefficients and partial pressures, oxygen and nitrogen will enter solution in water from air such that the percentage composition of oxygen increases from 21% in air to about 35% in water, whereas that of nitrogen decreases from 78% in air to approximately 65% in water.

The solubility of gases in water is markedly influenced by temperature, and solubilities of both oxygen and nitrogen decrease as temperature is increased. This change in solubility is not quite linear for either gas and is slightly more rapid for nitrogen.

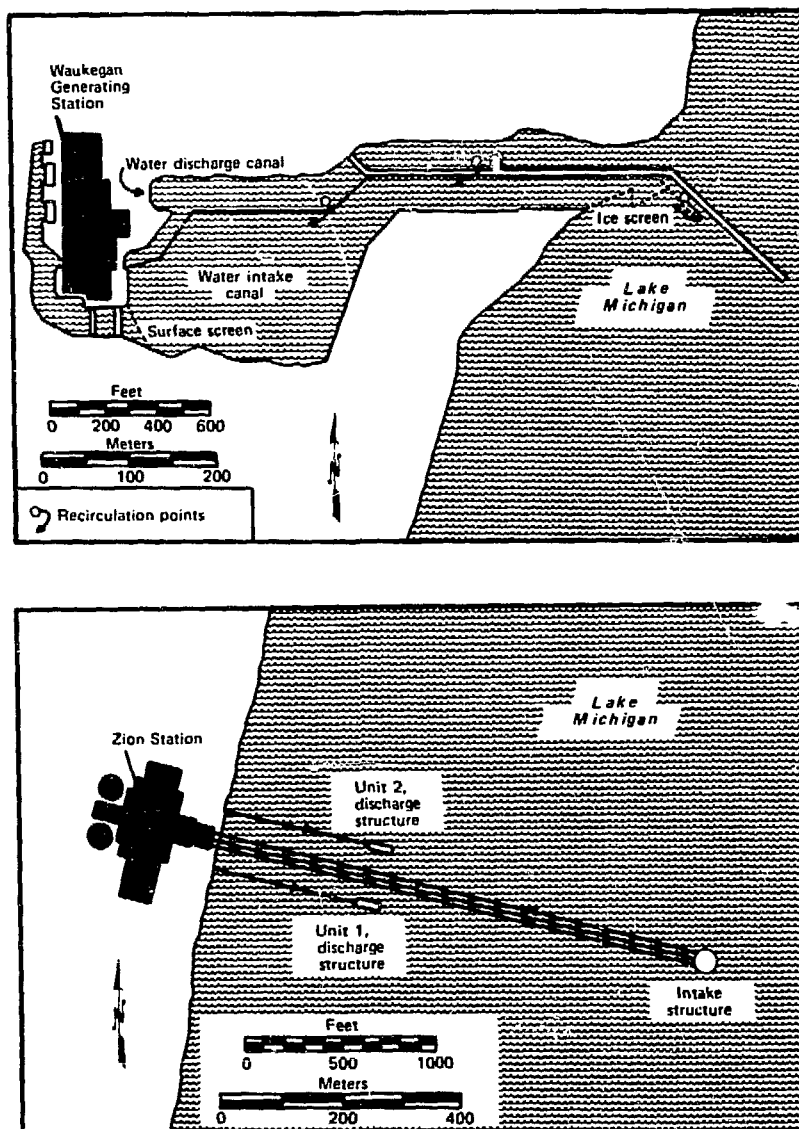


FIG. 1 Cooling-water systems at the Zion and Waukegan generating stations, southwestern Lake Michigan.

Solubility of gases in water is also strongly dependent on pressure such that, as the depth of water increases, solubility of both oxygen and nitrogen is also increased. For example, if a liter of water saturated with oxygen and nitrogen at the water surface ( $P = 1$  atm) is lowered adiabatically to a depth of 10 m ( $P = 2$  atm), saturations of both gases will be reduced (in relative terms) to half. Surface water at 110% saturation would be reduced to 100% saturation at a depth of about 2.5 m; 120% surface saturation would be reduced to 100% at about 4 m; and 130% surface saturation would be reduced to 100% at about 6 m.

Available information on the absolute levels of dissolved nitrogen in Lake Michigan and changes in percent saturation which accompany rapid alterations in water temperature is limited. We would assume, on the basis of ideal gas laws, that both absolute concentrations and relative changes in saturation as related to temperature would approximate those observed for dissolved oxygen for which a great deal of data are available. A limited series of tests was conducted to verify this assumption. Water samples were collected from the intake and discharge canals of the Waukegan Generating Station using a Kemmerer sampler, transferred to 250-ml glass-stoppered bottles, iced,



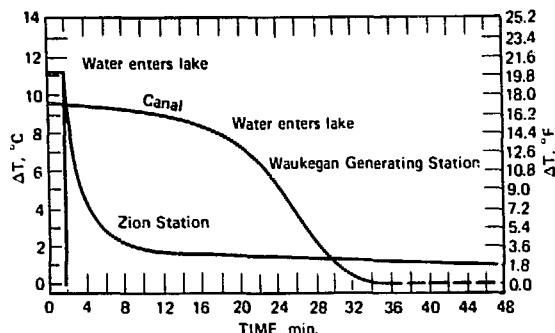


FIG. 2 Generalized temperature decay curves for the cooling-water discharges of the Zion and Waukegan generating stations.

and taken to the laboratory for analysis. Water temperature was recorded at the time of collection for future calculation of saturation values. A Fisher-Hamilton Model 29 gas partitioner having two chromatographic columns in series was used to measure concentrations of dissolved oxygen and nitrogen following the procedure of Swinnerton, Linnenbom, and Cheek (1962). The first column was 6.4 by 76 mm, filled with 30% hexamethyl phosphoramide on 60- to 80-mesh Chromosorb P. The second was 9.5 by 198 mm, filled with molecular sieve 13X. A helium carrier and a thermal conductivity detector were used. Each sample was analyzed in triplicate. The third injection into the chromatograph was made within 30 min of the first. There were no significant changes in oxygen or nitrogen concentrations between the first and last determinations for any sample. All peak areas were measured with a compensating polar planimeter.

### Laboratory Studies

Effects of gas supersaturation on survival of yellow perch (*Perca flavescens*) and rainbow trout (*Salmo gairdneri*) were studied. Both species are common to Lake Michigan and occur in the Zion-Waukegan region during

the winter period. In recent years the perch population has declined sharply in this region, and few individuals were taken in our field studies. Natural populations of fish, however, fluctuate widely on an annual basis, and the perch remains a species of importance in this region. Rainbow trout occur in the Zion-Waukegan region in considerable numbers during winter months. This species appears to remain along the shoreline during winter and is therefore in immediate proximity to heated effluents for both the Zion and Waukegan generating stations.

Fish used were obtained from commercial dealers since wild individuals were difficult to obtain during the winter months. One thousand yellow perch, ranging in length from 10 to 15 cm, were purchased in January from Fenders Fish Hatchery, Baltic, Ohio. Five hundred rainbow trout, ranging in length from 14 to 21 cm, were obtained in mid-February from Boone Springs Trout Farm, McHenry, Ill. Both stocks were maintained in the laboratory in 800-liter fiber glass tanks having a continuous flow of water from the Waukegan Generating Station intake canal. Lighting was supplied by banks of fluorescent tubes controlled by timers to approximate a natural photoperiod. Both stocks received an artificial diet (trout chow) daily and remained in good condition. Testing began in late March and continued through April 1972.

Controlled levels of gas supersaturation were achieved by using the differential solubility of gases in waters of various temperatures. Even though gas supersaturation can also be easily achieved using a pressurized system, previous studies (Speece and Leyendecker, 1969) indicate that the two methods may give different results regarding lethal limits for fish. The method used in this study duplicates the physical process that causes gas supersaturation in a steam generating facility. Ambient lake water, at approximately 100% air saturation, was pumped from the intake canal of the Waukegan Generating Station and equilibrated to constant temperature by passing it through a coil immersed in a refrigerated bath and aerated (Fig. 3). The water was then circulated through a second coil immersed in a heated bath, raising the temperature and decreasing the carrying

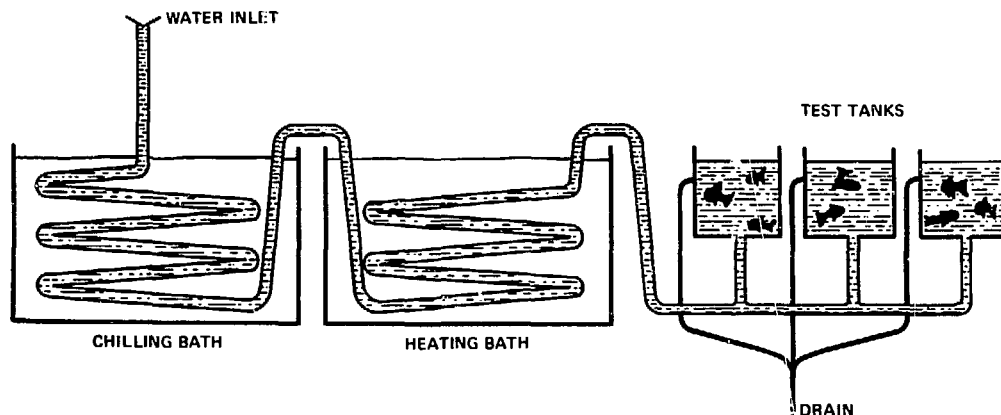


FIG. 3 Gas supersaturation apparatus.

capacity for dissolved gases accordingly. This heated, gas-supersaturated water was delivered to three 22-liter test tanks.

The apparatus was first adjusted to deliver water at the desired level of gas saturation, ranging from 110 to 130%. Groups of 10 trout or perch were then selected at random from the holding stocks and transferred to the test tanks. Fish transferred from ambient lake water to gas supersaturated water experienced an increase in temperature similar to that which a fish in the natural environment would experience when moving from ambient lake conditions to a comparable level of gas supersaturation in a thermal plume.

The fish were observed for 5 days or until 100% mortality had occurred. An attempt was made to note the onset of any external symptoms of gas-bubble disease including bubbles in the fins and gills, exophthalmus or popeye, and hemorrhaging around eyes, jaws, and gills. Such observations were restricted by the turbidity of the water and the possibility that excessive handling of test fish might influence the survival results. Actual times to death for individual fish were recorded to the greatest extent possible. However, in computing results, we reduced all mortality data to 24-hr increments in accord with standard bioassay procedure (American Public Health Association, 1960).

Dissolved oxygen levels in all test tanks were measured at regular intervals following the standard Winkler techniques. Levels of nitrogen supersaturation were computed from the corresponding values for dissolved oxygen. The rate of turnover of water in the test tanks was more than adequate to compensate for any reduction in oxygen levels due to respiration by the fish.

## Field Studies

Daily movements of fish into and out of the Waukegan discharge canal were determined by gill net sets at the mouth of the canal. An experimental-type net having duplicate 8-m (25 ft) panels of 1.3, 2.5, 3.8, 5.1, and 6.4 cm (square) mesh was used. The net was set for successive 6-hr periods, starting at 0600 hr and ending at 0600 hr the following day. Four sampling series were conducted (Jan. 6–7, Feb. 18–19, Mar. 9–10, and Apr. 5–6). Fish collected were identified as to species, weighed, and measured. Their position in the net indicated whether they were entering or leaving the canal at the time of capture. All specimens were examined for symptoms of gas-bubble disease.

Fish in the discharge canal proper were captured by electroshocking using a 230-volt boom shocker. Three attempts to sample by this method were made (Jan. 19 and 20 and Apr. 3). Because the conductivity of water is greatly reduced at low temperatures, this method proved unsatisfactory for capturing large numbers of fish. Those captured were examined for symptoms of gas-bubble disease.

Fish occupying offshore waters were sampled by trawling within three preselected transects. The first tran-

sect (W) was located directly offshore of the Waukegan Generating Station discharge, extending from the mouth of the canal perpendicular to the shoreline for a distance of 10 km. The second (D) extended from the Dead River region of Illinois Beach State Park offshore for a distance of 10 km. A third transect (Z) originated at the Zion Station and extended 7 km into the lake. The total area sampled was bounded by the Waukegan (W) and Zion (Z) transects and encompassed an area of 67 km<sup>2</sup>. Each transect was divided into sampling zones according to depth. Trawls were made at depths of 4, 6, 10, 20, 30, and 40 m with some minor variation due to bottom configuration.

A semiballoon trawl having a 13-m headrope, a 17-m footrope, and a 1.3-cm-mesh cod end was used. The trawl was towed on the bottom for 10 min at a speed of 2½ knots at each station. All tows were made parallel to the shoreline. Three series of trawls were made (Jan. 3, Feb. 14, and Mar. 30). All fish captured were identified as to species, weighed, and measured.

## RESULTS AND DISCUSSION

### Levels of Dissolved Gases

Relative (%) concentrations of oxygen and nitrogen in samples from the Waukegan Generating Station intake and discharge canals were determined on a total of 18 samples collected on three different occasions (Table 1). No appre-

TABLE 1 A Comparison of Oxygen and Nitrogen Saturation Levels for Lake Michigan Waters at the Waukegan Generating Station, Waukegan, Ill., as Determined by Gas Chromatography. Each Value Is an Average of Three Injections  $\pm$  the Standard Deviation

| Sampling date | Temp., °C | Oxygen                     |               | Nitrogen                   |               |
|---------------|-----------|----------------------------|---------------|----------------------------|---------------|
|               |           | Peak area, mm <sup>2</sup> | Saturation, % | Peak area, mm <sup>2</sup> | Saturation, % |
| Intake        |           |                            |               |                            |               |
| 4/03/72       | 3.0       | 18.9 ± 0.1                 | 97            | 35.4 ± 0.5                 | 97            |
|               | 3.0       | 19.0 ± 0.5                 | 98            | 35.8 ± 0.5                 | 99            |
|               | 3.0       | 19.0 ± 0.3                 | 98            | 36.3 ± 0.2                 | 101           |
| 4/11/72       | 9.0       | 15.8 ± 0.5                 | 109           | 30.1 ± 1.0                 | 112           |
|               | 9.0       | 15.6 ± 0.2                 | 107           | 29.8 ± 0.3                 | 110           |
|               | 9.0       | 15.5 ± 0.3                 | 107           | 29.7 ± 0.1                 | 110           |
| 4/14/72       | 12.0      | 13.3 ± 0.1                 | 98            | 24.7 ± 0.4                 | 97            |
|               | 12.0      | 13.3 ± 0.4                 | 98            | 25.0 ± 0.3                 | 98            |
|               | 12.0      | 13.5 ± 0.2                 | 100           | 25.1 ± 0.4                 | 99            |
| Discharge     |           |                            |               |                            |               |
| 4/03/72       | 12.0      | 18.5 ± 0.2                 | 119           | 35.2 ± 0.1                 | 120           |
|               | 12.0      | 18.4 ± 0.3                 | 119           | 34.9 ± 0.3                 | 119           |
|               | 12.0      | 18.6 ± 0.1                 | 120           | 35.3 ± 0.1                 | 121           |
| 4/11/72       | 17.0      | 15.1 ± 0.1                 | 124           | 28.5 ± 0.2                 | 125           |
|               | 17.0      | 15.9 ± 0.2                 | 132           | 28.5 ± 0.2                 | 124           |
|               | 17.0      | 15.4 ± 0.4                 | 127           | 29.2 ± 0.4                 | 127           |
| 4/14/72       | 20.0      | 13.1 ± 0.3                 | 115           | 25.1 ± 0.4                 | 115           |
|               | 20.0      | 12.6 ± 0.4                 | 111           | 23.6 ± 2.0                 | 109           |
|               | 20.0      | 12.7 ± 0.1                 | 111           | 23.5 ± 0.2                 | 108           |

cial differences in absolute concentrations between intake and discharge waters were noted for either gas on any one of the three occasions. Differences in relative (%) concentration of each gas between intake and discharge samples demonstrate the effects of typical operating conditions for the Waukegan Generating Station ( $\Delta T = 8^{\circ}\text{C}$ ) on dissolved gas levels at surface pressure and conform reasonably well to predicted levels. During midwinter when lake waters are coldest ( $0^{\circ}\text{C}$ ) and saturated with gases, supersaturation levels in the Waukegan discharge canal will be as high as 125%, whereas cooling waters from Zion Station may exceed 132% saturation (at surface pressure).

### Laboratory Observations

Yellow perch were unaffected by levels of gas supersaturation as high as 115% (Table 2). At saturation levels of 119% and 124%, test groups experienced 10% and 40% mortality, respectively. Surviving fish exhibited no external symptoms of gas-bubble disease at the end of the 5-day test period. Eighty percent mortality occurred in the test group exposed to 130% saturation. Surviving fish in this group had gas bubbles in the fins but no evidence of exophthalmos or hemorrhaging around the eyes, mouth, or gills. The 5-day median tolerance limit was approximately 126% saturation for this species.

TABLE 2 Percent Survival of Yellow Perch Exposed to Gas Supersaturation

| Gas saturation, % | Survival, %, after the indicated exposure times |       |       |       |        |        |        |        | Total mortality, % |
|-------------------|-------------------------------------------------|-------|-------|-------|--------|--------|--------|--------|--------------------|
|                   | 24 hr                                           | 48 hr | 72 hr | 96 hr | 120 hr | 144 hr | 168 hr | 192 hr |                    |
| 110               | 100                                             | 100   | 100   | 100   | 100    | 100    | 100    | 100    | 0                  |
| 115               | 100                                             | 100   | 100   | 100   | 100    | 100    | 100    | 100    | 0                  |
| 119               | 100                                             | 100   | 100   | 90    | 90     | 90     | 90     | 90     | 10                 |
| 124               | 100                                             | 90    | 90    | 90    | 70     | 60     | 60     | 60     | 40                 |
| 130               | 90                                              | 90    | 50    | 40    | 40     | 20     | 20     | 20     | 80                 |

Rainbow trout were somewhat more susceptible to gas-bubble disease than were the yellow perch (Table 3). Fish exposed to levels of 110% saturation or below did not experience mortalities or exhibit symptoms of the disease. Mortalities were observed at test levels of 115% and greater, the estimated 5-day median tolerance limit being 119% saturation.

Turbidity of the water made detailed observations on the relationship between development of external symptoms and mortalities impossible. However, a correlation was noted between level of saturation and time to death, both trout and perch succumbing more rapidly as the relative concentration of dissolved gases increased above lethal limits. Behavior of test fish also indicated that avoidance of extreme levels of gas supersaturation may exist for both species. Fish exposed to nonlethal or borderline lethal conditions remained at the bottom of the test tanks.

TABLE 3 Percent Survival of Rainbow Trout Exposed to Gas Supersaturation

| Gas saturation, % | Survival, %, after the indicated exposure times |       |       |       |        |        |        | Total mortality, % |
|-------------------|-------------------------------------------------|-------|-------|-------|--------|--------|--------|--------------------|
|                   | 24 hr                                           | 48 hr | 72 hr | 96 hr | 120 hr | 144 hr | 168 hr |                    |
| 110               | 100                                             | 100   | 100   | 100   | 100    | 100    | 100    | 0                  |
| 115               | 100                                             | 100   | 100   | 90    | 90     | 90     | 90     | 10                 |
| 118               | 60                                              | 60    | 60    | 60    | 60     | 60     | 60     | 40                 |
| 122               | 50                                              | 0     | 0     | 0     | 0      | 0      | 0      | 100                |
| 125               | 0                                               | 0     | 0     | 0     | 0      | 0      | 0      | 100                |

However, at higher, lethal levels of gas supersaturation, fish were clearly agitated, and it was necessary to cover the tanks to prevent the fish from jumping out. Similar tests in which temperature was increased without accompanying supersaturation indicate that this agitated state cannot be attributed to the change in water temperature.

There are no data available from other sources on susceptibility of yellow perch to gas-bubble disease. However, several others have considered the effects of gas supersaturation on rainbow trout. Rucker and Hodgeboom (1953) described a hatchery spring that delivered water 120% saturated with nitrogen and 80% saturated with oxygen. Trout fry were unable to tolerate nitrogen saturation levels of greater than 115%; thus the results compare closely with those of the present study. Rucker and Tuttle (1948) had earlier tentatively concluded that saturation levels as high as 110% were not excessive for this species but that 115% would be lethal. Shirahata (1966) conducted a detailed study on effects of nitrogen supersaturation on rainbow trout fry. Maximum tolerance levels of 130% were observed for fry in the early portion of the hatching-swim-up period. The tolerable level then decreased to 120% saturation as development progressed through absorption of the yolk sac, 110% shortly after feeding, and less than 110% for more advanced fry. The slightly lower tolerance limits observed by Shirahata may relate to his procedure of maintaining supersaturated water under pressure. A later comparison was made of the effects of nitrogen supersaturation as obtained in a pressure system with those observed using a nonpressure system (the present study involves a nonpressure system) for rainbow trout and goldfish (*Carassius auratus*) (Speece and Leyendecker, 1969). Both species were affected at lower levels of saturation when the pressure system was used. Rainbow trout were more susceptible to gas-bubble disease than were goldfish. Trout fingerlings were unable to tolerate 105% supersaturation of nitrogen for more than  $4\frac{1}{2}$  days, whereas goldfish survived 140% supersaturation for 48 hr and 105% for 5 days when tested in the pressurized system.

### Field Observations

Eight species were taken by gill netting at the mouth of the Waukegan discharge canal (Table 4). Three species

TABLE 4 Fish Collected by Gill Netting at the Mouth of the Waukegan Generating Station Discharge Canal, January–April 1972

| Common name     | Scientific name              | Number collected on |             |         |         |
|-----------------|------------------------------|---------------------|-------------|---------|---------|
|                 |                              | January 6           | February 18 | March 9 | April 5 |
| Rainbow trout   | <i>Salmo gairdneri</i>       | 4                   | 7           |         | 3       |
| Brown trout     | <i>Salmo trutta</i>          | 2                   | 4           |         | 9       |
| Brook trout     | <i>Salvelinus fontinalis</i> |                     | 1           |         | 2       |
| Coho salmon     | <i>Oncorhynchus kisutch</i>  |                     |             | 2       | 19      |
| Northern pike   | <i>Esox lucius</i>           |                     | 1           |         |         |
| Spottail shiner | <i>Notropis hudsonius</i>    |                     | 2           | 5       | 12      |
| White sucker    | <i>Catostomus commersoni</i> |                     | 2           |         | 15      |
| Carp            | <i>Cyprinus carpio</i>       |                     | 7           |         | 1       |
| Total           |                              | 6                   | 24          | 7       | 61      |

(spottail shiners, carp, and white suckers) inhabit the intake and discharge canals throughout the year. The remaining fish taken were mature salmonids and a single northern pike. Movements into and out of the discharge canal in January were restricted to large trout of the genus *Salmo*. Both the rainbow and brown trout are popular among local fishermen, and their presence in and around the Waukegan discharge is further evidenced by the active sport fishery that exists throughout winter and spring.

An increase in both number of species and number of individuals taken occurred in February. Half the fish captured were mature salmonids. The remainder were species that reproduce in the spring, and their occurrence at the canal mouth may relate to prespawning movements. A marked reduction in catch occurred in March. This is believed to be a sampling artifact rather than a true decrease in species abundance. Power requirements at the time of sampling were particularly high, and the station was operating all units at full capacity. Under such conditions, water velocity in the discharge canal is considerably more rapid than normal, a situation which makes it impossible to keep the bottom of the gill net down on the south side of the canal and which thus provides a zone of safe passage for the fish. However, this sampling series did demonstrate the appearance of coho salmon in inshore waters. The April sampling series provided the largest catch. More than 60% of the total was taken at this time, with the increase in abundance primarily due to the capture of large numbers of coho salmon, spottail shiners, and white suckers.

A diurnal periodicity in the movements of fish into and out of the discharge canal was noted (Fig. 4), the major proportion of fish being taken at night. These differences might be attributed to variations in efficiency of the gill net with time of day. However, the mouth of the canal is continuously lighted by floodlights, and the water is very turbid during the winter months. There was no obvious pattern to the direction of fish movement into or out of the canal during the various sampling periods. Direction was noted for 50 individuals, of which 20 were entering the canal and 30 were exiting. The majority of the fish

collected were alive at the time the net was lifted, and each was examined for symptoms of gas-bubble disease. Particular attention was given to these criteria: exophthalmus; bubbles in fins, gills, or surface mucous; and hemorrhaging around the eyes, mouth, or gills. None of the fish exhibited such symptoms.

Eight species were captured by electroshocking in the discharge canal. Four species (emerald shiners, golden shiners, gizzard shad, and green sunfish) were not taken in the gill net samplings. Electroshocking is not an effective method of sampling during winter, cold water being a poor conductor of electricity, and I assume that those fish captured represent only a small fraction of the fish actually present. A number of carp taken had large gas bubbles on the body surface. The bubbles were distributed over the body surface and seemed to be just under the skin. There were few bubbles in the fins and no indication of gas in the gills, roof of mouth, or around the eyes.

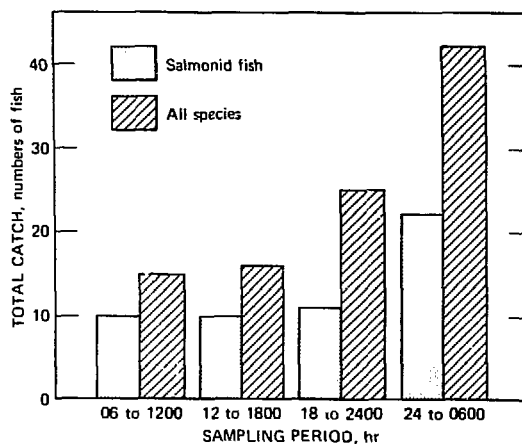


FIG. 4 Numbers of fish taken by gill netting at the mouth of the Waukegan Generating Station discharge canal at various times of day.

The offshore fish community differed markedly from that inhabiting the vicinity of the Waukegan discharge canal. Fourteen species of fish were taken, of which the great majority were rainbow smelt (Table 5). Only two salmonid fish were taken (both lake trout), one 1.3 km offshore from the Waukegan discharge and the other 5.2 km offshore from Zion Station. Four of the fourteen species taken by trawling (carp, gizzard shad, spottail shiners, and white suckers) were also captured by gill

to gas-bubble disease exists. The primary effect of such an aggregation is an enhanced sport fishery for those species, primarily rainbow trout, brown trout, and coho salmon, which come in contact with the two effluents.

## ACKNOWLEDGMENTS

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TABLE 5 Fish Collected by Trawling Offshore in the Zion-Waukegan Region of Southwestern Lake Michigan, January-March 1972

| Common name     | Scientific name               | Number collected on |             |          | Total |
|-----------------|-------------------------------|---------------------|-------------|----------|-------|
|                 |                               | January 3           | February 15 | March 30 |       |
| Alewife         | <i>Alosa pseudoharengus</i>   | 1                   | 0           | 17       | 18    |
| Rainbow smelt   | <i>Osmerus mordax</i>         | 499                 | 63          | 415      | 977   |
| Bloater         | <i>Coregonus hoyi</i>         | 1                   | 1           | 1        | 3     |
| Carp            | <i>Cyprinus carpio</i>        | 0                   | 0           | 1        | 1     |
| Gizzard shad    | <i>Dorosoma cepedianum</i>    | 1                   | 0           | 0        | 1     |
| Lake herring    | <i>Coregonus artedii</i>      | 0                   | 1           | 0        | 1     |
| Lake trout      | <i>Salvelinus namaycush</i>   | 1                   | 1           | 2        | 4     |
| Longnose sucker | <i>Catostomus catostomus</i>  | 0                   | 0           | 2        | 2     |
| Slimy sculpin   | <i>Cotus cognatus</i>         | 25                  | 1           | 22       | 48    |
| Spottail shiner | <i>Notropis hudsonius</i>     | 8                   | 3           | 20       | 31    |
| Stickleback     | <i>Pungitius pungitius</i>    | 3                   | 0           | 0        | 3     |
| Trout perch     | <i>Percopsis omiscomaycus</i> | 1                   | 0           | 1        | 2     |
| White sucker    | <i>Catostomus commersoni</i>  | 0                   | 1           | 3        | 4     |
| Yellow perch    | <i>Perca flavescens</i>       | 2                   | 0           | 10       | 12    |
|                 |                               |                     |             |          | 1107  |

netting or electroshocking in the Waukegan discharge. Six of the remaining 10 species (alewife, rainbow smelt, bloater, slimy sculpin, stickleback, and yellow perch) have been taken in the vicinity of the discharge at other times of year.

We can summarize the field observations by saying that apparently the heated effluents from the two power facilities affect different groups of fish during the period that gas supersaturation is prevalent. Waukegan Generating Station, with its onshore canal-type discharge, influences primarily such large salmonid fishes as rainbow trout, brown trout, and coho salmon moving along the shoreline. Zion Station, with an offshore jet-type discharge, results in changes in the temperature-gas supersaturation regime for waters occupied primarily by rainbow smelt.

Subtle changes in the natural distribution pattern of fish in the Waukegan-Zion region may have occurred as a result of attraction to or avoidance of the heated effluents. Ambient lake temperatures during this period are well below the preferred levels for all species, and fish responding to temperature alone might be expected to aggregate in waters of increased temperature and increased gas-saturation levels. However, all available data indicate that, even if aggregation of fish does occur, no problem related

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# Thermal Tolerance of the Adult Asiatic Clam

J. S. Mattice  
L. L. Dye

## ABSTRACT

Aspects of high- and low-temperature tolerance of the Asiatic clam, *Corbicula manilensis* Philippi, acclimated to temperatures ranging from 5 to 32°C were studied. Groups of 20 adult clams were exposed (1) continuously to high temperatures (24 to 39°C), (2) continuously to lower temperatures (2 to 20°C), or (3) for 30 min to temperatures between 37 and 43°C. Mortality was determined by lack of response to prying open the valves. Exposure to temperatures between 40°C (5°C acclimation) and 43°C (30°C acclimation) for 30 min caused 100% mortality. For continuous exposures, upper tolerance limits (50%) were between 24°C (5°C acclimation) and 34°C (30°C acclimation) and lower tolerance limits were between 2°C (15°C acclimation) and 12°C (30°C acclimation). Relationships between acclimation and resistance temperatures were linear. The upper incipient lethal temperature was 34°C. Use of results at 24 and 48 hr (and 96 hr for long-term heat stress) yielded significant overestimates of tolerance limits.

Since it was first observed in the United States in 1938 (Fox, 1969), the Asiatic clam, *Corbicula manilensis*, has increased rapidly in ecologic and economic importance. Most of the major drainage basins in the United States are now inhabited by *Corbicula* (Ingram, 1959; Sinclair and Ingram, 1961; Fuller and Powell, 1973; Fox, 1973). Where they are abundant, these clam populations are important in nutrient cycling and energy flow (e.g., they serve as food for fish, ducks, raccoons, and humans) (Fox, 1969; Fuji, 1955). The species seems to be a prime candidate for future freshwater aquaculture. Conversely, its ability to infest raw water intakes and canals and to clog power-plant condensers has been recognized by the national press (Anonymous, 1973a; 1973b) as well as the scientific community (Fox, 1969; Anonymous, 1963; Sinclair, 1964; Isom, 1971). In response to the interest in these clams, Sinclair (1971) presented an annotated bibliography of much of the recent literature on *Corbicula*.

Their interaction with raw water consumers and the concomitant necessity for controlling fouling have resulted in the major immediate impact of *C. manilensis* on man. Chlorination, usually continuous chlorination during some part of the clams' reproductive period, has been the most

usual method for reducing fouling by *Corbicula* (Isom, 1971). However, the extreme toxicity of chlorine to freshwater organisms (Brungs, 1973) and the possibility of creating stable toxic chloro-organics (Gehr, et al., 1974) suggest that this practice is not environmentally sound. Other chemicals such as general molluscicides have similar disadvantages (Isom, 1971). Isom (1971) suggested heat as a more environmentally sound alternative and presented preliminary evidence suggesting its efficacy. This potential method can be evaluated only after a thorough examination of the thermal tolerance of the clams.

Thermal means of preventing fouling by *Corbicula* should be aimed at veliger larvae or newly set juveniles, but it is important that we measure thermal tolerances of adults also. First, where fouling has previously occurred, adults, of necessity, must be the targets of treatment. Second, some power-plant fouling is a direct result of adult influx, apparently via water currents that carry them around or under intake screens. Third, the critical size for treatment at power plants is that which will still pass through the condenser tubing, which is about 1.97 cm in inside diameter (Coutant and Kedl, 1975), without clogging. Fourth, adult thermal tolerance in the area of the discharge must play a role in setting the upper boundaries for allowable heat treatment within the facility.

Two studies on aspects of the thermal tolerance of *C. manilensis* have been published previously (Isom, 1971; Habel, 1970). Habel acclimated clams to ~23°C and exposed them for 4 days to temperatures from 29 to 38°C. He found the upper incipient lethal temperature for this acclimation to be 34°C. Isom acclimated clams to 10°C and exposed them to temperatures ranging from 21 to 50°C. All exposures greater than 43°C killed the clams within minutes. Our studies were begun to extend Isom's preliminary work and to obtain thermal-tolerance data that could reasonably be extrapolated to real situations.

## MATERIALS AND METHODS

The Asiatic clams, *C. manilensis*, used in these tests were collected at Mile 23 on the Clinch River, Tennessee. Clams used in most tests ranged from 24 to 35 mm in longest

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dimension [shell length (SL)]; most were about 30 mm. In the cold-tolerance tests, some groups of clams 22 to 24 mm SL were used for comparison. Clams in the larger size group are about 4 years old, and those in the smaller size group are about 2 years old (Sinclair and Isom, 1963).

The clams were held in 190-liter aquariums supplied with constantly flowing well water and at temperatures maintained to  $\pm 0.5^{\circ}\text{C}$ . Acclimation was carried out at a rate of less than  $1^{\circ}\text{C}$  per day to temperatures of 5, 10, 15, 20, 25, 30, and  $32^{\circ}\text{C}$ . Clams did not survive attempted acclimation to 2 or  $35^{\circ}\text{C}$ .

As far as possible, clams held at all acclimation temperatures were tested simultaneously. Clams were held in the laboratory for 4 to 5 weeks before the tests began; this included the time for acclimation as well as 1 to 4 weeks at the acclimation temperature. Testing usually began in the early morning.

During acclimation and testing, clams were fed a powdered mixture of Biorell tropical fish flakes and trout chow. Food placed in each aquarium was calculated to be at least 0.01 gm dry weight of food for each clam per day. The amount of food given was based on (1) maximum growth rate of *Corbicula* calculated from data of Heinsohn (1958), (2) physical characteristics of the aquarium system (water turnover time, number of clams in the tank, etc.), (3) a filtration rate of about 1 liter/hr (Mattice and Nodvin, 1974) and (4) an assumed 50% assimilation.

Though each type of test began and ended in the same way, some of the details differed. Each experimental group consisted of 20 clams held in screened 4-liter plastic containers. Tests were initiated by direct transfer of the containers and clams, without water, from the acclimation tanks to tanks at the test temperatures. The combinations of acclimation and test temperatures are given in Table 1.

TABLE 1 Acclimation and Test Temperature Ranges Used for Measuring Thermal Tolerance of *Corbicula*

| Acclimation temperature, $^{\circ}\text{C}$ | Long-term high temp., $^{\circ}\text{C}$ |      | Long-term low temp., $^{\circ}\text{C}$ |      | High shock temp., $^{\circ}\text{C}$ |      |     |      |
|---------------------------------------------|------------------------------------------|------|-----------------------------------------|------|--------------------------------------|------|-----|------|
|                                             | 1973*                                    |      | 1973†                                   |      | 1974‡                                |      |     |      |
|                                             | Low                                      | High | Low                                     | High | Low                                  | High | Low | High |
| 5                                           | 24                                       | 36   | 2                                       | 5    | 32                                   | 42   | 37  | 43   |
| 10                                          | 24                                       | 36   | 2                                       | 5    | 36                                   | 44   | 37  | 43   |
| 15                                          | ¶                                        | ¶    | 2                                       | 10   | 40                                   | 46   | 37  | 43   |
| 20                                          | 24                                       | 36   | 2                                       | 10   | 40                                   | 48   | 37  | 43   |
| 25                                          | 26                                       | 39   | 2                                       | 15   | 42                                   | 50   | 37  | 44   |
| 30                                          | 30                                       | 39   | 2                                       | 15   | 40                                   | 46   | 37  | 44   |
| 32                                          | 34                                       | 39   | Not tested                              |      | Not tested                           |      | 37  | 44   |

\*Test temperatures were at  $2^{\circ}\text{C}$  intervals except for tests at midrange (i.e., 30, 31, and  $32^{\circ}\text{C}$ ).

†Test temperatures used included 2, 5, 7.5, 10, and  $15^{\circ}\text{C}$  where appropriate.

‡Test temperatures were at  $2^{\circ}\text{C}$  intervals except in the 15 and  $20^{\circ}\text{C}$  acclimation groups for which  $1^{\circ}\text{C}$  test intervals were used.

§Test temperatures were at  $1^{\circ}\text{C}$  intervals from the low to high temperatures listed.

¶Data are not presented; see text for explanation.

Controls were maintained in the acclimation tanks. The tolerance tests included exposing clams to:

1. Long-term stress at temperatures higher than acclimation temperature (long-term heat stress).

2. Long-term stress at temperatures colder than acclimation temperature (long-term cold stress).

3. 30-min shocks at temperatures above incipient lethal temperature (heat shock).

In the third group of experiments, the shock temperature was maintained to  $\pm 0.2^{\circ}\text{C}$  except during the initial 2 to 3 min when variation was  $\pm 0.4^{\circ}\text{C}$ . After the 30-min shock, clams were immediately transferred back to the original acclimation temperature. In all the experiments, mortality was tested at varying intervals; usually several times the first day, twice a day for the first week, and at longer intervals as the mortality rate decreased. When the rate of control deaths equalled the rate of experimental deaths, observations were terminated. The period of observations was from 3 to 5 weeks.

Death of the clams was determined in two ways. If the shell was open, the body was gently prodded (gape test). If the shell was closed, a knife was carefully inserted between the valves and gently twisted (twist test). If no closing response could be observed following either test, the clam was considered dead. In questionable cases the clam was isolated at the appropriate temperature for more careful observation in the next scheduled test.

Most tolerance limits and comparisons were based on median survival times. Median survival times at given test temperatures are calculated by assuming a normal distribution of percent mortality. In this procedure percent mortality (on a probit scale) is plotted against the log of death time (Fry, Hart, and Walker, 1946). In cases where death has also occurred in the controls, the number of control deaths was subtracted from the number of experimental deaths to give a corrected percent mortality at each time interval. A line was fitted by eye through the points to estimate the time to 50% death (see Fig. 1). The same graphic methods were used to determine the test temperatures causing 100% mortality in the heat-shock experiment. In the  $5^{\circ}\text{C}$  acclimation group, 15% of the controls died in less than 2 hr. In this case only, the maximum corrected experimental mortality was 85%. We used this value as our closest estimate to 100% mortality. The experiment was ended when the corrected percent mortality remained the same for 2 weeks. At this point we assumed that the effect of temperature on mortality was ended.

Since it is common to report temperatures yielding 50% mortality after 24, 48, and 96 hr, we made an analysis to compare the validity of the results obtained after each time interval. Regression lines were fit across acclimation temperatures for the incipient lethal temperatures obtained for the 24-, 48-, and 96-hr, and final time intervals. The regressions were compared to determine whether a single linear regression would adequately describe the relation for each of the different exposure times and the total



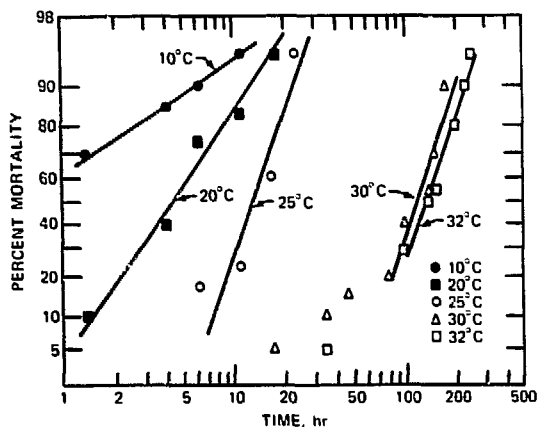


FIG. 1 Results of one long-term high-temperature test. Percent mortality (in probits) is plotted against log time for groups of clams acclimated to temperatures of 10, 20, 25, 30, and 32°C and exposed to 36°C.

effect. This was done by comparing the residual sums of squares and calculating an F statistic (Draper and Smith, 1966).

## RESULTS

### Long-Term Heat Stress

At each acclimation temperature a decrease in exposure temperature increased resistance time (Fig. 2). Although resistance times ranged over two or more orders of magnitude, the decreasing test temperature ultimately reached a temperature that failed to cause 50% mortality no matter how long it was applied. This limit is indicated by the oblique line at the right in Fig. 2. This line cuts through times of about 300 to 500 hr. Approximately

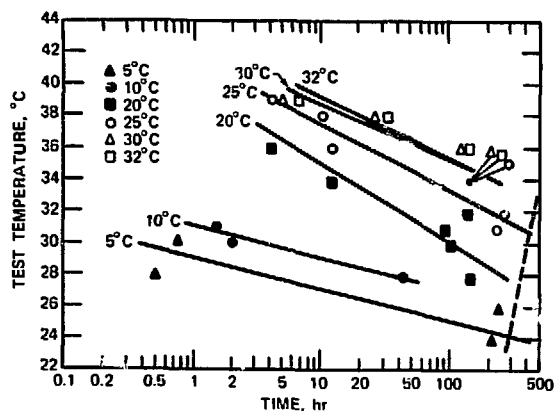


FIG. 2 Summary of time to 50% mortality (on a log scale) plotted against test temperature. Lines are calculated regressions for data at each acclimation temperature. The dashed line is a construct drawn to the right of the lowest test temperature that would cause 50% death at a given acclimation temperature.

one-third of the points plotted in Fig. 2 are times longer than 96 hr; i.e., 50% mortality does not occur in 96 hr. Extremely rapid mortality of all clams, including controls at 15°C acclimation, indicated the effect of a factor other than temperature. Since this seemed to be caused by a problem in developing the twist test, these results are not presented here.

Incipient lethal temperatures (50% mortality) based on 24-, 48-, and 96-hr testing intervals do not yield results equivalent to those obtained when the total effect of temperature is realized, i.e., when control and experimental death rates are equal [Fig. 3(a)]. Each of the 24-, 48-, and 96-hr regressions was significantly different from the total effect ( $P < 0.01$ ).

### Long-Term Cold Stress

Results of exposure to long-term cold stress indicated that a decrease in acclimation temperature enabled the clams to withstand lower test temperatures. Few combinations of acclimation and test temperatures resulted in 50% mortality. Clams could not be acclimated to 2°C. The data variation used to calculate the 50% mortality levels was greater than that for the long-term heat-stress data because of the difficulty in determining death at the lower temperatures. Small clams (22 to 24 mm SL) and larger

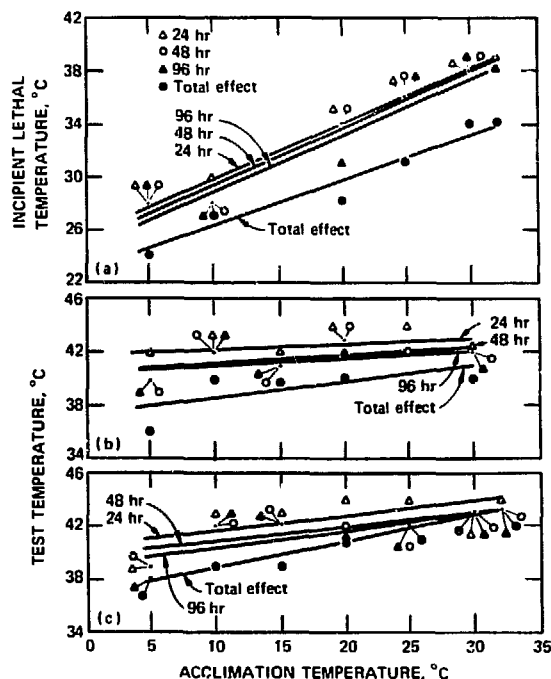


FIG. 3 Comparison of upper lethal temperatures (50%) obtained from observations made at 24, 48, and 96 hr with those obtained by continuing observations until experimental and control mortality rates are equal. Regression lines for each time interval are shown across the various acclimation temperatures for (a) long-term heat stress and for short-term heat shock in (b) 1973 and (c) 1974.

clams exhibited similar mortality when both were acclimated to 20°C and exposed to 2°C (Table 2). Neither group experienced mortality at 7.5°C. Minimum time to 50% mortality for any of the test temperatures was 200 hr.

### Short-Term Heat Shock

In the heat-shock tests the upper lethal temperature (50% mortality) varied depending on whether it was assessed at 24, 48, or 96 hr, or when experimental and control death rates were equal [Fig. 3(b) and 3(c)]. Although the object of the shock temperatures was to produce 100% mortality, the 50% level was used for comparisons because of the increased accuracy of estimate. Data for 1973 and 1974 are similar, but the 96-hr regression for the 1974 data was not significantly different ( $P > 0.25$ ) from the total effect. All other times were significant: For the 24-hr regressions from both 1973 and 1974 and the 48-hr regression from 1974,  $P < 0.01$ ; for the 48-hr regression from 1973,  $P < 0.025$ ; and for the 96-hr regression from 1973,  $P < 0.05$ .

TABLE 2 Acclimation and Incipient Lethal Temperatures and Times to 50% Mortality Determined for Clams Exposed to Long-Term Low Temperatures

| Acclimation temperature, °C | Incipient lethal | Time to 50% Mortality, hr |     |       |
|-----------------------------|------------------|---------------------------|-----|-------|
|                             |                  | 2°C                       | 5°C | 7.5°C |
| 5                           | ~2               |                           |     |       |
| 10                          | ~2               |                           |     |       |
| 15                          | ~2               |                           |     |       |
| 20                          | est. 4           | 290                       |     |       |
| 25                          | est. 7           | 510                       | 560 |       |
| 30                          | est. 9           | 220                       | 420 |       |
| 20S*                        | 2 < x < 7.5      | 200                       |     |       |

\*S = small clams (22 to 24 mm).

### Summary of Thermal Tolerance

The upper and lower incipient lethal temperatures (50% mortality) derived from the long-term heat and cold stress define a temperature tolerance range for adult *C. manilensis* (Fig. 4). The incipient lethal temperatures are shifted upward by acclimation to warmer temperature and downward by acclimation to cooler temperature. The upper ultimate incipient lethal temperature is about 34°C, and the lower ultimate incipient lethal temperature is about 2°C.

The temperatures that produce 100% mortality after a 30-min exposure also increase with acclimation to warmer temperatures. The 100% mortality levels obtained in 1973 and 1974 were not significantly different ( $P > 0.75$ ). The upper lethal temperature for 30-min exposure was 43°C but was only 3°C less at the 5°C acclimation temperature.

### DISCUSSION

The value of continuing observations over a long time period (until control and experimental death rates are

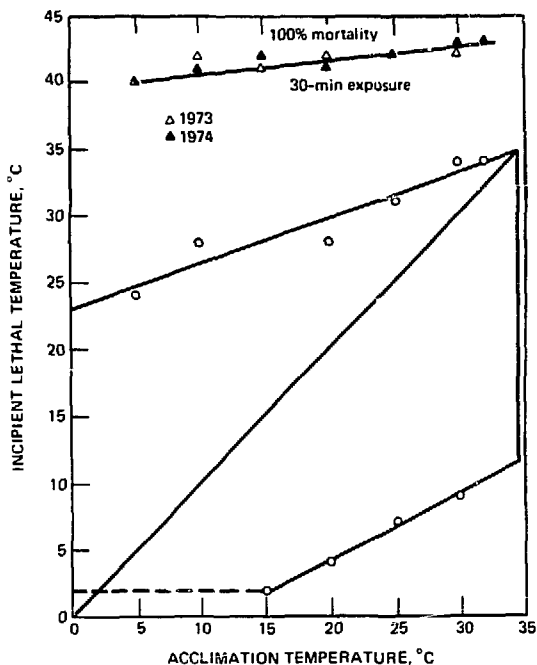


FIG. 4 Summary of the thermal tolerance of *Corbicula manilensis*. Long-term upper and lower thermal tolerance limits (50%) and temperatures resulting in 100% mortality after 30 min are shown. Upper and lower incipient lethal temperatures were about 34 and 2°C, respectively.

equal) was consistently demonstrated in our measures of thermal tolerance. Habel (1970) carried out heat-stress experiments on *Corbicula* for slightly over 96 hr after acclimation to ~23°C. His 50% mortality level agreed very closely with our 96-hr results at a similar acclimation temperature (25°C) but was 3.5°C higher than our total effect level for that acclimation group. Thus short observation periods result in an overestimate of tolerance. This overestimate would be expected to be greatest during long-term exposures. Dickie (1958) noted that median mortality levels of the Atlantic deep-sea scallop (*Placopecten magellanicus* Gmelin) at 120-hr exposures were about 0.8°C lower than those at 48-hr exposures. Our data demonstrate that the period of observation can have a much greater effect on determination of upper incipient lethal temperature for *Corbicula* than for the scallop.

Because of the length of time required for acclimation and testing (2 to 4 months), the feeding regime is important. The nutritional state of the clams apparently can alter the results of resistance tests quite significantly. The results of our long-term heat stress were considerably different from those obtained by Coutant (1973), who found that clams acclimated to 15 and 20°C were less resistant to heat stress than those acclimated to 5 and 10°C. He concluded that this reversal of the usual order resulted from the increased maintenance requirement of clams at

higher temperatures and from starvation. The clams Coutant used were collected from a potentially nutrient-poor environment (a water-supply tank) and were not fed while in the laboratory. Our results show the usual increase in resistance with increase in acclimation temperature. The most reasonable explanation for the difference is the effect of the nutritional state of the clams. It is still possible, however, that clams acclimated to higher temperatures received less than sufficient food because they had a higher maintenance energy requirement. The fairly constant slopes relating resistance and acclimation temperatures (Fig. 4) suggest, however, that this was not the case.

Part of the variability in our data, particularly at low temperatures, was undoubtedly caused by difficulty in accurately determining death. The criterion for death was a lack of response, but response includes a range of actions from immediate closure of the valves to a very slight movement of the mantle. At cold temperatures (10°C or less) the response could be extremely slow. It was at these low temperatures that the twist test was most valuable because the clams usually remained closed. On the other hand, at cold temperatures the twist test may have slightly stressed the clams. A more accurate criterion for death would have been helpful, but it probably would not have significantly affected the 50% mortality levels.

The potential for stressing clams with the twist test was demonstrated by the results of the long-term exposures to high temperature. Clams at the 15°C acclimation temperature were the first ones tested in this way. It was obvious when we observed the clams that the twist test had been applied too vigorously and the clams forced open in a manner that injured them. Further tests were carried out more carefully, with pressure being applied only to the extent that resistance was obvious. The data at 15°C acclimation was faulty and had to be discarded.

The long-term studies presented here have practical applicability to assessment of power-plant impacts. The instantaneous temperature changes to which the clams were subjected at the start of each of our tests may not be representative of the usual conditions near power-plant discharges, but rapid changes in the temperature of discharge waters can occur. A recent plant shutdown in Alberta, Canada (Ash, Chymko, and Gallup, 1974) resulted in a 16.9°C drop in temperature within 30 min. Certainly very rapid increases of temperature can occur at start-up or during fluctuations in the spatial location of the discharge plume as well.

This study is a first step in the overall examination of *C. manilensis* and its interaction with power plants. Except for the long-term cold stress on 2-year-old clams (22 to 24 mm), tests reported here were carried out on 4-year-old adults (24 to 35 mm). No significant difference was found in tolerance, but further testing of different age groups will be necessary since it is at the veliger larval stage that the clams infest and foul various types of intake structures. The heat-shock tests indicate the feasibility of using hot water

(e.g., at 43°C for 30 min) to remove adult clams small enough to pass through the condenser tubing. Either reversing water flow, where that is possible, or installing a feedback loop to recycle discharge water to the intake for short periods of time would provide the lethal water temperatures required. Thermal removal of clams before they can clog condensers or intake screens would prevent the costly shutdowns necessary for mechanical cleaning and avoid the long-term toxicity problems involved in using chlorine. However, results obtained using adult clams can only provide general guidelines for the potentiality of preventing fouling by the veligers. Final decisions regarding the efficacy of using heated water to prevent larval fouling will depend on studies of larval thermal tolerance.

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# The Effect of Acclimation on the Temperature Tolerance of Young Muskellunge Fry

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## ABSTRACT

Temperature tolerances of newly hatched muskellunge fry (*Esox masquinongy*) were determined in the laboratory. Three groups of fry were reared in a closed tank system under controlled environmental conditions for 31 days at 7, 15, and 25°C. The photoperiod was 12 hr of light alternating with 12 hr of darkness (L-D, 12:12). Biweekly water samples were taken to monitor the closed tank systems. Parameters measured were dissolved oxygen, alkalinity, pH, organic and nitrate nitrogen, phosphates, and ammonia, all of which were in normal concentrations. Critical thermal maximum (CTM) was used to indicate temperature tolerance. Experimental fish reared at 25°C exhibited the highest temperature tolerance (33.2 to 36.1°C) throughout their development. The CTM values for the 15°C experimental fish ranged from 29.8 to 34.4°C. These fish exhibited a decrease in temperature tolerance during their swim-up stage, followed by a slow recovery period indicated by a general increase in CTM values. Two days after the experiment began, 7°C experimental fry entered a torpid state that lasted for 10 days and was followed by a general decrease in thermal tolerance and a high mortality rate. The CTM values for this group ranged from an initial high of 34.0°C on the first day to 27.2°C at the termination of the experiment. Development never reached the free-swimming stage.

The muskellunge (*Esox masquinongy* Mitchell), one of the largest freshwater game fish in eastern North America, is found in waters from southern Quebec to Tennessee and northward through Illinois, Wisconsin, and Minnesota to southwestern Manitoba, where it inhabits warm, heavily vegetated lakes and rivers. A top carnivore, it feeds principally on fish, but waterfowl, muskrats, frogs, and mice have been found in stomachs of this species (Houder, 1952).

The muskellunge is abundant in Chautauqua Lake, New York, plays an important part in the maintenance of the aquatic ecosystem of this lake, and is the basis for one of

New York's most important sport fisheries (Heacock, 1946). To maintain the muskellunge fishery in the face of intense fishing pressure, the New York State Department of Environmental Conservation established a hatchery in 1904 at Bemus Point exclusively for the propagation of muskellunge. In 1964 this hatchery was moved to Pendergast Point, and a concentrated effort was made to raise muskellunge to fingerling size (12 to 20 cm) for stocking. Minimal temperature control was provided at the hatchery, but water temperatures at times have undergone wide fluctuations. Numerous die-offs have occurred at the hatchery, with 100% mortalities occurring in 1973 and 1974 at the swim-up stage of development. This stage is characterized by the development of a functional upper jaw and total absorption of the yolk sac. The swim-up stage terminates when fry begin actively swimming and feeding. The reason for the mortalities has not been determined (Youmans, 1974). We hypothesized that temperature may be an important factor in the development of muskellunge fry and may be a contributing factor in the mass mortalities at the hatchery.

Muskellunge spawn in late April or early May at water temperatures from 9 to 15°C. They lay their eggs in heavily vegetated areas where water varies in depth from 40 to 60 cm. Hatching occurs in 7 to 14 days, depending on water temperature (Scott and Crossman, 1973). Optimum temperature for incubation of eggs is 13°C (Galat and Eipper, 1969). Adult fish can tolerate temperatures as high as 32°C, but 24°C appears to be optimal (Scott, 1964; Ferguson, 1958).

Numerous studies have shown that temperature is a critical factor in the propagation and hatching of fish (Colby and Brooke, 1970; Hubbs, 1966; Kelley, 1968; Edsall, 1970; Combs, 1965; Allbaugh and Manz, 1964; Merriman, 1935). However, few studies have been concerned with the temperature tolerance of sa. fry and larval stages of fish. Allen and Strawn (1971) determined acclimation rates for juvenile channel catfish (*Ictalurus punctatus*). Fish changed from higher to lower temperatures required

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from 4 to 14 days to approach acclimation to the new temperature. Fish changed from lower to higher temperatures reacclimated in 1 to 3 days. Banner and Van Arman (1973) reported that bluegill sac fry (*Lepomis macrochirus*) were more resistant to thermal stress than eggs and that swim-up fry were more sensitive to high temperatures than fry at any other stage of development. Hokanson, McCormick, and Jones (1973) reported that newly hatched northern pike sac fry (*Esox lucius*) reared at temperatures from 6 to 17.6°C were more sensitive to high temperatures than were swimming larvae.

The purpose of our study was to delineate the effect of rearing temperature on the thermal tolerance of muskellunge fry and to determine periods of temperature sensitivity during the development of these fish. Temperature tolerance was determined by the critical thermal maximum (CTM) method.

aerated tap water. Temperatures were controlled to  $\pm 1.5^\circ\text{C}$ , except for the 15°C acclimation tanks, where temperature fluctuations as great as  $-3$  and  $+7^\circ\text{C}$  were sustained for up to several hours on six different occasions when the temperature control system failed. Daily temperatures were recorded to the nearest 0.1°C with a mercury thermometer. Details of the design and construction of the controlled-environment system used in this study are given by Spotila, Koons, and Terpin (1975). Tanks were checked daily for dead and diseased fish, which were removed and preserved. After swim-up, fry were fed a diet of live brine shrimp (*Artemia* sp.) two to four times a day. Water was analyzed biweekly for dissolved oxygen, pH, and alkalinity and every two weeks for ammonia, nitrates, organic nitrogen, and phosphates. Chemical characteristics of water in the acclimation tanks are summarized in Table 1. Two tanks were used at each temperature. Data from fry in one of the 15°C

TABLE 1 Water-Quality Data for Controlled-Environment System Used to Raise Muskellunge at 7, 15, and 25°C

| Parameter                    | Temperature  |             |              |
|------------------------------|--------------|-------------|--------------|
|                              | 7°C          | 15°C        | 25°C         |
| Dissolved oxygen, mg/liter   | 6.0-7.2      | 6.2-7.6     | 5.8-7.2      |
| Alkalinity, mg/liter         | 46-116       | 43-75       | 52-107       |
| pH                           | 7.1-8.0      | 7.4-7.9     | 7.5-8.0      |
| NH <sub>3</sub> -N, mg/liter | 0.178-0.67   | 0.071-0.195 | 0.178-0.184  |
| Organic nitrogen, mg/liter   | <0.009-0.056 | <0.009      | <0.009-0.009 |
| NO <sub>3</sub> -N, mg/liter | 0.090->4.50  | 3.00-4.50   | 2.86-3.00    |
| Orthophosphate, mg/liter     | 0.017-0.499  | 0.520-0.970 | 0.090-0.175  |
| Total phosphate, mg/liter    | 0.056-0.520  | 0.670-0.970 | 0.103-0.241  |

## METHODS AND MATERIALS

Newly hatched fry were obtained from the Pendergast Point Hatchery on May 17, 1974. Approximately 18,000 fry were transported in sealed plastic bags filled with water, injected with oxygen, and packed in ice. The temperature of the water during transport decreased from 14°C at the hatchery to 8°C on arrival at the laboratory. Approximately 4000 fry were immediately transferred to the 7°C acclimation tanks. The remainder of the fry were brought up to their experimental acclimation temperatures (15 and 25°C) at a rate of 2°C per hour.

Fry were reared in wooden closed-system tanks, 61 by 91 by 43 cm, under controlled environmental conditions for 31 days at 7, 15, and 25°C. Photoperiod was 12 hr of light alternating with 12 hr of darkness (L-D, 12:12), with the light period centered at 1400 EST. Illumination was provided by fluorescent lamps. Light intensity measured by an Epply pyranometer (0.3 to 3.0  $\mu\text{m}$ ) just above the water surface ranged from 0.01 to 0.02  $\text{cal cm}^{-2} \text{min}^{-1}$ . Net radiation measured with a Funk radiometer (all wavelengths) ranged from 0.01 to 0.02  $\text{cal cm}^{-2} \text{min}^{-1}$ . Visible light measured with a Weston light meter ranged from 68 to 180 ft-c ( $\bar{x}$  = 121 ft-c). Acclimation tanks were filled with aged (dechlorinated), continuously filtered and

acclimation tanks were discarded because of excessive temperature fluctuations and disease.

Critical thermal maxima (CTM) were determined between 1000 and 1500 EST in a lighted laboratory according to the method described by Hutchison (1961). Fry were placed in continually aerated water at their acclimation temperature ( $\pm 1^\circ\text{C}$ ) and heated at a rate of  $1.0 \pm 0.2^\circ\text{C}$  per minute. Water temperatures were measured with a standardized mercury thermometer graduated to 0.1°C.

A one-way analysis of variance (ANOVA) was performed on the mean CTM values for the three temperatures, and an SNK test (Sokal and Rohlf, 1969) was used to align the mean values according to their significance levels. Differences in CTM through time at each temperature were analyzed using a one-way ANOVA, and means were ranked according to their significance levels using the SNK test. Calculated F values for ANOVA were considered significant at the 5% level of probability ( $P = 0.05$ ) and highly significant at the 1% level ( $P = 0.01$ ).

## RESULTS

The loss of righting response was used as the end point for the CTM of fry reared at 25°C and of post-swim-up fry reared at 15°C. When placed in the test flask, these fry

swam continually until they approached their CTM. Then they began to gasp at the surface, sporadically increased their swimming speed, and suffered intermittent loss of equilibrium. They reached their CTM when they turned over, became stiff, and lost the ability to right themselves. When placed in water at their acclimation temperature, these fry recovered.

We found it very difficult to ascertain the CTM of fry reared at 7°C and of pre-swim-up fry reared at 15°C. These fry usually remained inactive throughout a test, rarely swimming, and moving only when probed with a glass rod. The only consistent end point was a dorsolateral bending of the posterior end of the body accompanied by general muscle rigor. Neither loss of righting response nor spasms occurred.

There was a highly significant statistical difference in the mean CTM of fry reared at 7 (28.8), 15 (31.9), and 25°C (34.5°C);  $F = 215.7$ ;  $P \leq 0.01$ . An increase in acclimation temperature resulted in an increase in CTM. These data are clarified in Table 2. There was a highly significant difference ( $P \leq 0.01$ ) between the mean CTM of fry from each acclimation temperature in comparison with those for fry at either of the other two acclimation temperatures (Table 2). At 25°C development occurred very rapidly,

TABLE 2 SNK Test for CTM of Muskellunge Fry Raised at 7, 15, and 25°C

| Acclimation temperatures | K value |
|--------------------------|---------|
| 7 vs. 25°C               | 5.69*   |
| 15 vs. 25°C              | 2.60*   |
| 7 vs. 15°C               | 3.09*   |

\* $P = 0.01$  (highly significant).

TABLE 3 Mean\* Critical Thermal Maximum (CTM) of Muskellunge Fry Reared at 7, 15, and 25°C

| Rearing temperature |         |      |         |     |         |
|---------------------|---------|------|---------|-----|---------|
| 25°C                |         | 15°C |         | 7°C |         |
| Day                 | CTM, °C | Day  | CTM, °C | Day | CTM, °C |
| 7                   | 36.1    | 1    | 34.4    | 1   | 34.0    |
| 3                   | 35.6    | 7    | 33.6    | 15  | 29.5    |
| 5                   | 35.5    | 31   | 33.2    | 13  | 29.3    |
| 1                   | 34.8    | 5    | 33.0    | 17  | 29.2    |
| 9                   | 34.7    | 27   | 32.9    | 19  | 28.9    |
| 11                  | 34.6    | 23   | 32.5    | 21  | 28.5    |
| 17                  | 34.2    | 21   | 32.3    | 23  | 28.5    |
| 13                  | 33.8    | 17   | 30.9    | 27  | 28.1    |
| 19                  | 33.5    | 13   | 30.9    | 31  | 27.9    |
| 15                  | 33.2    | 19   | 30.8    | 25  | 27.8    |
|                     |         | 15   | 30.7    | 29  | 27.2    |
|                     |         | 9    | 30.1    |     |         |
|                     |         | 11   | 29.8    |     |         |

\*Any two means not encompassed by the same vertical line are significantly different as determined by an SNK test ( $P = 0.05$ ).

with fry completing swim-up by day 5 (Fig. 1). The CTM remained high during this period, reaching a maximum of 36.1°C on day 7. There was a slow decrease in CTM during the remainder of the experiment. In general, this decrease was not statistically significant (Table 3). The fry ate voraciously, and food availability was a major problem. This resulted in a deterioration of the fishes' condition and some cannibalism. Mortality reached 50% by day 13 and 86% by day 19, at which point the experiment was terminated. The CTM reached a low of 33.2°C on day 15 and was 33.5°C on the last day of testing. Development occurred more slowly at 15°C; swim-up took place between days 8 and 12. The CTM ranged from 29.8 to 34.4°C (Fig. 2). At the beginning of swim-up, there was a sharp drop in CTM from 33.6°C on day 7 to 30.1°C on day 9 and then to 29.8°C on day 11. When compared with pre- and post-swim-up values, CTM was significantly lower during swim-up and for several days after (days 9 to 19) (Table 3). Once fry became free swimming, there was a slow increase in CTM; this indicates a gradual increase in temperature tolerance.

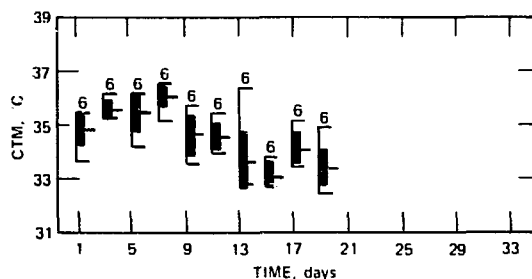


FIG. 1 Critical thermal maximum (CTM) of muskellunge fry reared at 25°C (L-D, 12 : 12). Vertical lines bounded by short horizontal lines are ranges; center horizontal lines are means; black rectangles represent  $\pm 2$  SE; and numbers are sample sizes.

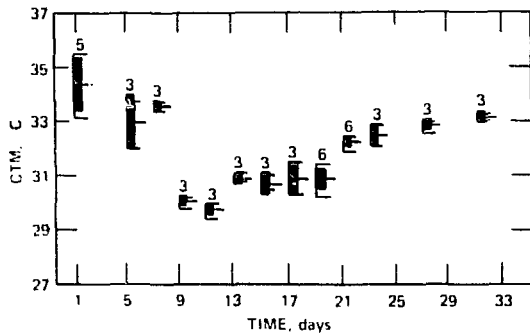


FIG. 2. Critical thermal maximum (CTM) of muskellunge fry reared at 15°C. Vertical lines bounded by short horizontal lines are ranges; center horizontal lines are means; black rectangles represent  $\pm 2$  SE; and numbers are sample sizes.

When taken from 14°C water and placed in 7°C water, muskellunge fry entered a state of torpor. The CTM declined from 34.0°C on the first day of the experiment to 29.3°C on day 13 (Fig. 3). Between days 2 and 12 these fry did not swim, did not respond to stimulation with a glass rod, and showed no end point that could be associated with their CTM. All fry tested during this period died owing to overheating. They appeared to be in an arrested state of development. We were able to determine the CTM of 7°C fry from day 13 on; it gradually declined from 29.3°C on day 13 to 27.2°C on day 29. The CTM for fry on days 13 to 17 was statistically different ( $P = 0.05$ ) from the CTM on day 1 and on days 21 to 31 (Table 3). These fry never reached the free-swimming stage. They failed to develop a functional upper jaw and died after absorbing their yolk sac.

## DISCUSSION

Rearing temperature had a highly significant effect on the temperature tolerance of muskellunge fry. These results indicate that the heat resistance of fry was readily altered by their past thermal history. This conclusion is consistent with the findings of other studies of fish (Allen and Strawn, 1971), salamanders (Hutchison, 1961; Spotila, 1972; Delsion and Whitford, 1973), and frogs (Brattstrom, 1968; Dunlap, 1969; Mahoney and Hutchison, 1969).

Fry raised at 25°C had the highest temperature tolerance and developed at the fastest rate. The decrease in temperature tolerance after swim-up was probably due to the deteriorating condition of the fry. Not enough food was available to sustain the high rate of metabolism of these fish.

At 15°C fry had a lower temperature tolerance, developed more slowly, and exhibited a marked reduction in temperature tolerance during swim-up (Fig. 2). The sharp drop in CTM during swim-up indicated that this was a critical stage in the development of the muskellunge. During this time fry changed from inactive nonfeeding organisms to free-swimming effective carnivores. This change was probably caused by changes at the cell, tissue,

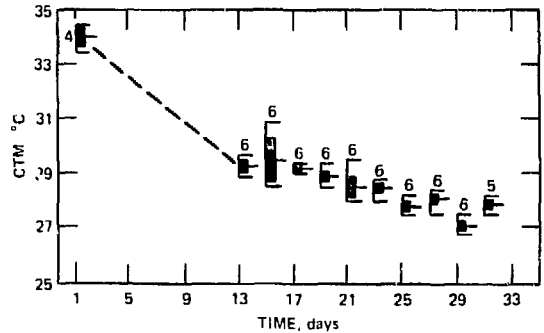


FIG. 3. Critical thermal maximum (CTM) of muskellunge fry reared at 7°C. Vertical lines bounded by short horizontal lines are ranges; center horizontal lines are means; black rectangles represent  $\pm 2$  SE; and numbers are sample sizes.

and organ levels in these fish. The manner in which these changes affect temperature tolerance is not known. A similar effect was reported for bluegill fry in the process of swimming up (Banner and Van Arman, 1973). Muskellunge fry reared at 25°C did not show an increase in temperature sensitivity during swim-up (Fig. 1). This may have been because of their rapid rate of development. These fish either passed quickly through any critical periods in the transition process from sac fry to the free-swimming stage or did not experience these critical periods because of a change in the developmental process at 25°C.

Fry maintained at 7°C entered a state of torpor and failed to develop properly. Arrested development of the 7°C fry suggests that low temperature inhibits key physiological processes in these fry. The gradual decline in CTM indicates that these fish did not complete acclimation to 7°C during the course of this experiment. Acclimation to low temperatures is a lengthy process. Newts (*Notophthalmus viridescens*) required 4 days to acclimate when transferred from 20 to 4°C (Hutchison, 1961). Channel catfish fry required 4 to 14 days to complete acclimation when transferred from 30 or 34 to 26°C (Allen and Strawn, 1971). In this experiment muskellunge fry were not able to adjust to 7°C. Physiological processes were disrupted, development was incomplete, and death resulted. These fry suffered a 46% mortality after 30 days, 66% by day 31, and 74% by day 35.

Muskellunge fry appear to be at least as tolerant of high temperatures as are larger individuals. Scott (1964) reported that the lethal temperature for juvenile muskellunge acclimated to 25°C was 32.5°C (resistance time of 400 min) as compared to a lethal temperature of 32.5°C for northern pike, *Esox lucius*, and 33°C for F<sub>1</sub> hybrids. In our experiment muskellunge fry acclimated to 25°C had CTM values ranging from 33.2 to 36.1°C. Hokanson, McCormick, and Jones (1973) reported that day-old larvae of the northern pike acclimated to 17.7°C had a lethal temperature (resistance time of 24 hr) of 24.8°C.

Temperature is an important factor in the development of muskellunge fry. It may be a contributing factor in the



mortalities of fry at the Pendergast Point Hatchery on Chautauqua Lake. It is probably not the only factor. The influences of heavy-metal contamination and disease remain to be investigated.

Nevertheless, it is clear that precise temperature control is vital to the successful propagation of muskellunge. This is especially true during swim-up. Maintenance of high temperature ( $\sim 25^{\circ}\text{C}$ ) during this period would ensure that fry would quickly pass through this critical phase of development. Temperatures could then be manipulated to keep metabolic rates of the fry in synchrony with the availability of live food supplies.

## NOTE ADDED IN PRESS

Recent experiments in our laboratory indicate that onset of spasms occurs in larval muskellunge the instant before loss of righting response. This is sometimes seen as only a slight shuddering of the operculum and/or total muscle rigor, but it always occurs. Thus the end point used in this study corresponds to onset of spasms. The end-point temperature varies no more than  $0.1^{\circ}\text{C}$  between spasms and the loss of righting response in larval muskellunge.

## ACKNOWLEDGMENTS

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# Thermal Tolerance of Stream Cyprinid Minnows

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S. E. Williams

## ABSTRACT

The responses to thermal stress of three cyprinid minnows (*Notropis lutipinnis*, *N. cummingsae*, and *Semotilus atromaculatus*), which inhabit tributaries of the Savannah River but are excluded from thermally stressed streams, were determined for fish tested individually and in groups. All species exhibited the same precipitous response, with excellent survival at five rates of heating, but rapid mortality occurred at 33 to 34°C. Slow heating rates did not elevate the upper limits of tolerance. Acclimation did not occur when temperature was continuously raised but did occur when the fish were subjected to constant sublethal temperatures.

Ten species of cyprinid minnows are found in various streams at the Savannah River Plant (SRP), a national environmental research park of the Energy Research and Development Administration near Aiken, South Carolina. Minnows are rare in streams thermally stressed by nuclear-production-reactor effluent, except during brief periods of ambient stream temperatures (McFarlane, this volume). Three minnow species, yellowfin shiner (*Notropis lutipinnis*), dusky shiner (*Notropis cummingsae*), and creek chub (*Semotilus atromaculatus*), are found sympatrically in first-, second-, and third-order tributaries of the Savannah River in upper coastal-plain South Carolina and Georgia. The species do not occur in equal abundance, however. The yellowfin shiner is the dominant species of first-order streams at SRP. The small dusky shiner is the most abundant minnow of the second- and third-order streams and may penetrate the lower reaches of first-order tributaries. The creek chub is not abundant and exhibits a very patchy distribution in first- and second-order streams.

Investigators of thermal tolerance usually acclimate the chosen species at one or more constant temperatures. Fish accommodate themselves to various thermal regimes with time; small differences in acclimation temperature alter their thermal responses significantly (see Hutchison, this volume; Murphy et al., this volume). This is undoubtedly important for fish found in bodies of water sufficiently large to resist rapid temperature fluctuation and able to offer the fish a behavioral choice in preferred temperature. Fish inhabiting small streams are not afforded this luxury and must withstand continual temperature fluctuations of

large magnitude. The streams at SRP may vary from 5 to 7°C within a 24-hr period and from 12 to 15°C over several days. Thus acclimation at a constant temperature may eliminate one variable but may be biologically meaningless or even stressful for fish from these streams. Shifting ectothermic animals that normally experience variable temperatures in their natural habitats to a constant-temperature regime in the laboratory may significantly alter or weaken their responses to various physiological stresses (see Sastry, this volume; Thorp and Hoss, 1975). Furthermore, the physical condition of any wild-caught fish may deteriorate after capture, and long acclimation periods can lead to physiologically weakened test subjects. This study reports on the thermal tolerances of unacclimated yellowfin shiners, small dusky shiners, and creek chubs.

## METHODS

The minnows were collected from first- and second-order streams at SRP during January 1975 and were tested individually and in groups. Yellowfin shiners were collected from a 13°C stream and tested the same day. Dusky shiners were collected from an 8°C stream and maintained near this temperature until they were tested during the next 2 days. One to three fish were placed into a 2-liter round flask heated by an external mantle at rates of 1.0°C/min (60°C/hr) and 0.1°C/min (6°C/hr). Two points were determined for each fish. As the water temperature increased, the fish exhibited violent muscular spasms and locomotory activity became disorganized. This point is quite distinct and represents the critical thermal maximum (CTM; see Hutchison, this volume). With further temperature elevation the minnows lay inverted at the surface and, eventually, opercular movement ceased. The cessation of opercular movement (COM) appears synonymous with the lethal temperature for this species since they do not recover when transferred to lower temperature as do other species, e.g., *Gambusia affinis* (unpublished data).

Group tests of yellowfin shiners and creek chubs were conducted in two adjacent, externally insulated, 75-liter aquariums. Temperatures were regulated by a constant-

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Moore: Birmingham-Southern College, Birmingham, Alabama.

temperature water bath. Fish collected from the streams were immediately placed in the aquariums and tested at three rates of temperature increase. The maximum rate was obtained when fish collected from a 16°C stream were placed into 21°C aquariums and raised to 34°C in 4.5 hr, i.e., at an average increase of 4°C/hr. Other fish collected at 17°C were placed in 21°C aquariums and raised to 34°C in 12 hr, an increase of ~1°C/hr. The slowest rate of increase, 0.125°C/hr, was achieved by placing fish collected at 11°C in 13°C aquariums and raising the water temperature 3°C/day for 8 days. In all experiments, temperatures were slowly raised at intervals of several hours during daylight and left constant overnight. Group tests produced data on lethal temperatures only (as COM). Natural daylight and photoperiods were maintained throughout.

The yellowfin shiners were 31 to 74 mm in total length and 0.19 to 3.62 g wet weight; dusky shiners were 25 to 46 mm and 0.19 to 0.89 g.

## RESULTS

The thermal tolerance, as percent survival, of the yellowfin shiner at five rates of temperature increase is shown in Fig. 1. Individually tested fish succumbed at lower temperatures with lower rates of temperature increase, a phenomenon demonstrated previously (Cox, 1974). Among the group tests, the 4°C/hr rate yielded 75% survival at 32°C, but complete fatality occurred within 40

min as the temperature approached 34°C. The 1°C/hr rate permitted 92% survival at 32°C but 101 fish died in a 15-min period as the temperature reached 33°C. The 0.125°C/hr rate produced only one fatality in reaching 34°C. These fish survived 2 hr at 34°C, but all died overnight even though the temperature was held constant at 34°C.

The dusky shiners exhibited a similar response at slightly lower temperatures when tested individually (Fig. 2). Creek chubs were not collected in sufficient numbers for adequate samples at all temperature rates. However, 24 were group tested simultaneously with yellowfin shiners at an increase rate of 4°C/hr, and the responses were virtually identical (Fig. 3). The critical thermal maximums and lethal temperatures, expressed as cessation of opercular movement, for individually tested yellowfin and dusky shiners are presented in Table 1.

## DISCUSSION

The precipitous nature of the response to thermal stress of the dusky stripe shiner, *Notropis pilsbryi* (Hickman and Dewey, 1973), is similar to that exhibited by the three cyprinids tested in our study.

Two aspects of the effect of different heating rates deserve comment. First, rapid heating appears to shift the upper limit of thermal tolerance to a slightly higher temperature. The mean CTM and COM values for both

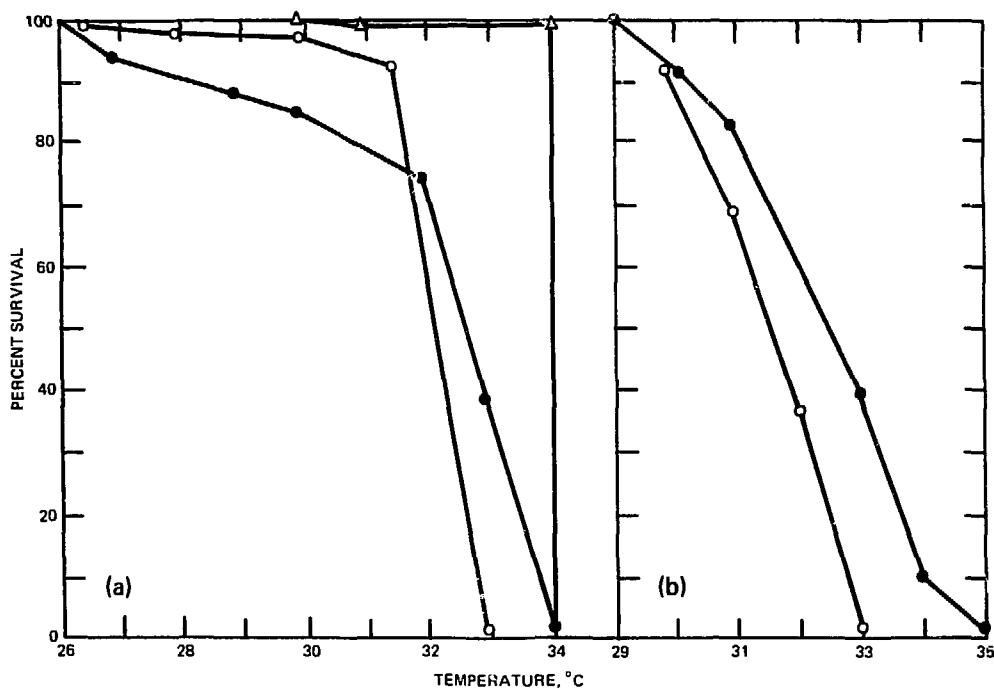


FIG. 1. Resistance of *Notropis lutipinnis* to thermal stress at five rates of temperature increase. (a) Groups: ●—●, 4°C/hr (36); ○—○, 1°C/hr (110); △—△, 0.125°C/hr (93). (b) Individuals: ●—●, 60°C/hr (23); ○—○, 0°C/hr (22).

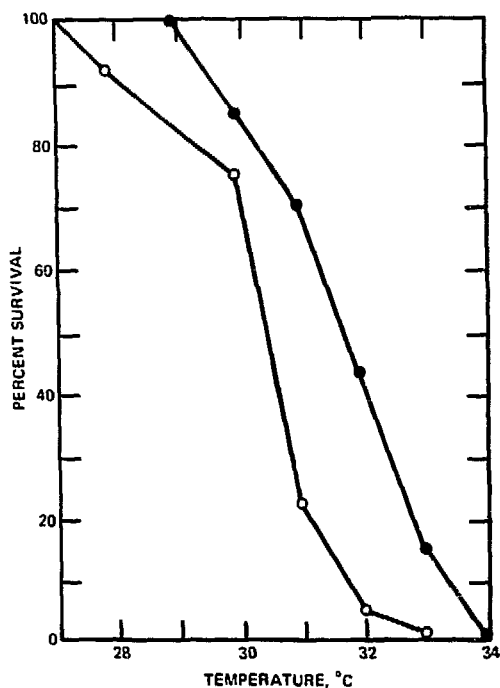


FIG. 2 Resistance of *Notropis cummingsae*, tested individually, to thermal stress at two rates of thermal increase.  $\circ$ — $\circ$ , 6°C/hr (36 fish tested).  $\bullet$ — $\bullet$ , 60°C/hr (60 fish tested).

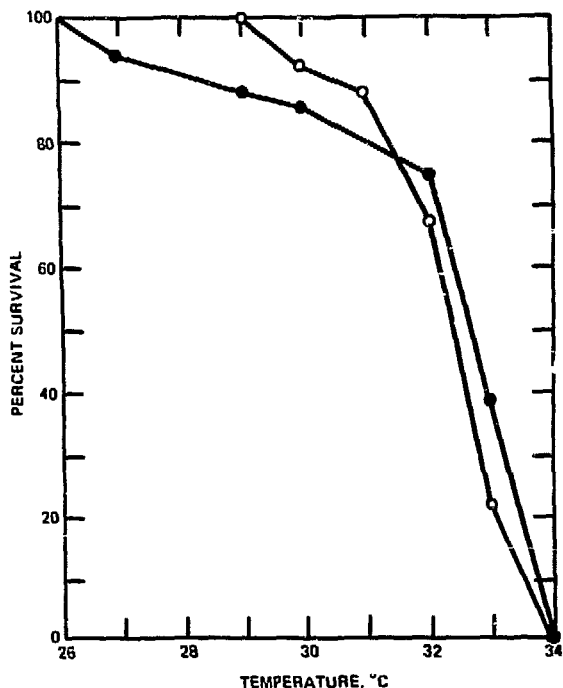


FIG. 3 Resistance of 24 *Semotilus atromaculatus* ( $\circ$ — $\circ$ ) and 36 *Notropis lutipinnis* ( $\bullet$ — $\bullet$ ), tested simultaneously as a group, to thermal stress at 4°C/hr temperature increase.

TABLE 1 Temperature Values\* for Critical Thermal Maximums (CTM) and Cessation of Opercular Movement (COM) for Yellowfin and Dusky Shiners

| Rate of increase, °C/hr | Yellowfin shiner |                 | Dusky shiner    |                 |
|-------------------------|------------------|-----------------|-----------------|-----------------|
|                         | CTM              | COM             | CTM             | COM             |
| 0.1                     | 29 ± 0.7(28–30)  | 32 ± 1.0(30–33) | 28 ± 1.1(27–30) | 31 ± 1.2(28–33) |
| 1.0                     | 30 ± 1.2(27–32)  | 33 ± 1.4(30–35) | 29 ± 1.3(27–32) | 32 ± 1.3(30–34) |

\*Mean ± standard deviation (range). All pairs differ significantly ( $P < 0.01$ ).

yellowfin and dusky shiners, regardless of size or recent thermal history, were consistently higher at the 1.0°C/hr heating rate than at 0.1°C/hr (Table 1). The slower rate may allow body temperature to keep pace with increasing water temperature, but slow heat application did not elevate the lethal temperature by acclimation. Only two of 405 fish tested survived temperatures in excess of 34°C, and these were exposed to the most rapid rate of heating (Fig. 1). Thus it appears that 34°C is the upper lethal limit for these species in winter when stream temperatures fluctuate from 18 to near 0°C.

The second aspect involves the effects of acclimation. These minnows can be acclimated to higher temperatures if they are exposed to a constant sublethal temperature for an extended period. For example, yellowfin shiners collected

in 13°C water had a mean CTM of 29°C and COM of 32°C (23 fish were tested) when tested on the day of collection. Four days of acclimation at 23°C raised the CTM to 33°C and COM to 34°C (14 fish tested) in all tests at the 6°C/hr rate of heating. Further acclimation at higher temperatures will extend the lethal temperature beyond 40°C. Yet 8 days of slow, gradual increase (0.125°C/hr) did not extend the lethal limit beyond the 34°C typical for nonacclimated fish. The principal effect of slow steady temperature increase seems to be reduction of mortality at low temperatures rather than elevation of the lethal limit.

#### ACKNOWLEDGMENTS

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# Thermal Tolerance and Respiratory Movement of Bluegill from Two Populations Tested at Different Levels of Acclimation Temperature and Water Hardness

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Charles T. Garten, Jr.  
Michael H. Smith  
Edward A. Standora

## ABSTRACT

Bluegill (*Lepomis macrochirus*) from two locations were acclimated at 16, 24, and 32°C and at two levels of water hardness (conductivity 77 and 110  $\mu\text{mho}$ ). Fish from a reservoir receiving thermal effluent had a higher critical thermal maximum (CTM) and lower rates of respiratory movement at each acclimation temperature than those from the control location. Respiratory movement and CTM increased as a function of acclimation temperature. Increased water hardness tended to decrease the CTM but had no consistent effect on respiratory movement.

Fish inhabiting thermally elevated zones of closed-loop reactor-cooling reservoirs may be subject to increased levels of water hardness and total dissolved solids because of continuous evaporation and the addition of supplemental water (Tilly, 1975). In such instances the possibility exists of synergistic effects between increased water hardness and increased water temperatures on fish physiology. Otto and Gerking (1973) found that increases in salinity lowered the thermal tolerance of Death Valley pupfish (*Cyprinodon* sp.) as acclimation temperature increased. Changes in respiratory breathing rates in fish have been suggested as one possible indicator of stress from environmental pollutants (Cairns et al., 1973). The purpose of our study was to test thermal tolerance and respiratory movement in bluegill from thermally differing portions of a closed-loop reactor-cooling reservoir system under varying combinations of acclimation temperature and water hardness.

## METHODS AND MATERIALS

Bluegill were seined in January and February 1975 from two reservoirs (Pond C and Par Pond) constructed in 1958 as part of a closed-loop reactor-cooling system located

at the Energy Research and Development Administration's Savannah River Plant (SRP). Pond C has an area of 67 ha and receives heated effluent from a nuclear production reactor. Water temperatures in the central part of the pond at times exceed 45°C and seldom fall below 25°C when the reactor is operating. Water from Pond C flows through a conduit beneath a rock dam into Par Pond, which has an area of 1200 ha. Water temperatures in the areas of Par Pond where fish were sampled range from 20 to 30°C in winter. Temperatures in other portions of the lake are normal for bodies of water at this latitude. A more complete description of the study areas is presented by Parker, Hirshfield, and Gibbons (1973).

The fish, and the water from which they were collected, were placed in glass containers and suspended in a 75-liter aquarium containing water at 16, 24, or 32°C ( $\pm 1^\circ\text{C}$ ). Over a period of 2 to 3 hr the fish were brought to their acclimation temperatures. Par Pond water was used as a control level of water hardness (concentration A: 37 ppm  $\text{CaCO}_3$ ; alkalinity, 3.5 ppm  $\text{CaCO}_3$ ; and conductivity, 77  $\mu\text{mho}$ , corrected to 25°C in a 1-cm cell). An elevated level of water hardness was obtained by evaporating Par Pond water (concentration B: 43 ppm  $\text{CaCO}_3$ ; alkalinity, 6.3 ppm  $\text{CaCO}_3$ ; and conductivity, 110  $\mu\text{mho}$ ). Hardness and alkalinity were measured by titration (American Public Health Association, 1971), and conductivity was measured by an in situ water-quality analyzer. Fish were acclimated for a minimum of 7 days and were not fed. Acclimation was considered complete after 60 hr (Holland et al., 1974). A 12:12 light-dark photoperiod was maintained during acclimation.

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Three to five fish at each of the acclimation-temperature and water-hardness treatment levels were tested. Respiratory movement, expressed as opercular beats per minute, was measured using an apparatus modified from that described by Spoor, Neihsel, and Drummond (1971). Fish were placed in plastic cylinders 12 cm long and 5 cm in diameter with stainless-steel-mesh electrodes on both ends and acclimated overnight. Then the electrodes were connected to an amplifier (Drummond and Dawson, 1974), and the opercular movement was registered on an oscillographic strip-chart recorder.

After the test of respiratory movement, thermal tolerance was measured according to procedures outlined by Holland et al. (1974). Critical thermal maximum (CTM) was determined by heating each fish in a 2-liter round-bottom flask at a rate of 1°C/min (Hutchison, 1961; Cox, 1974). Critical thermal maximum is defined as the temperature "at which locomotory activity becomes disorganized and the animal loses its ability to escape from conditions that will promptly lead to its death" (Lowe and Vance, 1955). Observations were made by one observer between 1200 and 1700 hr EDT. Total length and wet weight were measured as covariates to thermal tolerance. All fish tested were between 40 and 82 mm long ( $\bar{x}$  = 57, SD =  $\pm$ 0.93) and weighed 0.57 to 5.60 g ( $\bar{x}$  = 2.08, SD =  $\pm$ 1.07). Statistical analysis was accomplished using regression-analysis procedures of the Statistical Analysis System (Service, 1972) on a computer. Statistical significance throughout this work was indicated by  $P < 0.05$ .

## RESULTS AND DISCUSSION

Significant differences in CTM among acclimation temperatures ( $F_{2,45} = 397$ ) and between locations ( $F_{1,45} = 45$ ) and levels of water hardness ( $F_{1,45} = 6.3$ ) were detected using analysis of variance (ANOVA). The interaction between the location and acclimation-temperature treatments was also significant ( $F_{2,45} = 10$ ). All other interactions and the measured covariates, length and weight, were not significant. Acclimation temperature accounted for 85% of the total variation in the data, and the main effects of location and water hardness accounted for 5 and 1%, respectively [ $\omega^2$  statistic (Kirk, 1968)].

The results indicated that bluegill from the thermally elevated reservoir, Pond C, have a higher thermal tolerance than those from Par Pond. This is in agreement with the findings of Holland et al. (1974). At the 16°C acclimation temperature, the mean CTM of bluegill from Pond C was not significantly different from that of fish from Par Pond. At 24 and 32°C, fish from Pond C exhibited significantly higher CTM values than those from Par Pond (Table 1). Differences in thermal tolerance of fish from the two environments may be attributable to acclimatization or perhaps to genetic selection (Holland et al., 1974).

Increased water hardness tended to depress the mean CTM of bluegill at each location and acclimation-temperature treatment combination. Differences in CTM were significant between the two levels of water hardness at only

TABLE 1 Critical Thermal Maximum Estimates for Bluegill from Two Locations at Different Acclimation Temperatures

| Sample location | Acclimation temperature, °C |           |           |
|-----------------|-----------------------------|-----------|-----------|
|                 | 16                          | 24        | 32        |
| Pond C          | 31.5 (9)                    | 37.5 (10) | 41.4 (10) |
| Par Pond        | 31.4 (8)                    | 35.6 (10) | 38.5 (10) |
| Difference      | 0.1 (NS)                    | 1.9†      | 2.9†      |

\*Means at each acclimation temperature were significantly different from each other at both localities ( $P < 0.01$ ). Numbers in parentheses are sample size. NS is not significant.

†Overall  $F$  test for locality differences was significant ( $P < 0.001$ ), and Tukey's HSD test (Kirk, 1968) demonstrated significant differences among locality means at 24 and 32°C ( $P < 0.05$ ).

one of the treatment combinations. Pond C fish acclimated at 16°C had a CTM of 32.3°C at concentration A (control) and a CTM of 30.6°C at concentration B (elevated). Pond C fish may have the disadvantage of extreme directional selection for survival in an area of thermal elevation at the sacrifice of flexibility for varying conditions of water hardness. Opportunistic strategies, such as increased thermal tolerance, may result in reduced adaptability to other stressors.

A significant interaction among location, acclimation-temperature, and water-hardness effects ( $F_{2,45} = 19.4$ ) in an ANOVA of opercular movements by bluegill makes a simple interpretation of differences among the experimental groups difficult. The most important main effect in the analysis was acclimation temperature, which accounted for 46% of the total variation in response. The interaction among treatments accounts for 16% of the total variation in data. Although differences in mean opercular rate among various experimental groups were sometimes large (Table 2), sensitivity to differences was low because of the

TABLE 2 Mean Values of Respiratory Movement Rates (Opercular Beats/min) of Bluegill from Two Locations at Various Treatment Combinations of Acclimation Temperature and Dissolved Solids Concentration

| Sample location | Concentration | Acclimation temperature, °C |        |       |
|-----------------|---------------|-----------------------------|--------|-------|
|                 |               | 16                          | 24     | 32    |
| Par Pond        | A             | 47.9                        | 71.7   | 122.6 |
|                 | B             | 41.4                        | 66.0 * | 190.6 |
| Pond C          | A             | 37.9                        | 31.1 * | 147.1 |
|                 | B             | 24.3                        | 56.9   | 46.8  |

\*Pairs of means separated by asterisks are significantly different (Tukey's HSD test,  $P < 0.05$ ).

# Some Relationships of Upper Thermal Tolerances to Preference and Avoidance Responses of the Bluegill

Sherwood E. Peterson  
Robert M. Schutsky

## ABSTRACT

Temperature tolerance, preference, and avoidance studies were conducted on the bluegill (*Lepomis macrochirus*) in Conowingo Pond, lower Susquehanna River, Pa. Upper incipient lethal temperatures of 23.3, 29.3, and 35.8°C were determined for bluegill (mean total length 90 mm) acclimated at 1, 13, and 27°C, respectively. The results agree with published information on bluegill. Median resistance temperatures were plotted against linear regressions of preference and avoidance temperatures. Preferred temperatures were 4 to 5°C less than median resistance temperatures. Bluegill acclimated at temperatures  $\geq 13^\circ\text{C}$  avoided short-term (96 hr) lethal temperatures. Median resistance time for bluegill acclimated at 13 and 27°C leveled off and exceeded 96 hr near the avoidance temperature. Avoidance temperatures exceeded lethal temperatures at acclimation temperatures  $< 13^\circ\text{C}$ . No mortality occurred with bluegill acclimated at 1°C and subjected to a temperature differential  $\leq 21^\circ\text{C}$ . Low thermal responsiveness was observed most frequently in avoidance studies conducted at acclimation temperatures  $< 10^\circ\text{C}$  and in preference studies conducted at acclimation temperatures  $< 20^\circ\text{C}$ .

The ecology of Conowingo Pond, on the lower Susquehanna River in Pennsylvania, and connecting waters has been under study since June 1966. The research, conducted by Ichthyological Associates, Inc., was designed to determine the impact, if any, of the operation of the Peach Bottom Atomic Power Station Units 2 and 3 on aquatic life in the pond. Experimental studies initiated in November 1972 were designed to provide predictive data on (1) the behavior of fishes in or near the thermal discharge and (2) the effect of changes in temperature in or near the plume which may occur as a result of station operations.

Considerable information has accumulated on the effects of temperature on fishes in bibliographies by Beltz

et al. (1974), Coutant, Huber, and Pfuderer (1972), Coutant, Pfuderer, and Collier (1974), Coutant and Pfuderer (1973), Raney and Menzel (1969), and Raney, Menzel, and Weller (1974). Specific studies on preference and avoidance behavior of bluegill in response to a thermal gradient are available in Peterson, Schutsky, and Allison (1974a; 1974b) and Peterson and Schutsky (1975). Specific studies on temperature tolerance of bluegill are available in Banner and Van Arman (1973), Christianson and Tichenor (1968), Hart (1952), Hathaway (1927), Hickman and Dewey (1973), and Holland et al. (1974).

An important question concerning the operation of the plant was the relationship between thermal resistance and certain behavioral responses to temperature in the bluegill and whether these responses defined ranges of thermal tolerance. The objectives of the study were to determine whether thermal preference and avoidance responses were closely related to upper levels of thermal tolerance and whether mortality could be predicted using behavioral responses to a thermal gradient.

## MATERIALS AND METHODS

Test specimens ranging from 52 to 159 mm in total length (mean 90 mm) were collected by seine in Conowingo Pond and transported to the laboratory in 16-liter insulated and aerated bait carriers. Water temperatures varied less than  $\pm 0.5^\circ\text{C}$  during transport, and fish reached the laboratory within 3 hr.

Fish were held in the laboratory for 24 hr at their field acclimation temperature (the temperature of Conowingo Pond at the time of their capture) before they were tested. The holding tank was equipped with a  $25\text{-kg m}^{-1} \text{sec}^{-1}$  compressor and a constant-temperature circulator adjusted to maintain the acclimation temperature. The water temperature in the tank was monitored with a 7-day recording thermometer. Temperatures fluctuated less than  $0.5^\circ\text{C}$

Peterson and Schutsky: *Ichthyological Associates, Inc., Drumore, Pennsylvania.*



large degree of variation in opercular rate (coefficient of variation = 35.5%). Table 2 presents the mean opercular rates of bluegill at each of the selected treatments. The following trends are evident: (1) Opercular rate was not different between fish acclimated at 16 and 24°C but is sometimes more rapid at 32°C. (2) Par Pond fish exhibited more rapid opercular rates than Pond C fish. (3) Different levels of water hardness affected opercular rate at the 32°C acclimation temperature, but the trend was not consistent for fish from both locations.

The lower rates of respiratory movement in bluegill from Pond C relative to those from Par Pond are not necessarily indicative of a difference in oxygen consumption or metabolism. We did not measure the depth of opercular movement by bluegill, but the amount of oxygen exchanged across the gills is influenced by both the rate of opercular movement and the depth or amplitude of such movement. In response to oxygen shortage, rainbow trout increase the depth of opercular movement; freshwater eels increase both the frequency and depth of opercular activity (Van Dam, 1938, cited in Marshall, 1966). More recent work by Hughes and Roberts (1970) has shown that in trout the frequency of opercular movement is not as important as the stroke volume during the initial stages of heat stress. Their work indicated that, under conditions of greatest heat stress, the rapid small-amplitude pumpings move little water over the gills.

A relationship between the two response variables was explored using partial correlation. Critical thermal maximum and opercular rate were significantly correlated across the treatment combinations ( $r = 0.49$ ,  $N = 57$ ), but the partial correlation among these responses, adjusted for acclimation-temperature, location, and water-hardness effects, was not significant. The observed association between the variates was attributable to acclimation temperature because, even though water hardness did have some effect on the physiological responses, it was of minor importance in relation to the effects of acclimation temperature.

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during the test period, dissolved oxygen was maintained close to saturation, and pH was 7.4 to 7.9. Water used in the laboratory was from Muddy Run Pumped Storage Pond, a receiving body of Conowingo Pond. This water was filtered, passed through an ultraviolet sterilizer and temperature-control units, and recycled.

Preliminary studies (Peterson, Schutsky, and Allison, 1974a) indicated a high incidence of myxobacteria, *Chondrococcus columnaris* and *Aeromonas* sp., in tests conducted at temperatures  $\geq 15.6^{\circ}\text{C}$ . Using the ultraviolet irradiating units effectively controlled the spread of the pathogens in subsequent tests; i.e., no mortality was observed in control specimens.

Partitioned transparent plexiglass aquariums with a filtered-water flow system were used to test the effects of rapid temperature change. The flow rate was approximately 5 liters/min. These aquariums were of two sizes, a large unit divided into four 30-cm<sup>3</sup> compartments and a smaller unit divided by removable framed nylon screens into eight 15-cm<sup>3</sup> compartments. Each compartment was aerated. An automatic timer was adjusted weekly to simulate the natural photoperiod, with 0.5-hr periods of dusk and dawn. A light level of 215 lux at the surface of the water was maintained by lamps provided with Vita-Lite fluorescent bulbs (Duro-Test Corp., North Bergen, N. J.), the spectral energy distribution of which was similar to natural daylight.

All tests were conducted using equal numbers (usually 10) of control and experimental specimens. Jensen (1972a) indicated that increasing sample size significantly reduced the standard error of an estimated  $LT_{50}$  (temperature lethal to one-half the specimens) until the sample size reached about 30 fish. However, the studies by Peterson, Schutsky, and Allison (1974a) suggest that crowding may adversely affect tests at acclimation temperatures  $\geq 15.6^{\circ}\text{C}$ .

The procedure used to determine lethal temperatures was similar to that used by Brett (1952) and later by Edsall and Colby (1970) and Edsall, Rottiers, and Brown (1970). Fish were transferred directly from the holding tank to plexiglass aquariums at the test temperature. Control fish were transferred in a similar manner to plexiglass aquariums at their acclimation temperature. The temperature of the test water was measured at the beginning of the test and at 1, 5, 10, 15, 30, and 45 min and 1, 2, 3, 4, 24, 48, 72, and 96 hr. Responses to rapid temperature change (loss of equilibrium and mortality) were recorded when water temperatures were taken. Additional temperatures were often recorded between 4 and 96 hr. Tests were terminated when all specimens died or at the end of 96 hr. Cessation of opercular movement was used as the criterion for death.

Jensen (1972a; 1972b) indicated that, to increase precision in estimating an  $LT_{50}$  for a small sample, test temperatures should be as close to the projected  $LT_{50}$  value as possible. We hypothesized that avoidance temperatures would approximate  $LT_{50}$ . Thus experimental temperatures at or near avoidance temperatures were selected on the basis of preliminary tests. Generally, test temperatures at 0.5 $^{\circ}\text{C}$  intervals higher and lower than the avoidance

temperature were used. Responses to sublethal (highest test temperature at which no death occurred), upper incipient lethal (test temperature at which 50% of the fish died), and ultimate upper lethal temperature (lowest test temperature at which 100% of the fish died) were determined for each acclimation temperature (Hickman and Dewey, 1973).

Mortality curves were determined by the method of Fry, Hart, and Walker (1946). Resistance times of specimens were plotted as accumulated percentage mortality on semilog-probability paper. Median resistance times were determined graphically for three acclimation temperatures (1, 13, and 27 $^{\circ}\text{C}$ ).

The preference unit used was a modification of that developed by Meldrim and Gift (1971). The unit consisted of a trough 3.7 m long, 22.9 cm wide, and 22.9 cm deep, with a bottom of 24-gauge, type 304 stainless steel. Water was introduced at one end of the trough from a temperature-controlled circulating bath. As the water flowed the length of the trough, it was heated by three banks of infrared bulbs located beneath the stainless-steel bottom to form a horizontal thermal gradient. Each bank consisted of four 250-watt bulbs connected to a dimmer switch and a temperature regulator. The water drained from the opposite end of the trough and was filtered, aerated, and returned to the temperature-controlled bath. Thus, by regulating the intensity of each of the light banks and the temperature of the water bath, we maintained the desired thermal gradient.

Lighting was provided by three Vita-Lites extending the length of the trough. Tests were conducted at 430 lux, measured at the surface of the water. The pH was determined at the conclusion of each test.

In a test, the trough was filled with water of the acclimation temperature to a depth of 2.5 to 5.0 cm to permit horizontal but restrict vertical movement of test fishes. Acclimation temperature is that at which a fish was collected and held in the laboratory before testing. Fish were placed in the center of the trough and allowed free movement for 10 to 30 min before a thermal gradient of as much as 22 $^{\circ}\text{C}$  was established. Observations were made at 10-min intervals, and the temperature at the position of each fish was recorded by the use of 23 thermistors spaced equidistant throughout the length of the trough and connected to a central readout. Median body position (Fahmy, 1972) was used to determine the temperature at the position of each fish. Temperature at the upper and lower end of the gradient was recorded at this time. The test was usually terminated when more than one-half the specimens selected the same temperature continuously for 30 min. If a preferred temperature was not selected within 3 hr, the test was terminated because the test fishes may have reacclimated to a temperature other than the original acclimation temperature.

The avoidance design used was a modification of that employed by Meldrim and Gift (1971). Two temperature-controlled circulating baths were used as storage tanks. Water from each bath flowed by gravity into opposite ends of the troughs, drained at the centers, and was filtered and

recirculated to the temperature baths. The 5- $\mu$  filter consisted of a polypropylene bag filled with activated charcoal. Dye tests with malachite oxylate showed a sharp boundary at the center drain. The apparatus was thus divided into thermally distinct quadrants.

The methods used follow those of Meldrim and Gift (1971). Equal numbers of fish were placed into each quadrant. Two diagonally opposite quadrants contained water of the acclimation temperature (T) while the remaining quadrants contained water of increased temperature (T+). After a 5- to 30-min orientation period, the amount of time spent by each fish in each quadrant was measured for a trial period of 10 min. The number of times a fish entered each quadrant was multiplied by the amount of time it spent there to give a frequency distribution for each quadrant. A two-tailed t-test was used to determine if a significant difference ( $P \leq 0.05$ ) existed between the two frequency distributions. If both the (T+) quadrants were significantly avoided, the test was terminated. If the avoidance of one or both of the T+ quadrants was nonsignificant, the T and T+ temperatures were increased 2 to 3°C, and the same fish were then tested at the new temperatures.

Oxygen and pH were measured at the beginning and end of each test. The thermal conditions were monitored by a 24-channel temperature recorder connected to thermocouples at 20-cm intervals along the inner sides of both troughs.

A closed-circuit television system was used for observations. This allowed the unit to be completely enclosed for light control and permitted movement around the trough. The tests were recorded on videotape for subsequent analysis with the temperature-recorder printout. Tests were conducted at a light level of 430 lux at the water surface.

## RESULTS AND DISCUSSION

A typical mortality curve for bluegill is presented in Fig. 1. Complexity of mortality curves increased with resistance time. Complex curves were best represented by several straight lines (Fig. 1). A similar relationship was found by Edsall and Colby (1970) in temperature tolerance studies with the cisco, and by Tyler (1966) in a review of temperature tolerance studies on fishes and invertebrates.

The relationship between median resistance time, lethal temperature, and acclimation temperature is shown in Fig. 2. Median resistance temperature, considered to be the upper incipient lethal temperature for a 96-hr exposure, is represented by the point on each curve where resistance time exceeds 96 hr.

Sublethal, upper incipient lethal, and ultimate upper lethal temperatures were directly proportional to acclimation temperature (Table 1) and increased 0.5°C for each 1°C increase in acclimation temperature. The results of our study were compared with those of other investigators by interpolation of data presented in Table 1. Upper incipient lethal temperatures for bluegill acclimated to 20 and 22°C agree ( $\pm 1^\circ\text{C}$ ) with those reported by Hart (1952) (31.5°C)

and Hathaway (1927) (31°C). The results also agree ( $\pm 1^\circ\text{C}$ ) with upper incipient lethal temperatures (27.5, 33, and 36°C) reported by Banner and Van Arman (1973) for bluegill acclimated to 12.1, 19, and 26°C, respectively. Ultimate upper lethal temperatures (33 and 33.9°C) for bluegill acclimated to 20 and 22°C were similar to those reported by Hickman and Dewey (1973) (33.5°C) and Hathaway (1927) (34°C). Christianson and Tichenor (1968) also reported similar 24-hr upper incipient lethal temperatures (31 and 34°C) for bluegill acclimated to 15 and 30°C.

Preference and avoidance data were analyzed separately by a stepwise multiple regression to determine relationships with acclimation temperature (X) and mean total length of fish ( $X_1$ ). The acclimation temperature accounted for most of the variation in the dependent variable in preference and avoidance studies. Although the variance contributed by the length of the fish was significant ( $P \leq 0.05$ ), inclusion of the variable in the multiple regression equation resulted in only a slight reduction in the standard error of the estimate. Consequently the 95% confidence intervals on the population mean of the preferred and avoided temperatures were computed excluding length from the estimates. Confidence intervals were plotted with temperature-tolerance data (Fig. 3).

The equation that best describes the preference temperature of the bluegill is

$$\begin{aligned} Y_P &= 18.918 + 0.437X \\ R^2 &= 0.785 \\ s_{y \cdot x} &= 1.762 \end{aligned} \quad (1)$$

The avoidance temperature is best described by the following equation:

$$\begin{aligned} Y_A &= 27.392 + 0.226X \\ R^2 &= 0.443 \\ s_{y \cdot x} &= 2.043 \end{aligned} \quad (2)$$

Preference temperatures were directly proportional to acclimation temperatures and consistently lower (2.8 to 8.2°C) than avoidance temperatures (Table 1). We should note, however, that preferred temperatures corresponding to acclimation temperatures  $< 8^\circ\text{C}$  could not be determined because of the high incidence of low thermal responsiveness at lower acclimation temperatures. Low thermal responsiveness (LTR), which was defined by Meldrim and Gift (1971) as the inability of a fish to avoid areas in a thermal gradient which produce stressful conditions, has been observed with many species of fish at various acclimation temperatures.

At acclimation temperatures  $< 20^\circ\text{C}$ , fish often showed no distinct preference but continued to select the highest temperature available. This normally led to LTR if the temperature differential in the gradient approached or exceeded the predicted LTR temperatures (Fig. 4). We found a similar response in four other centrarchids tested from Conowingo Pond: green sunfish (*Lepomis cyanellus*),

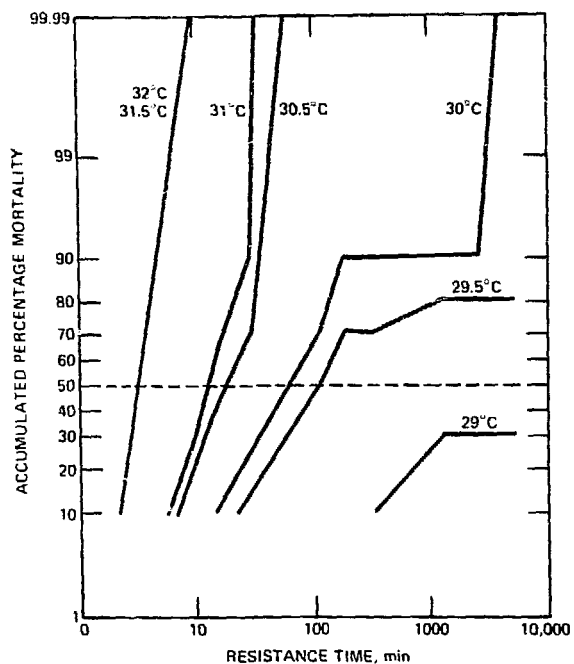


FIG. 1 Resistance times of bluegill (*Lepomis macrochirus*) acclimated to 13°C and exposed to lethal temperatures from 29 to 32°C. Fish exposed to 27 to 28°C survived the 96-hr exposure.

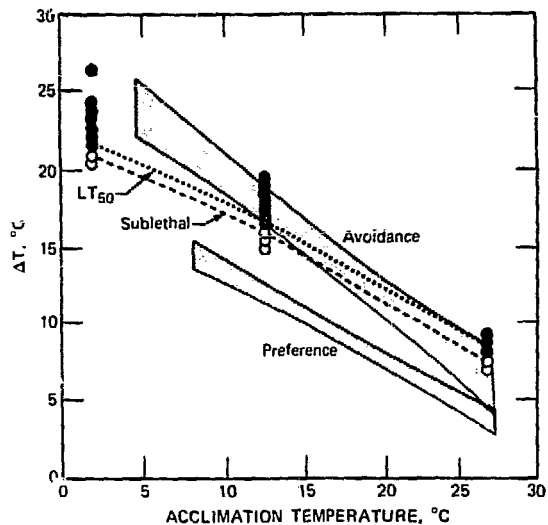


FIG. 3 Relationship between thermal resistance of bluegill (*Lepomis macrochirus*) and preference and avoidance temperatures, expressed as 95% confidence intervals of the population mean. Black circles indicate mortality within a test.

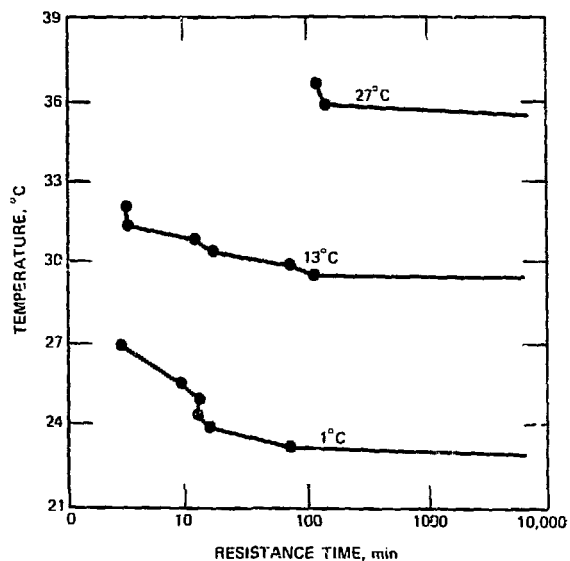


FIG. 2 Median resistance times of bluegill (*Lepomis macrochirus*) acclimated to 1, 13, and 27°C.

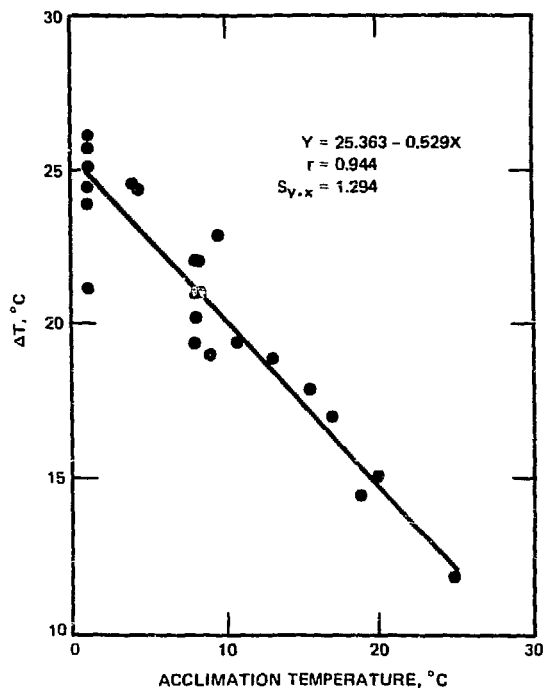


FIG. 4 Calculated regression line showing the relationship of low thermal responsiveness in bluegill (*Lepomis macrochirus*) to acclimation temperature and temperature differential (°C).

TABLE 1 Effect of Acclimation Temperature on Upper Temperature Tolerance of Bluegill (*Lepomis macrochirus*)

| Acclimation temperature, °C | Upper temperature tolerance, °C |                        |                       | Calculated temperature, °C |           |                            |
|-----------------------------|---------------------------------|------------------------|-----------------------|----------------------------|-----------|----------------------------|
|                             | Sublethal                       | Upper incipient lethal | Ultimate upper lethal | Preference                 | Avoidance | Low thermal responsiveness |
| 27.0                        | 35.0                            | 35.8                   | 36.0                  | 30.7                       | 33.5      |                            |
| 13.0                        | 28.0                            | 29.3                   | 30.0                  | 24.6                       | 30.3      | 31.5                       |
| 1.0                         | 22.0                            | 23.3                   | 23.5                  |                            | 27.6      | 25.8                       |

pumpkinseed (*Lepomis gibbosus*), smallmouth bass (*Micropterus dolomieu*), and largemouth bass (*Micropterus salmoides*). It also occurred in avoidance tests where fish showed LTR prior to or coincidental with significant avoidance.

The relationship between thermal resistance of bluegill and preference and avoidance temperatures is presented in Fig. 3. Preference and avoidance temperatures were expressed as the 95% confidence intervals of the population mean. The median resistance temperature ( $LT_{50}$ ) paralleled ( $1 \pm 0.3^\circ\text{C}$ ) and sublethal temperature and was within the 95% confidence intervals for avoidance temperature when acclimation temperatures were  $\geq 13^\circ\text{C}$ . Under this condition, the avoidance temperatures were an estimate of the upper incipient lethal temperatures for bluegill. Avoidance temperatures were not a reliable estimate of upper incipient lethal temperatures for bluegill acclimated to temperatures  $< 13^\circ\text{C}$ .

At acclimation temperatures  $< 13^\circ\text{C}$ , upper lethal temperatures can be estimated by a regression analysis of LTR. This analysis was performed, comparing the temperature differential above acclimation within a preference or avoidance test at which LTR first occurred (Y) with the acclimation temperature of the specimen (X). A high correlation ( $r = 0.94$ ) was found between the two variables, with the following equation best describing the relationship:

$$\begin{aligned} Y &= 25.363 - 0.529X \\ R^2 &= 0.891 \\ s_{y \cdot x} &= 1.294 \end{aligned} \quad (3)$$

Estimates of LTR temperatures based on this relationship were determined for two acclimation temperatures (Table 1 and Fig. 4). The LTR temperatures varied directly with acclimation temperature, increased  $0.5^\circ\text{C}$  for each  $1^\circ\text{C}$  increase in acclimation temperature, and were  $3.5$  to  $3.8^\circ\text{C}$  above sublethal,  $2.2$  to  $2.5^\circ\text{C}$  above upper incipient lethal, and  $1.5$  to  $2.3^\circ\text{C}$  above ultimate upper lethal temperatures.

If certain ranges of avoidance temperatures are reliable estimates for  $LT_{50}$ , then LTR temperatures must exceed the avoidance temperatures in that range. Indeed, at acclimation temperatures  $\geq 13^\circ\text{C}$ , where avoidance temperatures are estimates of  $LT_{50}$ , the LTR temperatures

exceed the avoidance temperatures by at least  $1.2^\circ\text{C}$ . In contrast, when avoidance temperatures are not reliable estimates of  $LT_{50}$ , LTR temperatures are less than avoidance temperatures. At an acclimation temperature of  $1^\circ\text{C}$ , where avoidance temperatures are not a reliable estimate of  $LT_{50}$ , the LTR temperature is  $1.8^\circ\text{C}$  less than the avoidance temperature. Despite the increased frequency of LTR and the apparent lack of thermal tolerance at low acclimation temperature, the estimated temperature differential to elicit an LTR response with bluegill acclimated to  $1^\circ\text{C}$  is  $24.8^\circ\text{C}$ . Thus temperatures that elicit LTR and avoidance are closely related.

In conclusion, bluegill actively avoided lethal temperatures when acclimation was  $\geq 13^\circ\text{C}$ . At acclimation temperatures  $\geq 13^\circ\text{C}$ , the  $LT_{50}$  could be estimated from the avoidance temperature ( $Y_A = 27.392 + 0.226X$ ). Sublethal and ultimate upper lethal temperatures were within the boundaries of the 95% confidence intervals of the population mean of avoidance temperature.

At acclimation temperatures  $< 13^\circ\text{C}$  avoidance temperatures were not reliable estimates of  $LT_{50}$ . However,  $LT_{50}$  could be estimated from analysis of temperatures required to elicit LTR. A potential problem, however unlikely, could exist in a field situation if the instantaneous temperature differential experienced by the bluegill approached or exceeded  $18.5^\circ\text{C}$ . The maximum discharge temperature change predicted at Peach Bottom Atomic Power Station Units 2 and 3 is  $11.7^\circ\text{C}$ . Thus, even when acclimation temperatures indicate the potential for LTR, the temperature differential would not be sufficient to induce such a response.

## ACKNOWLEDGMENTS

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# Further Studies of Cold-Shock Effects on Susceptibility of Young Channel Catfish to Predation

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## ABSTRACT

Previous experiments indicated that rapid temperature decreases of greater than  $\sim 6^{\circ}\text{C}$  tend to increase the susceptibility of juvenile channel catfish to predation by adult largemouth bass. This report presents confirmatory results with another stock of catfish and different predators. The rate of cooling is also shown to influence this measure of temperature stress. Cooling rates slower than  $1.5^{\circ}\text{C}/\text{min}$  did not induce selective predation on test fish, but rates of  $2^{\circ}\text{C}/\text{min}$  and instantaneous cooling resulted in significant selective predation over controls.

In this study we sought (1) to confirm earlier results (Coutant, Ducharme, and Fisher, 1974) which showed that instantaneous temperature decreases greater than about  $6^{\circ}\text{C}$  tended to increase the susceptibility of juvenile channel catfish (*Ictalurus punctatus*) to predation by adult largemouth bass (*Micropterus salmoides*), (2) to confirm and explain apparent decreased susceptibility to predation after low-magnitude cold shocks (Coutant, Ducharme, and Fisher, 1974), and (3) to determine whether the rate of cooling of catfish prey would influence this measure of temperature stress. Fish mortalities resulting from cold shock when the heat source is suddenly terminated at power plants could be minimized or eliminated if the tolerance of prey species to temperature decline was known with a high degree of certainty and this information used for modifications in power-plant design or operation (American National Standards Institute, 1974)

To adequately assure protection of fish, we must know tolerable decreases in temperature, not only for direct death but also for sublethal stresses that may markedly decrease long-term survival. These experiments simulated field conditions in which juvenile fishes that have resided in a warmed area long enough to become acclimated to the high temperatures disperse into surrounding waters when the heated discharge terminates. The small fish then encounter predators that have not themselves experienced the temperature drop.

## MATERIALS AND METHODS

We tested the relative survival of cold-shocked and unshocked prey by offering groups of the two simultaneously to adult largemouth bass and allowing about half of the combined number to be eaten. The protocol for predation was identical to that reported by Coutant, Ducharme, and Fisher (1974). The cold-shock exposure was modified in some tests to allow predetermined rates of cooling from the acclimation temperature to the predation temperature.

Channel catfish used as prey, obtained from the U. S. Fish and Wildlife Service, Cohutta National Fish Hatchery, Cohutta, Ga., averaged  $3.4 \pm 0.3(\text{SD})$  g in weight and  $7.9 \pm 0.4$  cm in length at the beginning of acclimation. Predator bass were captured by electrofishing from Watts Bar Reservoir, which is near our laboratory, and were maintained in laboratory tanks for several months before the tests.

Juvenile channel catfish from a common stock were randomly separated into groups, differentially marked, and held for more than 1 week in 1.2-m diameter cylindrical fiber glass tanks with water depths of 65 cm. One tank each was used for several warm acclimation temperatures and one for controls at the predation temperature ( $16$  to  $17^{\circ}\text{C}$ ). Fish were marked using a liquid nitrogen cold-branding technique (Coutant, 1972), test fish on one side and control fish on the other. In the first series of tests, which were directed at the first two objectives, test and control fish were mixed in groups of 20 each at the predation temperature and added simultaneously to the predator tank. The predator tank was identical to the acclimation tanks and contained four to five largemouth bass averaging about 1500 g. In the second series, warm-acclimated prey ( $30^{\circ}\text{C}$ ) were cooled at predetermined rates by adding cold water to a tank that initially contained water and test fish

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\*Undergraduate research participant through agreements with the Great Lakes Colleges Association.

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at the acclimation temperature. When the predation temperature (17°C) was reached, test and control fish (20 each) were mixed and added simultaneously to the predator tank.

After introduction of the prey, predators were allowed a maximum of 0.5 hr to eat approximately 50% of the prey fish. Then survivors were removed, and test and control fish were counted, weighed, and measured. The statistic used to express the difference in predation rates was the ratio  $d_p = i_1/i_2$  (Bams, 1967), where  $i_1$  and  $i_2$  are the instantaneous mortality rates of the test and control groups, respectively. The instantaneous mortality rate when time is a unit interval is given by  $i = -\ln s$ , where  $s$  is the survival proportion (i.e., number at the finish divided by number at the start). Group  $d_p$  was calculated for replicates by summing the numbers  $s$ . The chi-square analysis was used to test for homogeneity among replicate trials and for significance of observed differences in predation.

Because performance could vary with size, length and weight are analyzed as parameters that could affect rates of predation in this study. Test fish tended to be somewhat larger than controls because of more rapid growth at the elevated acclimation temperatures. To determine whether size affects predation rate, we reacclimated some test fish not used in the cold-shock predation tests to ambient temperature in their respective holding tanks for 2 weeks. Predation tests were then conducted on the reacclimated fish, the procedure being the same as in previous tests but without cold shock. The only variable in these tests was size difference, if we assume that reacclimation had no adverse effects on the test fish. Lengths and weights of survivors were immediately determined for each test.

Temperatures for all tanks were maintained within  $\pm 0.2^\circ\text{C}$  of the desired temperature by a control valve that

mixed hot, cold, and ambient-temperature water supplies at a flow rate of 6.0 liters/min. Artificial lighting was regulated on a 12-hr photoperiod.

## RESULTS

### Instantaneous Cold Shock

Channel catfish fed to predators immediately after a temperature decrease were preyed upon in greater numbers ( $P < 0.05$ ) than controls when the cold shock was 6 and  $10^\circ\text{C}$  (Table 1). A  $2^\circ\text{C}$  cold shock caused the shocked fish to be preyed upon in smaller numbers than controls (not significant,  $P > 0.05$ ).

The ratios of predation rates tended to increase above 1 (no preference) when the cold shock exceeded about  $3^\circ\text{C}$  (Fig. 1). Thereafter the increase was exponential. The  $d_p$  ratio had a tendency to decrease below 1 when the cold shock was less than  $3^\circ\text{C}$ .

Surviving catfish from test and control groups showed different weights and lengths in each trial (Table 2). The shocked fish were always slightly larger than the control fish.

Differential predation trials using test fish reacclimated to the ambient temperature showed that only the former  $2^\circ\text{C}$  test group was significantly ( $P < 0.05$ ) preyed upon in lesser numbers than controls (Table 3). The survivors of both reacclimated and control groups showed a difference in weight and length; reacclimated fish being larger than control fish.

### Prescribed Cooling Rates

Juvenile catfish cooled from  $30$  to  $17^\circ\text{C}$  at a rate of  $2.0^\circ\text{C}/\text{min}$  and those instantaneously cooled were significantly ( $P < 0.05$ ) more susceptible to predation than

TABLE 1 Average Percent Survival\* (Combined for Replicates) and Ratios of Instantaneous Predation Rates ( $d_p$ ) for Tests with Juvenile Channel Catfish Fed to Adult Largemouth Bass After Different Magnitudes of Cold Shock

| Magnitude of cold shock   | Number of replicates | Average percent survival |         | Combined chi square | Combined d <sub>p</sub> |
|---------------------------|----------------------|--------------------------|---------|---------------------|-------------------------|
|                           |                      | Shocked                  | Control |                     |                         |
| Instantaneous shock, † °C |                      |                          |         |                     |                         |
| 0                         | 5                    | 61‡                      | 60‡     | 0.38                | 0.96                    |
| 2                         | 7                    | 59                       | 54      | 0.70                | 0.86                    |
| 4                         | 5                    | 58                       | 69      | 2.61                | 1.45                    |
| 6                         | 5                    | 44                       | 62      | 6.51 §              | 1.72                    |
| 10                        | 4                    | 43                       | 85      | 32.2 §              | 5.27                    |
| Rate of cooling, ¶ °C/min |                      |                          |         |                     |                         |
| 0.6                       | 5                    | 57                       | 45      | 1.40                | 0.58                    |
| 0.8                       | 5                    | 63                       | 54      | 3.48                | 0.74                    |
| 1.5                       | 5                    | 59                       | 61      | 0.03                | 1.05                    |
| 2.0                       | 5                    | 32                       | 57      | 7.02 §              | 2.02                    |
| Instantaneous             | 5                    | 46                       | 72      | 7.12 §              | 2.36                    |

\*From 20 fish unless otherwise noted.

†To predation temperature of 16 to  $17^\circ\text{C}$ .

‡These tests had 15 fish at the start.

§Difference from equal numbers of survivors significant at  $P < 0.05$ .

¶From  $30^\circ\text{C}$  to predation temperature of  $17^\circ\text{C}$ .



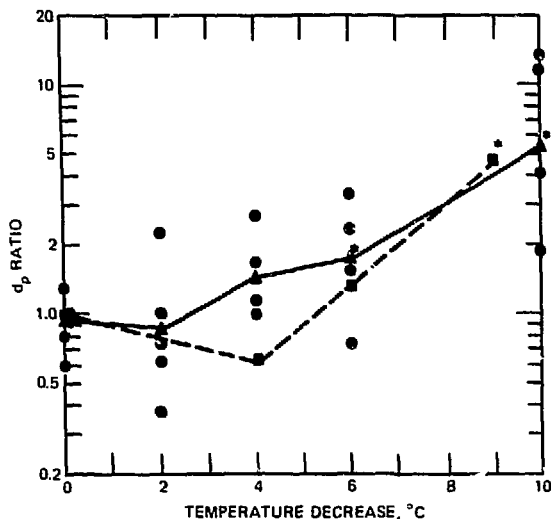


FIG. 1 Ratios of experimental instantaneous predation rates ( $d_p$ ) on cold-shocked and control juvenile channel catfish receiving temperature decreases shown (to a base temperature of 16°C). ●, individual tests with 20 shocked and 20 control fish. ▲, combined replicates. Asterisks (\*) indicate predation on test fish significantly greater ( $P < 0.05$ ). Earlier published data (■—■) are shown for comparison.

TABLE 2 Average Weights and Lengths\* ( $\pm$ SD) of Surviving Juvenile Channel Catfish

| Magnitude of cold shock, °C | Average weight, gm |               | Average length, cm |               |
|-----------------------------|--------------------|---------------|--------------------|---------------|
|                             | Shocked            | Control       | Shocked            | Control       |
| 2                           | 3.7 $\pm$ 0.5      | 3.3 $\pm$ 0.4 | 8.0 $\pm$ 0.4      | 7.8 $\pm$ 0.4 |
| 4                           | 3.6 $\pm$ 0.5      | 3.1 $\pm$ 0.6 | 7.9 $\pm$ 0.3      | 7.6 $\pm$ 0.4 |
| 6                           | 3.5 $\pm$ 0.8      | 3.1 $\pm$ 0.7 | 7.8 $\pm$ 0.6      | 7.6 $\pm$ 0.5 |
| 10                          | 5.4 $\pm$ 0.8      | 4.5 $\pm$ 0.5 | 8.4 $\pm$ 0.4      | 7.8 $\pm$ 0.4 |

\*Measurements made on first and last tests of each cold-shock temperature.

uncooled controls (Table 1). Rates of 1.5°C/min and less did not induce statistically significant selective predation. Anomalous results exhibiting excessive variability were obtained at a cooling rate of 0.3°C/min and are not considered valid (Coutant et al., 1974). Differential predation rates expressed as  $d_p$  ratios showed close similarity between 2°C/min and instantaneous cooling, no average differential predation at a rate of 1.5°C/min, and a slight trend toward higher susceptibility of controls at rates of 0.6 and 0.8°C/min (Fig. 2).

## DISCUSSION

Results of tests with instantaneous cold shock confirm those of earlier studies (Coutant, Ducharme, and Fisher, 1974) that increasing cold shock on juvenile channel catfish decreases their ability to survive predation (Fig. 1). Most significantly, there is a debilitation of catfish at cold shocks

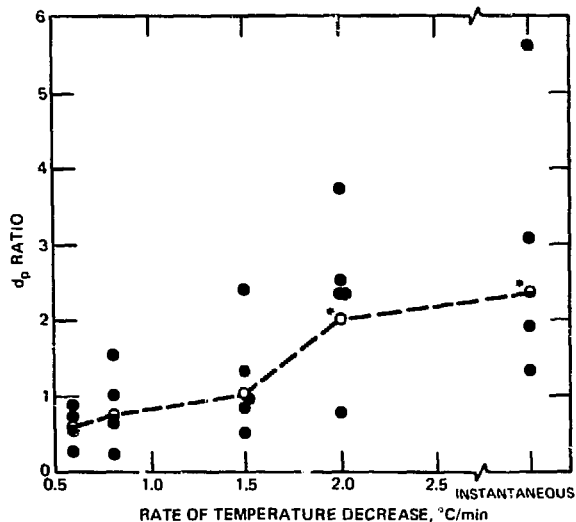


FIG. 2 Differential predation rates of adult largemouth bass on young channel catfish cooled at known rates compared with uncooled fish. A  $d_p$  of 1 indicates equal vulnerability. ●, individual tests. ○, combined replicates. Asterisk (\*) indicates significant inequality of predation ( $P < 0.05$ ).

TABLE 3 Differential Predation at 15°C on Reacclimated Test Fish and Control Fish After No Cold Shock

| Group number* | Combined chi square | Combined $d_p$ ratio† | Average weight, gm |         | Average length, cm |         |
|---------------|---------------------|-----------------------|--------------------|---------|--------------------|---------|
|               |                     |                       | Reacclimated       | Control | Reacclimated       | Control |
| 2             | 6.71‡               | 0.38 (2)              | 3.8                | 3.5     | 8.0                | 7.9     |
| 4             | 1.36                | 1.55 (2)              | 3.9                | 3.8     | 8.1                | 7.9     |
| 6             | 0.61                | 1.39 (2)              | 4.1                | 3.4     | 8.5                | 8.0     |
| 10            | 0.40                | 0.75 (1)              | 5.5                | 4.1     | 8.9                | 8.1     |

\*Indicates former cold-shock group.

†Numbers in parentheses are numbers of replicates.

‡Difference from equal numbers of survivors significant at  $P < 0.05$ .

of 6°C or more. Catfish subjected to low-level cold shocks less than 3°C showed a consistent but statistically non-significant trend toward decreased susceptibility to predation. This trend is also similar to earlier results. Absolute differences between these tests and earlier ones are probably due to slight deviations in experimental procedure or to the genetic pool of catfish. Low-level cold shocks might, as Coutant, Ducharme, and Fisher (1974) suggested, provide "nervous" stimuli that enable the fish to become more efficient in escaping predators.

Slight size differences between test and control fish might also account for the consistent trend shown at low-magnitude cold shocks. The former 2°C cold-shock group showed a significant predation ratio of less than 1, indicating that the larger reacclimated catfish were less susceptible to predation because of size difference only (Table 3). Groups originally acclimated to 4, 6, and 10°C above ambient temperature did not reveal any significant differential predation regardless of size difference, but these tests involved some statistical problems. The predators in these tests were consuming only about 35% of the initial prey (50% consumption is the most sound statistically). The catfish had grown to a size that made predation difficult.

The difference in fish length was probably a valid factor in this study and contributed to selection of prey. The smaller the prey, the more susceptible it was to predation. The  $d_p$  ratio significantly below 1 for a 2°C cold-shocked fish may, therefore, have resulted from lower susceptibility of test catfish to predation because of larger size. At low-level cold shocks, size difference may be the major factor determining susceptibility to predation. At higher magnitudes of cold shock, however, the effect of size differences is masked by the temperature effect. If this hypothesis is correct, low levels of cold shock on smaller fish would not provide a stimulus for a more efficient escape from predators. Further experimentation is needed to determine the exact relationship of size to predation.

Cooling rate appears to have a decided effect on thermal stress as represented by behavioral and/or performance changes that are reflected in susceptibility to predation. Although we tested only one temperature range (30 to 17°C), the effect may be a more universal one. For the 13°C range tested, rates slower than about 1.5°C/min seem to protect the young fish from cold-shock stress. This rate is much slower than initial cool-down rates of about 7°C/min calculated by Pilati (1973) for certain onshore discharges to large lakes in which the fish are assumed to follow the maximum available temperature as it disperses into the lake. Actual cool-down rates recorded at a stationary point in such a discharge spanned nearly the full 16°C temperature differential in 5 min when circulating pumps continued to operate after the heat source was abruptly terminated (Consumers Power Company, 1973). Most of that decline occurred within a 1-min period.

In the earlier work Coutant, Ducharme, and Fisher (1974) determined that "catfish held 1 hr in the cold water

were only slightly less susceptible to predation than were others tested immediately after the temperature change." This observation seems to be at variance with the results of rate-of-change experiments reported here. The critical difference between these two experiments, however, was the actual rate of cooling. In the earlier experiments cooling rate was instantaneous but predation was delayed; the experiments reported here evaluated different rates of slower cooling. Physiological accommodation assisting survival must, therefore, occur during the cooling period and not afterward.

These experiments have further expanded our knowledge of cold-shock stresses that can be tolerated by one important warm-water fish species. Comparing these results with other studies of young largemouth bass (Coutant, Ducharme, and Fisher, 1974) and bluegill (Wolters and Coutant, this volume), which yielded very similar patterns of cold-shock tolerance, we may be justified in considering these data as representative of warm-water fishes. This information would be useful to the electric-power industry for formulating guidelines in relation to siting, design, and operation of power stations so as to minimize the likelihood of local cold-shock damage.

The experiments to date have not, however, adequately addressed the question of significance of cold-shock damages to fish populations. The reported cold-shock fish kills (Coutant, Ducharme, and Fisher, 1974; Ash, Chymko, and Gallup, 1974), although often spectacular to the observer, may be of little ecological consequence. Conversely, the direct kill or unobserved increases in predation on cold-shocked prey species in the field may be highly significant for some populations and ecosystems. Further cold-shock analysis must consider the dynamics of fish populations affected by this source of potential ecological damage.

## ACKNOWLEDGMENTS

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# Acute Cold-Shock Resistance of Gizzard Shad

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## ABSTRACT

Groups of 10 adult gizzard shad were given acute cold shocks at 1, 2, 4, 6, 6.5, or 7°C after acclimating for at least 2 weeks to 15, 17.5, or 20°C. Time to loss of equilibrium and to cessation of opercular movement was determined. Shad acclimated between 15 and 20°C had a 10% lethal threshold of about 6 to 7°C for acute cold shock. Time to death (cessation of opercular movement) was much longer than time to equilibrium loss and would appear to have little ecological significance. Test temperatures of 1, 2, 4, and 6°C yielded mean time to loss of equilibrium and cessation of opercular movement. Regression lines from these data illustrate the time-dependent cold shock response of gizzard shad populations when acclimated to temperatures of 15 to 20°C.

During cool months, gizzard shad congregate in warmer areas of Tennessee power-plant discharges (Coutant, 1975). Fish kills of gizzard shad and other species resulting from cold shock have occurred when such an artificial heat source was interrupted (Norris and Gammon, 1972; Agerborg, 1930; Coutant et al., 1974). Since it is well-known that acclimation to heat occurs more rapidly than to cold (Prosser and Brown, 1961), fish populations that are attracted to these warmer waters become increasingly susceptible to sudden chilling. The objective of this study was to characterize the effects of cold shock on gizzard shad in terms of temperature and duration of exposure required to induce loss of equilibrium and cessation of opercular movement, using acclimation temperatures common to wintertime thermal discharges. Knowledge of tolerance limits of this susceptible species could produce design criteria for minimizing cold shock damages at power stations.

## MATERIALS AND METHODS

The resistance time response method described by Fry et al. (1946) and Brett (1952) was employed. Test populations of gizzard shad were collected by electrofishing in late February and early March from the discharge area of the Tennessee Valley Authority's Bull Run Steam Plant, a 950-Mw(e) coal-burning steam electric station on the Clinch River, Tennessee, United States. The temperature rise for a condenser flow of 1478 m<sup>3</sup>/min is about 10°C. Water

temperatures in the discharge cove and canal at collection times ranged from 17.4 to 19.4°C.

Groups of 10 adult shad having an average length of  $21.4 \pm 0.8$  (SD, standard deviation) cm and weight of  $147 \pm 34$  g were given an acute (instantaneous) cold shock to test temperatures of 1, 2, 4, 6, 6.5, or 7°C following acclimation for at least 2 weeks at 15, 17.5, or 20°C. Individual shad were observed at test temperatures to determine time to initial loss of equilibrium. Each fish was then marked by perforating a fin with a paper punch. The marked fish was then held in a shallow floating cage in the same test tank at the same test temperature to determine time to loss of opercular movement.

The three acclimation groups were maintained in circular tanks (1.2 m diameter) having a water depth of 65 cm. Acclimation temperatures were achieved by mixing 14°C ambient and 40°C heated well water. These temperatures were maintained within  $\pm 0.5^\circ\text{C}$  of target values. The flow rate to each acclimation tank was about 15 liters/min. The shad fed on the algae growing on the sides of the tank and were also provided fine trout chow.

A portable refrigeration unit coupled with a temperature controller was used to regulate test temperatures. A slow flow rate ( $<1$  liter/min) of prechilled well water at a temperature 1 or 2°C above test temperature was used to load the portable refrigeration system. This resulted in maintenance of test temperatures within  $\pm 0.2^\circ\text{C}$  of the desired value.

The data were summarized by least-squares regression lines with appropriate 95% confidence bands. The statistical test of residual sum of squares (Draper and Smith, 1966) was employed to test the differences among the regression lines for the three temperature acclimation groups. Significant differences ( $P < 0.05$ ) were identified by Duncan's New Multiple Range Test with unequal numbers of replications.

## RESULTS AND DISCUSSION

The responses of gizzard shad to various degrees of cold shock ranged from no apparent effect to complete mortality (Table 1). Approximately 10% mortality occurred near

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TABLE 1 Equilibrium Loss (LEQ) and Cessation of Opercular Movement (COM) in Gizzard Shad Following Acute Cold Shock; Response Is Indicated by Numbers of Fishes from the Initial 10\*

| Test temp., °C | Acclimation temp. of 15°C |     | Acclimation temp. of 17.5°C |     | Acclimation temp. of 20°C |     |
|----------------|---------------------------|-----|-----------------------------|-----|---------------------------|-----|
|                | LEQ                       | COM | LEQ                         | COM | LEQ                       | COM |
| 7.0            | 0                         | 0   | 0                           | 0   | 2R                        | 0   |
| 6.5            | 0                         | 0   | 5R                          | 3   | 10R                       | 10  |
| 6.0            | 10R                       | 1   | 10R                         | 9   | 10                        | 10  |
| 4.0            | 10                        | 10  | 10                          | 10  | 10                        | 10  |
| 2.0            | 10                        | 10  | 10                          | 10  | 10                        | 10  |
| 1.0            | 10                        | 10  | 10                          | 10  | 10                        | 10  |

\*Subscript R indicates individuals that experienced several cycles of equilibrium loss and revival over a 96-hr period.

6°C for shad acclimated to 15°C. This 10% lethal threshold moved up the temperature scale with an increase in acclimation temperature. The 10% lethal threshold thus approached 7°C for shad acclimated to 20°C. The threshold for initial equilibrium loss was slightly higher than the 10% lethal temperature for each acclimation group. Shad were observed to revive after loss of equilibrium when cold exposure was near the threshold. Some individuals experienced several cycles of equilibrium loss and revival before either entering a prolonged coma before death or completely regaining equilibrium. The timing of these cycles and of the identification of permanent equilibrium loss was not determined.

The timing of initial equilibrium loss and death was a function of exposure and acclimation temperatures (Figs. 1 and 2). Equilibrium loss generally occurred within seconds while death (cessation of opercular movement) occurred after many hours of nearly motionless cold coma. Times to equilibrium loss and death were shortest at the coldest test temperatures. There was not a great difference in resistance times among acclimation temperatures, but a trend toward longer survival at lower acclimation temperature could be seen (Fig. 2). The probability of difference between the three regression lines for loss of equilibrium was less than 0.75 but more than 0.5. The 20°C regression line for cessation of opercular movement differed ( $P > 0.05$ ) from the 15 and 17.5°C lines (Duncan's New Multiple Range Test). Our regression lines for times to equilibrium loss or death may be used in predicting the impact of cold shock on gizzard shad populations when the temperature of the warmed discharge area (the presumed acclimation temperature) and the ambient temperature that would prevail with the plant not operating (test temperature) are known.

The resistance time to death is a well-established parameter in scientific literature but when given alone is inadequate for evaluating the environmental effects of thermal discharges. Equilibrium loss dose (ELD) represents a much more critical thermal dose for survival of the organism, for it drastically reduces the ability to escape

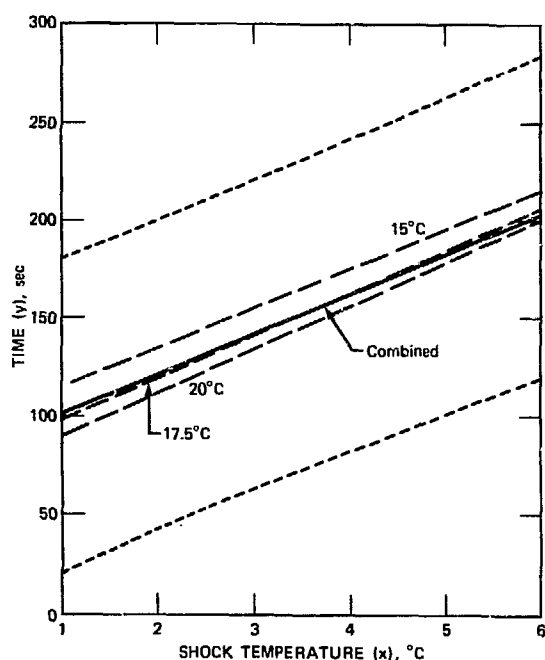


FIG. 1 Acute cold shock resistance of gizzard shad, *Dorosoma cepedianum*, as loss of equilibrium. Shad were acclimated in the laboratory for more than 2 weeks to 15, 17.5, and 20°C. Resistance times for respective acclimation temperatures are summarized as the regression of  $y$  on  $x$ :  $\hat{y} = 95.5 + 18.1x$ ;  $\hat{y} = 78 + 21.4x$ ;  $\hat{y} = 67.3 + 22.2x$ . Separate regression lines differed only within the  $0.5 < P < 0.75$  limits of probability. Resistance times for the combined acclimation temperatures are estimated by the solid regression line with a 95% confidence band for a new observation of  $y$ .

from the thermal conditions that are causing distress and imminent death (Cowles and Bogert, 1944). Physiological death dose (cessation of opercular movement) for gizzard shad in this study occurred many hours after ELD. This differs from the results found for heat shock of juvenile

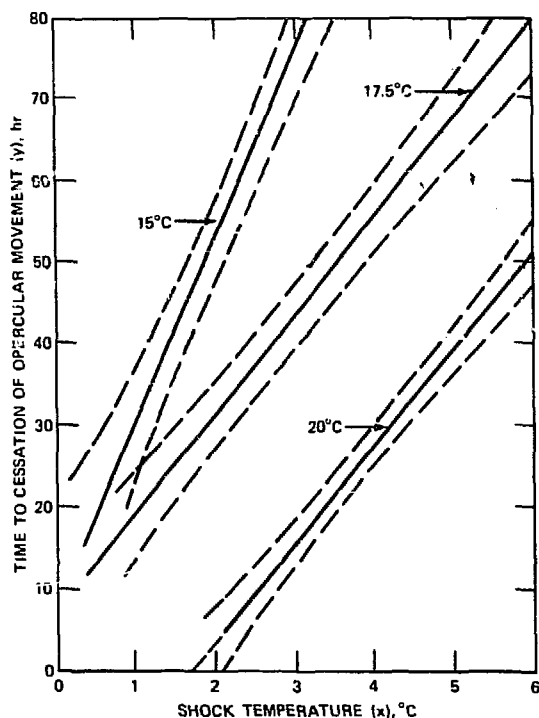


FIG. 2 Acute cold shock resistance of gizzard shad, *Dorosoma cepedianum*, as cessation of opercular movement. Shad were acclimated in the laboratory for more than 2 weeks to 15, 17.5, and 20°C. Resistance times for respective acclimation temperatures are summarized as the regression of  $y$  on  $x$ :  $\hat{y} = 5.1 + 23.3x$ ;  $\hat{y} = 6.8 + 12.2x$ ;  $\hat{y} = -20.2 + 12.3x$ . The 20°C regression line differed ( $P > 0.05$ ) from the 15 and 17.5°C lines. Regressions are illustrated with 95% confidence bands for the line.

chinook salmon and rainbow trout by Coutant and Dean (1972). They found ELD and death to occur close together at the higher lethal temperatures. The physiological death dose for gizzard shad in this study occurred long after the first clearly recognizable behavioral response to cold shock; hence it is of doubtful ecological significance. The ELD is not always the most critical criterion for judging potential ecologically significant thermal stress. Juvenile fishes receiving shock amounting to only a small fraction of that required to produce visible loss of equilibrium suffered stress sufficient to incite selective predation (Coutant, 1973). Mossman (1955) suggested that behavior associated with weakness may act as a stimulus for predator attack. Since adult gizzard shad are too large to be good prey to all but very large predators, their susceptibility to predation is of minor importance. However, during exposure to stressful temperatures, physical coordination appears to be progressively lost until fish completely lose control and invert. The equilibrium loss end point is useful, primarily because of its relatively high precision of identification.

Equilibrium loss is manifested in gizzard shad acclimated to warm temperatures (15 to 20°C) within a few

minutes by ambient temperatures below 6 to 7°C. If a power plant shuts down in cold weather, it must, therefore, resume operation in a short time in order to avoid the adverse effects of cold shock on fishes. Fishes in a state of equilibrium loss may be carried by current or their own nondirected swimming beyond the reinstated heat source where they would eventually die or they could succumb to physical factors, such as stranding on beaches. The precise effects of a shutdown would depend on many variables peculiar to a specific power-plant site, such as discharge configuration and flow velocities and patterns.

In summary, shad acclimated to between 15 and 20°C had a 10% lethal threshold of 6 to near 7°C for acute cold shock. Cessation of opercular movement occurred long after the loss of equilibrium; hence it is of doubtful ecological significance. Regression lines for times to equilibrium loss are summaries of an ecologically significant response (Fig. 1) and have potential usefulness in predicting the impact of cold shock on gizzard shad populations.

#### ACKNOWLEDGMENT

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# The Effect of Cold Shock on the Vulnerability of Young Bluegill to Predation

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## ABSTRACT

A laboratory study with young bluegill (*Lepomis macrochirus*) showed that cold shock of 9°C was sufficient to cause increased vulnerability to predation by unstressed adult largemouth bass. Equal numbers of control and test fish acclimated at higher temperatures than those of the test tanks were offered simultaneously to unstressed predators, and predation was halted when approximately 50% of the prey were eaten. Survival proportions and predation ratios were calculated from the remaining fish. At temperature differentials of 9°C or more, predation ratio was found to accelerate with increasing thermal shock.

This study attempts to measure the effect of various degrees of cold shock on the vulnerability of young bluegill (*Lepomis macrochirus*) to predation. Cold shock is an important environmental problem. Emery (1970) observed fish mortalities in Lake Huron caused by an abnormal decrease in water temperatures from an internal seiche. Dahlberg and Smith (1970) observed equilibrium loss in addition to mortality when water temperatures were abnormally cold along the Georgia coast. In addition to naturally occurring cold shocks, the termination of heated discharge during power-plant shutdown can easily cause fish mortalities (Williams et al., 1971; Ash, Chymko, and Gallup, 1974). In all these cases fish living in and acclimated to warm temperatures were suddenly exposed to cooler water.

Bluegill are warm-water fish often found in large numbers in warm-water discharges. Neill (1969) found bluegill most abundant in discharge areas at temperatures in the range of 30 to 33.5°C. Cox (1973), in a laboratory study, established a high critical thermal maximum for the species. Bluegill could, therefore, easily encounter conditions that might cause mortality from cold shock or behavioral stress at sublethal temperatures. Loss of equilibrium from cold shock could cause abnormally high predation as fish move from a warm-water discharge into cooler water and encounter predators that have not been subjected to temperature stresses. Recent studies with juvenile catfish and largemouth bass have shown increased predation on

these species because of equilibrium loss and/or lowered swimming efficiency when they are subjected to sublethal temperature decreases (Coutant, Ducharme, and Fisher, 1974). A study of this type is necessary to quantify the effect of cold shock on the vulnerability of young bluegill to predation.

This study does not deal with the simultaneous exposure of both prey and predators to sudden temperature decreases. The situation becomes much more complex in this case, as the predator response to temperature decreases must also be examined.

## METHODS AND MATERIALS

The experimental procedure was the same as that described by Coutant, Ducharme, and Fisher (1974). Young-of-the-year bluegill obtained from the National Fish Hatchery in Cohutta, Ga., were placed in a common holding tank. The fish, averaging  $40.3 \pm 2.8$  mm (SD) in length, were randomly separated into control and test groups. Control fish were cold branded on the left side with liquid nitrogen (Coutant, 1972) and acclimated at 16°C, the base temperature. Test fish were cold branded on the right side and acclimated to temperatures of 16, 20, 25, 30, and 32°C for 3 weeks before testing. Five largemouth bass, all approximately 19 to 24 cm in length, were used in a group as predators for all trials.

Predation took place in a 1.2-m circular tank with a water depth of 65 cm. No cover was available for the prey except the tank walls and the center outlet standpipe. Groups of test and control fish (20 each) were combined in a bucket (for less than 15 sec) and then simultaneously offered to predators. The predation was stopped by rapidly netting out the survivors when approximately 50% of the prey had been eaten. Since the predation usually took place within a few minutes after the prey were offered, it is

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unlikely that any cold-shock recovery occurred during the trial. None of the surviving prey were used for subsequent tests, but they were held for observation of direct effects of the cold shock. Three or four trials were run for each temperature shock until the fish stock was exhausted.

The predation ratio,  $d_p = i_1/i_2$  (Bams, 1967), was calculated from the survivors to express the difference in predation rates between the two groups of fish:  $i_1$  and  $i_2$  are the instantaneous mortality rates of the test and control groups, respectively. The instantaneous mortality rate,  $i$ , is given by  $i = -\ln s$  where time is a unit interval and  $s$  is the survival proportion (i.e., the number of fish at finish divided by the number at start). The average  $d_p$  for each test group was calculated by summing the survival proportions. Homogeneity of variance among replicate trials was tested for significance with chi-square analysis. Numbers of survivors in homogeneous trials were summed, and chi-square analyses were performed to test for significant differences between number of survivors in control and test groups.

## RESULTS

No selective predation was observed for the 16 or 20°C acclimation test groups, which represented no shock and a 4°C shock, respectively (Table 1). A 9°C temperature drop was sufficient to produce a significant ( $P < 0.05$ ) increase in selective predation ( $d_p > 1$ ), and above a 9°C differential the predation ratio increased greatly. At 30°C acclimation and above, there were usually no survivors in the test groups, but there was some variation in the calculated predation ratios because of the difficulty in halting predation when exactly 50% of the prey had been eaten.

The bass immediately recognized and pursued bluegill exhibiting the most abnormal swimming behavior. When fish from 30 and 32°C test groups were placed in 16°C water, they were almost instantly stunned and lost equilibrium and many sank to the bottom of the predation tank. The unshocked controls quickly swam to the perimeter of the tank and escaped predation.

All prey that survived the predators recovered from the initial shock. They seemed normal in the laboratory for several months thereafter.

## DISCUSSION

Increased temperature differential of cold shock caused a trend toward increased selective predation of bluegill, beginning between a 4 and 9°C drop in temperature. This result is similar to that seen with juvenile channel catfish and largemouth bass in similar experiments (Coutant, Ducharme, and Fisher, 1974). Because bluegill acclimated at the higher temperatures (30 and 32°C) were much more easily captured by the bass, predation of half the total prey occurred quite rapidly (in 1 min or less), and it was difficult to stop predation at exactly the 50% level. Values of  $d_p$  for 30 and 32°C appear to be much different, but essentially the same selective predation occurred in both cases; the difference was mostly due to unequal total numbers eaten. At smaller temperature changes predation took place much more slowly (requiring about 5 min until 50% of the prey remained) because the prey were active and escaped the predators by swimming around the perimeter of the tank.

The objective of the experiment was to determine tolerable temperature decreases. In all the tests described here, the cold shock did not cause direct death. Recovery was gradual, however; and symptoms of stress which increased predation were manifested. The temperature differential shown in these experiments to be detrimental for long-term survival among predators appears to be around 8 to 9°C.

The danger of cold shock may be greatest in the fall when ambient temperatures decline. Young fish are not yet large enough to withstand stress, and predators are still actively feeding. Low water temperatures during the winter could cause predator inactivity in ambient areas and thus lower the danger of predation. Studies of this type on many species and validation of field reality are necessary for the regulation of power-plant shutdowns. When the species composition and response to environmental temperature changes around a power plant are known, conscientious regulation of plant operation will lessen environmental impact.

## ACKNOWLEDGMENTS

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TABLE 1 Results of Combined Trials of Predation by Adult Largemouth Bass at 16°C on Juvenile Bluegill Suffering Acute Cold Shock

| Acclimation temperature, °C | Number of trials | Survival proportion |      | Instantaneous predation rate |      | $d_p$ | $\chi^2$ | P      |
|-----------------------------|------------------|---------------------|------|------------------------------|------|-------|----------|--------|
|                             |                  | Control             | Test | Control                      | Test |       |          |        |
| 16                          | 3                | 0.44                | 0.47 | 0.82                         | 0.76 | 0.93  | 0.08     | 0.68   |
| 20                          | 4                | 0.51                | 0.53 | 0.68                         | 0.64 | 0.94  | 0.114    | 0.67   |
| 25                          | 4                | 0.58                | 0.35 | 0.55                         | 1.05 | 1.91  | 4.37     | < 0.05 |
| 30                          | 4                | 0.86                | 0.09 | 0.15                         | 2.39 | 15.93 | 22.88    | < 0.05 |
| 32                          | 3                | 0.78                | 0.07 | 0.24                         | 2.71 | 11.29 | 39.76    | < 0.05 |



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# Effects of Various Time-Excess Temperature Histories on Hatching Success of Blueback Herring, American Shad, and Striped Bass Eggs

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## ABSTRACT

In the laboratory blueback herring (*Alosa aestivalis*), American shad (*Alosa sapidissima*), and striped bass (*Morone saxatilis*) eggs from the Chesapeake Bay region were subjected to time-excess temperature histories typical of those experienced by organisms entrained by power plants with a variety of design and operating criteria. The maximum excess temperature ranged from 7 to 20°C above ambient; the time of exposure was from 4 to 60 min; and the period of cooling to the final temperature was from 60 to 300 min. An excess temperature of 20°C resulted in nearly total mortality of eggs of all three species. An excess temperature of 15°C significantly reduced hatching success of blueback herring and American shad but not striped bass eggs. Excess temperatures of 7 and 10°C did not significantly affect hatching success of eggs of any of the three species.

Very little published research on the effects of temperature on fish eggs is of direct use in predicting the thermal effects on entrained fish eggs in steam electric stations with once-through cooling systems. In most studies eggs are subjected to some constant elevated temperature for all or nearly all of the experimental period (Albrecht, 1964; Bradford, Miller, and Buss, 1968; Edsall, 1970; Kelley, 1968; Price, 1940; Shannon, 1970). Such exposure histories are not representative of those experienced by organisms entrained in the once-through cooling system of a steam electric station, which removes large volumes of water from the environment and returns it at elevated temperatures. Organisms smaller than the mesh size of the intake screen—generally 9 to 13 mm—are carried along with the water.

Organisms passing through the plant are subjected to a very rapid rise in temperature, approximately equivalent to the temperature rise across the condensers, during passage through the plant to the point of discharge and then to decreasing excess temperatures as they are carried down the plume. The excess temperature is the increase above that

which would be observed if the plant were not operating. In the near field the rate of cooling along the plume is determined almost entirely by the intensity of mixing. Planktonic organisms may also be entrained into the thermal plume without passing through the plant. These organisms are subjected to a less rapid rise in temperature and to somewhat lower excess temperatures. The actual time-temperature histories experienced by planktonic organisms varies from plant to plant and, to a much smaller extent, with time at any given plant. The histories depend on the design of a plant, its operating load, and the characteristics of the receiving waters.

Numerous field studies conducted to assess the effects of entrainment by power plants have produced confusing and frequently contradictory results that generally do not permit separation of the effects of temperature, mechanical agitation and abrasion, pressure changes, and exposure to biocides.

Schubel and Auld (1972a; 1972b; 1973; 1974) and Schubel (1974; 1975) presented results of laboratory experiments in which American shad (*Alosa sapidissima*), alewife (*Alosa pseudoharengos*), blueback herring (*Alosa aestivalis*), and striped bass (*Morone saxatilis*) eggs were subjected to time-temperature histories typical of those experienced by organisms entrained by plants with a variety of design and operating criteria. Frank (1974) studied the effects of exposure of carp (*Cyprinus carpio* L.) eggs in different stages of development. Hoss, Coston, and Hettler (1971) studied the effects of thermal increases encountered during entrainment on post-larval and juvenile fishes, and Hoss, Hettler, and Coston (1973) investigated the thermal effects of entrainment on larvae of several species of fish.

In this study striped bass (*Morone saxatilis*), blueback herring (*Alosa aestivalis*), and American shad (*Alosa sapidis-*

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*simia*) eggs were subjected to a variety of time-temperature histories. The maximum excess temperature ranged from 7 to 20°C; the time of exposure to the maximum temperature differential ( $\Delta T$ ) was from 4 to 60 min; and the period of cooling to the final temperature (1°C above the base temperature) was from 60 to 300 min. The base temperature, the average surface-water temperature on the spawning grounds in Chesapeake Bay tributaries, ranged from 16.6 to 21.1°C.

## METHODS

### Egg Collection and Handling

Eggs stripped from a single female and artificially fertilized were maintained until the start of an experiment at the average surface-water temperature on the spawning grounds at the time the female spawned. At the start of an experiment, 7 to 14 subsamples of 100 to 125 viable eggs each were selected from a sample from a single female and placed in individual hatching boxes. Each box consisted of a polyvinyl chloride frame covered with nylon bolting cloth. One or two of the boxes were immediately returned to the constant-temperature bath set at base temperature. Each of the remaining subsamples was exposed to one of the time-temperature histories (Fig. 1), returned to the constant-temperature bath, and incubated until all eggs either hatched or died. The hatching boxes floated freely in 57-liter (15-gal) aquariums. Throughout this period the water in each aquarium was continuously aerated and gently stirred. Water was completely replaced every 48 hr by periodic additions of water from the spawning grounds (Schubel and Schiemer, 1972). Dead eggs were periodically removed, counted, and preserved for later study.

### Time-Temperature Exposure Histories

The time-temperature histories are plotted in Fig. 1. Each consists of an initial period of constant temperature (base temperature) followed by an abrupt rise to a maximum  $\Delta T$ , which was maintained for some specified time and was followed by a period of cooling to the final temperature, 1°C above base temperature. The final temperature is analogous to the far-field temperature that might be observed in nature. The time-temperature histories used in this study were patterned after those of Pritchard and Carter (1972), who developed relationships that give the maximum time of exposure of entrained organisms to any specified  $\Delta T$  for a variety of power-plant operating and design criteria. Each relationship is based on a specific maximum  $\Delta T$  corresponding to the temperature rise across the condensers and a particular set of cooling conditions, ranging from surface cooling only to cooling with various degrees of dilution. The excess temperature histories depicted in Fig. 1 are representative of power plants with a maximum  $\Delta T$  of 7, 8, 10, 15, or 20°C and a variety of discharge structures, with discharge velocities ranging from perhaps 50 to 300 cm/sec. In each experiment the excess temperature was keyed to the average surface-water temperature at the spawning grounds.

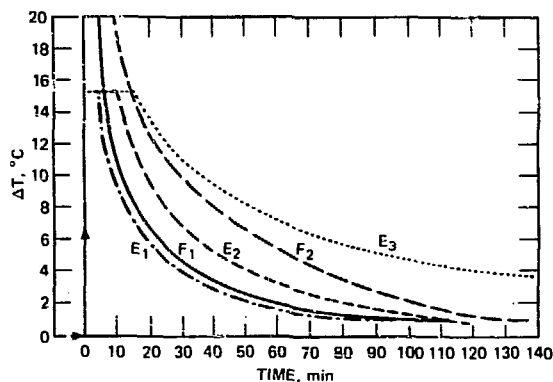
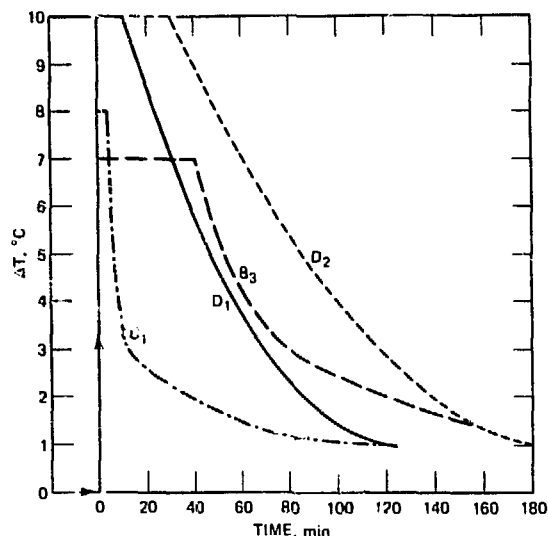


FIG. 1 Time-excess temperature histories used in experiments; these represent power plants with a maximum  $\Delta T$  of 7, 8, 10, 15, or 20°C. In each experiment a series of subsamples of eggs was selected from a single female. One subsample, the control, was held at the base temperature throughout the experiment. Each of the other subsamples was maintained for a period at the base temperature; abruptly raised to a maximum  $\Delta T$ , which was maintained for a specified time, as shown; cooled to the final temperature (1°C above base temperature); and then incubated at base temperature through hatching.

Each of the excess temperature histories was simulated in the following way. A 19-liter (5-gal) aquarium was filled with 11 to 15 liters of water at the temperature required to produce the desired  $\Delta T$ . The holding boxes containing the organisms were then quickly transferred to the aquarium. The water was maintained at the maximum excess temperature for some specified period and then cooled at a predetermined rate by adding colder water to the bottom through a hose placed below a small stirrer. The hose was

fixed at one end of the aquarium, and the holding boxes were restricted to the opposite end. The water in each testing aquarium was aerated and stirred throughout each experimental run. Temperature in the hatching box was monitored every 1 to 3 min during the first 20 min of each run and approximately every 5 min over the remainder of the run. All water was collected from the spawning grounds.

### Experimental Design

The experimental design was the partially balanced incomplete-block design shown in Table 1. The last three experiments were a repeat of the first three. The total design covers nine time-temperature treatments, but, in any one experiment, only six of the nine treatments, plus a control, were tested. This design was selected for several reasons. First, because of the natural variability in survival rates of eggs from different females, it was desirable to place them in different blocks. Second, because of the difficulty of conducting a complete set of nine experimental treatments in any one experiment, an incomplete design was chosen. Third, a partially balanced design was selected because it requires considerably less computation than an unbalanced design. This design tends to maximize and equalize the precision with which we can estimate  $\tau_c - \tau_i$ , the difference between the response of eggs to the control treatment and to the  $i$ th treatment.

TABLE 1 Experimental Design

| Experiment number | Time-temperature treatment* |                |                |                |                |                |
|-------------------|-----------------------------|----------------|----------------|----------------|----------------|----------------|
| 1                 | D <sub>1</sub>              | E <sub>2</sub> | F <sub>2</sub> | D <sub>2</sub> | E <sub>3</sub> | B <sub>3</sub> |
| 2                 | D <sub>2</sub>              | E <sub>3</sub> | B <sub>3</sub> | C <sub>1</sub> | E <sub>1</sub> | F <sub>1</sub> |
| 3                 | C <sub>1</sub>              | E <sub>1</sub> | F <sub>1</sub> | D <sub>1</sub> | E <sub>2</sub> | F <sub>2</sub> |
| 4                 | D <sub>1</sub>              | E <sub>2</sub> | F <sub>2</sub> | D <sub>2</sub> | E <sub>3</sub> | B <sub>3</sub> |
| 5                 | D <sub>2</sub>              | E <sub>3</sub> | B <sub>3</sub> | C <sub>1</sub> | E <sub>1</sub> | F <sub>1</sub> |
| 6                 | C <sub>1</sub>              | E <sub>1</sub> | F <sub>1</sub> | D <sub>1</sub> | E <sub>2</sub> | F <sub>2</sub> |

\*Symbols refer to curves in Fig. 1. A control sample was also run for each experiment.

## RESULTS

### *Alosa aestivalis* (Blueback Herring)

All six experiments were run with blueback herring eggs, one in duplicate. The stages of egg development at the start of the experiments ranged from early embryo (15 hr old) to tail-free embryo (28 hr old), and the base temperature ranged from 17.9 to 21.1°C. According to Bigelow and Schroeder (1953, p. 107), the optimum temperature range for blueback herring spawning in the tributaries to the Gulf of Maine is 21 to 24°C. Our observations indicate a similar range for the Chesapeake Bay region. The results are summarized in Table 2.

### *Alosa sapidissima* (American Shad)

All six experiments were run with American shad eggs, three in duplicate. Three of the six excess temperature histories in experiment 2 were lost because of an accident. The stages of development at the start of the experiments ranged from early embryo (20 hr old) to mid-embryo (38 hr old). The base temperature was 20.5°C in each of the experiments. According to Walburg (1960), spawning on the east coast generally occurs between 12 and 20°C. The results are summarized in Table 2.

### *Morone saxatilis* (Striped Bass)

Because of a variety of problems, only the first half of the experimental design was run with striped bass eggs. Although there was a loss of information for this set, the estimates are the minimum unbiased variance estimates of treatment effects. If the entire array had been run, the estimated variances would, of course, have been less since the variance generally decreases approximately as the square root of the number of observations. The stages of development tested ranged from early embryo (24 hr old) to mid-embryo (44 hr old), and the base temperature ranged from 16.6 to 19.6°C. According to Pearson (1938), spawning generally occurs between 15 and 21°C. The results are summarized in Table 2.

## DISCUSSION AND CONCLUSIONS

A two-way analysis of variance (ANOVA) was run on the combined results for each species. In these analyses each fish constitutes a block and each time-excess temperature history, including the constant-temperature history of the control, constitutes a treatment. Eggs from a variety of fishes were subjected to the same treatments to provide comparisons between treatments which would be free of possible variations from fish to fish. That subsequent analysis showed significant variations from fish to fish justified blocking in this way. Since eggs in different stages of development were tested, the effects of different fish and different stages of development are confounded, but the relative importance of these factors can be assessed as explained later. The data were transformed with an arc sine transformation before the analyses were run.

The results of the ANOVA are summarized in Table 3. In all the time-temperature histories tested, exposure to excess temperatures of 7 and 10°C did not significantly affect the hatching success of eggs of any of the three species in early- to mid-embryonic stages of development. After exposure to an excess temperature of 15°C, hatching success of striped bass eggs was not significantly reduced relative to that of the controls, but hatching success of American shad and blueback herring eggs was significantly reduced at excess temperatures of 15°C and higher. At an excess temperature of 20°C, except for one subsample of striped bass eggs, there was total mortality for all three species. Within 2 min of initial exposure to a  $\Delta T$  of 20°C, the eggs turned milky white and did not recover. The only other excess temperature history that significantly reduced

TABLE 3 Hatching Success of Controls and Experimental Subsamples of *Alosa aestivalis* (3R Fork Herring), *Alosa sapidissima* (American Shad), and *Morone saxatilis* (Striped Bass) Eggs\*

| Experiment number†        | Age of eggs, ‡ hrs | Stage of development‡                   | Base temp., °C | Absolute hatching success for various time-temperature histories § % |                    |                    |                     |                     |                     |                     |                     |                     |                     |
|---------------------------|--------------------|-----------------------------------------|----------------|----------------------------------------------------------------------|--------------------|--------------------|---------------------|---------------------|---------------------|---------------------|---------------------|---------------------|---------------------|
|                           |                    |                                         |                | Control                                                              | B <sub>3</sub> (7) | C <sub>1</sub> (8) | D <sub>1</sub> (10) | D <sub>2</sub> (10) | E <sub>1</sub> (15) | E <sub>2</sub> (15) | E <sub>3</sub> (15) | F <sub>1</sub> (20) | F <sub>2</sub> (20) |
| <i>Alosa aestivalis</i>   |                    |                                         |                |                                                                      |                    |                    |                     |                     |                     |                     |                     |                     |                     |
| 1                         | 21                 | Early embryo; start of somite formation | 17.9           | 88                                                                   | 89                 | NA                 | 83                  | 82                  | NA                  | 0                   | 0                   | NA                  | 0                   |
| 2                         | 29                 | Early embryo; start of somite formation | 17.9           | 87                                                                   | 96                 | 89                 | NA                  | 94                  | 0                   | NA                  | 0                   | NA                  | 0                   |
| 3                         | 25                 | Early embryo                            | 21.1           | 95                                                                   | NA                 | 33                 | 75                  | NA                  | 0                   | 0                   | NA                  | 0                   | 0                   |
| 4 a                       | 15                 | Very early embryo                       | 19.0           | 98                                                                   | 97                 | NA                 | 99                  | 95                  | NA                  | 50                  | 39                  | NA                  | 0                   |
| 4 b                       |                    |                                         |                | 91                                                                   | 92                 | NA                 | 90                  | 90                  | NA                  | 84                  | 47                  | NA                  | 0                   |
| 5                         | 28                 | Tail-free embryo                        | 20.3           | 98                                                                   | 55                 | 94                 | NA                  | 96                  | 93                  | NA                  | 50                  | 0                   | NA                  |
| 6                         | 28                 | Tail-free embryo                        | 21.1           | 56                                                                   | NA                 | 38                 | 92                  | NA                  | 0                   | 0                   | NA                  | 0                   | 0                   |
| <i>Alosa sapidissima</i>  |                    |                                         |                |                                                                      |                    |                    |                     |                     |                     |                     |                     |                     |                     |
| 1 a                       | 16                 | Very early embryo                       | 20.5           | 81                                                                   | 68                 | NA                 | 79                  | 72                  | NA                  | 0                   | 0                   | NA                  | 0                   |
| 1 b                       |                    |                                         |                | 76                                                                   | 66                 | NA                 | 79                  | 80                  | NA                  | 0                   | 0                   | NA                  | 0                   |
| 2 a                       | 20                 | Early embryo                            | 20.5           | 19                                                                   |                    |                    | NA                  |                     | 0                   | NA                  | 0                   | 0                   | NA                  |
| 2 b                       |                    |                                         |                | 24                                                                   |                    |                    | NA                  |                     | 0                   | NA                  | 0                   | 0                   | NA                  |
| 3                         | 38                 | Mid-embryo                              | 20.5           | 60                                                                   | NA                 | 32                 | 11                  | NA                  | 69                  | 0                   | NA                  | 0                   | 0                   |
| 4                         | 16                 | Very early embryo                       | 20.5           | 90                                                                   | 92                 | NA                 | 83                  | 91                  | NA                  | 0                   | 0                   | NA                  | 0                   |
| 5                         | 20                 | Early embryo                            | 20.2           | 96                                                                   | 59                 | 100                | NA                  | 95                  | 1                   | NA                  | 0                   | 0                   | NA                  |
| 6 a                       | 38                 | Mid-embryo                              | 20.5           | 99                                                                   | NA                 | 45                 | 100                 | NA                  | 16                  | 89                  | NA                  | 0                   | 0                   |
| 6 b                       |                    |                                         |                | 98                                                                   | NA                 | 30                 | 88                  | NA                  | 66                  | 83                  | NA                  | 0                   | 0                   |
| <i>Morone saxatilis</i> ¶ |                    |                                         |                |                                                                      |                    |                    |                     |                     |                     |                     |                     |                     |                     |
| 1                         | 36                 | Mid-embryo                              | 19.3           | 100                                                                  | 100                | NA                 | 98                  | 100                 | NA                  | 100                 | 98                  | NA                  | 0                   |
| 2                         | 44                 | Mid-embryo                              | 16.6           | 99                                                                   | 98                 | 97                 | NA                  | 98                  | 96                  | NA                  | 100                 | 68                  | NA                  |
| 3                         | 24                 | Early embryo                            | 19.6           | 100                                                                  | 100                | 99                 | NA                  | 100                 | 100                 | NA                  | 100                 | NA                  | NA                  |

\*Controls were held at the appropriate base temperature throughout the incubation period. Each of the experimental subsamples was exposed to one of the time-temperatures depicted in Fig. 1. The number of eggs per subsample ranged from 100 to 125. A complete experiment consists of a control treatment and six of the nine experimental time-temperature histories. Treatments not called for in the experiments are indicated by NA (not applicable). Blank spaces in columns indicate missing observations.

†Duplicate experiments are indicated by a and b following numbers.

‡Age and stage of development at the start of the experiment are given.

§Numbers in parentheses are maximum  $\Delta T$  (°C).

¶Only the first half of the experimental design was run for this species.

TABLE 3 Summary of Effects of Exposure to Different Time-Temperature Histories on Hatching Success as Indicated by Analysis of Variance\*

| Species          | Time-Temperature History† |                    |                     |                     |                     |                     |                     |                     |                     |
|------------------|---------------------------|--------------------|---------------------|---------------------|---------------------|---------------------|---------------------|---------------------|---------------------|
|                  | B <sub>3</sub> (7)        | C <sub>1</sub> (8) | D <sub>1</sub> (10) | D <sub>2</sub> (10) | E <sub>1</sub> (15) | E <sub>2</sub> (15) | E <sub>3</sub> (15) | F <sub>1</sub> (20) | F <sub>2</sub> (20) |
| Blueback herring | —                         | +                  | —                   | —                   | +                   | +                   | +                   | +                   | +                   |
|                  |                           | (<0.005)           |                     |                     |                     |                     | (all <0.005)        |                     |                     |
| American shad    | —                         | +                  | —                   | —                   | +                   | +                   | +                   | +                   | +                   |
|                  |                           | (<0.005)           |                     |                     |                     |                     | (all <0.005)        |                     |                     |
| Striped bass     | —                         | —                  | —                   | —                   | —                   | —                   | —                   | +                   | +                   |
|                  |                           |                    |                     |                     |                     |                     |                     | (<0.05)             | (<0.005)            |

\*A minus (—) indicates the absence of a significant effect. A plus (+) indicates a significant ( $p \leq 0.05$ ) effect. Levels of significance are indicated in parentheses.

†Numbers in parentheses are maximum  $\Delta T$  (°C).

atching success was  $C_1$  ( $\Delta T = 8^\circ\text{C}$ ), which had a significant effect on both blueback herring and American shad but not on striped bass.

In previous experiments Schubel (1974) subjected eggs of these same species to a variety of time-temperature histories. Four of these ( $B_3$ ,  $C_1$ ,  $D_1$ ,  $D_2$ ) were repeated in this study. Schubel found in his earlier experiments that none of these four histories had a significant effect on hatching success. The reason for the apparent difference in thermal sensitivity of blueback herring and shad eggs to time-temperature history  $C_1$  is unclear. That the base temperatures used in the experiments described here were higher than those used previously does not appear to be the causative factor since neither  $D_1$  nor  $D_2$  significantly reduced hatching success. The thermal dose associated with  $D_1$  and  $D_2$  was greater than that with  $C_1$ . The more rapid rate of cooling of  $C_1$  (relative to that of  $D_1$  or  $D_2$ ), coupled with the higher base temperature used in these experiments, may offer an explanation for the apparent difference revealed by the two studies.

The ANOVA also showed significant ( $P \leq 0.01$ ) variations in hatching success among eggs from different fish for blueback herring and American shad but not for striped bass. These differences are apparently caused by genetic differences of the parents and not by differences in the thermal sensitivity of eggs in different stages of development. If developmental stages differed in their sensitivity to exposure to the time-temperature histories studied, the effects should appear in experimental subsamples but not in controls; all control subsamples were held at the base temperature from within a few hours of fertilization until the eggs hatched. Comparing results of experiments with eggs in the same stage of development but from different females indicates that the variability among controls is approximately as great as that among experimental subsamples. The range of the stage of development tested in these experiments was very limited, but other experiments support the conclusions for these and similar time-temperature histories and associated thermal doses (Schubel, 1974). More experimentation is needed with eggs in other stages of development and with other thermal doses to assess unequivocally the relationship between thermal sensitivity and developmental stage.

There has been a tendency among biologists to favor a low  $\Delta T$  and a relatively large volume of cooling water for power plants. We might argue for the highest  $\Delta T$  that can be "safely" used since the higher the  $\Delta T$ , the smaller the volume of cooling water required for the same heat rejection. The experiments described here and elsewhere (Schubel, 1974) indicate that a  $\Delta T$  of 10 to  $12^\circ\text{C}$  may be optimal during the time of year that these and other important anadromous and semi-anadromous species spawn in the Chesapeake Bay region. Present state regulations limit the  $\Delta T$  in the Maryland waters of the Chesapeake Bay and its tributaries to  $5.6^\circ\text{C}$  ( $10^\circ\text{F}$ ). By raising the  $\Delta T$  and reducing the required volume of cooling water, we would reduce the mechanical effects of entrainment on popula-

tions since fewer organisms would be passed through the plant. The effects of biocides would also be reduced. Several investigators (e.g., Schubel, 1974; Adams, 1968) indicated that mechanical stresses during entrainment probably have a much greater impact on fish eggs and larvae than does temperature. Recent work by Coutant and Kedl (1975) with striped bass larvae indicates that "the condenser tube is unlikely as the locus of mechanical damages to fish larvae at operating power plants." The effects of the pumps, however, have not been adequately assessed.

The basic question, of course, is not what fractions of entrained fish eggs and larvae are killed but rather to what extent these losses affect fish populations in the system.

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# Behavioral Thermal Regulation by Rainbow Trout in a Temperature Gradient

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Norman Huggins

## ABSTRACT

This study compares the fluctuations of body temperature of fish selecting temperature in a laboratory gradient with the temperature of the fish's changing surroundings. Temperature preferences were determined for large (150- to 250-g) rainbow trout (*Salmo gairdneri*), which had been induced to swallow miniature radio thermometers in their food. Positions of individual fish in the horizontal temperature gradient apparatus were noted, and continuous records were made of the gut temperature by monitoring the pulse frequency of the radio thermometer. Body temperature fluctuated within considerably narrower limits than did the environmental temperatures encountered by the fish. It is concluded that this form of behavioral thermoregulation is aided by the relatively slow rates of heat exchange between a fish and the surrounding water.

The movement of fish in a temperature gradient which results in their concentration within a relatively narrow zone of temperature has been interpreted as temperature selection. The literature in this field consists, for the most part, of studies of the effects of thermal experience, expressed as acclimatization or acclimation, on temperature selection. In recent years interest has been focused on the mechanisms by which fish move into or select the favored temperature range (Crawshaw and Hammel, 1973).

Fish may be trained to discriminate between small temperature differences (Bull, 1936), and it logically follows that they will use their sensitive temperature receptors to recognize and seek out temperatures that are favorable to their physiological activities, performing this function with surprising precision. Sullivan (1954) considered temperature selection to be brought about by the congregation of fish in a narrow zone as a result of biases imposed by temperature on their movements and orientation as they swam about in a heterothermic environment. She analyzed the patterns of swimming activity and proposed a simple model of temperature selection, which she tested by writing the various components of activity on

cards, drawing them at random, and plotting the resulting distributions. Some of the distributions generated bore a close resemblance to those actually observed in temperature-selection experiments.

An opposing viewpoint to this mechanistic or kinetic interpretation is that fish in a temperature gradient move almost purposefully up or down the gradient toward the preferred temperature zone. Support for this interpretation is provided by experiments in which fish, when presented with two or more streams of water differing in temperature, consistently move into the favored stream (Fry, 1975).

A more sophisticated demonstration of an active process of temperature selection is the performance of fish in the thermoregulatory aquarium devised by Neill, Magnuson, and Chipman (1972). In this apparatus fish regulate the temperature of their environment by dividing their time between the halves of the experimental tank. Passage of a fish through a tunnel connecting the two chambers sequentially intercepts two light beams monitored by a pair of photoelectric cells that initiate either heating or cooling, depending on the direction of passage. Physiological mechanisms and the general nature of the process of temperature selection are under investigation in modified versions of this ingenious device (Beitinger, 1974; Neill and Magnuson, 1974; Reynolds and Casterlin, this volume; Fry and Evans, 1975).

We obtained the idea for this study from Crawshaw and Hammel (1973), who sought the location of the region in the fish's central nervous system which is responsible for behavioral thermoregulation. Their experimental tank consisted of an aquarium partitioned into two equal parts; each part was maintained at a different constant temperature throughout an experiment. The temperatures chosen bracketed the preferred temperature by about the same number of degrees on either side. Crawshaw and Hammel studied the effects of cooling the brainstem on the pattern of thermoregulatory movements as the fish moved back and forth between the two compartments. Fluctuations in deep body temperature were recorded by inserting a needlelike

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# Behavioral Thermal Regulation by Rainbow Trout in a Temperature Gradient

Robert McCauley  
Norman Huggins

## ABSTRACT

This study compares the fluctuations of body temperature of fish selecting temperature in a laboratory gradient with the temperature of the fish's changing surroundings. Temperature preferences were determined for large (150- to 250-g) rainbow trout (*Salmo gairdneri*), which had been induced to swallow miniature radio thermometers in their food. Positions of individual fish in the horizontal temperature gradient apparatus were noted, and continuous records were made of the gut temperature by monitoring the pulse frequency of the radio thermometer. Body temperature fluctuated within considerably narrower limits than did the environmental temperatures encountered by the fish. It is concluded that this form of behavioral thermoregulation is aided by the relatively slow rates of heat exchange between a fish and the surrounding water.

The movement of fish in a temperature gradient which results in their concentration within a relatively narrow zone of temperature has been interpreted as temperature selection. The literature in this field consists, for the most part, of studies of the effects of thermal experience, expressed as acclimatization or acclimation, on temperature selection. In recent years interest has been focused on the mechanisms by which fish move into or select the favored temperature range (Crawshaw and Hammel, 1973).

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We obtained the idea for this study from Crawshaw and Hammel (1973), who sought the location of the region in the fish's central nervous system which is responsible for behavioral thermoregulation. Their experimental tank consisted of an aquarium partitioned into two equal parts; each part was maintained at a different constant temperature throughout an experiment. The temperatures chosen bracketed the preferred temperature by about the same number of degrees on either side. Crawshaw and Hammel studied the effects of cooling the brainstem on the pattern of thermoregulatory movements as the fish moved back and forth between the two compartments. Fluctuations in deep body temperature were recorded by inserting a needlelike

thermistor into the fish's dorsal musculature. Using brown bullheads (*Ictalurus nebulosus*), they found that, after the fish had learned to move between the two temperature extremes, their movements were such that body temperatures fluctuated within relatively narrow limits. We attempted to duplicate the procedures of Crawshaw and Hammel with rainbow trout (*Salmo gairdneri*, Richardson), but our efforts were frustrated by the unwillingness of the experimental subjects to swim through the door connecting the two halves of the tank. We abandoned this two-choice temperature apparatus in favor of a conventional linear horizontal gradient trough, which relieved the fish of the task of swimming through or around physical barriers. The awkward "tether" of the thermistor was eliminated by replacing the thermistor with an internal radio thermometer. In our investigation we studied the relationship between changing environmental temperature and gut temperature of the fish.

## MATERIALS AND METHODS

### Experimental Animals

Rainbow trout yearlings weighing 150 to 250 g were purchased from a local fish hatchery and were acclimated in the laboratory at 10°C by holding them at this temperature for at least two weeks before the experiments. During their period in the laboratory, they were fed commercial trout food.

### Calibration of the Radio Thermometers

The radio thermometer consisted of a simple oscillator embodying a thermistor in its circuit such that signals in the a-m broadcast band (540 to 1600 kHz) were emitted in the form of short pulses at frequencies directly proportional to the surrounding temperature (MacKay, 1968). The miniature transmitter, along with its hearing-aid cell, consisted of a cylinder 15 to 20 mm long and 7 to 10 mm in diameter. We induced the fish to swallow the transmitter "pill" by coating it with hamburger and presenting it during feeding time. Usually the fish retained the transmitter for several days before it passed through the gut or was regurgitated. We protected the device from moisture and digestive fluids by dipping it in melted beeswax. The range of the transmitter was limited, and signals were received on the a-m band of a portable transistor radio at a distance of 30 cm. For this reason the broadcast signal was collected by an antenna imbedded along the bottom of the experimental tank.

Since the relation of pulses to temperature of such miniature transmitters is not stable (depending on the voltage of the galvanic cell), calibration was checked before and after an experiment. Calibration consisted of slowly warming or cooling the transmitter (one-fifth degree per minute) and noting the frequency of pulses at various temperatures. Curves were drawn relating frequency to temperature, and, from these, temperatures corresponding to a given frequency could be read.

## Determination of Preferred Temperature

Temperature preferences for individual trout were determined by recording consecutive positions of the fish in a 3.25-m gradient tank constructed from a design by Zahn (1963). Temperature gradients spanning 8°C established in the trough included the approximate preferred temperature as indicated from preliminary experiments. Observations were made at intervals of 15 sec, and core temperatures were simultaneously monitored by continuously counting the frequency of pulses emitted from the radio thermometer.

## Rates of Thermal Exchange Between Fish and the Surrounding Water

Several investigators have reported on the relationship between the body temperature of a fish and the temperature of its immediate surroundings (Fry, 1967; Hughes and Roberts, 1970; Stevens and Fry, 1970; Mueller, 1974; Neill and Magnuson, 1974; Reynolds and Casterlin, this volume; Beiting, 1974). Under conditions of changing environmental temperature, there may be a considerable lag between body and environmental temperature (Spigarelli et al., 1974; Dean, 1973). We determined rates of temperature equilibration for our experimental subjects by transferring individuals from the acclimation tank (10°C) to another constant-temperature bath maintained at 15°C. After the fish had assumed the temperature of the warmer bath, it was again returned to the 10°C tank, and the rate of equilibration to the cooler temperature was determined.

## RESULTS AND DISCUSSION

### Temperature Preference

Temperature appeared to be the dominant factor influencing fish distributions in the temperature-gradient trough (Fig. 1) since there was close agreement between the observed mode of distribution of fish and the location of the mean preferred temperature for the sample. Table 1 summarizes mean selected temperatures of the trout together with mean core temperatures as indicated by the radio thermometers. The mean selected temperature of the five subjects (16.7°C) is several degrees higher than that reported by Garside and Tait (1958) for two-year-old trout at the same acclimation level. No significance can be assigned the lower mean core temperature (16.3°C) since the error of the radio thermometer was of the order of one-third of a degree. The consistently narrower fluctuation of core temperature in comparison with the fluctuation of environmental temperature suggests that some form of behavioral thermoregulation of core temperature is taking place through the exploratory behavior of the fish in the temperature gradient (Fig. 2).

### Rates of Thermal Exchange

A typical curve relating changes in core temperature to corresponding changes in environmental temperature is shown in Fig. 3. The rate of heat exchange for warming is

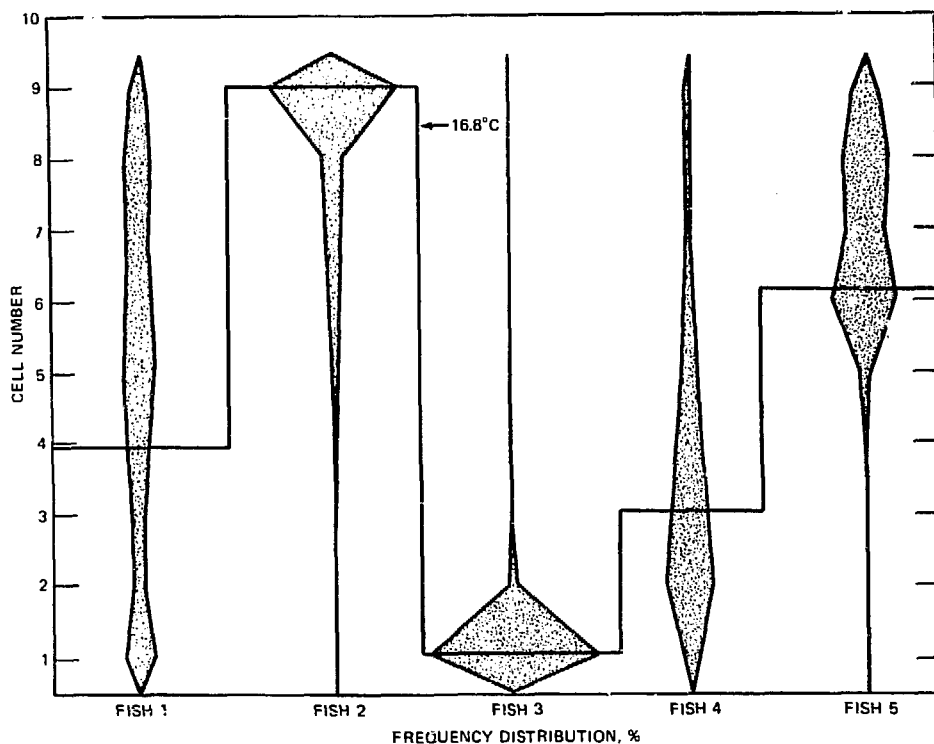


FIG. 1 Frequency distribution polygons of five rainbow trout in five successive temperature gradients. The position of the mean preferred temperature for the sample is connected by a 16.8°C isotherm.

TABLE 1 Summary of Mean Selected and Core Temperatures (°C) of Five Rainbow Trout in a 3.25-m Horizontal Temperature Gradient

| Fish | Mean selected temperature | Standard deviation | Mean core temperature | Standard deviation |
|------|---------------------------|--------------------|-----------------------|--------------------|
| 1    | 16.1                      | 1.74               | 15.5                  | 1.00               |
| 2    | 16.3                      | 1.10               | 16.2                  | 0.81               |
| 3    | 16.7                      | 0.12               | 16.6                  | 0.04               |
| 4    | 17.3                      | 1.50               | 16.6                  | 0.47               |
| 5    | 17.1                      | 0.21               | 16.5                  | 0.21               |

greater than that for cooling, a phenomenon noted by other investigators for some species (Spigarelli et al., 1974; Mueller, 1974). Mueller gives a detailed description of the variables associated with heat exchange in fish. If we know the rates of heat exchange between trout and the surrounding water and the intervals of time for which the fish remains at any environmental temperature in a temperature gradient, we can calculate core temperatures continuously throughout the course of an experiment (see Spigarelli et al., 1974, for details of this calculation). Cooling and

heating constants were derived from our data and inserted into the formula. Using a simple computer program, we calculated theoretical core temperatures at 15-sec intervals for each of the five preferred temperature experiments. Visual inspection of the computer-generated curves revealed no congruence between the observed and theoretical core temperatures, but in each experiment the amplitude of both sets of fluctuations lay between 1 and 2 degrees.

The results of this investigation demonstrate that the thermal inertia, or lag, between core and environmental temperature is sufficient to account for the conservative thermal regime that exists in the body of the fish moving back and forth in a temperature gradient. The laboratory study supports the recent field observations that fish in the neighborhood of a thermal plume are, in some instances, controlling their body temperatures by regular excursions into regions of varying temperatures (Spigarelli et al., 1974).

#### ACKNOWLEDGMENTS

This study was stimulated by informal conversation with Glen Parker of the University of Waterloo. Ken Jinde of the University of Guelph skillfully assembled the radio thermometer. The laboratory facilities were provided by funds from the U. S. Environmental Protection Agency.

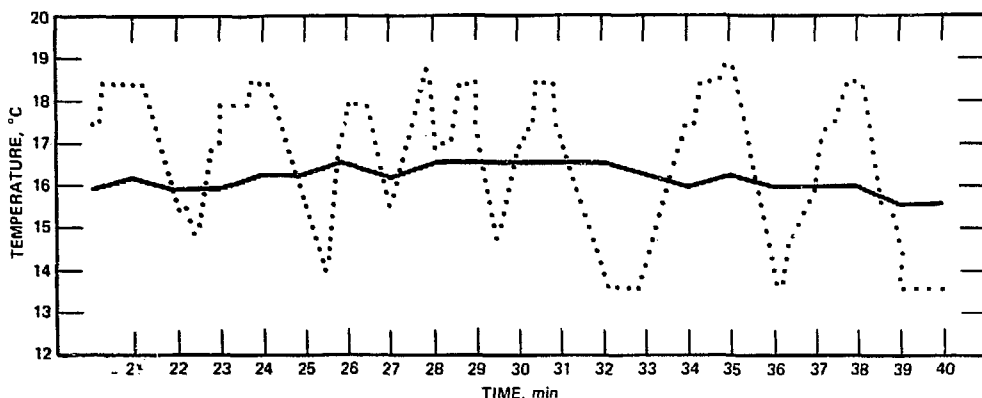


FIG. 2 Consecutive body and environmental temperatures of a rainbow trout moving back and forth in a horizontal temperature gradient. —, body temperature of a 200-g fish. . . . , temperature of surrounding water.

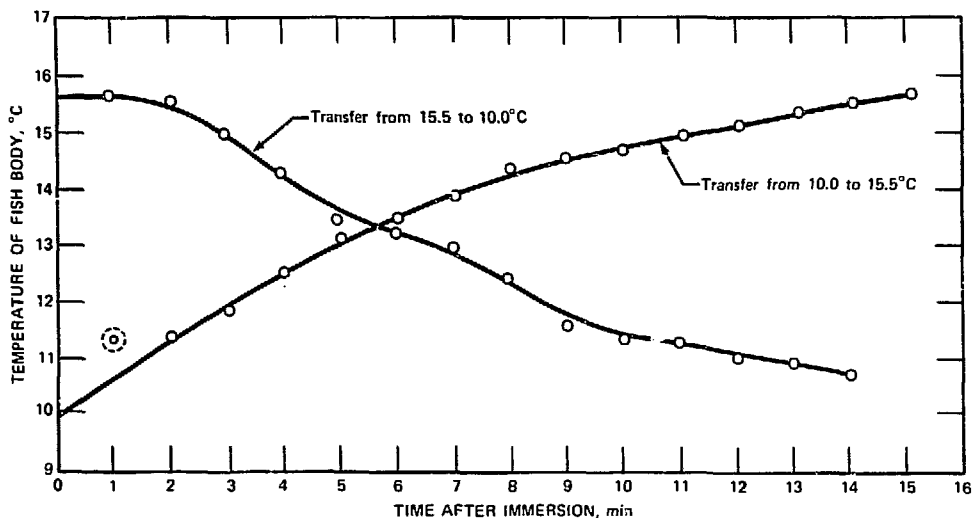


FIG. 3 Curves showing changes in core temperature of a 200-g rainbow trout when it is abruptly transferred from one constant temperature bath to another.

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# Behavioral Thermoregulation by Bluegill Exposed to Various Rates of Temperature Change

Thomas L. Beitinger

## ABSTRACT

Thermoregulatory behavior of bluegill (*Lepomis macrochirus*) exposed to three different combinations of temperature rate changes (heating rate twice the cooling rate; heating rate one-half the cooling rate; heating and cooling rates equal) was determined in a two-chambered, modified temporal temperature-preference system. During 3-day experimental periods, bluegill partitioned their time between the cooler and warmer tank halves relative to the ratio of heating and cooling rates. Variations in the rates of tank temperature change produced no significant differences in the preference or avoidance temperatures of bluegill compared with control fish, which experienced equal heating and cooling rates. These findings indicate that bluegill will respond behaviorally to variations in the magnitude of thermal reinforcement to maintain temperatures at selected levels.

Much attention has been focused on the responses of fish to temperature gradients, particularly in relation to ecological assessments of power-plant waste-heat effluents. Numerous investigators have reported temperature preference and avoidance statistics of various fish species (see Coutant, 1975) and the effects physical, chemical, and biological factors have on fish temperature behavior.

The influence of experimental thermal manipulations on thermoselection behavior of fish in the laboratory has been examined in research concerning the previous thermal history of test fish (e.g., Doudoroff, 1938; Fry, 1947), the type of gradient used (McCauley and Pond, 1971), thermal shock (Beitinger, 1974), and artificial heating and cooling of the rostral brainstem of fish in thermal gradients (Crawshaw and Hammel, 1974). These experimental manipulations have increased our understanding of the capability, stability, precision, and underlying physiology of the thermoselection behavior of fish.

This paper describes the responses of bluegill (*Lepomis macrochirus*) to variations in heating and cooling rates in a temporal temperature-gradient apparatus. The variations are comparable to changes in the steepness of a temperature gradient in an ecological framework or to changes in the

magnitude of thermal reinforcement in a psychological context.

## MATERIALS AND METHODS

### Apparatus

The test apparatus is described in detail by Beitinger et al. (1975). The design (Neill, Magnuson, and Chipman, 1972) substitutes a modified temporal temperature gradient for the strictly spatial gradient typical of most preferred-temperature studies. Each 50-liter test tank was divided into halves with a molded fiber glass partition. A tunnel in the partition allowed the fish to move from one half of the tank to the other. The temperature of the two halves always differed by 2°C. Fish movements through the tunnel were monitored by a pair of photoelectric cells. Passage of a fish into the warm tank half caused the temperature of the entire tank to increase at a constant rate, and, when a fish swam into the cool half, the temperature of the entire tank decreased at a constant rate. The design allows a fish to control its thermal exposure by moving from one half of the tank to the other.

Temperatures of each tank half were monitored by a thermistor-wheatstone bridge circuit connected to a multichannel analog recorder.

Four interrelated parameters were used to describe the thermoregulatory performance of each fish: (1) lower avoidance temperature; (2) upper avoidance temperature; (3) width of the preferred range; and (4) preferred temperature. The first two parameters are the same as the turn-around temperatures defined by Neill and Magnuson (1974). The width of the preferred range was computed as the arithmetic difference between the two avoidance temperatures, and the preferred temperature was operationally defined as the temperature midway between the two avoidance temperatures.

Although body temperatures were not measured, temperatures of the occupied tank half and those of the fish's

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body were assumed to differ by no more than 0.5°C because of the rapid thermal exchange rates for fish of this size.

## Procedures

Bluegill (mean  $\pm$  standard deviation = 91  $\pm$  10 mm total length) captured by electroshocking during summer from Lake Wingra in Dane County, Wis., were held in the laboratory at 25°C and an L-D cycle of 15 : 9 for at least 2 weeks before experimentation.

One fish was introduced into each test tank and allowed to experience the system for 2.5 days with tank halves set at static temperatures 1.0°C above and 1.0°C below its thermal acclimation state. Then temperature control was relinquished to each fish to begin the test period. Fish were randomly exposed to one of three combinations of temperature rate changes: 3.0°C/hr heating and 1.5°C/hr cooling, 1.5°C/hr heating and 3.0°C/hr cooling, and 3.0°C/hr heating and cooling (control). Thermoregulatory performance was assessed from data collected during the second, third, and fourth days of the test period.

In addition to collecting thermoregulatory performance data, I determined the time each fish spent in the various portions of the tank. The amount of time a fish occupied the tunnel area between tank halves was subtracted from the total time to compute the relative time spent in the cool or warm half. The expected proportion of time allocated to each of the halves is related to the rates of tank temperature change as expressed in the following equation

$$Pt_w = \frac{r_c}{r_c + r_w} \times 100 \text{ hence } Pt_c = 100 - Pt_w$$

where

$Pt_w$  = percent time in warm tank half

$Pt_c$  = percent time in cool tank half

$r_c$  = rate of cooling (°C/hr)

$r_w$  = rate of heating (°C/hr)

## RESULTS

Similarity in thermoregulatory performance of fish exposed to the three combinations of temperature rate change is indicated by comparisons of grand medians (Table 1) calculated from the data of individual fish. No significant differences were observed among the three groups for any of the four performance parameters as indicated by the high probabilities from both paired Mann-Whitney U tests ( $0.60 \leq P \leq 0.94$ ) and Kruskal-Wallis one-way analyses of variance (ANOVA) ( $0.82 \leq P \leq 0.97$ ); two of the ANOVA probabilities suggested that data of the three groups were significantly similar ( $P > 0.95$ ).

The responses of fish as expressed in median percent time in the warm tank half closely approached the theoretically expected values for all three combinations of temperature rate changes: controls, 49.4% (expected, 50.0%); slow cooling, 32.7% (expected, 33.3%); and slow heating, 66.7% (expected, 66.6%). Interquartile ranges (Table 2) calculated for each group demonstrate the small variability in response of individual bluegill.

## DISCUSSION

In this experimental design temperature acts as a positive reinforcement. When a fish occupying the warm tank half becomes thermally uncomfortable, its behavior is reinforced when it enters the alternate or cooler half. First and most directly, the alternate tank half is instantaneously 2°C cooler, and, second, the direction of tank temperature change is reversed from heating to cooling. In experiments

TABLE 1 Thermoregulatory Performance of Bluegill Exposed to Three Combinations of Temperature Rate Changes\*

| Rate change, C/hr<br>(increase : decrease) | N  | Avoidance temperatures         |                                | Preferred temperature          | Preferred range width         |
|--------------------------------------------|----|--------------------------------|--------------------------------|--------------------------------|-------------------------------|
|                                            |    | Lower                          | Upper                          |                                |                               |
| 3.0 : 3.0 (control)                        | 10 | 30.2<br>+0.7<br>-1.1           | 33.4<br>+0.5<br>-0.9           | 31.8<br>+0.4<br>-1.0           | 3.4<br>+0.3<br>-0.3           |
| 3.0 : 1.5 (slow cool)                      | 9  | 30.3<br>+0.3<br>-0.6<br>(0.87) | 33.7<br>+0.3<br>-0.5<br>(0.60) | 32.0<br>+0.4<br>-0.4<br>(0.65) | 3.3<br>+0.9<br>-0.5<br>(0.94) |
| 1.5 : 3.0 (slow heat)                      | 7  | 29.9<br>+0.8<br>-0.7<br>(0.88) | 33.6<br>+0.4<br>-1.2<br>(0.77) | 31.8<br>+0.7<br>-1.0<br>(0.91) | 3.3<br>+0.4<br>-0.5<br>(0.77) |
| Kruskal-Wallis ANOVA†                      |    | 0.96                           | 0.82                           | 0.82                           | 0.97                          |

\*N is sample size. Values in bold face are grand medians computed from individual fish data. The plus and minus values below each median represent the limits of the interquartile range (i.e., median + positive value = 75th percentile, and median - negative value = 25th percentile). Values in parenthesis are probabilities from Mann-Whitney U comparisons between data from fish of each experimental group and control fish.

†Kruskal-Wallis one-way ANOVA probabilities indicate the relationship of data from all three groups for each parameter.



TABLE 2 Percentage Time Spent in the Warm Tank Half by Individual Bluegill Exposed to One of the Three Different Combinations of Temperature Rate Changes

| Control<br>(3.0°C/hr increase :<br>3.0°C/hr decrease) | Slow cooling<br>(3.0°C/hr increase :<br>1.5°C/hr decrease) | Slow heating<br>(1.5°C/hr increase :<br>3.0°C/hr decrease) |              |
|-------------------------------------------------------|------------------------------------------------------------|------------------------------------------------------------|--------------|
| 46.2                                                  |                                                            |                                                            |              |
| 46.3                                                  | 31.8                                                       |                                                            |              |
| 47.5                                                  | 32.2                                                       |                                                            |              |
| 48.2                                                  | 32.3                                                       | 64.4                                                       |              |
| 48.6                                                  | 32.6                                                       | 66.2                                                       |              |
| 50.3                                                  | 32.7                                                       | 66.6                                                       |              |
| 50.4                                                  | 33.3                                                       | 66.7                                                       |              |
| 50.6                                                  | 33.8                                                       | 67.4                                                       |              |
| 51.0                                                  | 34.9                                                       | 69.3                                                       |              |
| 53.6                                                  | 35.4                                                       | 69.8                                                       |              |
| Median                                                | 49.4                                                       | 32.7                                                       | 66.7         |
| Interquartile<br>range                                | 47.2 to 50.7                                               | 32.2 to 34.4                                               | 66.2 to 69.3 |
| Number of<br>samples                                  | 10                                                         | 9                                                          | 7            |

in which there was no temperature differential between tank halves (Neill, Magnuson, and Chipman, 1972), bluegill regulated temperature in the same way as fish in tanks with the usual 2°C difference between halves. Successful thermoregulation in a tank with no temperature differential indicates (1) that bluegill are able to associate changes in spatial location with ultimate (but not immediate) changes in temperature and (2) that direction of temperature change alone can serve as a behavioral reinforcer. Within the context, the effect of variations in the magnitude of thermal reinforcement (i.e., rate of temperature change) on behavioral responses of bluegill was determined.

To thermoregulate while in this system, a fish must distribute its time between tank halves relative to the ratio of the rates of heating and cooling as expressed previously in an equation. Time-distribution data (Table 2) indicated that bluegill responded as expected to all three combinations of heating and cooling rates. The findings were corroborated by the similarity in thermoregulatory performance of the three groups of fish and confirmed that fish were actually controlling their thermal experiences and not merely "swamping out" extreme temperature fluctuations by randomly swimming between tank halves. If the latter had occurred, fish that experienced slow cooling and slow heating rates would have generated avoidance temperatures much higher and much lower, respectively, than control fish. The preferred temperatures of these fish were within ~0.5°C of those for bluegill in other studies (Beitinger, 1974; 1975) where heating and cooling rates were equal.

Similar findings have been reported in other studies of variations in the magnitude of thermal reinforcement and of thermoregulatory behavior. In temperature "titration" experiments (Rozin and Mayer, 1961; Rozin, 1968), individual goldfish were placed in small chambers and

heated from 25°C (acclimation temperature) to 41°C (ultimate upper lethal). Fish were conditioned to press a lever introducing a small jet of cold water that produced a transient decrease of about 0.3°C in the chamber per lever press. When the reinforcement was doubled (i.e., 0.5 to 0.6°C decrease per lever press), the frequency of presses decreased; however, fish maintained nearly the same temperature ranges (Rozin, 1968).

Weiss and Laties (1960; 1961) reported similar results for albino rats held under conditions too severe to permit physiological temperature regulation but, instead, given a behavioral means (a lever) of controlling their environmental temperature. Again, as the magnitude of thermal reinforcement was varied (the relationship between lever press to realized temperature change was altered) rats appropriately modified their response rate and, thereby, controlled their body temperatures at similar levels under a variety of experimental conditions.

The varying response rate relative to the temperature reinforcement by goldfish (Rozin, 1968) and rats (Weiss and Laties, 1960) is comparable to the varying time spent in the respective tank halves by bluegill in the experiments reported here. The responses of animals in three different experimental situations indicate an underlying similarity in behavioral temperature regulation.

## ACKNOWLEDGMENTS

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# Daily Ration, Temperature Selection, and Activity of Bluegill

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## ABSTRACT

Growth consequences of variations in daily ration were studied in relation to temperature preference. Yearling bluegills held at 25°C were fed earthworms at daily rations of 0, 2, 4, and 6% of body weight. After 2 weeks they (about seven fish per ration) were monitored in a temperature-preference device for 5 days. Preferred temperature, but not spontaneous activity, decreased at lower rations; these were adaptations expected for energy conservation. Thus bluegill would be expected to occur at their preferred temperature over long periods of time only if sufficient rations were available. At preferred temperatures they grew on rations of 4 and 6% of body weight but not of 0 and 2%.

Laboratory-derived temperature preferences have a narrow range and are stable for many fishes (Beitinger, 1974). Distributions in the field do not always correspond to laboratory preferences; e.g., fish distributed near outfalls from power plants may be concentrated in the thermal plume in relation to preferred temperature (Neill and Magnuson, 1974), but large numbers of fish occur in other areas of the lake and their distribution cannot be explained by temperature preference. Narver (1970) found in Babine Lake that sockeye salmon, *Oncorhynchus nerka*, made daily vertical migrations over a 10 to 12°C range and that fish spent most of their time in 4 to 6°C water, well below the preferred temperature of 15°C. Brett (1971) hypothesized that the daily migration was a behavior designed to conserve energy by minimizing metabolic rate during some periods of the day. He hypothesized that this behavior was a direct result of a low daily ration.

We believed that Brett's hypothesis might explain the behavior of some fishes in the vicinity of thermal plumes. If the hypothesis had some merit, fishes fed less should prefer cooler temperatures, which would, in turn, reduce their metabolic rate and make it more likely that growth would occur on low rations. We tested this idea with the bluegill, a species for which considerable information was already available.

## MATERIALS AND METHODS

### Experimental Animals

Bluegill used in these experiments were captured from lakes around Madison, Wis., by electroshocker and push

net. In the laboratory, fish were held in 200- to 800-liter stock tanks at 25 ± 2°C. The photoperiod was 12 hr dark and 12 hr light. Fish were fed a pelleted ration daily. Only fish between 75 and 125 mm total length were used in experiments, which were conducted between August 1974 and March 1975.

Two weeks before the temperature-selection phase of an experiment was to begin, 10 fish of similar size were selected from stock tanks. Each fish was blotted with a towel and weighed to the nearest 0.01 g on a top-loading balance. After being weighed, the fish was placed in an individual chamber (125 by 270 by 300 mm) of a 10-chamber growth tank. Chamber bottoms were covered with screen and connected with the larger water bath in which all the chambers were contained. Each chamber was equipped with an airstone. Temperature of the water was controlled at 25°C with a mercury contact thermometer and four submersible heaters. Temperatures were stable within ±0.2°C.

For 2 weeks before the temperature-selection experiments, each fish was fed a daily ration of live earthworms in an amount equal to 0, 2, 4, or 6% of its initial body weight per day. The ration was weighed to the nearest 0.001 g (wet weight). Fish were fed once a day, except those receiving 6%, which were fed in the morning and again in the afternoon each day of the experiment. Feeding continued throughout the experiment, including the period during which temperature selection was determined. During the temperature-selection experiments, fish were fed on whichever side of the tank they were on at the time of feeding. At the end of the 2-week acclimation period, fish were placed in the temperature selection apparatus derived from that described by Neill, Magnuson, and Chipman (1972) and as modified by Beitinger et al. (1975).

### Temperature-Selection Experiments

Fish were placed in separate 50-liter aquaria divided into halves by a barrier with a tunnel large enough for the fish to pass through. The tunnel for each tank was equipped with two photocells on one side and two lights directly across the tunnel. A fish passing through the tunnel would interrupt the light beams in a certain order and cause a

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flip-flop circuit to change state, thereby allowing us to record movements of the fish from one tank half to the other. Movement through the tunnels was recorded for each fish individually on a 24-channel event recorder and simultaneously on a small computer (ADS 1800E).

The change of state of the flip-flop circuit resulting from the passage of a fish through the tunnel was used to allow the fish to regulate tank temperatures.

Each tank half was equipped with a mercury contact thermometer, heaters, cooling coils, and an airstone. Each mercury contact thermometer was connected to a continuously operating reversible electric motor. When the fish was on one side of the tank the motors would turn clockwise, raising the setting of the mercury contact thermometer, and thus the water temperature,  $3^{\circ}\text{C}$  per hour. When the fish was on the other side of the tank, the motors operated counterclockwise, decreasing temperature  $3^{\circ}\text{C}$  per hour. Initially, tank halves were set  $2^{\circ}\text{C}$  apart ( $24$  and  $26^{\circ}\text{C}$ ) for 24 hr to allow the fish to learn that one side of the tank was cooler than the other. The  $2^{\circ}\text{C}$  differential between tank halves was maintained throughout the experiment and provided instantaneous reinforcement for the fish when it sought warmer or cooler temperatures.

### Data Collection and Analysis

Temperature was monitored continuously by thermistors placed in each half of each of the nine experimental tanks and was recorded to the nearest  $0.1 \pm 0.2^{\circ}\text{C}$  every 6 min from each tank half by a small computer. Temperature selection was monitored in this way for 5 days for each group of fish.

Activity of the fish was also monitored continuously and stored in the computer. Activity for these experiments is defined as the number of tunnel passes per hour by a fish.

Statistical tests used were the Mann-Whitney U-test and Spearman rank correlation (Siegel, 1956). The lowest level of significance was chosen to be  $P \leq 0.05$ . Probabilities reported are those obtained from the tables by Siegel. Two fish that died during experiments were excluded from analysis. Occasional short periods of electronic problems were also excluded from analysis.

## RESULTS

### Growth and Daily Ration

During the 2-week acclimation period at  $25^{\circ}\text{C}$ , before the temperature-selection experiments, only the 0% daily ration caused fish to lose weight. Fish receiving a ration of 2% of their initial body weight per day grew at an average rate of  $0.13\%/ \text{day}$ . During the temperature-selection phase of the experiments, both the 0 and 2% fish lost weight. The 2% fish ended the experiments having lost an average of  $0.25\%$  of the initial body weight per day. Fish receiving 4 and 6% daily rations grew throughout the experimental period. The initial mean weight for each of these groups was  $16.0 \text{ g}$  for 0% fish,  $15.6 \text{ g}$  for 2% fish,  $16.1 \text{ g}$  for 4% fish, and  $19.7 \text{ g}$  for 6% fish.

### Temperature Selection

The grand median temperature selected by all fish at all daily rations was  $30.0^{\circ}\text{C}$  during the 5-day temperature-selection period. Comparisons of median temperatures between groups of fish on different rations were significant only between the 0 and 4% daily rations ( $P < 0.01$ , Mann-Whitney U-test). The relationship between growth rate and median selected temperature is apparently somewhat curvilinear (Fig. 1). Fish losing weight were more variable in their median selected temperatures than were fish that were growing. The correlation coefficient between percent change in weight per day and median selected temperature was  $0.46$  (Spearman rank correlation coefficient,  $P < 0.01$ ). Median selected temperatures for each day of the experiment are shown in Fig. 2. The maximum daily median temperature for the 0 and 2% daily ration fish was distinctly lower than the maximum for the 4 and 6%. The relationship between initial weight of an animal and median selected temperature was not significant.

### Activity, Daily Ration, and Temperature Selection

There was no significant relationship between activity in the temperature-selection apparatus and median selected temperatures. Median activity per hour is highly variable from day to day, ranging from 2 to 64 tunnel passes per hour. In comparing grand median activities for each daily ration, we found the 0 and 6% daily ration fish had lower levels of activity than did the 2 and 4% fish (0%, 17; 6%, 16.9; 2%, 22; 4%, 22.2). These differences were not significant.

## DISCUSSION

### Temperature Selection

In previous studies (Beitinger, 1974; Neill and Magnuson, 1974), the temperature preference measured for bluegills was approximately  $31^{\circ}\text{C}$ . The grand median temperature selected by fish in this study was  $30.0^{\circ}\text{C}$ . Fish with a positive growth rate in these experiments had a median selected temperature of  $31.2^{\circ}\text{C}$ , and fish losing weight had a median selected temperature of  $29.8^{\circ}\text{C}$ . The significant difference between these two groups of fish ( $P < 0.025$ , Mann-Whitney U-test) indicates that fish were choosing lower temperatures when daily ration was less than that required for body maintenance. Although  $1.4^{\circ}\text{C}$  is a small change, it would result in a decrease in metabolic rate. Assuming a  $Q_{10}$  of 1.6 and an  $\text{O}_2$  consumption of  $1.222 \text{ mg/hr}$  at  $13^{\circ}\text{C}$  (Eggleston and McComish, 1973) for a 20-g fish, the change from  $31.2^{\circ}\text{C}$  to  $29.8^{\circ}\text{C}$  would result in a decrease of approximately 5% of the standard metabolism.

The group of fish receiving 2% daily ration was anomalous in regard to the other three groups. They tended to decrease their selected temperatures continuously throughout the experimental period after the first day. If this behavior is real, it indicates (1) that the experiments were not of sufficient duration to determine a final

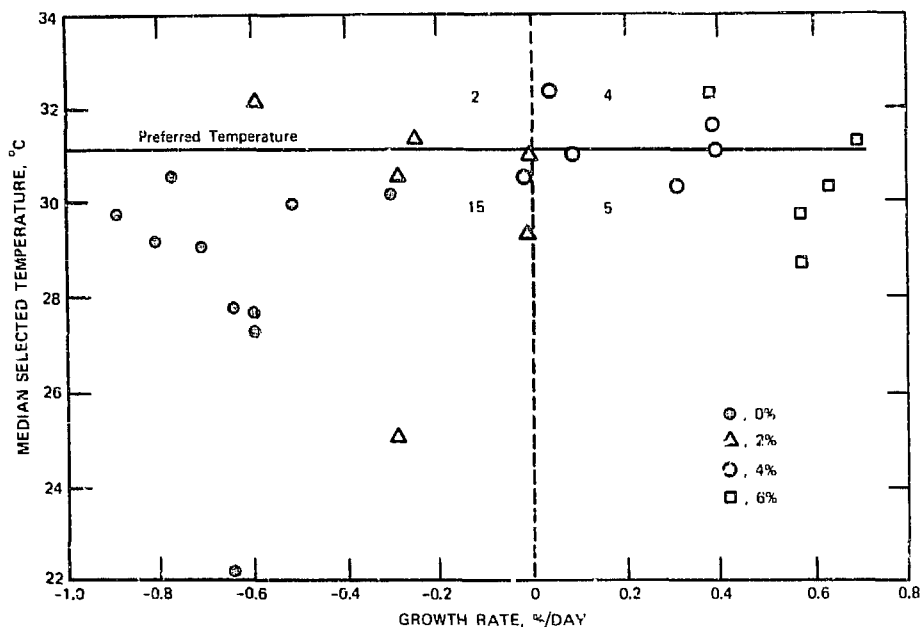


FIG. 1 Median selected temperature as a function of growth rate over 22 days, which included both the 2-week acclimation period and the period of thermoregulation. The preferred temperature is that determined by Beitinger (1974). The numbers in the four quadrats are total numbers of fish in each quadrat.

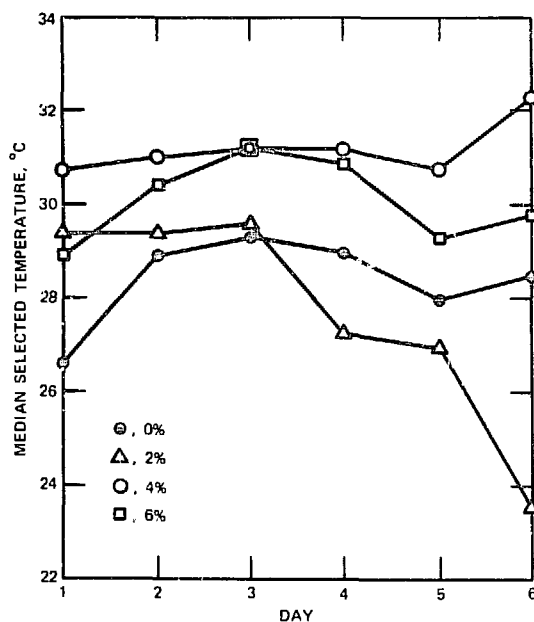


FIG. 2 Median selected temperatures for each day for four different rations. Day 1 began at noon, when thermoregulation began, and the experiments ended at noon on day 6.

temperature preferendum for the fish and (2) that the response of choosing lower temperatures occurs almost immediately after fish begin to lose weight. These fish had been growing at the 2% daily ration when temperatures were held at 25°C. We hypothesize from this that fish would select a temperature at which the feeding level would be sufficient to ensure growth. Long-term experiments will have to be run to test this idea.

#### Activity, Daily Ration, and Temperature Selection

Activity is a prerequisite for temperature regulation in this apparatus. Levels of activity may be quite low, however, and still provide stable temperatures. For example, a temperature range of 10°C could be maintained by movements averaging only 0.32 tunnel passes per hour. All the fish in these experiments had activities higher than this, the minimum being a median of two tunnel passes per hour. This activity level could result in a 3°C temperature range. There is no significant evidence that activity levels were related to the temperature selected.

Evidence that fish use decreased activity levels to conserve energy was not obtained in these experiments (Fig. 3). Activity was highly variable and followed no recognizable patterns.

To use these data for making predictions of behavior in the field, we must quantify several other parameters. Of

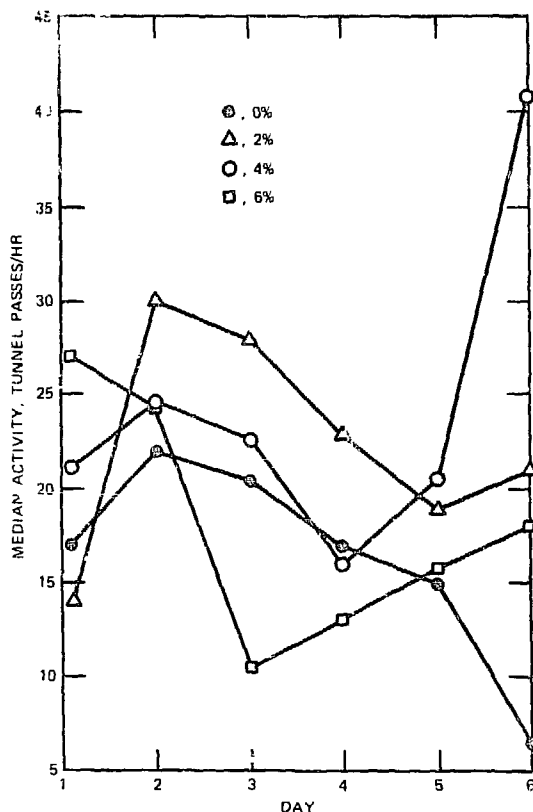


FIG. 3 Median activity for the four daily rations. Day 1 began at noon, and the experiments ended at noon on day 6.

primary importance is the relationship between temperature and the daily rations that will provide enough energy for body maintenance. Without these values it is impossible to predict temperature responses of a fish experiencing decreased food abundance. It is also important to know food availability in the field and, of course, the temperature distribution and its relationship to food availability.

Computer simulations of bluegill energetics by Kitchell et al. (1974) concluded that long-term residence in thermal plumes during summer was disadvantageous for a bluegill unless food was much more available there. Thus it seems reasonable to predict that residence time by an individual bluegill would depend at least in part on food availability.

Although the decrease in temperature preference at lower daily rations was slight, our data lend support to Brett's hypothesis. Bluegills appear to use temperature selection as a method of decreasing rates of weight loss during periods when daily rations are low, while at the same time they maintain rather high levels of activity. Such behavior would appear to be ecologically prudent in that it allows the animals to search large areas for food while reducing weight losses resulting from metabolic costs.

## ACKNOWLEDGMENTS

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# Thermal Preferenda and Behavioral Thermoregulation in Three Centrarchid Fishes

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## ABSTRACT

By use of a new electronic device that permits a fish to control ambient temperature by its movements, it was found that the modal preferenda of three centrarchid species differed more between night and day than did mean occupied temperatures; this reflects skewness of distributions. Some reasons for these differences are advanced.

Locomotor responses of fishes to thermal gradients can serve both physiological and ecological functions. The direct physiological function, in which temperature serves as both proximate and ultimate factor, is behavioral thermoregulation. The body temperature of a fish swimming in a gradient of temperature is probably related to the mean temperature recently occupied by the fish, modified slightly by metabolic heat and by slight differentials in heating and cooling rates due to circulatory adjustments. The size of the fish would affect the magnitude of the latter factors because of a changing mass-area relationship.

Beyond this physiological function of behavioral thermoregulation, temperature preferenda could serve an ecological function by acting as guideposts or sign stimuli by which fish could locate and remain in areas affording a suitable abundance of food, safety from predators, niche separation from competitors, or suitable reproductive sites (Norris, 1963; Lowe and Heath, 1969; Reynolds, 1973; Reynolds and Thomson, 1974; Neill and Magnuson, 1974).

Since in a thermal gradient fish tend to move continually throughout a range of preferred temperatures but spend more time at some temperatures than at others within the range, some measure of central tendency is needed. In experimental investigations of thermal preference, a number of different measures have been used. These include upper and lower turnaround or avoidance temperatures, range of occupied temperatures, midpoint of the range, mode, median, and mean.

The mode of the occupied temperature distribution, called the modal preferendum, has been used as a measure of central tendency in preference studies because it corresponds to the definition of the preferendum as that

temperature occupied most frequently by a fish (Pitt, Garside, and Hepburn, 1956). This statistic is unaffected by unusual values (Arkin and Colton, 1970). The mean temperature has also been used, particularly in studies related to regulation of body temperature. Median body temperatures of freshly caught fish have been used in field studies (Neill and Magnuson, 1974). The range of occupied temperatures (or some variously defined portion of the range) and the midpoint of the range have been used in some laboratory studies (Neill and Magnuson, 1974; Beitingger, 1974b).

If there is skewness in the distribution of occupied temperatures, the various measures of central tendency will fail to coincide. Therefore we described the thermal distribution of fish as completely as possible, using a number of different statistics, to facilitate comparisons among studies and to more fully elucidate all aspects of the fishes' behavior. Different statistics may be most useful in different contexts.

The final preferendum—defined by Fry (1947) as that temperature toward which a fish will finally gravitate regardless of its previous thermal history and at which acclimation and preferred temperature are equal—is generally assumed to be an innate and species-specific measure of thermal behavior. It is largely unaffected by environmental influences. Because of the rapidity of upward acclimation, the final preferendum of warm-water fishes is usually attained in less than 24 hr for fish acclimated above 15 to 20°C (Brett, 1956; Ferguson, 1958; Reynolds, 1973; Reynolds and Thomson, 1974; Crawshaw, 1975a). Therefore the final preferendum can be measured simply by leaving a warm-acclimated fish in a gradient for at least 24 hr. This allows it to adjust its temperature to the final preferendum and to acclimate to this temperature behaviorally and physiologically, as indicated by attainment of a stable preferendum (Reynolds and Thomson, 1974; Neill and Magnuson, 1974; Beitingger, 1974b).

Our goals in this study were to examine some aspects of thermoregulatory and thermal preference behavior in sev-



eral fish species to clarify the relationships among different measurements and, using the data collected, to compare our results with those of investigators who used different methods and measurements and thus to determine the stability and reproducibility of the results of independent investigations.

## METHODS AND MATERIALS

A newly designed electronic device, the Ichthyotron (Fig. 1), was used to measure thermal responses of fish.

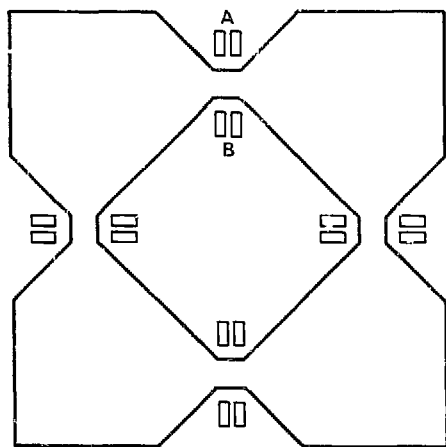


FIG. 1 Schematic diagram of the Ichthyotron; top view. It is 120 cm long and 46 cm high. Light sources are at position A, and phototransistors are at position B.

With this device a fish can control the water temperatures in the chambers, and thus the temperatures it experiences, by swimming between chambers along the temperature axis. This apparatus was inspired by a design described by Neill, Magnuson, and Chipman (1972). Our device differs in that it has four chambers instead of two; this permits simultaneous study of responses to two different variables along the two axes, e.g., temperature and light.

The fish do not pass through a tunnel, as in Neill's device, but through a constriction in which adjustable sliding gates permit them to swim between chambers while sequentially interrupting two light beams and thereby activating phototransistor relays. These, in turn, activate latching relays (one for each axis) which serve as a "memory" of the position of the fish and also activate heating or cooling units, lights, etc.

The size of the gate opening can be adjusted to the size of the fish to permit it to pass without missing the light beams. The height of the passageway can be adjusted to the height at which the species normally swims in the water column (top, middle, or bottom). Also, if desired, the gates can be adjusted so that only two of the four chambers (i.e., one axis) are used to study responses to a single variable.

Tests with a number of species have shown that fish will readily pass through such gates and almost always learn to operate the device proficiently within a few hours. Accord-

ingly, no lengthy training period has been found necessary; the fish simply respond to the thermal gradient of 1 to 2°C which normally exists between the two chambers.

The temperature differential between chambers was variable because of unequal heating and cooling rates, which were, in turn, affected by the difference between water and ambient room temperatures. However, Neill, Magnuson, and Chipman (1972) and Beiting (1974a) have shown that the differential between chambers and relative heating and cooling rates do not affect the thermoregulatory performance of fish.

Continuous records were made of the temperatures in each chamber, and event pens on the recorders, as well as on a separate event recorder, gave redundant records of the position of the fish. From these records temperatures occupied hourly were compiled separately for day and night, using pooled data for all individuals of a species.

Three centrarchid species were used: the bluegill sunfish, *Lepomis macrochirus* Rafinesque; the smallmouth black bass, *Micropterus dolomieu* Lacépède; and the largemouth black bass, *Micropterus salmoides* (Lacépède). Since these species have been extensively studied, results are amenable to comparisons with those of other investigators. The individuals used were captured in northeastern Pennsylvania and were maintained in aquaria at 20 to 24°C under a 12:12 light-dark photoperiod for at least 10 days. They were fed a variety of foods daily up to, but not during, the tests. The 12:12 photoperiod was maintained during the tests.

Nineteen bluegills (45 to 110 mm standard length), 20 largemouth bass (100 to 150 mm), and 20 smallmouth bass (100 to 145 mm) were used in the experiments. Each fish was tested singly for a 3-day period; after a 24-hr acclimation period, second- and third-day data were used.

The resulting temperature distributions were analyzed in degrees Fahrenheit because the recorders were calibrated in Fahrenheit and because most state and federal standards are expressed in Fahrenheit (cf. Stauffer, Dickson, and Cairns, 1974). For comparison with other studies and in accordance with scientific practice, we have presented our results in both Fahrenheit and Centigrade.

## RESULTS

Mean temperatures occupied by bluegills (Fig. 2, Table 1) did not differ significantly between day and night, nor did range, median, or midpoint temperatures. Modal preferenda did differ, however, increasing from 88°F (31.1°C) at night to 90°F (32.3°C) during the day. This reflects a difference in skewness of distributions;\* daytime distributions were negatively skewed ( $S_K = -0.9$ ), but nighttime distributions were not ( $S_K = 0$ ).

Bluegills were the most precise thermoregulators of the species studied and were slightly more precise in their thermoregulatory performance at night than during the day

\* $S_K$ , the Pearsonian coefficient of skewness, is calculated as (mean - mode)/standard deviation (Arkin and Colton, 1970).

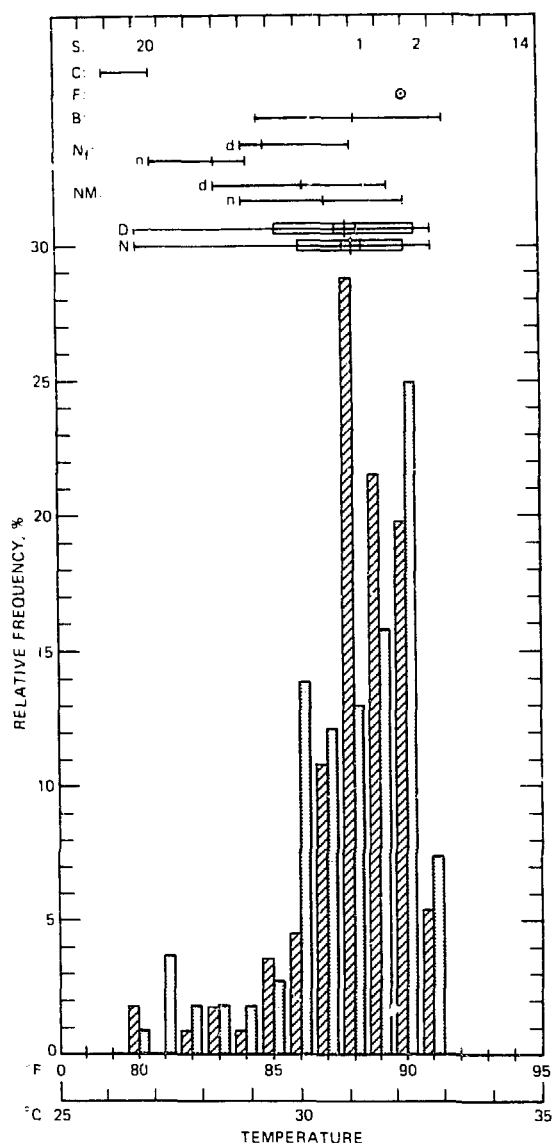


FIG. 2 Relative frequency distribution for bluegills (*L. macrochirus*) in relation to occupied temperatures. Pooled hourly (second and third day and night) data for 19 fish, 4.5 to 11 cm standard length.  $\square$ , night.  $\square$ , day. At the top are modified Dice-Leraas diagrams for day (D) and night (N), showing range and mean  $\pm 7$  standard errors (approximate 95% confidence limits) and  $\pm 1$  standard deviation. Shown for comparison are results of other independent investigations: NM, Neill and Magnuson (1974), laboratory preferred range and midpoints for day (d) and night (n); Nf, Neill and Magnuson (1974), median body temperatures and interquartile ranges for freshly captured fish, day and night; B, Beitinger (1974b), preferred range and midpoint (laboratory); F, Fry (in Ferguson, 1958), final preferendum; C, Crawshaw (1975b), range of intramuscular temperatures, day and night (laboratory); S, Stauffer, Dickson, and Cairns (1974), numbers of fish caught by rotenone at different temperatures.

(despite lower activity at night), as reflected in a smaller standard deviation and standard error of the mean in the nighttime distributions (Fig. 2).

For the two bass species, means, medians, modes, ranges, and midpoints all differed between day and night (Figs. 3 and 4, Table 1), and skewness ( $S_K$ ) shifted from negative values by day to positive values at night.

## DISCUSSION

Our results appear to agree substantially with a number of other independent investigations with these same species, including both field and laboratory studies. Some of the variations in results reported by different investigators apparently are due to the statistics used to report the results, as well as to differences in the techniques used to measure thermal preferenda. Despite this, all the results appear to indicate a satisfactory degree of reproducibility over time and geographic area.

Our differences in day-night modal preferenda of bluegills parallel the day-night median body temperatures of bluegills in summer immediately after capture in the field (Neill and Magnuson, 1974); however, our failure to find a significant difference in mean occupied temperature between day and night conforms to Crawshaw's (1975b) laboratory results for dorsal-muscle temperatures of bluegills. Our day and night means and medians for bluegills nearly coincide with Beitinger's (1974b) laboratory midpoint value, but our midpoint values are lower because we used a complete rather than a partial range. Our daytime mode (90°F, 32.3°C) for bluegills coincides with the final preferendum value of Fry and Pearson (1952) as cited by Ferguson (1958). The range of temperatures occupied by our bluegills (80 to 91°F, 26.7 to 32.8°C) encompasses most of the values reported by other investigators. That the upper end of the occupied range lies quite close to the reported ultimate upper incipient lethal temperature for this species [95°F, 35°C (Hart, 1952)], confirms the frequent observation that final preferenda often lie within a few degrees of the upper lethal temperature for warm-water species (Lowe and Heath, 1969; Reynolds and Thomson, 1974).

The field capture-site ambient temperature distributions of bluegills and smallmouth bass captured with rotenone by Stauffer, Dickson, and Cairns (1974) seem to bracket the laboratory preferred-temperature ranges (Figs. 2 and 3). This may indicate a decreased ability of these fishes to escape the rotenone at marginal temperatures above and below the preferred range. Fry and Pearson (1952) found that the maximum cruising speed of bluegills coincided with the final preferendum, and in other species the preferred range has been correlated with maximum scope for metabolic activity (Fry and Hart, 1948; Brett, 1956) and with maximum response to stimuli (Fisher and Elson, 1950).

Our results for smallmouth bass agree quite well with gut temperatures taken immediately after capture in a thermal effluent area by Trembley (1960) and also with the

TABLE 1. Comparison of Day and Night Temperature-Distribution Statistics for *L. macrochirus*, *M. dolomieu*, and *M. salmoides*

|           |    | <i>L. macrochirus</i> |           | <i>M. dolomieu</i> |           | <i>M. salmoides</i> |           |
|-----------|----|-----------------------|-----------|--------------------|-----------|---------------------|-----------|
|           |    | Day                   | Night     | Day                | Night     | Day                 | Night     |
| Mode      | °F | 90.0                  | 88.0      | 88.0               | 85.0      | 86.0                | 80.0      |
|           | °C | 32.3                  | 31.1      | 31.1               | 29.4      | 30.0                | 26.7      |
| Median    | °F | 88.0                  | 88.0      | 87.0               | 85.0      | 85.0                | 82.0      |
|           | °C | 31.1                  | 31.1      | 30.6               | 29.4      | 29.4                | 27.8      |
| Mean      | °F | 87.8                  | 88.0      | 86.8               | 85.5      | 84.3                | 81.9      |
|           | °C | 31.0                  | 31.1      | 30.4               | 29.8      | 29.1                | 27.7      |
| Range     | °F | 80-91                 | 80-91     | 80-94              | 78-93     | 76-92               | 71-91     |
|           | °C | 26.7-32.8             | 26.7-32.8 | 26.7-34.4          | 25.6-33.9 | 24.4-33.3           | 21.7-32.8 |
| Midpoint  | °F | 85.5                  | 85.5      | 87.0               | 85.5      | 84.0                | 81.0      |
|           | °C | 29.8                  | 29.8      | 30.6               | 29.8      | 28.9                | 27.2      |
| Skewness* | Sk | -0.9                  | 0.0       | -0.4               | +0.2      | -0.5                | +0.4      |

\*Sk is Pearson's coefficient of skewness, calculated as (mean-mode)/standard deviation.

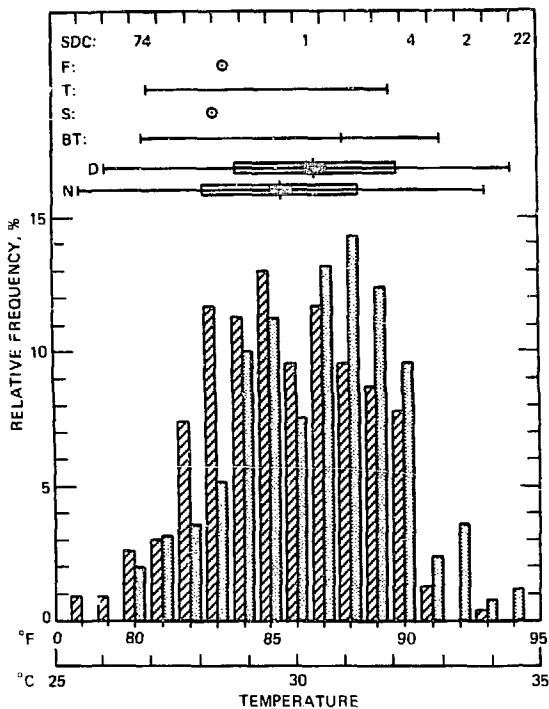


FIG. 3. Relative frequency distribution for smallmouth black bass (*M. dolomieu*) in relation to occupied temperatures. Pooled hourly second- and third-day and night data for 20 fish, 100 to 150 mm standard length.  $\square$ , night,  $\square$ , day. At the top are modified Dice-Leraas diagrams for day (D) and night (N). BT, Barans and Tubb (1973), summer mode and range of 95% of observations (laboratory); F, Fry (in Ferguson, 1958), final preferendum; SDC, Stauffer, Dickson, and Cairns (1974), number of fish captured with rotenone at different ambient temperatures; S, Stauffer et al. (1975), final preferendum calculated by regression; T, Trembley (1960), gut temperatures of fish caught in a thermal effluent area in the Delaware River.

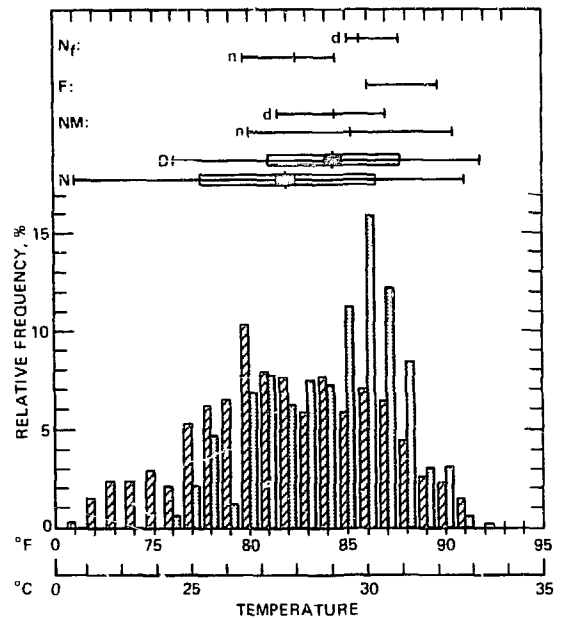


FIG. 4. Relative frequency distribution for largemouth black bass (*M. salmoides*), pooled data for 20 fish.  $\square$ , night,  $\square$ , day. At the top are Dice-Leraas diagrams for day (D) and night (N). F, Fry (in Ferguson, 1958); NM, Neill and Magnuson (1974), (laboratory); NF, Neill and Magnuson (1974), field data (intramuscular temperatures).

preferred range and daytime mode found in the laboratory by Barans and Tubb (1973) for summer-acclimated fish (Fig. 3). Preferendum values given by Stauffer et al. (1975) and by Fry (in Ferguson, 1958) are slightly lower but are still well within the range of our data.

Our results for largemouth bass (Fig. 4) are substantially in agreement with those of Neill and Magnuson (1974) and Fry (in Ferguson, 1958). Our daytime mode

(86°F, 30°C) best fits Fry's preferred range of 30 to 32°C and Neill's daytime median field body temperature (29.8°C). A point of possible interest is that our daytime mode for this species corresponds to a dip in the distribution curve for *M. dolomieu* (Figs. 3 and 4); this suggests a possible aspect of niche separation.

We conclude that when fish are isolated from extraneous stimuli, as in a laboratory gradient of proper design, they do tend to respond to temperature in a reproducible, characteristic, and species-specific manner. Although mean occupied temperatures may best reflect behavioral regulation of body temperature, modal thermal preferenda best reflect day-night differences in thermal behavior. Such circadian shifts in skewness may reflect ecological functions beyond thermoregulation, functions related to differences between day and night activities, with minimal effect on body temperatures. Some interaction between peripheral temperature receptors in the skin (Fry, 1964) and temperature-sensitive neurons in the rostral brainstem (Greer and Gardner, 1970; 1974; Crawshaw and Hammel, 1973; 1974) may account for the shifts in skewness. Acclimation levels probably depend on maximum body temperatures attained rather than directly on maximum ambient temperatures experienced.

Field distribution studies based on capture methods such as rotenone often may be susceptible to sampling artifacts, particularly if fish are better able to escape at optimum preferred temperatures.

Thermal effluents are often the only available source of water in the preferred temperature range for warm-water fishes in north temperate areas. The results of laboratory studies predict that many warm-water species, such as those studied here, should seek out and remain in such effluents indefinitely as a pure response to temperature. Tracking studies (e.g., Kelso, 1974) have shown that fish often evacuate thermal effluents at night, however, and Neill and Magnuson (1974) found the residence times of individual fish in an effluent to be relatively short.

Shifts in modal preferenda may be involved in the evacuation of thermal discharges at night, in conjunction with distraction by other abiotic and biotic factors. Such influences must be evoked to account for the differences between observed field distributions and those predicted by laboratory studies such as we have reported here.

## ACKNOWLEDGMENTS

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# Temperature Preference and Movement of Fish in Relation to a Long, Heated Discharge Channel

William B. Wrenn

## ABSTRACT

Four species of fish (six smallmouth bass, one largemouth bass, two flathead catfish, and one walleye) were tagged with temperature-sensing ultrasonic transmitters during 1972 and 1973 to monitor temperature preference and movement in relation to a 4.5-km heated discharge channel from Colbert Steam Plant on the Tennessee River. Summer effluent temperatures (33 to 35°C) did not create a thermal barrier to the movement of smallmouth bass, largemouth bass, and flathead catfish. One flathead catfish frequently selected temperatures above 30°C. The single walleye tagged remained in the heated discharge 30 days, during which time the effluent temperature increased from 19.2 to 27.4°C.

The development of temperature-sensitive ultrasonic transmitters for tagging fish has enhanced the study of fish behavior in relation to heated discharges from steam electric stations. Since temperature gradients associated with thermal discharges are usually complex, collection of a given fish species from a heated effluent provides only circumstantial information regarding thermal tolerance, preference, or avoidance. Although measurement of the body temperature (Spigarelli et al., 1974) of the captured fish provides additional information on thermal exposure, this technique does not allow continuous surveillance of temperature selection of an unrestrained fish.

The objective of this study was to monitor, via ultrasonic telemetry, temperature preference and movement of four species of fish known to frequent the heated discharge from Colbert Steam Plant on the Tennessee River. Two of the species, smallmouth bass (*Micropterus dolomieu*) and walleye (*Stizostedion vitreum*), are generally considered to be less temperature tolerant than the others, flathead catfish (*Pylodictus olivaris*) and largemouth bass (*Micropterus salmoides*).

## MATERIALS AND METHODS

### Study Area

The Tennessee Valley Authority's Colbert Steam Plant is located on Pickwick Reservoir in northwest Alabama (Fig. 1). It is a five unit fossil-fueled plant with a total

electrical generating capacity of 1396 Mw. At peak load, 56 m<sup>3</sup>/sec of water for condenser cooling is pumped from Pickwick Reservoir and discharged into an oblong basin, ~300 m long and 100 m wide. The heated effluent (5 to 8°C above ambient) passes from the discharge basin down the lower 4.5 km of Cane Creek before returning to the Tennessee River (Pickwick Reservoir) 1.6 km downstream from the plant intake. The water temperature seldom drops more than 1.0°C from the point of discharge to the mouth of the creek.

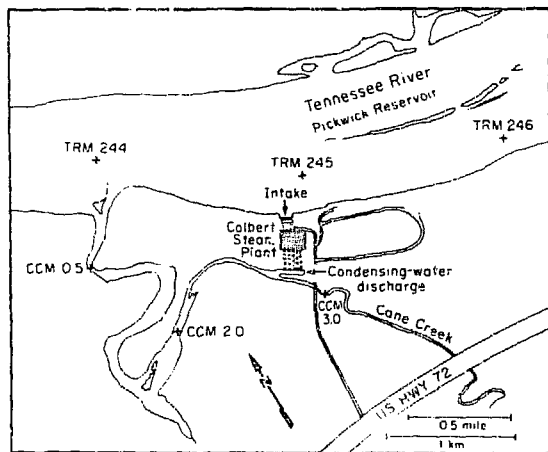


FIG. 1 Map of lower Cane Creek, which receives heated effluent from Colbert Steam Plant. CCM, Cane Creek mile; TRM, Tennessee River mile.

### Telemetry Devices and Tagging Methods

Temperature-sensing ultrasonic transmitters used in this study were developed (Rochelle and Coutant, 1973) and constructed at Oak Ridge National Laboratory. These sonic fish tags were of two sizes: 40 by 17 mm, weighing 23 g, or 32 by 17 mm, weighing 19 g. The thermistor was located at the end of a flexible polyvinyl appendage that extended 100 mm from the tag body. The entire assembly was covered with medical-grade silicone rubber to insure biolog-

Wrenn: Tennessee Valley Authority, Browns Ferry Nuclear Plant, Decatur, Alabama.

TABLE 1 Summary of Results of Experiments Using Temperature-Sensing Ultrasonic Fish Tags at Colbert Steam Plant, Tennessee River

| Species          | Date released  | Period tracked, days | Range of temperatures encountered, °C | Preferred temperatures,* °C | Maximum movement during 24-hr period, km |
|------------------|----------------|----------------------|---------------------------------------|-----------------------------|------------------------------------------|
| Largemouth bass  | Aug. 2, 1972   | 30                   | 27.0–35.0                             | 27–30                       | 0.2                                      |
| Smallmouth bass  | Sept. 13, 1972 | 17                   | 25.8–32.8                             | 26–27                       | 4.6                                      |
| Smallmouth bass  | Nov. 2, 1972   | 35                   | 7.8–19.5                              | 8–15                        | 0.1                                      |
| Smallmouth bass  | Apr. 19, 1973  | 5                    | 14.5–19.3                             | 17–19                       | 1.4                                      |
| Smallmouth bass  | Aug. 24, 1973  | 13                   | 28.0–33.4                             | 28–30                       | 0.3                                      |
| Walleye          | Apr. 27, 1973  | 30                   | 19.2–27.4                             | 19–27†                      | 0.4                                      |
| Flathead catfish | Aug. 25, 1972  | 34                   | 28.0–34.5                             | 31–33                       | 1.5                                      |
| Flathead catfish | Sept. 3, 1973  | 9                    | 28.2–30.7                             | 28–30                       | 0.1                                      |

\*Based on frequency and duration of location at a given temperature.

†Fish was always located within the heated effluent during period tracked.

ical compatibility when implanted in the body cavity. The carrier frequencies ranged from 60 to 110 kHz, and tag life (based on advertised capacity of fresh cells) ranged from 2.5 to 11 months. Temperature information was extracted from the ultrasonic signal by pulse-interval modulation. Calibration accuracy was within  $\pm 0.2^\circ\text{C}$  over a 5 to  $35^\circ\text{C}$  range.

The receiving system consisted of a directional hydrophone with a conical beam pattern of  $8^\circ$ , a tunable receiver modified to produce a digital output signal suitable for time-interval measurement (temperature extraction), headphones, a frequency counter, and a digital recorder. This system was used with a 4.8-m flat-bottom boat and a portable generator for tracking. Headphones increased range and improved signal detection while the boat was in motion.

Ten fish collected by electrofishing were tagged between August 1972 and October 1973. Since sufficient specimens of adequate size could not be collected from the heated discharge area when needed, eight fish were collected in Pickwick Reservoir 24 km upstream from Colbert Steam Plant and released at Cane Creek mile (CCM) 2.9, upstream from the heated discharge (Fig. 1). The largemouth bass and walleye, the only two nondisplaced specimens, were released at point of capture. Except for one flathead catfish and one smallmouth bass, which were held overnight before tagging and release, the fish were collected, tagged, and released on the same day. The fish were anesthetized (10 ppm quinaldine), and the transmitters were implanted in the body cavity via a midventral abdominal incision, with the thermistor protruding externally.

After recovering from anesthesia, the fish were released and were tracked continuously for 4 to 5 hr to check for any immediate ill-effects from tagging and to insure that

the transmitter was functioning properly. Subsequent tracking intervals varied from 2 to 3 hr of continuous observation to intermittent checks of 15 min. Although an attempt was made to locate each fish at least 5 days per week at different periods of the day, randomized observation periods were not established since the fish could not always be located.

After establishing a fish's location, we checked water temperature in its vicinity by a portable electric thermometer. Continuous temperature recorders, installed at five locations in lower Cane Creek, provided additional reference temperatures. Time of day, stream location, water temperature, and fish movement were recorded. Transmitted temperatures were determined from the calibration curve for each tag.

## RESULTS

A summary of the results for eight fish tagged with temperature-sensing transmitters is presented in Table 1. Results for each species are described below.

### Largemouth Bass

A largemouth bass (453 mm in total length and weighing 1.3 kg, tag number 28) was the first fish tagged. It was captured and released at approximately CCM 2.9 on August 2 and was tracked periodically until August 31. During the first 21 days this fish was always located in the ambient zone of Cane Creek, upstream from the heated discharge, except on one occasion when it moved downstream into the steep thermal gradient at the junction of the discharge basin and the creek. Transmitted temperatures, which corresponded to the temperatures measured in the vicinity of the fish, indicated that the largemouth bass selected a temperature of 27 to  $29^\circ\text{C}$  most frequently during the 21-day period; while in the gradient zone (4 hr),

the fish was in water that ranged from 31.0 to 32.2°C. Maximum daily temperatures of the heated effluent ranged from 33.5 to 35.0°C during the 21-day interval. The fish's daily movement during this period ranged from less than 20 to 150 m.

This fish was lost for a week and then was relocated on August 30 in the heated effluent (34.1°C) about 200 m downstream from the discharge basin. It remained at this location and temperature for about an hour before moving upstream into the heated discharge basin. Transmitted temperatures enroute to the discharge basin ranged from 34 to 35°C. After the fish moved into the discharge basin, the transmitted temperature dropped from 35.0 to 30.5°C. The measured effluent temperature in the general vicinity of the bass's location was 35.0°C. The next day this fish was again located in the heated discharge basin; the transmitted temperature was 28.7°C, but the measured effluent temperature was 34.5°C.

The possibility of a cool pocket in the heated effluent was substantiated by a check of the plant operating records, which indicated that two condenser circulating pumps for a unit that was down remained in operation (pumping 28°C river water) during this period. Stationary temperature monitors in the discharge basin upstream and downstream from the unit that was down showed a mean daily temperature of 34.5°C. Water temperature in Cane Creek upstream from the heated discharge was 28.2°C.

Based on the actual time of known location, the largemouth bass remained in the extreme effluent temperatures (34 to 35°C) about 4 hr and crossed the steep thermal gradient (5 to 7°C change within a distance of 10 m) between the heated discharge and the ambient zone of Cane Creek at least three times. On October 5 it was caught in the upper end of the discharge basin by a sport fisherman, who returned the transmitter, which had a dead battery. Expected tag life was 6 months.

### Smallmouth Bass

Of the six smallmouth bass tagged in the study, four were released in conjunction with the high summer effluent temperatures in July, August, and September, one was released in November, and one in April. Three of these fish (two released in the summer and the one in April) were lost within 5 days because of transmitter battery failure or the inability to relocate the fish. Premature battery failure was suspected to be responsible for the loss of numbers 25 and 31, which were released at summer conditions. The fish released in April was tracked periodically for 5 days. During this interval it moved from the ambient zone (16.5 to 16.9°C) of Cane Creek downstream 1.5 km into the heated discharge (19.3°C), where it was last located.

Smallmouth bass number 24 (495 mm in total length and weighing 2.2 kg) and number 26 (419 mm in total length and weighing 1.1 kg) moved downstream from ambient temperatures (27.5 to 29.5°C) into the heated discharge (32.8 to 33.4°C) within 24 hr after being released. Conclusions drawn from its relocation indicate that

number 24 swam through the heated discharge channel and back to Pickwick Reservoir during the first night after release. Rate of travel, estimated from subsequent observations of this fish, was 4.6 km in 12 hr. Number 26 moved downstream from CCM 2.9 into the heated zone within 2 hr after release and remained in the heated discharge (31.8 to 33.4°C) for 3 days before moving upstream into the cooler zone (28.0°C) of Cane Creek. Number 26 did not return to the heated discharge. Maximum rate of movement for this fish was 300 m in 3 hr.

Number 30 (565 mm in total length and weighing 2.7 kg), released in November, was the only smallmouth bass that did not move almost immediately into the heated discharge. This fish remained upstream from the discharge for 34 days, at temperatures ranging from 7.8 to 15.3°C (mean 12°C), before it moved into the heated effluent. In early December it moved from the ambient zone (11.5°C) into the heated discharge (19.5°C). Conclusions drawn from the relocation of Number 30 indicate that it moved across the steep thermal gradient at the junction of the discharge basin and Cane Creek within 24 hr. This fish was tracked only one day in the heated effluent before either it was lost or the battery failed. Expected tag life for number 30 was 6.5 months.

### Flathead Catfish

Both the flathead catfish tracked in this study were displaced from Pickwick Reservoir and released at CCM 2.9. The first, number 29 (711 mm in total length and weighing 5.4 kg), was tracked from Aug. 25 until Sept. 28, 1972. The second, number 33 (600 mm in total length and weighing 3.8 kg), was released on Sept. 3, 1973, and tracked for 2 weeks. Apparently the tag came out of this fish about 9 days after release, however. Temperature selection and movement of number 33 was, therefore, considered inconclusive.

During the first week after release, number 29 moved downstream (~1.5 km) into the heated discharge, back upstream to the ambient zone of Cane Creek, and then downstream (1.5 km) into the heated zone. With the exception of one occasion, when it moved ~100 m in 1 hr, number 29 was not tracked during an extensive move. Since most of the tracking was done between 0830 and 2000, peak movement apparently occurred after 2100 hours. Maximum rate of movement occurred on two occasions when number 29 moved at least 1.6 km in 10 hr or less. This fish appeared to have established a home range (Hart and Summerfelt, 1973) within the heated discharge channel near CCM 1.8.

A summation of the time of actual known location indicated that number 29 spent half the time in the heated effluent (31.0 to 34.5°C) and half in ambient temperatures (28 to 29°C); however, on the basis of probable locations between tracking intervals, it was in the heated zone of Cane Creek 24 days and in ambient zones 9 days. The temperature differential for the three times this fish moved



from ambient temperature into the heated effluent ranged from 4.0 to 5.5°C.

### Walleye

Only one walleye suitable for transmitter implantation was collected. This specimen, number 32 (575 mm in total length and weighing 1.5 kg), was released at CCM 2.2 (point of capture) on April 27. It was tracked for 30 days and was always located in the heated effluent. The transmitted tag temperature during this interval ranged from 19.2 to 27.4°C, which corresponded to instantaneous and recorded measurements of the effluent temperature. Ambient temperatures in Pickwick Reservoir ranged from 14 to 21°C for the same period. During the last 16 days the walleye was in the heated effluent, the temperature was above 25°C. The last signal from number 32 was on May 26; it was tracked about half the expected tag life. Daily movement was generally less than 50 m, but on one occasion it moved 450 m.

### DISCUSSION

Although the walleye remained in the heated discharge for at least 30 days, the maximum effluent temperature during this interval (27.4°C) was similar to the maximum temperature at which this species has been collected from the heated zone of Cane Creek. Bimonthly electrofishing and gill-net sampling during a 2-year study (Wrenn, 1974) yielded 26 walleye from the heated zone of Cane Creek; the species occurs here from early December (16°C) to mid-May (25°C). Walleye, uncommon in Pickwick Reservoir, also concentrate in small numbers in the tailwaters of Wilson Dam (24 km upstream in Pickwick Reservoir) during this same period at temperatures ranging from 5 to 18°C. Trembley (1960) reported that the maximum body temperature of a walleye (apparently not under heat stress) collected in a heated effluent on the Delaware River was 28.9°C. The upper lethal temperature (Smith and Koenst, 1975) for walleye acclimated at 26°C was 31.6°C under laboratory conditions. On the basis of these observations, I concluded that walleye occur in the heated zone of Cane Creek in the fall somewhat independent of water temperature, but, as effluent temperatures approach 30°C in the summer, the species returns to Pickwick Reservoir.

Two smallmouth bass tracked in the summer moved into the heated zone at 33°C, which is greater than the reported maximum summer preference for this species [31°C (Barans and Tubbs, 1973)] but less than the upper incipient lethal temperature [about 38°C (Larimore and Duever, 1968)]. Both fish returned to ambient conditions within 1 to 3 days, but one moved downstream 4.5 km to Pickwick Reservoir (where temperatures were 26 to 27°C) and the other returned upstream to the ambient zone of Cane Creek. In the heated zone of Cane Creek temperature conditions exceeding summer preference but less than lethal for smallmouth bass generally occur from mid-June through mid-September. Results obtained from specimens

tagged with temperature-sensing ultrasonic transmitters, as well as from specimens of this species collected at temperatures above 34°C (Wrenn, 1974), indicated that summer effluent temperatures in lower Cane Creek, although not necessarily preferred, would not be a barrier to movement of smallmouth bass, which are abundant in Pickwick Reservoir.

The smallmouth bass released in November apparently remained upstream from the heated discharge for over 30 days at temperatures (7.8 to 15.3°C; mean 12°C) somewhat below the winter preference temperature of 18°C reported for this species (Barans and Tubbs, 1973). This fish eventually moved into the heated zone, however, when the effluent temperature was 19.5°C.

Although the transmitters used in the smallmouth bass apparently had an abnormal rate of premature battery failure (Rochelle and Coutant, 1973), the extensive mobility demonstrated by smallmouth bass in this study also indicated that it is quite possible to lose track of the fish before the tag fails.

### ACKNOWLEDGMENTS

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# Thermal Ecology of Phytoplankton in a Desert Reservoir

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## ABSTRACT

The physical-chemical limnology and phytoplankton dynamics of Canyon Lake, Arizona, were investigated from February 1971 to July 1973. The reservoir is a warm monomictic lake with pronounced thermal stratification during the summer months. Chemically the lake is hard water of moderate to high alkalinity and salinity. Annual peaks in the phytoplankton standing crop were recorded during early spring and mid- to late summer, and significant depressions occurred during April-May and November to January. The spring peak was composed primarily of centric diatoms, whereas the summer peak was dominated by filamentous Cyanophyceae. The seasonal appearance and variation in population size of individual species were correlated in varying degrees with one or more physical-chemical parameters. Indications are that physical factors (particularly light and water temperature), rather than water chemistry, were the primary parameters regulating seasonal succession.

Although the morphology of algae is generally quite simple compared with terrestrial plants, the factors determining their distribution and abundance are complex. Despite the literally thousands of publications dealing with algal distribution, comparatively few generalizations can be made concerning the phytoplankton ecology of lakes (Lund, 1965; Hutchinson, 1966).

Although aquatic ecologists disagree on the relative importance of various environmental parameters regulating the growth and distribution of algae, it is clear that water temperature and light are important variables (Fogg, 1965; Round, 1965; 1971). This fact has been iterated in the publications of Patrick (1969; 1971), Patrick, Crum, and Coles (1969), and Round (1968), who present conclusive evidence that natural and induced changes in water temperature and/or light can affect both the qualitative and quantitative structure of algal communities.

Considerable descriptive and ecological data have been compiled for algal communities in temperate and humid-subtropical regions; however, information concerning the phytoplankton ecology of desert lakes (particularly thermal aspects) is very limited. Studies of tropical lakes in the arid regions of East Africa by Ross (1955), Talling (1966), Richardson (1968), Lind (1968), and Hecky and Kilham

(1973) provide some indication of the types of phytoplankton assemblages that may characterize desert lakes. Comparatively little seasonal or ecological information is available, however. In addition, no comprehensive ecological study of planktonic algae had been undertaken in the Sonoran Desert region of the United States before our study. Hostetter (1968) studied the planktonic diatoms in three mountain lakes in southern Arizona; Olsen and Sommerfeld (1970) conducted a cursory examination of the planktonic diatoms of six Sonoran Desert lakes in central Arizona; and Button and Blinn (1973) studied the algae in a high-elevation lake in northern Arizona. Other studies of Arizona algae have been confined to small ponds or creeks (Taylor and Colton, 1928; Kidd and Wade, 1963; 1965; Cameron, 1964a; Markey and Hevly, 1973) or to nonaquatic habitats (Cameron, 1960; 1964b; Luty and Hoshaw, 1967).

This paper presents information on the ecology (emphasizing thermal aspects) of dominant phytoplankton species in a desert lake. The data were gathered during an intensive study of reservoirs in central Arizona. The primary purpose of the study was to obtain information on species composition and seasonal variation of phytoplankton in a desert lake. The ultimate objective was to determine the environmental factors responsible for species succession in this habitat. Olsen (1975) presented a comprehensive treatment of the data.

## DESCRIPTION OF THE STUDY AREA

Canyon Lake is located in eastern Maricopa County, Arizona, 55 km east of the Phoenix metropolitan area, at 111° 26' west longitude and 33° 32' north latitude. The full-pool elevation of the lake is ~500 m above sea level. Morphometric characteristics of the lake basin are shown in Table 1. The lake is the third in a series of four impoundments on the Salt River of central Arizona and lies entirely within Tonto National Forest. Physiographically this region is in the Mountain Region subdivision found along the northern border of the Basin and Range Province (Wilson, 1962). The reservoir receives no industrial or urban effluent and at the present time is not subjected to induced thermal perturbation.

Olsen: Argonne National Laboratory, Argonne, Illinois.  
Sommerfeld: Arizona State University, Tempe, Arizona.

TABLE 1 Morphometric Characteristics of Canyon Lake

|                                          |      |
|------------------------------------------|------|
| Maximum depth, m                         | 38.1 |
| Mean depth, m                            | 18.6 |
| Maximum width, km                        | 0.95 |
| Mean width, km                           | 0.25 |
| Maximum length, km                       | 16.0 |
| Shoreline length, km                     | 45.6 |
| Surface area, km <sup>2</sup>            | 3.8  |
| Volume ( $\times 10^7$ ), m <sup>3</sup> | 7.1  |
| Volume development index                 | 1.46 |
| Shoreline development index              | 1.68 |
| Slope of basin, %                        | 18.5 |

This rugged part of the Sonoran Desert receives about 25 cm of precipitation per year. July and August are usually the wettest months, and December through March is another wet period. Drought conditions generally occur in spring (April through June) and fall (October and November). Average minimum air temperatures for January, the coldest month, are always above freezing in the desert region, and summer temperatures often exceed 45°C (Green and Sellers, 1964). Because of the high temperatures, low relative humidity, and prevailing winds, the evaporation rate during summer is extremely high. According to the heat budget calculated for Roosevelt Lake (Rinne, 1973), the high summer evaporation rates result in significant heat loss from the lake surface and probably exert a cooling effect on the epilimnion.

## METHODS AND MATERIALS

### Field

Water samples were collected monthly at four sites (Fig. 1) on Canyon Lake from February 1971 to July 1973. Samples from the surface, 2, 5, 10, and 20 m were collected at each site during 1971 and 1972, using a 2-liter nonmetallic Kemmerer water sampler. In 1973 only surface and 2-m samples were collected. Samples were placed in acid-washed (10% HCl) 2-liter polyethylene bottles and stored on ice until they were returned to the laboratory. Immediately after collection a 200-ml aliquot of each sample was filtered for PO<sub>4</sub> and NO<sub>3</sub>-N analyses using a Millipore Swinnex-47 filter apparatus with a 50-ml plastic syringe and a prerinsed 0.45- $\mu$  membrane filter (Millipore Corporation, Bedford, Mass.). The filtered subsamples were also stored on ice in plastic bottles. A colorimetric pH determination was performed on each unfiltered 2-m sample using a Hach field colorimeter model DR-EL (Hach Chemical Co., Ames, Iowa). Water temperature and dissolved oxygen were measured monthly at 1-m intervals from the surface to the bottom at each sampling site with a Hydrolab model TDO (Hydrolab Corp., Austin, Tex.). Temperature and dissolved oxygen function of the instrument were checked and calibrated before each sampling period. A Secchi disk reading was recorded at each sampling

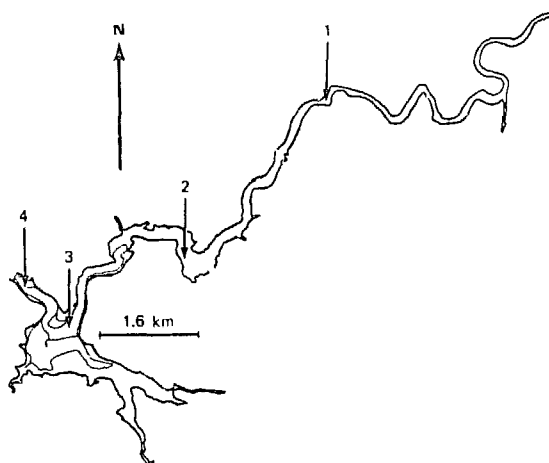


FIG. 1 Canyon Lake, Arizona, showing locations of the four regular sampling sites.

site using a standard 20-cm Secchi disk. Data on monthly average incident solar radiation and air temperature in the Canyon Lake area were supplied by the National Weather Service station at Phoenix.

### Laboratory

All water samples collected for subsequent phytoplankton analysis were stored in the dark at 5°C. Phytoplankton from 500 ml of each sample were concentrated to 25 ml by centrifugation at 3200 rpm for 20 min at 5°C with a refrigerated centrifuge. One milliliter aliquots of the concentrated phytoplankton sample were introduced into a Sedgewick-Rafter cell using a Hensen-Stempel pipet. The cell was then inverted, and the algal cells were allowed to settle for 20 min. A 1.0-mm ocular net micrometer was used to delimit fields for enumeration. Forty fields, ten on each of four counting cells, were counted at a magnification of 300X for non-diatom phytoplankton.

Because of the small cell sizes of most of the dominant diatoms found in the phytoplankton, it was necessary to identify and enumerate them separately. Permanent slides for identification and enumeration were prepared by placing 0.5 ml of the phytoplankton concentrate evenly over the surface of a 22-mm square cover glass. The cover glass was dried, incinerated on a hot plate at 450°C for 1 hr, and mounted in Hyrax on a microscope slide. Five random scans across the complete length of the cover glass were counted at a magnification of 300X using a 1.0-mm net micrometer.

When we returned to the laboratory after sample collection (an interval of 2 to 3 hr), we filtered a 250-ml aliquot of each original sample, using a 0.45- $\mu$  membrane, for dissolved metal analysis. The filtered sample was acidified with 1 ml of concentrated reagent grade HCl and stored in a polyethylene bottle at 5°C. All glassware,

pipettes, and filters used in analysis procedures were rinsed with 10% HCl, deionized water, and double-distilled water.

Analyses of the metals Ca, Mg, Na, K, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, and Zn were performed on the filtered, acidified samples (following appropriate dilution or concentration procedures) with a Perkin-Elmer atomic absorption spectrophotometer model 403 (Perkin-Elmer Corporation, Norwalk, Conn.). A detailed description of the specific procedures was presented elsewhere (Olsen, 1975). A Hach colorimeter and Hach reagents were used to analyze field-filtered samples for  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ ,  $\text{PO}_4$ ,  $\text{SiO}_2$ , B, and  $\text{SC}_3$ . Analyses for  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4$  were completed within 24 hr after sample collection. Hach reagents were also used in titrimetric tests for alkalinity, hardness, and chloride. Hach procedures are modifications of *Standard Methods* (1967) involving reduced sample size and use of premeasured reagents. The accuracy and precision of the Hach analytical system, evaluated using U.S. Environmental Protection Agency (EPA) reference samples, were found to be within  $\pm 10\%$  of the EPA calculated concentrations for all parameters tested. Accuracy and precision of  $\text{PO}_4$  and  $\text{NO}_3\text{-N}$  analyses were poor below 20 ppb, and detection limits were  $\sim 10$  ppb. Test analyses were run on  $\text{NO}_3$  and  $\text{PO}_4$  standard solutions before each sampling period to assure proper colorimeter operation.

Various statistical analyses were applied to the data (for samples for 2-m depth only) collected during the study to aid in interpretation (Olsen, 1975). We performed all statistical analyses by computer (Univac 1110, Sperry Univac, Bluebell, Pa.), using programs supplied by the Arizona State University Campus Computing Service. To normalize biological population data, we performed a logarithmic transformation on all cell numbers before analysis.

## RESULTS AND DISCUSSION

### General Observations on Limnology and Phytoplankton Ecology

The major limnological characteristics of Canyon Lake are summarized in Table 2. The lake can be described chemically as a hard-water lake with moderately high alkalinity and salinity. Dominant cations and anions in order of abundance are, respectively,  $\text{Na} > \text{Ca} > \text{Mg} > \text{K}$  and  $\text{Cl}^- > \text{HCO}_3^- > \text{SO}_4^{2-}$ . All major ions were subject to periodic dilution and concentration associated with alternating periods of drought and heavy precipitation.

Canyon Lake is a warm monomictic lake according to the definition of Hutchinson (1957). Thermal stratification with classical thermocline development occurred from mid-May to September and October, whereas the period of holomixis extended from November to mid-February. The thermocline was usually found at a depth of 4 to 6 m. The minimum water temperature during the winter homothermal period was 8 to  $10^\circ\text{C}$ , and maximum epilimnetic temperatures during summer approached  $30^\circ\text{C}$ . Hypolim-

TABLE 2 Summary of Limnological Parameters Monitored at Canyon Lake\*

| Parameter                      | Mean  | Standard deviation | Range        |
|--------------------------------|-------|--------------------|--------------|
| Chlorophyll a, $\text{mg/m}^3$ | 10.65 | 9.56               | 0.72-53.67   |
| Standing crop, no./ml          | 7095  | 8924               | 108-72766    |
| $\text{O}_2$ , ppm             | 9.69  | 2.13               | 5.5-15.5     |
| $\text{O}_2$ saturation, %     | 108.8 | 22.3               | 65.2-155.0   |
| pH                             | 8.30  | 0.34               | 7.60-8.85    |
| Secchi disk, m                 | 1.48  | 0.72               | 0.40-3.35    |
| Total alkalinity, ppm          | 137   | 23                 | 90-178       |
| Chloride, ppm                  | 252   | 78                 | 110-345      |
| Total hardness, ppm            | 178   | 30                 | 120-260      |
| Sulfate, ppm                   | 58    | 9.3                | 42-75        |
| Silica, ppm                    | 12.7  | 2.2                | 8.0-15.9     |
| $\text{PO}_4$ , ppm            | 0.082 | 0.066              | <0.010-0.310 |
| $\text{NO}_3\text{-N}$ , ppm   | 0.251 | 0.360              | <0.010-1.460 |
| Calcium, ppm                   | 42.9  | 6.7                | 32.0-63.8    |
| Magnesium, ppm                 | 12.7  | 3.3                | 9.3-18.0     |
| Sodium, ppm                    | 141   | 52                 | 75-232       |
| Potassium, ppm                 | 4.7   | 1.4                | 1.4-6.2      |
| Iron, ppb                      | 7.7   | 4.2                | 0.8-21.0     |
| Copper, ppb                    | 3.9   | 1.3                | 1.0-8.0      |
| Manganese, ppb                 | 3.0   | 4.7                | <0.5-27.0    |
| Zinc, ppb                      | 3.0   | 2.6                | 1.0-15.0     |

\*Based on all 2-m samples collected during the 30-month study period ( $n = 120$ ).

netic temperatures during the months of maximum epilimnetic temperatures (July and August) ranged from 17 to  $18^\circ\text{C}$  at 10 m to about  $15^\circ\text{C}$  near the lake bottom (20 to 35 m).

As the lake began to stratify thermally in April or May, the concentration of dissolved oxygen below the developing thermocline decreased. Oxygen concentration at 20 m remained above 2 ppm during the summer stratification period, but concentrations near the bottom sometimes fell to below 1 ppm. Oxygen was not completely depleted, apparently because of significant water movement through the hypolimnion as a result of water release from the upstream reservoir.

The primary nutrients,  $\text{PO}_4$  and  $\text{NO}_3\text{-N}$ , exhibited the most pronounced seasonal variation of any of the chemical parameters measured. In general, the two compounds were present in small quantities in the photic zone during the summer (10 to 50 ppb). Then, upon destratification and decline of the phytoplankton populations in the fall, concentrations began to increase, eventually reaching a peak during holomixis in December or January. The maximum concentrations ranged from 200 to 300 ppb for  $\text{PO}_4$  and 500 to 1500 ppb for  $\text{NO}_3\text{-N}$ . As the phytoplankton standing crop increased during late winter and early spring, the nutrient concentrations in the photic zone decreased, eventually reaching levels below 50 ppb by April. On the basis of the observed nutrient levels, we can describe Canyon Lake as eutrophic.

The periodicity of Canyon Lake phytoplankton corresponded roughly with the classic pattern described for moderate-sized eutrophic lakes (Hutchinson, 1966). Phytoplankton standing crop exhibited two distinct annual maxima. The first peak in late winter or early spring was composed almost entirely of centric diatoms, but the late summer maximum was primarily Cyanophyceae, with a few Chlorophyceae and pennate diatoms present as subdominants. No clearly defined fall peak was observed. Minimum phytoplankton growth was recorded during mid-winter when incident solar radiation was lowest and turbidity was high. The vernal diatom increase occurred at low water temperature commensurate with increasing solar radiation. Chrysophyceae and Chlorophyceae assumed dominance following decline of the vernal diatom populations when water temperature was increasing and nutrients were at lower levels. Blue-green algae became dominant during summer, when water temperatures approached the annual maximum and nutrients were at lowest annual concentrations. All phytoplankton populations declined rapidly during fall with onset of holomixis and decrease in incident solar radiation.

A factorial analysis of variance, where variance was partitioned into sites, was used to compare the four sampling sites in Canyon Lake (Fig. 1) for the entire 30-month study period (2-m depth samples only). No statistically significant (0.10 level) difference was found between the four sites for any of the physical-chemical and biotic parameters investigated.

### Thermal Ecology of Major Phytoplankton Taxa

*Cyclotella* sp. The centric diatom genus *Cyclotella* was the only true perennial taxon found in Canyon Lake. The dominant species was *C. meneghiniana*, but *C. atomus* and *C. striata* were also found in significant numbers. The three species always occurred together, and, since they could not be distinguished using the normal enumeration procedure, all were recorded as *Cyclotella* sp.

The periodicity of *Cyclotella* sp. in Canyon Lake was highly seasonal, with maximum population size developing in late winter to early spring [Fig. 2(a)]. Maximum cell numbers during the annual peak ranged from 10,000 to 20,000 per milliliter. Lowest annual populations were found from April to August and November to January. The winter depression in the population was associated with low light penetration (low Secchi disk values) and the lowest annual levels of incident solar radiation. The rapid increase in cell numbers to a vernal maximum was graphically correlated with the annual increase in incident solar radiation, the threshold level being ~250 to 300 langley/day (Olsen, 1975). The vernal bloom also coincided with increased light penetration.

The initiation and peak of the vernal *Cyclotella* bloom was associated with little or no increase in water temperature over the winter minimum of 8 to 10°C [Fig. 2(a)], but the rapid decline in cell numbers in the spring was closely associated with increasing water temperature. The decline

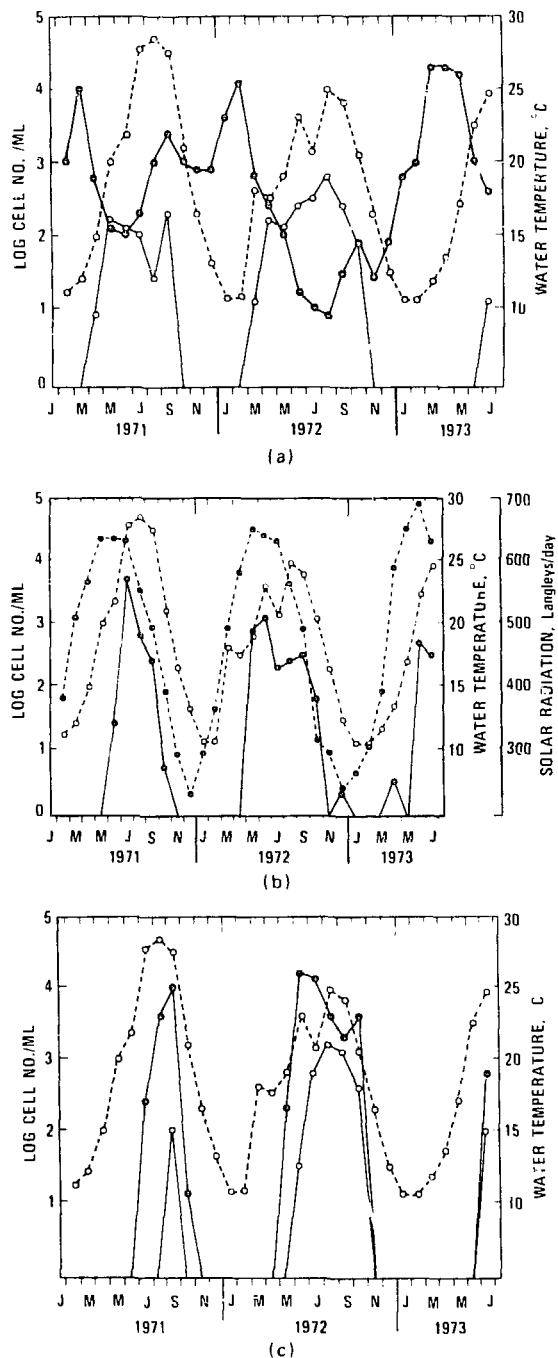


FIG. 2 (a) Monthly plot of numbers of *Cyclotella* (—●—), numbers of *Closteriopsis longissima* (---○---) and water temperature (○---○) at a depth of 2 m at site 3. (b) Monthly plot of numbers of *Synedra filiformis* (—●—), water temperature (○---○) at 2-m depth at site 3, and monthly average daily total incident solar radiation at the Phoenix, Arizona, Weather Station (○---○). (c) Monthly plot of numbers of *Spirulina laxissima* (—●—), numbers of *Anabaenopsis raciborskii* (---○---) and water temperature (○---○) at a depth of 2 m at site 3.

in *Cyclotella* numbers coincided with a rise in water temperature above 15 to 17°C at a depth of 2 m. This observation is in agreement with preliminary laboratory culture experiments, which indicate that *C. meneghiniana* isolated from Canyon Lake grows significantly faster at 10°C than at 20 or 25°C (Garmon, 1975). Correlation analysis of the *Cyclotella*-temperature relationship in Canyon Lake revealed a low but significant (0.05 level)  $r$  value of  $-0.446$ .

The *Cyclotella* bloom in Canyon Lake began when nutrient levels were at their highest annual concentrations. Concentrations of both  $\text{PO}_4$  and  $\text{NO}_3$  declined in an apparent direct relationship with increasing *Cyclotella* numbers during development of the vernal population peak, but no significant statistical correlation was found (Olsen, 1975). It is possible that decreased nutrient levels contributed to population decline, but the population peak in 1971 occurred at low nutrient concentrations, and in 1973 the peak was sustained for 2 months despite low nutrient levels.

Lin (1972) found a large increase and subsequent bloom of *C. meneghiniana* in a small eutrophic Canadian lake under a thin ice cover during spring in association with increasing light and high nutrient levels but with little or no increase in water temperature. He concluded that rising water temperature and silica depletion were the chief factors causing decline of the population in June.

Apparently light and water temperature are the principal environmental factors regulating *C. meneghiniana* periodicity in Canyon Lake and possibly in other aquatic habitats.

*Stephanodiscus astraea*. In Canyon Lake the centric diatom species *Stephanodiscus astraea* was represented by two concurrently occurring taxa, *S. astraea* var. *minutula* and the larger nominate variety *S. astraea*, in a ratio of about 10:1.

In terms of its periodicity *Stephanodiscus* was restricted to the cold-water period (October to March) and generally reached its maximum cell numbers (200 to 300/ml) in February or March. Decline of the population in spring coincided with increased water temperature and decreased nutrient concentrations (Olsen, 1975). Correlation analysis produced significant (0.05 level) negative correlation coefficients for water temperature ( $r = -0.814$ ), and incident solar radiation ( $r = -0.450$ ) and a weak but significant positive correlation with  $\text{PO}_4$  ( $r = 0.382$ ). Solar radiation did not seem to be directly responsible for the periodicity. It is apparent that the species is able to exist and multiply at the lowest annual light levels.

Thermal stratification also appeared to be an important influence on periodicity of *S. astraea* in Canyon Lake. The decline in population size in all 3 years of the study occurred with development of stratification, and population increase was correlated with holomixis (Olsen, 1975). Talling (1966) found *S. astraea* to increase markedly during isothermal mixing of the water column in Lake Victoria

(Africa) and postulated that thermal stratification controlled the periodicity of the species. *Stephanodiscus astraea* is a heavily silicified diatom with an unfavorable weight-to-volume ratio and a high sinking rate, which would be an obvious disadvantage during periods of stratification.

*Closteriopsis longissima*. The chlorococcalean alga *Closteriopsis longissima* was found in Canyon Lake only in spring and summer; maximum cell numbers were near 1000/ml. The periodicity [Fig. 2(a)] was closely related to water temperature and incident solar radiation. Correlation coefficients significant at the 0.05 level were found for water temperature ( $r = 0.705$ ) and incident solar radiation ( $r = 0.541$ ). The organism did not appear in the plankton until water temperatures had reached  $\sim 15^\circ\text{C}$  and light levels were at or near annual maxima. The decline in cell numbers in late summer seemed to coincide with decreasing water temperature and incident solar radiation. Cell numbers were also significantly correlated with light penetration (Secchi disk values), a fact that seems to emphasize the importance of light. *Closteriopsis* periodicity was also related to thermal stratification, but the population peak in September 1971, subsequent to turnover, indicated that stratification was not mandatory for maintenance of the population (Olsen, 1975).

The concentration of the primary nutrients  $\text{PO}_4$  and  $\text{NO}_3\text{-N}$  remained low, with no significant fluctuations during the periods when *C. longissima* was present (Olsen, 1975). Correlation analysis of the *Closteriopsis*-nutrient relationship yielded weak but statistically significant (0.05 level) correlation coefficients for  $\text{PO}_4$  ( $r = -0.498$ ) and  $\text{NO}_3\text{-N}$  ( $r = -0.430$ ). The morphology of *Closteriopsis* (long and narrow with greatly attenuated ends, a characteristic shared with other summer species) may very likely confer an advantage to the organism under low nutrient conditions since cells with a high surface-to-volume ratio generally possess lower half-saturation constants ( $K_s$ ) for nutrient assimilation (Fogg, 1965).

*Synedra filiformis*. In Canyon Lake the attenuated pennate diatom *Synedra filiformis* was restricted in occurrence to the warm-water months (May to October), when it generally achieved cell numbers of 2000 to 5000/ml. During the 3 years of the study, *S. filiformis*, did not appear in the plankton until incident solar radiation had reached annual peak levels [Fig. 2(b)]. This period also coincided with an increase in 2-m water temperatures to 20 to 25°C [Fig. 2(b)]. Not only did the peak in population size correspond to the peak in solar radiation but also the decline in cell numbers correlated with decreasing light. Statistical analysis of the population data revealed significant (0.05 level) negative correlations with the primary nutrients  $\text{PO}_4$  ( $r = -0.507$ ) and  $\text{NO}_3$  ( $r = -0.373$ ) and significant positive relationships with water temperature ( $r = 0.816$ ) and incident solar radiation ( $r = 0.556$ ). Graphic display of the data [Fig. 2(b)] suggests a closer relationship to light levels than to water temperature, although the

latter does appear to be important. We should point out that Holland and Beeton (1972) reported this species to be most abundant in Lake Michigan in mid-June when light levels would be expected to be high but when the reported water temperature was only  $\sim 10^{\circ}\text{C}$ . This suggests that light may be more important than temperature for maximum population development.

The population development and decline was associated with no major change in  $\text{PO}_4$  or  $\text{NO}_3\text{-N}$  concentrations (Olsen, 1975), but, since in the summer nutrients were at the lowest annual levels, it is conceivable that low primary nutrient concentrations may be a factor determining periodicity; i.e., *S. filiformis* may possess the ability to develop at low nutrient levels. This point is supported by the fact that in Lake Michigan much larger populations of *S. filiformis* were found in offshore areas where nutrient levels were lowest (Holland and Beeton, 1972).

*Spirulina laxissima*. The blue-green alga *Spirulina laxissima* is quite rare in North America but is common in many tropical lakes—particularly those of East Africa, where it is found in lakes of moderate to high alkalinity and salinity (Lind, 1968).

*Spirulina* was unquestionably the dominant summer phytoplankton organism in Canyon Lake. Maximum cell numbers ranged from 50,000 to 75,000/ml. This species was restricted in distribution to the period from May to October and exhibited a close association with water temperature [Fig. 2(c)]. The organism did not appear in the lake until water temperatures exceeded  $20^{\circ}\text{C}$  and declined with decreasing temperature in late summer or early fall. The correlation coefficient for the relationship ( $r = 0.757$ ) was significant at the 0.05 level. The periodicity appeared also to be correlated roughly with incident solar radiation ( $r = 0.330$ ) and thermal stratification, but neither parameter seemed to be as important as water temperature since the population persisted after a significant decline in both factors.

Concentrations of  $\text{PO}_4$  and  $\text{NO}_3\text{-N}$  during late spring and summer, as previously noted, remained quite low. This was probably a contributory factor to *Spirulina* development since blue-green algae are believed to possess lower half-saturation constants ( $K_s$ ) for nutrient assimilation, particularly  $\text{PO}_4$ , than other groups of algae (Hutchinson, 1966; Shapiro, 1973). *Spirulina* exhibited a low but significant (0.05 level) statistical correlation with  $\text{PO}_4$  ( $r = -0.396$ ) but not with  $\text{NO}_3\text{-N}$ .

*Anabaenopsis raciborskii*. The blue-green alga *Anabaenopsis* is uncommon in the plankton of temperate and subtropical lakes of Europe and North America but is often abundant in tropical lakes (Zafar, 1967; Ruttner, 1952; Lind, 1968).

The seasonal periodicity of *A. raciborskii* in Canyon Lake was closely related to that of *Spirulina* (correlation coefficient was 0.810) and other subdominant blue-green algae, but there was some tendency for *Anabaenopsis* to

appear in the lake slightly later in the summer. Its later appearance may be related to its reported ability to fix atmospheric nitrogen since combined forms of nitrogen were generally lowest in late summer (Olsen, 1975). Like *Spirulina*, *A. raciborskii* exhibited a close relationship to water temperature [Fig. 2(c)] and appeared able to develop significant populations at low nutrient levels. Correlation analysis produced somewhat low but significant (0.05 level)  $r$  values for  $\text{PO}_4$  ( $r = -0.345$ ) and water temperature ( $r = 0.564$ ).

## CONCLUSIONS

The phytoplankton community of Canyon Lake exhibited an annually repeated succession of dominant species. The seasonal appearance or variation in population size of individual species was correlated in varying degrees with one or more of the following parameters: incident solar radiation, water temperature, nutrient concentrations, thermal stratification, and light penetration.

Seasonal succession and variance in population size of dominant species were not correlated with the major element and ion constituents of Canyon Lake ( $\text{Ca}$ ,  $\text{Mg}$ ,  $\text{Na}$ ,  $\text{K}$ ,  $\text{Cl}^-$ ,  $\text{HCO}_3^-$ , and  $\text{SO}_4^{2-}$ ), despite the fact that significant fluctuations in concentration were recorded. Likewise, the concentration of trace elements monitored did not appear to be of major biological importance in Canyon Lake (Olsen, 1975).

The data suggest that water temperature and light were the most important factors affecting periodicity of individual species, but multiple parameters were implicated in all cases. The importance of water temperature was illustrated by the restriction of Cyanophyceae and certain Chlorophyceae to warm-water periods and by the inverse relationship between water temperature increase and centric diatom decrease in the spring. We should emphasize that the effects of light and water temperature could not be satisfactorily isolated in all cases and that caution should be exercised in interpreting data on thermal relations of primary producers in the absence of adequate solar-radiation and light-penetration information.

Implicit in the contention that water temperature is of major importance in aquatic ecosystems of desert regions is the implication that man-induced thermal perturbation (e.g., in the use of desert lakes for condenser cooling) may result in gross alteration in phytoplankton composition and periodicity with subsequent changes at other trophic levels.

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# Influence of Thermal Loading on the Ecology of Intertidal Algae

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## ABSTRACT

Thermal effluents from the Maine Yankee Atomic Power Company (operating intermittently from October 1972 to December 1974) increased water temperatures in the discharge area by 7 to 15°C. Plant operation and the removal of a causeway increased mixing and salinities in Montsweag Bay. Four small red algae immigrated into the area, but no species were lost from the system. Distribution and abundance patterns of the dominant algae, *Ascophyllum nodosum* and *Fucus vesiculosus*, were altered by the thermal discharge. The cover of *F. vesiculosus* decreased, whereas that of *A. nodosum* increased in 1973 but declined significantly in 1974. Reductions in biomass and percent cover were accompanied by changes in the growth dynamics of *A. nodosum*. Growth and survival in the discharge area were enhanced in 1973 but reduced in 1974. Growth was initiated earlier at all sites affected by the warm water. Plants at experimental sites not directly in the discharge channel grew at accelerated rates during the two years, but stressed plants in the discharge produced few or no viable apices in 1974. The net effect has been a compression and reduction of intertidal algae into a narrower and less dense band.

Benthic algae are the dominant life forms on most rocky intertidal shores in the North Atlantic (Lewis, 1964). These shores show a gradient from strongly algal-dominated sites in sheltered regions to algal-barnacle domination in wave-exposed sites (Ballantine, 1961). Numerous studies have shown that the individual members of these intertidal associations can withstand wide ranges of physical perturbations, including desiccation, salinity fluctuations, destructive light intensities, and temperature extremes (Zaneveld, 1969). Intertidal forms in Maine follow these generalizations and survive wide fluctuations in air and water temperatures and salinities.

Our interest in the tolerances of these algae was accentuated by the development of the Maine Yankee Atomic Power Company (MYAPCO). Studies were designed to determine the effects of thermal effluents on the composition, distribution, and abundance of intertidal algae

and on the ecology of the dominant alga, *Ascophyllum nodosum* (L.) Le Jol., which was studied extensively for two reasons. First, because of its dominance, we assumed that its demise could have a significant impact on the structure and integrity of the intertidal community, and, second, because of its abundance and wide distribution on both sides of the North Atlantic and of its sessile and perennial habit, it might be an effective and easily monitored indicator of changing environmental conditions in nearshore marine ecosystems.

## SITE AND ENVIRONMENT

Maine Yankee is located on Bailey Point in Montsweag Bay 6.5 km (4 miles) south of Wiscasset, Me. (Fig. 1). Montsweag Bay has semidiurnal tides, with a mean amplitude of 2.5 m. It is a unique system; tidal waters enter and leave from two directions, and there is a substantial inflow of fresh water from both ends of the bay. The Kennebec River supplies a major input of fresh water and pollution to the southern end of the bay, and the Sheepscot River estuary provides fresh water to the northern end of the bay. Before the removal of the causeway to Westport Island in the fall of 1974, tidal flows from both ends of the bay met north of the power plant. This was an area of still water, which, in the years before the power plant began operation, froze to form a surface ice plug during cold winters. Since the removal of the causeway, tidal flows meet south of Foxbird Island (Fig. 1).

Montsweag Bay and the surrounding river systems are drowned valleys that were modified by Pleistocene glaciation. Intertidal shorelines consist of granitic and basaltic ledges and sediment-covered coves and mud flats. Rockweeds (*A. nodosum* and *Fucus vesiculosus*) dominate the rocky ledge that surround most of the bay. Small green algae (*Enteromorpha* sp. and *Ulothrix flacca*) and diatoms

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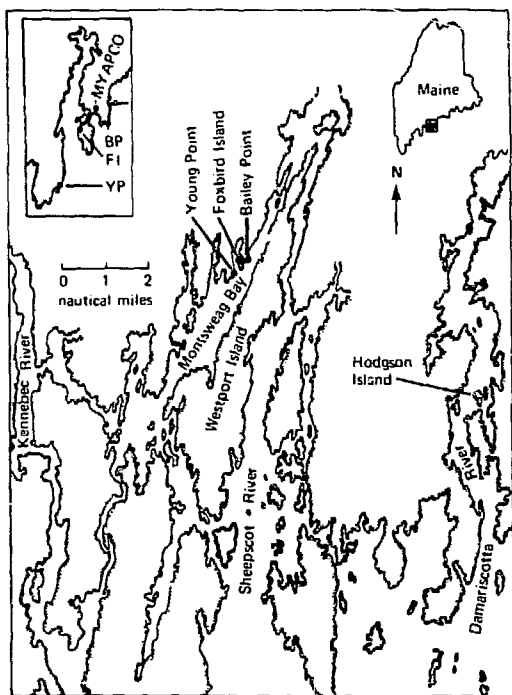


FIG. 1 Location of Maine Yankee Atomic Power Company and the control (Hodgson Island) and experimental (Bailey Point, Young Point, and Foxbird Island) sites.

appear seasonally on exposed rock. Small salt marshes, consisting of *Spartina alterniflora*, *Spartina patens*, and *Salicornia europaea* occur in pockets above the ledges and in most coves.

The lower intertidal area is covered by rockweeds, ulvoids (*Monostroma* and *Ulva* sp.), Irish moss (*Chondrus crispus*), and blue mussels (*Mytilus edulis*). Blue mussels are both epilithic and epiphytic and at times completely cover *A. nodosum*. Seasonally, a filamentous brown alga (*Pylaiella littoralis*) also occurs on *A. nodosum*. Subtidal areas are mostly shallow and sediment covered. At mean low water (MLW), 65% of Montsweag Bay lies under less than 2 m of water (McAlicie, 1972), and, except for scattered patches of eel grass (*Zostera marina*) and a few kelps (*Laminaria saccharina*) at the deeper southern end, the subtidal area is devoid of macroscopic plants. Benthic diatoms occur seasonally on mud flats, however, and small filamentous red or foliose green algae occur in areas where rocky ledges extend into subtidal regions.

The power plant operates between two regions of the bay. Intake is through a channel dredged to the main channel in Montsweag Bay, and discharge occurs on the mud flats of Bailey Cove on the northwest side of Foxbird Island (Fig. 1). Cooling water has a residence time of 3 to 5 min in the plant and undergoes a maximum temperature increase of 15°C. Power output was discontinuous throughout our study.

## METHODS

Three experimental and six control sites were established in Montsweag Bay. Additional controls were established in the Sheepscot River (river control), in the Damariscotta River (regional control), and on the open coast (estuary control). This paper discusses four sites in detail. Three of these sites, Foxbird Island, Young Point, and Bailey Point, located near or in the thermal discharge, are experimental sites. The fourth, Hodgson Island, is one of the regional controls (Fig. 1). Site selection was based on proximity to the discharge and on the accessibility of ledge. This eliminated loose and unconsolidated boulder or cobble beaches and provided for greater physical homogeneity among sites.

Hydrographic samples were collected monthly at high tide, 20 to 30 m seaward of the sites and at 0.15 and 3.0 m below the surface to correspond to intertidal depths. Temperature and salinity readings were made with a Beckman salinometer-thermometer, Model RS5-3, or a Yellow Springs Instrument Model 33 S C T meter. Extinction coefficients (*K*) were determined from secchi disk readings (Holmes, 1970). Continuous surface light measurements were recorded at MYAPCO with a pyrheliometer. Nearshore nitrate levels were examined by the method of Mullin and Riley (1955) for two years (June 1971 to August 1973) to determine its degree of correspondence to the growth cycle of *A. nodosum* and to mid-channel values determined by McAlicie (1972).

Biological measurements on intertidal algae include percent cover, biomass, growth, and mortality of *A. nodosum*. Aerial photographs, with Ektachrome and Ektachrome infrared film, were taken seasonally to detect stress or changes in algal coverage (Vadas and Manzer, 1971). The percent cover of algae, marsh grasses, attached invertebrates, and substrate types was measured seasonally (spring, summer, and fall). Three permanent vertical belt transects, 0.25 m wide and of variable length, depending on slope, were established from the latus line to mean low water or to mud flats subtending the ledge at each site. Transect data were taken in 1.0-m segments and analyzed according to tide height. The sites were divided into two zones, 1.2 and 2.2 m above MLW, based on U. S. Coast and Geodetic Survey Tide Tables.

Algal biomass was determined in the summers of 1971, 1973, and 1974 at Foxbird Island, the site nearest the discharge. Plants and animals were removed with scrapers from 25- by 25- or 25- by 12.5-cm quadrats. In 1971, randomly selected quadrats were taken from two zones stratified according to tide height. In 1973 and 1974, quadrats were taken from two randomly selected, vertical belt transects and were analyzed by zone to correspond to 1971 samples. Individual macroalgae were enumerated, cleaned of large epiphytes, and dried to a constant weight at 70°C.

We made growth and mortality studies of *A. nodosum* by studying two groups of 20 plants in each zone. Basal regions of these plants were identified with numbered

plastic fish tags (Howitt Plastics Company) attached by 12-lb test monofilament or waterproof plastic tape (3M Scotch Brand). Five apical tips per plant were identified with different colored plastic tapes attached by 8-lb test monofilament. Growth data for each zone, therefore, is represented by a minimum of 200 apical tip measurements. All tagging was done in June of each year, except in 1971, when the initial tagging procedure reduced mortality and was modified. Growth was also determined by monthly measurements of 100 apical tips from a minimum of 10 randomly selected plants from zone 2. Such measurements were made bimonthly on a small population within 150 m of the discharge pipes (inner Foxbird Island).

## RESULTS

### Hydrographic Data

Preoperational hydrographic patterns at the study sites are shown in Fig. 2. Surface and bottom salinities at the three experimental sites ranged from 10 to 30‰, commonly averaging 20 to 25‰. Lowest values occurred in April and May, coincident with maximum spring runoff. Highest salinities occurred during periods of minimum runoff, especially in the fall. This pattern, although variable, has not changed appreciably during power-plant operation. Hodgson Island (control) had a higher and less variable salinity regime. Nitrate ( $\text{NO}_3$ ) cycles in Montsweag Bay and the Damariscotta River were typical of inshore and estuarine waters. Values peaked in fall and winter and declined sharply in later winter and early spring. In some cases there was a late summer augmentation.

Radiation and temperature curves were similar but out of phase, especially from winter to spring and in the fall, when temperatures lagged behind changes in light. Radiation doubled in January and February while temperatures were still decreasing. The maximum rates of increase for light and temperature values occurred from January to April and April to June, respectively. Radiation reaching Montsweag Bay ranged from 100 to 600 langley/day. Values at Hodgson Island, 7 miles to the east, were assumed to be similar. Extinction coefficients ( $K$ ), however, varied considerably between Hodgson Island and the experimental sites. At Hodgson Island,  $K$  values were low (0.5) and indicated clear coastal waters; values in Montsweag Bay were higher (0.5 to 1.0). Turbidity increased in Montsweag Bay in 1972 and during power-plant operation, because of sediments resuspended from the discharge channel and of construction. Foxbird Island consistently had the highest extinction coefficients (1.3 to 2.0) because of construction at MYAPCO.

Temperatures at Hodgson Island changed little during these studies (Fig. 2d). Summer and winter temperatures ranged from 12 to 16°C and from 0 to 2°C, respectively. Preoperational temperatures in Montsweag Bay show greater extremes than Hodgson Island. Winter lows in the bay were similar, except when ice formed, but summer values were higher (15 to 20°C). The higher readings were caused by heating of the extensive shallow mud flats and

the flushing characteristics of the bay. As we anticipated, temperatures were altered in the vicinity of the discharge. At Bailey Point there was little evidence of change (Fig. 2c) in our samples taken at high tide, but, during the ebb, a flow of warm water moved north around Foxbird Island and raised temperatures in the Bailey Point area (personal communication, MYAPCO). During power-plant operation, temperatures at Foxbird Island and Young Point increased by 7 to 15°C. The variability in temperature was caused by reduced operating capacity and 1-day to 3-month shut-downs; e.g., in the summer of 1974. Differences in temperature between Foxbird Island and control sites in Montsweag Bay and the approximate level of power-plant operation are given in Fig. 2. Maximum temperatures at experimental sites ranged from 25 to 27°C.

### Species Composition

Thirty-four species of macroscopic algae and related plants were observed in preoperational phases (Vadas, 1971). Seasonal differences in the flora are caused mainly by life-history phenomena. Filamentous forms from the three major phyla (Chlorophyta, Phaeophyta, and Rhodophyta), fleshy green algae, benthic diatoms, and marsh grasses occur seasonally, but fleshy brown and red algae tend to be perennial.

No significant changes in dominance patterns or species composition have resulted from the heated discharge. Small populations of four red algae, *Bangia fuscopurpurea*, *Polysiphonia subtilissima*, *Rhodochorton purpureum*, and *Porphyra umbilicalis*, have colonized the bare ledges along the discharge channel on Foxbird Island, however, to raise the total taxa to 38 species.

### Percent Cover

The percent cover of *A. nodosum* and *F. vesiculosus* is given in Table 1. The percentages, taken seasonally, were converted with an arc sine transformation, averaged, and analyzed with analysis of variance (ANOVA). Detailed seasonal coverage by these and other biota and substrates is given elsewhere (Vadas, 1974). The dominant species at all sites was *A. nodosum*, with coverage ranging from 80 to 95%. Coverage by *F. vesiculosus*, the other species of importance, was more variable and ranged from 0 to 50%. The 4-year averages for *F. vesiculosus* at the four study sites ranged from 2 to 5%. The variability in algal coverage results from physical stress (mainly ice damage), the relatively short life-span of *F. vesiculosus* (3 to 5 years), and differential growth and mortality in *A. nodosum*.

Generally, algal coverage was highest in zone 2 (Table 1). The cover of *A. nodosum* at all sites, except Foxbird Island, remained constant or increased during power-plant operation. At the Foxbird Island site, changes were not apparent in the first year of operation, but effects were evident at this time at the northwest point of the island (inner Foxbird Island). By the following spring (1974), coverage by *A. nodosum* had decreased by 40%. *Fucus vesiculosus* began declining in zone 2 in 1972 and in

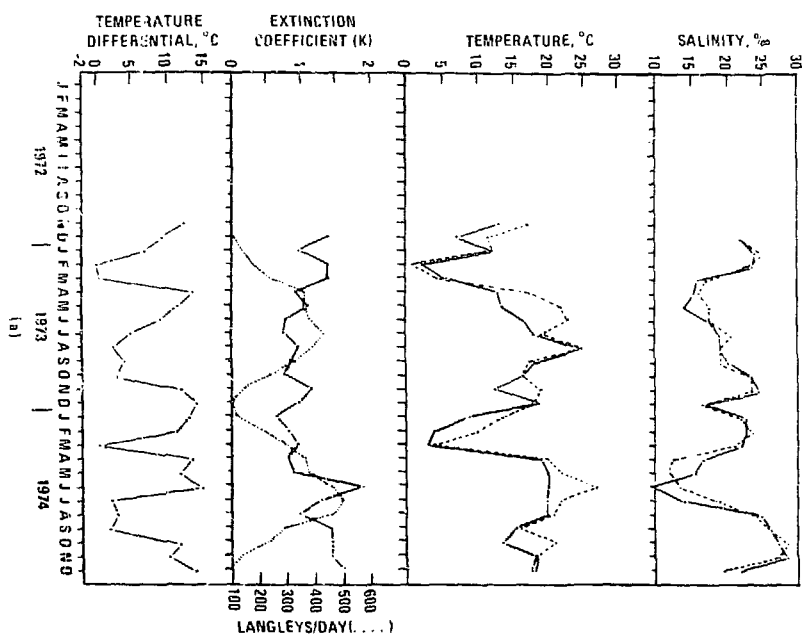
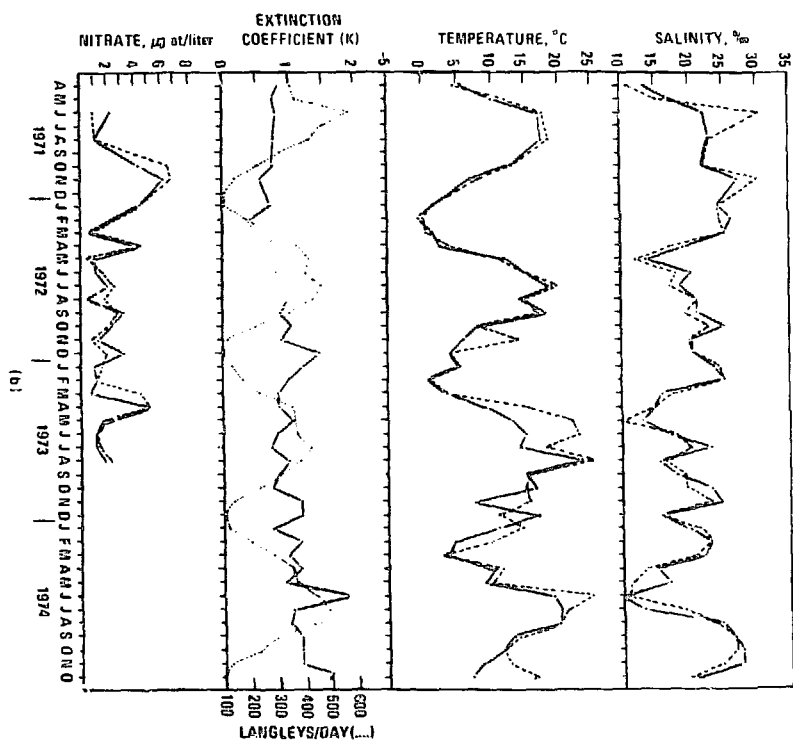


FIG. 2 Hydrographic data for control and experimental sites before and during thermal discharge.  
(a) Foxbird Island. (b) Young Point. (c) Bailey Point. (d) Hodgson Island. —, surface (0.15 m). - - -, bot-  
tom (3 m).



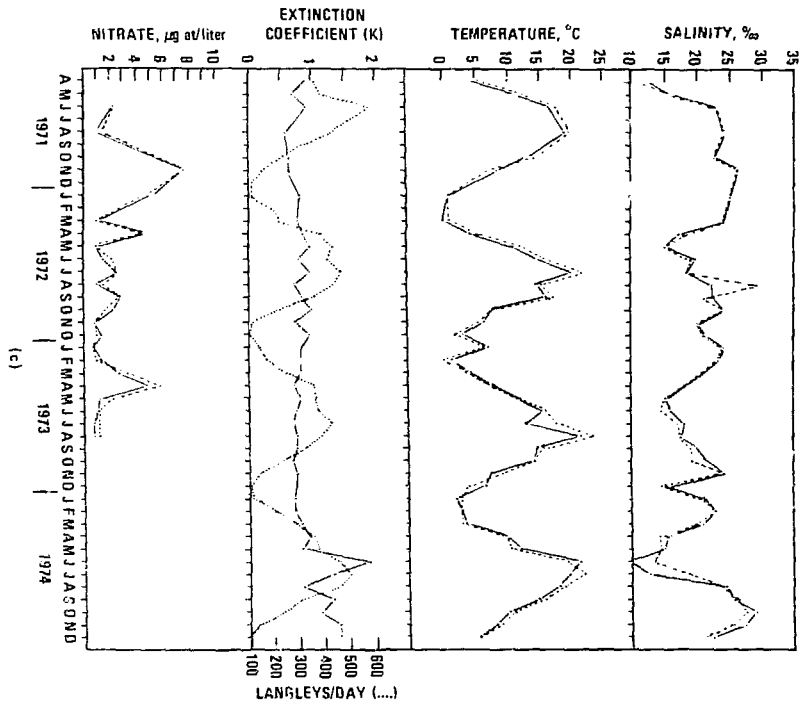


FIG. 2 (Continued)

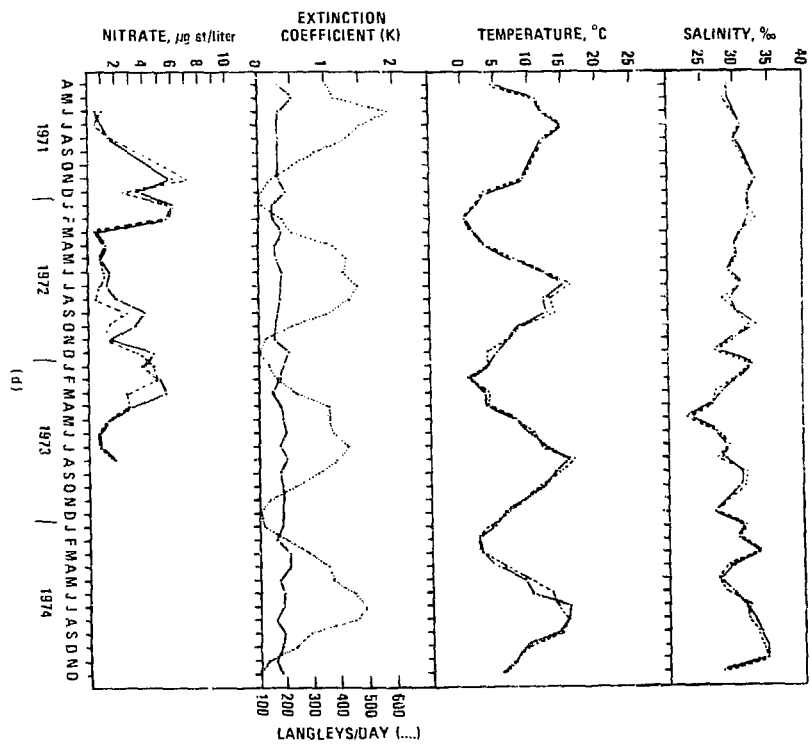


TABLE 1 Percent Cover of Dominant Algae, *Ascophyllum nodosum* and *Fucus vesiculosus*, Before and During Thermal Discharge at Control and Experimental Sites \*

| Zone 1                        |    |               |   | Zone 2 |    |               |   |
|-------------------------------|----|---------------|---|--------|----|---------------|---|
| Year                          | N† | $\bar{x}$     | s | Year   | N† | $\bar{x}$     | s |
| Hodgson Island (Control)      |    |               |   |        |    |               |   |
| 1971                          | 12 | 81.74 ± 1.65  |   | 1971   | 14 | 72.85 ± 3.91  |   |
| 1972                          | 9  | 61.33 ± 15.10 |   | 1972   | 9  | 52.77 ± 11.67 |   |
| 1973                          | 18 | 74.17 ± 7.79  |   | 1973   | 18 | 83.85 ± 3.83  |   |
| 1974                          | 27 | 74.99 ± 4.84  |   | 1974   | 27 | 85.27 ± 2.67  |   |
| Bailey Point (Experimental)   |    |               |   |        |    |               |   |
| 1971                          | 12 | 76.25 ± 4.89  |   | 1971   | 12 | 67.15 ± 6.23  |   |
| 1972                          | 16 | 71.82 ± 9.38  |   | 1972   | 17 | 73.37 ± 6.85  |   |
| 1973                          | 23 | 81.76 ± 2.69  |   | 1973   | 24 | 79.51 ± 3.76  |   |
| 1974                          | 21 | 75.74 ± 6.77  |   | 1974   | 21 | 78.16 ± 5.53  |   |
| Young Point (Experimental)    |    |               |   |        |    |               |   |
| 1971                          | 12 | 85.53 ± 1.26  |   | 1971   | 12 | 81.48 ± 5.21  |   |
| 1972                          | 14 | 77.81 ± 7.75  |   | 1972   | 16 | 78.29 ± 4.99  |   |
| 1973                          | 24 | 74.85 ± 7.72  |   | 1973   | 21 | 81.81 ± 3.75  |   |
| 1974                          | 21 | 73.17 ± 8.05  |   | 1974   | 27 | 78.20 ± 4.70  |   |
| Foxbird Island (Experimental) |    |               |   |        |    |               |   |
| 1971                          | 12 | 70.89 ± 4.25  |   | 1971   | 12 | 76.16 ± 5.12  |   |
| 1972                          | 15 | 74.55 ± 7.31  |   | 1972   | 12 | 64.37 ± 11.51 |   |
| 1973                          | 26 | 79.34 ± 3.51  |   | 1973   | 24 | 71.73 ± 6.14  |   |
| 1974                          | 27 | 53.26 ± 4.18  |   | 1974   | 24 | 48.48 ± 6.01  |   |

\*Original data were transformed with an arc sine function and analyzed with ANOVA.

†N is number of 0.25-m<sup>2</sup> quadrats.

zone 1 between November 1973 and April 1974 at Foxbird Island. Since that time it has not exceeded 2%. Similar reductions in algal coverage occurred at Young and Bailey points. An ANOVA between years at each site showed that decreases in both intertidal zones at Foxbird Island in 1974 and increases in zone 2 at Hodgson Island in 1973 and 1974 were significant ( $P \leq 0.05$ , Table 1).

Commensurate with the decline of algae in the discharge area was the increase in the amount of exposed ledge and mud. On some exposed ledges, sporelings of *A. nodosum* and *F. vesiculosus* colonized the mid- to low-intertidal regions during the fall of 1974. Algal coverage at Young and Bailey points increased slightly because of recruitment by *A. nodosum*.

## BIOMASS

Algal biomass in zones 1 and 2 at Foxbird Island for 1971, 1973, and 1974 are given in Table 2. Variation between quadrats was high (high variance-to-mean ratio) and indicated clumped distributions. Work with nested quadrats at other sites showed this to be a function of quadrat size (unpublished data). Such work was not done at Foxbird Island because of the destructive nature of this kind of sampling and the already precarious nature of the algal population. Standing-stock estimates for the site averaged 2.22 kg/m<sup>2</sup> before the power plant began operating. Algal biomass remained constant during the first year

of operation but decreased significantly (76%) during the second year.

## Growth Studies on *Ascophyllum nodosum*

Growth in *A. nodosum* follows a sigmoidal curve, beginning in March, becoming logarithmic during May and June, decelerating in summer and fall, and leveling off throughout the winter. Our sampling periods and data presentation (Fig. 3) reflect these basic features of growth. Data are given for zone 2, the optimal intertidal region for growth (Vadas, 1972). Emphasis is placed on growth differences in June, before plant shutdowns. The time and duration of shutdowns varied each summer and added to the variability of subsequent measurements. The mean lengths of apical tips for each period throughout the growing season were least variable at Hodgson Island. The mean length of apical tips in mid-June (1971–1974) ranged from 15 to 21 mm; 200 tips per year were measured. There is high overlap (95% confidence intervals) among the means for the four years, but the extreme values (15 and 21 mm) are different (Student's t-test,  $P \leq 0.05$ ). Another feature of comparative importance is the uniformity of growth based on relative lengths in June (Fig. 3).

In general, growth at Montsweag Bay sites was higher and more variable before power-plant operation. Mean tip lengths in June, August, and November were 20 to 40% longer than at Hodgson Island. Some of the variability was caused by periodic turbidity, which accounted for the low value at Foxbird Island in June 1972, when 15 cm (7500 m<sup>3</sup>) of sediment was removed from Bailey Cove (Takaes and Schnitker, 1973). The higher growth rates from June to November are caused by naturally occurring higher spring and summer temperatures.

The thermal discharge from MYAPCO affected the growth of *A. nodosum* at the three experimental sites. Apical tips, when produced, were initiated earlier and grew faster during power-plant operation. Growth increased by 50 to 100% at all sites except Foxbird Island. At Young and Bailey points, differences in tip lengths in June before and during power-plant operation were significant ( $P \leq 0.05$ ).

The results for Foxbird Island need additional explanation. Growth in June in 1972 and 1973 was reduced significantly over the previous preoperational year by high turbidity. Tip lengths at the other experimental sites were also affected at that time, but to a lesser extent. During the first winter and spring of power-plant operation, vegetation at Foxbird Island grew faster than at the other sites. Apical tips initiated during the spring of 1973 were significantly larger ( $P \leq 0.05$ ) than in 1971 and 1972. By winter, however, plants were suffering from thermal stress. During the second spring of power-plant operation, apical tips initiated the previous year grew another 20% in length at most sites in Montsweag Bay, except Foxbird Island (Fig. 3). The initiation of growth was inhibited in 1974, and not enough new apical tips were produced to permit tagging in June.

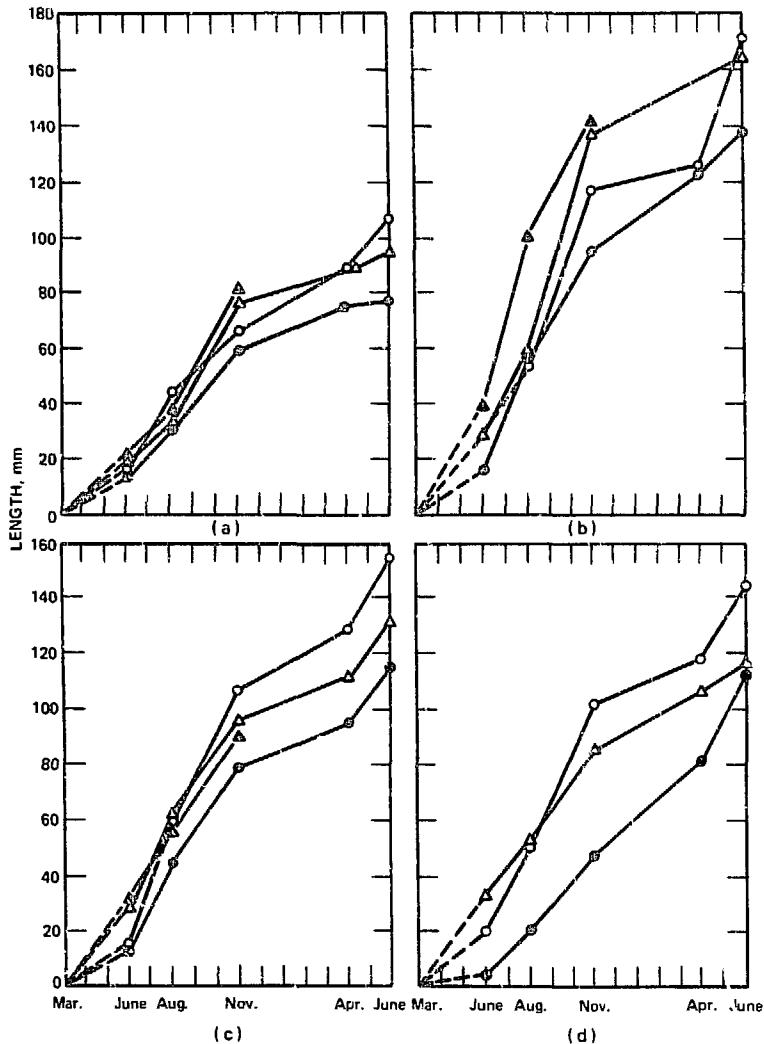


FIG. 3 Growth of *Ascophyllum nodosum* before and during thermal discharge at control [Hodgson Island (a)] and experimental [Bailey Point (b), Young Point (c), Foxbird Island (d)] sites. O, 1971–1972. □, 1972–1973. △, 1973–1974. ▲, 1974–1975.

### Mortality Studies on *Ascophyllum nodosum*

Mortality was measured concurrently with growth. Surviving plants and apical regions were expressed as a percent of those initially tagged at each of two tidal levels (Figs. 4 and 5). Mortality was defined as the loss of the upright portion of the alga. Plant survival was highest at Hodgson Island (control). Within sites, plants usually had higher survival rates in zone 2 than in zone 1, which was more prone to abrasion from debris and ice. The low yearly mortality at Hodgson Island (7 to 25%) also provided a check on tag losses before and during operation. At Bailey and Young points plant mortality increased significantly (30 to 50%) in both zones following the first full year of

plant operation. Survival at Foxbird Island, especially in zone 1, was variable but was relatively low in preoperational phases because of damage caused by ice movement from Bailey Cove. The subsequent increase in survival the first winter (1973) of plant operation paralleled the results at Bailey and Young points and indicates that the heated discharge ameliorated mortality from ice abrasion. Mortality increased significantly the second year (1973–1974), however, and in 1974–1975 had increased considerably before the onset of winter.

Tip survival was variable, especially at Young Point, but followed a pattern similar to that of whole plants. At the Hodgson Island site, which provided the most benign

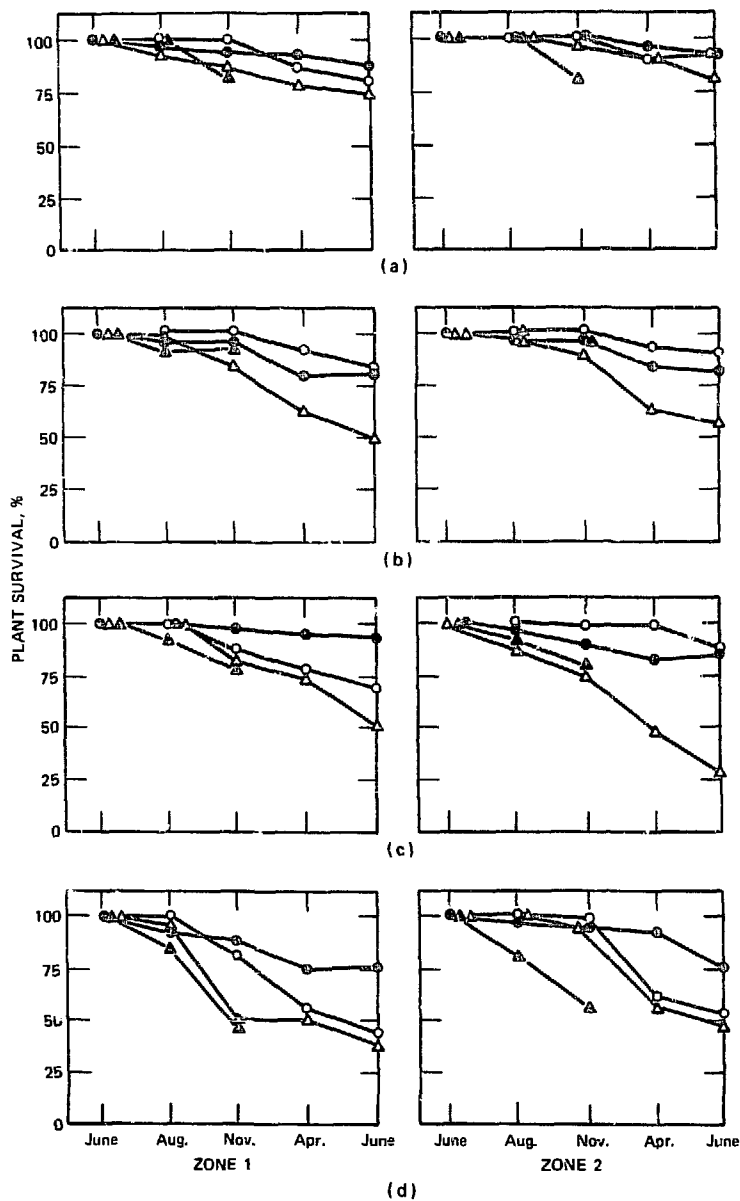


FIG. 4 Thallus (plant) survival, as percent of plants initially tagged, of *Ascophyllum nodosum* before and during thermal discharge at control [Hodgson Island (a)] and experimental [Bailey Point (b), Young Point (c), Foxbird Island (d)] sites. ○, 1971-1972. □, 1972-1973. △, 1973-1974. ▲, 1974-1975.

conditions, tip survival ranged from 40 to 65% after 1 year. Survival at the three experimental sites was lower and more variable than at Hodgson Island but not significantly different, except at Young Point, where survival was low in both zones in 1973-1974, and at Foxbird Island, where there were no apical tips to mark in 1974.

## DISCUSSION

The discontinuous discharge of heated effluents produced variable hydrographic changes in Montsweag Bay. The operation of the plant not only raised the water temperature but also increased salinity and turbidity in the discharge area. Maximum temperatures in the discharge



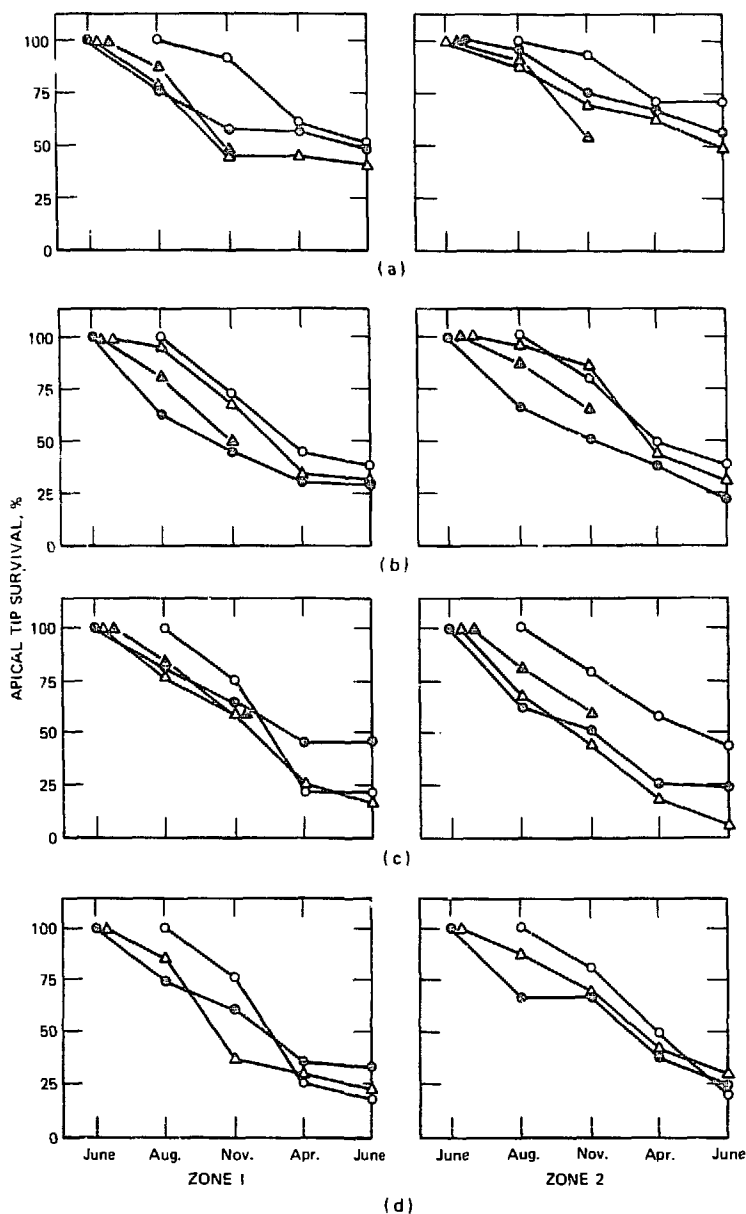


FIG. 5 Apical tip survival, as percent of plants initially tagged, of *Ascophyllum nodosum* before and during thermal discharge at control [Hodgson Island (a)] and experimental [Bailey Point (b), Young Point (c), Foxbird Island (d)] sites. ○, 1971-1972. □, 1972-1973. △, 1973-1974. ▲, 1974-1975.

channel were 27°C, or 8 to 10°C above preoperational summer highs. Because it has a limited mixing zone, MYAPCO was forced to reduce or curtail operations during the summers of 1973 and 1974. The discontinuous operation of the plant may have introduced complexities and effects not readily predictable or analyzable.

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The effects of the heated discharge were first observed at inner Foxbird Island in the spring of 1973, when apical tips of *A. nodosum* were not initiated. The site was abandoned that summer when MYAPCO built a boom to retain foam and enclosed this population. Plants at Foxbird Island (located 150 m from the inner population) and at

Young Point showed significant increases in growth during the same period (Fig. 3). Growth in all cases was initiated earlier at the three experimental sites. Normally, growth in Montsweag Bay was initiated sooner than at Hodgson Island, except in 1974–1975, when populations in both regions began growth at the same time. This was caused by the earlier occurrence of warm waters in 1974 (Fig. 2c and 2d).

These data are in good accord with short-term observations on the importance of temperature and light to intertidal macroalgae (Kanwisher, 1966). Kanwisher observed photosynthesis-to-respiration (P/R) ratios of 20 : 1 in these algae, doubling times of 120 hr for *A. nodosum* in saturated light ( $\sim 3000$  ft-c), and maximum photosynthetic rates that varied with environmental temperature. Since respiration was only 5% of photosynthesis, net photosynthesis would be greater at higher temperatures and longer day lengths. Our long-term field growth rates ranged from 2 to 3% per day for spring and summer; this would yield doubling times of 1 month. Growth rates calculated from monthly measurements ranged from 4 to 7% per day and gave doubling times of 2 weeks, which are near Kanwisher's maximum values.

The reasons for increased growth of plants in the thermal discharge area, therefore, are more interpretable. Increasing temperatures brought the plants nearer their maximum net photosynthetic capacity. Subsequent growth throughout the year at warm-water sites was more normal, as indicated by the similarities in growth rates (see curves in Fig. 3). This occurred because of MYAPCO's summer shutdowns and physiological processes controlling the balance between vegetative and reproductive growth. Enlargement of the receptacles occurs primarily from fall to spring (David, 1943), during periods of minimum vegetative growth. The absence of growth during the second spring of 1-year-old tips and of new apices at Foxbird Island suggests that the increased temperatures were altering P/R ratios rather than injuring apical cells directly. We are currently investigating this possibility with anatomical studies.

Several factors contribute to the natural mortality of *A. nodosum* in Maine's estuaries, with abrasion from ice being most important. With the onset of the thermal discharge, plant survival increased markedly throughout the winter of 1973. Commensurate with the spread of warm water was the absence of ice flows and intertidal ice. Thus the initial effect of the discharge brought about an amelioration of plant mortality. Increased survival of *A. nodosum*, at least in the initial year of plant operation, also may have been due to metabolic adaptation by the alga. Kanwisher (1966) observed respiratory  $Q_{10}$  values in *A. nodosum* of 1.5 and 2.0 for summer and winter, respectively. If seasonal adaptation occurs in Maine populations, it is possible that plants in the discharge area maintained lower  $Q_{10}$  values throughout winter, reduced their respiration rate, and maintained more favorable P/R ratios. Yokohama (1972) has also shown seasonal adaptation in temperature-respiration relationships in benthic algae.

The cause of the subsequent increase in mortality and the lack of growth in the following year is related to the thermal effluent, but the actual reasons are not clear. If Kanwisher's results are applicable in Maine, and given the fact that water temperatures at Foxbird Island have not exceeded  $27^{\circ}\text{C}$ , the growth and survival of *A. nodosum* should have increased rather than decreased. Another possibility, however, is that plants in the discharge area never adapted the second year because of the freezing air temperatures and maintained abnormally high  $Q_{10}$  values throughout the year. Similarly, the rapid demise of *F. vesiculosus* may have been caused by its inability to adapt climatically. Kanwisher (1966) showed that heat injury in *F. vesiculosus* occurred above  $30^{\circ}\text{C}$  in summer but between  $20$  and  $25^{\circ}\text{C}$  in winter. In addition, the rapid and extreme diurnal changes in winter temperatures (from  $-10^{\circ}\text{C}$ , during exposure, to  $18.5^{\circ}\text{C}$ , during submergence) may have weakened the thalli of both species. Finally, the observation that *A. nodosum* requires temperatures below  $10^{\circ}\text{C}$  for reproduction and mean maximums below  $22^{\circ}\text{C}$  for survival (Setchell, 1920) suggests that the deterioration of these populations was inevitable. Given this fact and the winter temperature extremes noted above, it is perhaps surprising that these algae survived as long as they did.

Although the floral compositions of other marine ecosystems were altered extensively by thermal discharge (North, 1969; Wood and Zieman, 1969; Thorhaug, 1974b) and a truncated survival curve was demonstrated for tropical marine algae in thermal discharges in Florida (Thorhaug, 1974a), we have not observed a reduction in the number of intertidal species. Instead, a few small forms colonized ledges in the discharge area. Of these, only *P. subtilissim* was a warm water form (Taylor, 1957). All the other immigrants occur on the open coast of Maine, and they are more likely to have invaded these areas because of the increase in salinity and mixing brought about by the removal of the causeway and by the operation of MYAPCO rather than because of the presence of warm water.

In summary, distribution (zonation) and abundance (percent cover and biomass) patterns were altered by the discharge. The events characterizing these changes, summarized in Fig. 6, involve, first, the demise of *F. vesiculosus* from the highest and lowest intertidal zones at Foxbird Island; second, the slight expansion of *A. nodosum* during the first year of plant operation into areas previously occupied by *F. vesiculosus*; third, the compression of *A. nodosum* and understory algae during the second and third years of operation into a narrower and less dense band, accompanied by a reduction in biomass; and, fourth, colonization of bare rock by small populations of red algae.

This study shows that the growth of *A. nodosum* can be related to causal factors in the environment. Because of its widespread distribution, abundance, and consistency in growth and phenology in Maine and in the North Atlantic in general (MacFarlane, 1932; Printz, 1956), *A. nodosum* may be a valuable indicator of environmental change caused by thermal effluents.

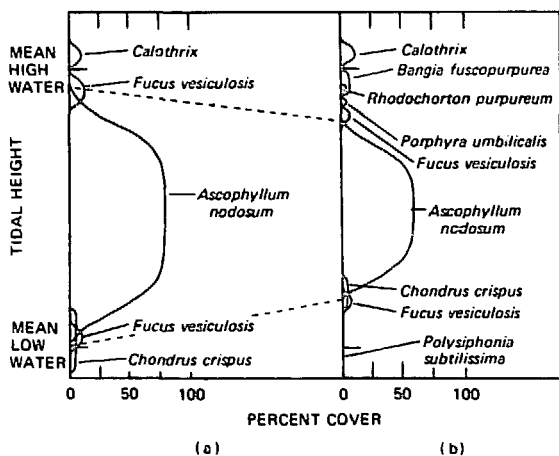


FIG. 6 Graphic illustration of the effects of a thermal discharge on benthic marine algae at Foxbird Island. (a) Before thermal discharge. (b) After 2-year exposure to thermal discharge. Dashed lines represent compression of vertical distribution of *A. nodosum*.

## ACKNOWLEDGMENTS

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Contribution 85, Ira C. Darling Center, Department of Oceanography, University of Maine, Walpole.

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# Relationship Between Thermal Loading and Parasitism in the Mosquitofish

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J. Whitfield Gibbons  
Gerald W. Esch

## ABSTRACT

The relationship between thermal loading and parasitism was examined in 980 mosquitofish, *Gambusia affinis*, taken from areas of varying thermal conditions at the Savannah River Plant (SRP) near Aiken, S. C. Collections were made at 2-week intervals from May to August 1974 and again from January to February 1975. The metacercariae of two strigeid trematodes, *Ornithodiplostomum ptychocheilus* and *Diplostomum scheuringi* were the only species of parasites recovered. *Ornithodiplostomum ptychocheilus* was always found encysted in the brain and eyes, whereas *D. scheuringi* was restricted exclusively to the body cavity. The density of the body-cavity parasite was highest in fish from areas of ambient temperatures and declined in fish from areas with higher water temperatures. The density of the brain parasite, on the other hand, was higher in fish from areas directly receiving thermal effluent than in fish from ambient-temperature areas. The body-cavity parasite was absent from mosquitofish from Pond C, but the infection percentages were relatively consistent in fish from other areas in the Par Pond system regardless of water temperature. The brain metacercariae were recovered from 95% of fish from Pond C. There were higher infection percentages of brain parasites in fish from thermally altered areas of Par Pond than in those from other parts of the Par Pond system. Three different hypotheses can be formulated to account for the relationship between thermal loading and the observed abundance and distribution patterns for these two species of parasites: (1) a direct differential influence of temperature on the physiological tolerance of the various life-cycle stages of the parasite; (2) a direct differential impact of temperature on the distribution of the molluscs that serve as the first intermediate hosts; and (3) an effect of temperature on the abundance and distribution of the potential definitive hosts within the study area.

species populations capable of surviving in thermally altered habitats. One species that has shown this ability is the mosquitofish, *Gambusia affinis*.

Several life-history studies have been conducted on *G. affinis* (Kuntz, 1913; Hildebrand, 1917; Krumholz, 1948; Rosen and Bailey, 1963). Mosquitofish are a surface-living fish found primarily in the littoral zone. Most individuals have a life expectancy of only 1 to 2 years. Size ranges from 8 mm for the young to 50 mm for the larger females. Since mosquitofish produce several broods a year, with a gestation period of about 1 month, they can produce large local populations over short periods of time.

Because *G. affinis* live in shallow water, they are exposed to higher temperatures than most fish would normally experience. Not only are they able to survive in areas directly receiving thermal effluent but in some areas they may be the only fish species present. Thus mosquitofish living in areas of elevated water temperature may be under heat stress near their upper thermal limits (Carlander, 1969; Hart, 1952).

Very little is known about the effect of thermal loading on parasite populations in aquatic vertebrates. Because of their high thermal tolerance and the presence of large local populations in heated areas, mosquitofish are an ideal organism for such studies. The objective of our study was to compare the parasite fauna in *G. affinis* taken from a variety of thermally stressed aquatic habitats.

## MATERIALS, METHODS, AND STUDY AREAS

A total of 980 *G. affinis* were sampled at 2-week intervals from 10 locations at the Savannah River Plant (SRP) near Aiken, S. C., from May to August 1974 and in January and February 1975. Of the 10 locations, nine were in Par Pond, Pond B, or Pond C (Fig. 1 and Table 1), and one was an abandoned farm pond not shown on the map. Collections were made from two sites on Pond C, six sites on Par Pond proper, and one on Pond B.

Increases in man's use of energy have resulted in the discharge of an increased amount of waste heat into a variety of aquatic habitats. One approach to understanding the impact of thermal loading on ecosystems is to study

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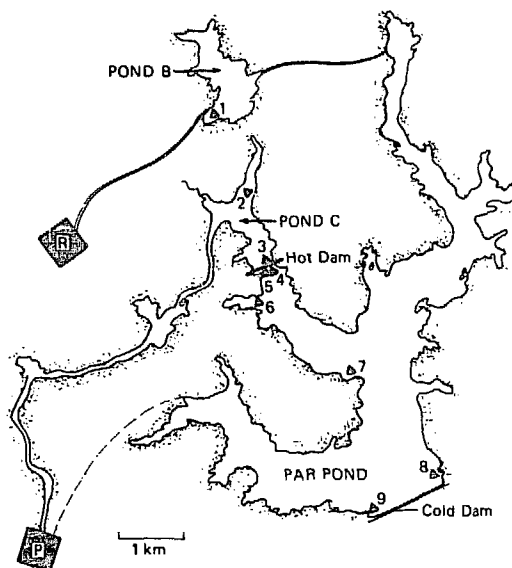


FIG. 1 Map of Par Pond system showing sampling sites (1 to 9).

TABLE 1 Mean Number ( $\pm$ Standard Deviation) of Brain and Body-Cavity Metacercariae in *Gambusia affinis* from Heated and Ambient Areas of the Par Pond System

| Temperature*<br>and sampling<br>station | Summer          |                 | Winter          |                |
|-----------------------------------------|-----------------|-----------------|-----------------|----------------|
|                                         | Body cavity†    | Brain           | Body cavity     | Brain          |
| 40°C                                    |                 |                 |                 |                |
| Pond C (2)                              | 0.0 $\pm$ 0.0   | 29.5 $\pm$ 20.2 | 0.0 $\pm$ 0.0   | 14.2 $\pm$ 7.1 |
| Pond C (3)                              | 0.0 $\pm$ 0.0   | 13.1 $\pm$ 10.2 | 0.0 $\pm$ 0.0   | 9.8 $\pm$ 5.5  |
| 35°C                                    |                 |                 |                 |                |
| Par Pond (4)                            | 16.2 $\pm$ 8.7  | 7.5 $\pm$ 6.9   | 5.4 $\pm$ 4.0   | 4.9 $\pm$ 4.5  |
| Par Pond (5)                            | 8.3 $\pm$ 8.3   | 8.6 $\pm$ 7.0   | 4.2 $\pm$ 4.0   | 2.1 $\pm$ 2.2  |
| Par Pond (6)‡                           | 78.7 $\pm$ 70.6 | 5.3 $\pm$ 4.8   | 19.4 $\pm$ 24.4 | 1.8 $\pm$ 1.6  |
| Slightly<br>above ambient               |                 |                 |                 |                |
| Par Pond (7)                            | 25.5 $\pm$ 50.1 | 8.1 $\pm$ 10.3  | 16.1 $\pm$ 20.0 | 1.0 $\pm$ 1.4  |
| Ambient                                 |                 |                 |                 |                |
| Par Pond (8)                            | 15.4 $\pm$ 19.3 | 2.2 $\pm$ 2.2   | 10.1 $\pm$ 10.3 | 1.2 $\pm$ 1.8  |
| Par Pond (9)                            | 15.0 $\pm$ 18.7 | 2.7 $\pm$ 2.6   | 7.8 $\pm$ 7.2   | 0.7 $\pm$ 1.1  |
| Pond B (1)                              | 2.6 $\pm$ 2.9   | 6.5 $\pm$ 6.6   | 3.8 $\pm$ 6.7   | 6.4 $\pm$ 6.5  |

\*Elevated temperatures are mean maximums for summer months; mean maximum temperatures for winter would be approximately 8 to 10°C less.

†All means are based on 20 *Gambusia* (10 females and 10 males) from each site.

‡Temperature for station 6 approximates that for stations 4 and 5.

Dick's Pond, a 1-ha abandoned farm pond, has never received any form of thermal input. Pond C is a 67-ha precooling reservoir. Water temperatures in the center often rise to 50°C in the summer and seldom fall below 30°C in the winter. Par Pond, an 1120-ha cooling reservoir, receives heated effluent from Pond C. The thermal influence in Par

Pond extends several hundred meters beyond the point of entry at the Hot Dam and gradually declines to ambient in the more distant parts of the reservoir (Holland et al., 1974). Pond B, an 81-ha impoundment undergoing post-thermal recovery, is at ambient temperature year round. More extensive descriptions of the Par Pond system and

Dick's Pond are given by Parker, Hirshfield, and Gibbons (1973) and Benke and Benke (1975). Fish were collected at 2-week intervals during both sampling periods using a 2-m seine (4-mm mesh) or a dip net (8 openings/cm<sup>2</sup>). The total length of the fish was measured, and the sex was determined by the presence or absence of a specialized anal fin, the gonopodium. The fish were carefully dissected and examined for internal parasites. After recording the number of each species of parasite and the site of infection, we then fixed, stained, and identified the parasites.

## RESULTS

Mosquitofish harbored only two species of parasites, both larval trematodes of the family Strigeidae. The two species are metacercaria of *Diplostomum scheuringi*, which is found free in the body cavity, and *Ornithodiplostomum ptychocheilus*, which is found encysted in the brain and eyes. The sites of infection for the parasites did not overlap.

Before evaluating the relationship between parasitism and thermal loading, we had to determine if the length of the fish influenced the intensity of infection for each parasite species. In each sampling period there was a highly significant relationship ( $P < 0.0001$ ) between body length and the mean number of both brain and body-cavity parasites. If length is adjusted in an analysis of covariance, however, the results obtained are almost identical to the results of a simple analysis of variance without considering length as a covariate. Thus it is necessary to consider only the results of the simple analysis of variance in evaluating the relationship between thermal loading and parasitism in mosquitofish.

The impact of thermal loading on the abundance and distribution patterns for the two species of parasites cannot be readily demonstrated when the infection percentages in different localities are compared. Although body-cavity parasites were not found in *G. affinis* from Pond C, the infection percentages were relatively constant (80 to 100%) throughout the remainder of the Par Pond system and certainly did not suggest any kind of correlation between temperature and infection during either summer or winter. The brain parasite, *O. ptychocheilus*, was found in fish taken in Pond C, as well as in all other sites sampled within the Par Pond system. The infection percentages of the brain parasite were slightly higher in thermally altered than in ambient areas of Par Pond. This trend was most apparent during winter when 87% of the fish from Coleman's Cove were infected (Fig. 1, station 4, a heated area) and only 45% of the fish from the west side of the Cold Dam (station 8, an ambient area) were infected.

The mean number of brain and body-cavity parasites was significantly related ( $P < 0.0001$ ) to location, and, thus, thermal condition. In the summer sample, the mean number of body-cavity parasites in *G. affinis* was found to be highest in areas with ambient water temperature and to decrease in density in thermally influenced areas (Table 1). On the other hand, the mean number of brain parasites was found to be highest in fish from areas of high thermal

loading, whereas the density decreased progressively in *G. affinis* taken from areas of decreasing water temperature. Even though the mean number of each parasite species was generally lower in winter than in summer, the same basic patterns were observed in both seasons.

The relationship that exists between thermal loading and the two parasite species can be seen more clearly when each species is considered as a percentage of the collective parasite infrapopulation (an infrapopulation is defined as all individuals of a single parasite species within an individual host). In the summer sample (Fig. 2), the percentage of the brain parasite was highest in fish from areas of high thermal

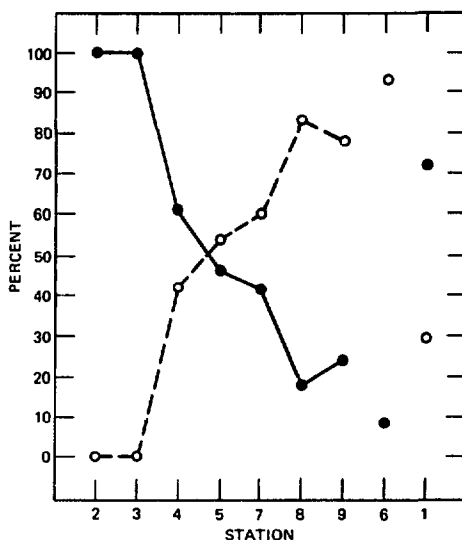


FIG. 2. Percentage of brain (●—●) and body-cavity (○—○) metacercaria within the collective parasite infrapopulation in summer. Water temperature was consistently elevated at stations 2 to 6.

loading and lowest in fish from areas with lower temperature. The percentage of the body-cavity parasites was highest in fish from areas with ambient temperature and decreased in fish from thermally altered areas. Similar trends existed for the percentage of each parasite species in the winter sample (Fig. 3).

Despite the overall consistency in infection patterns, fish at 3 of the 10 locations had levels of parasitism which departed from the observed patterns and required closer examination. Mosquitofish from station 6, near the boat dock in an area directly influenced by thermal loading, had much higher body-cavity-parasite densities than those from any other sample site (Table 1). In both summer and winter samples, the mean number of body-cavity parasites in mosquitofish from the boat dock area was several times greater than the mean number in those from the Cold Dam (stations 8 and 9). However, the mean number of brain parasites in mosquitofish from the boat dock area was comparable to levels normally observed at the Cold Dam.

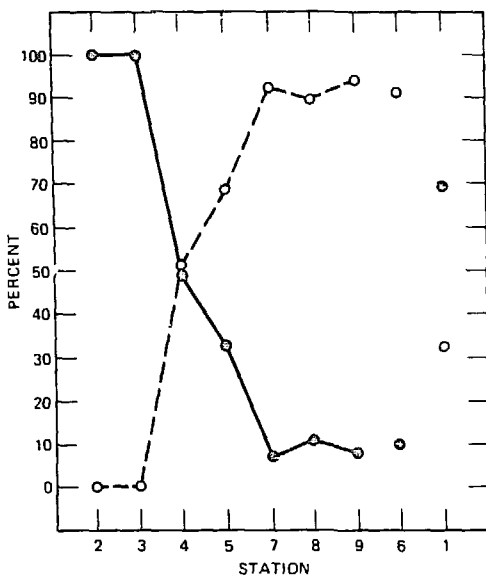


FIG. 3 Percentage of brain (●—●) and body-cavity (○-○) metacercaria within the collective parasite infrapopulation in winter. Water temperature was consistently elevated at stations 2 to 6.

The parasite fauna in Pond B (station 1) mosquitofish also did not follow the trends seen in most of the locations in Par Pond and Pond C. The densities of each parasite species in *G. affinis* from Pond B represented an intermediate position between that observed in areas receiving thermal effluent and ambient temperature areas. Mosquitofish in Dick's Pond were completely devoid of parasites.

## DISCUSSION

Our results indicate a differential distribution pattern for the brain and body-cavity metacercaria in the mosquitofish taken from the Par Pond reservoir system. Furthermore, the observed distribution pattern for each species of parasite appears to be related to water temperature, except at stations 6 and 1 and Dick's Pond. We should note that in largemouth bass, *Micropterus salmoides*, the infection percentages of the acanthocephalan, *Neoechinorhynchus cylindricus*, were consistent between thermal and ambient sampling stations even though the infrapopulation densities were consistently higher in thermally altered areas of Par Pond (Eure and Esch, 1974). This pattern is consistent with both species of parasites in *G. affinis*, but the densities of body-cavity metacercariae were highest in fish from ambient areas.

At the present time, three hypotheses have been formulated to explain the relationship between thermal loading and parasitism in *G. affinis*. Briefly, these are: (1) a direct differential influence of temperature on the physiological tolerance of the parasites; (2) a direct differential impact of temperature on the distribution of the molluscs that serve as the first intermediate hosts; and (3) an effect

of temperature on the abundance and distribution of the potential definitive hosts within the study area.

The first hypothesis is concerned with the effects of temperature on the physiological tolerances of the various larval forms of the parasite. For example, Vernberg (1961) reported that cercaria of *Zoogonus rubellus*, a parasite of marine fishes, were killed by short exposures to temperatures above 36°C. Elevated water temperature could also influence egg viability, miracidial penetration into the first intermediate hosts, and maturation rates of the developing intramolluscan larval stages, could increase the rate of cercarial release from the infected snails, and could possibly alter the behavioral patterns of the cercaria. Each of these phenomena, either alone or in conjunction with any of the others, could be responsible for producing the observed parasite distribution patterns.

The second hypothesis is related to the impact of temperature on the distribution of the molluscan first intermediate hosts and/or on a possible differential effect on the infected snails within the Par Pond system. The snails that serve as first intermediate hosts for *D. scheuringi* and *O. ptychocheilus* are *Helisoma* sp. and *Physa* sp., respectively. *Helisoma trivolvis* is the only species found at the Hot Dam, but *H. trivolvis* and *H. anceps* are both found in the area around the Cold Dam (Tilly, 1974). In conjunction with the differential distribution of snails in Par Pond, there is also a seasonal difference in the densities of snails found in the heated areas, as compared with the ambient-temperature areas. In the summer months the density of *H. trivolvis* is higher at the Cold Dam locations than at the Hot Dam, where water temperature (consistently >35°C) may be close to the thermal tolerances of the snails. On the other hand, although the ambient-temperature locations would present conditions acceptable for normal survivability of the snails in summer, the opposite might be true for winter. Thus low water temperature in the unheated areas might cause a reduction in the density of the snails in winter, but in the heated areas, densities would be higher since the water temperatures there would be similar to those in ambient-temperature locations in the summer. The density of snails may also influence the incidence of their infection (Ewers, 1964; Ollerenshaw, 1958; Wright, 1960) and, in turn, the degree of infection in mosquitofish. In addition to the possibility that the differential distribution of snails may have an influence on the abundance and distribution of the metacercaria in *G. affinis*, it is also possible that temperature may differentially affect infected snails; e.g., Vernberg and Vernberg (1963) reported that *Nassarius obsoletus*, individually infected with a variety of larval trematodes, have reduced thermal resistance, as compared with non-parasitized individuals.

Finally, the third hypothesis is related to the distribution of potential definitive hosts for the parasites. Brisbin (1974) reported that the species diversity and abundance of waterfowl and wading birds within the Par Pond system appear to be influenced by thermal loading. The abundance

and species diversity of diving and dabbling ducks, the probable definitive host for *D. scheuringi*, were significantly lower in the area of Par Pond presently receiving thermal effluent than in areas of ambient water temperature. On the other hand, such wading birds as the great blue heron, the American egret, and the common egret are more frequently seen feeding in Pond C and the area surrounding the Hot Dam of Par Pond. The distribution of these birds, the potential definitive host for *O. ptychocheilus* (Hoffman, 1960), follows the observed distribution and abundance pattern for the brain metacercaria in *G. affinis*.

Despite the overall consistency in parasite distribution in relation to thermal characteristics of the Par Pond system, there were three sample sites, the boat dock #6, Pond B (1) and Dick's Pond, which did not fit the observed patterns. Water temperatures in Pond B are ambient. Thus the distribution of brain and body-cavity metacercaria in Pond B mosquitofish might be expected to resemble those observed in ambient areas of Par Pond. However, fish from Pond B had parasite burdens for brain and body-cavity metacercaria intermediate between the densities observed in the thermally elevated and ambient-temperature areas of Par Pond. These observations appear anachronistic, but we should point out that Pond B is presently undergoing post-thermal recovery and exhibits a number of biotic characteristics (species diversity and abundance of fish, reptiles, and aquatic vegetation) which place it in an intermediate position between what might be expected for the heated and ambient areas of Par Pond (Parker, Hirshfield, and Gibbons, 1973). Mosquitofish taken from the boat dock, also a thermally influenced area, had densities of parasite fauna resembling those of fish from areas of ambient water temperature. Changes in the distribution of the definitive host, in combination with a possible alteration in the density or species composition of the molluscs present in the area, could account for the altered parasite fauna. Studies now under way should shed some light on the peculiar distribution pattern of parasites in boat dock mosquitofish. Dick's Pond mosquitofish were not infected by either parasite: the absence of the larval trematodes probably reflects the absence of the appropriate definitive and/or intermediate hosts.

Differences between summer and winter densities of brain and body-cavity parasites strongly indicate seasonal changes in parasite population dynamics. Kennedy (1970) developed a "general systems theory" that helps explain the factors involved in the population biology of some parasitic helminths. He suggested that the principal factor controlling the intensity of parasitic infection in intermediate hosts is the availability of infective larvae. It has been shown that high water temperature (e.g., in the summer months) increases both the maturation rates of the intramolluscan larval stages and the cercarial emergence from the first intermediate hosts (Spall and Summerfelt, 1970). In the summer mosquitofish could be exposed to more cercaria than in the winter and would, accordingly, have higher levels of infection. In midwinter the lower water tempera-

ture would reduce the maturation time, retard cercarial release, and, thus, result in lower worm burdens in the fish. Casual observations made during the winter revealed large numbers of dead snails at all sampling locations; this would probably cause a reduction in the numbers of infected snails and, hence, in the production of cercaria available for infection of mosquitofish.

Previous studies by Eure and Esch (1974) and Bourque and Esch (1974) have shown that thermal effluent in the Par Pond reservoir system may act in such a way as to increase the abundance of the acanthocephalan, *Neoechinorhynchus cylindricus*, in largemouth bass, *Micropterus salmoides*, or alter the species diversity of enteric parasites in the yellow-bellied turtle, *Pseudemys scripta*. In each of these instances, the impact of thermal loading was probably related to changes in the feeding behavior of the definitive host. In the case of *G. affinis*, however, the feeding behavior of the intermediate host is not directly related to parasitism, because the two species of parasites are acquired via penetration of cercaria through external surfaces. Instead, species composition and intrapopulation densities (Esch, Gibbons, and Bourque 1975) are probably more closely related to the influence of thermal loading on the larval forms of the parasite, the snails serving as the first intermediate hosts, or on the distribution and abundance of the definitive hosts. Clearly, additional research is necessary to evaluate the various relationships between thermal loading and host-parasite population dynamics. The importance of such information will not only contribute to our knowledge of how thermal loading affects the various components of an ecosystem but will also lend information to our understanding of the ecology of host-parasite relationships.

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# Effects of a Thermal Effluent on the Ostracods of Par Pond, South Carolina

■ Marcia Bowen

## ABSTRACT

The effects of a thermal effluent on the distribution and abundance of freshwater ostracods were investigated. Hot and cold water and thermal recovery sites were sampled for their ostracod fauna, using both tow and core methods. An unusual assemblage of both tropical and temperate ostracods was collected. Neither high maximum nor high minimum temperatures had an effect on total numbers of ostracods per sample, but high minimum temperatures were correlated with low species diversity. Variations in the seasonal cycles of ostracods were related to temperature differences. Thermal stresses may have indirectly affected the ostracod population by eliminating rooted vegetation.

Ostracods are among the most ubiquitous of microcrustacea, occurring in such diverse habitats as the intestines of salamanders (Zinn, 1940) and crayfish gills (Tressler, 1959). In addition, they may be found in almost any freshwater environment, including lakes, streams, swamps, and cave waters (Tressler, 1959). Despite their abundance, this group remains little investigated, and much about their biology remains unknown.

One of the most puzzling questions about ostracods concerns the biological factors that control their distribution and abundance. Environments of certain species have been described, and certain factors have been correlated with the appearance of species. Yet there are few data, from either the laboratory or the field, which quantitatively illustrate the effects of different environmental factors on the dispersion of ostracod populations.

Temperature is one of the more obvious environmental factors that would be expected to influence the diversity and abundance of ostracod fauna. Ostracods are unique in that they seem to have the highest temperature tolerance of any aquatic metazoan (Wickstrom and Castenholz, 1973), and they have been found living in waters of thermal springs with temperatures as high as 45 to 50°C (Mason, 1939; Tagliasacchi-Masala, 1969). There is no consensus on the direct effects of temperature on distribution, however. Many investigators have concluded that the temperatures in their study areas had a negligible effect (Hoff, 1942; Puri, Bonaduce, and Malloy, 1964; McKenzie, 1964; King and Kornicker, 1970). Hoff (1943) found that, in neighboring ponds of the same temperature, a particular species of ostracod would appear earlier in one pond than in others.

Sergeeva and Vinogradskaya (1972) noticed no differences in the fauna (including ostracods) of heated and unheated portions of a cooling reservoir.

Other studies confirm the hypothesis that ostracods are distributed according to temperature. Robertson (1876) noticed that on cool days ostracods in his aquarium burrowed in the mud and on warmer days they swam near the surface. He also reported that, when one of two barrels of water from the same source was heated, one species of ostracod drastically increased in numbers, apparently thriving in the warmer water. In his study of organisms of Algerian thermal springs, Mason (1939) postulated the existence of certain thermally resistant races of nonthermal ostracods, in addition to thermal and nonthermal groups. Tagliasacchi-Masala (1969) divided ostracods of thermal springs into three types, thermal, nonthermal, and ubiquitous. According to Nuttal and Fernando (1970) the ostracod fauna of Lake Nipigon, Ontario, is divided into two types, warm water and cold water. Thermal tolerances of some ostracods are known. *Cypridopsis vidua* was abundant in cooling ponds at 29.4°C (Mason, 1939) and has been found in waters from 0 to 30°C (Elofson, 1941). *Heerocypris balnearia* has been found only in waters from ~40 to 50°C (Mason, 1939). *Aurila floridans*, a marine ostracod, does not tolerate temperatures lower than 6°C nor higher than 36°C (Kornicker, 1964).

Certain physiological effects of temperature are known. *Hirschmania viridis* has been found to keep its oxygen consumption constant at temperatures of 10 to 20°C; at other temperatures it conforms to Arrhenius's law (Hagerman, 1969b). Temperature also affects the time of development. Eggs kept at 28 to 30°C developed twice as fast as those kept at 17 to 19°C, which, in turn, developed twice as fast as those kept at 9 to 11°C (Wolgemuth, 1914). However, temperature exerts no discernible influence on the length of time spent in larval stages 7 and 8 (Hagerman, 1969a). Increased temperature can increase the speed of rejuvenation from a torpid state, a resistant state caused by drying (Delorme and Donald, 1969). This, in turn, could alter timing of the seasonal cycle.

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The purpose of this study is to evaluate the influence of thermal effluent from a nuclear production reactor on the diversity and abundance of the ostracod fauna in a cooling reservoir in the southeastern United States.

## MATERIALS AND METHODS

### Study Areas

The areas of study were two lakes on the grounds of the Energy Research and Development Administration's Savannah River Plant near Aiken, S. C. (Fig. 1). Par Pond, a 1133-ha man-made lake, was constructed in 1958 as a source of cooling water for a nuclear reactor (Parker, Hirschfield, and Gibbons, 1973). The lake receives thermal effluent from the reactor after it has traveled through ~14 km of cooling canals and ponds. A thermal gradient is produced in the arm of the lake receiving the effluent. Although summer temperatures may reach nearly 50°C in this arm, the rest of the lake has ambient temperatures (Gibbons, Hood, and Forney, 1971). The lake has been characterized as moderately productive, with the low nutrient level characteristic of lakes in South Carolina (Tilly, 1973). There is an abundance of rooted vegetation, including *Scirpus cyperinus* and *Typha latifolia* on the shoreline and *Eleocharis quadrangulata*, *E. equisetoides*, *Myriophyllum heterophyllum*, *Utricularia* sp., *Najas guadalupensis*, and *Potamogeton* sp. in the littoral zone. Par Pond is well populated with turtles, alligators, and fish, especially centrarchids (Parker, Hirschfield, and Gibbons, 1973).

Pond B, an 81-ha lake that received a thermal effluent from Reactor R from 1961 to 1964, now has temperatures ambient for the region. It is less productive than Par Pond and has a lower diversity of aquatic macrophytes and vertebrates. This is especially noticeable in the area where the thermal effluent once entered (Parker, Hirschfield, and Gibbons, 1973).

Of the four sampling sites chosen (Fig. 1), three were on Par Pond: the Hot Dam, which receives thermal effluent from Nuclear Reactor P; the Cold Dam, ~4.5 km distant, which remains at ambient temperatures; and North Cove, chosen because of the unusual species found in cove areas (Tilly, 1975a). North Cove received thermal effluent from Reactor R from 1961 to 1964. The fourth site was located in Pond B, near the former entry of thermal effluent from Reactor R. Areas were sampled weekly from June 12 to Aug. 14, 1973. Samples were also taken on Oct. 3, 16, and 30, and Nov. 5, 1973.

### Sampling of Ostracods

There is apparently no agreement in the literature on a generally acceptable collecting method for freshwater ostracods. Kornicker (1964) claims that one of the greatest problems confronting those who study ostracods is that of obtaining an unbiased sample. A major dilemma is that ostracods live both beneath and on top of the substratum. Two methods were employed in an effort to solve this problem.

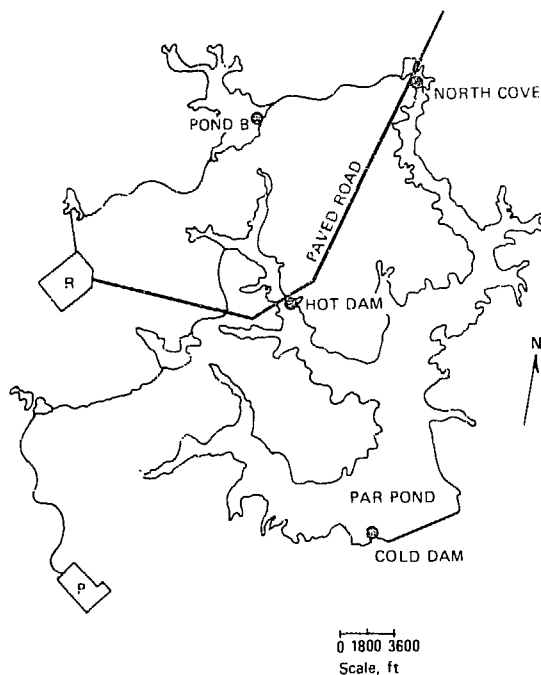


FIG. 1 The Par Pond System, showing sampling sites (●). P and R are nuclear reactors.

A number 20 mesh plankton net, 40.6 by 15.3 cm, was towed with its bottom edge just touching the substratum at uniform speed over a 5-m course. Five tows were taken at each station. This method is similar to that used by Armitage and Davis (1967).

To sample ostracods living in the water column and in the substratum, I used a plexiglass corer, 5.0 cm in diameter and 122.0 cm in length. The method, which is similar to that of Smyly (1957), involved directing the tube through the entire water column plus 5.0 cm of the substratum. The corer was removed from the substratum, and the contents were strained through a number 20 plankton net. Five cores were taken at each station whenever tows were made.

In August 1974, an artificial substratum was placed at the Hot Dam sampling site. The artificial substratum, constructed of hardware cloth of 1.27-cm mesh, was 91 cm wide and 261.0 cm long. Tufted bottle brushes, 3.8 cm in diameter and 11.4 cm long, with 16.5-cm handles, were used to facilitate attachment of ostracods. Brushes were placed upright, standing 19.2 cm tall in rows of eight, 14 cm apart. Sixteen rows 15 cm apart were made. Tows were taken biweekly in August, three each through this artificial substratum and outside of it, to compare the ostracod fauna of both areas.

All samples were refrigerated, and as many as possible within the period of 1 week were hand sorted. Unsorted samples were strained through a 100-mesh screen. All samples were preserved in a solution of 6 parts 95% alcohol and 1 part glycerin (Tressler, 1959). Adult ostracods from

three of the five summer samples taken weekly at each site, together with all five of the fall samples taken bimonthly at each site, were identified and counted.

### Physical Characteristics

Weekly water temperatures were taken at each site using submerged maximum–minimum thermometers. Maximum, minimum, and temperatures at the time of sampling were recorded. Additional temperature data for 1972 and 1973 were also obtained.

## RESULTS

### Temperature

Maximum temperatures were higher at the Hot Dam than at all other sites (Table 1). This difference in temperature occasionally exceeded 30°C in winter months but was

TABLE 1 Mean Weekly Temperatures (°C) in Par Pond and Pond B

| Site       | June–Aug. 1972 |              | Oct. 1972–Sept. 1973 |              |
|------------|----------------|--------------|----------------------|--------------|
|            | Mean maximum   | Mean minimum | Mean maximum         | Mean minimum |
| Hot Dam    | 40.1           | 34.2         | 35.3                 | 26.6         |
| Cold Dam   | 33.3           | 27.4         | 22.7                 | 17.5         |
| North Cove | 30.9           | 26.1         | 21.7                 | 16.1         |
| Pond B     | 31.5           | 30.7         |                      |              |

only 5°C in the summer months. The Student's t-test for paired data indicated that both maximum and minimum temperatures at the Hot Dam were significantly higher than those at the Cold Dam and North Cove ( $P < 0.005$ ), whereas these cooler sites did not differ significantly between each other. At Pond B, t-test results indicate that, although maximum summer temperatures were significantly lower than those at the Hot Dam, minimum temperatures were as high and were significantly higher than those at North Cove and the Cold Dam.

### Total Numbers

Ostracods were most abundant at the Cold Dam. Numbers per tow were four times those at any other site (Table 2). Total numbers were next highest at the Hot Dam and then at North Cove. Few ostracods were collected at Pond B. At all sites, tow numbers were much higher than core numbers, even when corrected so that volumes were equivalent. These results were confirmed by an analysis of covariance. Date of sample, site, method, and sample numbers (for within-sample variance) were the variates. First maximum, then minimum, temperatures were used as covariates. A logarithmic transformation was used to ensure homogeneity of variance.

Both sites and methods were significantly different in their total numbers ( $P < 0.0001$ ), but the dates of sampling and numbers were not significantly different ( $P > 0.40$  in

all cases). Neither maximum nor minimum temperatures were significant with respect to total numbers (probabilities of 0.9045 and 0.2696, respectively).

The samples at all sites were characterized by extremely high variance (Table 2). The variance-to-mean ratio is substantially greater than 1, a property characteristic of a contagious distribution. (Pielou, 1969).

### Community Structure

Seven species of ostracods were found in Par Pond and Pond B. All species were collected from only one site, the Cold Dam. *Cypridopsis vidua* was the most abundant ostracod there; but *Cypretta viridis* also occurred in substantial numbers (Fig. 2). There were also low numbers of *Strandesia spinulosa*, *Darwinula stevensoni*, *Stenocypris* sp., *Metacypris americana*, and *Physocypris pustulosa* at the Cold Dam. The Hot Dam had a fairly even distribution of four species at the beginning of the summer: *S. spinulosa*, *Stenocypris* sp., *P. pustulosa*, and *Cypridopsis vidua*. As the summer progressed, all species diminished except for *P. pustulosa*, which made up more than 80% of the population during the latter half of the period. North Cove tows and cores were composed mainly of *Cypretta viridis*, *Cypridopsis vidua*, *P. pustulosa*, and *M. americana*, with *S. spinulosa* and *D. stevensoni* occasionally appearing. The benthic species, *M. americana* and *D. stevensoni*, were more common in cores than tows. Few ostracods were found at Pond B; averages were 0.33 per tow and 1.76 per core. *Physocypris pustulosa* was most abundant, with *Cypridopsis vidua*, *Cypretta viridis*, and *D. stevensoni* also occurring.

All sites were different in ostracod composition (Fig. 2). A Shannon–Weaver diversity index (Pielou, 1966) was calculated for each sample to indicate quantitative differences. Analyses of covariance were performed using these diversity indexes. First maximum, then minimum, weekly temperatures were used as the covariates, with site, date, and sample as the variates. Minimum (but not maximum) temperature was found to be nearly significant ( $P = 0.0899$ ) with respect to diversity (Table 3). The

TABLE 2 Mean and Variance of Total Numbers of Ostracods in Par Pond and Pond B

| Site and method | Number of samples | Mean number | Standard deviation |
|-----------------|-------------------|-------------|--------------------|
| Cold Dam        |                   |             |                    |
| Tow             | 27                | 333.93      | 252.74             |
| Core            | 31                | 39.58       | 26.22              |
| Hot Dam         |                   |             |                    |
| Tow             | 31                | 79.35       | 85.21              |
| Core            | 35                | 22.63       | 28.70              |
| North Cove      |                   |             |                    |
| Tow             | 30                | 20.69       | 28.71              |
| Core            | 32                | 13.66       | 15.03              |
| Pond B          |                   |             |                    |
| Tow             | 30                | 0.33        | 0.84               |
| Core            | 34                | 1.76        | 2.26               |

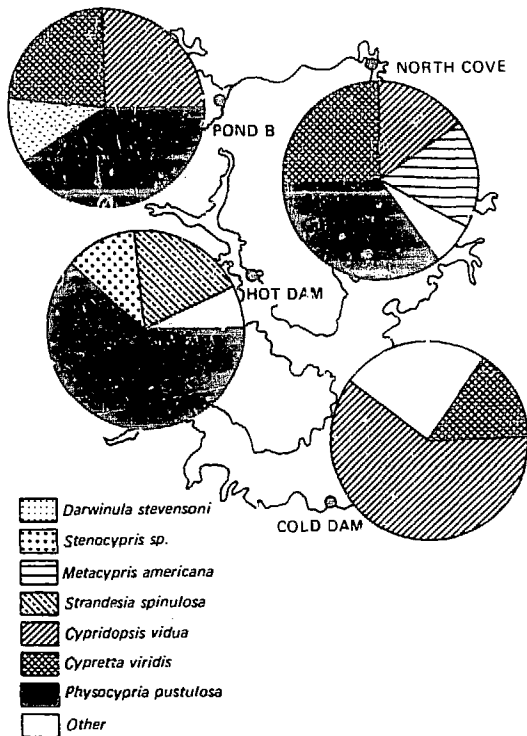


FIG. 2 Species assemblages, based on percentages of ostracods at the four sampling sites. The block labeled "Other" may also include the species listed when they occur in percentages too small to differentiate.

TABLE 3 Analysis of Covariance for Shannon-Weaver Diversity with Maximum and Minimum Weekly Temperature as Covariate

| Source      | Degrees of freedom | Probability > F |         |
|-------------|--------------------|-----------------|---------|
|             |                    | Maximum         | Minimum |
| Temperature | 1                  | 0.2238          | 0.0899  |
| Date        | 8                  | 0.4238          | 0.4300  |
| Site        | 3                  | 0.0001          | 0.0001  |
| Method      | 1                  | 0.4425          | 0.4386  |
| Sample      | 2                  | 0.2128          | 0.2676  |

correlation coefficient of these factors, minimum temperature and diversity, was 0.205, ( $P < 0.002$   $n = 216$ ). In the analyses of covariance, only the sites were significantly different. The Duncan multiple range test (Steel and Torrie, 1960) showed that the Cold Dam cores were significantly more diverse than all the other samples, followed by the Cold Dam tows and North Cove cores. The Hot Dam, and then Pond B, samples (both cores and tows) showed the lowest diversity (Table 4).

### Seasonal Cycles

Two species of ostracods had seasonal cycles that were of particular interest to this study. *Strandesia spinulosa* appeared first at the Hot Dam and somewhat later at the Cold Dam (Fig. 3). Numbers of the species plummeted at the Hot Dam during the time that minimum temperatures drastically increased (by  $15.5^{\circ}\text{C}$ , from  $20$  to  $35.5^{\circ}\text{C}$ ). Cold Dam *S. spinulosa* were not exposed to this coupling of high minimum and high maximum temperatures, and the population did not fluctuate. *Stenocypris* sp. also appeared earlier at the Hot Dam than at the Cold Dam (Fig. 4). The species appeared at both sites only after a minimum temperature of  $28$  to  $30^{\circ}\text{C}$  was reached, and numbers at the Hot Dam dropped when the minimum temperature fell below this point.

### Artificial Substratum

Before installation of the artificial substratum (August 5), few ostracods were found at the Hot Dam. The substratum was colonized in less than 3 days, however, by species that were rare in the surrounding areas. These included *Cyprretta viridis*, *Cypridopsis vidua*, *Stenocypris* sp., and *S. spinulosa*. Numbers of ostracods were significantly higher after installation of the artificial substratum than before (Fig. 5) and Shannon-Weaver diversity indexes remained high for the entire month of August (Fig. 6). Unusual results were seen in waters surrounding the artificial substratum. Numbers first increased sharply, then decreased. Diversity steadily increased until August 19, as additional species moved into the area. After this time both diversity and numbers decreased. A Mann-Whitney U test (Sokal and Rohlf, 1969) showed no significant differences between the species diversity of the

TABLE 4 Duncan Multiple Range Test for Site Differences in Shannon-Weaver Diversity Indexes\*

|                   | Pond B,<br>Core | Pond B,<br>Tow | Hot Dam,<br>Core | Hot Dam,<br>Tow | North<br>Cove, Tow | North<br>Cove, Core | Cold Dam,<br>Tow | Cold Dam,<br>Core |
|-------------------|-----------------|----------------|------------------|-----------------|--------------------|---------------------|------------------|-------------------|
| Mean<br>diversity | 0.047           | 0.164          | 0.369            | 0.430           | 0.552              | 0.618               | 0.802            | 1.082             |

\*Columns underscored by the same line are not significantly different. Columns not underscored by the same line are significantly different.

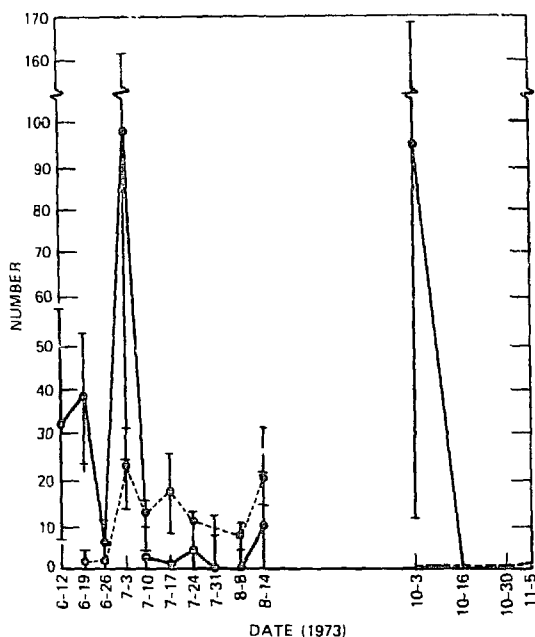


FIG. 3 Mean numbers of *Strandesia spinulosa* collected in tows in Par Pond and Pond B. Vertical lines indicate standard errors of the mean. —, Hot Dam. ----, Cold Dam.

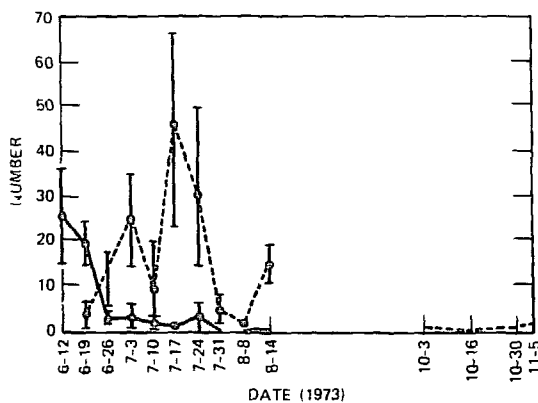


FIG. 4 Mean numbers of *Stenocypris* sp. collected in tows in Par Pond and Pond B. Vertical lines indicate standard errors of the mean. —, Hot Dam. ----, Cold Dam.

artificial substratum and that outside it in samples for the entire month of August. Similarly, there was no significant difference in total numbers per tow between the two areas.

The first half of the artificial substratum experiment coincided with a reactor shutdown, when all thermal effluent ceased. Temperatures were, therefore, cooler for the first half of the experiment. Maximum temperatures then increased from an average of 31°C for the first 2 weeks in August to 35°C on August 19 (after the reactor resumed operation) and 37°C on August 26 (Vigerstad,

1975). The increase in temperature coincided with the sharp drop in species diversity in waters outside the substratum. A similar decrease in diversity did not occur within the substratum.

## DISCUSSION

Of the two sampling methods used in this study, tows generally seemed to be superior and in most cases could be used exclusively. Greater numbers of ostracods per unit volume were collected by towing than by core samples, and representatives of all species were collected in Par Pond. Even benthic species, *M. americana* and *D. stevensoni*, which would be expected to be collected only by cores, were found in substantial numbers in tows.

Quantitative sampling of ostracods was a problem. Sample variance was so high that general trends were hard to elucidate. A similar problem was noted by Pierre (1973), who used 12 replicates per collection. Ferguson (1944) found that certain species in his samples disappeared and then reappeared within a few days without explanation. The contagious distribution that I hypothesize for ostracods in Par Pond might explain this phenomenon. Pierre

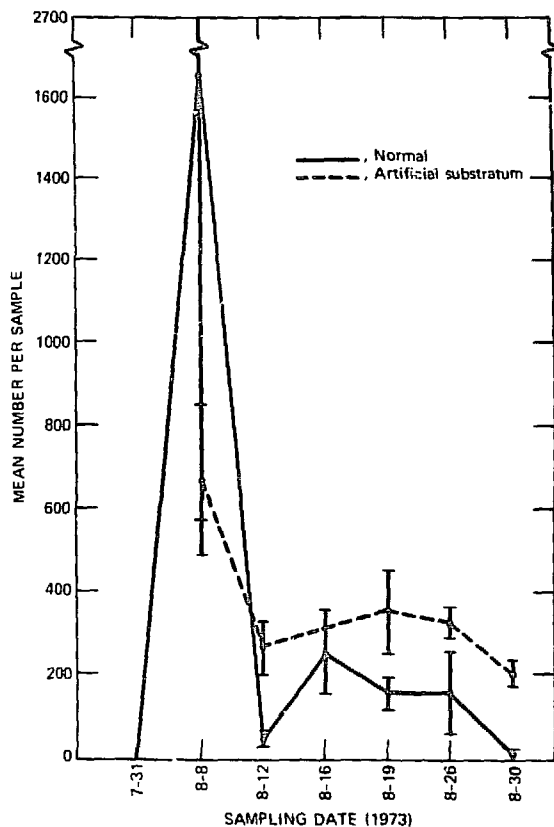


FIG. 5 Mean numbers of ostracods collected in tows at Hot Dam during the artificial substratum experiment. Vertical bars indicate standard errors of the mean.

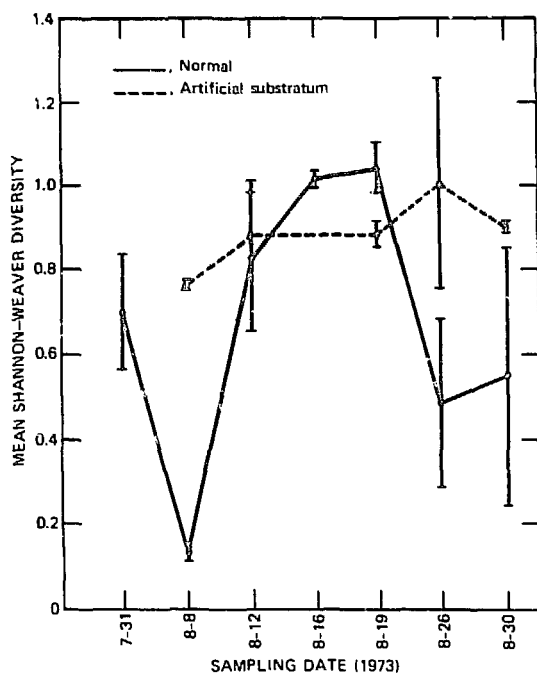


FIG. 6 Mean Shannon-Weaver diversities of samples collected at the Hot Dam during the artificial substratum experiment. Vertical bars indicate standard errors of the mean.

(1973) also concluded that ostracod populations are aggregated; his variance-to-mean ratios overlap those of the ostracods from Par Pond. The problem of finding a sampling method that truly reflects the actual population dynamics of a contagiously distributed ostracod population remains unsolved.

This study was hampered by a sampling regime restricted to the summer and autumn months. Even though ostracods are most common at these times, further sampling would have made results more clear-cut and, in general, enhanced the study.

Four of the seven species of ostracods in Par Pond, *Stenocypris* sp., *S. spinulosa*, *Cyprretta viridis* and *M. americana*, are generally assumed to be tropical species, and, for this reason, their appearance is unusual. The other three species found are common American species (Tressler, 1947) and two, *Cypridopsis vidua* and *P. pustulosa*, have been collected previously in South Carolina (Ferguson, 1952).

The genus *Strandesia* was once thought to be confined to South and Central America, and its presence in North America was thought to be the result of a northerly migration (Hoff, 1944). Also, *S. spinulosa* has been found in Macedonia (Petrovski, 1964) and in Puerto Rico (Delorme, 1975). The genus *Stenocypris* was assumed to be restricted to Asia, Africa, and Australia until it was discovered in Yucatan (Furtos, 1936). The species from Par Pond does not correspond to any of the new world species

(Ferguson, 1962) and probably is new. *Cyprretta viridis* has been collected previously only in Australia and New Zealand (Chapman, 1963; Furtos, 1934) and *M. americana* only in Yucatan (Furtos, 1936).

Particularly interesting is the appearance of the genera *Cyprretta*, *Strandesia*, and *Stenocypris* together in the same body of water. These three genera, along with *Chlamydotheca*, are a typical assemblage found in southern North America, South America, Asia, and Africa (Furtos, 1934). Although Furtos included southern North America within the range of these four genera, we must assume that she meant areas south of the United States since no species of *Strandesia* or *Stenocypris* had been found in the United States at that time. According to Furtos, the appearance of these genera indicates a similarity of tropical environments across the hemispheres. If this is true, then the Par Pond environment may be similar to tropical environments where the species are normally found.

Although ostracods have been collected in freshwaters of Alabama, Arkansas, Florida, Georgia, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, and Texas (Tressler, 1947), most of the collections were made before the first *Strandesia* was found in the United States and long before the first *Stenocypris* was found in Florida (Ferguson, 1962). In general, ostracod fauna in the United States has been only patchily investigated and is less well known than that of other microcrustaceans. Further collections in the Southeast today might reveal that the formerly tropical genera *Stenocypris*, *Strandesia*, and *Cyprretta* are now common to this area and that the ostracod fauna has changed in this region within recent years.

A northerly progression of tropical ostracods into temperate regions is probably aided by warm-water habitats such as Par Pond. This could explain the presence of *S. spinulosa* and *Stenocypris* sp. in substantial numbers at the Hot Dam, where the warm water may provide a suitable environment for their establishment. This does not explain the presence of tropical species in low numbers at the Cold Dam and North Cove. Thermally resistant members of these species may have spread to cooler portions of the lake until all suitable habitats were colonized. These species may also have invaded the newly-formed Par Pond merely because it was a new habitat with few predators and competitors and not because of its thermal characteristics.

This study has shown temperature to be related to community structure of ostracods but not to their total number. The analysis of covariance and correlation analysis using Shannon-Weaver diversity indexes to characterize the community structure showed minimum temperature to be correlated to diversity. The Hot Dam and Pond B had higher minimum temperatures and lower species diversity than the other sites; the Cold Dam and North Cove had lower minimum temperatures and higher species diversities. Other factors may contribute to the diversity of a site, however.

Pond B is the least favorable site for ostracods; it had the lowest numbers and species diversity of all the sampling

stations. After the nutrient-rich Savannah River water ceased to enter Pond B as a thermal effluent, nutrient levels dropped drastically (Tilly, 1975b). A probable result of this would be lowered primary productivity, which, in turn, could cause lower numbers of ostracods (King and Kornicker, 1970). Low nutrient levels also included low alkalinity (Tilly, 1975b). The scarcity of calcium in Pond B may limit all organisms with calcified exoskeletons, including ostracods. Experiments in which calcium phosphate and lime have been added to experimental ponds resulted in higher numbers of ostracods (Pierre, 1973). Ostracods in the low-alkaline Lake Nicaragua were thin shelled and fragile (Swain and Gilby, 1964). That two genera of ostracods, *Strandesia* and *Stenocypris*, whose shell ornamentation indicates they are weakly calcified (Hartman, 1964) did not appear in Pond B may further indicate the dependence of these species on dissolved calcium.

The two sites lowest in diversity also have few vascular plants, probably an indirect result of thermal stress (Parker, Hirschfield, and Gibbons, 1973). Plants often serve directly or indirectly as a food resource for ostracods (Monokov, 1972), and ostracod abundance has been correlated with presence of plants (McKenzie, 1964; Hagerman, 1969a; Hulings and Puri, 1964).

In this study the results of the artificial substratum experiment testing the effects of simulated rooted aquatic macrophytes indicated that the presence of artificial rooted vegetation increased species diversity and numbers. Similar results were found when an area of the Hot Dam where rooted vegetation normally grows was compared with areas of the Hot Dam without plants (M. Bowen, unpublished data). Rooted vegetation may provide food, an attachment point, and/or a refuge from predators. It appears that, in addition, the artificial plants offer some sort of protection from the effects of the thermal effluent. Diversity and numbers were lower in areas surrounding the artificial substratum only when waters were affected by a thermal effluent. The most obvious explanation for this would be that plants provide a holdfast in areas of higher flow caused by thermal effluent. Thus, although a thermal effluent may not directly affect those organisms tolerant enough to withstand elevated temperatures, it may eliminate the rooted vegetation that certain species require.

Two tropical species, *Strandesia spinulosa* and *Stenocypris* sp., appeared at different times at the Hot Dam and the Cold Dam. Perhaps the warm temperatures that occur early in the year, a direct result of the thermal effluent, explain the early appearance at the Hot Dam. Apparently the species require a certain minimum temperature. For *Stenocypris* sp., the minimum appears to be 28 to 30°C. Fluctuations in both species coincided with changes in the minimum temperature. The appearance and disappearance of *Stenocypris* sp. seem to be related to the attainment of a temperature of 28 to 30°C. Fluctuations in numbers might also be a result of the natural seasonal cycle or the patchy distribution, however.

In previous studies the relation of seasonal cycles to temperature has shown mixed results. Some studies have correlated temperature with the appearance of marine ostracods (Hulings and Puri, 1964), and other studies have shown no effect of temperature on their seasonal appearance (McKenzie, 1964; Hoff, 1943). The evidence in Par Pond points to the seasonal appearance of ostracods as a temperature-related phenomenon worthy of further investigation.

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# Seasonal Distribution of a Podocopid Ostracod in a Thermally Altered Area of Tampa Bay, Florida

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Norman J. Blake

## ABSTRACT

As part of a general ecological survey to determine the effects of a thermal effluent, quantitative samples of living benthic ostracods were collected from November 1973 through June 1974 and during August 1974 in the vicinity of a power plant located on Tampa Bay, Florida. Sampling stations were in the areas of maximum thermal influence and at a control site. For each sampling period, samples taken for organic carbon, salinity, sediment and water temperature, and dissolved oxygen showed that the areas were quite similar, except for temperature, which was found to have the greatest difference among stations of the effluent and ambient areas on a monthly basis. This study concentrates on the distribution of *Haplocytheridea setipunctata*, the dominant ostracod in the study area. Significant differences between population means were detected for all months except May and June. Maximum numbers of living *H. setipunctata* occurred at all stations in June, with densities of 55,000 individuals/m<sup>2</sup> in the area of thermal influence. A positive correlation of temperature with population density was found to occur from March through June at all stations. Temperature was found to play a significant role in limiting populations during the warmer months of the year. Carapaces of this ostracod were found to be depleted of calcium carbonate in individuals occurring in the thermal effluent.

Because of their importance as ecological indicators, recent marine ostracods have been the subject of many distributional investigations in the past few decades (Puri and Hulings, 1957). Geologically, fossil ostracods are useful in determining environmental conditions existing at the time of sediment deposition. Knowledge of environmental conditions in which recent ostracods are living can be used to project back in time to estimate conditions when the same species are found in preholocene sediments (Kornicker and Wise, 1960). From a biological point of view, marine benthic ostracods are important because they are part of

the meiobenthos and may contribute significantly to the productivity of the benthic community (McIntyre, 1969).

According to Maddocks (1974), a total of 79 papers describing benthic marine ostracods of the Gulf of Mexico have been published. Many of these deal directly with the influence of environmental factors on ostracod populations and their distributions. The literature pertaining to the west coast of Florida suggests that several factors, particularly salinity, organic content of sediments, and temperature, work in unison to regulate ostracod distributions (Hulings and Puri, 1964). However, most of the ecological studies in the Gulf of Mexico were based on a limited number of samples using the distribution of dead ostracods. In recent years living populations of ostracods have been shown to be subject to considerable fluctuations and to exhibit less diversity than nonliving communities (Kornicker and Wise, 1960; Baker and Hulings, 1966; King and Kornicker, 1970). Thus it is essential that conclusions concerning extant ostracod communities be based on living individuals. It is also important to determine the extent of influence of each measurable environmental factor considered important in regulating ostracod populations. To do so under field conditions would require an area where primarily one variable could be considered. The marine environment of Tampa Bay adjacent to the Florida Power Corporation's P. L. Bartow Plant provides sites that meet this requirement, with temperature being the apparent predominant variable. Our study was undertaken to determine the extent of influence of thermally elevated conditions on the populations of the podocopid ostracod *Haplocytheridea setipunctata* (Brady).

## MATERIALS AND METHODS

The study area is located on the west side of Old Tampa Bay adjacent to the P. L. Bartow Plant of the Florida Power Corporation. The plant itself, a fossil-fueled facility operat-

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ing since Sept. 30, 1958, is just north of the St. Petersburg city limits, on a landfill commonly referred to as Weeden Island (Fig. 1). Presently the plant generates 482 Mw of power, using 390,000 gpm of salt water for cooling. This results in a maximum temperature differential of 7.8°C. Water enters the plant from the intake channel constructed by the Florida Power Corporation for shipping operations and is pumped out at a velocity of 51 m/min into a tidal discharge canal.

Site selection for this study was based primarily on the existence of substantial temperature differences between the thermally affected area and the surrounding cooler waters of the natural environment. Additional considerations included the homogeneous nature of the sediments and the relatively uniform salinities, reflecting only seasonal changes (Blake, Doyle, and Pyle, this volume). Two stations each were chosen in the areas of maximum thermal influence (area 1,  $\Delta T > 4^\circ\text{C}$ ) and of ambient thermal influence (area 2). On the basis of previously collected data, precautions were taken to limit environmental variability (with the exception of temperature) among stations. All stations were established on sandy bottoms devoid of macrobenthic vegetation. Scattered sea-grass beds occurring in area 2 were specifically avoided since it has been shown that the presence of sea grass can influence species composition of ostracod populations (Waller, 1961; Hulings and Puri, 1964; McIntyre, 1969).

The original experimental design included data from two stations in a third sampling area, defined as an area of minimal thermal influence ( $\Delta T = 1$  to  $4^\circ\text{C}$ ). Shortly after collections began, however, it became apparent that uncontrollable variables in this area would make valid comparisons with the other stations impossible. For example, in December 1973 we noted that stands of the sea grasses *Syringodium filiforme* and *Halodule wrightii* had begun to develop quite near this sampling site. During the early spring of 1974, beds of these grasses had become established, and they remained throughout the study. Although living ostracods were taken from the samples collected at both stations in this area, at no time were living specimens of *Haplocytheridea setipunctata* (the dominant species of the other two areas) encountered. Very small numbers of carapaces of dead *H. setipunctata* found in these samples could possibly be accounted for through transportation or might be the remains of populations that existed in the geologic past. For these reasons, data from the area of minimal thermal influence was not considered in the analyses.

Five benthic faunal samples were taken from each station at monthly intervals from November 1973 through June 1974 and during August 1974. Samples were taken to a depth of 10 cm using a diver-operated hand-held PVC core with an inside diameter of 5 cm. Although King and Kornicker (1970) found cores inadequate to assess the variability in the ostracod communities of Redfish Bay, Texas, in our study the cores supplied an ample number of individuals per sample. All samples were retrieved from an

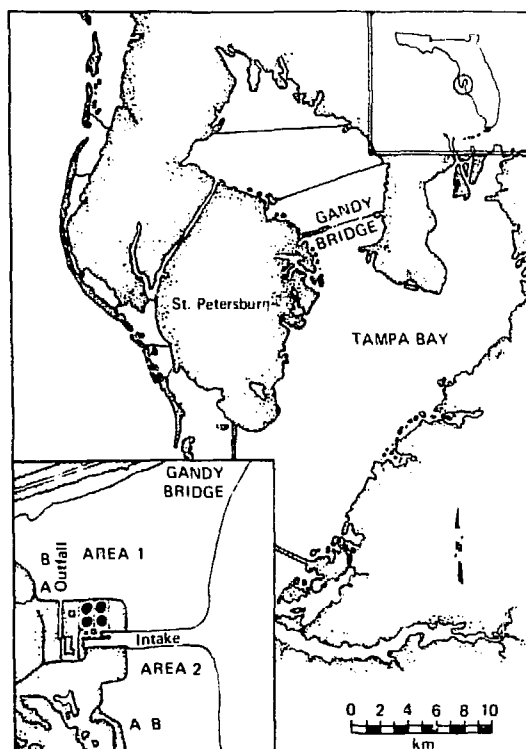


FIG. 1 Geographic setting and location of stations.

area of  $\sim 100 \text{ m}^2$  around the station marker. The samples were sieved in the field through a 250- $\mu\text{m}$  mesh screen and returned to the laboratory where they were relaxed in 0.15% propylene phenoxetyl (McKay and Hartzband, 1970) and fixed in 10% Formalin containing a 200-mg/liter concentration of rose bengal (Manson and Yovich, 1967). Although samples were sorted, identified, and counted for the presence of all living ostracod species, the results presented here are discussed primarily in relation to the dominant species, *Haplocytheridea setipunctata*. An individual was considered to be alive if the carapace gaped when propylene phenoxetyl was added and if it contained healthy, intact appendages. Other ostracod species will be considered only with respect to occurrence of live individuals at each station since the sampling technique and the use of a 250- $\mu\text{m}$  screen was designed to confidently recover only the *H. setipunctata* populations. The probability of complete recovery of additional genera is low (Sandberg, 1974).

Individuals from the five samples for each station were combined and mathematically converted to number of individuals per square meter for all months. Statistical analyses were performed to detect differences between the stations in area 1 and area 2 and those within areas. Raw numbers of individuals were transformed by the formula  $\log(X + 2)$  and were tested for homogeneity of variance

(Sokal and Rohlf, 1969). Transformed numbers were then used to perform an analysis of variance (ANOVA) for each month's data, totalling nine individual tests. When significant differences were detected, Duncan's new multiple range test (Duncan, 1955) was used to determine location of occurrence. All statistics were tested at the 95% confidence level as determined by  $P/X_i \pm 2(\sigma^2/N)^{1/2}$ .

Physical and chemical conditions measured in the field at the time of collections include salinity and water temperature measured with a salinometer, sediment temperatures taken with a telethermometer, and dissolved oxygen levels determined by the Winkler technique. Sediment samples were taken to a depth of 10 cm with a 5-cm PVC core for the months of November 1973 and February, May, and August 1974. Since sediments have been shown to remain relatively constant throughout the year (Blake, Doyle, and Pyle, this volume), only seasonal ranges for organic carbon and percentage carbonate will be presented here. These values were obtained through the use of an induction furnace and a carbon analyzer. Statistical analyses were performed by correlation coefficients to determine the correlation of salinity and temperature with populations of *H. setipunctata*. Where applicable, significance of the *r* values were determined by calculating the significant levels of *t* (Wang 700 Series). Raw numbers of *H. setipunctata* were transformed by  $\log(X+1)$  for these procedures.

## RESULTS

### Environmental Conditions

A summary of environmental data, as seasonal ranges, appears in Table 1, with a description of the yearly sediment conditions for each station.

Although salinity remained relatively uniform among stations, seasonal fluctuations were recorded, from an average high of 29.9‰ in May to an average low of 20.1‰

in August. Computation of correlation coefficients revealed no constant relationship with differences in *H. setipunctata* populations. In most cases values of *r* were found to be insignificant.

It was difficult to determine a pattern for monthly variations in levels of dissolved oxygen. Each station fluctuated unpredictably in relation to the others and to seasonal differences in temperature. The lowest value recorded for any station was 4.5 mg/liter, which is generally not considered limiting for benthic invertebrates (Simon, 1974).

Throughout the study, organic carbon content of the sediments was found to contribute a very small percentage of the total sample of each station. The contents of stations 1A, 1B, and 2A were comparable, with a yearly average of ~0.09%. The seasonal average of station 2B was somewhat lower (0.056%). It is unfortunate that these measurements could only be made on a seasonal basis since it has been suggested that this environmental factor may play a particularly important role in ostracod distributions (Hulings and Fari, 1964; Tietjen, 1969; Hulings, 1971). The importance of this factor may have been minimal for our study since observations revealed large populations existing in sediments with extremely low organic carbon content (e.g., 33,000/m<sup>2</sup> with 0.058% organic carbon at station 2A in November 1973).

Sediment and water temperatures of the study area were shown to be closely associated because of a rapid interchange resulting from the nature of the sandy sediments surrounding the plant (Blake, Doyle, and Pyle, this volume). Since measurements taken during our study support the earlier findings, further discussions of temperature will be based on temperature of the sediments. Temperatures were found to exhibit the greatest variations seasonally; minimum lows occurred at all stations in December, and maximum highs were recorded in August.

TABLE 1 Range of Physical and Chemical Measurements and Sediment Description

| Station | Water temperature, °C | Sediment temperature, °C | Salinity, ‰ | Dissolved oxygen, mg/liter | Sediment description                                                                                   |
|---------|-----------------------|--------------------------|-------------|----------------------------|--------------------------------------------------------------------------------------------------------|
| 1A      | 19.7–37.7             | 20.0–36.8                | 20.5–29.4   | 6.07–9.16                  | 96–98% medium-grained sand<br>1.2–2% silt–clay<br>0.049–2.58% organic carbon<br>0.071–0.142% carbonate |
| 1B      | 19.7–36.8             | 19.8–36.4                | 20.3–29.3   | 6.03–9.30                  | 96–98% medium-grained sand<br>1.0–2.0% silt–clay<br>0.047–0.272% organic carbon<br>0.077% carbonate    |
| 2A      | 12.1–31.0             | 14.0–30.7                | 19.7–28.9   | 4.35–10.0                  | 97–98% medium-grained sand<br>1.2–2.0% silt–clay<br>0.156–0.175% organic carbon<br>0.089% carbonate    |
| 2B      | 11.2–30.8             | 14.5–30.5                | 19.9–29.2   | 7.50–11.28                 | 97–98% medium-grained sand<br>1.2–2.0% silt–clay<br>0.013–0.099% organic carbon<br>0.056% carbonate    |

The overall seasonal range was from 14.5 to 36.8°C. Stations in the heated effluent (area 1) averaged 4.9°C higher than stations in the ambient environment (area 2). This was not a constant relationship, however. Seasonal differences ranging from a low  $\Delta T$  of 2.1°C in May to a high  $\Delta T$  of 6.7°C in January were detected. During the maximum stress period in August, when temperatures in the effluent area reached 36.8°C, the  $\Delta T$  was 6.3°C. The effects of this parameter on distributions of *H. setipunctata* are discussed later.

### Additional Ostracod Species

Twelve other species of living ostracods were collected in the two sampling areas during the study. Their area of occurrence and the stations at which they were encountered are shown in Table 2. Although differences could not be tested statistically, it appears that, on the average, more species were found at station 2B and the fewest species occurred at station 1B.

TABLE 2 Location of Occurrence of Additional Live Species of Ostracods Collected During the Study

| Species                       | Station |    |    |    |
|-------------------------------|---------|----|----|----|
|                               | 1A      | 1B | 2A | 2B |
| <i>Cytherura johnsoni</i>     |         |    | X  | X  |
| <i>Cyprideis floridana</i>    |         |    |    | X  |
| <i>Cushmanidea elongata</i>   | X       |    | X  | X  |
| <i>Haplocytheridea bradi</i>  | X       |    | X  | X  |
| <i>Hemicythere amygdala</i>   |         |    |    | X  |
| <i>Hulingsina sulcata</i>     |         |    | X  | X  |
| <i>Megacythere robusta</i>    |         |    |    | X  |
| <i>Parasterope pollex</i>     | X       | X  | X  | X  |
| <i>Polycopse</i> sp.          | X       |    |    | X  |
| <i>Proteochoncha concinna</i> | X       |    |    |    |
| <i>Sarsiella zostericola</i>  | X       | X  |    |    |
| Total                         | 6       | 2  | 5  | 9  |

### Monthly Densities of *H. setipunctata*

Table 3 gives the number of living individuals retrieved in the five 5-cm cores at all stations for months of low (November), medium (May), and high (June) abundances of *H. setipunctata*.

Figures 2 and 3 summarize the seasonal variation in densities of *H. setipunctata*. Significant differences were found between stations within a single area as well as between stations in different areas. Therefore, since data could not be pooled to obtain values to represent populations in a given area, stations are discussed individually.

Living specimens were found in the thermally influenced area at station 1B from January through June and at station 1A from March through June (Fig. 2). The occurrence of 101 individuals/m<sup>2</sup> at station 1B in January was not found to be significantly different from the complete absence of the organism from station 1A for this month, however. The lowest densities for each of the stations in area 1 were found in the winter, presumably

TABLE 3 Living *H. setipunctata* Collected at Each Station During Months of Low, Medium, and High Density\*

| Month and station | Total number of live ostracods | Average number per core | Range  |
|-------------------|--------------------------------|-------------------------|--------|
| November          |                                |                         |        |
| 1A                | 1                              | 0.2                     | 0-1    |
| 1B                | 1                              | 0.2                     | 0-1    |
| 2A                | 250                            | 50.0                    | 32-53  |
| 2B                | 33                             | 6.6                     | 2-11   |
| May               |                                |                         |        |
| 1A                | 55                             | 11.0                    | 0-20   |
| 1B                | 76                             | 15.2                    | 0-48   |
| 2A                | 152                            | 30.4                    | 6-54   |
| 2B                | 74                             | 14.8                    | 8-28   |
| June              |                                |                         |        |
| 1A                | 546                            | 109.2                   | 67-209 |
| 1B                | 371                            | 74.2                    | 52-126 |
| 2A                | 326                            | 65.2                    | 43-79  |
| 2B                | 510                            | 102.0                   | 36-167 |

\*Combined data for five samples taken each month at each station using 5-cm PVC cores.

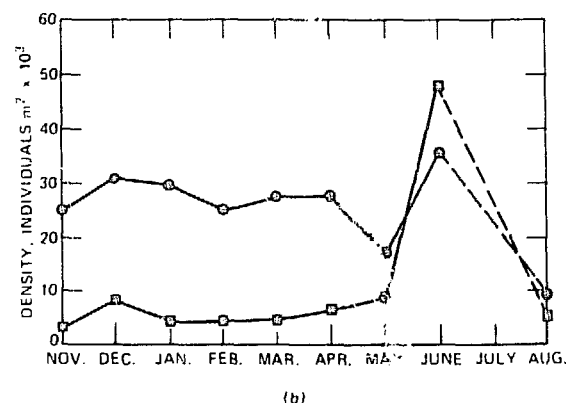
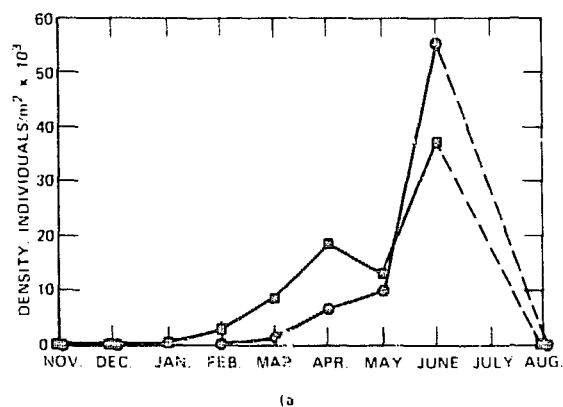


FIG. 2 Seasonal variations in living populations of *H. setipunctata*. (a) Area 1. (b) Area 2. ○—○, Station A. □—□, Station B.

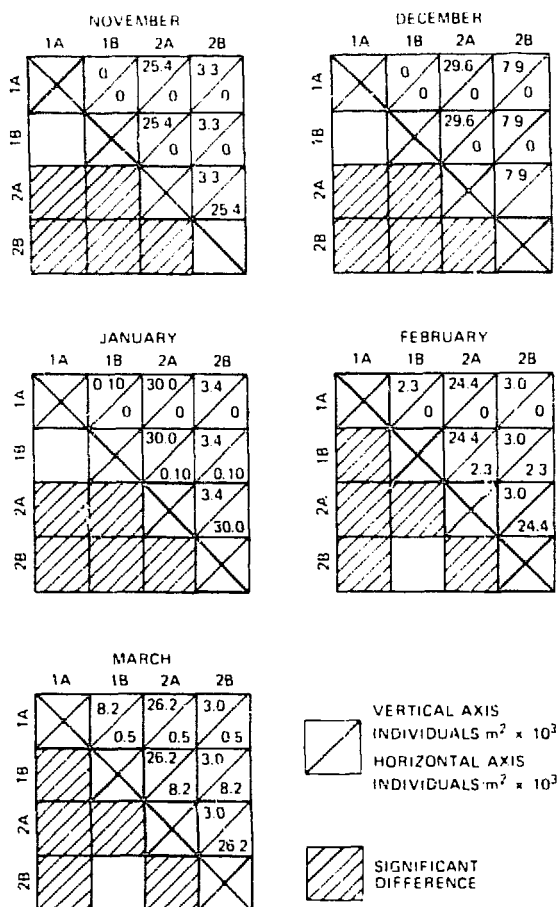


FIG. 3 Significant differences between stations as a result of analysis of variance and Duncan's new multiple range test.

when initial establishment of populations in this area was taking place. The maximum densities occurred in June ( $55,000/m^2$  at station 1A, which was the maximum density recorded throughout the entire study). Stations in area 1 were found to be statistically different in March and April and statistically equivalent in May and June (Figs. 2 and 3). Viable populations were not found at either station in November, December, and August.

In comparison, living populations were found during all months at both stations of area 2 (ambient area) (Fig. 2). These stations were found to be statistically different for each month, except May and June, when they were found to be statistically equivalent (Figs. 3 and 4). Maximum densities for both stations occurred in June, and the lowest populations were recorded in August.

Numbers of individuals of *H. setipunctata* at stations in area 1 were found to be statistically different from those at stations in area 2 from November through January. In February and March populations at 1B were statistically the

same as those occurring at the same time at station 2B. In April, 1A and 1B became statistically equal with populations of 2B and 2A, respectively. No significant differences were found to occur for the months of May and June. In August, however, areas 1 and 2 were again statistically different.

### Temperature vs. Population Density

The graphs in Figs. 5 and 6 represent the seasonal variation of densities of *H. setipunctata* with respect to monthly fluctuations in temperature. Because of a warmer than average winter, temperatures taken from November through February were found to fluctuate unpredictably. Unusually warm temperatures were recorded in November and January, but temperatures appeared to be nearer to normal winter conditions in December and February. With the exception of August, the lowest densities of *H. setipunctata* for all stations were recorded in the winter months and the highest in the summer. Throughout the winter, when sediment temperatures fluctuated between  $26.7$  and  $14.5^\circ C$  in the ambient area, *H. setipunctata* populations appeared to vary independently of temperature (i.e., there were  $\sim 8000$  individuals/ $m^2$  at  $14.5^\circ C$  at 2B in

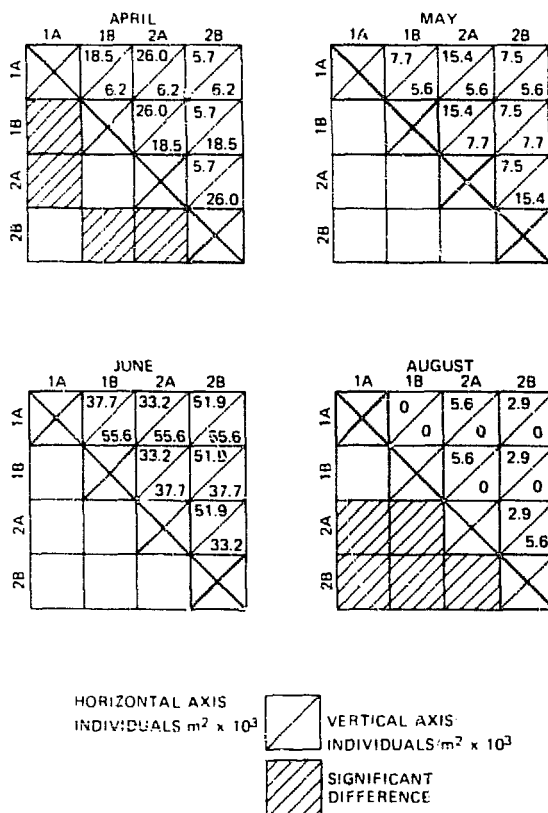


FIG. 4 Significant differences between stations as a result of analysis of variance and Duncan's new multiple range test.

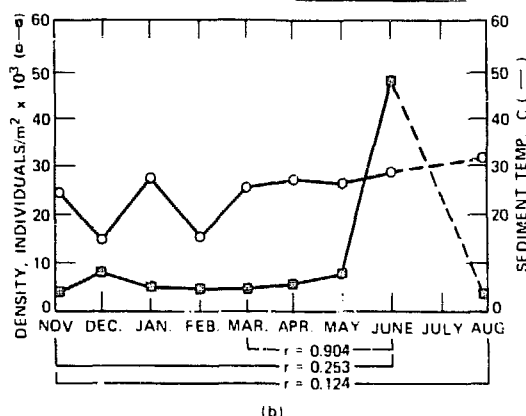
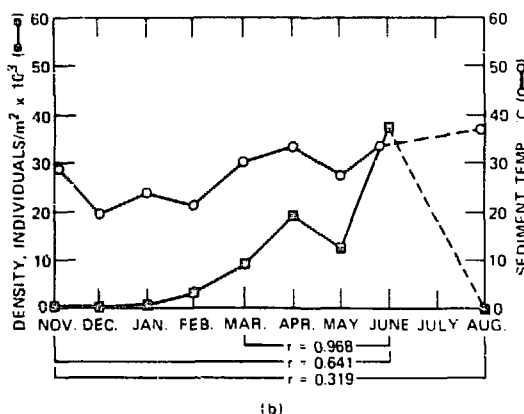
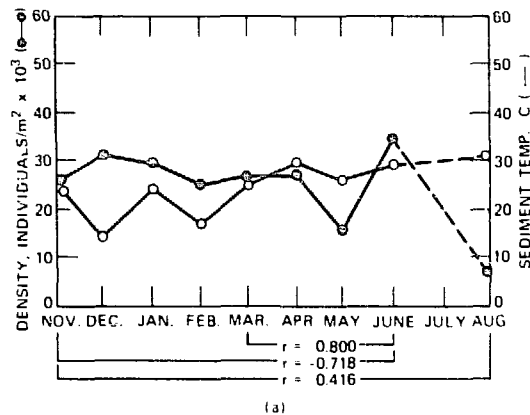
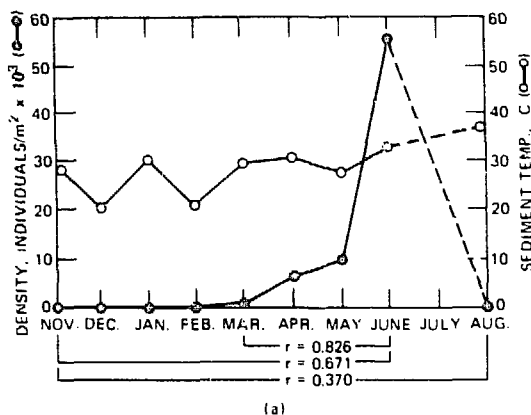


FIG. 5 Seasonal variations in the densities of living populations of *H. setipunctata* in area 1 with respect to temperature.  $r$  = correlation coefficient. (a) Station A. (b) Station B.

FIG. 6 Seasonal changes in the densities of living populations of *H. setipunctata* in area 2 with respect to temperature.  $r$  = correlation coefficient. (a) Station A. (b) Station B.

December and 3500 individuals/m<sup>2</sup> at 26.7°C at 2B in January). No living individuals were detected in the area of thermal influence during the colder months. As temperature began to increase from March through June, numbers of *H. setipunctata* appeared to increase correspondingly in both areas. In August, when temperatures exceeded 35°C in area 1, living individuals had disappeared from both 1A and 1B. Recorded temperatures for area 2 in August were somewhat lower (30°C), and populations had decreased to the lowest abundances observed there (5602 individuals/m<sup>2</sup> at 2A and 2954 individuals/m<sup>2</sup> at 2B). The calculated correlation coefficients seemed to correspond in part to these patterns (Figs. 5 and 6). A strong positive correlation resulted for all stations when analysis was restricted to the relationship existing from March through June. Coefficients computed from November through June for area 1 still suggested correlation, though less strongly. Corresponding values from the stations in the ambient area were found to differ; this indicates that the influence of temperature could not be correlated. When the August values were added, the coefficients became statistically insignificant.

### Decalcification Effects

During sorting procedures we observed that the carapaces of living individuals collected from the area of thermal influence were noticeably different from those from the ambient area. Most of the *H. setipunctata* from area 1 were considerably more opaque and often crumbled on being touched. At no time was this evident in specimens from the ambient area. Microscopic examination of *H. setipunctata* from the effluent area revealed that calcification was incomplete and the calcium carbonate remaining within the structure was milky white in color. The normal coloration of the *H. setipunctata* carapace is white; however, it is also transparent. This phenomenon seemed to be seasonally related; the minimum number of affected organisms was observed in March and the maximum in June. The area of calcium carbonate depletion varied from large to small patches throughout the carapace structure. In some cases only small patches of calcium carbonate remained, leaving the organic matrix of the carapace and the organism vulnerable to the environment. In other individuals only the area surrounding the pore canals was affected.

## DISCUSSION

Populations of *H. setipunctata* existing in a thermally affected area have been shown to exhibit statistically different distributions from those occurring in a similar habitat in an adjacent unaffected area during some months of the year. Definite problems have arisen in interpreting the data collected in the study, however. The most urgent consideration should be the occurrence of seasonal differences between stations in the same area. There are two plausible explanations for these variations in the ambient area. First, it has been well documented that populations of benthic invertebrates may exhibit patchiness of distribution (Boughay, 1973). Except for one month, the difference in populations in area 2 maintained a constant relationship, with consistently lower densities occurring at 2B. This may indicate patchy distribution. Second, field observations made during the study revealed eruptive increases in polychaete densities at station 2B. In May, when numbers of *H. setipunctata* decreased markedly at station 2A, there was a concurrent increase in numbers of the polychaete *Omphis cremita oculata*. This polychaete did not increase in density at 2B. The polychaete was still present in large numbers at 2A when *H. setipunctata* densities of 2B surpassed those of 2A in June. In addition, frequent fluctuations in ostracod populations have been observed in the field by others (see, e.g., discussion by Hulings, 1971).

The seasonal differences in the area of thermal influence seem to have resulted from a slightly different process. Since no living *H. setipunctata* were found to exist at either station during the first three months or the last month of the study, we can reasonably assume that the organism was not present. If living specimens were truly absent, how did they suddenly reappear in February and March? There are two possible explanations for the appearance and disappearance of *H. setipunctata* in the effluent area. First, juveniles could have been reintroduced by winter-wind-driven currents. The migration of ostracods has been suggested in the literature (Hagermann, 1966; 1969; Whatley and Wall, 1969; 1975). Once an organism has been introduced into a favorable environment, it establishes and multiplies rapidly. *H. setipunctata* first appeared at 1B; this indicates an initial establishment of populations for the area at this station. Within a month they were detected in small numbers at station 1A. It is felt that, if their introduction was as hypothesized, a lag time set up by temporal differences in establishment could account for the seasonal differences. Second, they may have produced resting eggs as effluent conditions became unfavorable during the late summer months; this would account for their complete disappearance in August. Certain species of ostracods are well known for overwintering by this method (McGregor, 1969; Whatley and Wall, 1969).

Although it is evident that no single factor is responsible for limiting ostracod distributions in this study area throughout the year and that they are more likely controlled by a number of environmental factors acting synergistically, the importance of temperature in regulating

ostracod abundances during the warmer months of the year is certain. Various investigators have suggested that one factor in particular may play an especially important role in some situations. For example, Tietjen (1969) found that abundances increase with increasing plant detritus. Tressler and Smith (1948) and Waller (1961) suggested temperature as a primary regulator of ostracod populations; Waller noted a definite correlation between temperature and seasonal abundances of ostracods in St. Andrews Bay, Florida. It appears that, if circumstances allowed one parameter to dominate by becoming critical, that parameter could override the effects of others. Data from our study suggest that there may be an optimum temperature for *H. setipunctata* in the Bartow area, and, when it is surpassed, populations are reduced or completely eliminated. The data indicate that this optimum may fall in the range from 31 to 35°C. Additional laboratory studies are required to define the critical limits accurately. Although temperature may not necessarily be limiting the distributions of *H. setipunctata*, it is felt that conditions resulting from increased temperature are directly responsible for the decline in populations in August.

We do not know whether calcium carbonate loss from the carapaces of *H. setipunctata* in the effluent area is temperature related, but the process of calcium carbonate deposition by certain invertebrates has been shown to be temperature related. Ruddy, Feng, and Campbell (1975) documented that shell deposition of the American oyster, *Crassostrea virginica*, was affected by prolonged exposure to elevated temperatures. Shells of oysters maintained in a heated effluent were found to be altered in both shape and thickness. Bock (1975a) reported that foraminiferal samples collected from a heated effluent from another power plant on Tampa Bay were found to be brittle and lacking in calcium carbonate. He reported that living populations averaged ~50% of the total number of foraminiferans collected. This is somewhat unusual since living foraminiferans, like ostracods, are frequently found among tests of dead individuals, but the latter normally account for a much higher percentage of the sample. Bock (1975a) attributed this occurrence to the complete dissolution of many tests in the effluent because of the predominance of quartz sediments. Many geologists feel that this type of sediment is somehow responsible for a significant lowering of the pH (Bock, 1975b). In our study sediments from the ambient as well as the effluent area are composed of quartz sand. Since the carapaces of organisms from the ambient area were not found to be lacking in calcium carbonate, we feel that this was not the causative agent. Physiological problems involving the deposition process may have caused reduced quantities of calcium carbonate in organisms encountered in this study since molting in ostracods requires a complete recalcification of the carapace structure (Turpen and Angell, 1971). Although calcium carbonate depletion in *H. setipunctata* appears to be definitely related to effluent conditions, further investigations are required to determine the actual cause of the phenomenon. It is



noteworthy that the greatest abundances of living ostracods occurred during the month of June, just prior to the loss of carapaces by the described effect. This certainly resulted in a considerable loss to the geologic record of the effluent area.

In summary, temperature appears to play a larger role in regulating populations of *H. setipunctata* in the thermally affected area than it does in the ambient area. Since temperatures in the effluent area are much higher than those in the ambient area, they are more likely to exceed the optimum and lethal limits for this organism and contribute to the observed complete elimination of living *H. setipunctata* from the effluent area.

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# Chronic In Situ Exposure of the Bay Scallop to Power-Plant Effluents

John F. Studt  
Norman J. Blake

## ABSTRACT

The bay scallop, *Argopecten irradians concentricus* (Say), was transplanted to the vicinities of two power-plant installations, the P. L. Bartow site (Tampa Bay, Fla.) and the Anclote estuary site (near Tarpon Springs, Fla.). The mean increase in temperature over ambient at the P. L. Bartow site was 6°C near the outfall, and the mean thermal increase at the Anclote site was 2°C near the outfall. Scallops were exposed to a thermal gradient at three stations ( $\Delta 1, 3^\circ\text{C}$ ;  $\Delta 2, 4^\circ\text{C}$ ;  $\Delta 4, 5^\circ\text{C}$ ) at the P. L. Bartow site for 1 month, and at a  $\Delta T, 1$  to  $2^\circ\text{C}$ , station at Anclote for over 5 months. Incidence of mortality was higher at the thermally altered stations than at the ambient stations. Sublethal effects (disruption of the normal reproductive cycle) were found at all thermally altered stations.

Since the bay scallop is a species highly sensitive to environmental perturbations (Blake, 1972), it should be considered in detail when it is a natural inhabitant of a study area that may be subjected to environmental alteration. This study addresses the problem of long-term in situ exposure of bay scallops to power-plant effluents. Subtle sublethal effects of thermal addition on the "natural" reproductive cycle of the bay scallop were assayed.

Temperature has long been recognized as an important controlling factor in the marine environment (Gunter, 1957; Kinne, 1970), and the effects of temperature on the reproductive cycles of marine invertebrates have been extensively considered in the literature (Orton, 1920; Thorson, 1950; Giese, 1959; Loosanoff and Davis, 1963). Early investigations dealing with the bay scallop elucidated general reproductive trends as a function of temperature (Belding, 1910; Gutsell, 1930). More recently, investigations by Sastry (1961; 1963; 1966; 1968; 1970) have contributed greatly to our present knowledge of the reproductive physiology of the bay scallop. He has devoted much effort to investigating the relationships of food and temperature with reproduction.

Many of the past investigations involving alteration of the natural temperature regimes of marine animals have been conducted under laboratory conditions to exclude as

many extraneous factors as possible. Vernberg and Vernberg (1974), however, pointed out the importance of interactions of various factors in the environment. The need now exists for study by in situ exposure of marine organisms to thermal alteration, e.g., the investigation by Tinsman and Maurer (1974) on the American oyster and the work described here.

## MATERIALS AND METHODS

The investigations at the P. L. Bartow and the Anclote power plants were conducted from January through February 1974 (Bartow study) and from September 1974 through February 1975 (Anclote study). These intervals coincide with gonad maturation and spawning periods reported for the southern populations of the bay scallop by Gutsell (1930) and Sastry (1966; 1970).

The Anclote power plant, located at the entrance of the Anclote River to the Gulf of Mexico, ~50 km north of the mouth of Tampa Bay, is a fossil-fueled installation presently producing 515 Mw of power at a pumping rate of 7562 m<sup>3</sup>/min; it is expected to expand to a total of 1030 Mw. The plant began operating in September 1974, at the same time this study was initiated, with a thermal addition of 1 to 2°C over ambient water temperature. Scallops are the dominant epifaunal bivalve in the Anclote study area (Blake et al., 1973), but a decline in the 1974 population necessitated the transplanting of specimens from the Crystal River estuary for this study.

The Bartow power plant, located on the west side of the mouth of Old Tampa Bay, first started generating power in September 1958 and presently produces 482 Mw at a pumping rate of 418 m<sup>3</sup>/min. The thermal alteration at the outfall is 6°C over ambient. Scallops occur as sparse populations in the Tampa Bay system (Dragovich and Kelly, 1964; personal observations). Since collection from sparse populations is logistically difficult, scallops were transplanted to the Bartow site from the Anclote population to ascertain the effects of the 6°C increase in

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temperature (over twice the increase occurring at the Anclote study area).

Since the bay scallop is a motile bivalve, it was necessary to use covered cages, which excluded two major predators, the stone crab, *Menippe mercenaria*, and the blue crab, *Callinectes sapidus*. The cages, made of pine and plastic-coated galvanized wire, were 1 sq m in area and 25 cm deep. They were suspended 0.5 m above the sediment on four lengths of metal conduit. Each cage could accommodate more than the 100 scallops placed in it. Duggan (1973) found that he could put up to 50 scallops per 0.09 sq m in cages before crowding caused adverse effects. He also found that placement in the water column did not affect growth.

At the Bartow site an ambient station (A, 21 to 23°C), and three experimental stations (B, Δ4 to 5°C; C, Δ2 to 4°C; and D, Δ1 to 3°C) were established (Fig. 1). In the vicinity of the Anclote power plant, one ambient station (1) and two experimental stations (2 and 3) were selected (Fig. 2). During the study period temperatures ranged from 10 to 24°C at station 1. Station 2 was essentially always encompassed by the plume (Δ1 to 2°C), and station 3 incurred only periodic exposures, caused by unusual wind and current conditions. Fifty scallops were placed at each of the four Bartow stations. At the Anclote site 200 scallops were placed at each of the three stations.

Between Jan. 23 and Feb. 20, 1974, scallops were collected from the four stations near the Bartow power

plant. Five scallops were collected weekly, and the number of dead was noted at each station. The scallops were transplanted to the laboratory and fixed in Dietrich's fixative for at least 24 hr. The tissues were then dehydrated, cleared, imbedded in Paraplast (Sherwood Medical Industries), sectioned at 6 μm, and stained with hematoxylin and eosin. Sagittal sections, including gonadal tissue from each scallop, were observed at 430X and maximum diameters of 30 oocytes were measured.

Once every three weeks from Oct. 15, 1974, to Feb. 18, 1975, 20 scallops were collected from each of the three Anclote stations, and the number of dead scallops at each station was noted. Fourteen of each group of 20 scallops were dissected into four parts: (1) gills and mantle, (2) gonad, (3) digestive gland, and (4) muscle. After drying for 72 hr at 60°C in preweighed aluminum pans, the dissected tissues were weighed, ashed for 6 hr at 550°C, and weighed a final time. Tissue indexes were calculated from the ash-free weights (gonad index = gonad weight/total weight X 100, digestive-gland index = digestive-gland weight/total weight X 100, muscle index = muscle weight/total weight X 100). The remaining six scallops from each station were fixed in Dietrich's fixative and histologically analyzed as described earlier.

Water samples were collected at each of the three stations in the Anclote study area at ~3-week intervals. Levels of salinity, total suspended load (by weight of

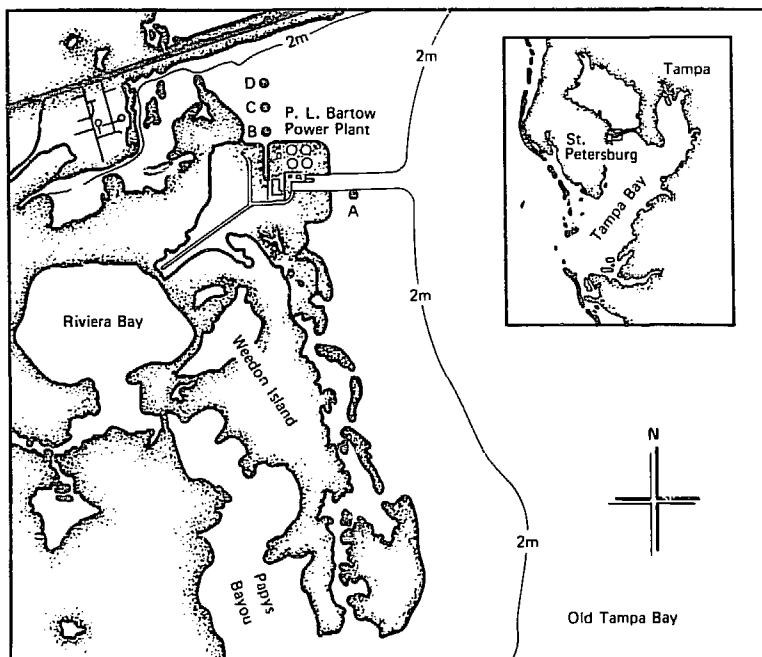


FIG. 1 Station locations in the vicinity of the Bartow power plant. Station A, ambient; station B, effluent (Δ4 to 5°C); station C, effluent (Δ2 to 4°C); station D, effluent (Δ1 to 3°C).

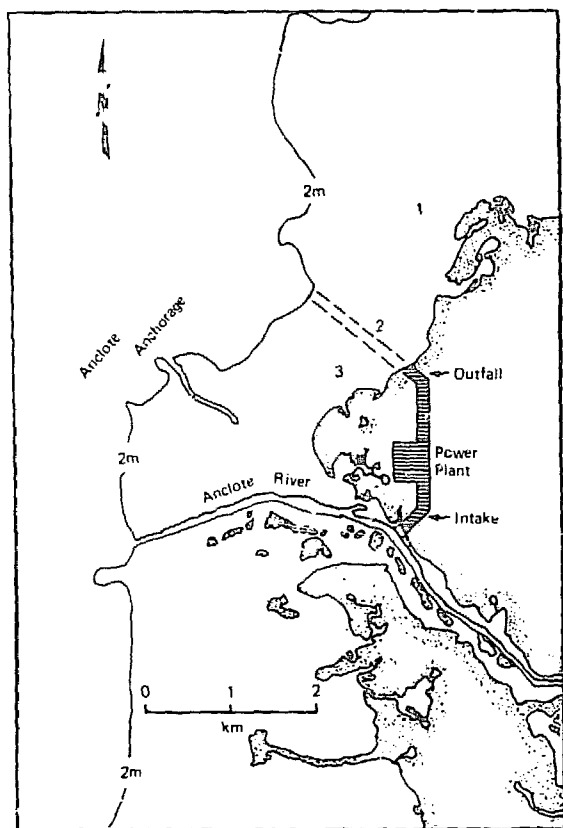


FIG. 2 Station locations in the vicinity of the Ancote power plant. Station 1, ambient; station 2, effluent; station 3, periodically exposed.

filtered material), and chlorophyll *a* (Strickland and Parsons, 1968) were determined.

To determine where significant differences occurred in the tissue indexes, we used a multiway analysis of variance (randomized complete-block design) and Duncan's new multiple range test. The level of significance used throughout was  $P < 0.05$  (Steel and Torrie, 1960).

## RESULTS

### Ancote

Chlorophyll *a* concentrations (Fig. 3) during the period from Oct. 10 to Nov. 26, 1974, were greater at both experimental stations 2 and 3 than at station 1. Similar chlorophyll *a* concentrations occurred at all three stations between Dec. 12, 1974, and Jan. 28, 1975 (Fig. 3). Total suspended load (Fig. 4) was, with one exception, greater at station 2 (the station within the plume) than at either the ambient station (1) or the periodically exposed station (3). All total-suspended-load values fluctuated with wind conditions (Fig. 4). Salinity was generally 28 to 30‰ and always fell within the tolerance limits of the bay scallop (20 to 38‰) as reported by Gutsell (1930). Temperature was

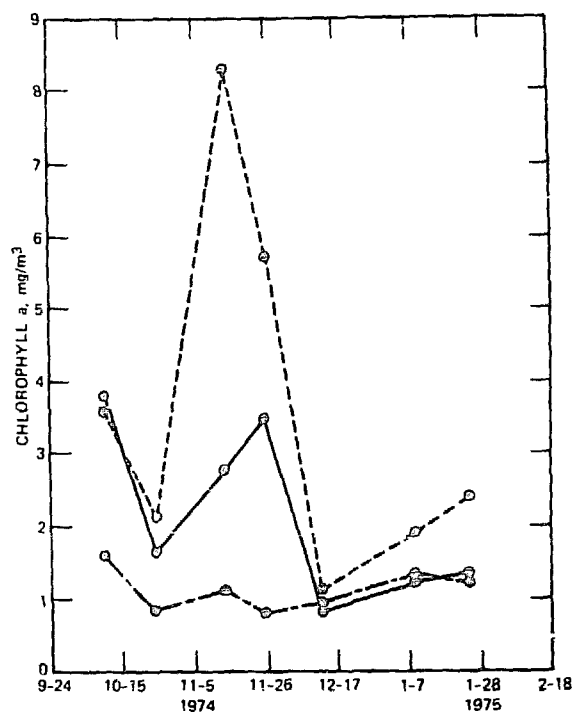


FIG. 3 Chlorophyll *a* concentrations at the three Ancote stations (1, ---; 2, ---; 3, —) during the study period.

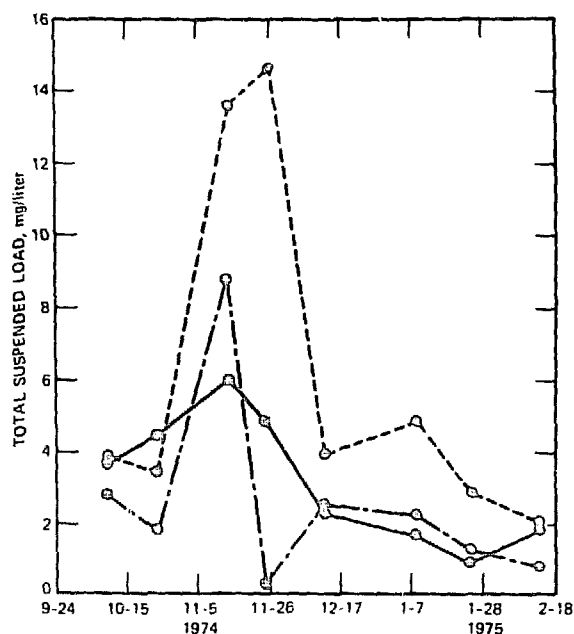


FIG. 4 Total suspended load at the three Ancote stations (1, ---; 2, ---; 3, —) during the study period.

2°C higher at station 2 than either of the other stations. Periodically, however, during certain wind and current conditions, station 3 was exposed to the plume and the coincident 2°C temperature increase and greater total suspended load.

During the 5 months of exposure, greater mortality was observed at the thermally altered station (2) than at the ambient station (Table 1)

1974 (Table 2). Slight fluctuations in the index occurred, but it did not show the inverse relationship to gonad index that Sastry (1966; 1970) found during rapid gonad growth. Indeed, during the rapid gonad growth, Sept. 24 to Nov. 5, 1974, the digestive-gland index increased or remained the same at all three stations (Fig. 5).

The muscle index (Fig. 5) at all three stations decreased rapidly during early gonadal growth, Sept. 24 through Oct.

Table 1 Observed Mortality of Caged Scallops Located in and Adjacent to Thermally Altered Areas of Anclote and Bartow Power Plants, Florida

| Date          | Mortality*  |           |           |           |
|---------------|-------------|-----------|-----------|-----------|
|               | Station 1 * | Station 2 | Station 3 |           |
| Anclote       |             |           |           |           |
| Oct. 15, 1974 | 1           | 0         | 3         |           |
| Nov. 5, 1974  | 3           | 0         | 1         |           |
| Nov. 26, 1974 | 5           | 4         | 0         |           |
| Dec. 17, 1974 | 3           | 3         | 2         |           |
| Jan. 7, 1975  | 0           | 1         | 1         |           |
| Jan. 28, 1975 | 2           | 5         | 2         |           |
| Feb. 18, 1975 | 0           | 2         | 6         |           |
| Mar. 11, 1975 | 0           | 10        | 3         |           |
| Total         | 14          | 25        | 18        |           |
| Mortality*    |             |           |           |           |
| Date          | Station A   | Station B | Station C | Station D |
| Bartow        |             |           |           |           |
| Jan. 23, 1974 | 0           | 0         | 0         | 0         |
| Jan. 30, 1974 | 1           | 5         | 3         | 1         |
| Feb. 6, 1975  | 0           | 6         | 4         | 2         |
| Feb. 13, 1974 | 2           | 4         | 2         | 1         |
| Feb. 20, 1974 | 0           | 4         | 3         | 0         |
| Total         | 3           | 19        | 12        | 4         |

\*Mortalities were recorded and dead individuals were removed on each sampling date.

The gonad index (Fig. 5) at station 1 increased from the onset of the experiment to a peak value on Dec. 17, 1974 (Fig. 5). After the peak the index started a steady decline indicative of spawning (Sastry, 1966; Giese, 1959). The gonad index at both stations 2 and 3 increased between Sept. 24 and Nov. 5, 1974. From November 5 to 26 at these stations the gonad index decreased, possibly as a result of partial spawning. From November 26 to December 17, the gonad index at both stations again increased. After December 17, the index at station 3 began to decrease as did that at station 1. The decrease in gonad index and, therefore, the spawning were delayed 3 weeks or more at station 2. The peak amplitude of the gonad index was significantly higher at station 1 than either of the experimental stations on Dec. 17, 1974 (Fig. 5, Table 2).

The digestive-gland index (Fig. 5) was significantly different among the three stations only once, Nov. 26,

15, 1974 (Fig. 5). At station 1 the decline continued until December 17, but at the two experimental stations it increased again until November 26. At station 1 the muscle index was significantly lower than either of the experimental stations on both November 26 and December 17.

Trends similar to those found for the gonad index occurred in mean oocyte diameter (Fig. 6). The decrease at all three stations in mean oocyte diameter on November 26 indicates the possibility of partial spawning (Sastry, 1966). Many oocytes had reached vitellogenesis at all three stations by December 17, after which spawning commenced at stations 1 and 3. Spawning was apparently delayed at station 2 until after Jan. 7, 1975. Numerous phagocytes and resorption were observed at station 2 in the last sample (Feb. 18, 1975), when temperatures above 25°C had been recorded. At no other time was resorption or phagocytic activity observed.

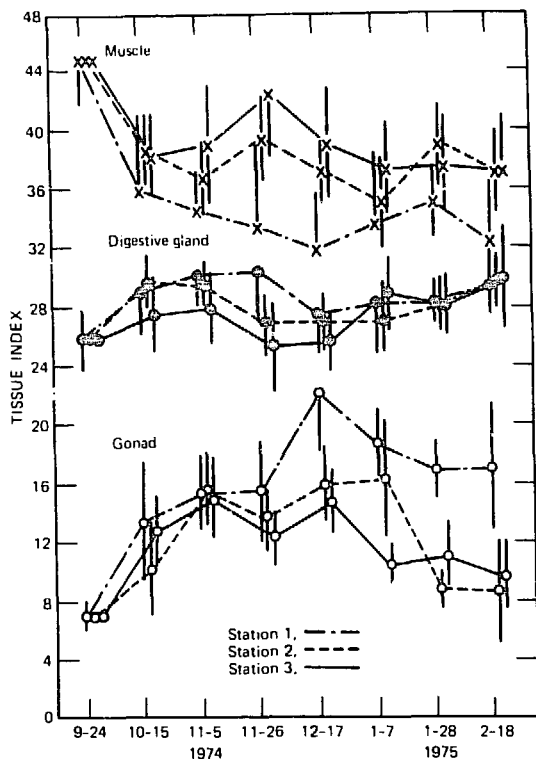


FIG. 5 Tissue indexes of scallops from the Anclote study area. Fourteen scallops were collected from each station once every 3 weeks for determination of mean tissue indexes. Vertical lines represent 1 standard deviation from the mean.

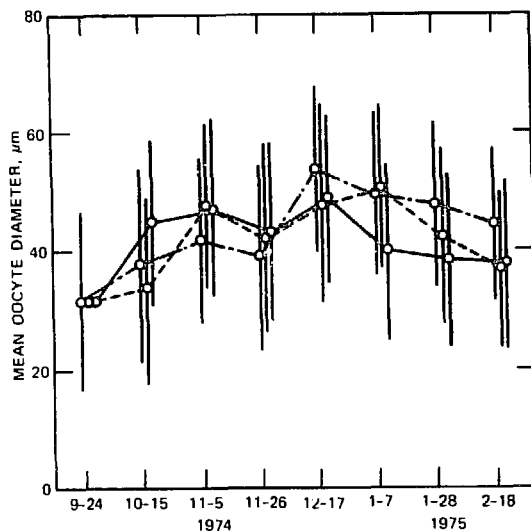


FIG. 6 Mean diameter of oocytes ( $\mu\text{m}$ ) from the three stations (1, ---; 2, - - -; 3, —) in the Anclote study area. Thirty oocytes from each of six scallops were measured once every three weeks for each station to determine mean diameter. Vertical lines represent one standard deviation from the mean.

TABLE 2 Results of Duncan's New Multiple Range Test\* ( $P < 0.05$ ), for Differences Between Tissue Indexes of the Three Anclote Stations† (1, 2, 3)

| Date          | Ranked means |             |             |                       |             |             |              |             |             |
|---------------|--------------|-------------|-------------|-----------------------|-------------|-------------|--------------|-------------|-------------|
|               | Gonad index  |             |             | Digestive-gland index |             |             | Muscle index |             |             |
| Oct. 15, 1974 | 2            | 3           | 1           |                       |             |             |              |             |             |
|               | 10.2         | <u>12.8</u> | <u>13.5</u> | NS                    |             |             | NS           |             |             |
| Nov. 5, 1974  |              | NS          |             | NS                    |             |             | 1            | 2           | 3           |
|               |              |             |             |                       |             |             | <u>34.6</u>  | <u>36.7</u> | <u>39.1</u> |
| Nov. 26, 1974 | 3            | 2           | 1           | 3                     | 2           | 1           | 1            | 2           | 3           |
|               | <u>12.4</u>  | <u>13.7</u> | <u>15.6</u> | <u>25.3</u>           | <u>26.8</u> | <u>30.4</u> | <u>33.6</u>  | <u>39.5</u> | <u>42.5</u> |
| Dec. 17, 1974 | 3            | 2           | 1           |                       |             |             | 1            | 2           | 3           |
|               | <u>14.7</u>  | <u>15.9</u> | <u>22.1</u> | NS                    |             |             | <u>31.8</u>  | <u>37.2</u> | <u>39.0</u> |
| Jan. 7, 1975  | 3            | 2           | 1           |                       |             |             |              |             |             |
|               | 10.4         | 16.2        | 18.6        | NS                    |             |             | NS           |             |             |
| Jan. 28, 1975 | 2            | 3           | 1           |                       |             |             | 1            | 3           | 2           |
|               | 8.7          | 10.9        | 16.8        | NS                    |             |             | <u>35.0</u>  | <u>37.6</u> | <u>38.9</u> |
| Feb. 18, 1975 | 2            | 3           | 1           |                       |             |             | 1            | 2           | 3           |
|               | <u>8.4</u>   | <u>9.5</u>  | <u>16.7</u> | NS                    |             |             | <u>32.2</u>  | <u>36.9</u> | <u>37.0</u> |

\*Steel and Torrie, 1960.

†Means of the three stations ( $n = 14$ ) ranked from low to high for each sampling date. Underlined means are not significantly different. NS indicates no significant differences among means.

## Bartow

Scallop mortality after 1 month of exposure was found to be directly related to proximity of the effluent. The highest death rate, which occurred at the most thermally altered station (B), was over six times that occurring at the ambient station (A). Mortality at the least thermally altered experimental station (D) was similar to that occurring at station A (Table 1).

Noticeable trends between temperature and mean oocyte diameter were found at the four Bartow stations (Fig. 7). At station A oocyte diameter steadily increased throughout the study period. By the end of the experimental period many of the oocytes were in the cytoplasmic growth stage or were entering vitellogenesis. Few of the oocytes from station D reached vitellogenesis although early increase in size did occur. There was little change in oocyte diameter from stations B or C, and many of the oocytes were resorbed.

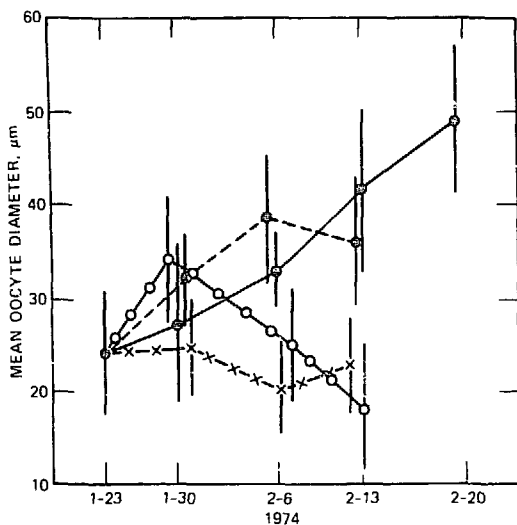


FIG. 7 Mean diameter of oocytes ( $\mu\text{m}$ ) from the four stations (A, —○—; B, x—x; C, —○—; and D, ---x---) at the Bartow site. Thirty oocytes from each of five scallops were measured once a week at each station to determine mean diameter. Vertical lines represent one standard deviation from the mean.

## DISCUSSION

The effects observed may be attributed to the "thermal effluent," which includes increased chlorine, heavy metals, and total suspended load, as well as increased temperature (Roosenburg, 1969). We believe that the increased temperature and total suspended load of the thermal effluents studied were major factors responsible for the effects observed. Previous investigations have shown the importance of temperature in the reproductive physiology of the bay scallop (Sastri, 1966; 1968; 1970; Sastri and Blake, 1971), which is at the southern limit of its geographic range

in the two study areas (Abbott, 1974). Kennedy and Mihursky (1971) have shown that *Mya arenaria* and several other bivalves living near their southern range limits were less tolerant of temperature increases than species not near their southern limits.

Increased mortality has been used by many investigators to indicate deleterious environmental change. Sublethal effects can be just as devastating to a population, however, especially if the reproductive potential is lowered (Lehmkuhl, 1974). Our study takes into account these more subtle long-term indicators of the success of a population.

The mortality that occurred at the more thermally altered Bartow stations can be attributed to the thermal effluent; the benthic community of the Bartow study area has been radically altered by the thermal effluent (Blake, Doyle, and Pyle, 1974). The increase in temperature may have put metabolic demands on the animal that it was unable to meet, at least during that particular stage of its life cycle (growth and gametogenesis). Gennette and Morey (1971) have shown that gravid oysters are more susceptible to high temperatures than oysters in the vegetative reproductive stage. Exposure for 1 month to a 1 to 3°C increase in temperature at Bartow yielded mortality similar to that found at ambient temperatures. Longer exposure (5 months) to a 1 to 2°C increase in temperature at Anclote produced mortality greater than that found at ambient temperatures. These results emphasize the need for long-term exposure to experimental conditions. Increased total suspended load can adversely affect a suspension feeder in two basic ways: (1) by clogging the feeding mechanisms (Jorgensen, 1966) and (2) by increasing metabolic demand because of greater sorting activity of the palps. One or both of these factors could have been operating in conjunction with elevated temperature at the Anclote site to produce the increased mortality. The effects of slight elevation in temperature might have been insufficient had the increased total suspended load and other effluent effects been absent.

Sastri (1970) has shown variation in reproductive response of the bay scallop to temperature with varying latitude. Although northern populations spawn at or near the summer temperature maximum, southern populations spawn during the fall decrease in temperature. Populations in the Anclote study area were previously observed spawning during the fall decrease in temperature (Blake et al., 1973), and during this study we observed spawning well into the winter. Many invertebrates have extended spawning periods of lower amplitude at their southern (more equatorial) range limit (Booolootian, 1966; Giese, 1959). Decreasing an already lowered amplitude of reproduction may be deleterious to reproductive success, especially if the species spawns only once or twice a year, as the bay scallop does. This decrease, combined with reduced spawning duration, may lower reproductive success even further. Knowledge of reproductive success can be obtained only by observing the population dynamics of a species in a thermally altered area over several years.

Sastry (1966; 1970) found that the digestive gland of the scallop supplies necessary food reserves for rapid gonad growth. Results from our study indicate that the muscle may be a substitute reserve energy source during early, rapid gonad growth. The higher muscle indexes observed at both Anclote stations 2 and 3 show a direct relationship to the greater food supply available at these stations; this indicates that the muscle may be an energy-storage area.

Work by Sastry and Blake (1971) indicates that, when food is abundant, reliance on the digestive-gland reserves for gonad growth is reduced. Although the source of reserve energy is different, our study also indicates that, if food is abundant, reliance on energy reserves is decreased.

Investigations by Sastry (1968) and Sastry and Blake (1971) have shown that the bay scallop requires food and a certain minimum temperature to initiate oocyte growth. Sastry (1970) found that oocytes from scallops in Beaufort, N. C., developed to vitellogenesis in May but were then resorbed. The temperature ranges given indicate that the resorption occurred when the temperature increased to 25°C or more. Our results indicate that recently spawned scallops exposed to temperatures above the necessary minimum initiate oocyte development. Scallops exposed to temperatures at or above 26°C did not begin oocyte growth, however, or after growth initiation the oocytes degenerated. These results also indicate that scallops exposed to 25°C or greater after completion of the normal reproductive cycle resorb the remaining oocytes. The bay scallop apparently requires a range of temperature to complete the reproductive cycle. Once oocyte growth is initiated, temperatures must remain below a maximum of 25°C for completion of the reproductive cycle.

We observed a range of responses—resorption of oocytes, decreased gonad index amplitude, delay of spawning, and mortality. The most severe responses, mortality and reproductive degeneration (resorption of oocytes), occurred after brief exposure (1 month) to a thermal effluent with a 4 to 5°C increase in temperature over ambient or after lengthy exposure (5 months) to a thermal effluent with a 1 to 2°C increase in temperature over ambient. Therefore, we cannot dismiss deleterious effects of thermal effluents if only short-term observations have been made.

## CONCLUSIONS

1. Mortality is greater in thermally altered areas than in ambient areas.

2. Increased mortality occurred during lengthy exposure to a thermal effluent with an elevation in temperature of 1 to 2°C over ambient and increased total suspended load.

3. Reduction in reproductive amplitude occurred due to exposure to a thermal effluent with an elevation in temperature of 1 to 2°C over ambient and a concurrent elevation in total suspended load.

4. Oocytes are resorbed and reproduction ceases if scallops are continuously exposed to temperatures exceeding 25°C during the early cytoplasmic growth stage.

## ACKNOWLEDGMENT

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# Effects of the Palisades Nuclear Power Plant on Lake Michigan

Robert S. Benda  
John Gulvas

## ABSTRACT

An 18-month study (May 1972 to October 1973) was undertaken to determine the effects of once-through cooling water from the Palisades nuclear power plant on the biota of Lake Michigan. The Palisades facility [821 Mw(e)] began operations in early 1972. The cooling-water intake is submerged 1006 m offshore at a minimum depth of about 7.6 m. The heated discharge enters the lake directly at the shoreline. In the original once-through cooling system of the plant, the total flow rate was about 30,686 liters/sec, with a maximum  $\Delta T$  of 13.9°C. The size range of 36 surface plumes was 6.9 to 352.1 ha, with an average size of 93.7 ha at the 1.7°C isotherm.

From May 16, 1972, to Oct. 25, 1973, a total of 651,712 fish weighing 19,842.2 kg were impinged on the traveling screens. Alewife (*Alosa pseudoharengus*), slimy sculpin (*Cottus cognatus*), spottail shiner (*Notropis hudsonius*), and yellow perch (*Perca flavescens*) made up 58.6, 27.5, 7.2, and 4.4%, respectively, of the total number. Coregonids and salmonids and smelt (*Osmerus mordax*) made up 0.2 and 1.0% of the total, respectively. The fish appeared seasonally on the screens, with none of the 37 recorded species being collected regularly. In addition, 4768 crayfish (*Orconectes propinquus*) were counted, mostly during late spring and early summer. No significant change in the abundance of prominent species in lake populations occurred after the introduction of the thermal discharge, but the spawning period for one-third of the yellow perch collected was advanced by approximately 3 weeks in 1973.

Fifty-two sets of  $^{14}\text{C}$  samples were collected on 25 separate days. The average loss in primary productivity for a heated discharge was 32.7% and for a nonheated discharge, 17.9%, indicating that mechanical stress caused approximately one-half of the decrease.

Live-to-dead ratios were established for groups of zooplankters in 119 sets of samples for the intake and discharge. All groups studied showed that the discharge samples usually had a higher percentage of dead organisms than did the intake samples. Below a discharge temperature of 34°C, the average loss of the discharge over the intake was 6.1 to 14.9% for all groups. However, once discharge

temperatures exceeded 34°C, losses were 75 to 90%, indicating that lethal levels had been reached.

Seining, gill netting, and trawling data revealed that the heated water attracted fish species (including salmonids) according to their temperature preference. This attractant provided an active sport fishery during several weeks of the year.

In general, the operation of the Palisades plant with the once-through cooling system did not have a significant impact on the lake.

This paper summarizes the results of 18 months of postoperational study conducted by the authors from May 1972 to October 1973 as part of a 7-year monitoring program to determine the effects of the Palisades nuclear generating plant on Lake Michigan (Consumers Power Company, 1975).

Consumers Power Company began construction of the Palisades nuclear power plant in March 1967. An environmental survey program was initiated in 1968 in response to concerns expressed about the effects of the discharge of heated effluents by various regulatory agencies and with regard to standards being developed by the Michigan Water Resources Commission. Consumers Power Company was also under commitment that, should the effects of the discharge on the aquatic biota prove to be significant, corrective measures would be taken. Operations were begun in December 1971 at 20% of rated power and gradually increased to 60% in August 1972, 85% in January 1973, and 100% of full power [700 Mw(e)] in April 1973. The plant was shut down because of various operational problems in August 1973 and remained off-line for the rest of the study period. The construction of cooling towers for the plant provided a closed-cycle cooling system instead of once-through cooling after 1973. The cooling towers were constructed to satisfy the intervenors' concern for possible environmental damage and were built during the study.

Benda: Natural Science Division, Aquinas College, Grand Rapids/Michigan.  
Gulvas: Consumers Power Company, Jackson, Michigan.

The program objectives were:

1. Preoperational surveys of the benthos, plankton, and fish to obtain a knowledge of the state of the lake, both in the vicinity of the plant and in areas away from any possible influence of the thermal plume.

2. Postoperational mapping of the thermal plumes.

3. Postoperational surveys of the benthos, plankton, and fish to determine the thermal influence, if any, on the biota.

The preoperational surveys began in May 1968 and formed a base line against which postoperational data were compared to determine effects of the heated discharge on the biota of the area. An additional study was begun in 1972 to determine whether psammolittoral organisms were significantly affected by elevated temperatures during times of onshore winds and currents. Psammon were sampled during June and October of 1972 and 1973. The T. W. Beak Company and the Institute for Fisheries Research of Michigan conducted aspects of the 7-year monitoring program.

## DESCRIPTION OF THE PLANT AND STUDY AREA

The 197-ha plant site is located in Van Buren County on the eastern shore of Lake Michigan, approximately 7 km south of South Haven and 26 km north of Benton Harbor and St. Joseph.

The Palisades facility has a rated capacity of 700 Mw(e) with an ultimate output of up to 821 Mw(e). The plant uses a pressurized-water-reactor system, with steam originally being condensed by means of a once-through condenser cooling system using Lake Michigan water to dissipate the waste heat. The waste heat at rated capacity increased the cooling-water temperature a maximum of 19°C above ambient at the intake. Cooling water is taken from Lake Michigan through an intake crib located 7.6 m below the lake's surface, 1.8 m from the lake bottom, and 1006 m from the shoreline. The crib is a 17.4-m-wide, 17.4-m-long, 3.7-m-high box with a steel plate for its top and 0.05-m vertical bars spaced 0.25 m apart around the sides. Water flows horizontally between the vertical bars at a measured velocity of 0.21 to 0.29 m/sec and subsequently is diverted to a 3.4-m intake pipe rated at 2.9 m/sec at full-flow conditions.

The heated discharge enters Lake Michigan directly at the shoreline through two 62-m structures extending into the lake. The total flow rate through the original once-through cooling system was about 30,686 liters/sec, with an average discharge current of 58.5 cm/sec (range 38.9 to 72.3 cm/sec). This cooling system was used until the mechanical draft evaporative cooling towers were completed in early 1974. These towers convert the circulating-water system to essentially closed-cycle cooling with a blowdown of 98.5 liters/sec discharged at the shoreline to the lake. This blowdown, when diluted by 4546 liters/sec

of lake water, is no more than 3°C above ambient lake temperature.

The shoreline at the Palisades plant is regular along the length of the study area without large indentations. It is of the basic "high sand dunes" type, 30 ft or higher, described by Carlson (1971) and subject to erosion by wind and water. Lake currents are reported to flow parallel to the shoreline 54% of the time because of southwest or northwest winds. At other times the flow is offshore or calm (Hough, Davis, and Keeley, 1967).

The offshore area is predominantly sand and small gravel and is thus unstable during most of the year, leaving little possibility for plants to attach or take root. Analysis of the sediments indicated that they consisted mostly of a fine to medium sand with occasional small amounts of mud and detritus. A typical scuba-diving transect showed the near shore area to be a heavy sandbar that moved frequently as wave, wind, and current action changed. Considerable amounts of organic detritus were observed in the shallow areas and out to the 7.5-m level. Beyond this level the sand was washed clean. The whole bottom contour was made up of wave ripple marks, varying in height from 2.5 to 8 cm. There was an extensive gravel bar, 4 to 6 m wide, at the 6- to 8-m depth that ran the entire length of the survey area parallel to the shore.

The immediate discharge area (approximately 6 m from the point of discharge) was the only area that showed any observable change. The velocity of the discharge water washed the bottom clean of sand, and only large gravel and rocks remained.

## METHODS

Periodic thermal plume measurements gave an indication of the area of lake surface, depth, and shoreline area that might be affected by temperature changes. A total of 21 plumes were measured by survey personnel during the study, and an additional 15 were measured by Argonne National Laboratory personnel. The Argonne personnel used on-board computers for plume measurements, and the authors used a temperature probe. Surface temperature measurements were taken every 50 m along predetermined transect lines. Directional lights on shore enabled the boat to stay on line, and distances were measured using a Tellurometer radar unit. Surface temperatures were taken until the 1.7°C isotherm from ambient was reached. Temperature profiles were also recorded along each transect to establish the depth of the plume.

The plant intake water flowed through 0.95-cm mesh traveling screens, and all fish impinged on the Rex traveling screens were sluiced into a collection basket, identified, counted, weighed, and measured on a daily basis. Representative samples were sometimes examined, where individual counts were impractical. The screens were usually run twice each 24-hour period, except during heavy runs of fish when they were run continuously.

In 1973, 23 gill-net sets were made using a 38-m variable-mesh gill net. These sets were made at the mouth

of the discharge and in the immediate vicinity of it, under both heated and nonheated discharge conditions.

Live-to-dead ratios were established for zooplankton organisms in the intake and discharge water by collecting surface-water samples from the intake bay and, approximately 90 sec later (calculated plant passage time), from the discharge at the shoreline. It was assumed that the intake and discharge samples represented the same water mass, differing only in respect to passage through the plant. Various amounts of water were poured through a No. 10 mesh Wisconsin plankton net. The net was submerged in a bucket of water to minimize physical stress to the organisms. The amount of water sampled varied directly with the time of year and the numbers of organisms present. More water was sampled in the winter than the summer to keep the numbers of organisms at a predetermined minimum of 100 per sample.

Originally the zooplankton were incubated for 4 hr in jars at intake temperature, then stained with 10 drops of 1% neutral red vital stain (Dressel, Heinle, and Grote, 1972; final concentration of stain was 1:120,000 in each jar), and incubated an additional 30 min. After the organisms were stained, they were removed by sieving and were washed in water to remove excess stain. The washed organisms were then preserved in acidic 5% formalin solution and frozen until time became available to sort and count them. Freezing prevented the leaching of stain by the preservative. In late spring 1973, both motility counts and stained motility counts were used to determine ratios and verify the results of the staining technique. This was done using zooplankton counting wheels divided into eight small compartments and counting live and dead organisms. The staining greatly facilitated the ease of distinguishing live from dead organisms by enabling investigators to view the movement of the internal organs rather than just wait for actual swimming motion.

## RESULTS AND DISCUSSIONS

### Plume Configuration

The temperature rise through the plant varied from less than 5.5°C during the period of 20% operation to 15.5°C at 60% operation with only one circulating pump operating. The maximum temperature rise with both pumps operating was 10°C during 60% operation and 13.5°C during 100% operation. Maximum discharge temperature measured in the discharge canal reached 38.5°C in July 1973. Plume behavior was affected by many factors, with wind and resultant lake conditions being dominant. On the basis of wind direction measurements and plume behavior, the following generalities can be made: (1) The majority of the plumes measured (26 of 36) traveled in a northerly direction, either northwest out into the lake or along the north shore, and (2) the winds associated with these plumes came predominantly from the south, either south or southwest. Occasionally plumes went in the opposite direction of the wind owing to lake movement established by previous prevailing winds.

The plume was deepest at the point of discharge and usually floated to the surface where it became thinner until it eventually became indistinguishable from the ambient lake water. The early spring plumes were usually deep for extended areas, reaching bottom over much of the identifiable area of influence. Three sinking plumes were encountered during the period of once-through cooling operation; on Apr. 6, 1972, and Mar. 8 and 23, 1973.

A problem throughout the study was to accurately determine ambient lake temperature conditions. The input of solar energy during the 2 to 3 hr required to chart a plume caused additional surface warming, which made plotting the edge and shoreward areas of the plume extremely difficult. The areas and respective lengths of thermal plumes that could be defined in the neighborhood of the 1.7°C level above ambient are tabulated in Table 1. The average size of the plumes mapped was 93.7 ha (range 6.9 to 352.1 ha) in surface area. Pack ice erosion was observed only in the immediate area of the discharge, and none was observed down the shoreline.

### Fish

The 57 species of fish sampled since the survey began are listed in Table 2. Twenty-five species were not collected in the preoperational studies, but two species collected in those studies have not been sampled since plant operation began.

Table 3 shows the total number, weight, and period of greatest impingement from May 16, 1972, to Oct. 25, 1973. A total of 651,712 fish weighing 19,842.2 kg were impinged on the screens. Of this total, four species, alewife, slimy sculpin, spottail shiner, and yellow perch, made up 58.6, 27.5, 7.2, and 4.4%, respectively, of the total number of fish, and 62.3, 5.8, 4.3, and 18.1, respectively, of the total weight of fish. Longnose sucker, burbot, and white sucker made up 3.7, 1.7, and 1.4%, respectively, of the total weight of fish, although their combined totals made up less than 0.003% of the total number of fish. The most numerous groups of fish appeared in the vicinity of the plant intake on a seasonal basis.

Except for yellow perch, the number of game fish impinged was small in comparison to nongame fish in the collections. The salmonids (steelhead, coho, chinook, lake trout, and brown trout) totaled 139 individuals weighing 72.6 kg, and coregonids (cisco, lake whitefish, round whitefish, and bloaters) totaled 551 individuals weighing 55.8 kg. The majority of coregonids were impinged in just a few days, and these usually followed an upwelling in the lake. The only other economically important fish were smelt, totaling 7565 individuals weighing 193.7 kg, and northern pike, totaling 18 individuals weighing 6.8 kg.

The condition of the impinged fish varied from excellent to battered, dead, or decomposed. More than 50% of the fish showed signs of physical damage, including descaling, hemorrhaging, cuts, and gouges. It is doubtful that many of the fish in this condition could have survived

TABLE 1 Summary of Thermal Plume Measurements at the Palisades Plant

| Date      | Approximate plant load, Mw(e) | Temperature of isotherm determined, °C | Isotherm increase above, °C | Approximate area, ha | Plume length, km (main axis) | Average depth of plume, m | Range of depth of plume, m | Direction of plume travel* |
|-----------|-------------------------------|----------------------------------------|-----------------------------|----------------------|------------------------------|---------------------------|----------------------------|----------------------------|
| 4/06/72   | 365                           | 3                                      | 0.44-0.78                   |                      | 0.6                          |                           |                            | SO                         |
| 6/06/72   | 415                           | 20.6                                   | 1.7-2.2                     | 80.9                 | 2.7                          | 0.6                       | 0.6-1.2                    | SO                         |
| 6/13/72†  | 420                           | 18                                     | 1.5-3.5                     | 86.2                 | 1.9                          | 0.5                       | 0.2-1.5                    | NS                         |
| 6/13/72†  | 420                           | 19                                     | 2.0-3.2                     | 126.3                | 3.7                          | 0.3                       | 0.2-1.5                    | NS                         |
| 6/19/72†  | 415                           | 17                                     | 1.4-2.1                     | 38.9                 | 2.4                          | 0.3                       | 0.2-1.5                    | NS                         |
| 6/20/72†  | 412                           | 17                                     | 2.3-2.5                     | 22.3                 | 0.8                          | 0.3                       | 0.2-1.5                    | NS-NO                      |
| 6/29/72†  | 412                           | 17                                     | 1.4-2.1                     | 35.6                 | 1.0                          | 0.5                       | 0.2-1.8                    | NS-NO                      |
| 6/20/72†  | 412                           | 19                                     | 3.0-3.2                     | 27.9                 | 1.0                          | 0.3                       | 0.2-0.8                    | NO                         |
| 6/20/72†  | 412                           | 19                                     | 1.8-2.3                     | 31.6                 | 1.3                          | 0.3                       | 0.3-0.5                    | NS-NO                      |
| 7/10/72   | 427                           | 21.6                                   | 2.2-2.8                     | 129.5                | 2.7                          | 0.6                       | 0.6-3.0                    | NS                         |
| 7/11/72   | 428                           | 21.1                                   | 2.2                         | 352.1                | 7.1                          | 0.6                       | 0.3-1.5                    | NS                         |
| 7/17/72†  | 185-415                       | 24                                     | 1.5-1.8                     | 43.3                 | 1.3                          | 0.8                       | 0.3-0.9                    | NS                         |
| 7/18/72†  | 415                           | 24                                     | 1.5-2.0                     | 138                  | 2.6                          | 0.9                       | 0.3-1.5                    | O                          |
| 7/19/72†  | 420                           | 25                                     | 0.5-1.5                     | 96.3                 | 1.1                          | 1.2                       | 0.5-1.5                    | O-NS                       |
| 7/21/72   | 426                           | 26.6                                   | 1.7-2.2                     | 8.1                  | 1.3                          | 0.6                       | 0.2-1.2                    | NS                         |
| 7/27/72   | 428                           | 13.4                                   | 2.2                         | 23.1                 | 0.6                          | 0.9                       | 0.2-1.5                    | O                          |
| 8/07/72†  | 432                           | 15                                     | 3.7                         | 28.7                 | 1.0                          | 2.4                       | 2.0-3.0                    | SS                         |
| 8/08/72†  | 433                           | 16                                     | 3.5                         | 65.2                 | 1.1                          | 2.4                       | 2.0-3.0                    | NS                         |
| 8/10/72   | 435                           | 18.9                                   | 0.6-1.7                     | 68.8                 | 4.0                          | 1.8                       | 1.5-2.7                    | NS                         |
| 8/25/72   | 418                           | 19.4                                   | 1.7-2.2                     | 29.1                 | 0.6                          | 0.9                       | 0.6-2.4                    | O                          |
| 10/10/72† | 434                           | 15                                     | 1.7-2.1                     | 34.4                 | 0.6                          | 0.8                       | 0.5-1.2                    | O                          |
| 10/11/72† | 435                           | 17                                     | 3.5                         | 58.7                 | 0.6                          | 2.4                       | 2.0-3.0                    | NS                         |
| 10/13/72† | 433                           | 14                                     | 1.3-1.4                     | 27.9                 | 0.8                          | 1.5                       | 0.9-2.0                    | NO                         |
| 3/08/73   | 581                           | 6.7                                    | 2.2-3.3                     | 263.1                | 2.4                          | 3.0                       | 1.2-6.1                    | NO-NS                      |
| 3/23/73   | 583                           | 6.1                                    | 1.7-3.3                     | 137.6                | 1.8                          | 3.0                       | 0.3-3.4                    | O-SS                       |
| 5/18/73   | 702                           | 13.4                                   | 1.1-2.2                     | 87.0                 | 1.9                          | 1.1                       | 0.9-1.2                    | NO                         |
| 6/05/73   | 728                           | 20.0                                   | 1.7-2.8                     | 214.5                | 2.4                          | 0.6                       | 0.3-0.9                    | NS                         |
| 6/06/73   | 726                           | 22.2                                   | 1.1-3.9                     | 105.2                | 1.6                          | 0.9                       | 0.3-1.8                    | NS                         |
| 6/12/73   | 719                           | 23.9                                   | 0.6-2.8                     | 226.6                | 3.2                          | 0.6                       | 0.3-0.9                    | NS                         |
| 6/20/73   | 717                           | 24.5                                   | 2.2-2.8                     | 40.5                 | 1.6                          | 0.9                       | 0.3-0.8                    | NO                         |
| 6/25/73   | 715                           | 21.1                                   | 2.2                         | 101.2                | 1.9                          | 0.6                       | 0.3-0.8                    | NS                         |
| 7/09/73   | 710                           | 22.8                                   | 1.7-3.3                     | 111.7                | 2.4                          | 0.8                       | 0.6-0.9                    | NO                         |
| 7/20/73   | 713                           | 23.9                                   | 1.7                         | 193                  | 4.0                          | 0.6                       | 0.3-1.1                    | NS                         |
| 7/24/73   | 716                           | 23.9                                   | 2.2                         | 6.9                  | 0.2                          | 0.6                       | 0.3-0.9                    | O                          |
| 7/31/73   | 714                           | 25.0                                   | 1.7-2.2                     | 144.1                | 2.6                          | 0.5                       | 0.3-0.6                    | NS                         |
| 8/02/73   | 715                           | 21.1                                   | 1.7                         | 22.7                 | 3.5                          |                           |                            | SS                         |

\*NS = north along shore; NO = north offshore; O = offshore; SO = south offshore; SS = south along shore.

†Survey was conducted by Argonne National Laboratory.

if they had been put back into the lake, considering the possibility of increased infection and predation pressures.

Overall, the results of gill netting, seining, and trawling were generally poor, and the small sample size prevented statistical comparisons among the areas. Several inferences concerning the inshore movement and distribution of fish can be made from combining data from all three collecting methods.

Table 4 shows the fish collected during each set of six 5.5°C temperature intervals. Lake trout and brown trout were collected up to 15.5°C, steelhead and coho up to 21°C, and yellow perch up to 38°C. Observations at the discharge area, when the temperature ranged from 10 to

15.5°C, revealed hundreds of trout and salmon swimming in the discharge, and numerous fishermen were observed fishing offshore of the discharge pilings. Data obtained from the seining and trawling collections showed species concentrated in the discharge area at temperatures elevated above ambient. Alewife, spottail shiners, centrarchids, and carp were consistently collected or observed more often in the discharge area than in areas that were at or close to ambient temperatures. This was especially evident in the Apr. 14, 1973, daytime seining collection, when the discharge area temperature was 15 to 18°C (11°C above ambient), and 155 coho, steelhead, and brown trout were collected. This number exceeded the total for these species in all other areas that day by a factor of 10.

TABLE 2 Species Collected by Seining, Trawling, Gill Netting, Observation, and Impingement on the Traveling Screens from 1968 of the Preoperational Studies to Oct. 25, 1973, of the Postoperational Studies\*

| Common name               | Scientific name                     | Pre-op.† | Common name            | Scientific name                 | Pre-op.† |
|---------------------------|-------------------------------------|----------|------------------------|---------------------------------|----------|
| Sea lamprey               | <i>Pteromyzon marinus</i>           |          | Buffalo fishes         | <i>Ictiobus</i> spp.            |          |
| Lake sturgeon             | <i>Acipenser fulvescens</i>         |          | Quillback carpsucker   | <i>Carpiodes cyprinus</i>       | X        |
| Bowfin                    | <i>Amia calva</i>                   |          | Northern redhorse      | <i>Moxostoma macrolepidotem</i> |          |
| Longnose gar              | <i>Lepisosteus osseus</i>           |          | Redhorse sucker        | <i>Moxostoma</i> spp.           | X        |
| Gizzard shad              | <i>Dorosoma cepedianum</i>          |          | Golden redhorse        | <i>Moxostoma erythrurum</i>     | X        |
| Alewife                   | <i>Alosa pseudoharengus</i>         | X        | White sucker           | <i>Catostomus commersoni</i>    | X        |
| Chinook salmon            | <i>Oncorhynchus tshawytscha</i>     | X        | Longnose sucker        | <i>Catostomus catostomus</i>    | X        |
| Coho salmon               | <i>Oncorhynchus kisutch</i>         | X        | Channel catfish        | <i>Ictalurus punctatus</i>      | X        |
| Brown trout               | <i>Salmo trutta</i>                 |          | Yellow bullhead        | <i>Ictalurus natalis</i>        |          |
| Rainbow trout (steelhead) | <i>Salmo gairdneri</i>              |          | Brown bullhead         | <i>Ictalurus nebulosus</i>      |          |
| Lake trout                | <i>Salvelinus namaycush</i>         | X        | Black bullhead         | <i>Ictalurus melas</i>          | X        |
| Lake whitefish            | <i>Coregonus clupeaformis</i>       |          | Trout perch            | <i>Percopsis omiscomaycus</i>   | X        |
| Lake herring (cisco)      | <i>Coregonus artedii</i>            | X        | Burbot                 | <i>Lota lota</i>                | X        |
| Bloater                   | <i>Coregonus hoyi</i>               | X        | Brook stickleback      | <i>Eucalia inconstans</i>       |          |
| Rouge whitefish           | <i>Prosopium cylindraceum</i>       | X        | Nine-spine stickleback | <i>Pungitius pungitius</i>      |          |
| American smelt            | <i>Osmerus mordax</i>               | X        | Slimy sculpin          | <i>Cottus cognathus</i>         | X        |
| Central mudminnow         | <i>Umbra limi</i>                   |          | Mottled sculpin        | <i>Cottus bairdi</i>            | X        |
| Northern pike             | <i>Esox lucius</i>                  | X        | Large-mouth bass       | <i>Micropterus saxoides</i>     |          |
| Grass pickerel‡           | <i>Esox americanus vermiculatus</i> | X        | Smallmouth bass        | <i>Micropterus dolomieu</i>     | X        |
| Goldfish                  | <i>Carassius auratus</i>            |          | Black crappie          | <i>Pomoxis nigromaculatus</i>   |          |
| Carp                      | <i>Cyprinus carpio</i>              | X        | White crappie          | <i>Pomoxis annularis</i>        |          |
| Longnose dace             | <i>Rhinichthys cataractae</i>       | X        | Rock bass              | <i>Ambloplites rupestris</i>    |          |
| Creek chub                | <i>Semotilus atromaculatus</i>      | X        | Green sunfish          | <i>Lepomis cyanellus</i>        |          |
| Lake chub                 | <i>Hybopsis plumbea</i>             |          | Bluegill               | <i>Lepomis macrochirus</i>      | X        |
| River chub                | <i>Hybopsis micropogon</i>          |          | Pumpkinseed            | <i>Lepomis gibbosus</i>         |          |
| Common shiner‡            | <i>Notropis cornutus</i>            | X        | Yellow perch           | <i>Perca flavescens</i>         | X        |
| Spotfin shiner            | <i>Notropis spilopterus</i>         | X        | Johnny darter          | <i>Etheostoma nigrum</i>        | X        |
| Emerald shiner            | <i>Notropis atherinoides</i>        | X        | Log perch              | <i>Percina caprodes</i>         |          |
| Spottail shiner           | <i>Notropis hudsonius</i>           | X        |                        |                                 |          |

\*Species list was compiled from the present study and from associated Michigan Department of Natural Resources pre- and postoperational studies.

†Twenty-five species collected in postoperational surveys were not collected in preoperational surveys.

‡Two species collected in preoperational surveys were not collected in postoperational surveys.

The main conclusions from the seining, trawling, and gill-netting data were that the discharge acted as an attractant to alewife, carp, centrarchids, trout, salmon and spottail shiners and that they abandoned the warmer areas as temperature preferences were exceeded. The gill-net data indicate that a majority of the fish occurring in the area of the Palisades plant inhabited the discharge area at some time during the year. Whether the attractant was temperature or current would be difficult to determine, but the discharge definitely attracted fish for whatever reason and provided an active fishery during different times of the year, usually during late fall, spring, and early summer. This is consistent with the results of studies by Benda and Proffitt (1974), Romberg et al. (1974), and Gammon (1973), which showed that fish species were attracted to thermal discharges; but, as the temperatures were elevated, they left the plume area.

## Crayfish

The crayfish (*Orconectes propinquus*) was the only benthic organism observed impinged on the traveling screens. A total of 4768 crayfish impinged, and the majority of these were alive when they were sluiced off the screens. Like the fish, *Orconectes propinquus* showed seasonal movement into the area of the intake. The time of greatest impingement was from May to August.

## Zooplankton

The most abundant and consistently appearing zooplankton groups were the cyclopoid and calanoid copepods and the *Bosmina* cladocerans. The copepods appeared in every sample, but the *Bosmina* appeared in large numbers primarily in the summer months. The dominant forms were *Bosmina longirostris*, *Diaptomis*, *Limnocalanus*, *Asplanchna* rotifers, and *Daphnia retrocurva*.

TABLE 3 Total Number\* of Fish Collected Daily from Traveling Screens at the Palisades Nuclear Power Plant from May 16, 1972, to Oct. 25, 1973

| Species                | Total number | Percent of total | Total weight, kg | Percent of total | Period of greatest impingement        |
|------------------------|--------------|------------------|------------------|------------------|---------------------------------------|
| Alewife                |              |                  |                  |                  |                                       |
| Young                  | 339,807      | 52.1             | 12,464           | 62.8             | October 1972, 1973                    |
| Adult                  | 42,565       | 6.5              |                  |                  | April 1973                            |
| Slimy sculpin          | 179,376      | 27.5             | 1,146            | 5.8              | March–April–May 1972, 1973            |
| Spottail shiner        | 46,897       | 7.2              | 854.9            | 4.3              | October–April 1973; December 1972†    |
| Yellow perch           |              |                  |                  |                  |                                       |
| Adult                  | 27,330       | 4.2              | 3,589.6          | 18.1             | October–December (esp. December) 1972 |
| Young                  | 1,084        | †                |                  |                  | † June 1972, 1973                     |
| Smelt                  |              |                  |                  |                  |                                       |
| Adult                  | 6,691        | †                | 193.7            | 1.0              | April–May 1973                        |
| Young                  | 874          | †                |                  |                  | June–August 1973                      |
| Trout–perch            | 3,485        | †                | 62.6             | †                | May 1972; April–June 1973             |
| Longnose sucker        | 1,294        | †                | 732.4            | 3.7              | June 1973                             |
| Bleater                | 519          | †                | 46.3             | †                | May–August 1973, 1973                 |
| Burbot                 | 493          | †                | 344.7            | 1.7              | December–March 1973                   |
| Nine-spine stickleback | 409          | †                | 2.3              | †                | April–June 1972, 1973                 |
| White sucker           | 395          | †                | 283.4            | 1.4              | December–January 1973                 |
| Channel catfish        | 167          | †                | 20               | †                | December–January 1973                 |
| Lake trout             | 81           | †                | 41.3             | †                | January–March 1973                    |
| Black bullhead         | 76           | †                | 3.2              | †                | April–June 1972                       |
| Chinook salmon         |              |                  |                  |                  |                                       |
| Young                  | 28           | †                | 12.2             | †                | June–July 1973                        |
| Lake whitefish         | 28           | †                | 7.7              | †                | August 1973                           |
| Johnny darter          | 27           | †                | 0.4              | †                | May 1972, 1973                        |
| Coho salmon            | 24           | †                | 9.5              | †                | May–June 1972, 1973                   |
| Northern pike          | 18           | †                | 6.8              | †                | October 1973                          |
| Total                  | 651,712      |                  | 19,842.2         |                  |                                       |

\*An additional 18 species were represented by fewer than 10 individuals each.

†Indicates less than 1% of the total.

TABLE 4 Number of Fish Collected in a 38-m Variable Gill-Net Set at or Near the Mouth of the Discharge, 1973

| Species         | Temperature range, °C |      |           |       |       |       |
|-----------------|-----------------------|------|-----------|-------|-------|-------|
|                 | 0.6–4.5               | 5–10 | 10.5–15.5 | 16–21 | 22–27 | 33–38 |
| Gizzard shad    |                       |      | 3         |       |       |       |
| Alewife         |                       | 36   | 192       | 12    | 134   |       |
| Coho            |                       |      |           | 2     |       |       |
| Brown trout     |                       | 3    | 13        |       |       |       |
| Lake trout      |                       | 7    | 9         |       |       |       |
| Steelhead       | 3                     | 5    | 18        | 6     |       |       |
| Northern pike   |                       |      |           | 1     |       |       |
| Smelt           |                       | 1    | 1         |       |       |       |
| Carp            |                       |      | 5         | 1     |       | 2     |
| Spottail shiner |                       | 3    | 14        |       |       |       |
| Channel catfish |                       |      |           |       |       | 1     |
| White sucker    |                       | 7    | 23        | 2     | 3     |       |
| Longnose sucker |                       |      |           |       | 4     |       |
| Trout–perch     |                       |      | 1         |       |       |       |
| Burbot          | 2                     |      |           |       |       |       |
| Slimy sculpin   |                       | 1    | 2         | 5     |       |       |
| Perch           |                       | 4    |           | 3     | 61    | 39    |
| Bluegill        |                       |      |           |       |       | 1     |
| Black crappie   |                       | 1    |           |       |       |       |
| Total           | 5                     | 68   | 281       | 32    | 202   | 43    |

Live-to-dead ratios were established for calanoid and cyclopoid copepods, naupli larva, *Bosmina* sp. and *Daphnia* sp. cladocerans, and rotifers for intake samples and both heated and nonheated discharge samples. Percent mortalities were established for all samples containing 25 or more specimens of each group. A total of 119 separate samples were counted, 83 from a heated discharge and 36 from an unheated discharge from May 1972 to October 1973.

Owing to sampling variability, exact mortality rates of zooplankton organisms were difficult to establish. In many instances the discharge samples showed lower ratios of dead organisms than did the intake samples taken 90 sec earlier. It should be noted that, once the discharge temperature reached 32 to 38°C, the percent mortality rates surpassed 90% in most groups. This indicated that the upper lethal temperature level of the zooplankton was reached and exceeded in these samples. During the summer months (late June to August), the discharge temperatures could have exceeded 32 to 38°C daily, but, owing to plant shutdowns during 100% operation, this could not be determined. The majority of the samples, however, showed low percent differences between the intake and discharge samples. Table 5 shows the averages and ranges of all samples for all groups. On the basis of all the samples collected, an average mortality was calculated for the 119 intake samples, 36 nonheated discharge samples (in an attempt to establish mechanical damage), and 83 heated discharge samples. If the mortality rates for all the zooplankton groups are averaged for the nonheated discharge samples and the heated discharge samples, they would show that approximately 31% of the total mortality was caused by mechanical damage and that 69% was caused by the addition of heat.

The percent mortalities are lower than those found by Storr (1974) on Lake Ontario, but the high mortality rates coincide with his data after temperatures exceed 35°C.

### Phytoplankton

Details on phytoplankton have already been published (Benda, Gulvas, and Neal, 1975). Fifty-two sets of <sup>14</sup>C samples were collected on 25 separate days. If the samples are averaged collectively for the entire study period (using samples showing increases and decreases in discharge primary productivity over intake samples), but divided into effects of a heated discharge and a nonheated discharge, the average loss for the heated discharge was 32.7% and the average loss for the nonheated discharge was 17.9%, indicating that mechanical stress caused approximately one-half of the decrease in primary productivity.

### Scuba-Diving Observations

During the surveys of the lake bottom carried out by divers, no rooted aquatics were observed in the vicinity of the plant. The only algal growth was observed on many of the rocks in the discharge area. The discharge structure itself was covered with *Cladophora* as were many of the large rocks on the shore and many of the seawalls in the area, both within and outside the area of thermal influence. *Cladophora* on the discharge structure were observed to die off during both summers after the discharge temperatures exceeded 27°C.

Observations at the time of the diving surveys indicated that many fish inhabit the area of the intake crib and actively swim in and out of the structure. Numerous yellow perch, alewife, and longnose and white suckers were observed in the area of the crib. The riprap around the base

TABLE 5 Average Mortality for Groups of Zooplankton from Intake, Heated, and Nonheated Discharge Samples at Palisades Nuclear Plant, May 16, 1972—Oct. 25, 1974

|                                | n   | Copepoda |          |           |          |          |          | Cladocera |          |         |          | Rotifera |          |
|--------------------------------|-----|----------|----------|-----------|----------|----------|----------|-----------|----------|---------|----------|----------|----------|
|                                |     | Calanoid |          | Cyclopoid |          | Nauplius |          | Bosmina   |          | Daphnia |          | Rotifers |          |
|                                |     | Av., %   | Range, % | Av., %    | Range, % | Av., %   | Range, % | Av., %    | Range, % | Av., %  | Range, % | Av., %   | Range, % |
| Intake                         | 119 | 16.0     | 1-91     | 21.3      | 1-60     | 20.9     | 0-79     | 20.6      | 0-83     | 26.7    | 0-100    | 8.1      | 0-35     |
| Heated discharge               | 83  | 30.9     | 3-93     | 28.7      | 4-83     | 33.2     | 0-80     | 26.7      | 0-73     | 33.5    | 0-91     | 20.3     | 3-70     |
| Nonheated discharge            | 36  | 21.5     | 8-46     | 27.0      | 12-66    | 18.5     | 5-43     | 25.2      | 8-67     | 25.3    | 0-67     | 14.7     | 8-20     |
| Percent difference             |     |          |          |           |          |          |          |           |          |         |          |          |          |
| Intake and heated discharge    |     | 14.9     |          | 7.4       |          | 12.3     |          | 6.1       |          | 6.8     |          | 12.2     |          |
| Intake and nonheated discharge |     | 5.5      |          | 5.7       |          | -2.4*    |          | 4.6       |          | -1.4*   |          | 6.6      |          |

\*The average percent mortality was higher in intake samples than in nonheated discharge samples.



of the crib provided habitat for crayfish, johnny darter, and slimy sculpin, which accounts for the presence of these organisms in the screen counts.

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# Effects of Power-Plant Operation on the Biota of a Thermal Discharge Channel

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## ABSTRACT

Phytoplankton productivity and bacterial heterotrophic uptake potential for glucose were measured monthly for 4 years at the James Stuart Generating Station on the Ohio River. Over a 3-year period, the plant opened four new units and a cooling tower. The average annual  $\Delta T$  was 8.5, 15.0, 11.2, and 9.5°C for the years 1971 to 1974. The response of phytoplankton production and bacterial heterotrophy as a function of ambient temperatures and the  $\Delta T$  were determined. Temperatures greater than 34°C were inhibitory regardless of ambient temperature. Any combination of  $\Delta T$  and ambient temperature not exceeding 25°C was usually stimulatory for algal production. The inhibition, where it occurred, was reduced as the discharge water moved down the 1.57-km channel to the Ohio River. Once the warm water was mixed with river water, primary production was stimulated 60% of the time by the mild temperature increase (2.4 to 3.6°C).

Mortality increased significantly with prolonged exposure of the zooplankton to elevated temperatures for 1 to 2 hr in the discharge channel after passing through the plant. Benthic and drift invertebrates were eliminated from the discharge channel in the summer and autumn of 1973 when temperatures exceeded 40°C; however, both returned in 1974 and persisted throughout the summer, when temperatures did not exceed 37°C. The release of once-through condenser cooling water to a discharge channel allowed an analysis of the time-delayed effects of temperature elevation that simply cannot be resolved when the heated waters are returned directly to the river.

Electric-power generating stations frequently disperse waste heat to the environment by adding it to surface waters (Parker and Krenkel, 1970). Cooling water is heated in the condenser tubes and discharged either directly into the river or lake from which it was taken or into a channel system or pond for cooling before release back into a natural water system. This paper concerns the effects of retention of entrained and resident organisms in a 1.57-km discharge canal at the James Stuart Station at Aberdeen, Ohio.

We have summarized some aspects of 4 years of year-round sampling of physical and biological parameters

associated with operation of the Stuart Station, the largest generating plant on the Ohio River (2400-Mw capacity). We have examined the effect of temperature elevation on algal and bacterial metabolism and on zooplankton and benthic invertebrate survival in the discharge canal. In particular, we have looked at the seasonal variation and the sensitivity of these biological measurements to year-to-year variability in plant operation.

## SITE

The James Stuart Generating Station began operation with one 600-Mw unit, using once-through condenser cooling, in October 1970. A second unit began operation in May 1971; a third in March 1972; and a fourth, with a natural-draught cooling tower, in early 1974. Each unit is equipped with two pumps that have a combined pumping capacity of 855 m<sup>3</sup>/min. The heated effluent is discharged after aeration into the enlarged channel of a small stream. The channel, 1.57 km long, empties into the Ohio River through a weir that had a 6.4-m opening from 1970 to 1973 and a 19.2-m opening in 1974. The plume of heated water in the Ohio River was usually confined to the top meter of water and did not extend more than half the distance across the river. The Ohio River is 30 m deep at this point, the location of an old navigational lock and dam. The plant uses an Amertrap system for cleaning its condenser tubes, rather than chlorine, so that all effects on entrained organisms are either mechanical or thermal. Retention of the water and entrained organisms in the discharge channel for 45 to 90 min was accompanied by less than a 3°C drop in temperature on most days.

Four locations were sampled: station I was the control station, situated on the Ohio River upriver from the plant; station II was at the entry point of the heated-water outfall into the discharge channel, where the temperature elevation averaged from 9.2 to 15.6°C above ambient over 4 years; station III was located 30 m up the discharge channel from the weir and 1.57 km from station II; and station IV was

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situated in the Ohio River 300 m below the weir where the heated water mixed with the Ohio River proper. Changes in entrained organisms from station I to II, III, or IV reflect a gradient of decreasing temperature differential and increasing time of exposure to heated water.

## METHODS

Primary production rates of the phytoplankton were determined by the  $^{14}\text{C}$  method of Steemann-Nielsen (1952) as modified for the International Biological Program (Vollenweider, 1969). Routinely, duplicate light bottles (125 ml) and a single dark bottle were filled with water collected at a 1-m depth, injected with  $4\ \mu\text{Ci}$  of  $\text{NaH}^{14}\text{CO}_3$ , and resuspended *in situ* at 10-cm and 1-m depths for 3- to 5-hr incubations that included solar noon. At the end of the incubation period, we fixed the samples with Lugol's iodine, transported them directly to the laboratory, and filtered them on  $0.45\text{-}\mu\text{m}$  filters at a vacuum of 0.5 atm. Dried filters were counted for  $^{14}\text{C}$  assimilated in a toluene cocktail. We used an external standard ratio to correct for counting efficiency in a Packard scintillation spectrometer. By using the tables of Saunders, Trama, and Bachman (1962), we were able to calculate the available  $\text{CO}_2$  from the bicarbonate alkalinity, pH, and temperature, assuming equilibrium conditions.

To measure algal density and biomass, we used settled samples and an inverted-phase microscope after Utermohl (1958). We counted 10 to 25 fields at 400 power in each of two or three 12-ml sedimentation chambers.

Bacteria stained with acridine orange were enumerated by epifluorescent microscopy following the method of Francisco, Mah, and Rabin (1973). Bacterial heterotrophic uptake potential of glucose- $^{14}\text{C}$  ( $V_{\text{max}}$ ) was measured by the substrate-dilution technique of Wright and Hobbie (1966), uncorrected for respiration of  $^{14}\text{CO}_2$  (Hobbie and Crawford, 1968).

The viable bacteria were enumerated by counting colonies grown on 2 ml of total-count medium on 47-mm filters. The maximum number of colonies found after 3, 7, and 13 days at or near ambient temperature ( $\pm 2^\circ\text{C}$ ) were used for maximum colony-forming units.

Oxygen consumption of the planktonic community was measured by the loss rate of oxygen in darkened BOD (biochemical oxygen demand) bottles (300 ml) incubated *in situ* for 3 to 5 hr. Oxygen tension before and after the incubation was quantified by the azide-modified Winkler method (American Public Health Association, 1967).

For zooplankton entrainment mortality, eight weekly samples were taken from July through September 1974 at stations I, II, and III. On each date, three 10-liter Juday samples (20-mesh net) were taken at each station at a depth less than 1 m. Within 30 min of sampling, three 1-ml subsamples were drawn from each sample (concentrated to 100 ml) with a Hensen-Stempel pipet, placed in a Sedgwick-Rafter cell, and examined under a dissecting microscope. Dead zooplankton were distinguished from those living by heartbeat and movement. After treatment with

Lugol's iodine fixative, the total number of zooplankton were counted. The percent dead was calculated for each station from these data.

Benthic macroinvertebrates were collected with the use of an Ekman grab (15.2 by 15.2 cm) from a substrate composed of a grey silty-mud mixture with variable portions of sand and detritus. Duplicate bottom samples were taken from each site; each sample included two Ekman grabs washed through a U. S. Standard No. 30 sieve.

For drifting invertebrates, basket samplers filled with 5-cm-diameter ceramic balls were suspended at a 1-m depth in 1973. Duplicate 10-plate Dendy samplers were placed at station III in the summer of 1970 (before plant operation) and the summer of 1974 (after full operation had begun). Sampling frequency was about once a month for both.

Settled solids, determined as milliliters per liter in Imhoff cones, were high from October to April and uniformly low during the remainder of the year.

## RESULTS

### Primary Production

In 1974 the algal primary production in the Ohio River ranged from about  $3\ \text{mg C m}^{-2}\ \text{day}^{-1}$  in December to  $3220\ \text{mg C m}^{-2}\ \text{day}^{-1}$  in August. We used the BMD-02R software package, a stepwise multiple regression-partial correlation analysis, to correlate the observed seasonal variation with 10 independent variables. Temperature, total alkalinity, Secchi depth, discharge, and dissolved reactive phosphorus explained 72% of the variation in primary production (observed  $F = 5.87 > \text{critical } F_{0.05(5,12)} = 4.68$ ). Most important to primary production rates were positive linear correlations with temperature ( $r = 0.695$ ), Secchi depth as a measure of light penetration ( $r = 0.645$ ), maximum light intensity on that day ( $r = 0.484$ ), and negative correlations with discharge ( $r = -0.585$ ) and current [ $r = -0.425$  (not significant), critical  $r_{0.05(df\ 16)} = 0.468$ ].

Seasonally, algal cell volume in the Ohio River varied from  $0.45$  to  $9.7 \times 10^9\ \mu\text{m}^3/\text{liter}$ . No difference in numbers or biomass per liter was noticed between the river (I) and stations II, III, and IV (Hater, 1975). In order of importance for the amount that the residual variation was reduced, 72% of the variation in algal cell biomass was explained by river discharge, dissolved reactive phosphate, current velocity, and maximum daily insolation (observed  $F = 8.307 > \text{critical } F_{0.05(4,13)} = 5.87$ ). Algal cell biomass, more than the rate of production, was negatively correlated with discharge ( $r = -0.759$ ), river stage ( $r = -0.637$ ), and settled solids ( $r = -0.575$ ). Variation in river flow, which determined the washout rate of suspended algae and resuspended bottom silts, was most important in controlling the algal biomass present. Temperature, light, nutrient concentrations, and light penetration were most important for determining the rate of production of the existing algal biomass.

In comparing the primary production of the high-temperature station (II) (at a 10-cm depth) with that of the

control station (I) for the years 1971 to 1973, we found marked inhibition in the summer and some stimulation in the winter. Frequently station IV, with a modest 3°C  $\Delta T$ , was stimulated in comparison with station I. Data from more than 600 bottle experiments on 55 dates over 4 years were reduced by clumping the proportionate change in primary production of heated stations compared with station I by 5°C intervals for ambient temperature and for temperature increase ( $\Delta T$ ). The result (Fig. 1) shows the photosynthetic response surface for variation in ambient temperature and  $\Delta T$ , in which small temperature increases (<5°C) are stimulatory over a range of ambient temperatures up to 20 to 25°C and large  $\Delta T$ 's are stimulatory only at cold ambient temperatures (<5°C). This graph represents only the mean response; the average coefficient of variation for the points plotted was 57% of the mean. Thus on any given date the pattern may deviate significantly from this mean response. Variation caused by mechanical effects,

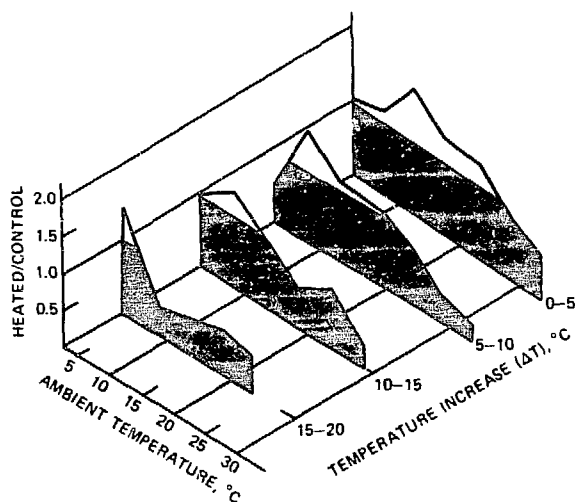


FIG. 1 (Primary production at heated stations)/(primary production at station I) as functions of ambient temperature and  $\Delta T$ , 1971–1974.

time lag, and seasonal variation in phytoplankton composition confounds the analysis. In general, any absolute temperature (ambient temperature +  $\Delta T$ ) less than 25°C caused stimulation of primary production. At ambient temperatures greater than 25°C, however, all temperature increases, on the average, caused inhibition of plankton production (Table 1).

The retention of entrained phytoplankton in the discharge channel allowed a significant recovery of primary production between stations II and III. We tested the null hypothesis that the difference between primary production rates at stations II and III was zero, using a group-comparison *t*-test. The observed  $t = 2.038 > \text{critical } t_{0.05(82)} = 2.00$ ; thus there appears to be a significant difference in the production rates at the two stations. Since no differences in species composition or biomass of algae were found between station I and III, either the algae at station III had become acclimated to the warmer water during the hour of retention in the channel or the apparent recovery was caused by a drop in temperature of about 3°C between station II and III. The average production rate at station II was 6.56 mg C m<sup>-3</sup> hr<sup>-1</sup> and at station III, 9.57 mg C m<sup>-3</sup> hr<sup>-1</sup>. The average difference between  $\Delta T$  at station II and that at station III was 2.2°C in 1971, 2.8°C in 1972, 0.3°C in 1973, and only 0.1°C in 1974 on the dates both stations were evaluated (Table 1). Thus recovery might come both from acclimation to changed temperature and from temperature decreases at station III.

To determine whether the observed photosynthetic rate changes at station II were caused by the temperature rise or by mechanical damage from passing through the condenser tubes and the aerators, we collected duplicate bottles of water from the control station (I), transferred them to hot-water station II or III, and incubated them at a depth of 10 cm after injecting them with NaH<sup>14</sup>CO<sub>3</sub>. On 16 dates, when the average primary production of entrained phytoplankton at station II was 7.96 mg C m<sup>-3</sup> hr<sup>-1</sup> (coefficient of variation = 146%), bottle-transferred phytoplankton at II had an average production of 10.9 mg C m<sup>-3</sup> hr<sup>-1</sup> (coefficient of variation = 177%). Although the transferred algae appeared to fix carbon at a higher rate, a group-comparison *t*-test showed that the difference be-

TABLE 1 Effect of Temperature Elevation on Primary Production of Plankton

| Year | Station I               |                                                        | Station II*             |                       | Station III*            |                       | Station IV*             |                       |
|------|-------------------------|--------------------------------------------------------|-------------------------|-----------------------|-------------------------|-----------------------|-------------------------|-----------------------|
|      | Average temperature, °C | Primary product, mg C m <sup>-3</sup> hr <sup>-1</sup> | Average $\Delta T$ , °C | Heated PP†/Control PP | Average $\Delta T$ , °C | Heated PP†/Control PP | Average $\Delta T$ , °C | Heated PP†/Control PP |
| 1971 | 14.7                    | 18.6                                                   | 11.2                    | 0.93 (53%)            | 9.0                     | 0.91 (47%)            | 2.3                     | 1.04 (35%)            |
| 1972 | 15.7                    | 11.1                                                   | 15.6                    | 0.88 (71%)            | 12.8                    | 0.89 (69%)            | 3.2                     | 1.11 (45%)            |
| 1973 | 16.7                    | 32.4                                                   | 11.8                    | 0.52 (75%)            | 11.5                    | 0.62 (75%)            | 2.5                     | 1.35 (38%)            |
| 1974 | 16.6                    | 69.6                                                   | 9.2                     | 1.01 (33%)‡           | 9.1                     | 0.89 (61%)            | 3.2                     | 1.55 (31%)            |

\*Numbers in parentheses indicate the percentage of dates on which this station was reduced more than 10% of station I (control).

†PP, primary production rate.

‡In 1974 station II includes only January to May.

tween the paired determinations was not significantly different from zero (observed  $t = 0.971 < \text{critical } t_{0.05(30)} = 2.042$ , NS). Thus stimulation or inhibition of primary production at higher temperature stations appeared to be primarily a temperature response.

For water samples that were transferred from the hot-water stations to the control station, the primary production was increased in every case to a rate higher than that at the hot-water stations. Since the number of cases was small ( $n = 6$ ), we cannot give, from in situ studies, more information about the form of the recovery time of algae from thermal shock. Preliminary laboratory experiments show that the time required for recovery after a thermal shock depends on the severity and length of the shock (Miller, Kallendorf, and Reed, 1972; Miller and Kallendorf, 1973).

The sensitivity of the monitored primary production rates to changes in operation of the plant from year to year is of interest. During most of 1971, the plant operated two units. In 1972 and 1973 three units were operated during the summer, and in 1974 four units were operated. During 1974 the condenser cooling function of one unit was handled by a natural-draught cooling water tower. The  $\Delta T$  increased from 1971 to 1972 and 1973; it dropped in 1974 because of the operation of the cooling tower, even though capacity was at an all-time high. The increasing and then decreasing inhibition of algal production at stations II and III over that period agree with the pattern of temperature increase and decrease from 1971 to 1974 (Table 1). If we examine the proportion of experimental dates on which algal production was reduced, we see that the same pattern emerges (Table 1, numbers in parentheses). At station IV the mild temperature increases were, on the average, stimulatory to algal production, and the percentage of dates on which inhibition was observed have been reduced compared with stations II and III.

### Bacteria

In the Ohio River waters, bacteria were present in modest numbers ranging from 2 to  $20 \times 10^5$  per milliliter. The highest numbers occurred in winter and spring when river current and discharge were greatest. The bacterial density measured by colony-forming units was only 60% less than the total acridine-orange bacterial count, in contrast to bacterial densities for most freshwater systems. Among the 14 genera isolated and identified, the most common genera were *Escherichia*, *Pseudomonas*, *Clostridium*, and *Bacillus* (Smiddy, 1974), forms typical of soil or fecal contaminants. Seasonally, bacterial density, as measured by viable counts on nutrient broth (tryptone-glucose-beef extract), correlated directly with turbidity as measured by settled solids ( $r = 0.54$ ,  $df = 14$ ,  $p < 0.05$ ) (Fig. 2), and inversely with Secchi depth ( $r = -0.6$ ,  $df = 23$ ,  $p < 0.01$ ).

In the discharge channel (station III), the density of viable bacteria and total counts were usually higher than at the control station (I); see Table 2.

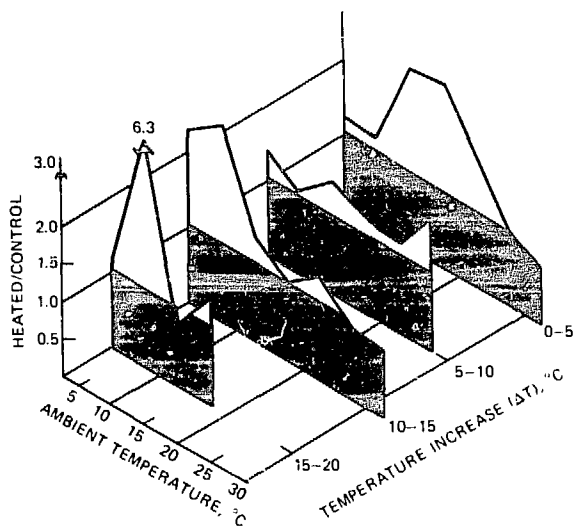


FIG. 2 (Heterotrophic uptake potential for glucose at heated stations)/(uptake potential at station I) as functions of ambient temperature and  $\Delta T$ , 1972–1973.

We used the kinetic bioassay with glucose- $^{14}\text{C}$  to determine the magnitude of the temperature-related effects on metabolism of bacteria in the channel. The ratio of  $V_{\text{max, II}}/V_{\text{max, I}}$  for 3 years clumped by ambient temperature and  $\Delta T$  showed, as expected, an increase in  $V_{\text{max}}$  at warmer temperatures (Fig. 2). In general, at most ambient temperatures a small  $\Delta T$  was stimulatory, the maximum stimulation occurring at a combination of ambient temperature and  $\Delta T$  giving an absolute temperature of 20 to 25°C (Table 3 and Fig. 2). The high  $\Delta T$  ( $>15^\circ\text{C}$ ) was stimulatory only at lower ambient temperatures. The maximum uptake rate occurred at an absolute temperature between 20 and 25°C.

Community metabolism was increased substantially 63% of the time (1970 to 1971) at the stations in the discharge channel. Oxygen depletion was not observed. In 1971 the average percentage of oxygen saturation in the heated-water channel was 101%, whereas the average percentage in the Ohio River was 82%. The increased temperature did reduce the oxygen concentration in the hot-water channel to as low as 4.5 mg  $\text{O}_2$ /liter, primarily as a function of reduced solubility at higher temperatures.

### Zooplankton

Zooplankton mortality, measured during the summer of 1974, was significantly increased by retention at high temperatures during the transit time from station II to station III. These stations had an average  $\Delta T$  of 7 to 8°C over the study dates, with a maximum  $\Delta T$  of 10°C. The ambient temperatures decreased seasonally from 28.5 to 20.5°C over that period. Zooplankton density was maximal during July (200 animals/liter) and declined through mid-September to less than 10 animals/liter. The decline was probably caused by the effect of declining temperature

TABLE 2 Comparison of Maximal Density of Bacterial Colony-Forming Units at the Temperature Nearest Field Temperatures at Stations I and III and the Ratio of Those Counts, III/I, as a Measure of Proportional Changes in Bacterial Density During 1973-1977.

| Date     | Station I                   |                                                                       | Station III                 |                                                                       | III/I proportion change |
|----------|-----------------------------|-----------------------------------------------------------------------|-----------------------------|-----------------------------------------------------------------------|-------------------------|
|          | River water temperature, °C | Bacterial density at nearest temperature, 10 <sup>3</sup> bacteria/ml | Canal water temperature, °C | Bacterial density at nearest temperature, 10 <sup>3</sup> bacteria/ml |                         |
| Apr. 15  | 9.5                         | 93.5                                                                  | 22.0                        | 78.6                                                                  | 0.84                    |
| Apr. 29  | 13.5                        | 119.0                                                                 | 24.6                        | 134.0                                                                 | 1.12                    |
| May 29   | 19.5                        | 35.0                                                                  | 32.0                        | 335.0                                                                 | 9.57                    |
| June 23  | 27.0                        | 12.0                                                                  | 40.3                        | 22.0                                                                  | 1.83                    |
| July 15  | 28.0                        | 38.0                                                                  | 39.5                        | 75.0                                                                  | 1.97                    |
| Aug. 12  | 25.0                        | 119.7                                                                 | 41.0                        | 172.7                                                                 | 1.44                    |
| Sept. 5  | 29.5                        | 1.7                                                                   | 44.0                        | 4.6                                                                   | 2.7                     |
| Sept. 30 | 25.0                        | 3.2                                                                   | 37.0                        | 1.2                                                                   | 0.37                    |
| Nov. 11  | 11.6                        | 0                                                                     | 21.9                        | 0.5                                                                   |                         |
| Dec. 16  | 7.0                         | 75.0                                                                  | 18.0                        | 123.0                                                                 | 1.64                    |
| Jan. 6   | 4.5                         | 53.0                                                                  | 12.0                        | 90.0                                                                  | 1.69                    |
| Jan. 27  | 8.0                         | 156.0                                                                 | 18.0                        | 266.0                                                                 | 1.70                    |
| Feb. 24  | 5.8                         | 193.5                                                                 | 14.0                        | 131.0                                                                 | 0.67                    |
| Mar. 17  | 7.2                         | 117.0                                                                 | 17.8                        | 126.0                                                                 | 1.08                    |

TABLE 3 Effect of Temperature Elevation on Heterotrophic Potential for Glucose (V<sub>max</sub>)

| Year      | Station I                      |                                                              | Station II* |                  |            | Station III* |                  |            | Station IV* |                  |            |
|-----------|--------------------------------|--------------------------------------------------------------|-------------|------------------|------------|--------------|------------------|------------|-------------|------------------|------------|
|           | Average annual temperature, °C | V <sub>max</sub> , µg C liter <sup>-1</sup> hr <sup>-1</sup> | ΔT, °C      | V <sub>max</sub> | III/I      | ΔT, °C       | V <sub>max</sub> | III/I      | ΔT, °C      | V <sub>max</sub> | IV/I       |
| 1970-1971 | 15.2                           | 4.02                                                         | 11.1        | 5.08             | 1.68 (66%) | 8.1          | 4.96             | 1.46 (57%) | 2.1         | 4.78             | 1.27 (60%) |
| 1972      | 16.4                           | 10.4                                                         | 15.1        | 24.3             | 1.83 (71%) | 12.4         | 12.8             | 1.37 (66%) | 2.5         | 12.0             | 1.48 (50%) |
| 1973      | 15.6                           | 3.82                                                         | 11.2        | 9.7              | 2.43 (90%) | 11.7         | 6.5              | 1.95 (60%) | 2.3         | 5.0              | 2.21 (50%) |

\*Numbers in parentheses are the percentage of the time that V<sub>max</sub> from the heated station was more than 10% greater than V<sub>max</sub> from the control station (I).

on zooplankton reproduction rates. In September, increased discharge effectively increased the washout rate. The total densities were not significantly different between stations. However, the average percentage of dead zooplankton (cladocera, copepods, and rotifers) increased from station I (16.1%) to station III (37.8%); see Fig. 3. A two-way analysis of variance to partition the variance in the percentage dead (arc sine transformation) between sample date and station showed that both main effects were significant (Table 4).

The mortality was taxon selective, being especially high in the larger cladocera, *Daphnia* sp. (Reed et al., 1974). In July, at station III, 100% of the cladocera were dead compared with 22.7% of the copepoda and 0% of the rotifers.

### Benthic and Drift Invertebrates

Before the plant began operation in 1970, Dendy plate samplers (929 cm<sup>2</sup>), exposed for 44 days during June and July at station III, had 74 organisms per trap, representing four taxa. In October, after 123 days of incubation, one unit of the plant had become operational, and the Dendy samplers contained an average of 66 organisms per sampler (Hatch and Gammon, 1973).

Benthic and drift invertebrates were eliminated from the discharge channel during the summer of 1973 as generating capacity and discharge temperatures rose. Basket traps yielded 54 to 107 individuals, of 10 taxa, per trap at station I and 12 to 71 individuals, of 9 taxa, per trap at station IV, but no macroorganisms were found on 25 traps in the canal at stations II and III.

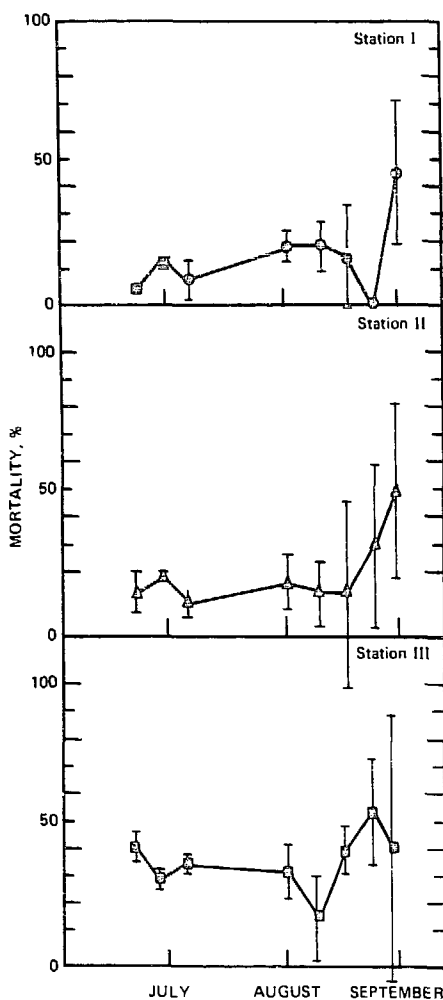


FIG. 3 Percentage of zooplankton found dead in the Ohio River (station I) and in the discharge channel (stations II and III) at the James Stuart Station during summer of 1974.

Drift invertebrates returned to the high-temperature stations during the summer of 1974. Summer densities from the Dendy samplers ranged from 13 to 151 individuals of 5 taxa at station I and from 33 to 74 individuals of 4 taxa at station IV. Samples from station III reached 17 to 21 individuals of 6 taxa by September to October (Reed et al., 1974).

Benthic invertebrate density and diversity for station I were similar during 1973 and 1974. The groups in order of numeric importance were oligochaetes, midge larvae (Chironomidae, Diptera), *Ch. borus* (Culicidae, Diptera), fingernail clams (Sphaeriidae, Mollusca), and damselfly nymphs (Agrionidae, Odonata). In the spring of 1973, stations I, III, and IV had very similar maximal densities, 2000 to 3000 organisms/m<sup>2</sup> (station II had 1000 organisms/m<sup>2</sup>). The density at stations I and IV fluctuated

TABLE 4 Analysis of Variance of Percentage of Zooplankton Found Dead at Stations I, II, and III on Eight Dates from July to September 1974 (Arc Sine Transformation)

|                  | df | Sum of squares | Mean square | F ratio | Significance level |
|------------------|----|----------------|-------------|---------|--------------------|
| Dates            | 7  | 5,130.40       | 732.91      | 2.93    | 0.025              |
| Stations         | 2  | 3,191.95       | 1,595.50    | 6.38    | 0.005              |
| Dates x stations | 14 | 4,284.83       | 306.05      | 1.22    | NS*                |
| Within subgroups | 48 | 11,992.86      | 249.85      |         |                    |

\*Not significant.

around that level as particular groups emerged; but almost all organisms at stations II and III had been eliminated by late June as the maximum temperature reached 40°C. By May 1974 the density at station III was higher than ever (about 5000 organisms/m<sup>2</sup>) and was not significantly different from densities at stations I and IV. In contrast to 1973, in 1974 the density at station III remained as high as that at station I throughout the summer.

## DISCUSSION

In a series of reports concerning the biological effects of thermal effluents in the Ohio River, few significant effects on entrained organisms except *Clad* have been identified. This is particularly true for plants which discharge cooling water directly into the river with only a short total retention time (Bryant et al., 1972; WAPORA, 1970; 1971). Given field-sampling variation, complete mortality of most taxa would not be detected downstream after a 10X dilution with river water within a few seconds of being injected into the river. The use of a discharge canal isolates the cooling waters and the entrained organisms while prolonging their retention at elevated temperatures (Rankin, Buck, and Forester, 1974). Since the present study has continued for 4 years as generating capacity increased, we were able to evaluate the sensitivity of several biotic groups to increasing  $\Delta T$ 's.

Phytoplankton production was stimulated by small temperature increases over a broad range of ambient temperatures. At cooler ambient temperatures, increases up to an absolute temperature of about 25°C were stimulatory, as others have also found (Morgan and Struss, 1969; Gurtz and Weiss, 1974; Butz et al., 1974). Higher absolute temperatures were frequently inhibitory or coupled with latent mortality from mechanical damage (Storr, 1974; Morgan and Struss, 1969). In this discharge channel the phytoplankton productivity recovered significantly as the water flowed toward the discharge weir.

The response of phytoplankton production to abrupt temperature increases is a sensitive index of the compara-

tive health of the algae (Fig. 1). For temperatures less than 25°C, such a set of relationships between ambient temperature, the temperature differential, and response of the planktonic algae would allow a plant manager to adjust the pumping rate through the cooling condensers to minimize damage to the algae. Although damage to the algae may not legally be an important impact by itself, we have shown them to be a sensitive indicator of thermal stress which correlates with physiological damage and mortality of other components of the food chain.

Bacteria are an important energy-processing component since river systems derive much of their annual energy budget from allochthonous inputs. Except for a short period in midsummer, it would appear from total counts that most bacteria in the Ohio River do not exist as isolated organisms but, rather, that they are attached to particulate material. Thus, when the amount of silt and detritus in the water is increased during high flow periods, bacterial density increases (Smiddy, 1974).

Bacterial density increased in the discharge channel at the Stuart Station compared with surface waters in the Ohio River. Either the bacteria were multiplying during the hour residency in the channel or the numbers in the discharge water were being increased by sediment resuspension or channel sloughing. In the Ohio River the volume of settleable solids increased with depth as the suspended solids settled out. Since the water intake withdraws water from the top to the bottom of the water column, the water column was mixed completely in the cooling-water intake and hence in the heated discharge (Table 2). As a result the average bacterial density would be increased at the outfall. In fact, the Secchi depth was shallower and the settled solids greater in the discharge channel except during the spring or winter when rainfall runoff increased the current velocity of the river more than that of the channel. It would appear, therefore, that bacterial density in the discharge channel was not changed by entrainment.

In this study the heterotrophic potential of the bacterial population declined between station II and station III, both in average  $V_{max}$  and in the proportion of times when temperature increases were stimulatory (Table 3). Uptake potential for bacteria using glucose was inhibited, on the average, at higher absolute temperatures than was algal production. However, the maximal stimulation over a wide range of ambient temperatures occurred at about the same absolute temperature, namely, at about 25°C (Figs. 1 and 2).

Zooplankton entrained in cooling waters suffered a significant increase in mortality as the water passed down the channel. Mortality was 5.1% higher at the discharge (station II) and 21.7% higher at the weir (station III) than in the river (station I). Apparently, increased numbers of animals were dying in the discharge channel, and the sensitivity of the animals was greatest in midsummer when the absolute temperature was the highest. Storr (1974) could not find a significant latent effect at two power plants. Carpenter, Peck, and Anderson (1973) found that

mortality of zooplankton at the condenser discharge was only 15%; however, the eventual mortality could be as high as 70%. Recovery of mobility up to 20% has been reported 4 hr from the time of discharge (Butz et al., 1974). In the present study, temperatures greater than 30°C had significant zooplankton mortalities associated with them, although it was impossible to separate mechanical damage from thermal damage with these data.

Benthic and drift invertebrate densities in the discharge channel were sensitive indicators of plant operation. The density of entrained organisms (algae, bacteria, and zooplankton) was not observed to change in the discharge channel. Rather, because of the rapid replacement of the organisms in the channel, only their state of health, as measured by a key metabolic rate (photosynthesis or heterotrophic uptake potential), or their physiological state (living or dead) could be detected. Benthic invertebrate and drift invertebrate populations, on the other hand, are relatively immobile and must endure all variation in physical conditions or be eliminated. Presence or absence, therefore, might be a valid indicator of year-to-year variation in thermal conditions in the channel.

Invertebrates found in the benthos and on suspended artificial substrates were eliminated from the discharge channel in 1973 when the maximum temperature reached 44.2°C and the average  $\Delta T$  was 11.2°C. In 1974, macroinvertebrates returned and remained throughout the summer. During this time the maximum temperature did not exceed 36.8°C, and the average  $\Delta T$  was 9.5°C. We have another empirical limit of perhaps 37°C above which stationary macroinvertebrates normally found in the Ohio River simply do not survive. Benda and Proffitt (1974) similarly reported a minimal biomass in drift samplers in Indiana at stations in a canal reaching a temperature of 39.4°C. Trembley (1960; cited by Parker and Krenkel, 1969, pp. III-46) reported the ability of organisms to repopulate the hottest stations below the Martins Creek Plant over the winter. Although drift and benthic invertebrates are eliminated by a critical high temperature in excess of 40°C, recolonization apparently takes less than 1 year.

We have summarized the extent of thermal effects on entrained microorganisms and macroinvertebrates which occur in the discharge canal of the Stuart Station. Each group has several attributes that allow us to determine the impact of temperature elevation or mechanical damage. The Stuart Station has little effect on these groups in the Ohio River as a whole. It is doubtful that many statistically significant effects could be isolated after the heated waters have been diluted (at least 5 : 1 in the 300 m required to reach station III). Except for the proportion of the total river flow that actually passes through the plant (never to exceed 15%), little ecological change in the Ohio River can be shown. With our expanding empirical data base, management of condenser pumping rates and operation of cooling towers could be designed to meet requirements for any acceptable level of biological change in the effluent.



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# Seasonal Abundance, Biomass, Diversity, and Trophic Structure of Fish in a Salt-Marsh Tidal Creek Affected by a Coastal Power Plant

Mark Homer

## ABSTRACT

Monthly measurements were made of the seasonal abundance, biomass, species diversity, and trophic composition of fish inhabiting the tidal creeks of salt marshes receiving thermal discharge near Crystal River, Fla. In the warm months (May through September 1974), mean abundance in the creek receiving thermal discharge was 0.46 individuals/m<sup>2</sup> and mean biomass was 2.2 g (preserved weight)/m<sup>2</sup>. In a control area, creek values in the warm months were 6.77 individuals/m<sup>2</sup> and 9.1 g (preserved weight)/m<sup>2</sup>, respectively. During the cold months (October 1974 through February 1975) there were 0.48 individuals/m<sup>2</sup> and 8.3 g (preserved weight)/m<sup>2</sup> in the discharge area and 0.58 individuals/m<sup>2</sup> and 7.4 g preserved weight/m<sup>2</sup> in the control area. In all months except May 1974 and February 1975, species diversity as species per 1000 individuals was higher in the control creek than in the discharge creek. No apparent differences in fish trophic structure were observed.

The landward area surrounding the Florida Power Corporation's Crystal River, Fla., power plant is dominated by *Juncus-Spartina* salt marshes. Marshes such as these are known to export large quantities of organic material (de la Cruz, 1973) to the adjacent estuarine bays, providing an important input of energy to the diverse food webs of the estuaries (Kemp, McKellar, and Homer, 1974). Tidal creeks function as avenues for the movement of materials between the salt-marsh areas and the shallow estuarine bays. Following the tides in and out of these creeks are many species of fish that use marsh areas for food or protection. This unique association of fish accounts for one type of movement of materials between the marshes and the bays.

Lindall (1973), Carr and Adams (1973), and others have noted that estuaries and their associated marshes serve as nurseries for as many as 75% of the important

commercial species. These species, along with the non-commercial species using estuaries, serve to maintain important food-web links between the marshes and the bays. These creek inhabitants upgrade the salt-marsh products into protein, some of which is then transported to the bay food webs or to the estuarine bird populations (Day et al., 1973).

Man's use of coastal zones as waste repositories changes the processes occurring in salt marshes and estuaries producing new associations. Few studies have examined the impact of thermal effluents on salt marshes or the inhabitants of tidal creeks. The present study compares quantitative data collected on salt-marsh fish in a thermally affected tidal creek with fish inhabiting creeks outside the range of the thermal plume. Preliminary data on seasonal abundance, biomass, species diversity, and trophic structure are presented here for a 10-month period, May 1974 through February 1975. Concurrent studies on the productivity of invertebrates, plants, and many other aspects of the affected estuaries were included in companion projects (Florida Power Corporation, 1974).

## SITE DESCRIPTION

The study site was located near the Florida Power Corporation Crystal River power plant on Florida's west central Gulf Coast, 112 km north of Tampa (Fig. 1). At the time of this study, two oil-burning generating units with a combined generating capacity of 897 Mw had been in operation for 7 years. An 856-Mw nuclear unit was still under construction. The power plant used a once-through cooling system, pumping in offshore seawater through a 12.5-km intake canal. This water is used as the condenser

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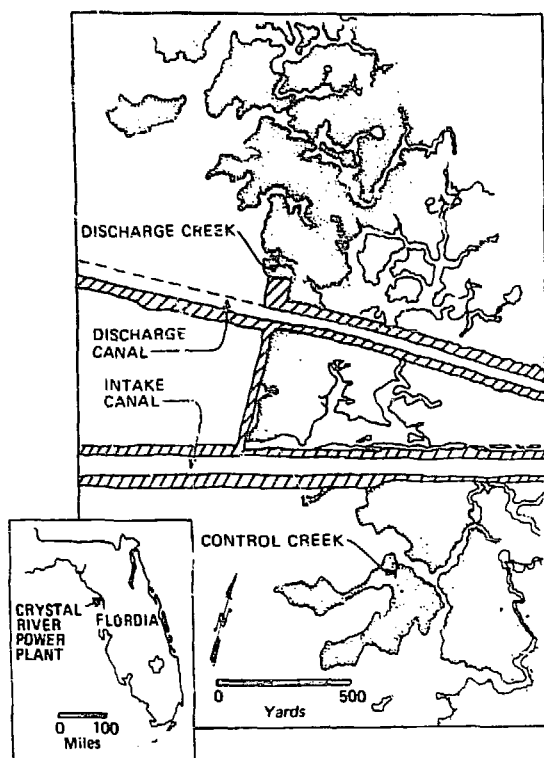


FIG. 1 Location of the two tidal-creek sampling areas.

coolant and is discharged into a 3.8-km canal that terminates near a shallow estuarine bay.

The influences of the power plant on adjacent marsh ecosystems include the discharge of thermal effluents and the alteration of hydrographic patterns and sedimentation processes. During a flood tide, the thermal discharge from the plant ( $6^{\circ}\text{C}$  maximum rise above ambient temperature) is pushed up onto the surrounding salt marsh north of the plant through tidal creeks (Klauswitz et al., 1974). The discharge canal, dredged through salt marsh, had disrupted the normal drainage patterns of water, leaving the marsh over an ebb tide (Cottrell, 1974).

The two tidal creeks discussed in this paper are similar in a number of ways. Both are small creeks, the intake creek had a drainage area of  $1560\text{ m}^2$ , and the discharge creek area was  $2090\text{ m}^2$ . Tides in both areas were semidiurnal, generally with unequal highs and lows. At slack high tide water depths varied from 1.2 to 1.7 m at the mouths of both creeks. With an average daily tidal range of 1 to 1.5 m, both creeks drained fairly completely at least once daily. A fringe of *Spartina alterniflora* extends some 2 to 3 m from both tidal creek banks into the marsh, where *Juncus roemerianus* becomes dominant. Alteration of drainage patterns associated with discharge and construction has caused extensive erosion, leaving numerous limestone outcroppings in the discharge creek (Cottrell, 1974). The

sediment base of the control creek is about 80 to 130 cm thick and is underlain by limestone. The use of high-salinity offshore water as coolant has caused the salinity of the water in the discharge creek to be generally 5 to 6‰ higher than that of the control creek.

## METHODS

Collections in two tidal creeks, one in the discharge area and one outside the influence of the plant were made monthly from May 1974 through February 1975.

Finfish and blue crabs (*Callinectes sapidus*) were collected with a modification of a block net technique described by Adams (1974). During a slack high tide, a 16-m-long by 2-m-high nylon net having a 1.6-mm bar mesh was stretched across the mouth of the creek from berm to berm. The lead line ( $\frac{3}{4}$ -in. case-hardened chain) was jammed into the sediment to prevent escapement under the net. The top of the net was hung on stakes approximately 2 ft above the waterline to prevent such species as mullet (*Mugil* sp.) and needlefish (*Strongylura* sp.) from jumping over the net. During the subsequent low tide, the stranded fish and crabs were collected by hand and dip net, returned to the laboratory, and preserved in a 10% Formalin solution within an hour following collection. Preserved wet weights were recorded to the nearest 0.1 g. Standard lengths were taken to the nearest 1 mm. Almost all individuals were weighed and measured, except for silversides (*Menidia beryllina*). Disk lengths were measured for the Atlantic stingray (*Dasyatis sabina*), and carapace widths were taken for blue crabs.

Standard surveying techniques were used to survey both creeks for surface areas. Temperature and salinity measurements were taken at the time of net setting, usually at 6 to 8 locations in the creek.

## RESULTS

### Temperature and Salinity

Shown in Fig. 2 are temperature and salinity measurements taken at the time of each collection. The temperature difference between the discharge and the control creek averaged  $5$  to  $6^{\circ}\text{C}$  during the summer months (May through September). During the fall and winter months (October through February), temperatures in the discharge creek averaged about  $1.5^{\circ}\text{C}$  higher than those in the control creek. Salinity in the control creek exhibited greater seasonal variation than in the discharge creek. Highest salinities ( $23$  to  $26\text{‰}$ ) were recorded during the fall months (September through November), whereas salinities ranged from  $15.9$  to  $21\text{‰}$  during the other months. Salinities were uniformly high in the discharge creek, averaging about  $27\text{‰}$ . Salinity differences between the two creeks ranged from no difference (October 1974) to  $10\text{‰}$  (February 1975) and averaged about  $5\text{‰}$  during the 10-month period. Young (1974), during a study of the marsh in the area surrounding the Crystal River power plant, recorded water temperatures and salinities in the general vicinity of the two creeks and noted similar seasonal patterns.

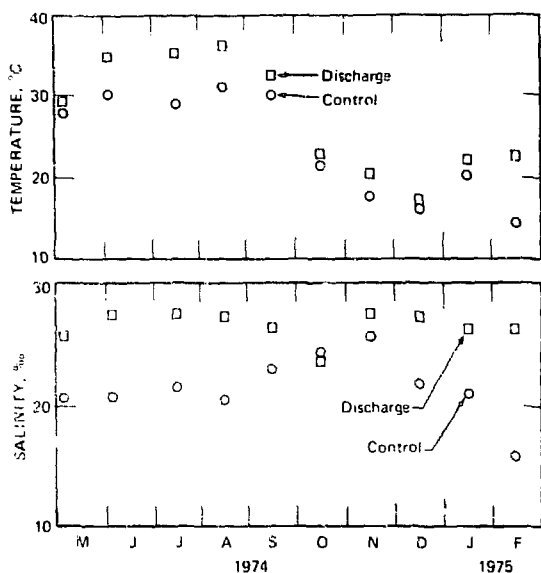


FIG. 2 Seasonal changes in sampled creek temperature and salinity.

### Seasonal Abundance and Biomass

Shown in Fig. 3 are the seasonal trends of abundance and biomass of the finfish collected in both creeks from May 1974 through February 1975. The abundance levels in the control creek were high during late spring and early summer, decreased steadily until late fall, then remained fairly constant at a low level through February. Biomass levels in the discharge creek were low until September when large catches of mullet were collected. After December, biomass in the discharge creek dropped steadily through February and was similar to, but somewhat lower than, the control-creek biomass.

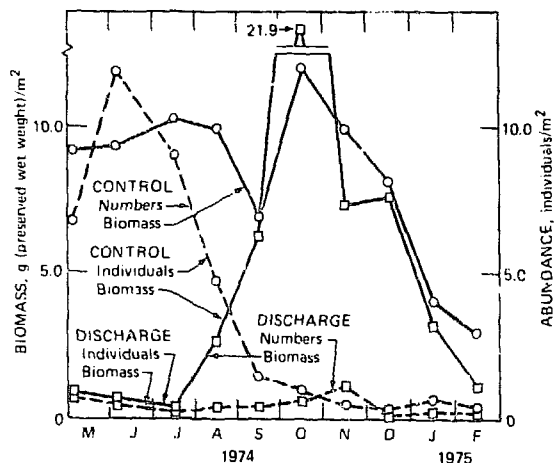


FIG. 3 Seasonal abundance (individual/m<sup>2</sup>) and biomass (g/m<sup>2</sup>) of tidal creek finfish populations.

Average biomass during the 10-month sampling period was 8.3 g/m<sup>2</sup> in the control creek and 5.2 g/m<sup>2</sup> in the discharge creek; the discharge area had fewer but larger fish (control, 3.67 individuals/m<sup>2</sup>; discharge, 0.47 individuals/m<sup>2</sup>).

In late spring and summer, populations were high (6.77 individuals/m<sup>2</sup>) in the control creek and low (0.46 individuals/m<sup>2</sup>) in the discharge creek. In the fall and winter months, an average of 0.58 individuals/m<sup>2</sup> were collected in the control creek, and 0.48 individuals/m<sup>2</sup> were collected in the discharge creek.

### Species Diversity

Table 1 lists the 59 species of finfish collected thus far in the salt-marsh tidal creeks. The control creek contained 55 species, and the discharge creek contained 38 species. Of the 59 species, 21 were collected only in the control creek and 4 were collected only in the discharge creek. None of these 25 species were ever very abundant.

Species accounting for over 80% of both the total abundance and biomass in the control creek were *Mugil cephalus*, *Menidia beryllina*, *Fundulus similis*, *F. grandis*, *Eucinostomus argenteus*, and *Mugil trichodon*. In the discharge creek, species accounting for over 80% of both the total abundance and biomass were *M. cephalus*, *F. similis*, *F. grandis*, *Pogonias cromis*, *E. argenteus*, and *M. beryllina*.

Species diversity as species/1000 fish is given in Fig. 4. Both creeks exhibited similar seasonal patterns in species diversity, with the highest diversity generally found during the summer and fall months. Diversity as species/1000 fish in the control creeks ranged from 13 to 24 (averaging 18), and in the discharge creek, it ranged from 9 to 20 (averaging 14). Diversity was greater in the control creek (1 to 13) during each of the monthly samples except for May 1974 and February 1975.

### Trophic Composition

Shown in Fig. 5 are seasonal trends in trophic structure as percent biomass of herbivores and midcarnivores of the total population. Species were grouped in categories (Table 1) according to classification by W. E. Odum (1971) and various estuarine fish dietary studies (Darnell, 1958; Carr and Adams, 1973; Harrington and Harrington, 1961; Diener, Inglis, and Adams, 1974; W. E. Odum, 1971; Homer, 1974, unpublished). Both creeks exhibited similar patterns for each trophic group. On a weight basis, midcarnivores dominated the samples from May through August with herbivores dominant from October through January. On the basis of percent of numbers of total catch, however, the midcarnivores in both creeks were always dominant (Fig. 6). Trophic structure was similar in both creeks except during June and July 1974.

### DISCUSSION

The most striking differences between the two areas were the changes in summer and winter abundance and

TABLE 1 Monthly Average Abundance and Biomass and Trophic Classification of Salt-Marsh Fish

| Species                              | Common name           | Number, individuals/m <sup>2</sup> |           | Biomass, g/m <sup>2</sup> * |           | Trophic level† |
|--------------------------------------|-----------------------|------------------------------------|-----------|-----------------------------|-----------|----------------|
|                                      |                       | Control                            | Discharge | Control                     | Discharge |                |
| <i>Carcharhinus limbatus</i> ‡       | Blacktip shark        |                                    |           |                             |           | TC             |
| <i>Sphyrna tiburo</i> ‡              | Bonnethead            |                                    |           |                             |           | TC             |
| <i>Dasyatis sabina</i>               | Atlantic stingray     | <0.01                              |           | 0.26                        |           | MC             |
| <i>Lepisosteus oculatus</i> ‡        | Spotted gar           |                                    |           |                             |           | TC             |
| <i>Elops saurus</i>                  | Ladyfish              | <0.01                              | <0.01     | 0.06                        | <0.01     | MC             |
| <i>Myrophis punctatus</i>            | Speckled worm eel     | <0.01                              |           | <0.01                       |           | MC             |
| <i>Brevoortia patronus</i> §         | Gulf menhaden         | <0.01                              |           | <0.01                       |           | O              |
| <i>Harengula pensacolae</i>          | Scaled sardine        | <0.01                              | <0.01     | <0.01                       |           | MC             |
| <i>Opisthonema oglinum</i>           | Threadfin herring     | <0.01                              |           | <0.01                       |           | MC             |
| <i>Anchoa hepsetus</i>               | Striped anchovy       | 0.01                               |           | 0.03                        |           | MC             |
| <i>A. mitchilli</i>                  | Bay anchovy           | 0.37                               | 0.07      | 0.10                        | 0.01      | MC             |
| <i>Synodus foetens</i>               | Inshore lizardfish    | <0.01                              |           | <0.01                       |           | TC             |
| <i>Arius felis</i> ‡                 | Sea catfish           |                                    | <0.01     |                             | 0.06      | MC             |
| <i>Opsanus beta</i> ‡                | Gulf toadfish         |                                    |           |                             |           | MC             |
| <i>Hyporhamphus unifasciatus</i>     | Halfbeak              | <0.01                              | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Strongylura marina</i>            | Atlantic needlefish   | 0.02                               | <0.01     | 0.19                        | 0.02      | TC             |
| <i>S. notata</i>                     | Redfin needlefish     | <0.01                              | <0.01     | 0.04                        | 0.03      | TC             |
| <i>Adinia xenica</i>                 | Diamond killifish     | <0.01                              | <0.01     | <0.01                       | <0.01     | H              |
| <i>Cyprinodon variegatus</i>         | Sheepshead killifish  | 0.09                               | 0.02      | 0.06                        | 0.03      | H              |
| <i>Floridichthys carpio</i>          | Goldspotted killifish | 0.12                               | 0.03      | 0.14                        | 0.03      | O              |
| <i>Fundulus grandis</i>              | Gulf killifish        | 0.13                               | 0.04      | 0.80                        | 0.30      | MC             |
| <i>F. similis</i>                    | Longnose killifish    | 0.15                               | 0.10      | 0.56                        | 0.36      | MC             |
| <i>Lucania parva</i>                 | Rainwater killifish   | <0.01                              | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Gambusia affinis</i> ‡            | Mosquitofish          |                                    |           |                             |           | MC             |
| <i>Poecilia latipinna</i>            | Sailfin molly         | <0.01                              | 0.02      | <0.01                       | 0.02      | H              |
| <i>Menidia beryllina</i>             | Tidewater silverside  | 2.14                               | 0.10      | 1.43                        | 0.07      | MC             |
| <i>Syngnathus floridae</i>           | Dusky pipefish        | <0.01                              |           | <0.01                       |           | PC             |
| <i>Echeneis naucrates</i> ‡          | Sharksucker           |                                    |           |                             |           |                |
| <i>Caranx hippos</i>                 | Crevalle jack         | <0.01                              |           | <0.01                       |           | MC             |
| <i>Oligoplites saurus</i>            | Leatherjacket         | <0.01                              | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Selene vomer</i> ‡                | Lookdown              |                                    |           |                             |           | MC             |
| <i>Trachinotus falcatus</i>          | Permit                | <0.01                              |           | <0.01                       |           | MC             |
| <i>Lutjanus griseus</i>              | Grey snapper          |                                    | <0.01     |                             | <0.01     | MC             |
| <i>Eucinostomus argenteus</i> ¶      | Spotfin mojarra       | 0.40                               | 0.03      | 0.52                        | 0.08      | MC             |
| <i>E. gula</i> ‡                     | Silver jenny          | <0.01                              |           | 0.01                        |           | MC             |
| <i>E. lefroyi</i> ‡                  | Mottled mojarra       |                                    | <0.01     |                             | <0.01     | MC             |
| <i>Orthopristes chrysopterus</i> ‡   | Pigfish               |                                    |           |                             |           | MC             |
| <i>Archosargus probatocephalus</i> ‡ | Sheepshead            |                                    |           |                             |           | MC             |
| <i>Lagodon rhomboides</i>            | Finfish               | 0.04                               | <0.01     | 0.28                        | 0.05      | MC             |
| <i>Bairdiella chrysura</i>           | Silver perch          | <0.01                              |           | <0.01                       |           | MC             |
| <i>Cynoscion nebulosus</i>           | Spotted sea trout     | <0.01                              |           | <0.01                       |           | TC             |
| <i>Leiostomus xanthurus</i>          | Spot                  | <0.01                              | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Pogonias cromis</i>               | Black drum            | <0.01                              | <0.01     | 0.06                        | 0.38      | MC             |
| <i>Sciaenops ocellata</i>            | Red drum              | <0.01                              | <0.01     | 0.04                        | 0.09      | MC             |
| <i>Chaetodipterus faber</i>          | Spadefish             | <0.01                              | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Mugil cephalus</i>                | Striped mullet        | 0.03                               | 0.03      | 2.98                        | 3.00      | H              |
| <i>M. curema</i>                     | White mullet          | <0.01                              | <0.01     | 0.09                        | 0.09      | H              |
| <i>M. trichodon</i>                  | Fintail mullet        | 0.14                               | <0.01     | 0.55                        | 0.08      | H              |
| <i>Polydactylus octonemus</i> ‡      | Atlantic threadfin    |                                    |           |                             |           | MC             |
| <i>Bathygobius soporator</i> ‡       | Frillfin goby         |                                    |           |                             |           | MC             |

TABLE 1 (Continued)

| Species                                           | Common name           | Number, individuals/m <sup>2</sup> |           | Biomass, g/m <sup>2</sup> * |           | Trophic level† |
|---------------------------------------------------|-----------------------|------------------------------------|-----------|-----------------------------|-----------|----------------|
|                                                   |                       | Control                            | Discharge | Control                     | Discharge |                |
| <i>Gobiosoma robustum</i>                         | Code goby             | <0.01                              | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Microgobius gulosus</i>                        | Clown goby            | 0.01                               | <0.01     | <0.01                       | <0.01     | MC             |
| <i>Scomberomorus maculatus</i>                    | Spanish mackerel      | <0.01                              |           | <0.01                       |           | MC             |
| <i>Prionotus tribulus</i>                         | Big-headed sea robin  | <0.01                              |           | <0.01                       |           | MC             |
| <i>Paralichthys albigutta</i>                     | Gulf flounder         | <0.01                              | <0.01     | 0.02                        | <0.01     | TC             |
| <i>Achirus lineatus</i> ‡                         | Lined sole            | <0.01                              |           | <0.01                       |           | MC             |
| <i>Symphurus plagiosa</i>                         | Blackcheek tonguefish | <0.01                              |           | <0.01                       |           | MC             |
| <i>Sphoeroides nephelus</i>                       | Southern puffer       | <0.01                              |           | <0.01                       |           | MC             |
| <i>Chilomycterus schoepfi</i>                     | Striped burrfish      | <0.01                              |           | <0.01                       |           | MC             |
| <i>Callinectes sapidus</i>                        | Blue crab             | 0.04                               | 0.01      | 0.40                        | 0.30      | MC             |
| Total monthly average excluding crabs             |                       | 3.67                               | 0.47      | 8.26                        | 5.20      |                |
| Total monthly average including crabs             |                       | 3.71                               | 0.48      | 8.66                        | 5.50      |                |
| Total number of finfish collected                 |                       | 57,300                             | 9789      |                             |           |                |
| Total number of finfish collected plus crabs      |                       | 57,900                             | 9956      |                             |           |                |
| Total biomass of finfish collected, kg            |                       |                                    |           | 128.7                       | 108.7     |                |
| Total biomass of finfish collected plus crabs, kg |                       |                                    |           | 134.9                       | 114.9     |                |

\*Biomass in grams preserved weight per square meter.

†TC, top carnivore; MC, midcarnivore; O, omnivore; H, herbivore; PC, primary carnivore.

‡Species collected in other nearby creeks or collected by Adams (1974).

§Tentative identification; taxonomy of this genus in the Gulf still uncertain.

¶Many of these individuals were <35 mm but by body dimensions were probably *E. argenteus*.

biomass patterns. During the summer months, biomass in the discharge creek was only 24% of that in the control creek, and numbers in the discharge creek were only about 7% of those in the control creek. This difference may be attributed to the exclusion from the discharge creek of the juvenile and young of a number of species, such as the silversides, killifish, mullet, mojarra, and others, which were abundant in the control creek. This exclusion may have been in response to the thermal effluent from the power plant or to the loss of sediment from the bottom of the discharge creek. Carr and Giesel (1975), in a study of the effects of a Jacksonville, Fla., power plant located on a salt marsh, found a similar exclusion occurring during the summer months. Nugent (1970), in a study of the effects of the Florida Power and Light Corporation plant at Turkey Point on fish inhabiting a subtropical estuary, took some of his samples in mangrove-lined tidal creeks. During the summer months he found significantly greater numbers of fish in the control-creek area. However, because of his sampling methods, no juvenile or young fish were taken. Adams (1974), in a study of fishes inhabiting the shallow bays adjacent to the present study marsh area at Crystal River, Fla., found only slightly higher levels of biomass in the control area during the summer months. Gallaway and

Strawn (1974), studying the effects of thermal discharge on marine fishes in Galveston Bay, Texas, concluded that detrimental effects on fish during summer months were negligible. These studies indicate a greater effect of heated effluents during the warm months on the shallow waters of tidal creeks than on the relatively deeper bay waters receiving thermal discharge.

During the fall and winter months, the biomass in the discharge creek exceeded that of the control area by 12%, although abundance in the discharge creek during this period was 85% of the control creek. Greater numbers of adult fish appeared during these months in the discharge creek. The striped mullet, *Mugil cephalus*, was the dominant fish, although adult red drum, *Sciaenops ocellata*, black drum, *Pogonias cromis*, and sea catfish, *Arius felis*, also occurred in the discharge creek during this period. Young (1974), in a study of the salt-marsh grasses in the present study area, found significantly higher levels of dead material in the discharge area. There was also some indication of higher export rates of this material from the discharge marsh. This, then, might have attracted mullet, which graze on this material (W. E. Odum, 1968). The discharge creek was located only about 360 m from the discharge canal. Gallaway and Strawn (1974) observed

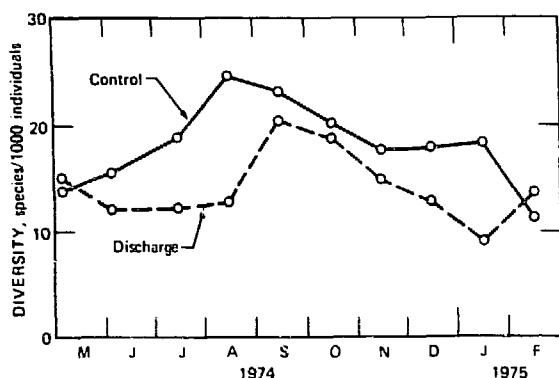


FIG. 4 Seasonal trends in species diversity as species/thousand individuals.

"tremendous numbers of adult mullet" at the mouth of the discharge canal during their study. Young (1974) also measured a fall "bloom" in discharge marsh-grass production. The timing of this bloom (September through October) correlates well with the movement of greater numbers of adult mullet into the discharge creek.

As noted by many investigators (Harrington and Harrington, 1961; Wiley, Koo, and Cronin, 1972; Weaver and Holloway, 1974), salt marshes serve as nursery grounds for the juveniles and young of many species of fish. Numerous investigators (Reid, 1954; Springer and Woodburn, 1960; Sykes and Finucane, 1966; Carr and Giesel, 1975) have found that many of the young-of-the-year appear in estuaries during the spring and early summer months. Based on seine hauls from discharge and control creeks in Jacksonville, Fla., Carr and Giesel (1975) stated that the discharge from the power plant at their study site "had a detrimental effect on the capacity of two marshland creeks to serve as a nursery area for juvenile fishes during the hot season."

Apparently the discharge creek sampled during the present study suffered a similar detrimental effect during the hot months. However, a number of these displaced

species appeared in the discharge creek during the fall and winter months in comparable numbers and biomass and exhibited similar age class structures. This suggests that juvenile fish are avoiding the discharge creek during the hot months and are adding to nearby areas. Either the displaced population is surviving in the other areas or there is a large enough pool of these species in the vicinity of the discharge creek to supply a population in the colder period.

### Trophic Structure and Fish Diversity

Diversity may be viewed as a measurement of the information pool and adaptability of a system. Diversity in some systems may decline under conditions of stress (Cairns, 1971) or when there is an additional energy resource. Heat may be regarded as a stress to some organisms and a resource to others.

Diversity was found to be lower in the discharge area during 8 of the 10 monthly samples (Fig. 4). The seasonal trends of diversity in both creeks were very similar, however, with lower values found during the spring and early summer, higher values in late summer and the fall, and lower values again during early winter. Adams (1974) found the same diversity relationships among fish inhabiting shallow estuarine bays adjacent to the discharge and control creeks of the present study. Diversity, calculated from several papers (Reid, 1954; Kilby, 1955; Tagatz and Dudley, 1961; McFarland, 1963; Norden, 1966; McErlean et al., 1973; Adams, 1974; Cherr, 1974), indicates similar seasonal trends for shallow-water fish in Texas, northern Florida, Louisiana, and North Carolina. The greatest impact of thermal discharges on diversity appears to occur during those months when high diversities are normally found, whereas data from Adam's (1974) study and the data collected so far in this study show no enhancement of diversity during the cooler months.

Trophic structure (Figs. 5 and 6) and the basic food-web connections appear to be intact in the discharge creek during the warm months. The lower abundance and biomass levels of fish in the discharge creek indicate, however, a lower energy processing regime in this area.

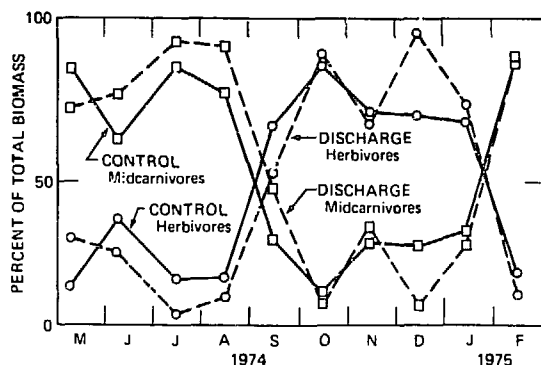


FIG. 5 Seasonal trends in tidal-creek fish trophic structure.

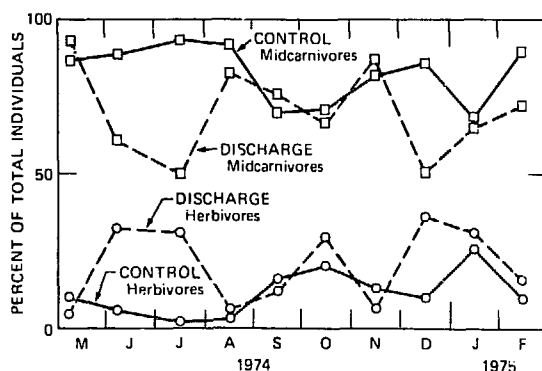


FIG. 6 Seasonal trends in tidal-creek fish trophic structure.

Naylor (1965) suggested that juvenile fish inhabiting thermal-discharge areas may show higher growth rates, i.e., faster turnover times. Preliminary analysis of growth rates (not presented here) indicates that the discharge-creek fish did not exhibit higher growth rates.

During the cooler months, in addition to the trophic structure and relationships in the discharge creek being similar to those in the control creek, the levels of abundance and biomass were also similar. During this period more energy may have been processed in the discharge creek, particularly with the presence of greater numbers of adult mullet. W. E. Odum (1976) suggested that a mullet will filter 100 g of material for every 1 g ingested and that a single adult mullet might be responsible for altering 45 m<sup>2</sup> of the top 0.5 cm of sediment per year.

### Energy Diagram

Often, when dealing with complex systems containing many interactions, it is desirable to symbolically represent the system so that these interactions are seen in overview. Using energy language (H. T. Odum, 1971), we developed a food-web diagram for the tidal-creek fish collected during this study (Figs. 7 and 8). The major species are shown separately, and the remaining species are grouped according to trophic level. The histograms within each of the food-web components represent the average grams per square meter of that species or group collected during the

warm (May through September) months (Fig. 7) and the cool (October through February) months (Fig. 8). The pathways representing the flow of energy from component to component are based on stomach-analysis data collected during this study (not reported here) and various other gut studies cited in Kemp, McKellar, and Homer (1974).

As given in Fig. 7, the biomass of control-creek fish at all trophic levels exceeded the discharge-creek biomass figures during the spring and summer months. During this period Smith (1974) measured higher shallow day community respiration rates in the discharge area at Crystal River, and Young (1974) recorded the highest marsh-grass net-production rates in the site of the present study. These measurements, along with the scouring of discharge-creek sediments, indicate greater marsh export rates in the discharge area to the bay systems; so the marsh materials may have been processed in the bays rather than in the creeks.

### ACKNOWLEDGMENTS

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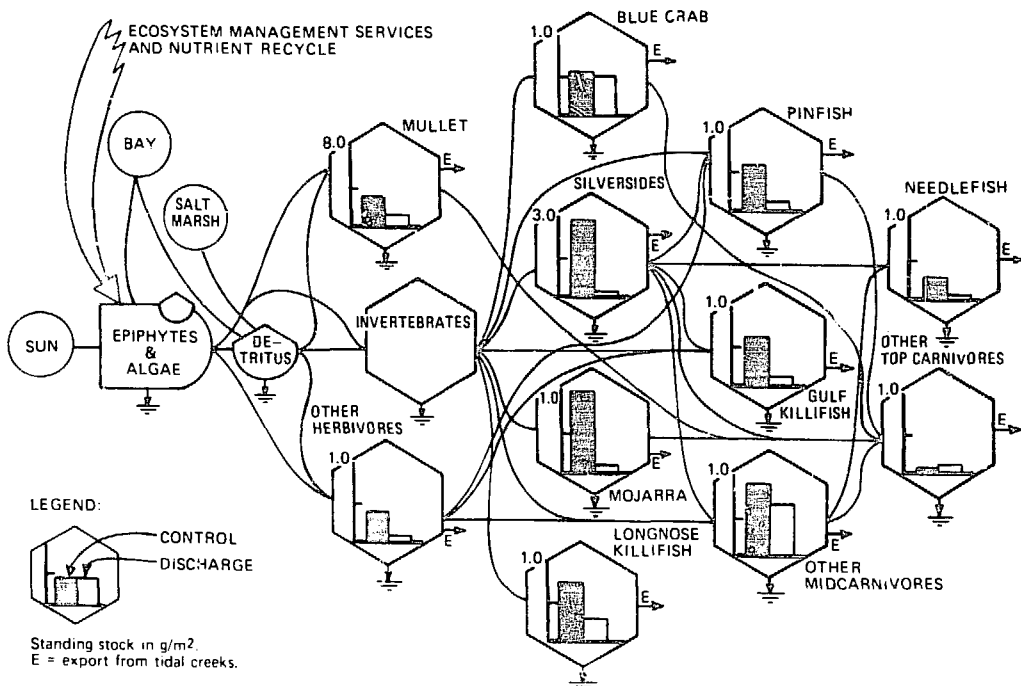


FIG. 7 Tidal-creek food-web energy diagram. Average standing stocks (g/m<sup>2</sup>) from May through September 1974.



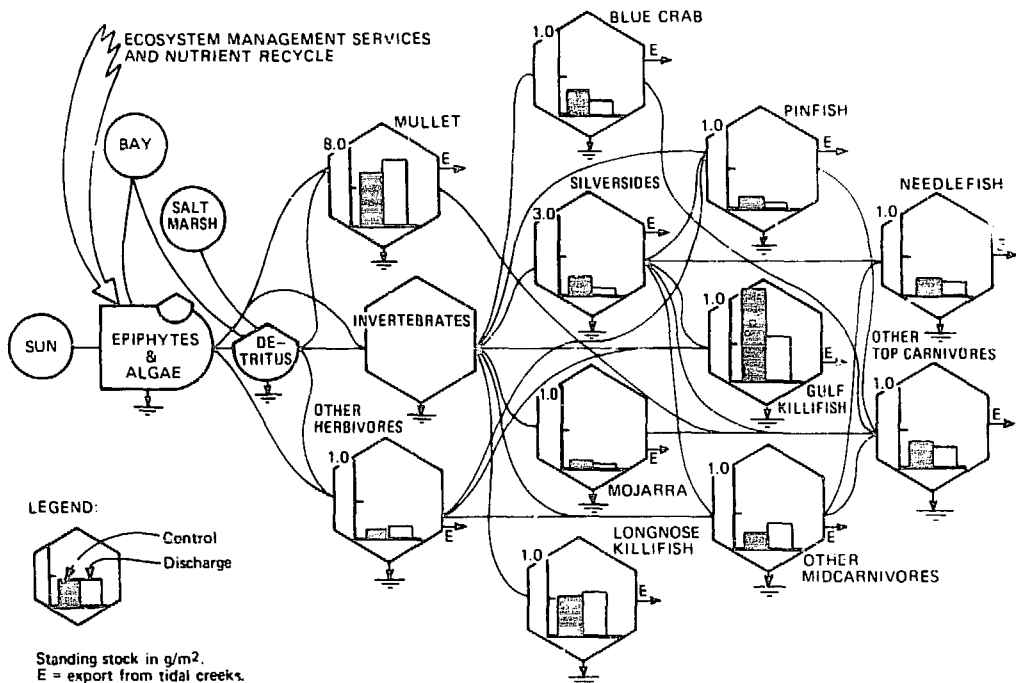


FIG 8 Tidal-creek food-web energy diagram. Average standing stocks (g/m<sup>2</sup>) from October 1974 through February 1975.

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# Fish Diversity in Adjacent Ambient, Thermal, and Post-Thermal Freshwater Streams

Robert W. McFarlane

## ABSTRACT

The Savannah River Plant area is drained by five streams of various sizes and thermal histories. One has never been thermally stressed, two presently receive thermal effluent, and two formerly received thermal effluent from nuclear production reactors. Sixty-four species of fishes are known to inhabit these streams; 55 species is the highest number obtained from any one stream. Thermal effluent in small streams excludes fish during periods of high temperatures, but the streams are rapidly reinvaded when temperatures subside below lethal limits. Some cyprinids become extinct in nonthermal tributaries upstream from the thermal effluents after extended periods of thermal stress. This extinction is similar to that which follows stream impoundment. Post-thermal streams rapidly recover their fish diversity and abundance. The alteration of the streambed and removal of overhead canopy may change the stream characteristics and modify the post-thermal fish fauna.

The Savannah River Plant (SRP), a National Environmental Research Park of the Energy Research and Development Administration near Aiken, S. C., provides an excellent opportunity to examine the effects of thermal effluent on fish diversity in freshwater streams. Five streams drain the SRP area and flow, approximately parallel, for various distances before entering the Savannah River (Fig. 1). The largest stream, in both length and flow, has never been subjected to thermal stress. Two smaller streams have received thermal effluent from nuclear reactors for more than 20 years. Two other streams also received thermal effluent for some years but no longer do so. One of these has subsequently been dammed upstream to create a large cooling reservoir. This study contrasts the past and present fish fauna of these streams to assess the effect of thermal stress on fish distribution.

## THE STREAMS

The streams of the SRP area have several features that increase their usefulness for studying the effects of thermal stress on aquatic ecosystems. The production reactor effluents raise the stream temperatures to a much higher level (in places  $>50^{\circ}\text{C}$ ) than that normally encountered

with industrial nuclear reactors. This produces a thermal gradient ranging from temperatures too high for most living organisms, through a thermophilic bacteria and algae zone, down to near ambient temperature when the water enters the Savannah River. In addition, since the streams are sufficiently narrow to allow for near bank-to-bank and top-to-bottom thermal constancy, the only refuge from the hot water lies in tributaries or adjacent sloughs. This means that most fish can exist in the main stream only during periods of reactor shutdown and near ambient temperatures.

The effluent water, which is removed from the Savannah River, pumped through the reactors, and released into the streambeds, has greatly increased water flow, erosion, and subsequent siltation. The streambeds have been broadened and aggraded; macrophyte vegetation has been eliminated and overhead canopy reduced, and deep silt deposits occur in some peripheral zones. Since the streams are relatively unaffected by industrial or sewage pollution, lotic fauna alterations are primarily direct or secondary results of thermal stress.

Upper Three Runs is a deep, swift blackwater stream 39 km long with a drainage area of  $490\text{ km}^2$ . It has never been thermally stressed, and much of it flows beneath an overhead canopy of bottomland hardwood trees.

The thermal-stressed streams are Four Mile Creek (20 km long, with a  $90\text{-km}^2$  drainage basin) and Pen Branch (16 km long, with a  $90\text{-km}^2$  basin), which carry heated effluent for 11 and 10 km, respectively, before entering a large cypress-gum swamp (Langley and Marter, 1973). Ambient-temperature tributaries extend a considerable distance upstream from the thermal confluence of each (Fig. 1).

The post-thermal streams are Steel Creek (15 km long, with a  $90\text{-km}^2$  basin), which received thermal effluent from 1954 to 1968 (Langley and Marter, 1973), and Lower Three Runs (38 km long, with a  $465\text{-km}^2$  basin), which received thermal effluent from 1953 to 1958. Since 1958 the dam and reservoir (Par Pond) constructed at the upper end of Lower Three Runs have reduced streamflow considerably.

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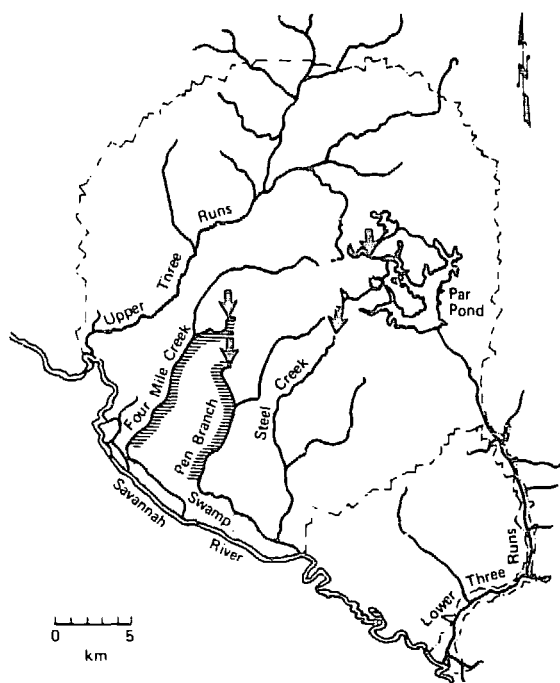


FIG. 1 Streams of the Savannah River Plant area. Waterways currently receiving thermal effluent are Four Mile Creek and Pen Branch. Arrows indicate the sources of present and past thermal effluent.

## FISH FAUNA

The southeastern coastal plain has a very diverse fish fauna, and the Savannah River drainage is known to contain 102 species (Dahlberg and Scott, 1971). Seventy-nine of these species have been found in the streams, swamp, and river of the SRP site, an upper coastal plain locality (Patrick, Cairns, and Roback, 1966; Freeman, 1954; 1958; Dahlberg and Scott, 1971; Clugston, 1973a; 1973b; Humphries, 1965; Smith et al., 1972; Parker, Hirshfield, and Gibbons, 1973; D. J. Tinney and J. W. Gibbons, unpublished manuscript, J. L. Greene, unpublished data). The studies cited, conducted from 1951 to the present, used a multitude of collecting techniques, including seines, dip nets, traps, electrofishing, angling, and fish toxins. Seines, dip nets, angling, and fish toxin were used in this study.

The streams have not all been fished with equal intensity, but 64 species (Table 1) are known to occur in one or more of them. Rare or seasonally occurring species may be added to the faunal lists of individual streams with future capture effort, but the presence or absence of the more common species is adequately known. Since data on relative abundance are insufficient to calculate diversity indexes, except for Steel Creek, this report will be limited to a discussion of species numbers.

Upper Three Runs supports 55 fish species. One species, the black-banded sunfish, *Enneacanthus chaetodon* (Free-

man, 1954; personal observations), is found nowhere else in the Savannah River drainage (Dahlberg and Scott, 1971).

At least 33 species of fish were present in Four Mile Creek before thermal stress (Freeman, 1954), and 29 species have been collected recently (Tinney and Gibbons, unpublished manuscript). The total fish fauna collected is 49 species. Pen Branch was known to contain 16 species before thermal stress (Freeman, 1954), and 20 species have been collected subsequently (Tinney and Gibbons, unpublished manuscript; Greene, unpublished data), raising the known fauna to 30 species. A minimum of 33 species are known to have reinvaded these two thermally-stressed streams in recent years, at least during intervals of cold-water flow.

Only 15 species were recorded from Steel Creek before thermal stress (Freeman, 1954). Recent collectors found 22 species present (Greene, unpublished data). Intensive fishing of Steel Creek from August until December 1974 yielded 3616 fish of 45 species, raising the total known fauna to 50 species. One species, the sawcheek darter, *Etheostoma serriferum*, had not been collected previously at the SRP site. The present species diversity in Steel Creek indicates that post-thermal recovery of the fish community is almost complete.

Freeman (1958) found 24 species in the upper part of Lower Three Runs in 1951–1952 and 35 species in the entire stream in 1957–1958 before the dam was completed. More-recent studies have raised the total to 53 species (Clugston, 1973b).

## THE EFFECT OF THERMAL STRESS

The apparent fish diversity in the thermal streams as discerned from current species lists is deceptive. First, the abundance of fish, in both numbers and species, is high only in periods of reactor shutdown but not during thermal effluent flow. Second, the fauna upstream from the thermal effluent is now exceedingly depauperate in both numbers and species.

With the exception of the mosquitofish, *Gambusia affinis*, there are very few fish in the main stream when thermal effluent is present. Mosquitofish are abundant in shallow peripheral areas at temperatures up to 41°C. A few centrarchids may be present at lower temperatures, but the tremendous numbers and the behavior of mosquitofish in the thermal habitat, contrasted with that in a similar but nonthermal habitat, suggests that predation has been greatly reduced.

During reactor shutdown the stream returns to ambient temperature and is immediately invaded by numerous fish from adjacent nonthermal areas. These fish range broadly until thermal effluent flows once more. They must then retreat downstream or into contiguous nonthermal areas. A tiny millpond adjacent to an abandoned dam on Four Mile Creek is a frequently used refuge. This pond is rotenoned after each ambient period to determine the species composition and relative abundance of the retreating fish. Thus far, 24 species have been collected, and species composition has been quite variable.

The diversity and abundance of species found in the headwater tributaries of Four Mile Creek and Pen Branch upstream from the thermal effluent are greatly reduced in contrast to comparable areas of Upper Three Runs or Steel Creek. Three species (*Esox niger*, *Notropis lutipinnis*, and *Etheostoma hopkinsi*) were collected from Pen Branch and four species (*Esox niger*, *Notropis lutipinnis*, *Notropis hypselopterus*, and *Gambusia affinis*) from Four Mile Creek during this study. Collecting efforts revealed that the otherwise normal-appearing first- and second-order tribu-

taries of these streams were essentially devoid of fish. Thus the thermal effluent appears to exert considerable influence on the stream fauna some distance removed from the heat stress. It is interesting to compare this alteration with that resulting from the damming of Lower Three Runs to form Par Pond.

Ten cyprinids, seven of which were collected from localities now upstream of the Par Pond dam before its construction, have been recorded from Lower Three Runs (Table 1). *Notropis petersoni* and *Notemigonus crysoleucas*

TABLE 1 Stream Fish of the Savannah River Plant\*

| Species and<br>common name                            | U3R | 4M | PB | SC | L3R | Par | Species and<br>common name                              | U3R | 4M | PB | SC | L3R | Par |
|-------------------------------------------------------|-----|----|----|----|-----|-----|---------------------------------------------------------|-----|----|----|----|-----|-----|
| <i>Lepisosteus osseus</i> ,<br>longnose gar           | +   | Δ  |    | +  | +   |     | <i>Noturus gyrinus</i> ,<br>tadpole madtom              | +Δ  | +  |    | +Δ | +   | Δ   |
| <i>L. platyrhincus</i> , Florida gar                  | +   | Δ  |    | +Δ |     |     | <i>N. insignis</i> , margined madtom                    | +Δ  |    |    | Δ  | +   |     |
| <i>Ambloplites rupestris</i> , bowfin                 | +   | +Δ |    | Δ  | +   | +   | <i>N. leptacanthus</i> ,<br>speckled madtom             | +Δ  | +  | +Δ | +Δ | +   |     |
| <i>Anguilla rostrata</i> ,<br>American eel            | +   | +Δ |    | +Δ | +   | +   | <i>Chologaster cornuta</i> , swampfish                  | +   |    |    |    | +   |     |
| <i>Alosa aestivalis</i> ,<br>blueback herring         |     | +  |    |    | +   | +   | <i>Aphredoderus sayanus</i> ,<br>pirate perch           | +Δ  | +Δ | +Δ | +Δ | +   | +Δ  |
| <i>Dorosoma cepedianum</i> ,<br>gizzard shad          |     | Δ  |    |    | +   | +   | <i>Fundulus lineolatus</i> ,<br>lined topminnow         | +Δ  | +Δ | +Δ | +Δ | +   |     |
| <i>Umbra pygmaea</i> ,<br>eastern mudminnow           | +   |    | +  | +Δ | +   | +   | <i>Gambusia affinis</i> ,<br>mosquitofish               | +Δ  | +Δ | +Δ | +Δ | +   | +Δ  |
| <i>Esox americanus</i> ,<br>redfin pickerel           | +   | +Δ | +  | +Δ | +   | +   | <i>Labidesthes sicculus</i> ,<br>brook silverside       | +   | +Δ |    | +Δ | +   | +Δ  |
| <i>E. niger</i> , chain pickerel                      | +Δ  | +Δ | +Δ | +Δ | +   | +   | <i>Acanthiarchus pomotis</i> ,<br>mud sunfish           | +   | +  | +  | Δ  | +   |     |
| <i>Cyprinus carpio</i> , carp                         | +   | +  |    | Δ  |     |     | <i>Centrarchus macropterus</i> , flier                  | +   | +  | +  | +Δ | +   |     |
| <i>Hypognathus nuchalis</i> ,<br>silvery minnow       |     | +  |    |    |     |     | <i>Elassoma zonatum</i> ,<br>banded pygmy sunfish       | +Δ  | +  |    | Δ  | +   | Δ   |
| <i>Nocomis biguttatus</i> ,<br>bluehead chub          | +Δ  | +  | +  | +  | +   |     | <i>Enneacanthus chaetodon</i> ,<br>black-banded sunfish | +Δ  |    |    |    |     |     |
| <i>Notemigonus crysoleucas</i> ,<br>golden shiner     | +   | Δ  | +  | +Δ | +   | +Δ  | <i>E. gloriosus</i> , bluespotted sunfish               | Δ   |    | +  | +Δ | +   |     |
| <i>Notropis chalybaeus</i> ,<br>ironcolor shiner      | +   | +  |    |    | +   |     | <i>E. obesus</i> , banded sunfish                       | +   |    | +  | +  |     |     |
| <i>N. cummingsae</i> , dusky shiner                   | +Δ  | +Δ | +Δ | +Δ | +   |     | <i>Lepomis auritus</i> ,<br>redbreast sunfish           | +   | +Δ | +  | +Δ | +   | +Δ  |
| <i>N. hudsonius</i> , spottail shiner                 | +   |    |    | Δ  |     |     | <i>L. cyanellus</i> , green sunfish                     | +   |    |    |    | +   |     |
| <i>N. hypselopterus</i> ,<br>sailfin shiner           | +   | +Δ |    |    | +   |     | <i>L. gibbosus</i> , pumpkinseed                        | Δ   | +Δ |    | +Δ |     |     |
| <i>N. lutipinnis</i> ,<br>yellowfin shiner            | +Δ  | +Δ | +Δ | +Δ | +   | +Δ  | <i>L. gulosus</i> , warmouth                            | +   | +Δ | +  | Δ  | +   | +   |
| <i>N. maculatus</i> , taillight shiner                | Δ   | +  |    | +Δ | +   |     | <i>L. macrochirus</i> , bluegill                        | +   | +Δ | +  | +Δ | +   | +   |
| <i>N. niveus</i> , whitefin shiner                    |     |    |    |    | +   |     | <i>L. marginatus</i> , dollar sunfish                   | +   | +Δ | +  | +Δ | +   | +Δ  |
| <i>N. petersoni</i> , coastal shiner                  | +   | Δ  |    | +Δ | +   | +   | <i>L. microlophus</i> ,<br>redecor sunfish              | +   | Δ  | +  | +Δ | +   | +   |
| <i>Semotilus atromaculatus</i> ,<br>creek chub        | +Δ  | Δ  | Δ  | +Δ | +   |     | <i>L. punctatus</i> , spotted sunfish                   | +   | +Δ | +  | +Δ | +   | +   |
| <i>Erimyzon oblongus</i> ,<br>creek chubsucker        | +   | +Δ | +  | +Δ | +   |     | <i>Micropterus salmoides</i> ,<br>largemouth bass       | +   | +Δ |    | +Δ | +   | +   |
| <i>E. sucetta</i> , lake chubsucker                   | +   | Δ  | +  | +Δ | +   | +   | <i>Pomoxis annularis</i> ,<br>white crappie             |     |    |    |    | +   |     |
| <i>Hypentelium nigricans</i> ,<br>northern hog sucker | +   | +  |    | +Δ |     |     | <i>P. nigromaculatus</i> ,<br>black crappie             | +   | +  | +  |    | +   | +   |
| <i>Minytrema melanops</i> ,<br>spotted sucker         | +   | +Δ |    | +  | +   | +   | <i>Etheostoma fricksium</i> ,<br>Savannah darter        | +   |    |    |    | +   |     |
| <i>Ictalurus brunneus</i> ,<br>snail bullhead         |     |    |    | Δ  |     |     | <i>E. fusiforme</i> , swamp darter                      |     | +  | +  | +Δ | +   | +Δ  |
| <i>I. catus</i> , white catfish                       |     |    |    | Δ  | +   |     | <i>E. hopkinsi</i> , Christmas darter                   | Δ   | Δ  | Δ  | Δ  |     |     |
| <i>I. natalis</i> , yellow bullhead                   | +   | +Δ | +  | Δ  | +   | +   | <i>E. olmstedii</i> ,<br>tessellated darter             | Δ   | Δ  |    | Δ  | +   |     |
| <i>I. nebulosus</i> , brown bullhead                  | +   | +  | +  | +  | +   | +   | <i>E. serriferum</i> , sawcheek darter                  |     |    |    | Δ  |     |     |
| <i>I. platycephalus</i> , flat bullhead               | +   | +Δ |    | +Δ | +   | +   | <i>Percina flavescens</i> , yellow perch                | +   | Δ  |    | +  | +   | +   |
| <i>I. punctatus</i> , channel catfish                 | +   | Δ  |    |    | +   | +   | <i>Percina nigrofasciata</i> ,<br>blackbanded darter    | +   | +  |    |    | +   |     |

\*Abbreviations are U3R, Upper Three Runs; 4M, Four Mile Creek; PB, Pen Branch; SC, Steel Creek; L3R, Lower Three Runs;

Par, Par Pond. Plus (+) indicates recorded presence of species based on literature survey; Δ, collected during this study.

# Distribution and Abundance of Fish Populations in the Middle Wabash River

Terry C. Teppen  
James R. Gammon

## ABSTRACT

A field investigation was made of the distribution and abundance of fish within a 161-km portion of the Wabash River to determine effects of heated effluents as well as changes in water quality on ichthyofaunal communities within the river. Twenty-six sampling stations were electro-fished, sequentially, four times in 1974 with extended sampling efforts made in the vicinity of two power-generating stations studied since 1967 and 1968.

During August an overall rise in river temperature of 4°C was observed from upstream to downstream, with several chemical factors also showing slight increases.

Although the majority of species populations were influenced either negatively or positively by the gradient of river conditions available to them, the only statistically significant parameters found in the analysis of community structure involved a lower diversity by weight below Terre Haute and a greater abundance of fish above the Cayuga generating station. Decreases occurred downstream in populations of redhorse (*Moxostoma* sp.), sauger (*Stizostedion canadense*), longear sunfish (*Lepomis megalotis*), and gizzard shad (*Dorosoma cepedianum*), with increases downstream observed in flathead catfish (*Pylodictis olivaris*), shortnose gar (*Lepisosteus platostomus*), longnose gar (*L. osseus*), and bowfin (*Amia calva*) populations. Carp (*Cyprinus carpio*) were present in large numbers throughout the study area with a tremendous population increase evident in recent years. Although species associations were variable among the segments, overall community parameters remained relatively unaffected.

From an ecological point of view, the thermal properties of water and the attending chemical, physical, and biological relationships are doubtless the most important factors in maintaining the fitness of water as an environment. Recent years have brought a growing concern for the quality of this environment as it relates to increased demand for power-generating stations. Discharges of heated water from those power plants using once-through cooling have been compared to the discharges of organic wastes in lowering the waste assimilative capacity of a stream (Parker and

Krenkel, 1969). Heated effluents from these plants, possible synergistic effects of heat combined with other pollutants, and the impact of the power-plant cooling process as an aquatic predator must all be measured and evaluated.

Unlike man, who has gained a considerable degree of control over his environment, fish have only limited mechanisms of temperature regulation. They must either avoid intolerable temperatures or acclimate to temperatures within evolved tolerances. Laboratory determinations of these tolerances and the acclimation process have received considerable attention (Fry, 1947; Sullivan, 1954; Brett, 1956; Ferguson, 1958), but investigations into the ability of natural fish populations to adapt to and survive thermal changes are relatively recent (Trembley, 1960, 1961; Alabaster, 1964; Proffitt, 1969; Allen, Boydstrum, and Garcia, 1970; Gibbons, Hook, and Forney, 1972; Gammon, 1971; 1973). Attempts to incorporate both aspects into a coordinated study have been related by Ferguson (1958), Coutant (1970), Neill and Magnuson (1974), and Holland et al. (1974).

Species survival in an aquatic ecosystem that is continually undergoing modification through natural agencies as well as man's activities is partly dependent on thermal adaptive ability. Feeding rates (Alabaster, 1964; Gibbons, Hook, and Forney, 1972), resistance to disease (DeMont and Miller, 1972; Smith, 1972), and spawning activity (Cairns, 1956; Clark, 1969), to name a few, are essential processes influenced by temperature. In this light, population structures may be used as indicators of environmental changes. It is the total impact of altered life processes which will ultimately determine the fate of a species.

This study examines the ichthyofaunal community structure of the middle Wabash River, including the influences of two large electric-power generating stations under investigation since 1967 and 1968 (Gammon, 1971; 1973). These early studies yielded valuable information about the behavioral responses of fishes to differing thermal conditions in the immediate vicinity of the two power

are the only two minnows found in Par Pond at present. Elsewhere at the SRP site, *Notropis petersoni* is predominantly a resident of deep pools, and *Notemigonus* primarily inhabits lentic sloughs. Thus these two species might logically appear to be the best adapted for persistence after impoundment. An intensive search of Par Pond tributaries during this study produced only two individual *Notropis lutipinnis*, the dominant species of headwater tributaries elsewhere in the SRP area. *Notropis cummingsae*, *Notropis hypselopterus*, *Nocomis leptcephalus*, and *Semotilus atromaculatus* are no longer present upstream from the dam. The outstanding characteristic of Par Pond tributaries is their total lack of fish, except in the immediate vicinity of the pond. All of these minnows, except *Semotilus*, were present, however, in these tributaries after 5 years of thermal effluent flow from a point near the upper reaches of the stream (Freeman, 1958).

Apparently 5 years of thermal stress eliminated only one of the seven cyprinid species inhabiting the stream. Impoundment and continued thermal stress of the reservoir have eliminated three additional species, and another is almost gone. The species that have disappeared from the impounded waterway are also absent or greatly reduced in abundance in Four Mile Creek and Pen Branch, which have been thermally stressed for more than 20 years.

The bottoms of SRP streams are primarily sand and deep detritus deposits, both ill-suited for minnow reproduction. Gravel beds, the preferred substrate, are rare. It is possible that most cyprinids found here are recruited from downstream areas, and such impoundments as the Par Pond dam eliminate this population replacement so that the species eventually becomes extinct upstream.

If upstream populations are the result of immigration, the presence of lethal thermal effluent for much of the year is a barrier almost as effective as a permanent, physical obstacle in the waterway. The rapid invasion of streams in the absence of thermal stress and the greater abundance of fish in downstream tributaries lend further credence to this hypothesis.

## ACKNOWLEDGMENTS

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plants, as well as habitat preferences in thermally unaffected parts of the river. Thermal-preference ranges were estimated for all important species in the river, but insufficient information was acquired to associate these localized behavioral responses with permanent changes in the species populations over extended periods of time. The measurement and assessment of permanent changes in species populations and the possible alterations of community structure demanded a detailed study concerning the nature of the community within an extended portion of the river.

## THE RIVER AND STUDY AREA

The Wabash is a turbid, largely free-flowing river, which drains two-thirds of Indiana as well as portions of Illinois and Ohio. Little or no rooted aquatic vegetation is present in the middle and lower reaches of the river, and bottom substrate is composed primarily of sand and gravel with finer silts deposited near the shore. Its waters are economically important for agriculture, municipalities, and industrial use, including electric-power generation. At one time commercial and sport fishing were important, but in recent years clam shell harvesting has yielded more economic benefits. Water quality is affected importantly by contributions from strip mining, agriculture, municipal waste, and industry in general.

The 161-km-long (100 mile) portion studied extends from Independence, Ind., to the mouth of Big Creek in Illinois, has an average gradient of 11 cm/km (7.2 in./mile), and an average low flow estimated at 2250 cfs.

Sampling stations were 1 km in length and located on the outside of moderate river bends at intervals of 3 to 7 miles. Previous studies established that the abundance and variety of fish were distributed nonrandomly, with comparatively fast water on the outside bends yielding about 50% more individuals and 25% more species than slow-moving and midstream stretches of the river (Gammon, 1973). The faster areas also tend to be avoided by gizzard shad (*Dorosoma cepedianum*), which favor the quieter areas. Landmarks, such as tributaries, buildings, marked trees, and rocks, defined the 1-km stations. Individual sampling stations were labeled according to their river mile (RM) distance from the mouth of the Wabash River (Fig. 1).

Two areas adjacent to large fossil-fuel generating stations were also included in the study. The Cayuga station (1011 Mw) at RM 248.6 began operating with one unit in October 1970 and an additional unit in May 1972. Ecological studies under way since 1968 are summarized elsewhere (Gammon, 1973). The older Wabash station began operations in 1953 and was enlarged to its present capacity (970 Mw) in 1968. Ecological impact at this station has been studied since 1967 (Gammon, 1971; 1973). Both plants elevate intake water about 8°C before discharging it back to the river. Another small power station at RM 200.2 was included in the study, but its

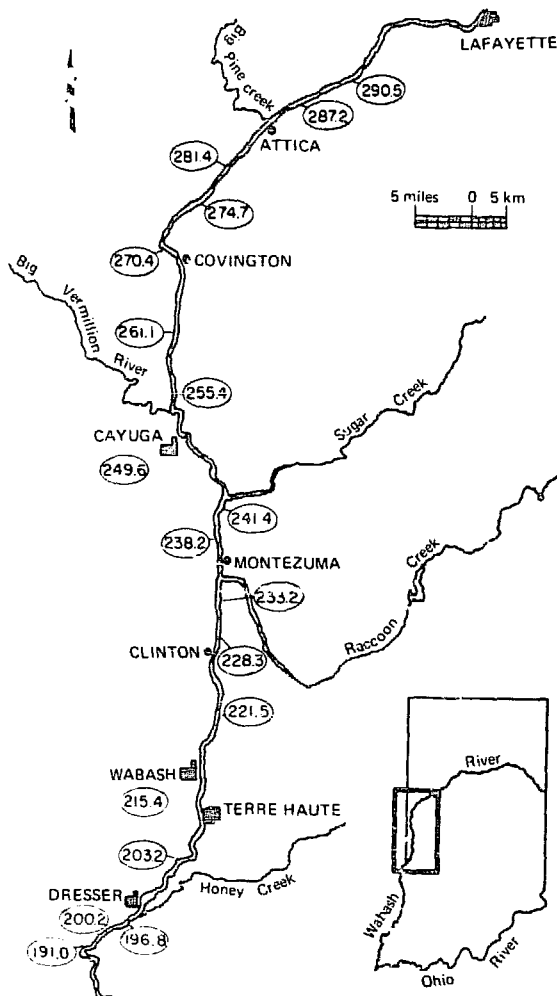


FIG. 1 Map of study area including sampling stations and electric-power generating plants according to distance from the mouth of the Wabash River.

impact appears to be minimal. Retirement of this plant has recently been announced by Public Service of Indiana.

The portion of the river studied was subdivided into five segments as follows: (1) RM 290.5 to 255.4, known as the segment "above Cayuga"; (2) the "Cayuga" segment in the vicinity of RM 249.6; (3) RM 241.4 to 221.5, called the "intermediate" segment; (4) the "Wabash" segment at RM 215.4; and (5) the lowermost segment from RM 203.2 to 191.0, referred to as "below Terre Haute."

The Indiana State Board of Health (ISBH), in conjunction with the U. S. Environmental Protection Agency, included this portion of the Wabash River in an extensive program of water-quality analysis (Indiana State Board of Health, 1975). During early August 1974, chemical and microbial determinations were made in a thorough survey of outfalls, discharges, tributaries, and the Wabash River



proper. Many of these ISBH sampling stations coincided with our own.

## METHODS

Fish were collected by means of a Smith-Root type V1 electrofisher, which converted 115-volt a-c from a 3000-watt generator into pulsed d-c current. The current was fed into five short, stainless-steel anodes extending 8 ft in front of the boat: three long woven-copper cathodes hung from the bow. Pulse configuration consisted of a triangular wave (fast rise, slow decay) interrupted into 60 pps (pulses per second) and producing between 300- and 425-volt d-c at 4 to 6 mps. These specifications were shown to be effective in drawing fish within easy netting distance while causing minimal damage to the fish themselves (Taylor, Cole, and Sigler, 1957; Vincent, 1971; Novotny and Priegel, 1974).

The collecting procedure, developed and refined during 1973 and fully implemented in 1974, consisted in electrofishing through the sampling stations sequentially from upstream to downstream. Four complete passes through all stations were made during 1974, two in July, one in August, and one in September/October. Each required five consecutive days of effort. The third trip was coordinated with the sampling program of the ISBH.

Each sampling station was electrofished from beginning to end as close to the shoreline and submerged cover as possible. Work assignments among the three-man crew were rotated to minimize sampling bias. Captured fish were transferred immediately to an on-board live well. At the end of each station, fish were identified, weighed, and measured. Temperatures were recorded at the beginning and end of each station with a YSI model 33 telethermistor.

The data from each collection were recorded on magnetic tape, and analytic programs were developed for computing various parameters, including total number of fish captured per kilometer, total weight of fish caught per kilometer, number of species captured, and Shannon-Weiner diversity indices (Shannon, 1948) based on both numbers and weights.

Information was also generated about individual species populations, including number caught, weight caught, mean weight, and mean total length.

## RESULTS

The mean temperature of the 161-km study area during the July and August trips can be divided into three distinct parts (Fig. 2): the cooler portion above Cayuga varying slightly from 23 to 24°C, an intermediate plateau between the two power plants measuring around 26°C, and the lower portion below Terre Haute warmed to approximately 27.5°C. Thus an overall rise of 4°C occurs from upstream to downstream within the study area, with localized areas of thermal loading directly downstream from the power stations. Temperature peaks below the power stations represent samples taken within the thermally mixed area of the river.

The remaining chemical trends shown in Fig. 2 are based on only a portion of the ISBH study and are drawn from those sampling sites located in mid-river. Similar patterns are exhibited in 5-day BOD and total solids, with a slight increase from upstream to downstream and a noticeable peak at RM 228 below the Clinton Laboratories of Eli Lilly & Co., a pharmaceuticals manufacturer. The lowered 5-day BOD above and below the Cayuga plant appear to be related to entry of tributary waters into the Wabash River.

Ammonia nitrogen remained low throughout the study area, whereas nitrate-nitrite nitrogen levels, which range from 0.4 to 0.5 mg/liter in the upper reaches, rather suddenly double in concentration downstream from the Big Vermillion River and remain high throughout the lower reaches.

Dissolved oxygen (DO) concentrations were observed to fluctuate rather erratically throughout the study area, even though determinations were restricted to those taken early in the day to minimize effects of photosynthesis. The DO concentrations were transformed to percent saturation to eliminate the effects of temperature. In general, DO saturation remains close to 100%  $\pm$  10% within the entire study area and exhibits no definite trends. A study in 1967

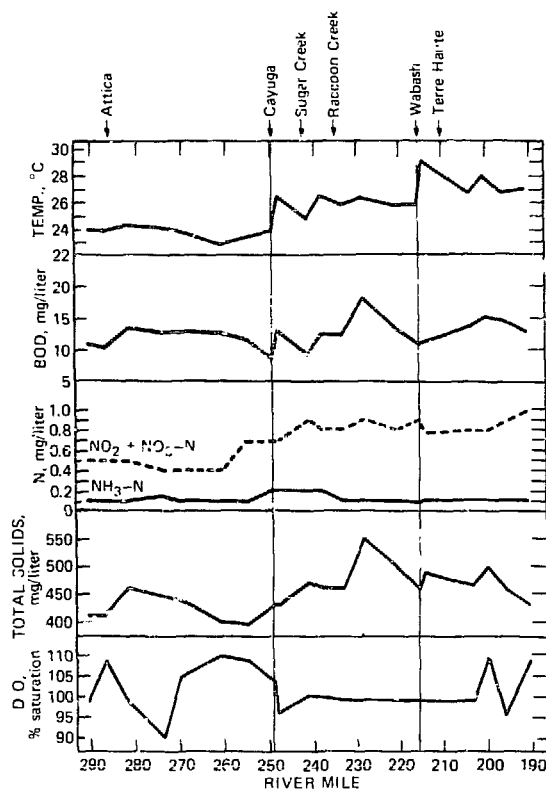


FIG. 2 Physical (July to August 1974) and chemical (August 1974) characteristics of the study area. (Chemical data from Indiana State Board of Health, 1975).

reported a drop in DO to only 3.0 mg/liter in a 13-mile stretch below Terre Haute (Nedved, 1971), but no comparable decrease was noted in 1974.

Thirty-six species of fish were collected in 1974, with exclusion of all fry and smaller species of Cyprinidae, Atherinidae, and Percidae from the analysis. The elimination of these individuals seems justified in that they were not taken as effectively as larger, older fish owing to the selectivity of the electrofisher. Woolcott (1974) found similar results using a d-c boat electrofisher. Those species captured inconsistently in electrofishing samples and not included in the analysis are the following: silverjaw minnow (*Ericymba buccata*), silvery minnow (*Hybognathus nuchalis*), silver chub (*Hybopsis storerianus*), river chub (*Neomystus micropogon*), emerald shiner (*Notropis atherinoides*), striped shiner (*N. chryscephalus*), spotfin shiner (*N. spilopterus*), suckermouth minnow (*Phenacobius mirabilis*), bluntnose minnow (*Pimephales notatus*), bullhead minnow (*P. vigilax*), brook silversides (*Labidesthes sicculus*), logperch (*Percina caprodes*), and blackside darter (*P. maculata*).

Five community parameters were derived from the catch data: (1) the number caught per kilometer, (2) the weight in kilograms caught per kilometer, (3) the number of species caught at each sampling station, (4) a Shannon-Weiner (S-W) diversity index based on numbers, and (5) a S-W diversity based on weights. Means and standard errors for each station are illustrated in Fig. 3.

These data were subjected to a nested analysis of variance to test for statistical differences among the five segments at the 95% level (Table 1). Marked variability was evident among the individual sampling stations for all community parameters except for S-W diversity based on weights.

The five segments differed statistically only with respect to the number of fish caught per kilometer and the S-W diversity based on weights. A Student-Newman-Keuls (S-N-K) multiple-range comparison test (Newman, 1939; Keuls, 1952) analyzing numbers caught per kilometer indicated three different groupings among the five segments with two of the three groups overlapping: (1) the intermediate segment and the segment below Terre Haute, (2) the segment below Terre Haute, Wabash, and Cayuga, and (3) the segment above Cayuga. The uppermost segment yielded significantly more fish than any of the other segments.

The S-N-K test applied to the S-W diversities (weight) indicated two overlapping groups: (1) below Terre Haute and above Cayuga and (2) all segments except below Terre Haute. The S-W diversity based on weight distribution among the species was statistically lower below Terre Haute than it was for any of the other segments.

With respect to these community parameters then, the fish populations in the vicinity of the two power plants are not statistically different from other portions of the river. The lowered diversity of fish below Terre Haute appears to be the result of the combined impact exerted on the river

ecosystem by industry, strip mining, and Terre Haute; although synergistic effects resulting from thermal additions cannot be ruled out.

The distribution and abundance of a significant portion of fish captured in 1974 in the middle Wabash are summarized graphically in Figs. 4 and 5. To remain consistent in restricting the analysis only to those stations situated on the outside of river bends, we included in this analysis only four of the eight stations sampled at Cayuga and four of the six stations sampled at Wabash. Three of the four stations at Cayuga and two of the four at Wabash were thermally elevated sections of the river. Few species of fish were restricted only to one region of the study area, while, on the other hand, few species were evenly represented throughout.

Gizzard shad were the most abundant species captured and appeared in virtually every collection in 1974. They were most abundant above Cayuga and least abundant in the intermediate segment (Table 2). Numbers of gizzard shad caught at the two generating stations were similar to the numbers caught in other segments of the river. Their profuse numbers have dominated catches at Cayuga and Wabash since studies began (Tables 3 and 4).

River carpsucker (*Carpionodes carpio*) were more abundant at and above Cayuga than in the segments downstream. Populations were noticeably higher at Cayuga and Wabash, which seemingly attracted and held individuals from adjacent areas, than in other segments of the river. However, the density declined overall in the vicinity of the Cayuga plant and never regained the 1968 and 1969 levels at the Wabash plant.

Golden redbreast (*Moxostoma erythrum*) were more abundant in the upstream segments, including Cayuga, and all but absent from Wabash and below Terre Haute. It was postulated that this species would probably be eliminated from the Cayuga area and downstream (Gannon, 1973), but they have maintained constant population levels since 1972 (Table 3). Spottail redbreast (*M. macrolepidotum*) and silver redbreast (*M. unisurum*) were present in fewer numbers, but were distributed much the same as were golden redbreast.

Freshwater drum (*Aplodinotus grunniens*) were fairly evenly distributed throughout, with obviously reduced abundance at the Cayuga and Wabash plants and depressed populations below Terre Haute. Populations of drum have shown no definite trends over the years at the power stations.

Sauger (*Stizostedion canadense*) were much more abundant above Cayuga and also fairly abundant in the intermediate segment. They were present in somewhat lower numbers at Cayuga and Wabash and nearly absent below Terre Haute. Since the Cayuga plant began operations in 1970, the catch of sauger has been roughly half as great as in previous years. At the Wabash plant catches were substantial in 1968 but much diminished thereafter.

Longear sunfish (*Lepomis megalotis*) were essentially restricted to the uppermost segment, although a few fish

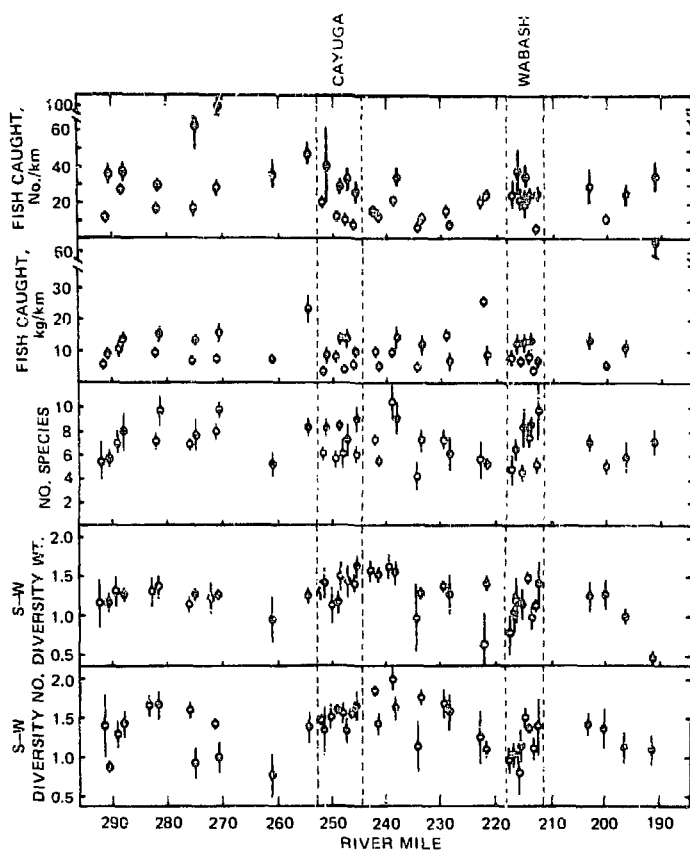


FIG. 3 Mean values ( $\pm 1$  SE) of the community parameters for each sampling station.  $\circ$ , 1973.  $\bullet$ , 1974.

TABLE 1 Results of Nested Analysis of Variance Tests for Differences Among Five Segments and Among Sampling Stations Within the Study Area

| Community parameter      | Probability of difference (segments) | Probability of difference (station) |
|--------------------------|--------------------------------------|-------------------------------------|
| Number caught per km     | $P > 0.05$                           | $P > 0.01$                          |
| Weight caught per km     | Ns*                                  | $P > 0.01$                          |
| Number of species caught | Ns                                   | $P > 0.01$                          |
| S-W diversity (weight)   | $P > 0.05$                           | Ns                                  |
| S-W diversity (number)   | Ns                                   | $P > 0.05$                          |

\*Not significant.

were caught in the power-plant segments. Abundance at Cayuga has declined sharply during the past several years, but no evident trends appear at the Wabash plant.

Carp (*Cyprinus carpio*) were the second most abundant species numerically and the most significant in terms of

weight (Fig. 5). The pattern for this species was toward an even distribution and abundance throughout the study area, with an unusually high density in the downstream reaches. An immense increase in catch of carp in recent years bears witness to their proliferation throughout the middle Wabash. However, they were neither more nor less abundant at the two power plants than in other segments in 1974.

Flathead catfish (*Pylodictis olivaris*) were well-represented from Cayuga downstream, but nearly absent above Cayuga. Preliminary collections from this uppermost segment in 1973 also yielded no individuals of this species. The Cayuga power plant has led to an expansion of this species population within the Cayuga segment and may also have increased the population in the intermediate segment. Channel catfish (*Ictalurus punctatus*) were taken sporadically in unexpectedly low numbers throughout the study area, with the greatest concentrations occurring below Terre Haute. None have been collected at Cayuga during the past 2 years, although small catches were made in earlier years.

Shortnose and longnose gar (*Lepisosteus platostomus* and *L. osseus*) were quite abundant but sporadic in their

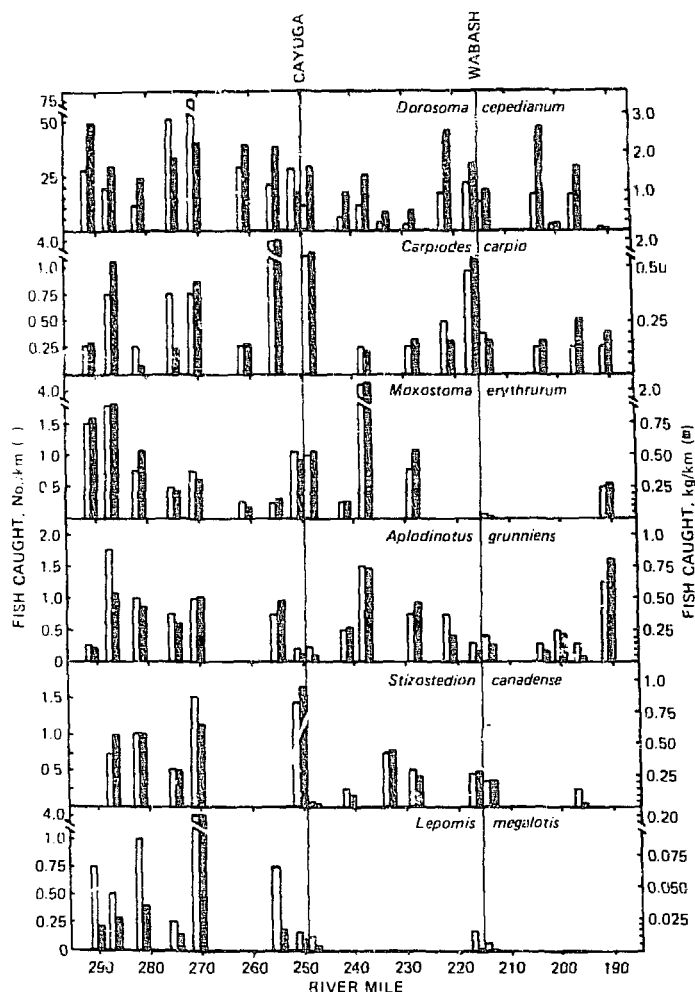


FIG. 4 Number and weight per kilometer for several species of fish captured within the study area.

distribution. Both species were notably abundant below Terre Haute. At Cayuga they exhibited an abundance comparable to other segments, but they have declined markedly since 1968. At Wabash the shortnose gar densities have increased.

Among the centrarchid basses, the spotted bass (*Micropterus punctulatus*) was the most frequently captured species. At only one sampling station below Terre Haute were they conspicuously abundant, and they were completely absent from collections in much of the upper segment. Populations at the two power plants were comparable to other areas. Most of the spotted bass collected at Cayuga and Wabash and at the intermediate segment were small, young individuals, indicating that these segments may serve as significant spawning and nursery areas. Smallmouth bass (*M. dolomieu*) were mostly limited to the segment above Cayuga where they occurred in greater numbers than spotted bass. Largemouth bass (*M. salmoides*)

were taken in small numbers in all segments, but they were most common at and above Cayuga.

Before 1974 bowfin (*Amia calva*) were rarely taken by electrofishing or netting near the power plants. In 1974, however, they were taken more commonly throughout the study area and were especially prominent below Terre Haute. No obvious attraction or avoidance of the power-plant segments was evident in this species.

Several other species were captured in small numbers unevenly throughout the middle Wabash. Species that were more abundant upstream than downstream included smallmouth and black buffalo fish (*Ictiobus bubalus* and *I. niger*), bluegill (*Lepomis macrochirus*), white crappie (*Pomoxis annularis*), and black crappie (*P. nigromaculatus*). Those species which were more prominent in the lower portion of the study area included skipjack herring (*Alosa chrysochloris*), quillback carpsucker (*Carpiodes cyprinus*), and bigmouth buffalo (*Ictiobus cyprinellus*).

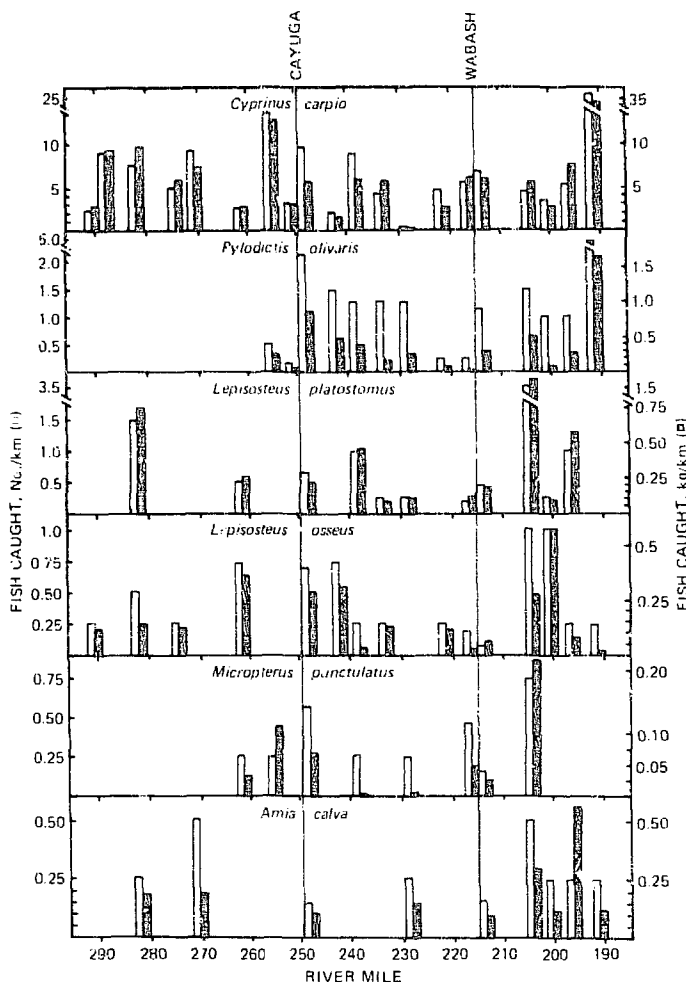


FIG. 5 Number and weight caught per kilometer for several species of fish captured within the study area.

## DISCUSSION

The complex and dynamic nature of river ecosystems affords little comfort to those seeking well-defined, explicit models of running-water communities. A few studies have attempted to correlate changes in fish species assemblages within extended river systems to temperature (Trembley, 1960; Patten, Thompson, and Gronlund, 1970) or to human modification of aquatic environments (Larimore and Smith, 1963; Mills, Starrett, and Bellrose, 1966), but rarely have they attempted to deal with the complex nature of combined effluents as they relate to community structure in fish populations. Evaluation is difficult because environmental conditions, normally variable in rivers over space and time, tend to fluctuate even more as man's development of the river basin proceeds.

If the environmental factors comprising habitat were uniform throughout the ecosystem, communities made up

of similar species would also be distributed uniformly. However, the communities sampled were markedly different from one area to another with no species populations exhibiting uniform distribution. Habitats within the sampling stations were superficially similar, although several turned out to be noticeably different on closer examination.

Among the community parameters measured, intersegmental differences were minor, perhaps because of considerable variability among individual stations and samples. Only the S-W diversities based on weights were uniform among sampling stations and, when subjected to analysis of variance, indicated a significantly lower diversity below Terre Haute than elsewhere. Whitaker and Wallace (1972), in a study of the fishes of Vigo County, Indiana, attributed low catches below Terre Haute to industrial effluents in the Terre Haute region. Thus it appears from the analysis that

TABLE 2. Mean Number of Fish Caught per Kilometer of Electrofishing in Five Segments of the Study Area During 1974

| Common name                     | Scientific name                           | River mile           |                 |                      |                           |                      |
|---------------------------------|-------------------------------------------|----------------------|-----------------|----------------------|---------------------------|----------------------|
|                                 |                                           | 290.5<br>to<br>255.4 | Cayuga<br>249.6 | 241.4<br>to<br>221.5 | Wabash<br>Valley<br>215.4 | 203.2<br>to<br>191.0 |
| Silver lamprey                  | <i>Ichthyomyzon unicuspis</i>             | 0.17                 | 0.02            |                      | 0.05                      | 0.18                 |
| Shovelnose sturgeon             | <i>Scaphirhynchus platyrhynchus</i>       | 0.04                 | 0.05            | 0.05                 | 0.05                      |                      |
| Longnose gar                    | <i>Lepisosteus osseus</i>                 | 0.25                 | 0.52            | 0.30                 | 0.14                      | 0.63                 |
| Shortnose gar                   | <i>L. platostomus</i>                     | 0.29                 | 0.50            | 0.30                 | 0.32                      | 1.19                 |
| Bowfin                          | <i>Amia calva</i>                         | 0.11                 | 0.10            | 0.05                 | 0.26                      | 0.31                 |
| Skipjack herring                | <i>Alosa chrysochloris</i>                | 0.07                 | 0.18            | 0.30                 | 0.76                      | 0.25                 |
| Gizzard shad                    | <i>Dorosoma cepedianum</i>                | 33.93                | 16.47           | 8.10                 | 18.35                     | 9.00                 |
| Goldeye                         | <i>Hiodon alosoides</i>                   | 0.04                 |                 | 0.15                 | 0.42                      | 0.13                 |
| Mooneye                         | <i>H. tergisus</i>                        | 0.04                 | 0.05            | 0.40                 | 0.05                      | 0.06                 |
| Redfin pickerel                 | <i>Esox americanus</i>                    |                      |                 |                      | 0.10                      |                      |
| Carp                            | <i>Cyprinus carpio</i>                    | 5.93                 | 8.10            | 3.95                 | 6.44                      | 9.69                 |
| River carpsucker                | <i>Carpodacus carpio</i>                  | 1.00                 | 0.93            | 0.21                 | 0.64                      | 0.19                 |
| Quillback carpsucker            | <i>C. cyprinus</i>                        |                      |                 | 0.05                 | 0.18                      | 0.06                 |
| Highfin carpsucker              | <i>C. velifer</i>                         |                      | 0.20            |                      | 0.05                      |                      |
| White sucker                    | <i>Catostomus commersoni</i>              | 0.04                 |                 |                      |                           |                      |
| Blue sucker                     | <i>Cycleptus elongatus</i>                | 0.07                 | 0.05            | 0.30                 |                           |                      |
| Hog sucker                      | <i>Hypentelium nigricans</i>              | 0.21                 |                 |                      |                           | 0.06                 |
| Smallmouth and black<br>buffalo | <i>Ictiobus bubalus</i> & <i>I. niger</i> | 0.31                 | 0.31            | 0.15                 |                           | 0.06                 |
| Bigmouth buffalo                | <i>I. cyprinellus</i>                     |                      |                 | 0.15                 | 0.05                      | 0.15                 |
| Silver redhorse                 | <i>Moxostoma anisurum</i>                 | 0.50                 | 0.24            | 0.35                 | 0.12                      |                      |
| Shorthead redhorse              | <i>M. macrolepidotum</i>                  | 0.61                 | 0.29            | 0.80                 | 0.05                      | 0.06                 |
| River redhorse                  | <i>M. carinatum</i>                       | 0.07                 |                 |                      |                           |                      |
| Golden redhorse                 | <i>M. erythrurum</i>                      | 0.82                 | 0.97            | 1.05                 | 0.03                      | 0.13                 |
| Channel catfish                 | <i>Ictalurus punctatus</i>                | 0.18                 |                 | 0.05                 | 0.15                      | 0.25                 |
| Flathead catfish                | <i>Pylodictus olivaris</i>                | 0.07                 | 1.65            | 1.10                 | 0.71                      | 2.06                 |
| American eel                    | <i>Anguilla rostrata</i>                  |                      | 0.11            |                      | 0.05                      |                      |
| White bass                      | <i>Morone chrysops</i>                    | 0.14                 |                 | 0.10                 | 0.25                      | 0.13                 |
| Bluegill                        | <i>Lepomis macrochirus</i>                | 0.25                 | 0.10            | 0.10                 | 0.17                      | 0.06                 |
| Longear sunfish                 | <i>L. megalotis</i>                       | 1.04                 | 0.13            |                      | 0.11                      |                      |
| Smallmouth bass                 | <i>Micropterus dolomieu</i>               | 0.29                 |                 | 0.05                 |                           | 0.06                 |
| Spotted bass                    | <i>M. punctulatus</i>                     | 0.11                 | 0.45            | 0.10                 | 0.32                      | 0.31                 |
| Largemouth bass                 | <i>M. salmoides</i>                       | 0.21                 | 0.22            | 0.05                 | 0.12                      | 0.06                 |
| White crappie                   | <i>Pomoxis annularis</i>                  | 0.32                 | 0.42            | 0.20                 | 0.09                      | 0.19                 |
| Black cr. ppie                  | <i>P. nigromaculatus</i>                  | 0.50                 |                 | 0.05                 | 0.07                      | 0.13                 |
| Sauger                          | <i>Stizostedion canadense</i>             | 0.54                 | 0.42            | 0.30                 | 0.38                      | 0.06                 |
| Freshwater drum                 | <i>Aplodinotus grunniens</i>              | 0.79                 | 0.20            | 0.70                 | 0.34                      | 0.56                 |
| Total                           |                                           | 49.94                | 32.68           | 19.45                | 30.82                     | 26.00                |
| Distance electro-<br>fished, km |                                           | 28.00                | 20.84           | 20.00                | 30.52                     | 16.00                |
| Total number of fish            |                                           | 1456                 | 681             | 389                  | 941                       | 416                  |
| Total number of<br>species      |                                           | 30                   | 28              | 29                   | 32                        | 28                   |

the Terre Haute region exerts a discernable impact on the ichthyofaunal community in the lower part of the study area and possibly beyond.

The only other statistically significant difference occurred above Cayuga with respect to number of fish caught. The increased catch in this segment appears attributable to concentrations of gizzard shad. Further, all community parameters at the two power stations were similar to those from other segments, indicating that no pronounced differences exist at the community level.

At the population level several patterns of response to point-source discharge of pollutants are possible (Fig. 6).

An indigenous species population may elicit (A) a strong positive response, (B) a weak positive response, (C) no response, (D) a weak negative response, or (E) a strong negative response. These are, of course, arbitrary categories within a continuum of potential responses. Theoretically, strong responses should be easily detected, whereas progressively smaller departures from the extremes should become increasingly difficult to identify. These patterns would be especially evident in sedentary species. Similar models involving benthic communities have been widely used in water-quality surveys. Patterns in more mobile species would be less easy to detect, however, and assessment of

TABLE 3 Number of Fish Caught per Kilometer of Electrofishing at the Cayuga Segment, all Stations Included, During 1968 to 1974 (Arrow indicates beginning of plant operations, October 1970)

| Species                                   | 1968  | 1969  | 1970↓  | 1971  | 1972   | 1973  | 1974  |
|-------------------------------------------|-------|-------|--------|-------|--------|-------|-------|
| <i>Scaphirhynchus platyrhynchus</i>       |       |       | 0.01   | 0.08  | 0.69   |       | 0.03  |
| <i>Polyodon spathula</i>                  |       |       |        |       |        | 0.05  |       |
| <i>Lepisosteus osseus</i>                 | 0.81  | 0.88  | 0.40   | 0.20  | 0.69   | 0.37  | 0.37  |
| <i>L. platostomus</i>                     | 1.74  | 1.12  | 0.63   | 0.63  | 0.89   | 0.27  | 0.30  |
| <i>Amia calva</i>                         |       |       | 0.02   | 0.01  | 0.01   |       | 0.08  |
| <i>Alosa chrysochloris</i>                | 0.54  | 0.21  | 0.07   | 0.02  | 0.25   | 0.08  | 0.16  |
| <i>Dorosoma cepedianum</i>                | 17.76 | 7.05  | 11.90  | 8.13  | 30.78  | 11.11 | 34.84 |
| <i>Hiodon alosoides</i>                   | 0.18  | 0.30  | 0.20   | 0.21  | 0.11   |       | 0.06  |
| <i>H. tergisus</i>                        |       | 0.12  | 0.12   | 0.12  | 0.03   | 0.04  | 0.05  |
| <i>Cyprinus carpio</i>                    | 0.37  | 0.53  | 0.70   | 0.29  | 1.26   | 1.10  | 5.43  |
| <i>Coxpiodes carpio</i>                   | 1.63  | 0.77  | 0.93   | 0.42  | 0.93   | 0.80  | 0.61  |
| <i>C. cyprinus</i>                        | 0.01  | 0.26  | 0.12   | 0.05  | 0.12   | 0.04  | 0.07  |
| <i>C. velifer</i>                         | 0.01  | 0.02  | 0.07   |       | 0.01   |       | 0.14  |
| <i>Catostomus commersoni</i>              | 0.06  | 0.02  | 0.02   |       |        |       |       |
| <i>Cycleprus elongatus</i>                | 0.02  | 0.02  | 0.01   | 0.02  | 0.02   | 0.08  | 0.04  |
| <i>Hypentelium nigricans</i>              | 0.11  |       | 0.05   |       |        |       |       |
| <i>Ictiobus bubalus</i> & <i>I. niger</i> | 0.11  | 0.09  | 0.06   | 0.02  | 0.09   | 0.17  | 0.17  |
| <i>I. cyprinellus</i>                     | 0.05  | 0.07  | 0.03   |       | 0.03   | 0.05  | 0.15  |
| <i>Moxostoma anisurum</i>                 | 0.60  | 0.40  | 1.42   | 0.19  | 0.16   | 0.04  | 0.23  |
| <i>M. macrolepidotum</i>                  | 1.94  | 1.35  | 0.76   | 0.71  | 0.58   | 0.22  | 0.39  |
| <i>M. carinatum</i>                       |       |       | 0.01   |       |        |       |       |
| <i>M. erythrum</i>                        | 1.05  | 1.44  | 0.76   | 0.35  | 0.69   | 0.60  | 0.68  |
| <i>Ictalurus punctatus</i>                | 0.08  | 0.14  | 0.10   | 0.03  | 0.01   |       |       |
| <i>Pylodictus olivaris</i>                | 0.10  | 0.07  | 0.04   | 0.09  | 0.29   | 0.95  | 0.93  |
| <i>Anguilla rostrata</i>                  | 0.06  |       | 0.02   | 0.16  | 0.01   | 0.03  | 0.04  |
| <i>Morone chrysops</i>                    | 0.07  | 0.05  | 0.13   | 0.04  | 0.17   | 0.04  | 0.04  |
| <i>Ambloplites rupestris</i>              | 0.07  | 0.05  |        |       | 0.01   |       |       |
| <i>Lepomis macrochirus</i>                | 0.02  | 0.05  | 0.03   | 0.02  | 0.03   | 0.04  | 0.08  |
| <i>L. megalotis</i>                       | 0.62  | 0.16  | 0.06   |       | 0.03   |       | 0.10  |
| <i>Micropterus dolomieu</i>               | 0.14  | 0.28  | 0.07   |       | 0.01   |       | 0.03  |
| <i>M. punctulatus</i>                     | 0.65  | 0.35  | 0.11   |       | 0.04   | 0.19  | 0.33  |
| <i>M. salmoides</i>                       |       |       |        |       | 0.02   | 0.10  | 0.11  |
| <i>Pomoxis annularis</i>                  | 0.06  | 0.02  | 0.11   | 0.09  | 0.07   | 0.11  | 0.26  |
| <i>P. nigromaculatus</i>                  |       | 0.02  | 0.03   |       |        | 0.05  |       |
| <i>Stizostedion canadense</i>             | 0.99  | 0.70  | 0.63   | 0.30  | 0.43   | 0.13  | 0.30  |
| <i>Aplodinotus grunniens</i>              | 0.18  | 0.30  | 0.07   | 0.05  | 0.09   | 0.08  | 0.14  |
| Total                                     | 29.03 | 16.84 | 18.71  | 11.20 | 37.91  | 16.75 | 46.38 |
| Distance electrofished, km                | 61.60 | 43.00 | 166.51 | 66.30 | 112.95 | 24.12 | 41.64 |
| Total number of fish caught               | 1836  | 727   | 3115   | 1028  | 4377   | 413   | 1984  |
| Total number of species                   | 29    | 29    | 34     | 25    | 31     | 25    | 29    |

population levels would demand a refined study over extended periods of time to recognize significant changes. Influence of climatic patterns should also be measured in these cases.

If only the impact of the Cayuga plant, the Wabash plant, and the Terre Haute complex are considered, to the exclusion of such other inputs as towns, isolated industries, and streams, the following trends are evident. The sedentary flathead catfish responded strongly and positively to thermal additions at the Cayuga plant after its start-up in 1970 (Funk, 1954; Gammon, 1973). The pattern of response found in 1974 resembles closely that of curve A (Fig. 6). Longear sunfish, another relatively sedentary species (Gerking, 1953), exhibited a pattern similar to curve D, with obvious declines in thermally enriched areas.

Gizzard shad and carp display no obvious response to local conditions within the river, but patterns would seem to indicate that present levels of thermal loading and other pollution are neither detrimental nor beneficial to the numbers. Sauger gives the appearance of being somewhat depressed at the Cayuga and Wabash plants and almost totally depressed below Terre Haute. The River carpsucker population appears to be mildly attracted to power-plant segments and slightly depressed below Terre Haute. Golden shiner seems comparably high at Cayuga, but shows little tolerance for conditions at Wabash and below Terre Haute. Freshwater drum seems slightly inhibited near the power plants and below Terre Haute. Spotted bass were captured in small numbers, but they seem to be attracted to warm areas perhaps for reproductive activity. The two species

TABLE 4 Number of Fish Caught per Kilometer of Electrofishing at the Wabash Plant, all Stations Included, During 1968 to 1974

| Species                                     | 1968  | 1969  | 1970  | 1971  | 1973  | 1974  |
|---------------------------------------------|-------|-------|-------|-------|-------|-------|
| <i>Scaphirhynchus platyrhynchus</i>         |       | 0.02  | 0.09  |       |       | 0.08  |
| <i>Polyodon spathula</i>                    |       |       |       | 0.05  |       |       |
| <i>Lepisosteus osseus</i>                   | 0.52  | 1.12  | 0.43  | 0.63  | 0.89  | 0.54  |
| <i>L. platostomus</i>                       | 0.59  | 0.61  | 0.35  | 1.72  | 1.43  | 1.28  |
| <i>Amia calva</i>                           |       |       |       |       |       | 0.35  |
| <i>Alosa chrysochloris</i>                  | 0.74  | 0.85  | 0.09  | 0.09  |       | 0.47  |
| <i>Dorosoma cepedianum</i>                  | 21.67 | 21.92 | 5.89  | 24.55 | 9.34  | 18.69 |
| <i>Hiodon alosoides</i>                     | 0.15  | 0.31  | 0.56  | 1.02  | 0.27  | 0.23  |
| <i>H. tergisus</i>                          |       | 0.07  |       | 0.12  | 0.04  | 0.04  |
| <i>Esox americanus</i>                      |       |       |       |       |       | 0.08  |
| <i>Cyprinus carpio</i>                      | 0.59  | 0.70  | 1.17  | 1.02  | 2.06  | 8.07  |
| <i>Carpiodes carpio</i>                     | 2.07  | 1.65  | 0.56  | 1.78  | 0.12  | 0.62  |
| <i>C. cyprinus</i>                          | 0.07  | 0.15  |       |       | 0.04  |       |
| <i>C. velifer</i>                           |       |       | 0.04  |       |       | 0.04  |
| <i>Cycleptus elongatus</i>                  |       | 0.02  |       |       |       |       |
| <i>Hypentelium nigricans</i>                |       | 0.02  |       |       |       |       |
| <i>Ictiobus bubalus</i> and <i>I. niger</i> | 0.07  | 0.05  | 0.04  | 0.03  | 0.12  | 0.08  |
| <i>I. cyprinellus</i>                       | 0.07  | 0.10  | 0.04  | 0.12  |       |       |
| <i>Moxostoma anisurum</i>                   |       |       |       |       | 0.04  | 0.04  |
| <i>M. macrolepidotum</i>                    | 0.07  | 0.03  | 0.04  | 0.12  |       |       |
| <i>M. erythrum</i>                          | 0.30  | 0.03  | 0.04  | 0.03  |       | 0.16  |
| <i>Ictalurus punctatus</i>                  | 0.59  | 0.12  | 0.22  | 0.06  | 0.12  | 0.08  |
| <i>Pylodictus olivaris</i>                  | 0.44  | 0.10  | 0.56  | 0.15  | 0.70  | 1.32  |
| <i>Anguilla rostrata</i>                    |       |       |       |       | 0.04  | 0.04  |
| <i>Morone chrysops</i>                      | 0.07  | 0.07  | 0.09  | 0.06  | 0.08  | 0.08  |
| <i>Lepomis cyanellus</i>                    | 0.07  | 0.02  |       |       |       |       |
| <i>L. macrochirus</i>                       |       | 0.03  |       |       |       | 0.04  |
| <i>L. megalotis</i>                         | 0.07  |       |       |       |       | 0.23  |
| <i>Micropterus dolomieu</i>                 |       | 0.03  |       |       |       |       |
| <i>M. punctulatus</i>                       | 0.30  | 0.05  | 0.13  | 0.06  |       | 0.16  |
| <i>M. salmoides</i>                         |       |       |       |       |       | 0.12  |
| <i>Pomoxis annularis</i>                    | 0.07  |       | 0.04  | 0.03  |       | 0.08  |
| <i>Stizostedion canadense</i>               | 0.96  | 0.24  | 0.04  | 0.03  | 0.08  | 0.08  |
| <i>Aplodinotus grunniens</i>                | 0.22  | 0.05  |       | 0.03  |       | 0.23  |
| Total                                       | 33.51 | 28.28 | 10.44 | 30.69 | 15.35 | 33.19 |
| Distance electrofished, km                  | 13.52 | 58.83 | 23.08 | 33.20 | 25.79 | 25.79 |
| Total number of fish caught                 | 453   | 1664  | 241   | 1019  | 396   | 856   |
| Total number of species                     | 21    | 25    | 19    | 20    | 15    | 26    |

gar, as well as bowfin, find the region below Terre Haute particularly conducive to their needs, but this may be due to the proximity to adjacent bayous downstream which function as reproductive habitats (Whitaker and Wallace, 1972).

This simplistic analysis of species populations, however, ignores the complications imparted by at least four basic realities: (1) that many species are considerably more mobile than others; (2) that not all contributions to the river are point-source discharges; (3) that adjacent sources may be sufficiently close so as to reinforce similar positive or negative effects, or even to neutralize each other; and (4) that shifting climatic patterns may produce a new population equilibrium.

The analysis of variance of community parameters confirmed our impression in the field that attempts to select similar sampling stations were not always successful. A few sampling stations, despite being located on the

outside of moderate river bends, were a good deal deeper and more sluggish than others. We believe that both attributes resulted in a certain degree of unrepresentative sampling and contributed markedly to the observed variability between stations. Refinements in this sampling program should include careful determinations of mean depth and velocity of current, as well as an evaluation of electrofishing effectiveness, which may be biased in a number of respects.

Although fry and smaller species of fish were excluded from the analysis owing to ineffective capture, fish larger than 6 kg were also absent from collections, even though such are not uncommon in the Wabash. It is possible that the increased body surface of larger fish enables them to detect the electric field at greater distances and thus escape before a galvanotaxic response is imposed. It may also be that larger fish inhabit deeper waters than can be effectively electrofished. Various pulse rates and voltage settings may



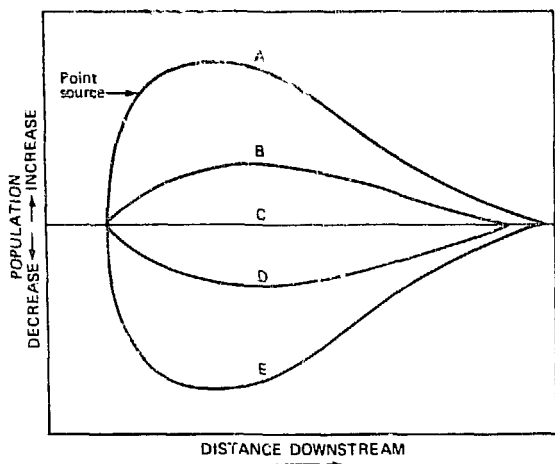


FIG. 6 Theoretical responses of sedentary fish populations to point-source effluents.

determine efficiency in capturing fish (Taylor, Cole, and Sigler, 1957). Novotny and Priegel (1974) have found that optimal electrical impulse rates are higher in such species as trout, carp, bullhead, largemouth bass, and smallmouth bass, whereas walleye, yellow perch, bluegill, white bass, and yellow bass are more susceptible to lower pulse rates. Electrode configuration may also importantly influence effectiveness.

One of the purposes of this study was to relate, if possible, the populations of fish to environmental conditions zoned within the middle Wabash River. The community of fishes throughout the study segment was clearly dominated by gizzard shad and carp, which were comparably numerous in all locations and were neither strongly depressed nor concentrated unduly by any of the three major influences. The electric-power generating stations influenced several species populations, though to different degrees. Negative responses to the generating stations were strongly elicited by longear sunfish populations and noticeably by redear and sauger. The influence of Terre Haute virtually eliminated sauger in the lower study section. A much more local negative response to generating plants was evident in drum populations, which were also depressed below Terre Haute.

A clear positive response to the generating plants is apparent only for the flathead catfish. Weak, local attraction to heated portions of the river appears characteristic of carp, longnose and shortnose gar, and bowfin. The gar and bowfin populations flourish especially well below Terre Haute, perhaps because of the proximity to reproductive bayous and not because of positive enhancement of the physical and chemical environment.

With few exceptions, this shift in species composition occurs without changing important community characteristics (e.g., abundance and diversity) and represents adaptations of the community along an environmental gradient. In

essence, changes within species populations occur, but the basic integrity of the community is maintained.

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# Seasonal Distribution and Abundance of Ohio River Fishes at the J. M. Stuart Electric Generating Station

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## ABSTRACT

Distributions of Ohio River fishes were studied in the vicinity of a 2400-Mw coal-fired electric generating plant. Three thermally elevated zones, two ambient river zones, and a backwater zone were sampled intensively from June 1974 through February 1975 following the completion of all four units of the power plant. Less intensive collections were made preceding and during construction from 1970 to 1973. Overall variations in water temperature in 1974 to 1975 ranged from 6 to 40°C in the heated zones, 4 to 30°C in the ambient river zones, and 5 to 27°C in the backwater zone. Seasonal and spatial differences in abundance, diversity, and faunal associations were largely influenced by temperature. Notable changes in species populations from 1970 to 1975 were observed which were attributed to power-plant operation. Although seasonal definitions in terms of summer, fall, and winter were generally used, they were of very limited value, as demonstrated by annual fluctuations in community parameters. Apparently near-freezing temperatures in the ambient river zones, as well as high summer temperatures in the effluent canal, limit the time fish can spend in these areas and force them to seek more hospitable temperatures. This suggests that there are critical winter as well as summer months, with spring/fall transitional periods in between in the vicinity of thermal effluents.

As the demand for electric-power generation increases, so does the problem of waste-heat disposal. Along the Ohio River 35 electric generating plants with a total capacity of 25,000 to 26,000 Mw use river water for condenser-cooling purposes, the majority with a once-through cooling system. By 1985 it is estimated the generating capacity will rise to over 35,000 Mw (Butz et al., 1974).

The relationship of temperature to fish has been studied with varied emphasis and methods by many investigators. Early investigators studied fish tolerance to elevated temperature in the laboratory (Fry, 1947; Pitt, Garside, and Hepburn, 1956; Brett, 1956). More recent laboratory

studies have dealt with physiological response to elevated temperature (Cairns, 1956), temperature shock and avoidance (Meldrim and Gift, 1971), and seasonal temperature selectivity (Barans and Tubb, 1973; Otto, 1973). There has also been an increased effort to determine fish response to temperature in the field under both ambient (Dendy, 1945; Ferguson, 1958) and thermally elevated conditions in lakes and streams (Dryer and Benson, 1957; Trembley, 1960, 1961; Alabaster, 1962; Proffitt, 1969; Proffitt and Benda, 1971; Gammon, 1971, 1973) during the critical summer months. Other investigators have touched on seasonal trends (Churchill and Wojtalik, 1969; Barkley and Perrin, 1972; Neill and Magnuson, 1974; McNeely and Pearson, 1974) and have related seasonal and spatial differences in abundance, diversity, and faunal associations to temperature (Recksiek and McCleave, 1973).

This paper describes the seasonal, spatial, and temperature-related differences in abundance, diversity, and ichthyofaunal associations in the vicinity of the J. M. Stuart electric generating plant observed during 1974 to 1975. A short analysis of data collected in 1970 (Norris and Gammon, 1971), 1971 and 1972 (Hatch and Gammon 1973), and 1973 (Lesniak and Gammon, 1974) is included with the analysis of the 1974 to 1975 collections.

## LOCATION AND DESCRIPTION OF STUDY AREA

The Ohio River drains an area covering 3.34 by 10<sup>7</sup> ha of the eastern and midwestern United States and is the major eastern artery of the Mississippi River basin. From the river's origin in Pennsylvania to its confluence with the Mississippi, it is impounded by navigational dams that have created a series of narrow, deep, flowing, thermally unstratified lakes with attendant backwaters at the mouths of tributaries.

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# The Macrobenthic Community of a Thermally Altered Area of Tampa Bay, Florida

Norman J. Blake  
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Thomas E. Pyle

## ABSTRACT

An ecological study in the vicinity of the P. L. Bartow Power Plant (482 Mw, maximum  $\Delta T = 7.2^{\circ}\text{C}$ ) included surveys of physical and chemical parameters at approximately monthly intervals. Assuming that seagrasses were originally present on the discharge side of the plant, as they are now on the intake side, over 81 ha of seagrasses have been destroyed. Only small patches of *Halodule wrightii* now grow near the outer edges of the thermal plume ( $\Delta T = 3^{\circ}\text{C}$ ). *Thalassia testudinum* transplanted to the discharge area failed to survive for 30 days where mean water temperature was  $31^{\circ}\text{C}$  or greater. A summer survey of the benthic invertebrates yielded 104 species of polychaetes, molluscs, crustaceans, and echinoderms from the intake side of the plant and 60 species from the discharge side. The difference was not attributable to detectable variations in sediments.

The food webs of most tropical and subtropical estuaries are largely dependent on the productivity of benthic macrovegetation such as *Thalassia testudinum*. In addition, such seagrasses serve as a substrate for epiphytes, which may contribute significantly to the primary production (Humm, 1964). Nutrient cycling within the macrophytes appears to be rapid since a *Thalassia* blade may reach maturity and disintegrate, along with the epiphytes, to detritus all within 10 weeks (Thorhaug, 1974). Fenchel (1970), who studied the decay process of *Thalassia*, concluded that the organic matter entered the macroinvertebrate assemblage primarily through detrital-feeding amphipods. It is apparent, therefore, that thermal additions that even slightly alter the macrophytic productivity may affect the faunal components of the community.

Tampa Bay, which is located on the west coast of Florida, contains macrophytic components typical of a subtropical environment and macroinvertebrate components typical of a temperate Carolinian environment. Our study was designed to assess the present condition of the

macrobenthic community adjacent to a fossil-fueled power plant located on this estuary.

## STUDY AREA

The P. L. Bartow Power Plant of the Florida Power Corporation is located near the mouth of Old Tampa Bay (Fig. 1). This plant, which in its present capacity has been operable since 1963, generates a maximum of 482 Mw of power, heats 390,000 gpm of cooling water with a temperature rise of  $7.2^{\circ}\text{C}$ , and releases the water at 83.8 cm/sec. Maximum recorded intake temperature is  $33.5^{\circ}\text{C}$ , and maximum discharge temperature is  $40.7^{\circ}\text{C}$ .

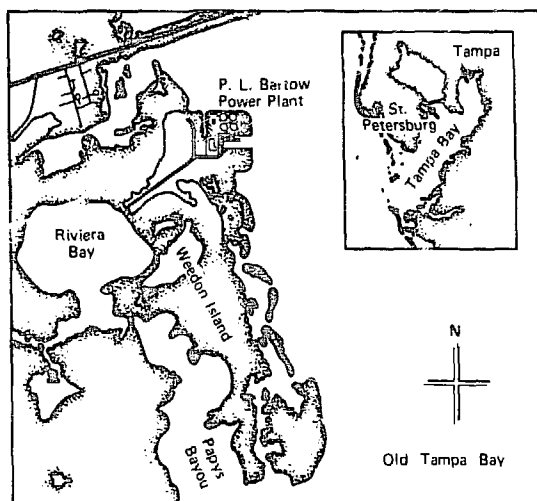


FIG. 1 Geographic setting of the P. L. Bartow power plant.

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effluent canal in winter indicates a response to the warmer water since fish selected the highest temperatures available.

Skipjack herring occurred equally in all zones except the effluent canal in fall (Fig. 6) and over a wide range of temperature, 16 to 30°C. Winter catches were relatively low and restricted to the effluent canal and thermal river zones. Gammon (1973) found skipjack herring preferred temperatures of 26 to 28.5°C. Trautman (1957) observed fish in the Ohio River to school in the middle near the surface where they fed on the vast schools of emerald and minnie shiners (*Notropis velucellus*) present there. Dryer and Benson (1957) correlated the seasonal abundance of this species directly with that of threadfin shad. This also was the case in this study, as individuals were observed feeding on the large schools of emerald shiners and young-of-the-year gizzard shad present in all zones in fall. The decline in numbers of skipjack herring in winter corresponded well with the decline in abundance of forage species. However, some individuals were attracted to the effluent area, most likely in response to the cooling of the river in winter.

Freshwater drum were scarce in all zones during summer and fall. The majority were collected at 6 to 11°C in February in the thermal zones.

Sauger avoided the thermally affected zones in summer and occurred in low numbers in the ambient river zones in fall (Fig. 6). Highest catches were observed in the thermal river zones and effluent canal in winter. The avoidance of the thermal zones in summer and fall agrees well with Gammon (1973), but the fall and winter ranges of 14 to 21°C and 8 to 11°C find little support in the literature, being well below the 26 to 28°C preferendum defined by Gammon (1973). The 19.2°C optimum defined for Norris Lake fish (Dendy, 1945) seems more applicable to this study but is well above the temperatures at which the majority of sauger were collected.

Both spotted (*Minytrema melanops*) and white suckers (*Catostomus commersoni*) were observed over the same temperature ranges of 25 to 27°C in summer and 16 to 19°C in fall and were restricted to the backwater zone. Golden redhorse were occasionally collected in the ambient river zones and in small numbers in the backwater zone. This has been the most deleteriously affected species in this study; it was found in all zones including the effluent canal before plant start-up in 1970. Gammon (1971, 1973) also found both golden and shorthead redhorse (*Moxostoma macrolepidotum*) to be substantially reduced from all areas below the thermal discharge. Barkley and Perrin (1972) made a similar observation for spotted suckers in Lake Catherine. Minckley (1963) defines backwater areas as the optimum habitat for the spotted sucker in the Ohio River. The final preferendum of 26 to 27.5°C (Gammon 1973) for *Moxostoma* may also apply to the other two sucker species in this study. Brett (1956) cites the catostomids as being the least heat tolerant of all North American fishes, except for salmonids. Trembley (1960) observed white suckers to avoid all thermal areas in summer and fall. The absence of

these species in winter may be due to an upstream migration in the tributaries away from the backwater areas since these migrations for spawning purposes have been observed as early as January (McSwain and Gennings, 1972).

These findings are similar to others in locations where the annual temperature variations were similar (Trembley, 1960, 1961; Gammon, 1971, 1973; Neill and Magnuson, 1974). The various habitats of the Ohio River were visited and occupied by fish on a seasonal basis in which temperature was the primary influential factor. However, the concentration of forage species also attracted predator species to certain areas. Dryer and Benson (1957) and Barkley and Perrin (1972) felt that movements and distribution of predator species was largely dictated by the availability of forage species, but the range of thermal conditions available was significantly less than in the other studies.

Most fish were eliminated from the effluent canal in summer and were most numerous in the backwater and ambient river zones; only those species with high summer temperature optimums remained in and near the effluent area. Density and biomass increased in all zones, except the effluent canal, with the advent of cooler temperatures and an abundance of forage species in fall. The reverse of summer events took place in winter. Most species vacated the colder ambient river zones and moved into the warmer effluent canal, thermal river, and backwater zones. Biomass increased especially in these areas owing to the increase in numbers of such species as carp, river and quillback carpsucker, and smallmouth buffalo.

Natural seasonal temperature variations appear to play an important role in the normal distributions of fish and their movements between the river proper and backwater areas. The backwaters are cooler in summer and warmer in winter than the river proper. This is especially important in winter when river temperatures plummet to near-freezing levels. Fish leave the river at this time and enter the backwater areas, reentering the river in spring when temperatures become warmer. Species with low temperature optima, such as spotted and white suckers and golden redhorse, use the backwaters year-round. The thermal river zones and effluent canal are warmer in winter than both ambient river and backwater zones. Species are attracted that would otherwise use the backwater zone in winter, and this accounts for the winter increases in diversity, density, and biomass in the thermally altered zones from November through February.

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Gizzard shad were the most numerous species overall, the largest catches occurring in the ambient river zones in November (Fig. 6). They avoided the effluent canal in all seasons. The observed summer temperature range of 26 to 34°C is slightly higher than the 28.5 to 31°C range observed by Gammon (1973) and may be due to the age/size differences between the two studies. The 28.5 to 31°C range was derived from individuals averaging 240 mm, whereas the range of 26 to 34°C was derived from fish averaging near 190 mm. The attraction of fish to all zones except the effluent canal in fall and an avoidance of all zones except the backwater in winter indicates an avoidance of the warmer summer and colder winter temperatures.

Bluegill dominated D net catches in every zone except the effluent canal in fall (Fig. 7) and were collected over a wide range of temperature. There was a preference for the thermally elevated areas in fall but for the backwater zone in winter. An upper temperature tolerance of 34°C agrees well with other field results (Proffitt and Benda, 1971; Neill and Magnuson, 1974), as does the bluegill's wide temperature range (McNeely and Pearson, 1974). It has a weakly developed preference for fall temperatures of 14 to 24°C and avoids the colder ambient river temperatures in winter.

Quillback carpsuckers were attracted to the thermal river zones in summer and winter with temperature ranges of 26 to 32°C and 10 to 16°C. This agrees with the 25 to 32°C range and 29 to 31°C optimum derived by Gammon (1973) and indicates a preference for warmer temperatures in all seasons.

River carpsuckers were well-distributed throughout the study area and selected different zones seasonally (Fig. 6). The summer temperature range of 26 to 32°C is somewhat lower than the defined optimum of 31.5 to 34.5°C (Gammon, 1973). This may be due again to the age/size factor in that those individuals from which the optimum of 31.5 to 34.5°C was derived were mostly young (Gammon 1971, 1973) but those collected in this study were entirely large adults. Fish preferred the backwater and thermal river zones in summer. Catches were relatively low in fall, and the highest catches occurred in the effluent canal in winter. These observations, along with the fall and winter temperature ranges of 16 to 22°C and 12 to 16°C, indicate the fish had a weak thermal response in summer and fall but had an overwhelming attraction to the warmest winter temperatures available in avoidance of the colder ambient river zones. This response accounts for the increase in the river carpsucker population since plant operation began.

Smallmouth buffalo were distributed much the same as carp and river carpsucker (Fig. 6). They were collected in every zone except the effluent canal in summer. Catches were relatively low in fall, and the highest catches were observed in the thermal river, effluent canal, and backwater zones in winter. The summer temperature range of 22 to 32°C is lower than the optimums of 31 to 34°C (Gammon, 1973) and 34°C (Proffitt and Benda, 1971) and again was

based on large fish. This species responded to thermal additions in the same manner as carp and river carpsucker by avoiding the hottest zones in summer and fall and by being attracted to the warmest temperatures in winter.

White crappie were second only to bluegill in D net catches in all zones (Fig. 7). The fall temperature range of 18 to 26°C agrees more closely with the optimum 27 to 28.5°C derived by Gammon (1973) than does the summer range of 26 to 31°C. Highest catches occurred in the backwater zone at 5 to 8°C, well below the defined optimum, but that temperature range can hardly be termed the preferred temperature range. It is more likely that fish were seeking the warmer backwater temperatures in avoidance of the colder river, but it is puzzling that no catches were made in the warmer effluent canal.

Largemouth bass were collected only in summer and fall and selected temperatures of 24 to 31°C and 18 to 21°C, respectively. Existing information varies, but a summer optimum of 26 to 28°C (Ferguson, 1958) seems reasonable. Other field studies show higher ranges of 29 to 31°C (Otto, 1973) and 30 to 32°C (Neill and Magnuson, 1974). Fish avoided the effluent area in both seasons, preferring the backwater zone in summer and ambient river zones in fall. The increased numbers of forage species, e.g., young-of-the-year gizzard shad and emerald shiners (*Notropis atherinoides*) may have attracted fish to the river zones in fall. Kutkuhn (1968) found largemouth bass stomachs to contain over 80% gizzard shad by volume. Spotted bass responded much the same as largemouth bass, except that they preferred the river zones in summer and were observed in the effluent canal in winter.

White bass exhibited a very distinct seasonal trend, with highest catches in the thermal and ambient river zones in fall and in the effluent canal in winter (Fig. 6). Summer catches were relatively low and occurred mainly in the ambient river zones. The summer temperature range of 26 to 29°C agrees with the 28 to 29.5°C final preferendum defined by Gammon (1973). Barans and Tubb (1973), in seasonal laboratory tests with adult and juvenile Lake Erie fish, found four distinct seasonal temperature optimums of 30 to 32°C (summer), 14 to 25°C (fall), 19 to 25°C (winter), and 16 to 21°C (spring) for 80% of adult test fish acclimated to field temperatures. These results, based solely on temperature selection, agree somewhat with the fall range of 16 to 28°C in this study. The winter range of 12 to 16°C in this study is much lower than that for the Lake Erie fish and can be attributed to the unavailability of temperatures higher than 16°C. Individuals observed in fall were larger than those observed in summer and winter, when catches were comprised of smaller fish. This lends support to the inference that the fall peaks were due partly to the increased availability of forage species in the river zones in fall. Barkley and Perrin (1972) found in Lake Catherine the numbers of white bass to be correlated directly with the seasonal occurrence of threadfin shad (*Dorosoma petenense*). However, the preference for the

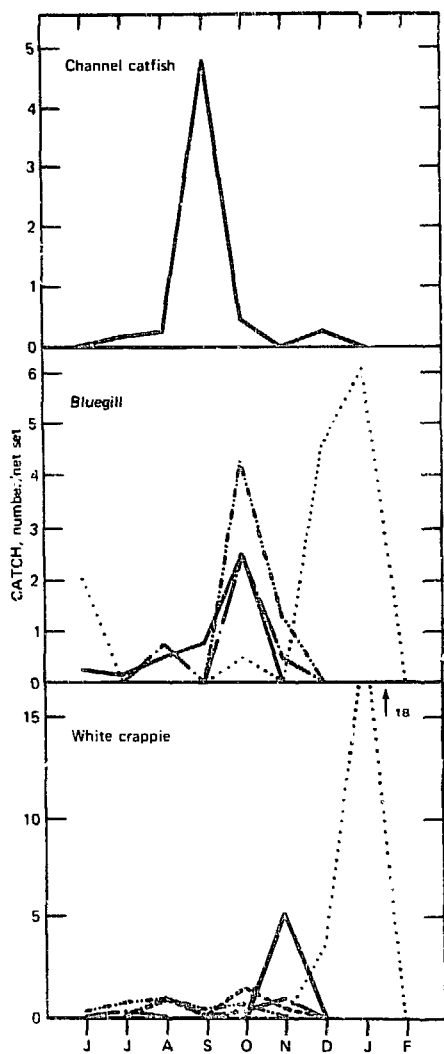


FIG. 7 Seasonal abundance for three species collected in D nets in zone 1 (—), zone 2 (---), zone 3 (-.-), zone 4 (.....), and zone 5 (.....), 1974 to 1975.

Benda, 1971). These are all summer observations and closely agree with the 30 to 34°C found in this study.

Channel catfish were collected in D nets in the effluent canal in summer and fall (Fig. 7) and by electrofishing in winter. The summer and autumn temperature ranges of 32 to 36°C and 30 to 32°C, respectively, and a preference for the effluent canal indicates a response to temperature. Gammon (1973) observed the highest catches of channel catfish in hoop nets at 33°C and defined a final preferendum of 30 to 32°C. This agrees with the fall range of 30 to 32°C in this study when catches were highest, suggesting that this range is optimum.

Winter electrofishing catches of channel catfish were highest during one collection period in February after a

TABLE 3 Temperatures (°C) Selected Seasonally by 18 Species Captured by Electrofishing and in D Nets, 1974 to 1975

| Species              | Summer | Fall  | Winter |
|----------------------|--------|-------|--------|
| Channel catfish      | 32–36  | 30–32 | 9–14   |
| Flathead catfish     | 24–36  | 18–29 |        |
| Longnose gar         | 30–34  | 24–28 | 12–16  |
| Carp                 | 26–34  | 16–20 | 5–16   |
| Gizzard shad         | 26–34  | 10–22 | 4–10   |
| Bluegill             | 22–34  | 14–24 | 5–8    |
| Quillback carpsucker | 26–32  |       | 10–16  |
| River carpsucker     | 26–32  | 16–22 | 12–16  |
| Smallmouth buffalo   | 22–32  | 18–26 | 6–14   |
| White crappie        | 26–31  | 18–26 | 5–8    |
| Largemouth bass      | 24–31  | 18–21 |        |
| White bass           | 26–29  | 16–28 | 12–16  |
| Skipjack herring     |        | 16–30 | 10–16  |
| Freshwater drum      |        | 22–30 | 6–11   |
| Spotted bass         |        | 16–21 | 6–15   |
| Sauger               |        | 14–21 | 8–11   |
| Spotted sucker       | 25–27  | 16–19 |        |
| Common white sucker  | 25–27  | 16–19 |        |

brief rise in water level and a rise in temperature. The majority of individuals collected were large adults, averaging 425 mm in length, taken in the thermal river zones. All fish were covered with a film of mud, indicating that they had recently emerged from a period of inactivity on the bottom. Harlan and Speaker (1951) found that fish in Iowa rivers increased activity as water levels rose, but they found that the activity was dependent upon temperature. They also found that there was no activity below 5°C and that full-scale movements did not begin until temperatures reached 15°C in spring. The catch of channel catfish in February may indicate an early spring response at a temperature of about 10°C and was limited to the thermal river zones.

Flathead catfish were taken in small numbers in both the ambient and thermal river zones by electrofishing and in the effluent canal in D nets. Gammon (1973) found that flathead catfish populations increased because of improved reproductive success below power plants on the Wabash River. He found a preferendum of 31.5 to 33.5°C, which is close to the 24 to 36°C range in this study. However, a fall range of 18 to 29°C and an even distribution of fish throughout the river zones indicates no definable temperature response; yet individuals were collected at high temperatures in the effluent canal as previously mentioned. Further sampling may prove inconclusive since this species has never been numerous in the study area (Table 1).

Except for their absence in the effluent canal in summer and fall, carp were equally represented in all zones; they preferred the effluent canal and backwater zone in winter. The summer preferendum of 26 to 34°C agrees with the 33 to 35°C range observed by Gammon (1973), and a preference for 32°C (Pitt, Garside, and Hepburn, 1956; Alabaster, 1962; Neill and Magnuson, 1974) seems reasonable.



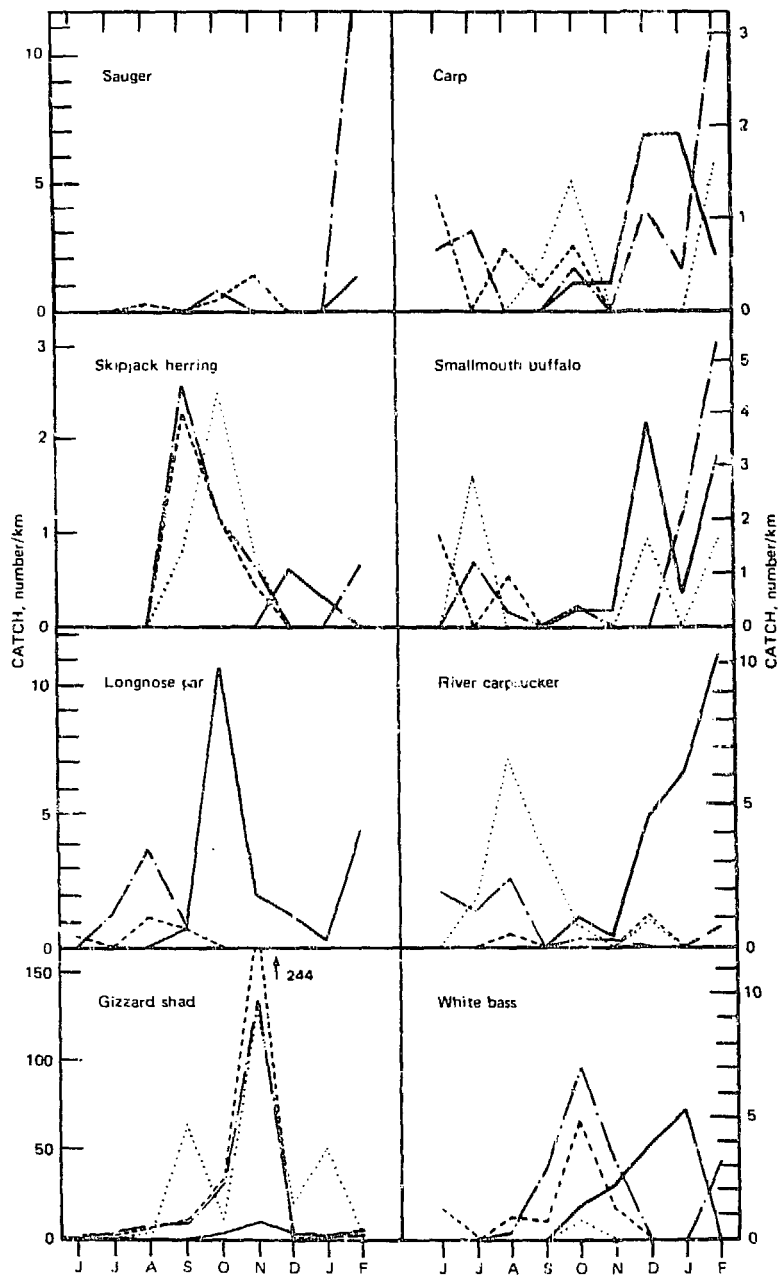


FIG. 6 Seasonal abundance for eight species collected by electrofishing in zone 1 (—), zones 3 and 4 (---), zones 2 and 6 (---), and zone 5 (....), 1974 to 1975.

16°C in winter (December to February). Available seasonal temperatures of 24 to 40°C in summer, 11 to 33°C in fall, and 4 to 16°C in winter indicate that longnose gar were selecting nearly the highest temperatures available in each season. This, along with the observations that they selected the thermally affected areas (Fig. 6) and appeared in the

study area immediately after 1 year of plant operation (Table 1), indicates that temperature is the major influence in their distribution. A final preferendum of 33 to 35°C was defined for Wabash River longnose gar (Gammon, 1973), 30 to 32°C in Lake Monona (Neill and Magnuson, 1974), and near 34°C in the White River (Proffitt and

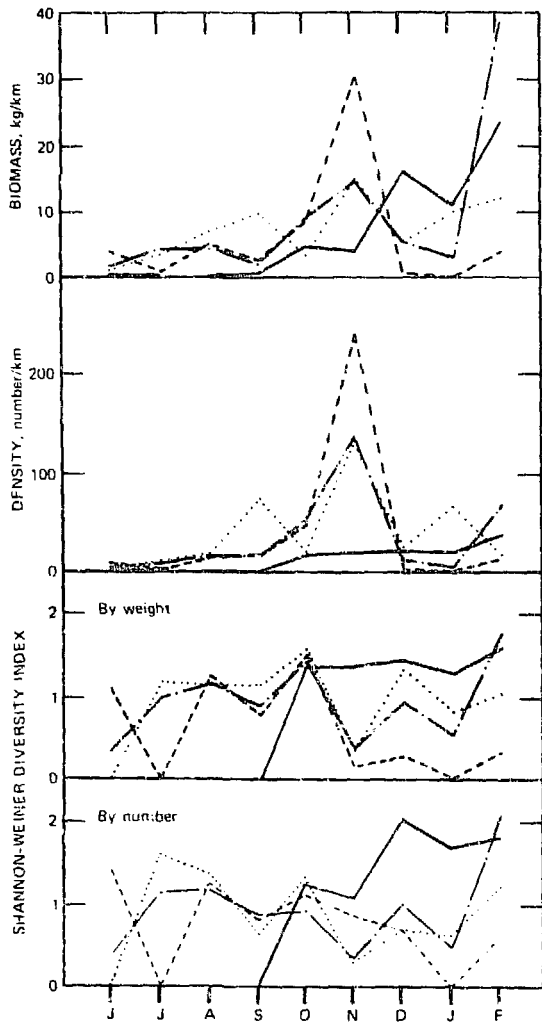


FIG. 4 Seasonal diversity by number and weight, density, and biomass in zone 1 (—), zones 3 and 4 (---), zones 2 and 6 (---), and zone 5 (....) for electrofishing catches, 1974 to 1975.

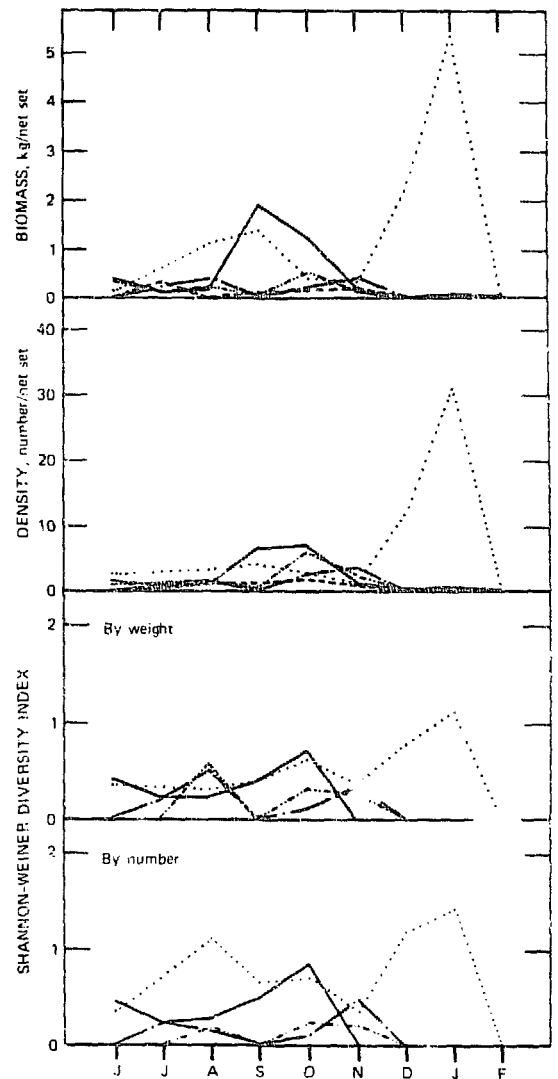


FIG. 5 Seasonal diversity by number and weight, density, and biomass in zone 1 (—), zone 2 (---), zone 3 (---), zone 4 (-.-.-), and zone 5 (....), for D net catches, 1974 to 1975.

1956; Pennsylvania Department of Health, 1962). Fish respond readily to spatial and temporal variations of temperature in natural rivers and are capable of selecting specific temperatures that are optimum for various ecological and physiological processes (Alabaster, 1962; Gammon, 1973); such responses may not be subject to food availability within or outside their optimum temperature range (Neill and Magnuson, 1974).

Responses to seasonal and spatial variability in temperature, food availability, and habitat were demonstrated by all the major species collected. A wide range of temperature was available in the study area, but the presence of three distinct types of habitat (effluent canal, river zones, backwater) and a seasonal occurrence of forage

species make strict definitions of optimum temperature preferences difficult. However, each species was distributed over defined temperature ranges seasonally (Table 3), but the reasons for this varied among species. In this study, fish responded primarily to temperature, food availability, and habitat and, in some instances, to water level.

Seasonal temperature ranges for each species were defined by observing the temperatures within which the majority of the seasonal catch of each species occurred. For example, the majority of longnose gar were collected at temperatures of 30 to 34°C in summer (June to August), 24 to 28°C in fall (September to November), and 12 to

TABLE 2 Average Number of Each Species per Net Set Caught in D Nets, Summer–Fall 1970 to 1974

| Species                                               | 1970 | 1971 | 1972 | 1973 | 1974–1975 |
|-------------------------------------------------------|------|------|------|------|-----------|
| Bluegill ( <i>Lepomis macrochirus</i> )               | 3.26 | 0.52 | 1.07 | 3.68 | 0.63      |
| White crappie ( <i>Pomoxis annularis</i> )            | 1.74 | 0.52 | 1.17 | 1.36 | 0.37      |
| Channel catfish ( <i>Ictalurus punctatus</i> )        | 0.04 | 0.22 | 0.10 |      | 0.26      |
| White bass ( <i>Morone chrysops</i> )                 | 0.15 | 0.03 | 0.12 |      | 0.10      |
| Spotted bass ( <i>Micropterus punctulatus</i> )       | 0.44 | 0.01 | 0.06 | 0.04 | 0.05      |
| Spotted sucker ( <i>Minytrema melanops</i> )          |      | 0.07 | 0.07 | 0.07 | 0.04      |
| Freshwater drum ( <i>Aplodinotus grunniens</i> )      |      |      |      |      | 0.04      |
| Longear sunfish ( <i>Lepomis megalotis</i> )          | 0.11 |      | 0.01 | 0.54 | 0.03      |
| Gizzard shad ( <i>Dorosoma cepedianum</i> )           | 0.11 | 0.01 | 0.02 | 0.04 | 0.03      |
| Carp ( <i>Cyprinus carpio</i> )                       | 0.11 | 0.12 |      | 0.07 | 0.02      |
| Flathead catfish ( <i>Pylodictis olivaris</i> )       | 0.11 | 0.04 | 0.04 | 0.25 | 0.02      |
| River carpsucker ( <i>Carpic des carpio</i> )         |      | 0.04 | 0.10 | 0.11 | 0.02      |
| Warmouth ( <i>Lepomis gulosus</i> )                   |      | 0.04 | 0.01 | 0.04 | 0.02      |
| Black crappie ( <i>Pomoxis nigromaculatus</i> )       | 0.44 |      | 0.01 | 0.04 | 0.02      |
| Yellow bullhead ( <i>Ictalurus natalis</i> )          |      | 0.01 |      |      | 0.02      |
| C. white sucker ( <i>Catostomus commersoni</i> )      |      |      |      | 0.04 | 0.02      |
| Longnose gar ( <i>Lepisosteus osseus</i> )            | 0.11 | 0.06 | 0.02 |      | 0.01      |
| Golden redbreast ( <i>Moxostoma erythrum</i> )        |      | 0.03 | 0.06 |      | 0.01      |
| Quillback carpsucker ( <i>Carpionodes cyprinus</i> )  |      | 0.03 |      |      | 0.01      |
| Orangespotted sunfish ( <i>Lepomis humilis</i> )      |      |      |      | 0.04 | 0.01      |
| Rockbass ( <i>Ambloplites rupestris</i> )             |      | 0.01 |      |      | 0.01      |
| Smallmouth bass ( <i>Micropterus dolomieu</i> )       |      |      |      | 0.07 | 0.01      |
| Smallmouth buffalo ( <i>Ictiobus niger</i> )          |      |      |      |      | 0.01      |
| Sauger ( <i>Stizostedion canadense</i> )              |      | 0.03 | 0.05 |      |           |
| Silver redbreast ( <i>Moxostoma valenciennianum</i> ) |      |      |      | 0.04 |           |
| Skipjack herring ( <i>Alosa chrysochloris</i> )       | 0.04 |      |      |      |           |
| Shortnose gar ( <i>Lepisosteus platostomus</i> )      |      | 0.01 |      |      |           |
| Average number/net set                                | 6.15 | 1.85 | 2.93 | 7.68 | 1.77      |
| Total number of nets set                              | 27   | 67   | 82   | 28   | 90        |
| Total number species                                  | 12   | 18   | 15   | 14   | 21        |

winter, some channel catfish do appear to overwinter in the canal. In the past, anglers have taken channel catfish throughout the winter in the canal (Hatch and Gammon, 1973).

### Seasonal Changes in Species Associations, 1974 to 1975

Seasonal and spatial differences in relative abundance for species collected by electrofishing were particularly striking (Fig. 6). As mentioned previously, summer catches in the effluent canal were near zero but were dominated by longnose gar and gizzard shad in fall and by river carpsucker and white bass in winter. Gizzard shad dominated catches in the remaining zones in all seasons with few exceptions, but the rank order of species thereafter varied considerably. In the thermal river zones, river carpsucker and longnose gar dominated summer catches, white bass and skipjack herring dominated during fall, and freshwater drum, channel catfish, and sauger were most common in winter. The ambient river zones were dominated by smallmouth buffalo and carp in summer, by white bass and skipjack herring in fall, but with catches decreasing to near zero in winter. Catches in the backwater zone through fall most

commonly included river carpsucker and spotted sucker and, in the fall, skipjack herring. However, white crappie and carp dominated winter collections in the backwater zone.

D-net catches were generally dominated by bluegill and white crappie in all zones and seasons except that channel catfish dominated summer and fall catches in the effluent canal, as shown previously (Fig. 7).

### Importance of Temperature, Season, and Habitat

Much emphasis has been placed on temperature and thermal addition in lakes and rivers as determining fish distribution to the near-exclusion of all other factors (Brett, 1956, 1960; Ferguson, 1958; Hela and Laevastu, 1962; Proffitt, 1969; Proffitt and Benda, 1971; Gammon, 1971, 1973). Other investigators cite habitat diversity, turbidity, and agroindustrial pollution as major factors (Mills, Starret, and Bellrose, 1966; Whittaker and Wallace, 1972), but it is often difficult to distinguish distinct abiotic influences and evaluate their individual influences (Larimore and Smith, 1963). Temperature is the most important abiotic factor (Hynes, 1970), but it does not always work alone (Cairns,

TABLE 1 Average Numbers of Each Species per Kilometer Caught by Electrofishing, Summer–Fall 1970 to 1974

| Species                                                 | 1970  | 1971  | 1972  | 1973  | 1974  |
|---------------------------------------------------------|-------|-------|-------|-------|-------|
| Gizzard shad ( <i>Dorosoma cepedianum</i> )             | 6.98  | 1.19  | 4.77  | 1.48  | 7.39  |
| White bass ( <i>Morone chrysops</i> )                   | 0.13  | 0.29  | 0.09  | 0.14  | 0.30  |
| River carpsucker ( <i>Carpionodes carpio</i> )          | 0.10  | 0.19  | 0.20  | 0.15  | 0.18  |
| Longnose gar ( <i>Lepisosteus osseus</i> )              |       | 0.26  | 0.09  | 0.12  | 0.18  |
| Skipjack herring ( <i>Alosa chrysochloris</i> )         | 0.60  | 0.02  | 0.04  | 0.02  | 0.10  |
| Spotted sucker ( <i>Minytrema melanops</i> )            | 0.02  | 0.03  | 0.03  | 0.02  | 0.10  |
| Carp ( <i>Cyprinus carpio</i> )                         | 0.08  | 0.09  | 0.16  | 0.06  | 0.08  |
| Smallmouth buffalo ( <i>Ictiobus bubalus</i> )          | 0.02  |       | 0.09  | 0.13  | 0.07  |
| Flathead catfish ( <i>Pylodictis olivaris</i> )         | 0.05  |       | 0.01  | 0.01  | 0.05  |
| Bluegill ( <i>Lepomis macrochirus</i> )                 | 0.10  | 0.12  | 0.08  | 0.14  | 0.04  |
| Quillback carpsucker ( <i>Carpionodes cyprinus</i> )    | 0.03  | 0.12  | 0.07  | 0.04  | 0.03  |
| White crappie ( <i>Pomoxis annularis</i> )              | 0.01  | 0.06  | 0.01  | 0.03  | 0.03  |
| Longear sunfish ( <i>Lepomis megalotis</i> )            |       | 0.01  | 0.03  | 0.05  | 0.03  |
| Sauger ( <i>Stizostedion canadense</i> )                | 0.58  | 0.73  | 0.14  | 0.04  | 0.02  |
| Spotted bass ( <i>Micropterus punctulatus</i> )         | 0.10  | 0.03  | 0.09  | 0.01  | 0.02  |
| Common white sucker ( <i>Castostomus commersoni</i> )   |       | 0.01  | 0.01  |       | 0.02  |
| Freshwater drum ( <i>Aplodinotus grunniens</i> )        | 0.07  | 0.03  | 0.05  | 0.06  | 0.01  |
| Channel catfish ( <i>Ictalurus punctatus</i> )          | 0.15  | 0.19  | 0.12  |       | 0.01  |
| Largemouth bass ( <i>Micropterus salmoides</i> )        | 0.06  | 0.21  | 0.02  | 0.09  | 0.01  |
| Smallmouth bass ( <i>Micropterus dolomieu</i> )         | 0.02  |       | 0.06  | 0.12  | 0.01  |
| Striped bass ( <i>Morone saxatilis</i> )                |       |       |       |       | 0.01  |
| Black crappie ( <i>Pomoxis nigromaculatus</i> )         |       |       |       |       | 0.01  |
| Golden redbreast ( <i>Moxostoma erythrurum</i> )        | 0.16  | 0.18  | 0.08  | 0.03  | 0.01  |
| Warmouth ( <i>Lepomis gulosus</i> )                     | 0.01  | 0.01  |       | 0.01  | 0.01  |
| Goldeye ( <i>Hiodon alosoides</i> )                     | 0.02  |       |       |       | 0.01  |
| Mooneye ( <i>Hiodon tergisus</i> )                      |       |       |       |       | 0.01  |
| Muskellunge ( <i>Esox masquinongy</i> )                 |       |       |       |       | 0.01  |
| Highfin carpsucker ( <i>Carpionodes velifer</i> )       | 0.03  | 0.21  | 0.01  |       |       |
| Logperch darter ( <i>Percina caprodes</i> )             | 0.01  | 0.03  |       |       |       |
| Shortnose gar ( <i>Lepisosteus platostomus</i> )        |       | 0.01  | 0.01  | 0.01  |       |
| Rockbass ( <i>Ambloplites rupestris</i> )               |       |       | 0.01  |       |       |
| Shorthead redbreast ( <i>Moxostoma macrolepidotum</i> ) |       | 0.01  | 0.01  |       |       |
| Bigmouth buffalo ( <i>Ictiobus cyprinellus</i> )        |       | 0.01  |       |       |       |
| Goldfish ( <i>Carrasius auratus</i> )                   |       | 0.01  |       |       |       |
| River redbreast ( <i>Moxostoma carinatum</i> )          | 0.01  |       |       |       |       |
| Total number collected/km                               | 9.33  | 4.03  | 6.25  | 2.76  | 8.75  |
| Total distance fished, km                               | 27.52 | 54.16 | 34.64 | 33.96 | 47.22 |
| Total number species                                    | 23    | 25    | 25    | 21    | 31    |

the thermal river zones in February. Both density and biomass approached zero in the ambient river zones during December through February.

In general, the four community parameters derived from D-net catches exhibited the same basic pattern from June through February (Fig. 5). The most striking aspect was the large D-net catch in the backwater zone during December and January, a time when almost no fish were caught elsewhere. In summer, D-net catches in the backwater zone were also somewhat greater than in other zones. November appeared to be a transitional period where catches in all zones except the effluent canal were approximately equal.

In summer the effluent canal contained some heat-tolerant species susceptible to capture by D nets, such as channel catfish, flathead catfish, longnose gar, and bluegill.

The catches increased still further in September and October when the temperature decreased below 32°C, with channel catfish predominating and white crappie appearing also. Contrasting with the electrofishing catch, the D nets caught very few fish in the effluent canal during winter, possibly (1) because the aforementioned species vacated the canal during the winter months whereas other species susceptible to capture by electrofishing entered the canal or (2) because these species were actually present in the canal but were inactive and did not enter the D nets. The bluegill and white crappie apparently moved out of the effluent canal during winter, as evidenced by the fact that none were caught in the canal although large catches were made in the backwater zone in December and January. Judging from a few individuals taken by electrofishing and D nets in

creased, especially during high-water periods, the plume was forced against the north shore. Temperatures ranged seasonally from 12 to 41°C in the effluent canal (zone 1), 7 to 32°C in the thermal river zones (zones 3 and 4), 4 to 30°C in the ambient river zones (zones 2 and 6), and 5 to 27°C in the backwater zone (zone 5) during sampling periods from 1970 to 1975 (Fig. 3). The effluent canal was the warmest area in all seasons and was followed by the thermal river zones. The ambient river zones were cooler than the effluent canal and thermal river zones in all seasons, but warmer in summer and cooler in winter than the backwater zone. Vertical stratification was apparent only in summer and early fall, and the heated water was restricted to the upper 2 m.

### Changes in Fish Catches from 1970 to 1974

Table 1 summarizes the electrofishing catch during summer and fall only. Throughout the study period the sampling effort was roughly comparable. Most species either appeared sporadically in the catch or fluctuated irregularly through the study period. A few, however, exhibited trends. Longnose gar (*Lepisosteus osseus*) were absent from the 1970 catch, but they have been collected each year since 1971. White bass (*Morone chrysops*) were particularly abundant in 1971 and 1974. Catches of sauger (*Stizostedion canadense*) and golden redbreast (*Moxostoma erythrum*) decreased steadily and dramatically. However, young sauger were collected during the winter of 1974–1975.

The D-net catches were dominated by bluegill (*Lepomis macrochirus*) and white crappie (*Pomoxis annularis*), which

fluctuated greatly annually in relative abundance (Table 2). Spotted bass (*Micropterus punctulatus*) were taken in abundance only in 1970.

### Seasonal Changes in Community Structure, 1974 to 1975

An unusually complete series of collections obtained during 1974 and early 1975 permitted seasonal comparisons. Shannon–Weiner diversity indices by number and weight were comparable for all seasons (Fig. 4). Diversity through September was comparable in all zones except the effluent canal (zone 1), whereas in October diversity was similar in all zones including the effluent canal. Diversity in the effluent canal remained high through winter but declined in the remaining zones. The thermal river zones (zones 3 and 4), however, which were 3 to 5°C warmer than the ambient river zones (zones 2 and 6), had higher diversity indices.

Density and biomass units resulted in similar patterns of relative abundance in summer and fall but differed in winter. Biomass steadily increased from June to February, whereas density steadily increased to a maximum in November and then sharply declined in winter. Diversity and biomass were comparatively low in all zones through August. Catches were highest in the backwater zone (zone 5) in September, highest in both ambient and thermal river zones in October, and highest in the ambient river zones in November. Density steadily increased in the effluent canal from September through February. Biomass was highest in the effluent canal in winter months and in

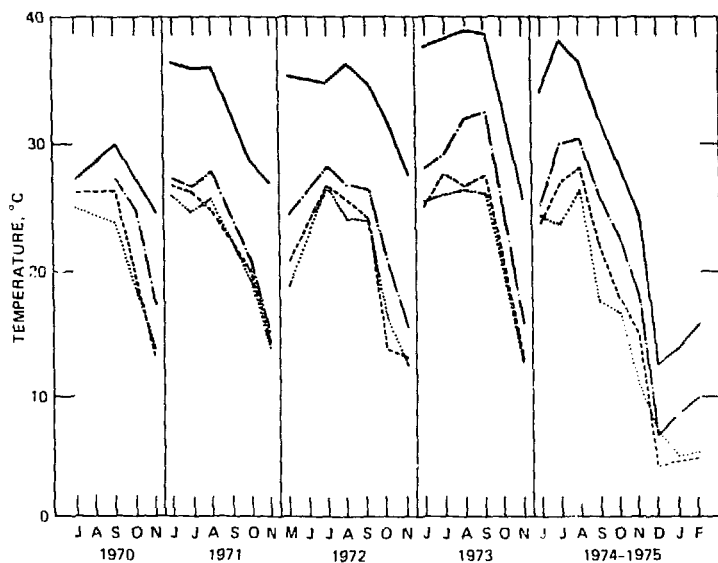


FIG. 3 Seasonal temperature variations in zone 1 (—), zones 3 and 4 (---), zones 2 and 6 (----), and zone 5 (.....), 1970 to 1975.

night during the test runs in 1974. Each zone was fished at monthly intervals by moving the boat in a slow and steady course downstream over the littoral slope and near snags and stumps whenever possible. This method proved to be selective for surface dwelling and littoral species. Relative abundance was based on numbers per kilometer and kilograms per kilometer.

D nets constructed of metal frame and covered with 1.9-cm wire mesh were set monthly at eight locations on the bottom in zones 1 to 5 (Fig. 1) and lifted at 24-hr intervals. Relative abundance was based on number of fish per each net-day and kilograms of fish per each net-day.

Upon capture, fish were placed in a holding tank filled with water from the zone sampled. Total lengths were measured to the nearest millimeter, and weights were taken to the nearest gram or ounce; fish were then returned unharmed to the river.

Temperatures were recorded with thermistors at net depth where nets were lifted and at 1.5 m at the beginning and end of electrofishing runs. Additional temperature surveys for plume determinations were conducted in June, July, and October 1974 and January 1975. Plant intake and discharge temperatures have been monitored hourly for each unit since plant operation began.

An echo sounder with a bow-mounted transducer was used for depth determinations in all areas of the study site.

Raw data were compiled in sequence and placed on magnetic tape. Programs written in Basic-plus were used to compute relative-abundance and weight indices, Shannon-Weiner diversity indices, and mean monthly temperatures.

## RESULTS AND DISCUSSION

### Thermal Characteristics

Condenser cooling water was taken in by the plant from the north shore of the river 1.0 km upriver from the weir, heated 8 to 15°C, and passed into the effluent canal 1.6 km above the weir. Temperatures taken during electrofishing runs in 1974 to 1975 showed a 2°C change from the discharge area to just behind the weir; however, differences as high as 11°C have been recorded (Hatch and Gammon, 1973). Heated water was then jetted out into the river through the weir, where it mixed rapidly near the surface. The mixing zone extended a little more than 2 km downriver from the weir in all seasons (Fig. 2). During our observations the plume stayed close to shore and within the north half of the river. During previous years (1970 to 1973) the plume did not come back into the north shore for at least 1.0 km downriver of the weir (Hatch and Gammon, 1973). These differences in plume structure were attributed to the variation in the width of the weir from 1970 to 1973 and 1974 to 1975. River discharge also influenced plume structure. As the volume of flow in-

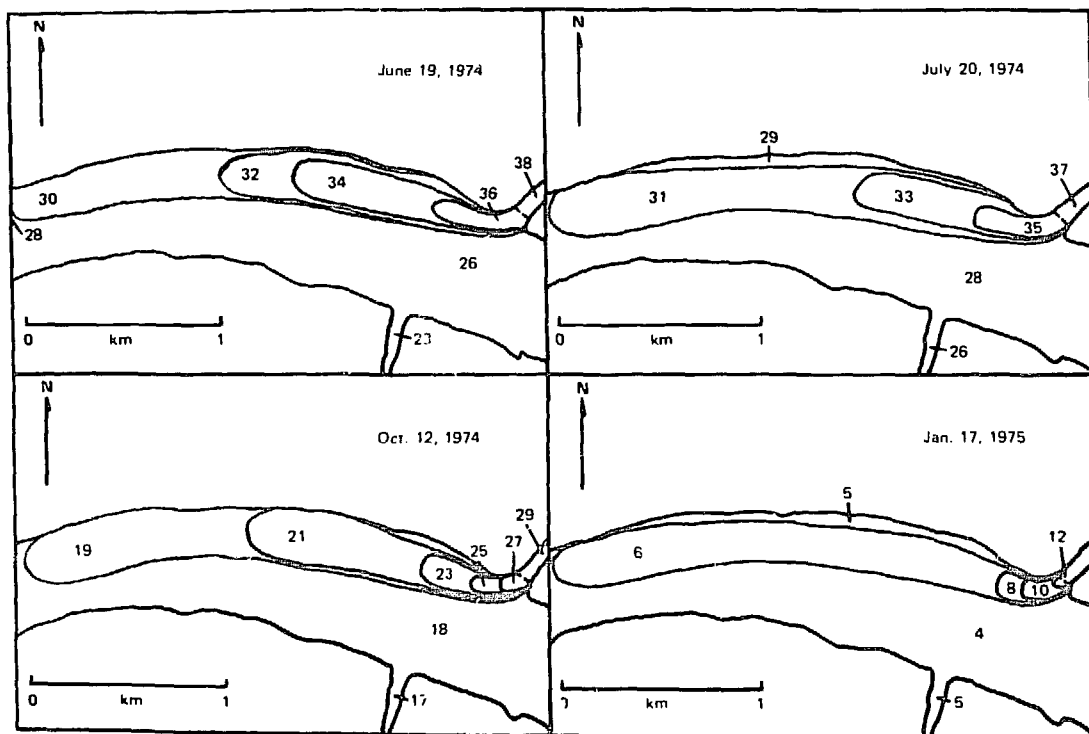


FIG. 2 Plume structure, in °C at 1.5-m depth, during four sampling dates, 1974 to 1975.

The J. M. Stuart plant is a four-unit complex. Three 610-Mw units using once-through cooling began operating in September 1970, May 1971, and March 1972 and one 610-Mw unit using a natural-draft cooling tower began operation in February 1974. The three older units each draw water with dual pumps rated at  $13.5 \text{ m}^3/\text{sec}$  (307 mgd), whereas the newest unit requires  $0.8 \text{ m}^3/\text{sec}$  (19 mgd).

The study site encompasses 3.36 km of the Ohio River in Brown and Adams counties, Ohio. 1.57 km of Little Three Mile Creek in Adams County, Ohio, and 0.61 km of Kennedy Creek in Mason County, Kentucky, near Maysville, Ky., and Aberdeen, Ohio. Each type of habitat characteristic of the Ohio River is present, and six zones ranging from 0.40 km to 1.57 km in length were selected on the basis of their proximity to the thermal discharge and habitat type.

Zone 1 (1.57 km) is the dredged-out portion of Little Three Mile Creek that serves as the discharge canal (Fig. 1). Heated water is pumped continuously through two submerged pipes at the head of the canal and is jetted out into the Ohio River through a concrete and steel weir at the canal mouth. Width of the weir opening has varied from 6.3 m (1970 to 1974) to the present 18.9 m. Width of the canal varies from 30 m at the discharge to 75 m behind the weir with a uniform depth of 4 to 5 m. Zones 2 (0.40 km) and 6 (1.10 km) are located immediately and 2.2 km upriver from the weir. Both are characterized by the typical steep-sloped littoral shelf of the Ohio River, but zone 6 affords more variation of habitat in the form of overhanging trees and submerged stumps. Zone 3 (0.80 km)

begins in a deep eddy (7 to 9 m) created by the weir and gradually becomes shallower, ending over a shallow (3 m) sand and mud flat. Zone 4 (1.06 km) begins over this same sand and mud flat and gradually becomes deeper, ending in a rocky steep-sloped littoral shelf. Both zones afford varied habitat ranging from open shoreline to overhanging trees and submerged stumps. Zone 5 (0.61 km) is located in Kennedy Creek, which typifies the sluggish backwater areas common to the mouths of Ohio River tributaries. Width of this zone ranges from 30 to 40 m, and the depth is a uniform 3 to 4 m. There is an abundance of woody submergent cover and overhanging trees.

Water depth is largely controlled by the Meldahl (48 km downriver) and Greenup (97 km upriver) navigational dams. Normal pool level is 10.1 m, and seasonal variations range from 9.5 to 15.0 m.

## METHODS AND MATERIALS

Relative-abundance indices and determinations of seasonal distributions for the different species of fish were obtained through use of a boat-mounted electrofishing device and stationary D nets.

Electrofishing was conducted by passing a direct current through an electrode system, similar to that described by Frankenberger (1960), mounted on a 5-m fiber glass skiff. Fishing was conducted at night from 1970 to 1973 because the river was low and relatively transparent when the study began and daytime electrofishing was not as effective. However, the increased turbidity of the river during the past 2 or 3 years has made it unnecessary to electrofish at night; comparable catches were made day and

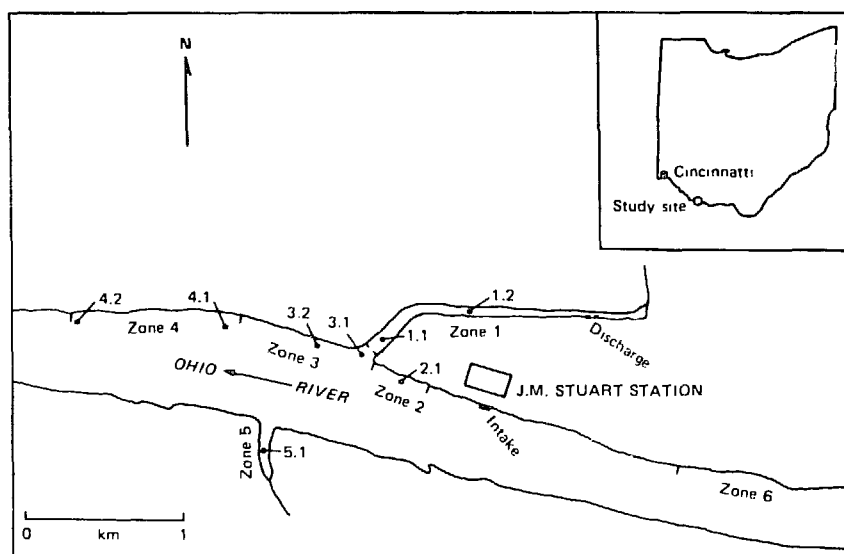


FIG. 1 Location of study site showing electrofishing zones and D net locations. Arrow indicates direction of river flow.

## METHODS

### Hydrographic Surveys

Physical and chemical parameters were measured at 21 stations (Fig. 2) at approximately monthly intervals from December 1972 through October 1973. Additional samples were taken at least three times during a tidal cycle for analysis of dissolved oxygen by the Winkler technique (American Public Health Association, 1971). Surface and bottom water temperatures and salinities were determined at each station with a salinometer. Turbidity was measured at the stations during three of the surveys with a turbidimeter, and the organic and inorganic content of the total suspended load was measured on six surveys by filtering in the field 150 to 200 ml of water through preweighed 0.45- $\mu$ m filters. The filters were oven-dried at 60°C, reweighed, ashed at 550°C, and weighed again.

Recording instruments were used at selected stations during each survey for measurements of current velocity, surface temperature, and relative tidal heights. Since the area immediately adjacent to the plant is ~0.5 m deep (mean low water), current measurements were of limited value and served primarily to delineate general circulation patterns.

### SEDIMENTS

Samples for sediment analysis were collected at the stations shown in Fig. 3 with a benthic sampler (Baird et al., 1972), which removes intact sediment plugs to a depth of 20 cm. Sediments were processed and analyzed for grain

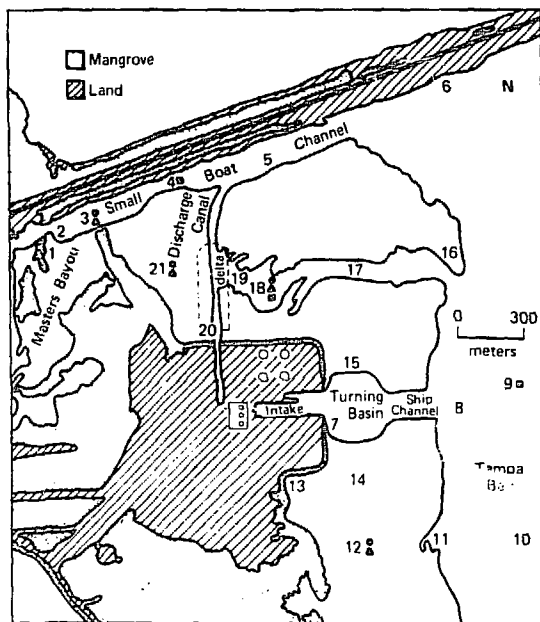


FIG. 2 Locations of hydrographic stations.  $\Delta$ , temperature.  $\circ$ , current.  $\blacksquare$ , tide. Channel lines indicate the 2-m contour.

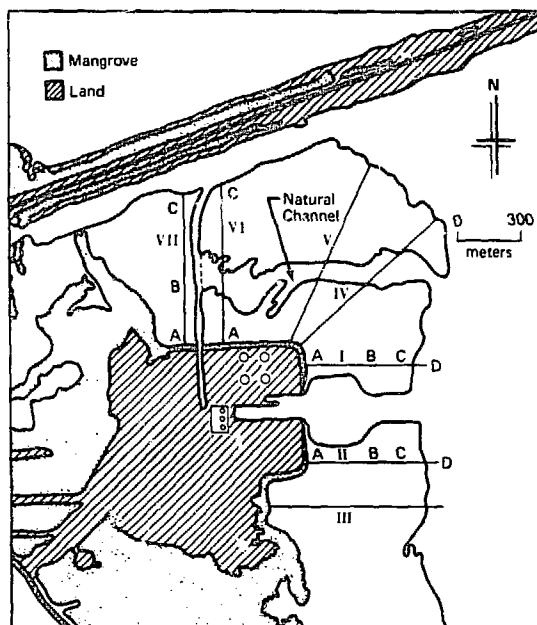


FIG. 3 Locations of sediment and invertebrate sampling stations (letters) in relation to grass transects (Roman numerals). Channel lines indicate the 2-m contour.

size (Folk, 1965) and classified by weight percent of granule as large sediment ( $>2.0$  mm), sand ( $62.5$   $\mu$ m to  $2.0$  mm) and silt-clay ( $<62.5$   $\mu$ m). Organic content was determined by standard combustion methods, and carbonate was determined by acid digestion. Sediment temperatures at depths of 2 and 10 cm were taken at selected stations with thermistors.

### Benthic Vegetation

Seven transects (Fig. 3) were visually mapped by divers during the winter of 1972 and the summer of 1973 to determine the relative degree of bare sediment, seagrass, and algae. In addition, since benthic macrophytes may show pronounced seasonal variation, periodic overflights using methods of Feigl, Pyle, and Clingan (1972) were coupled with spot checks by divers to determine the extent of variability over areas not examined in the transects.

In June 1973, an attempt was made to transplant *Thalassia testudinum* to the discharge side of the power plant. The seagrass was removed from the bottom in 15- by 15- by 20-cm plugs and placed in 1-m-square wooden boxes. Three boxes were placed near each of the three invertebrate sampling stations of transect VII (Fig. 3). Three other boxes were placed in the area from which the seagrass was removed near the outer edge of transect IV (Fig. 3).



## Benthic Macroinvertebrates

Bottom samples to be used for determination of species composition and abundance were taken in the summers of 1972 and 1973 with the benthic sampler. The sampler used for collection removed an 81-cm<sup>2</sup> sediment plug to a depth of 20 cm. Samples were sieved in the field through 500- $\mu$ m-mesh screen and were fixed in 10%-neutral buffered formalin. The macroinvertebrates were stained with 0.15% rose bengal (Mason and Yevich, 1967), sorted from the nonliving and plant material, and identified to the species level wherever possible. Invertebrate counts were compared using the t-test for paired comparisons (Steele and Torrie, 1960). Significant differences were accepted at the 0.05 probability level.

During the summer of 1972, 14 samples were taken from each of 12 stations on the intake side of the plant to determine the degree of variability in the benthic faunal assemblages. Based on the analysis of these samples and an analysis of 14 samples taken from the discharge area, we established a macroinvertebrate sampling program. During the summer of 1973, 9 samples were taken from each of 14 stations along the four transects (I, II, VI, and VII) shown in Fig. 3. Eight of these samples were used for faunal analysis, and one was used for sediment analysis.

## Salinity and Temperature

Salinities remained relatively constant from station to station (<2‰ range). Salinities did change between surveys, however. The highest mean salinity (31.8‰) occurred during the January 1973 survey, and the lowest mean (25.1‰) occurred during the April 1973 survey. Salinities in the shallow parts of the bay may decrease rapidly after periods of heavy rainfall, and it is likely that this area is frequently exposed to salinities less than those recorded in the study.

Stations 11 and 12 (Fig. 2), south of the plant, were chosen as representative of ambient thermal conditions in the shallow waters adjacent to the plant. The temperature values observed at these two stations were averaged for each survey, and this average is called ambient. Ambient temperature values ranged from 15.3°C in February 1973 to 29.9°C in June 1973.

Figure 4 shows the mean temperature-increase contours for the surface waters in the vicinity of the discharge. During the course of the study, ~15 ha were subjected to a mean temperature rise >4°C, 69 ha to a mean rise >3°C, 95 ha to a mean rise >2°C, and about 174 ha to a mean rise >1°C.

## Suspended Particulate Matter and Turbidity

Consistent differences in the total suspended loads of the intake and discharge stations were not observed. The mean values for the 21 stations ranged from 5.50 mg/liter (SD =  $\pm 1.35$ ) in November 1972 to 17.34 mg/liter (SD =  $\pm 10.31$ ) in April 1973. The variation observed during the April survey was a result of varying wind conditions. The mean percent organic content of the total suspended load

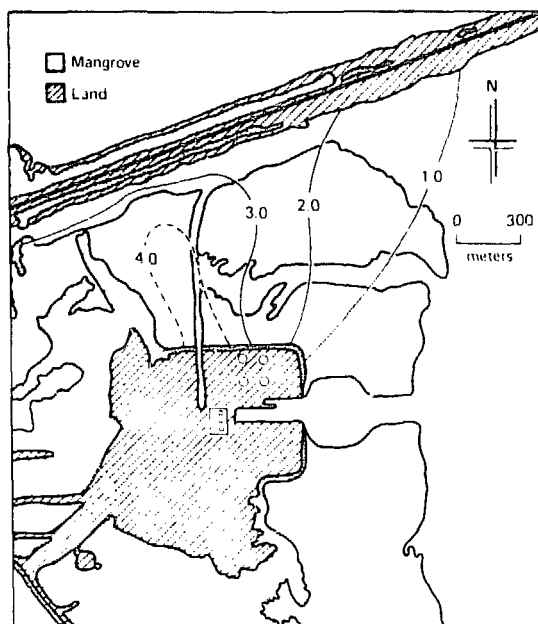


FIG. 4 Mean temperature increase over ambient (contour intervals are at 1°C). Channel lines indicate the 2-m contour.

ranged from 25.1% in October 1973 to 50.0% in April 1973. Formazin turbidity units (FTU) ranged from 3.5 in January 1973 to 9.5 in April 1973.

## Dissolved Oxygen

Values of dissolved oxygen ranged from 2.6 to 13.0 ppm. Predictably, the higher values occurred during the winter months. The lowest values occurred in the bayou west of the plant during the morning hours of June 27, 1973, on a flood tide. The water over the seagrasses often became supersaturated with oxygen during the daylight hours, and, in the immediate vicinity of the discharge channel, saturation values at times exceeded 200%.

## Sediments

The classical sediment characteristics are relatively uniform over the study area. Sediments average 97% sand and 2.6% silt-clay particles by weight; grains larger than sand comprise only 0.4%. The sand-sized and larger particles are predominantly quartz, with only 3.2% by weight as carbonate. Although the total organic content of the sediment was only 1.6% by weight, the sampler used was not adequate to show possible differences in the fine particulate detritus which might occur in the near surface between sand and seagrass areas.

Porous, permeable, sandy deposits that surround the power plant allow rapid equilibration of temperatures between sediments and the water column. Insertion of the thermistors may have further facilitated equilibration. Thus

sediment temperatures at depths of 2 and 10 cm were usually within 0.5 to 1.0°C of those of the overlying water (Fig. 5).

### Benthic Vegetation

The results of the visual observations of divers and of aerial overflights were used to estimate the distribution of the dominant seagrasses in different areas adjacent to the plant (Fig. 6). Three seagrasses, *Thalassia testudinum*, *Syringodium filiforme*, and *Halodule wrightii*, were observed on the eastern (intake) side of the plant; *Halodule* and *Syringodium* were the dominant species. To the northeast of the plant, where the mean temperature rise was 2°C or less, all three seagrasses were present, but *Thalassia* and *Halodule* dominated. During May and June of 1972 and 1973, the *Thalassia* plants to the northeast bore fruit, an unusual occurrence for Tampa Bay (Humm, 1975).

*Halodule wrightii* was the only seagrass found on the northern side of the plant. It occurred only near the outer edge of the plume (where the mean temperature rise is 3°C or less). An area of ~81 ha<sup>2</sup> on the northern side of the plant was bare sand during most of the year.

Figure 7 shows the biomass of *Halodule wrightii* near station B of transect II and station C of transect VI for July and August 1973. The biomass near Station B(II) was 5 to 10 times as great as that at station C(VI). In addition, the

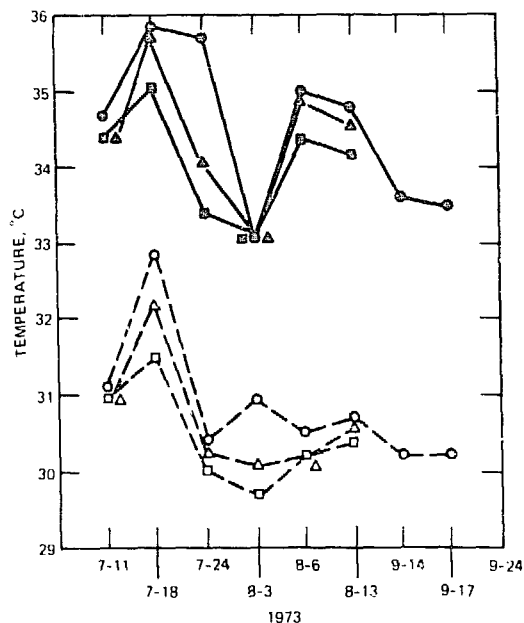


FIG. 5 Mean water and sediment temperatures near two stations in the intake and discharge areas. Temperatures were measured every 20 min for 6 hr. Near station B(II): —○—, water; —△—, sediment at 2 cm; —■—, sediment at 10 cm. Near station C(VI): ---○---, water; ---△---, sediment at 2 cm; ---■---, sediment at 10 cm.

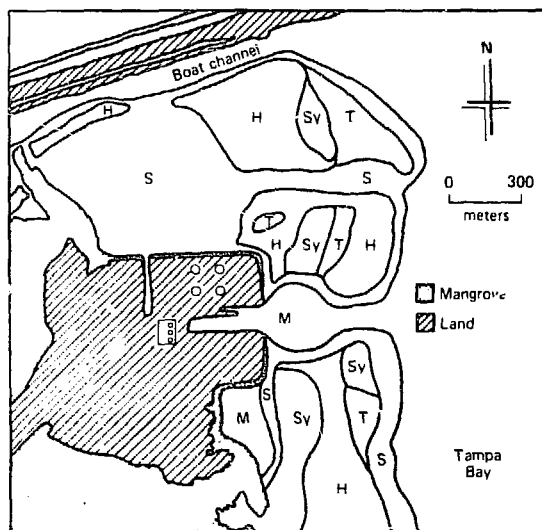


FIG. 6 Distribution of dominant seagrasses in the area adjacent to the P. L. Barrow power plant. Channel lines indicate the 2-m contour.

S = sand  
M = mud  
T = *Thalassia testudinum*  
Sy = *Syringodium filiforme*  
H = *Halodule wrightii*

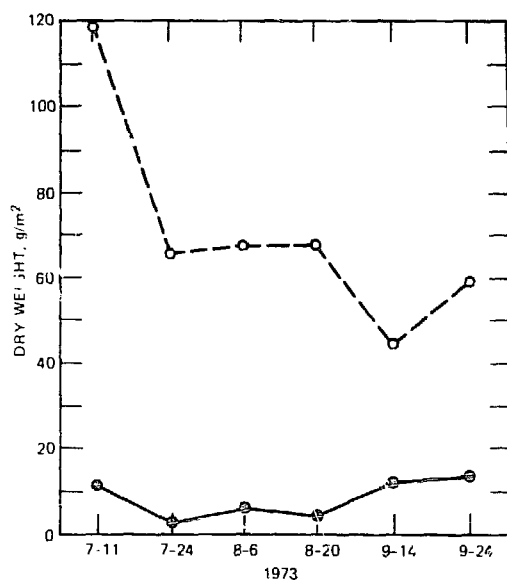


FIG. 7 Biomass of *Halodule wrightii* at two stations during the summer months. ---○---, station B(II), intake; —○—, station C(VI), effluent.

average length of emergent stems over the sampling period was 20.2 cm at B(II), compared with 10.0 cm at C(VI) (Holsing, 1974).

Two species of algae, *Gracilaria foliifera* and *Ulva lactuca*, were abundant on the intake side during the winter and spring of 1972–1973. Within the small embayment south of the plant, *Gracilaria foliifera* was the macrophyte observed in quantity. On the effluent side of the plant during this same period, *Gracilaria foliifera* was observed near the outer reaches of the plume in the *Halodule* beds. During the summer of 1973, blue-green algae formed mats 2.5 mm thick over much of the area covered by a mean temperature rise of 3.0°C or greater.

When the plugs of *Thalassia* were transplanted, the water temperature over the *Thalassia* beds at the end of transect IV was approximately 29°C. At stations A(VII), B(VII), and C(VII), the water temperatures were 35, 33, and 31°C, respectively. The seagrass transplanted to the stations along transect VII began to die within 1 week. By the end of 30 days, the *Thalassia* blades were mostly disintegrated into a detrital mat on the surface of the sediment. Although the roots and rhizomes were still present in the sediment, new growth did not occur in these boxes. The seagrass that was removed from the beds at the end of transect IV and placed in the same area survived for 6 months until a winter storm eroded the substrate from the boxes.

### Benthic Macroinvertebrates

Figure 8 shows the cumulative number of bivalves and polychaetes at six intake stations and one effluent station. Three curves are evident: one for the intake stations containing seagrasses, one for the intake stations where only sand was observed, and one for the sandy discharge

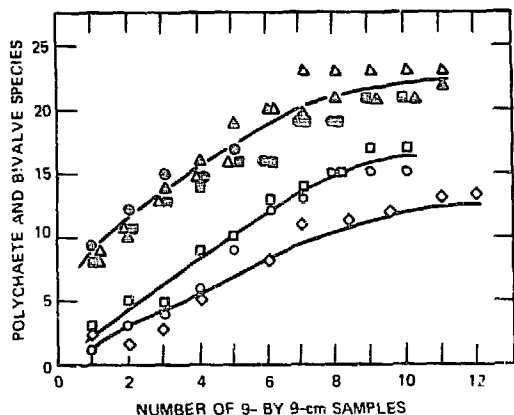


FIG. 8 Cumulative number of bivalve and polychaete species taken in successive samples from stations in the intake and discharge areas. ○, station C(I), □, station B(II), ▲, station A(II), △, station C(II), □, station D(II), ○, station D(I), ◇, station A(VI).

area (Table 1). Seven of the 9- by 9-cm samples yielded 82.3 to 100% of the total number of bivalve and polychaete species collected in the 14 samples.

A total of 104 species of polychaetes, molluscs, crustaceans, and echinoderms were identified from the intake transects, and 60 species were identified from the discharge transects. Assuming that the intake and discharge habitats were at one time very similar, 35 species belonging to these groups have been excluded from the immediate vicinity of the discharge, and 9 species have been added.

The polychaete, *Prionospio heterobranchiata texana*, and the bivalve, *Mysella planulata*, were the most abundant species to appear consistently in samples taken from either the intake or discharge transects, either in a seagrass habitat or a sand habitat. In addition, these two species occur in significantly greater numbers ( $P < 0.05$ ) along the discharge transects than along the intake transects.

The numbers of individuals per square meter which belong to the three major taxonomic groups collected from the intake and discharge transects are shown in Table 2. The mean numbers of individual polychaetes, molluscs, and crustaceans along transect I were not significantly different ( $P > 0.05$ ) from those along transect II. Also, mean numbers along transect VI did not differ significantly from

TABLE 1 Description of Infaunal and Sediment Sampling Stations

| Transect and station | Distance from shore, m | Depth, cm (mean high water) | Bottom description                                        |
|----------------------|------------------------|-----------------------------|-----------------------------------------------------------|
| <b>I</b>             |                        |                             |                                                           |
| A                    | 15                     | 63                          | 100% sand                                                 |
| B                    | 330                    | 125                         | 25% sand<br>5% <i>Halodule</i><br>70% <i>Syringodium</i>  |
| C                    | 480                    | 130                         | 5% sand<br>90% <i>Halodule</i><br>5% <i>Syringodium</i>   |
| D                    | 665                    | 150                         | 100% sand                                                 |
| <b>II</b>            |                        |                             |                                                           |
| A                    | 15                     | 65                          | 100% sand                                                 |
| B                    | 330                    | 105                         | 0–20% sand<br>0–20% <i>Ulva</i><br>80% <i>Halodule</i>    |
| C                    | 480                    | 120                         | 80% sand<br>10% <i>Halodule</i><br>10% <i>Syringodium</i> |
| D                    | 665                    | 155                         | 100% sand                                                 |
| <b>VI</b>            |                        |                             |                                                           |
| A                    | 15                     | 75                          | 100% sand                                                 |
| B                    | 240                    | 180                         | 100% sand                                                 |
| C                    | 760                    | 140                         | 40% sand<br>60% <i>Halodule</i>                           |
| <b>VII</b>           |                        |                             |                                                           |
| A                    | 15                     | 55                          | 100% sand                                                 |
| B                    | 350                    | 60                          | 80% sand<br>20% <i>Halodule</i>                           |
| C                    | 750                    | 90                          | 40% sand<br>60% <i>Halodule</i>                           |

# Effects of Thermal Constancy and Seasonal Temperature Displacement on Community Structure of Stream Macroinvertebrates

James M. Ward

## ABSTRACT

Diurnal and seasonal thermal constancy, a greatly delayed seasonal temperature maximum, and summer cool and winter warm conditions characterize the stream environment below a deep-release dam in Colorado. Low diversity index and equitability values and changes in macroinvertebrate species composition may result from failure of the temperature regime to provide the thermal stimuli essential for various life-cycle phenomena. It is hypothesized that the following sublethal effects, directly or indirectly resulting from the modified temperature regime, may further alter macroinvertebrate community structure: (1) reduction of niche overlap and a shift toward an equilibrium community as a consequence of reduced environmental fluctuation; (2) more intense competition associated with greater productivity; (3) elimination of major invertebrate predators; and (4) failure of the limited temperature range to provide optimal temperatures for various physiological processes. Effects of the temperature regime on diversity patterns should be considered in dam construction and operation.

Stream macroinvertebrate communities are considerably altered below deep-release dams in comparison with those in unregulated streams of the same region. Effects on abundance vary because of complicating factors (such as flow fluctuations, oxygen conditions, and sediment deposition), but species diversity is invariably reduced (Pearson, Kramer, and Franklin, 1968; Hilsenhoff, 1971; Hoffman and Kilambi, 1971; Isom, 1971; Spence and Hynes, 1971a; Fisher and LaVoy, 1972; Lehmkuhl, 1972; 1974).

Seasonal temperature constancy, e. g., high winter and low summer temperatures, has been considered as directly influencing macroinvertebrate community structure by failing to provide the thermal stimuli essential for various life-cycle phenomena (Coutant, 1968; Pearson, Kramer, and Franklin, 1968; Nebeker, 1971; Lehmkuhl, 1972; 1974). Little attention has been given to effects of the delayed

seasonal maximum or the diurnal temperature constancy in streams below deep-release dams.

A year-round study below a Colorado mountain reservoir revealed a macroinvertebrate community impoverished in species, but rich in density and biomass (Ward, 1974). This stream provided an ideal opportunity to assess the effects of a modified thermal pattern in an otherwise favorable environment. This paper presents some hypothetical relationships based on the influence of the thermal environment on competitive, physiological, and predator-prey interactions to explain the low macroinvertebrate diversity and altered species composition below this and other deep-release dams.

## METHODS AND MATERIALS

The study was conducted on the South Platte River below Cheesman Lake (elevation 2087 m above sea level), a deep-release storage reservoir built in 1905 in the mountains of central Colorado. Sampling sites 1 to 4 were established on rubble riffles 0.25, 2.4, 5, and 8.5 km, respectively, below the dam.

Macroinvertebrates, aufwuchs, and various physicochemical parameters were sampled about every 28 days for a 1-year period.

Macroinvertebrate species-diversity values are based on the Shannon-Weaver index with logarithms to base 2 (bits per individual). Equitability values were calculated from the table of Lloyd and Ghelardi (1964).

A detailed description of the thermal pattern and other biological and physicochemical data has been presented elsewhere (Ward, 1974).

## RESULTS

A generally favorable physicochemical environment characterized the stream in the study section. The rapid

those along transect VII. However, significantly more individual polychaetes ( $P < 0.05$ ) were collected along the effluent transects (VI and VII) than along the intake transects, and significantly fewer crustaceans ( $P < 0.05$ ) were collected along the effluent transects than along the intake transects. No significant difference ( $P > 0.05$ ) was found between the number of individual molluscs along the intake and the effluent transects. The large variation in the number of individual molluscs (Table 2) collected at the different stations in the intake area was caused by the occurrence of the epibenthic gastropods, especially *Bittium varium*, in the seagrass areas.

## ACKNOWLEDGMENTS

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TABLE 2 Number of Individual Polychaetes, Molluscs, and Crustaceans Collected from Stations Along the Intake (I and II) and Effluent (VI and VII) Transects (individuals/m<sup>2</sup>)

| Transect and station | Polychaetes | Molluscs | Crustaceans | Total  |
|----------------------|-------------|----------|-------------|--------|
| I                    |             |          |             |        |
| A                    | 2813        | 160      | 430         | 3,403  |
| B                    | 7410        | 1,999    | 1583        | 10,992 |
| C                    | 2982        | 10,517   | 2475        | 15,974 |
| D                    | 690         | 10,870   | 5781        | 17,341 |
| Mean                 | 3473        | 5887     | 2567        |        |
| II                   |             |          |             |        |
| A                    | 2721        | 4,428    | 1245        | 8,394  |
| B                    | 5073        | 5,043    | 2014        | 12,130 |
| C                    | 4335        | 42,250   | 1276        | 47,861 |
| D                    | 507         | 9,071    | 7303        | 16,881 |
| Mean                 | 3159        | 15,198   | 2959        |        |
| VI                   |             |          |             |        |
| A                    | 6119        | 7964     | 3582        | 17,665 |
| B                    | 6334        | 2860     | 276         | 9,470  |
| C                    | 8809        | 5279     | 615         | 14,703 |
| Mean                 | 7087        | 5367     | 685         |        |
| VII                  |             |          |             |        |
| A                    | 3,782       | 2009     | 276         | 6,067  |
| B                    | 10,024      | 841      | 584         | 11,449 |
| C                    | 10,101      | 3860     | 307         | 14,268 |
| Mean                 | 7969        | 6710     | 389         |        |

flow fluctuations, lowered dissolved oxygen, and siltation reported below dams by other investigators were not encountered here. The favorable conditions were reflected by the presence of an abundant macroinvertebrate fauna at all sites during all seasons.

### Temperature Pattern

A greatly altered temperature pattern was found below the dam. Although the degree of alteration decreased downstream, the lowermost site (8.5 km below the dam) still exhibited an abnormal temperature regime. The mean annual temperature increased downstream (5.8, 7.2, 7.9, 8.8°C at sites 1 to 4, respectively) although in winter the temperature was constant throughout the study section or decreased slightly downstream. The annual temperature range was 3 to 13°C at the upper three sites and 3 to 14°C at site 4. At the upper sites, however, the seasonal maximum (13°C) did not occur until the end of October, at the time of reservoir overturn, whereas site 4 exhibited a maximum (14°C) in early August. There was also a considerable difference between the diurnal constancy immediately below the dam and the relatively great daily fluctuations at the lowest site.

### Macroinvertebrate Community

Figure 1 shows three measures of species diversity as a function of the distance below the dam. The number of species increased progressively downstream concomitant

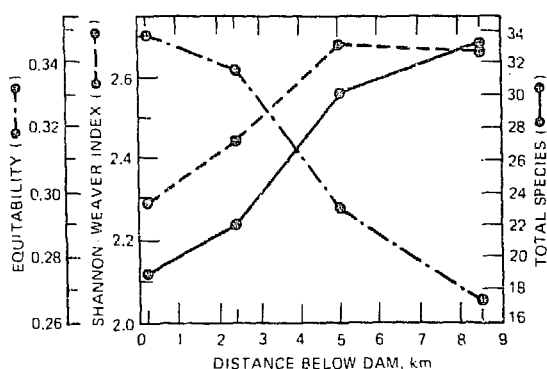


FIG. 1 Three measures of species diversity as a function of the distance below Cheesman Dam, Colorado.

with an approach toward more normal temperature conditions. Even at the lowest site, however, temperature conditions did not return to normal. This was best exemplified by the absence of surface ice at times when the river was completely ice covered only a few kilometers farther downstream. The number of species at site 4 was still low; many more species occur farther downstream (Ward, unpublished).

Additional insight is provided by the diversity values. Unpolluted streams usually have Shannon-Weaver diversity index values ( $\bar{d}$ ) between 3.0 and 4.0 (Wilhm, 1970). The

highest index value obtained for the South Platte River was 2.7, and the progressive increase of  $\bar{d}$  in a downstream direction resembles the recovery section of a polluted stream. Equitability values between 0.6 and 0.8 are thought to characterize unpolluted streams, and even slight organic pollution produces values below 0.5 (Weber, 1973). In the South Platte River equitability ranged from 0.27 to 0.35 and decreased in a downstream direction; this accounts for the leveling of  $\bar{e}$  values from site 3 to site 4 even though the number of species increased.

A few species accounted for a large majority of the standing crop. Four species, two mayflies (*Baetis* sp. and *Ephemerella inermis*) and two dipterans (*Cardiocladius* sp. and *Simulium arcticum*), comprised 87% of the organisms in the study section. *Baetis* sp. and *Cardiocladius* sp. made up 69% of the total.

Table 1 shows the percentage composition and number of species for the major taxa in the study sites. Table 2 indicates possible temperature effects on the diversity of major taxa in other studies below deep-release dams.

In this study Ephemeroptera and Plecoptera exhibited exceptionally low diversity. Trichoptera, Coleoptera, and Diptera were somewhat reduced in diversity in comparison with their diversity in similar unregulated streams. Amphipoda and Gastropoda, not normally present in granitic foothills streams in Colorado, were apparently favored by the effects of the dam. These results agree in general with the findings of other investigators (Table 2).

Ephemeroptera, Plecoptera, Trichoptera, Coleoptera, Gastropoda, Hydracarina, and Oligochaeta exhibited a downstream increase in diversity within the study area. Diptera, Amphipoda, and Platyhelminthes exhibited similar diversity patterns within the length of the study section. Diptera seem best able to withstand conditions below dams and may comprise a major portion of the benthic fauna

TABLE 1 Percentage Composition by Density and Biomass at Site 1 and Number of Species in Major Taxa at Sites 1 and 4, South Platte River, Colorado\*

| Taxon            | % composition |         | Number of species |        |
|------------------|---------------|---------|-------------------|--------|
|                  | Density       | Biomass | Site 1            | Site 4 |
| Diptera          | 50.4 S        | 44.2 S  | 10                | 10     |
| Ephemeroptera    | 43.2 S        | 35.0 S  | 2                 | 6      |
| Trichoptera      | 4.5 D         | 15.3 D  | 3                 | 5      |
| Plecoptera       | 0 I           | 0 I     | 0                 | 1      |
| Coleoptera       | + I           | + S     | 1                 | 2      |
| Amphipoda        | 1.3 D         | 4.1 D   | 1                 | 1      |
| Gastropoda       | 0 I           | 0 I     | 0                 | 1      |
| Oligochaeta      | + I           | 1.3 I   | 1                 | 4      |
| Platyhelminthes  | + S           | + S     | 1                 | 1      |
| Hydracarina      | 0 I           | 0 I     | 0                 | 2      |
| Total insects    | 98.6          | 94.5    | 16                | 24     |
| Total noninsects | 1.4           | 5.5     | 3                 | 9      |

\*Symbols and abbreviations are: +, less than 1.0%; I, increase downstream [at least twice as important (%) at site 4]; D, decrease downstream [at least twice as important (%) at site 1]; and S, similar in importance (%) at sites 1 and 4.

TABLE 2 Temperature-Indicated Influences of Deep-Release Dams on Diversity of Macroinvertebrates in the Receiving Stream

| Taxon                             | Effect* | Temperature influence             |
|-----------------------------------|---------|-----------------------------------|
| Ephemeroptera                     | D       | Low summer temperature†,‡         |
|                                   | D       | Seasonal temperature alteration § |
|                                   | D       | Summer cooling, winter warming ¶  |
| Plecoptera                        | D       | Low summer temperature†           |
|                                   | D       | Summer cooling, winter warming ¶  |
| Trichoptera                       | D       | Low summer temperature†           |
|                                   | D       | Summer cooling, winter warming ¶  |
| Coleoptera                        | D       | Low summer temperature†           |
|                                   | D       | Summer cooling, winter warming ¶  |
| Amphipoda                         | I       | Cool summer temperature ‡         |
|                                   | I       | Summer cooling, winter warming ¶  |
| Total macroinvertebrate diversity | D       | Seasonal temperature alteration § |
|                                   | D       | Summer cooling, winter warming ¶  |
|                                   | D       | Low summer temperature†,‡         |
|                                   | D       | Low constant temperature**        |

\*I, increased diversity; D, decreased diversity.

†Pearson, Kramer, and Franklin, 1968 (*Baetis* sp. increased).

‡Spence and Hynes, 1971a.

§Lehmkuhl, 1972; 1974.

¶Hilsenhoff, 1971.

\*\*Hoffman and Kilambi, 1971.

(Pearson, Kramer, and Franklin, 1968; Hilsenhoff, 1971; Hoffman and Kilambi, 1971; Lehmkuhl, 1972).

A diverse mayfly and stone fly fauna occurs in a downstream study area, but some taxa, such as gastropods and amphipods, were not found and apparently are restricted to the stream section below the dam.

## DISCUSSION

Figure 2 shows hypothetical interrelationships between the factors suggested as largely responsible for the reduced macroinvertebrate diversity in streams below deep-release dams. It is postulated that these factors result, in large part, from the altered thermal regime.

### Seasonal Constancy

The seasonal thermal constancy below dams may affect diversity in a number of ways. Most natural streams have environmental conditions conducive to what Hutchinson (1953) calls nonequilibrium communities. Unlike equilibrium communities, in which competitive exclusion reduces niche overlap, nonequilibrium communities occur where temporal fluctuations create environmental conditions that alternately favor different species. Much of the diversity of macroinvertebrates in unregulated streams may be a result of this phenomenon. Ide (1935) noted that the number of species of the mayfly genus *Stenonema* occurring in different parts of an Ontario stream was positively correlated with the temperature range. Reduced niche overlap would be expected under the constant conditions in tropical streams, but little information is available (Hynes, 1970).

The relatively constant environmental conditions found in streams below deep-release dams may increase coactive

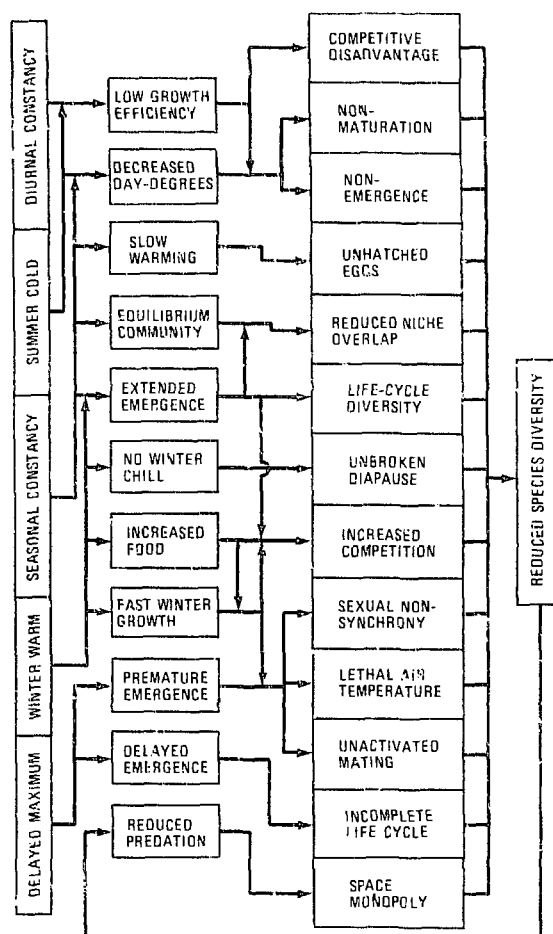


FIG. 2 Hypothetical thermal effects of deep-release dams on macroinvertebrate diversity.

patterns and cause a shift toward an equilibrium community. Although only part of the constancy is directly caused by thermal conditions, other factors may result indirectly from the temperature pattern. For example, the epiphytic algal community, which is seasonal and highly variable in unregulated mountain streams, provided an abundant year-round food source below Cheesman Dam.

Certain species (e.g., *Ephemerella g. grandis* and *Rhythrogena doddsi*) occurred rarely and only at the lowest sampling site but are common below the study section. These may represent "fugitive species" (Hutchinson, 1951) which are unable to find an unfilled niche in the more constant environment farther upstream. It is thought that site 4 represents an ecotone between the altered temperature upstream and the natural temperature downstream.

The necessity of near freezing followed by higher temperatures to stimulate hatching may explain the absence of some mayflies (Lehmkuhl, 1972) and perhaps other insects below deep-release dams.

*Gammarus lacustris* is one species favored by the conditions below the dam. This amphipod is a cold stenotherm normally found only at higher elevations in Colorado and is able to complete its life cycle under constant temperature conditions (Smith, 1973). However, Cairns (1972) emphasizes that a seasonal cycle of temperature is essential for the maintenance of most aquatic communities. Seasonal constancy also influences emergence and larval maturation, which are discussed in subsequent sections.

### Diurnal Constancy

The biological significance of daily variations in stream temperature remains virtually uninvestigated. It has been suggested that temperate stream organisms are adapted to diurnal temperature fluctuations and that a constant temperature would be generally disadvantageous (Hubbs, 1972), but the reasons for this have not been elucidated. Perhaps various physiological processes have different temperature optima within the same organisms, and the nearly constant diurnal temperatures below dams fail to provide the thermal range necessary to allow all body processes to operate optimally, at least for a time. The assumption that the marine abyssal environment is ideal in its thermal constancy was challenged by Moore (1966), who stated that, because of the different temperature optima of various enzyme, nerve, and reproductive processes, a thermally constant environment may call for unusual specialization. Thus species with body processes requiring a wide daily temperature range for optimal energetic efficiency may have a competitive disadvantage.

Diurnal fluctuations as great as 6°C at site 4 contrasted with the nearly constant temperature at site 1. The likelihood that the diurnal temperature pattern is important for many stream organisms in temperate or mountain regions warrants further study.

### Summer Cold Conditions

The ~4°C temperature of the hypolimnion of deep stratified reservoirs results in summer temperatures below deep-release dams which are considerably colder than those of unregulated streams and even those of nonthermal springs in the same region.

Species diversity may be reduced by the elimination of species for which the number of day-degrees is inadequate for larval maturation or emergence. Reduced growth efficiency at low temperatures may eliminate species even though the temperature is within the tolerance range of the organism (Edgington and Hildrew, 1973), presumably by causing a competitive disadvantage. *Baetis rhodani*, however, exhibited similar growth in isothermic and normal streams in Ireland (Fahy, 1973). Species capable of such metabolic adaptation would certainly be favored in streams below dams. It is notable that *Baetis* was the most abundant organism in this study, comprising 39.4% of all macroinvertebrates in the study section. Although growth rates were not measured, its seasonal pattern of larval

densities was remarkably similar at sites 1 and 4 and apparently is not influenced by the different temperature regimes (Fig. 3). *Baetis* exhibited similar relative abundance at all sites. *Simulium arcticum*, on the other hand, exhibited quite different seasonal patterns at the two sites, and was nearly four times more abundant at site 4 than at site 1.

### Winter Warm Conditions

The warmer winter temperatures below deep-release dams have various direct and indirect effects on stream macroinvertebrates.

Species that require winter chill to break egg or larval diapause will be eliminated (Lehmkuhl, 1972). The requirement of some species for near-freezing temperatures followed by higher temperatures to stimulate hatching has already been mentioned in regard to seasonal constancy.

Species that do not require low temperatures for hatching may be eliminated indirectly if their winter

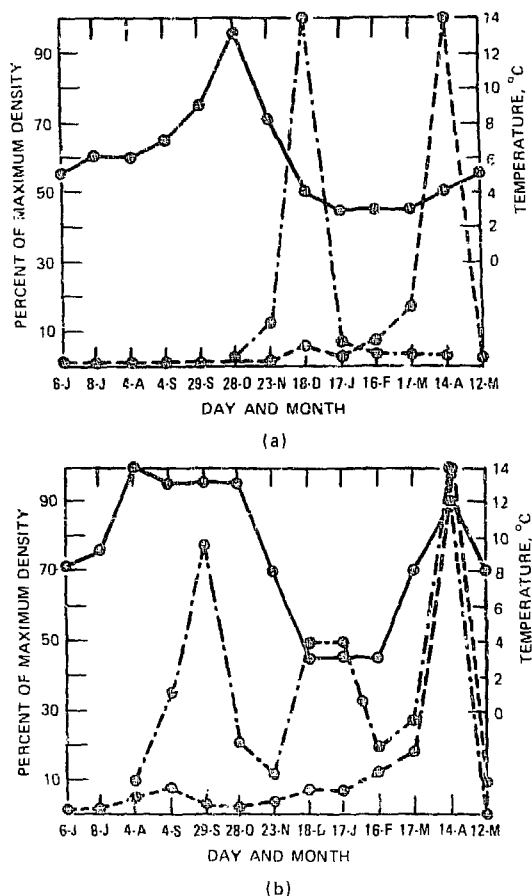


FIG. 3 Seasonal temperature pattern and density of *Baetis* and *Simulium* at sites 1 (a) and 4 (b), South Platte River, Colorado. —•—•—, temperature; - - - - *Simulium arcticum*; · · · · *Baetis* sp.



growth rate is accelerated. A 1°C increase in water temperature caused Hydropsychidae to emerge two weeks earlier in the Columbia River (Courtant, 1968). Premature emergence may eliminate species in several ways (Nebeker, 1971). Emergence may occur at a time when air temperatures are lethal to aerial adults. Surviving adults may not reproduce because mating mechanisms are not activated at low air temperatures or because of nonsynchronous emergence of males and females.

Winter warm conditions and seasonal constancy may result in extended emergence for some species in regions with warm winter air temperatures. After 8 years of personal observation, I concluded that adult mayflies and midges can be found on the South Platte River during any month of the year. Extended emergence may result in increased life-cycle diversity and increased and more constant productivity. Since the niche of a species may change throughout the life cycle, the temporal spacing of overlapping generations may occupy niches that would have been available for other species. Multivoltine cycles are comparatively rare among stream invertebrates (Hynes, 1970). The increased and more constant productivity may also reduce species diversity by increasing the intensity of competition. The effect of a year-round constant food supply on niche overlap was mentioned previously. In addition, results of a few studies indicate that high densities may eliminate rare species from a population (Hohn, 1961; Patrick, 1970), apparently by more intense competitive interactions. Yount (1956), studying a constant-temperature spring, suggested that species diversity is inversely correlated with community productivity. In addition to increased and more constant macroinvertebrate productivity resulting from extended emergence, the warmer winter temperatures and the absence of ice may allow a high year-round production of *aufwuchs* in the nutrient-rich water below deep-release dams. The high productivity below some deep-release dams, partly a result of altered temperature, may therefore offer an additional explanation for low macroinvertebrate diversity.

### Delayed Seasonal Maximum

At the upper three sites, the temperature maximum (13°C) was delayed until the end of October. Dam construction on the Columbia River is also causing a shift of maximum temperatures later into the year (Templeton and Courtant, 1971). Edington (1966) suggested that the alteration of seasonal temperature maxima in streams may be of significance, but little attention, or even recognition, has been given to the delayed seasonal maximum below hypolimnial-release dams.

It is conceivable that the absence of rising temperatures in summer may fail to stimulate hatching in some species and, depending on their life-history patterns, may either delay or cause precocious emergence in others. Potential effects of premature emergence have already been discussed. Delayed emergence could eliminate species by

resulting in emergence during midwinter rather than autumn.

Since the seasonal temperature range and other physicochemical factors were similar for the four sites on the South Platte River, much of the downstream difference in diversity must be a result of differences in diurnal temperature patterns and seasonal displacement.

### Reduced Predation

Major predator species may be among those eliminated or reduced in numbers below deep-release dams. Plecoptera are especially affected (Spence and Hynes, 1971a). In Colorado only one species of stone fly was found in the study section, and it was not found at site 1. This species and many other stone flies occur in abundance in a downstream study area. Fish diversity may also be reduced below hypolimnial-release dams, and this may further reduce predation pressure (Spence and Hynes, 1971b). Paine (1966) found that removal of predators from the intertidal zone decreased the diversity of the community and that organisms were affected whether or not they were directly preyed upon. An increased standing crop also followed removal of the top carnivore. It is of further interest that the most diverse benthic intertidal community studied by Paine was associated with the greatest annual temperature fluctuation.

The stream environment below the dam in Colorado is in some ways analogous to the rocky intertidal zone since space rather than food is a major limiting factor.

It is hypothesized that temperatures below dams reduce the number and abundance of predators; this further reduces diversity by allowing space monopolization by dominant prey species. *Baetis*, which reached densities greater than 12,000 organisms/m<sup>2</sup>, exemplified this.

The standing crop of macroinvertebrates is thought to be limited primarily by low growth efficiency at site 1 and primarily by space at site 4. If this is true, it would explain the downstream increase in standing crop in the study section and the downstream decrease in equitability (Fig. 1).

### Implications for Dam Construction and Operation

Poor project design has been cited (Pfitzer, 1967) as one reason for the poor fisheries in certain cold tail waters of the southeastern United States. The penstock intake may be at a depth that results in the discharge of cold water in spring and early summer only; this provides a thermal regime unsuitable for either a trout or warm-water fishery.

Hubbs (1972) warns against fixed outlets in dams, which, if water is released from a stratified reservoir, allow the release of either cold or warm water only; this can cause thermal shock as the water level drops. He points out that a combination of a deep and a shallow outlet may have the same effect if discharge is shifted from the upper (warm) to the lower (cold) outlet as the reservoir level falls, and he suggests that dams be constructed to allow water to be

# Thermal Effluent in Coastal Rivers: A Contribution to the Location of Power Plants

R. C. Baird  
J. A. Watson

## ABSTRACT

A study of possible environmental impact was made at a fossil-fueled power-generating plant at Port Wentworth, on the Savannah Front River near the city of Savannah, Ga. A major contribution of this study to future site planning concerns the potential advantages of locating power plants in the vicinity of the freshwater-saline interface of a large river system. Evidence is presented which indicates that the Port Wentworth plant is near the region of maximum faunal change from a predominantly freshwater biological community to one dominated by marine and estuarine forms. Individuals found in such regions are in or near physiologically marginal habitats and are not likely to contribute significantly to resident populations either upriver or downriver. The region from 1 to 3‰ salinity appears to be near the tolerance minimum of most euryhaline estuarine-marine forms. Most of the freshwater organisms passing downriver will likely be lost to reproducing populations. A discussion of potential damage to populations of anadromous or freshwater-tolerant nektonic species is presented along with some ecological considerations of riverine ecosystems in general.

With the proliferation of power-generating facilities has come an increasing awareness of potential environmental impact, a greatly expanded research effort to determine the degree and extent of such impact, and considerably more stringent laws governing power-plant operating limitations. The resulting accumulation of ecological data from a variety of environments and power-plant sites has made it apparent that the nature and often the degree of environmental impact, although site specific, can be predicted, at least qualitatively, if we know the plant design and the physical and biological characteristics of the site environment. This predictive capability can be used to locate power-generating facilities as well as to provide direction for future environmental research, particularly research directed toward expanding and refining predictive capabilities.

With the expected increase in power demand in the coming years and the probable continued high cost of expanding facilities, efficiency and fiscal economy will have to be among the important considerations in power-plant design and location. At present, once-through cooling systems remain the most efficient and economical means of providing cooling water for steam condensers used in generating electricity. Locations that provide sufficient water for once-through cooling yet incur minimum environmental damage are prime prospects for power-plant site selection. A major contribution of this study of possible environmental impact at a power-generating plant on the Savannah Front River concerns the potential advantages of locating power plants in the vicinity of the freshwater-saline interface of a large river system. Evidence is presented to support the view that, from an environmental impact standpoint, such locations may be acceptable for power plants with once-through cooling systems. A discussion of potential damage to populations of anadromous or freshwater-tolerant nektonic species is presented along with some ecological considerations of riverine and estuarine ecosystems in general.

## PHYSICAL CHARACTERISTICS

Where relatively large volumes of freshwater enter the sea, there is a zone where the salinity of the water increases beyond 1‰ as it proceeds into the true estuarine regions of a coastal river. This zone, formally defined by estuarine ecologists as the limnetic-oligohaline boundary (e.g., Day, 1951; Carriker, 1967), is the area corresponding to the freshwater-saltwater interface discussed here. Although it is defined by salinity ranges, it is generally not a fixed geographic point along a coastal river but is a zone of transition which moves upriver and downriver in response to tides and river flow. In addition, density differences between freshwater and saline water often cause vertical

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Watson: Geo-Marine, Inc., Richardson, Texas.

taken from a varying combination of reservoir depths to maintain a more normal diurnal temperature regime in the receiving stream.

The section of the South Platte River below the dam, with a high standing crop of macroinvertebrates and ice-free conditions, provides a productive year-round trout fishery. If we accept the premise that environments with low species diversity are especially susceptible to perturbation, however, the situation is less desirable. Cairns (1972), in reference to thermal pollution, listed some requirements to preserve aquatic communities. These were, among others, to maintain a normal species diversity and productivity and a temperature pattern not disruptive to the organisms' seasonal cycles. Perhaps these same requirements would be appropriate considerations in the construction and operation of deep-release dams.

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stratification in which bottom waters exhibit higher salinity than surface waters (this is the well-known salt-wedge effect of the headwaters of riverine-estuarine systems). For purposes of definition, the saline-freshwater interface is the zone near the upper limit of saltwater intrusion of riverine estuarine systems in which the mean annual salinity is less than 1‰ throughout the water column but may exceed this value during unusual conditions (e.g., very low freshwater river flow or extremely high tides).

To illustrate the general nature of the physical environment in interface regions, some results are presented of an ecological investigation of a power plant located at Port Wentworth, Ga. (Fig. 1), on the Savannah River near the city of Savannah. The data were compiled from numerous sources, the principal ones being reports by the Environmental Protection Agency (1971), Inman (1971), the Georgia Water Quality Control Board (1972), Rees (1972), Stickney and Miller (1973), the U. S. Geological Survey (1974), and the Georgia Department of Natural Resources (1974). The Savannah River, a medium-sized river of the coastal plain of the central and southeastern United States, typifies conditions found in many such riverine systems. Salinity values for a 10-year period (1958 to 1966, 228 observations) ranged from 0.03 to 11.38‰ with a mean of 0.13‰. Salinity exceeded 0.21‰ at only six observations. These represent both surface and near-

bottom measurements taken approximately 1.5 miles upriver from the plant site. Salinity measurements taken at the cooling-water intake of the plant at a mean depth of about 7 ft during a period of low river flow (Fig. 2) illustrate both the extreme variability in salinity and the marked influence of tidal factors in the region of the interface. Other salient features of the interface environment are as follows: There are marked seasonally correlated changes in temperature with a mean of 20°C and a range from 7 to 29°C. Mean monthly freshwater inflow (1972 to 1973) ranged from 7407 cfs in October to 34,870 cfs in April, reflecting seasonal patterns in rainfall and runoff. Average maximum annual velocity at ebb and flood were 2.5 and 1.5 fps, respectively, with a mean tidal range of about 7 ft. The river is generally well mixed, and several reports indicated little thermal stratification. Turbidity was high, ranging from 15 to 95 Jackson turbidity units (JTU), with total suspended solids varying from 8 to 52 mg/liter. Bottom sediments near the plant were primarily sand with some mud. The plant is located on a navigational channel maintained to a depth of 30 ft, with abrupt changes in depth near shore. High flow rates, frequent maintenance dredging, and ship traffic contribute significantly to continuous removal of deposited sediments on the river bottom.

## BIOLOGICAL CHARACTERISTICS

Numerous investigators have noted the major and rather abrupt faunal and floral replacement occurring in the region of the saline-freshwater interface (see, e.g., Nash, 1947; Pennak, 1953; Carriker, 1967; Green, 1968). A graphic representation (Fig. 3) indicates that productivity, number of species, and biomass appear to decline abruptly in the interface region and increase rapidly both upstream and downstream. Upstream the fauna and flora are dominated by freshwater species that are generally unable to live in saline environments and are often phylogenetically remote from saltwater forms. Estuarine and marine species are dominant downstream. The change in species composition appears to affect almost all phyla, including attached or rooted plants, algae, plankton, benthic invertebrates, and most nektonic forms. The reader is referred to Pennak (1953), Carriker (1967), and Green (1968) for extensive reviews of the major taxa.

The causes for the zonation appear to involve numerous factors, and the location, nature, and degree of faunal and floral replacement vary according to the specific environmental characteristics and rates of salinity change for each river system. Reviews by Green (1968), Gunter, Ballard, and Venkataramaiah (1973), and others indicate that the region from 1 to 3‰ salinity is near the tolerance minimum of most euryhaline estuarine-marine forms and that ~5‰ is the maximum salinity tolerance for most freshwater forms. Thus problems of osmoregulation are a major factor involved. Green (1968) also noted the effect of reduced salinities on the ability of estuarine polychaete larvae and copepods to swim. Gunter, Ballard, and Venkataramaiah (1973) noted the generally smaller sizes and

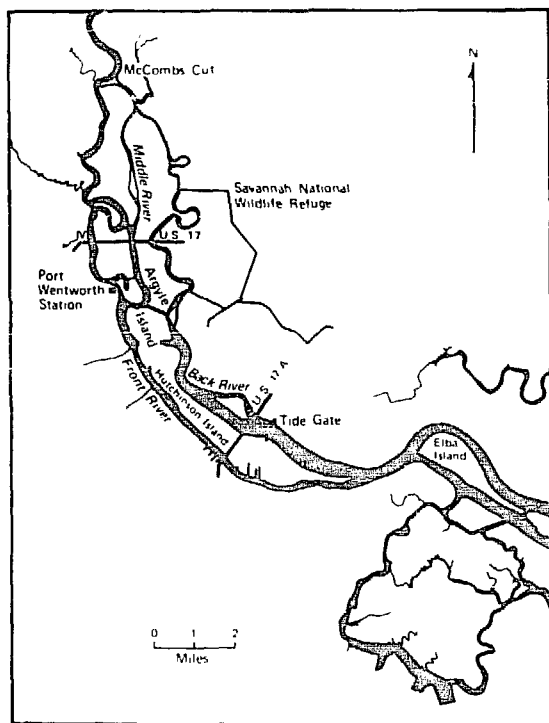


FIG. 1 Lower Savannah River.

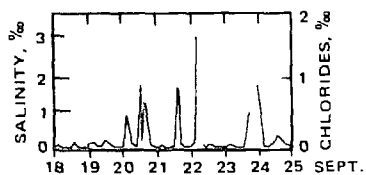


FIG. 2 Salinity data from September 1972 and September and October 1974 at Port Wentworth power plant. Samples were taken at intake pumps, and depth of sampling in the water column varies with tidal stage.

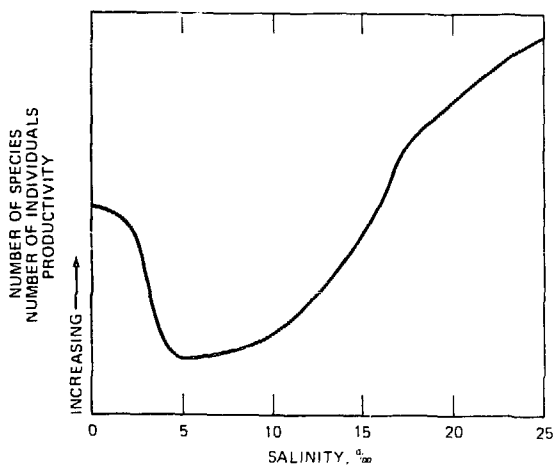
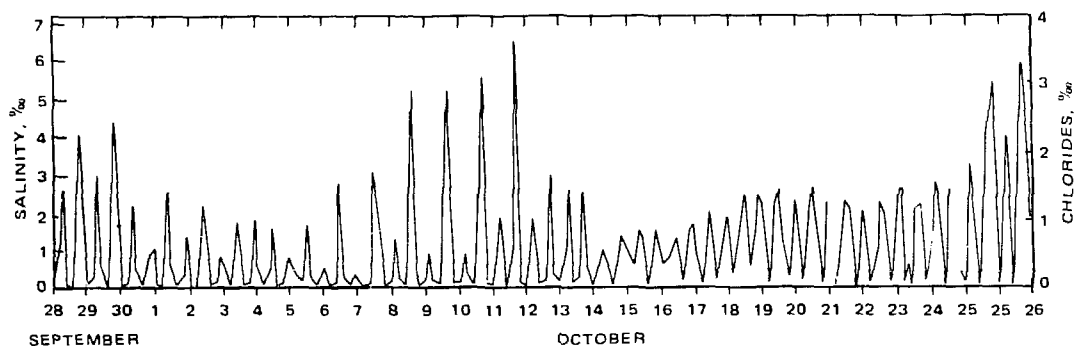


FIG. 3 General representation of the nature of change in biota with salinity in riverine-estuarine systems. (Adapted from Pennak, 1953, and others.)

growth rates of juvenile shrimp and of some estuarine fish in salinities less than about 8‰. Factors affecting interspecific competition are of critical importance, and species living outside their adaptive optima may well be excluded from an area by competition from other species (Riley, 1967). Conditions in freshwater regions of a large river are generally much different from those found in the oligoha-

line and estuarine regions nearer sources of saline water. Nash (1947), Pennak (1953) and Patrick, Cairns, and Roback (1966), among others, noted the major differences in reproductive strategies of organisms from the two environments, the tendency of freshwater forms to be highly adapted to small specialized habitats within riverine systems, the increased turbidity and subsequent reduction of the euphotic zone in rivers, the generally small size of freshwater species, and the often considerable phylogenetic differences among freshwater and marine forms.

A general summary of the faunal change in the Savannah River from the plant site at Port Wentworth to Elba Island, about 6 miles downriver, was compiled from a number of sources, principally reports by the U. S. Public Health Service (1961), the Georgia Water Quality Control Board (1972), Rees (1972), and Stickney (1972). These reports indicate that zooplankton and phytoplankton populations, particularly in the main channel, are generally in low abundance at Port Wentworth. With few exceptions, the organisms listed were freshwater in character. Both drift and benthic organisms were also primarily freshwater forms, but a few brackish water species were noted. Diurnal measurements of dissolved oxygen made at Elba Island indicated little diurnal fluctuation and, therefore, low primary production. Nektonic invertebrates were generally rare at Port Wentworth; a freshwater form of the glass shrimp *Palaemonetes* was the most abundant. Downstream estuarine shrimp of the genus *Penaeus* were abundant, as was the blue crab *Callinectes sapidus*. No encrusting invertebrates (barnacles and oysters) were noted at Port

Wentworth, but they were present several miles downstream. The fish fauna was also markedly different at the two sites. The ichthyofauna at Port Wentworth (Table 1) is essentially freshwater in composition and includes several euryhaline marine or estuarine species that are common in the tidal freshwater areas of coastal streams of the southeast (Smith, 1971), as well as a number of anadromous fishes that enter freshwater to spawn. The downstream fishes (Table 2) are dominated by marine elements known to use estuaries as nursery areas.

In summary, the evidence from the Savannah River study and other riverine-estuarine systems indicates that the region of the limnetic-oligohaline interface is one of maximum faunal change from a predominately freshwater community to one dominated by marine and estuarine forms. Individuals found in interface regions are in or near physiologically marginal habitats of high variability and stress and are not likely to contribute significantly to resident populations upriver or downriver. Most of the freshwater organisms passing downriver are lost to reproducing populations. The primary freshwater nektonic species found above the interface region are generally demersal spawners, and, in any case, eggs and larvae passing downriver again are likely to be lost to source populations. Most of the estuarine marine nekton below the interface region are either demersal spawners or spawn in marine environments offshore (McHugh, 1967), and optimal habitats for juveniles appear to be seaward of the interface region. This region is ecologically significant as a route whereby anadromous or freshwater-tolerant marine fish species move between estuarine and freshwater environ-

TABLE 2 Savannah River Fish Species From Study near Elba Island

| Family         | Scientific name                 | Common name           |
|----------------|---------------------------------|-----------------------|
| Clupeidae      | <i>Brevoortia tyrannus</i>      | Atlantic menhaden     |
|                | <i>Opisthonema oglinum</i>      | Thread herring        |
| Engraulidae    | <i>Anchoa hepsetus</i>          | Striped anchovy       |
|                | <i>Anchoa mitchelli</i>         | Bay anchovy           |
| Ictaluridae    | <i>Ictalurus catus</i>          | White catfish         |
| Ariidae        | <i>Arius felis</i>              | Sea catfish           |
|                | <i>Bagre marinus</i>            | Gaff-topsail catfish  |
| Batrachoididae | <i>Opsanus tau</i>              | Oyster toadfish       |
| Pomatomidae    | <i>Pomatomus saltatrix</i>      | Bluefish              |
| Carangidae     | <i>Chloroscombrus chrysurus</i> | Atlantic bumper       |
|                | <i>L'omer setapinnis</i>        | Atlantic moonfish     |
| Sclacnidae     | <i>Bairdiella chrysura</i>      | Silver perch          |
|                | <i>Cynoscion regalis</i>        | Weakfish              |
|                | <i>Larimus fasciatus</i>        | Banded croaker        |
|                | <i>Leiostomus xanthurus</i>     | Spot                  |
|                | <i>Micropogon undulatus</i>     | Atlantic croaker      |
|                | <i>Stellifer lanceolatus</i>    | Star drum             |
| Mugilidae      | <i>Mugil cephalus</i>           | Striped mullet        |
| Trichiuridae   | <i>Trichiurus lepturus</i>      | Atlantic cutlass fish |
| Stromateidae   | <i>Peprilus alepidotus</i>      | Harvest fish          |
| Bothidae       | <i>Etropus crossotus</i>        | Fringed flounder      |
|                | <i>Paralichthys lethostigma</i> | Southern flounder     |
| Soleidae       | <i>Trinectes maculatus</i>      | Hogchoker             |
| Cynoglossidae  | <i>Symphurus plagiusa</i>       | Blackcheek tonguefish |
| Tetraodontidae | <i>Sphoeroides maculatus</i>    | Northern puffer       |

ments. In determining potential environmental impact by a power plant located in an interface region, we must focus our concern on attempts to assess and recommend designs that minimize impact to such populations, particularly in the early life-history stages of vulnerable anadromous species (which are often of major sport or commercial importance).

## POTENTIAL ENVIRONMENTAL IMPACT

The entrainment (passage of organisms through condensers of once-through cooling systems) of plankton and drift organisms in the interface region is not considered to be a problem of major environmental concern. Planktonic organisms are generally sparse in such areas and, in any case, will encounter a salinity barrier not far downstream. Even when a large percentage of the mean river flow is processed and thermally loaded, little threat to the indigenous plankton populations upstream or downstream is envisioned.

As indicated, the benthic community near the interface region is in a highly stressed environment in which populations are not likely to contribute to source populations upriver or downriver. The benthos is not generally vulnerable to entrainment, except for certain larval stages, and, if thermal discharges are located near deep fast-flowing main channels, the density gradient is such that the heated water tends to remain near the surface. Therefore, in many cases the benthic community will not be subjected to elevated temperatures. (Figure 4 illustrates the depth of the 5°F above ambient isotherm at slack tide at Port Went-

TABLE 1 Savannah River Fish Species, Mile 22, Port Wentworth, Georgia

| Family         | Scientific name                 | Common name         |
|----------------|---------------------------------|---------------------|
| Lepisosteidae  | <i>Lepisosteus oculatus</i>     | Spotted gar         |
| Amiidae        | <i>Amia calva</i>               | Bowfin              |
| Anguillidae    | <i>Anguilla rostrata</i>        | American eel        |
| Clupeidae      | <i>Alosa sapidissima</i>        | American shad       |
| Clupeidae      | <i>Brevoortia tyrannus</i>      | Atlantic menhaden   |
| Clupeidae      | <i>Dorosoma cepedianum</i>      | Gizzard shad        |
| Engraulidae    | <i>Anchoa mitchelli</i>         | Bay anchovy         |
| Synodontidae   | <i>Synodus foetens</i>          | Inshore lizard fish |
| Cyprinidae     | <i>Cyprinus carpio</i>          | Carp                |
| Cyprinidae     | <i>Notropis lutipinnis</i>      | Yellowfin shiner    |
| Ictaluridae    | <i>Ictalurus catus</i>          | White catfish       |
| Ictaluridae    | <i>Ictalurus punctatus</i>      | Channel catfish     |
| Atherinidae    | <i>Menidia menidia</i>          | Atlantic silverside |
| Percichthyidae | <i>Morone chrysops</i>          | White bass          |
| Percichthyidae | <i>Morone saxatilis</i>         | Striped bass        |
| Centrarchidae  | <i>Micropterus salmoides</i>    | Largemouth bass     |
| Centrarchidae  | <i>Lepomis microlophus</i>      | Redear sunfish      |
| Centrarchidae  | <i>Lepomis auritus</i>          | Redbreast sunfish   |
| Centrarchidae  | <i>Pomoxis nigromaculatus</i>   | Black crappie       |
| Centrarchidae  | <i>Pomoxis annularis</i>        | White crappie       |
| Mugilidae      | <i>Mugil cephalus</i>           | Striped mullet      |
| Bothidae       | <i>Paralichthys lethostigma</i> | Southern flounder   |
| Soleidae       | <i>Trinectes maculatus</i>      | Hogchoker           |

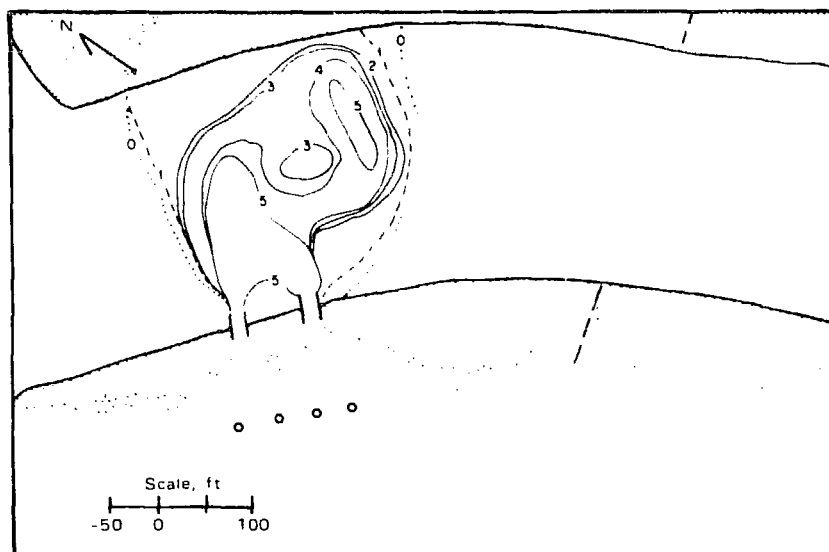


FIG. 4 Distribution of 5°F delta isotherm vs. depth at Port Wentworth power plant, Sept. 7, 1972.

worth.) The frequent dredging and ship traffic often found in coastal rivers may well prevent significant populations of benthic organisms from occurring at all. Unless thermal discharges are located in shallow areas or on small tributaries, significant damage to rooted or attached aquatic plants also appears limited.

#### FRESHWATER-TOLERANT MARINE NEKTONIC SPECIES

With few exceptions, significant populations of marine or estuarine invertebrate nekton do not enter freshwater (Gunter, Ballard, and Venkataramaiah, 1973), and population centers are in salinities above those of the interface region. Small juveniles and adults of freshwater-tolerant marine fish species (e.g., silversides, anchovies, and mullet) might at times be vulnerable to impingement or entrainment at intake structures. Since these species are extremely abundant and have high reproductive potentials, even impingement of large numbers of individuals is not likely to affect resident populations significantly. In addition, these species are primarily estuarine marine forms reproducing in marine environments and appearing most abundantly in shallow habitats (Rees, 1972; Carr and Giesel, 1975). Additional precautions taken to reduce vulnerability of anadromous fishes to impingement-entrainment and thermal effects, discussed in the following section, will also apply to these species.

#### ANADROMOUS FISH SPECIES

Many large riverine-estuarine systems have populations of anadromous fish species which ascend the river to spawn

in freshwater environments. The distance upriver of spawning sites, the distribution of eggs and larvae, the tolerance of eggs and larvae to salinity, and the habitat selection, diet, and size of juveniles entering saline waters are all important considerations in assessing potential impact by power plants located near the interface region. Reproductive potential, natural mortality, and the density-dependent effects on growth and survival which compensate for observed mortality need to be considered in evaluating the significance of observed or expected plant-induced mortality to anadromous fish populations. Since the number and taxonomic composition of anadromous species vary among riverine-estuarine systems, the salient biological factors must be evaluated for each region. To illustrate, we discuss the anadromous fishes of the Savannah River in relation to vulnerability to power-plant-induced mortality in the saline-freshwater interface region. Many of these fish species have been intensively studied, and the synopsis that follows includes information from Sykes (1956), Talbot (1966), Godwin (1968), Street (1969), Adams (1970), Smith and Gardner (1970), the Georgia Game and Fish Commission (1971), Rees (1972), McBay (1974), and others.

The anadromous fishes of the Savannah River include three species of shad (*Alosa* sp.), sturgeon (*Acipenser* sp.), and the striped bass (*Morone saxatilis*). The clupeid (*Alosa*) species all migrate far upriver (60 to 120 miles) to spawn, and eggs, larvae, and young juveniles develop well upriver. Adult and older juvenile clupeids descend the river and at times occur in abundance in the interface region. Less is known of the sturgeon, which appears to migrate and spawn far upriver. Striped bass, like the other species,

# An Analysis of the Applicability of EPA's Draft Water-Temperature Criteria: A Site-Specific Case-History Evaluation

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## ABSTRACT

The applicability of the Environmental Protection Agency's draft water-temperature criteria in evaluating the impact of thermal discharges from Appalachian Power Company's Glen Lyn Power Plant on the New River in Virginia is analyzed. The draft water-temperature criteria are applied to important representative fish species in New River. Since field studies have been conducted at the site, it is possible to analyze the predicted impact using the draft water-temperature criteria and the actual impact as documented by the field studies. No data on water-temperature criteria were available for some important representative fish species at the plant site, but, for species for which data were available, the predicted impact was corroborated by the field studies. It was not possible, however, to make a realistic impact assessment using the draft water-temperature criteria at the Glen Lyn site. Field data on the aquatic community and on fish preference and avoidance temperatures proved more useful in evaluating the impact of the thermal discharges.

Of the 1273 steam power plants projected for operation by the year 1983, 966 plants were operating by 1972. Approximately 70% of these are located on river systems, and the remainder use cooling waters from lakes, oceans, and estuaries (Rosen et al., 1975). Steam electric power plants operating since Jan. 1, 1974, with a capacity of 25 to 500 Mw and new plants with at least a 25-Mw capacity projected for operation by 1983 must apply for a variance to mandatory closed-cycle cooling unless their discharge is to contained cooling ponds or unless land availability limits the consideration of alternatives. Regulations published by the Environmental Protection Agency (EPA) set forth the various types of demonstrations needed to obtain a variance to closed-cycle cooling (*Federal Register*, 1974). Public Law 92-500, Section 316(a), provides that, if an owner or operator, after opportunity for a public hearing, can demonstrate to the satisfaction of the relevant permitting

authority that any effluent limitation proposed for the control of the thermal component of discharge from a power plant will require effluent limitations more stringent than necessary to assure the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on the body of water into which the discharge is made, the permitting authority may impose a less stringent, alternative effluent limitation. The burden of the proof is on the discharger to demonstrate that an alternative effluent limitation will indeed protect the biological community. Rules and regulations governing Section 316(a) published in the *Federal Register* set forth three types of demonstrations by which a discharger can submit evidence for a less stringent effluent limitation.

Type 1: The use of on-site data to demonstrate absence of prior appreciable harm to the biological community.

Type 2: A demonstration using the EPA's draft water-temperature criteria that conditions at the site would protect important representative species.

Type 3: The use of a combination of biological and engineering data to demonstrate no adverse effects.

The objective of this paper is to evaluate the applicability of EPA's draft water-temperature criteria, which are used in a type 2 demonstration, by applying the criteria to the important representative fish species in the New River in Virginia at Appalachian Power Company's Glen Lyn Power Plant. The draft water-temperature criteria were published in September 1974 by the Water Planning Division, Office of Water and Hazardous Materials, Environmental Protection Agency, in Appendix B of a draft manual on Section 316(a) entitled "316(a) Technical Guidance Thermal Discharges."

We have been involved in field and on-site laboratory studies evaluating the impact of heat discharges from Glen Lyn Power Plant on aquatic life in New River for more than

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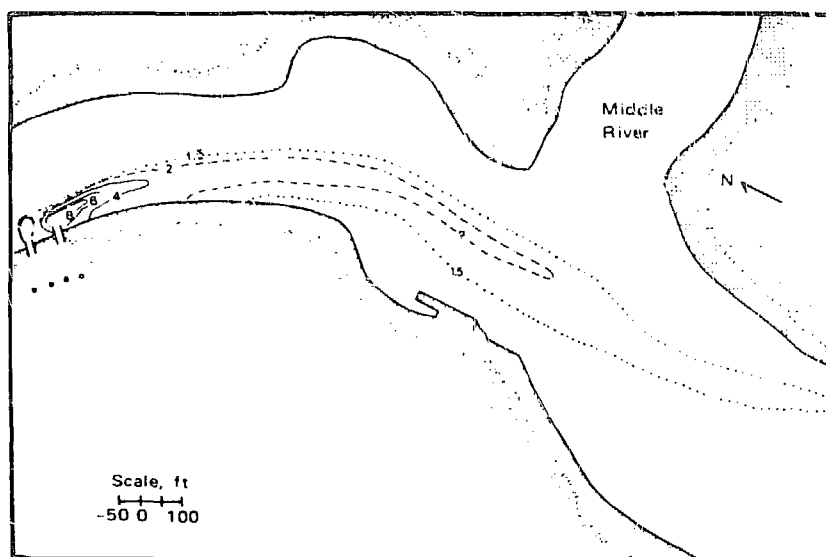


FIG. 6 Surface isothermal contour chart ( $^{\circ}\text{F}$ ) at Port Wentworth power plant, Sept. 7, 1972, during ebb tide.

removed by the plant, the potential population loss to osmotic barriers downriver, and the fecundity and natural mortality of the species must be considered. On the basis of our net sample data, we estimated that total yearly egg loss caused by plant operation ranges from 461,000 to 554,000, or one-half the reproductive potential of a single female.

Impingement potentially represents a significant source of mortality to resident fish populations, and serious consideration should be given to designing intake structures that reduce impingement mortality. Locating intakes on the main channel, as opposed to in intake canals or productive natural small tributaries, reduces vulnerability. Intake screens, bypasses, and reduced current flows may also reduce impingement effects.

## SUMMARY AND CONCLUSIONS

The evidence reviewed both here and elsewhere indicates that, from the standpoint of environmental impact, the location of power plants with once-through cooling systems near the freshwater-saline boundary of a large river system may be acceptable. An important potential impact appears to involve plant-induced mortality at various life-history stages of anadromous fish species, particularly those which spawn near the interface region. Proper location of intake and discharge sites and use of design criteria to reduce impingement-entrainment effects should be prime considerations. Mortality of other elements of the biota appears not likely to significantly affect populations of balanced communities upstream and downstream. Only recently has ecological information about the interface region been accumulated, and at present the ecology of such regions remains poorly understood. Addi-

tional studies designed to evaluate the ecological consequences of power-plant location are needed, and results should be incorporated into a comprehensive strategy for power-plant site selection. Current regulations concerning power-plant operating limitations are often inadequate because of an insufficient data base from which to make both environmentally and economically sound decisions.

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spawn from late winter (February and March) until late May, by which time eggs and larvae are no longer found. Striped bass eggs are released free, are denser than freshwater (specific gravity 1.005), and hatch after 36 to 72 hr. Spawning occurs in freshwater, and, although eggs are found at least 50 miles upstream, peak spawning areas are not far above the interface region (2 to 5 miles). Fry and fingerlings were not found to inhabit waters having salinities in excess of 3.2‰ and appear to utilize near-shore or shallow-water nursery areas, primarily in freshwater upstream of the interface area. Feeding studies of fry and fingerling striped bass in the Savannah and Ogeechee Rivers indicate that their diet is composed of prey organisms known to live in primarily freshwater habitats. Subadults and adults are often found in abundance in the interface region.

In summary, it appears that, with the possible exception of striped bass, which can be considered potentially vulnerable, eggs and larvae of most anadromous fishes are not vulnerable to entrainment at power-plant intake sites located near the interface region of the Savannah River. However, in view of the apparent osmotic barrier at about 3‰ salinity, eggs and larvae passing the interface region can be expected to suffer high mortalities and to make little if any contribution to existing populations. The apparent depauperate nature of the interface environment is indicative of the marginality of the interface region in general as a suitable habitat for fry and fingerling striped bass.

### THERMAL EFFECTS

It is well known that increased temperatures can affect fishes in numerous ways (Krenkel and Parker, 1969;

Sylvester, 1972; Clark and Brownell, 1973; Carr and Giesel, 1975) and that fishes are often attracted to thermal discharges. Figures 4, 5, and 6 illustrate a thermal plume from a power plant discharging into the main channel of the Savannah River. Of importance are the rapid dissipation of heated water with each tide (at any given geographic point), the limited extent of surface area affected, the shallow depth of the 5°F isotherm, and the limited exposure of much of the cross-sectional area (mean depth 30 ft) of the river to markedly elevated temperatures. Fishes may be attracted to the thermal discharge area, but its extent and rapidly changing dimensions and the fishes' ability to avoid unfavorable habitats and their tolerance to short-term elevations in temperature (Schubel and Auld, 1974; Romberg et al., 1974; Storr and Schlenker, 1974; Copeland, Laney, and Pendleton, 1974) make it unlikely that thermal loading, as illustrated here, causes significant mortality to fish populations. In addition, spawning seasons occur at times when river temperatures are relatively low, further reducing potential thermal-induced mortality.

### ENTRAINMENT-IMPINGEMENT

Entrainment effects may not be significant to fish eggs and larvae of anadromous fishes in the interface region since most fish either spawn upstream or, in the case of striped bass, face high mortalities in any case from the osmotic barrier immediately seaward. Older juveniles often found in this region are probably large enough to be able to avoid intake structures if they are properly screened. If eggs and larvae are entrained, estimates can be made of the significance of plant-induced mortality to source populations. The residence time and total amount of water

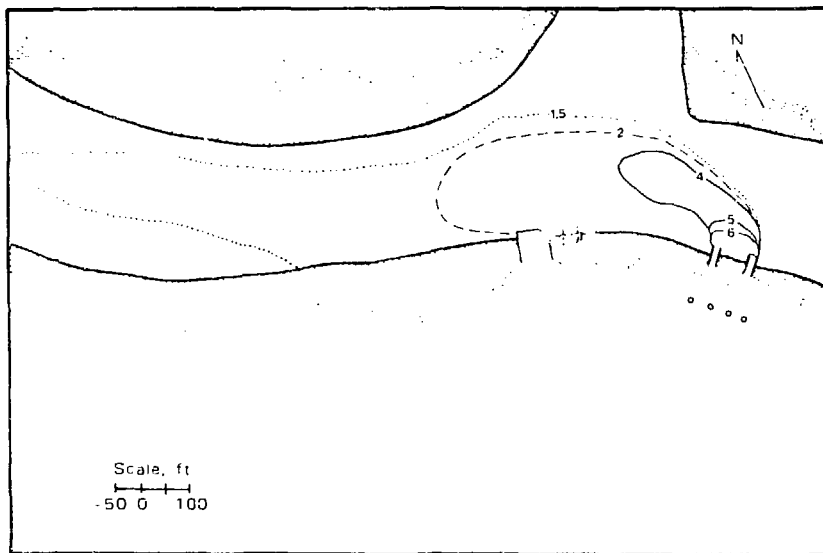


FIG. 5 Surface isothermal contour chart (°F) at Port Wentworth power plant, Sept. 8, 1972, during flood tide.

6 years. Extensive data on the aquatic communities were available for the type 1 demonstration of no appreciable harm. These data also offered the opportunity to test the applicability of the type 2 demonstration. Generally, a type 2 demonstration is recommended when on-site data are not available or for new plants that are not yet operating.

Basically EPA's draft water-temperature criteria consists of two principal recommendations. The first states that the maximum weekly average temperature should not exceed: (1) one-third of the range between the ultimate upper incipient-lethal temperature and the optimum temperature for growth; (2) a temperature that, if rapidly dropped to ambient temperature (i.e., causing cold shock), would cause mortality to important species; (3) a temperature that would prevent successful reproduction and development; or (4) a temperature that would prohibit the maintenance of normal species diversity. The second recommendation in the draft water-temperature criteria states that at no time should the temperature exceed the applicable maximum temperature for short-term summer and spawning exposures for important species.

We attempted to apply EPA's draft water-temperature criteria to the Glen Lyn site and to evaluate their applicability to that site. It would be presumptuous to assert that the Glen Lyn site is representative of conditions at a large number of other sites, but some of the information generated there may be useful elsewhere.

## SITE DESCRIPTION

From its origin in North Carolina, the New River, one of the headwaters of the Ohio River, flows northwest through three physiographic provinces (Blue Ridge, Ridge and Valley, and the Appalachia Plateau) in Virginia and ultimately merges with the Greenbriar River at Hinton, W. Va., to form the Kanawha River. The main stream of the New River consists of a series of extended riffles and long pools. The river is shallow under normal flow, and the bottom is composed primarily of tilted bedrock and gravel in the riffles and silt in the pools. The width of the river in the vicinity of Glen Lyn Power Plant is ~140 m.

Records of New River flow from 1915 to 1965 at Glen Lyn, Va., show that the 17-day consecutive low flow with a recurrence interval of 10 years was ~32 m<sup>3</sup>/sec. Once in 50 years a peak flow of 4361 m<sup>3</sup>/sec can be expected at Glen Lyn (Virginia Division of Water Resources, 1967).

Appalachian Power Company's fossil-fueled Glen Lyn plant is located at New River kilometer 153. The plant, built in 1919, had an installed capacity of 401 Mw. The present operating capacity is 350 Mw because three small generating units within the plant are obsolete. The plant is located on the west bank of the river immediately adjacent to the Highway 460 bridges. Of the 14.9 m<sup>3</sup>/sec of cooling water discharged by the plant, one unit (105 Mw) discharges 5.35 m<sup>3</sup>/sec directly into the New River ~46 m above the Route 460 bridge. The second unit (245 Mw) discharges 7.6 m<sup>3</sup>/sec via a 274-m effluent canal into the

East River, upstream from its confluence with the New River, and 2.0 m<sup>3</sup>/sec into the New River. The maximum theoretical rise in temperature in the discharge is 7.8°C under full load conditions.

Preliminary efforts to develop a thermal plume dispersion model in the New River have been conducted by Polk (1970) and Marchman, Shatz, and Kobeson (1975). To date, however, no definitive model exists. Temperature-transect data above and below the plant are available (Cairns et al., 1971; Cairns, Hendricks, and Dickson, 1973; Long, 1969; Robinson, 1970). By plotting data from these studies, we were able to estimate the location in the river of the 5.6 and 2.8°C isotherms under low-flow conditions of 36.8 to 42.5 m<sup>3</sup>/sec. From these estimates, we calculated the area encompassed by the isotherms. Below the power plant, ~2 ha of New River surface were contained within the 5.6°C isotherm under "worst" conditions (i.e., low flows and high temperature). An estimated 6 ha of surface were contained within the 2.8°C isotherm in the New River. The temperature-transect data revealed that very little vertical temperature stratification existed in the turbulent and shallow New River. Figure 1 shows the size of the 5.6 and 2.8°C isotherms in the river at the power plant. Under conditions of high flows, which existed during most of the year, the size of the area within the 5.6 and 2.8°C isotherms would be smaller. Sufficient temperature-transect data were not available, however, to delineate isotherms under these conditions.

## FIELD STUDIES CONDUCTED AT GLEN LYN Applying Draft Water-Temperature Criteria to Important Representative Fish Species

On-site biological studies, both laboratory and field, have been conducted at Glen Lyn Power Plant. Preference and avoidance temperatures and cold shock data for fish were determined in an on-site mobile laboratory (Cherry et al., 1975; Cherry, Dickson, and Cairns, 1975a; 1975b; Cairns et al., 1975), and field distribution studies of the fish community in relation to the heated discharge were conducted also (Stauffer, Dickson, and Cairns, 1974; Stauffer, 1975). Cairns et al. (1971) and Cairns, Hendricks, and Dickson (1973) studied the impact of the heated discharges from Glen Lyn Power Plant on the benthic macroinvertebrate community. The impact on algae and macrophytes has also been monitored (Rodgers, 1975). Dickson et al. (1975), using the results of these studies to prepare a type 1 demonstration on the impact of the heated discharge on the aquatic community, concluded that: A diverse and productive fish community exists in the New River around the power plant, and the heated discharges do influence fish species abundance. Channel catfish were shown to prefer the heated areas during certain times of the year, thus causing changes in the fish community structure. That most species (46 out of 48) have been collected in the heated discharge at some time during the year, however, indicates only a minor impact on total number of species.

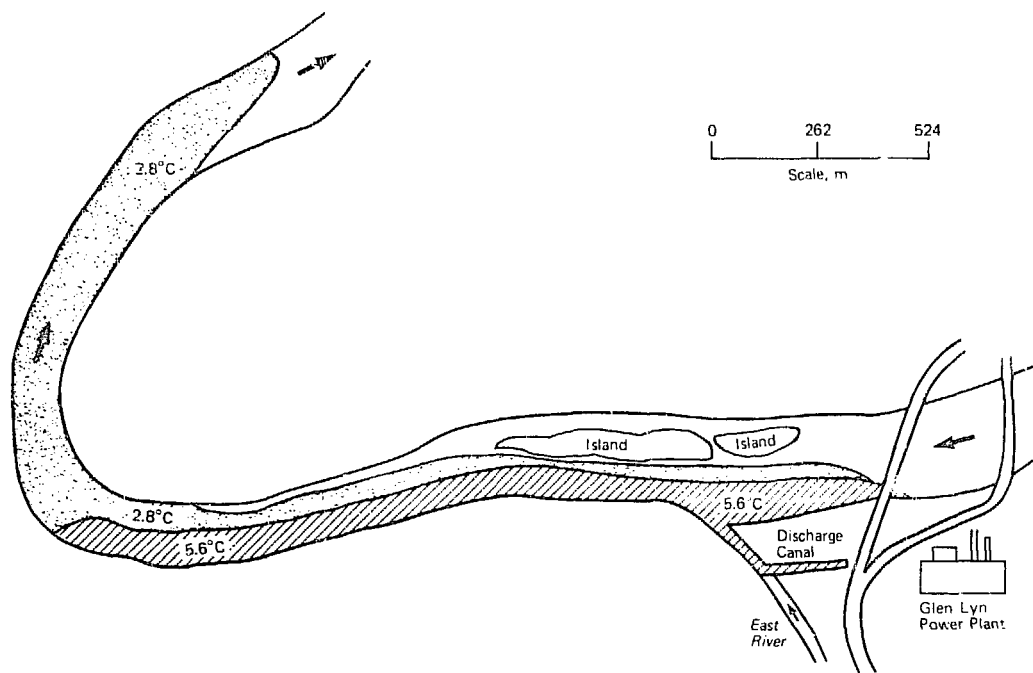


FIG. 1 Estimates of the 5.6 and 2.8°C isotherms in New River as a result of the heated discharge from Glen Lyn Power Plant under conditions of low flow (36.8 to 42.5 m<sup>3</sup>/sec) and high ambient river temperatures.

The condition of several species of fish (channel catfish, spotfin shiner, flathead catfish, etc.) had been shown to be poorer in the heated discharge than in ambient temperature areas of New River. The heated discharge from the Glen Lyn plant did not create a thermal barrier that prevented the spawning migration of the two migratory species present, the white sucker and the northern hog sucker. On the basis of available data, Dickson et al. (1975) concluded that a complete shutdown of the Glen Lyn Power Plant during winter conditions is not likely to cause cold-shock mortality.

The macroinvertebrate community in the New River at Glen Lyn was diverse and abundant and was dominated by mayflies and caddis flies. Immediately below the heated discharges, the macroinvertebrate community was less diverse and altered in composition. Species known to be sensitive to pollutional stress were present; however, this indicates only minor stress on the community. Recovery downstream was rapid, and changes in the macroinvertebrate community were limited to the area within the 5.6°C isotherm (i.e., an area of ~2 ha at low-flow conditions).

While specific entrainment studies on drifting macroinvertebrates were not conducted at Glen Lyn, laboratory studies on specimens from the New River indicated that most species should be able to tolerate temperatures present in the plume (i.e., a maximum of 33°C under low-flow high-temperature conditions). These data, along with data available on the fish, macroinvertebrate, and

plant communities, indicate that entrainment at Glen Lyn does not significantly alter the ecological balance in the river.

Algae and macrophyte communities were shown to be affected by the heated discharges. Although changes in diversity and dominance (green to blue-green algae) occurred in the heated areas, the area in which the changes occurred was limited (i.e., <2 ha).

Dickson et al. (1975) concluded that the heated discharges from Glen Lyn were having a demonstrable impact on aquatic life immediately below the power plant. It was their professional judgment, however, that no appreciable harm occurred to the populations of shellfish, fishes, and wildlife in the New River and that operation of the Glen Lyn plant had not significantly disturbed the indigenous populations in that reach of the river.

It is not within the scope of this paper to include all the data supporting these conclusions. Readers interested in a more detailed analysis of the field studies should refer to the primary sources cited or to Dickson et al. (1975).

### Determination of Thermal Regime

To apply the EPA's draft water-temperature criteria, we must determine the water-temperature regime present. We assumed that determination of the water-temperature regime should be based on a historical record of ambient temperatures and heated-discharge temperatures. At Glen Lyn Power Plant, we had information on inlet and outlet temperatures every 2 hr for 4 years of operation (1970 to

1974). In a type 2 demonstration, the recommended water-temperature criteria for important representative species are compared to the maximum weekly average temperatures by months. One of the first problems encountered when applying the criteria is how to determine the maximum weekly average temperature. There are at least two ways to obtain this information. First, the maximum weekly average temperature for each week of the 4-year data base could be determined by averaging the means of the 2-hr readings by week. This would give 52 temperature values, with each point representing the mean of the four weekly maximum average temperatures for a specific week in the year [Fig. 2(a)]. A second approach is to select the highest weekly average temperature from the 16 possibilities (based on 4 weeks per month and 4 years of record) and plot this single value by month [Fig. 2(b)]. Both approaches provide a maximum weekly average temperature. Specifically, the first provides a mean of the maximum weekly average temperatures for each week of the year for the historical data base. The second approach simply uses the highest maximum weekly average temperature observed in each month of the year based on the historical data. In this paper we use the second approach because it better represents the temperature extremes and thus provides the greatest margin of safety. Comparing Fig. 2(a) and 2(b) reveals that the temperature regime represented in Fig. 2(b) is slightly higher than that in Fig. 2(a).

### Selecting Important Representative Species

The Environmental Protection Agency (1973) has defined important species as: (1) species with high yield to commercial or sport fisheries, (2) species that are important links in food chains, (3) species having large biomass in the existing ecosystem (if desirable) and (4) rare and endangered species. Ideally, the water-temperature criteria should be applied to important representative organisms from all trophic levels in the aquatic community, but, in reality, data on the thermal responses of aquatic organisms other than fish is so depauperate that it is generally acceptable to apply the criteria to fish species in a type 2 demonstration. Extensive fishery studies conducted at the Glen Lyn site (Stauffer, Dickson, and Cairns, 1974; Stauffer et al., 1975) enabled us to tentatively establish what fish species in the New River qualify as important representative species. Of the 48 species collected by Stauffer and coworkers in a 2-year study of the distribution of fish in relation to the Glen Lyn plant's thermal discharge, none is considered to be rare or endangered (Department of the Interior, 1975). Bryson et al. (1975), evaluating the sport fish of New River in the vicinity of Glen Lyn Power Plant, found that, on the basis of fisherman's catch and preference, the channel catfish was the most important sport fish in the area. No commercial fishery exists in the New River. Such popular sport fish as the smallmouth bass and rock bass are also found in the vicinity of the Glen Lyn plant and represent top-level carnivores. The white sucker, a bottom-feeding omnivore

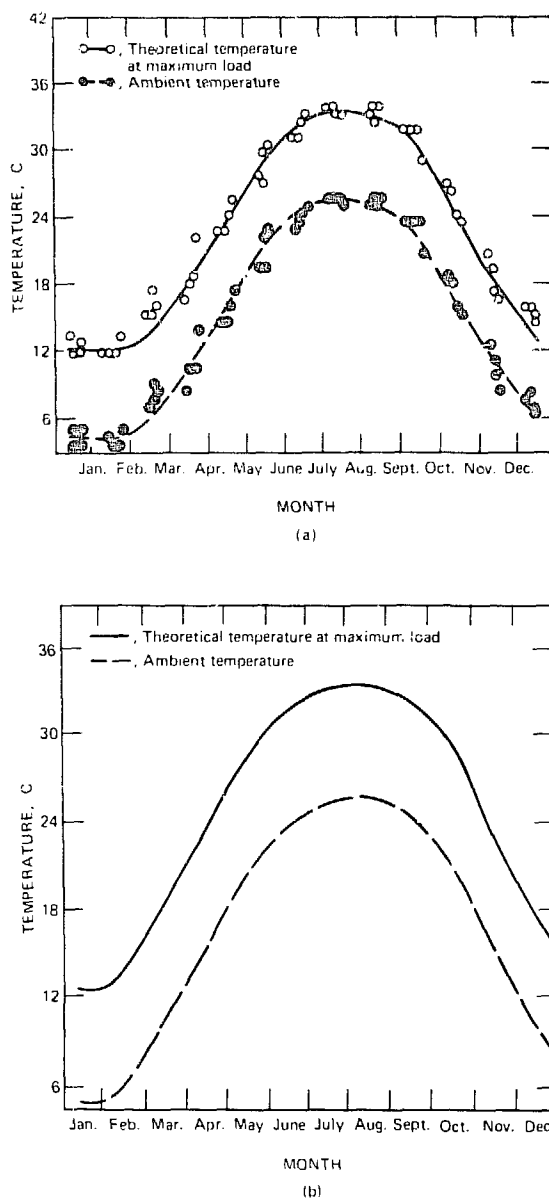


FIG. 2 Ambient New River temperatures and theoretical heated discharge temperatures at maximum power-plant load at Glen Lyn, Virginia, 1970-1974. (a) Maximum weekly average temperature for each week. (b) Maximum weekly average temperature for each month.

that is important in the food chain, was found to be abundant in the New River (Stauffer, 1975). Stauffer also found three species of forage fish (spotfin shiner, rosyface shiner, and stone roller) to be very abundant in the New River near Glen Lyn. These sport fish, omnivore, and forage species comprised the seven important representative fish species for the Glen Lyn site.

# Impact of Heated Discharges on Important Representative Fish Species Using the Draft Water-Temperature Criteria

In the following discussions in addition to the draft water-temperature criteria, we have also used several additional temperature parameters to evaluate the impact of the heated discharge from Glen Lyn Power Plant on important representative fish species (Table 1). Specifically, we used temperature preference and avoidance data recommended by Coutant (1974) and Stauffer et al. (1975).

Channel Catfish (*Ictalurus punctatus*). Relationships of the temperature criteria of channel catfish to ambient river and maximum weekly average temperatures at Glen Lyn Power Plant are shown in Fig. 3. The short-term maximum temperature for survival during summer months ( $T_7$ , 36.0°C) is not exceeded by maximum discharge temperatures; therefore channel catfish could survive in all regions of Glen Lyn plant effluent plume for at least short periods of time. During the summer the heated discharge does not exceed the optimal temperature for growth ( $T_3$ , 32.0°C) nor the upper allowable limits for growth ( $T_1$ , 32.4°C), which is defined as one-third the range between optimum temperature for growth and the ultimate upper incipient-lethal temperature. Thus during warm weather sufficient

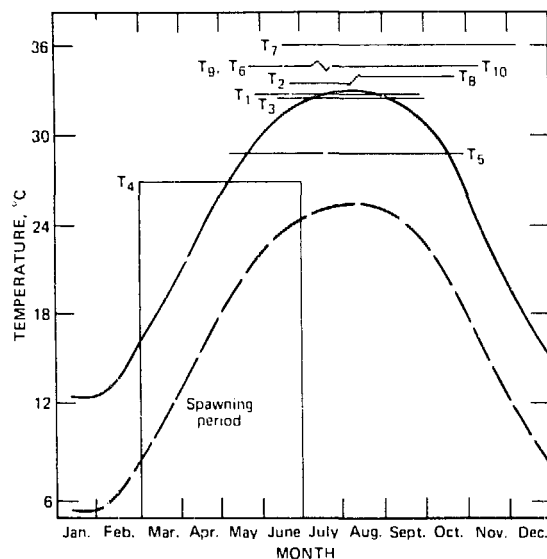


FIG. 3 Relationship of temperature criteria for channel catfish to weekly maximum discharge and ambient river temperatures at Glen Lyn Power Plant. —, theoretical temperature at maximum load. ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

TABLE 1 Critical Temperature Criteria ( $T_1$  to  $T_{10}$ ) for Important, Representative Species Sampled in New River, Glen Lyn, Virginia\*

| Species         | Temperature criteria† |                   |                        |                   |                   |       |                   |             |           |                |
|-----------------|-----------------------|-------------------|------------------------|-------------------|-------------------|-------|-------------------|-------------|-----------|----------------|
|                 | $T_1$                 | $T_2$             | $T_3$                  | $T_4$             | $T_5$             | $T_6$ | $T_7$             | $T_8^{f,j}$ | $T_9^c$   | $T_{10}^{d,j}$ |
| Stone roller    |                       | 32.0 <sup>a</sup> |                        |                   |                   | 35.0  |                   | 24.1, 26.8  | 22.8–23.9 | 33.0           |
| Rosyface shiner |                       | 33.0 <sup>a</sup> |                        |                   |                   | 35.0  |                   | 26.2, 28.8  | 28.9–30.0 | 33.0           |
| Spotfin shiner  |                       | 36.0 <sup>a</sup> |                        |                   |                   | 35.0  |                   | 29.8        |           | 36.0           |
| White sucker    |                       |                   |                        | 10.0 <sup>g</sup> | 17.8 <sup>g</sup> | 30.6  |                   |             | 14.1–20.4 |                |
| Channel catfish | 32.4                  | 33.3 <sup>b</sup> | 32.0 <sup>c</sup>      | 27.0 <sup>g</sup> | 28.9 <sup>g</sup> | 35.0  | 36.9 <sup>g</sup> | 33.8        | 33.9–35.0 | 35.0           |
| Rock bass       |                       | 36.0 <sup>a</sup> |                        |                   |                   | 35.0  |                   | 28.9, 30.2  | 30.0      | 35.8           |
| Smallmouth bass | 30.0                  | 35.0 <sup>a</sup> | 25.5–28.9 <sup>d</sup> | 23.9 <sup>h</sup> | 26.1 <sup>i</sup> | 35.0  |                   | 28.3, 30.1  |           | 33.0           |

\*References are: <sup>a</sup>Cairns et al., 1975; <sup>b</sup>Hart, 1952; <sup>c</sup>Kümbri, Noble, and Hoffman, 1970; <sup>d</sup>Horning and Pearson, 1973; <sup>e</sup>Stauffer, 1975; <sup>f</sup>Stauffer et al., 1975; <sup>g</sup>Brungs, 1974; <sup>h</sup>Henderson and Foster 1956; <sup>i</sup>Webster 1945; <sup>j</sup>Cherry, Dickson, and Cairns, 1975a; 1975b; Cherry et al., 1975.

†Symbols are:

- $T_1$ , one-third the range between optimum temperature for growth and ultimate upper incipient-lethal temperature.
- $T_2$ , ultimate upper incipient-lethal temperature.
- $T_3$ , optimum temperature for growth.
- $T_4$ , maximum weekly average temperature for spawning and egg incubation.
- $T_5$ , short-term maximum temperature for spawning and egg incubation.
- $T_6$ , highest temperature at which the species was recorded in situ.
- $T_7$ , short-term maximum temperature for survival during the summer.
- $T_8$ , final laboratory temperature preference.
- $T_9$ , final field temperature preference.
- $T_{10}$ , laboratory avoidance temperature when acclimated at 30°C (86°F) ambient temperature.

growth would not occur in fish in that portion of the plume more than 5.6°C above ambient (Fig. 1). This represents a minor section of the river (i.e., 2 ha) at low-flow conditions.

Channel catfish usually spawn between April and July if water temperature is near 23.9°C. Based on the maximum weekly average temperature regime at Glen Lyn (Fig. 3), spawning could occur in mid-May and continue through June. With increasing ambient temperatures from mid-May through June, the temperatures in the heated effluent exceeded the maximum weekly average temperature for spawning and egg incubation ( $T_4$ , 27.0°C). Thus, if a specimen spawned in mid-May to the end of May, conditions would be unfavorable within the 5.6°C isotherm (i.e., ~2 ha), and, if a specimen spawned near the end of June, success would be inhibited within the 2.8°C isotherm (i.e., ~6 ha). The short-term maximum temperature for spawning ( $T_5$ , 28.9°C) was exceeded in the thermal plume only within the 5.6°C isotherm; thus conditions preventing the survival of embryos (temperatures greater than 28.9°C) would be limited to an area of ~2 ha during the spawning season. We must emphasize that the isotherms and areas cited were estimated during low-flow (36.8 m<sup>3</sup>/sec) conditions. Flow rates during the early spawning season would be higher (113.3 to 226.6 m<sup>3</sup>/sec). Thus the heat would be more rapidly dissipated and the areas of the 5.6 and 2.8°C isotherms would be smaller than those estimated.

Other criteria recorded at Glen Lyn for the channel catfish show that the final field and laboratory preferences ( $T_8$ , 33.8°C;  $T_9$ , 33.9 to 35.0°C), the laboratory avoidance temperature when acclimated at 30°C ( $T_{10}$ , 35.0°C), and the highest temperature at which channel catfish were collected at Glen Lyn ( $T_6$ , 35.0°C) were 2.0 to 3.0°C above the theoretical maximum weekly average discharge temperatures. Hence, channel catfish should be preferentially attracted to the heated discharge and should not avoid any areas in the plume, even during periods of maximum temperatures.

In summary, maximum discharge temperatures should not inhibit survival of adult channel catfish in the New River at Glen Lyn. During the summer months, growth may be inhibited in a small area of the thermal plume, and spawning will be inhibited in a small area below the power plant. It is unlikely, however, that the loss of ~2 ha of spawning habitat would be detrimental to the channel catfish population at Glen Lyn since large amounts of similar undisturbed habitat are available above, adjacent to, and below the power plant. In addition, Stauffer (1975), in his comprehensive field study, found an abundance of channel catfish in the immediate area of the power plant.

**Smallmouth Bass (*Micropterus dolomieu*).** Temperature criteria of smallmouth bass in relation to maximum weekly average ambient and discharge temperatures at the Glen Lyn Power Plant are shown in Fig. 4. The maximum weekly average temperature in the discharge during full plant load exceeded one-third of the range ( $T_1$ , 30.0°C) between the

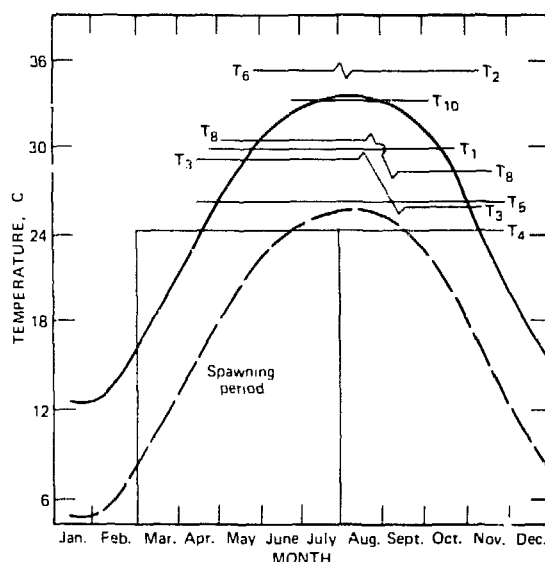


FIG. 4 Relationship of temperature criteria for smallmouth bass to weekly maximum discharge and ambient river temperatures at Glen Lyn Power Plant. —, theoretical temperature at maximum load. ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

optimum temperature for growth ( $T_3$ , 25.5 to 28.9°C) and the ultimate upper incipient-lethal temperature ( $T_2$ , 35.0°C). Thus, during the summer months, the upper allowable limits for growth of smallmouth bass are exceeded in the area of the thermal plume where temperatures are 2.8°C above ambient (i.e., ~6 ha). This area also corresponds to the region where temperatures in the summer months would exceed the optimum temperature for growth ( $T_3$ , 25.5 to 28.9°C). Smallmouth bass spawn between March and July, most commonly at water temperatures from 15 to 18.3°C (Hubbs and Bailey, 1938). On the basis of the maximum weekly average temperature regime at Glen Lyn [Fig. 2(b)], spawning would start in mid-April and continue to mid-June. Spawning could occur anywhere in the system for the first 6 weeks of the spawning season (March to mid-April); after that, the 7.8 ΔT caused by the heated discharge would exceed the maximum weekly average temperature for spawning and egg incubation ( $T_4$ , 23.9°C). From mid-April to mid-May, the smallmouth bass would be excluded from spawning in the area bounded by the 5.6°C isotherm (i.e., ~2 ha) and by June would be excluded in an area of ~6 ha below the plant. From late June and thereafter, the bass would spawn only in deep pools or in cool tributaries since the ambient river temperature at that time exceeds  $T_4$  (23.9°C). The thermal plume exceeded the short-term maximum temperature for spawning ( $T_5$ , 26.1°C) in approximately the same areas in which successful spawning was prevented (i.e., in most of the area within the 2.8°C isotherm, ~6 ha). Estimates of the area precluding some reproductive func-



tion are probably high since they are based on isotherms generated under low-flow conditions (i.e., 36.8 m<sup>3</sup>/sec). Historically flows during the spawning season for the New River have been much larger.

Final laboratory preferences ( $T_8$ , 28.3 to 30.1°C) exceeded all ambient temperature conditions but were approximately 2.8 to 5.4°C below the maximum weekly average discharge temperatures. Laboratory avoidance data showed that exclusion from the plume would not be expected until temperatures exceeded 33.0°C ( $T_{10}$ ). These data suggest that the smallmouth bass would not be in the hottest part of the discharge from July to September. Stauffer (1975) collected smallmouth bass at temperatures of 35.0°C ( $T_6$ ), however. This suggests that the smallmouth bass generally avoid the hottest parts of the plume but will venture into these areas for at least short periods of time.

Overall, the heated discharge at Glen Lyn should eliminate a small area (2 to 6 ha) for reproductive functions. The loss of this area does not appear to adversely influence the population in New River in the vicinity of Glen Lyn, however. Bass were frequently seined in the shallows near the plant (Cherry, unpublished data) and were shown to be present in the plume and above and below the power plant by Stauffer (1975).

**White Sucker (*Catostomus commersoni*).** Temperature criteria of white suckers in relation to maximum weekly average ambient and discharge temperatures at Glen Lyn Power Plant are illustrated in Fig. 5. Data are not available on the short-term maximum temperature for survival of this species, but Hart (1947) found a 24-hr  $LT_{50}$  for specimens acclimated at 25.0°C to be 29.2°C. The maximum weekly

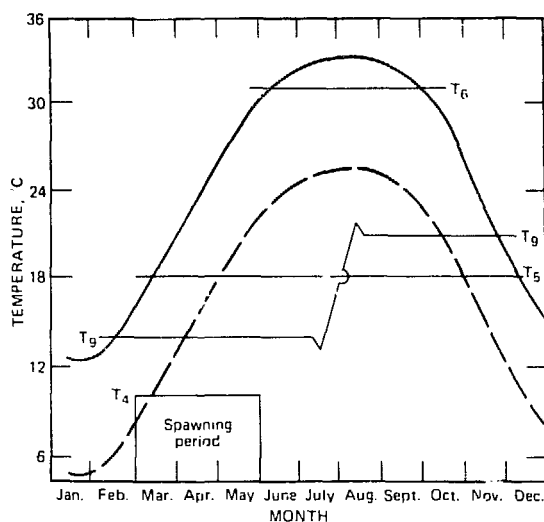


FIG. 5 Relationship of temperature criteria for white sucker to weekly maximum discharge and ambient river temperatures at Glen Lyn Power Plant. —, theoretical temperature at maximum load. ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

average temperature for growth for the white sucker (27.8°C) is exceeded in the thermal plume from May to mid-October (Brungs, 1974). Therefore conditions conducive to sufficient growth for this species are presumably precluded in an area of ~6 ha below the power plant. White suckers generally spawn in shallow areas in cooler tributary streams. Ambient New River temperature conditions generally exceed the maximum weekly average temperature for spawning ( $T_4$ , 10.0°C) during the March to June spawning season (Trautman, 1957). White suckers have been observed spawning in the tributaries near the Glen Lyn plant (Stauffer, unpublished data). The short-term maximum temperature for spawning ( $T_5$ , 17.4°C) is exceeded by ambient river temperatures for half the spawning season (i.e., May to June). The thermal discharge from Glen Lyn Power Plant probably does not entirely inhibit white sucker spawning since Stauffer (1975) observed spawning in the discharge canal at the plant.

Field preference temperatures ( $T_9$ , 14.1 to 20.4°C) were exceeded by ambient river temperatures from April to November. During the winter months (December to March), the white sucker should be able to inhabit areas in the thermal plume. In summary, the major impact of the heated discharge on the white sucker would be to create conditions that impair the growth of the fish below the plant from May to November. Impact of the heated discharges on reproductive functions is probably negligible.

**Rock bass (*Ambloplites rupestris*).** The temperature requirements of the rock bass in relation to maximum weekly average ambient and discharge temperatures at Glen Lyn Power Plant are presented in Fig. 6. Data on short-term maximum temperature and temperature requirements for growth and reproduction of this species are not available. The final laboratory and field preference temperatures ( $T_8$ , 28.9, 30.2°C, and  $T_9$ , 30.0°C) are higher than the maximum ambient temperature. All other temperature criteria [highest temperature at which rock bass were collected ( $T_6$ , 35.0°C), upper laboratory avoidance temperature ( $T_{10}$ , 35.8°C) and ultimate upper incipient-lethal temperature ( $T_{12}$ , 36.0°C)] were higher than the maximum discharge temperature.

On the basis of these data, the effects of the heated discharge at Glen Lyn Power Plant on the rock bass population should be negligible. The impact of the heated discharge on growth and reproduction is unknown, but rock bass have been observed to spawn in shallow weed beds in unheated areas immediately upstream from the plant. Fry and juveniles are also abundant in this habitat (Cherry, unpublished data).

**Stone roller (*Campestris anomalum*).** Temperature requirements of stonerollers in relation to maximum weekly average ambient and discharge temperatures at the Glen Lyn Power Plant are shown in Fig. 7. Data were not available on the short-term maximum temperature for

survival of the species, the maximum weekly average temperature for growth, or the short-term maximum temperature for spawning. The spawning season for the stone roller is short (occurring from mid-April to June), with an exceptionally high maximum spawning temperature of 27.0°C (Smith, 1935, cited in Carlander, 1969). The discharge temperatures exceed the 27.0°C level during half the spawning season. Overall, the stone roller was able to spawn in all areas of the plume during the first half of the season and was prevented from spawning successfully in the 5.6°C isotherm area (~2 ha) during the latter part of May. The loss of this area did not appear to be detrimental to the stone roller population, however, since this species is highly abundant in the river (Stauffer, 1975).

The stone roller will interact in temperatures that are near or just above the maximum ambient conditions; field and laboratory preferences ranged from 22.8 to 26.8°C. During July and August the discharge temperatures exceeded the ultimate upper incipient-lethal temperature for this species by 1.0°C ( $T_2$ , 32.0°C; maximum weekly average discharge temperature, 33.0°C). The stone roller was collected by Stauffer at temperatures as high as 35.0°C ( $T_6$ ).

In summary, the heated discharge did not prevent the stone roller from using the areas influenced by the plume. Reproduction probably decreased in an area of ~2 ha below the plant during the last half of the spawning season, however. The overall impact of the heated discharge on the

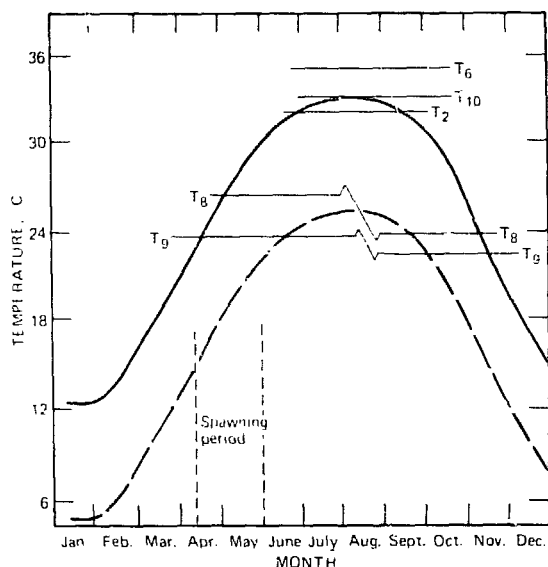


FIG. 7 Relationship of temperature criteria for stone rollers to weekly maximum discharge and ambient river temperatures at Glen Lyn Power Plant. —, theoretical temperature at maximum load. ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

stone roller population appeared to be negligible; this species was abundant in the New River at Glen Lyn during the period covered by our study.

Spotfin Shiner (*Notropis spilopterus*). Temperature requirements of spotfin shiners in relation to maximum weekly average ambient and discharge temperatures at Glen Lyn Power Plant are illustrated in Fig. 8. Unfortunately, data were not available to evaluate the effects of the heated discharge on the growth, survival, and reproduction of the spotfin shiner exposed to short-term maximum temperatures. However, the available data on this species indicate that its distribution was not restricted by the heated discharge. The spotfin shiner preferred water temperatures higher than ambient ( $T_8$ , 29.8°C). No formal field preference was assigned to this species since Stauffer (1975) found that it was abundant at all temperatures at Glen Lyn, even when the plume temperatures were maximal ( $T_6$ , 35.0°C). The ultimate upper incipient-lethal temperature ( $T_2$ , 36.0°C) and the upper laboratory avoidance temperature ( $T_{10}$ , 36.0°C) exceeded the maximum temperature present at Glen Lyn. Thus the available information suggests that there was no impact of the heated discharge on the spotfin shiner. The species response was highly eurythermal, and Stauffer (1975) has shown that numerically it is the most successful fish species in the New River in the vicinity of Glen Lyn.

Rosyface Shiner (*Notropis rubellus*). Figure 9 shows the temperature requirements of the rosyface shiner in relation

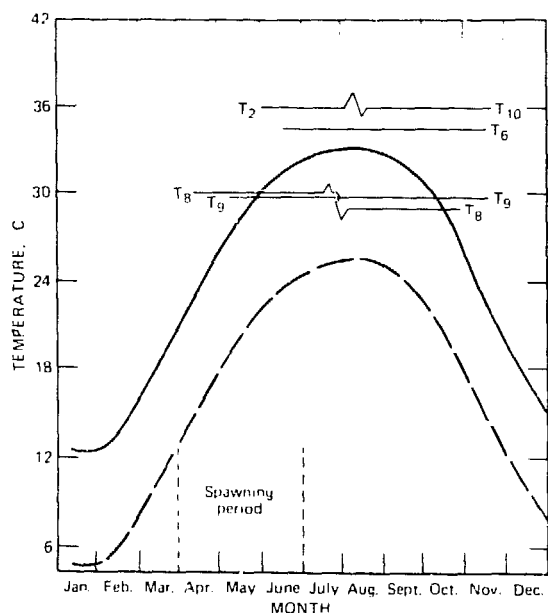


FIG. 6 Relationship of temperature criteria for rock bass to weekly maximum discharge and ambient river temperature at Glen Lyn Power Plant. —, theoretical temperature at maximum load. ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

to the maximum weekly average ambient and discharge temperatures at Glen Lyn Power Plant.

Data on the short-term maximum temperature for survival, maximum weekly average temperature for growth, and the short-term maximum temperature for spawning are not available for this species. Near the end of the spawning period (May to June), the maximum discharge temperature exceeded the maximum spawning temperature (28.9°C) by ~2.8°C (Pfeiffer, 1955, cited in Miller, 1964). Therefore, spawning activities would probably be excluded from the area bounded by the 5.6°C isotherm (i.e., ~2 ha).

Laboratory and field preference temperatures were greater than the maximum discharge temperatures by at least 2.8°C. Although we would not expect the rosyface shiner to be present continuously in the hottest parts of the plume, it could well be present for short periods of time since the ultimate upper incipient-lethal temperature ( $T_2$ , 33.0°C) and the upper laboratory avoidance temperature ( $T_{10}$ , 33.0°C) equalled the maximum weekly average discharge temperature. This tentative conclusion was supported by field collections of the rosyface shiner in situ at 35.0°C ( $T_6$ ).

On the basis of these data, the rosyface shiner does not appear to be excluded from the area below the plant because of the heated discharge. A small area (~2 ha) would not be conducive to successful reproduction of this species, however. On-site studies indicate that the rosyface shiner population at Glen Lyn was abundant despite the

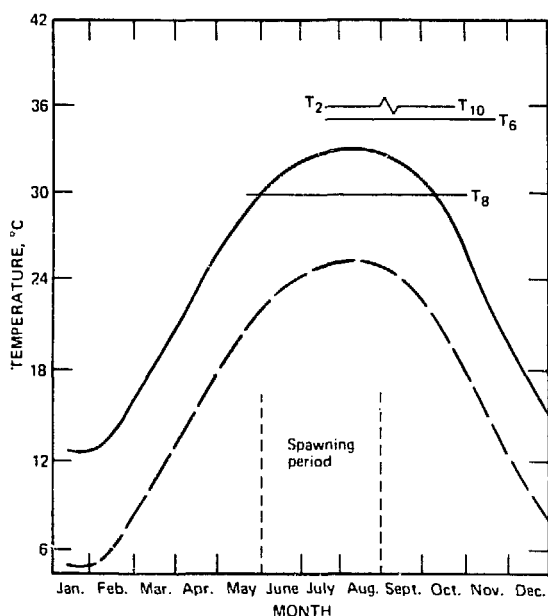


FIG. 8 Relationship of temperature criteria for spotfin shiner to weekly maximum discharge and ambient river temperature at Glen Lyn Power Plant. —, theoretical temperature at maximum load, ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

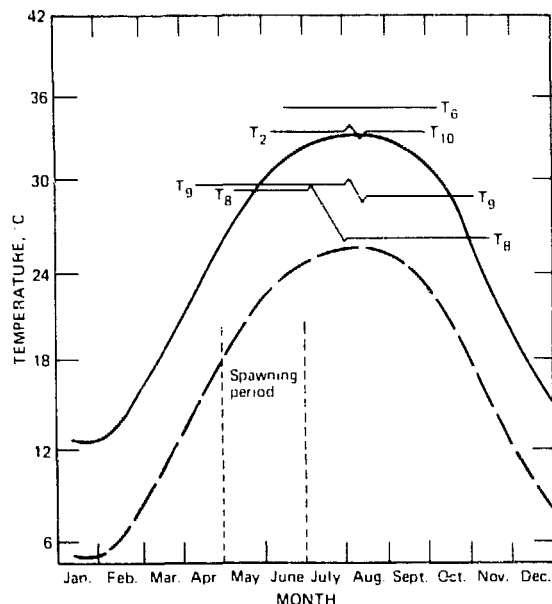


FIG. 9 Relationship of temperature criteria for rosyface shiner to weekly maximum discharge and ambient river temperatures at Glen Lyn Power Plant. —, theoretical temperature at maximum load, ---, ambient temperature. (See Table 1 for explanation of criteria symbols.)

loss of habitat in the plume. This abundance was probably caused by availability of adequate habitat in other parts of the river near Glen Lyn.

#### APPLICABILITY OF EPA'S DRAFT WATER-TEMPERATURE CRITERIA TO THE GLEN LYN PLANT

To effectively apply EPA's draft water-temperature criteria to important representative species, first, we must have a reasonable basis for selecting important representative species, and, second, we must have information on the thermal responses of those species. At Glen Lyn Power Plant, sufficient field studies had been conducted to identify the important fish species. As shown in the preceding section (Table 1), however, data pertaining to some of EPA's draft water-temperature criteria were not available for many of the species. Thus, using only these criteria, we had difficulty effectively determining if the heated effluents from the plant were significantly disturbing the populations of these species in New River.

If the preference, avoidance, and field-observation parameters were removed from Table 1, it is highly unlikely that an evaluation of impact could be made for any species other than channel catfish and smallmouth bass. Data on growth, successful spawning activity, and short-term maxima were not available for the stone roller, rosyface shiner, white sucker, spotfin shiner, and rock bass. Available evidence, however, indicated that these five species were

extremely important components of the fish community. Substituting other species for which data on the EPA's draft water-temperature criteria were available would not be acceptable since the objective of the Section 316(a) process is to provide a relevant site-specific evaluation.

If laboratory preferences and avoidances and field data for fish were not available, using the remaining parameters in Table 1 (i.e., T<sub>1</sub>, T<sub>2</sub>, T<sub>3</sub>, T<sub>4</sub>, T<sub>5</sub>, and T<sub>7</sub>) and knowledge of the distribution of the thermal plume (Fig. 1), we could predict that the growth and reproductive success of the channel catfish and smallmouth bass would be inhibited in an area of ~2 ha in the New River. Water temperatures in the discharge and in the river itself would prevent successful spawning of the white sucker. From these data, we could also predict that the channel catfish and smallmouth bass would be found in the heated areas most of the time. Conclusions about the impact of the heated plume on the other important fish species would not be possible. It seems highly unlikely that any conclusive decision about the impact of the heated effluents could be reached using only these data. To make effective decisions on the significance of the impact using the draft EPA criteria, we need information on each of the criteria, as well as on the areal extent of impact and the availability of spawning areas in the immediate vicinity of the plant. If this information is available, then the criteria should allow for a reasonable assessment of impact. When we apply EPA's draft water-temperature criteria to the Glen Lyn Power Plant, however, their applicability is limited.

It is only through using combined data on the draft EPA water-temperature criteria, fish preference and avoidance temperatures, field studies at the site, and the thermal plume distribution that we can make a reasonable and realistic assessment. Using only the draft EPA water-temperature criteria will generally make it possible to show a demonstrable effect in most aquatic systems, but additional site-specific studies are needed to determine whether these effects are significantly affecting the aquatic life in the system.

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# The Connecticut River Ecological Study: A Synopsis

■ Daniel Merriman

## ABSTRACT

This paper recounts some salient features of an extensive study of the thermal effects of the Connecticut Yankee Atomic Power Company's electric generating plant on biota of the lower Connecticut River. The work includes a description of the plume, an examination of the anadromous shad population, a discussion of the affected ichthyofauna and entrainment, and an account of alterations in benthic fauna. This study has several distinctive attributes, among them that it was begun before the Water Quality Act (1965) and that it had a long-term before-and-after character, beginning in 1965 before the plant began operating and continuing during operation (1968-1973). Ecological alterations observed to date appear to be well within the limits of acceptability, and in large measure, wrought by mechanical rather than thermal factors.

In contradistinction to its name, the Connecticut River is in geographical fact shared by four of the six New England states. From its source at the Canadian border and for nearly two-thirds of its length, it forms the boundary between New Hampshire and Vermont; then it flows through the middle of Massachusetts and Connecticut to empty into the eastern end of Long Island Sound. In its 660-km course, it drains a valley of some 29,000 km<sup>2</sup> and, with the exception of the Susquehanna, carries the largest volume of any river on the east coast of the United States north of Georgia. The rainfall in this region is relatively evenly distributed throughout the year, and the average daily rate of flow of the Connecticut over the past half century has been about 450 m<sup>3</sup>/sec (although it has varied over a wide range during this span of time).

There are 17 dams along the length of the river, the southernmost being at Enfield, near the Connecticut-Massachusetts border, 110 km upstream from the river mouth. The last 65 km of the river, from Hartford, Conn., to the river mouth, is subject to a heavy volume of commercial shipping - mainly oil barges and tankers. It is with this 65-km section that the Connecticut River Ecological Study was chiefly concerned from 1965 to 1973.

The Connecticut Yankee Atomic Power Company's electric generating plant is situated on the east shore of the

river at Haddam Neck, 26 km from Long Island Sound and 39 km below Hartford (Fig. 1). It is important to note that the navigation channel, which is 5 to 7 m deep, hugs the west shore opposite the plant in this region where the river is 500 to 600 m wide. It is also important to note that this region of the river is tidal—not in the sense that it is in any way saline or brackish, but rather that there is a rise and fall of approximately 0.75 m. The minimum daily average tidal flow provides an additional 425 m<sup>3</sup>/sec over and above the normal current.

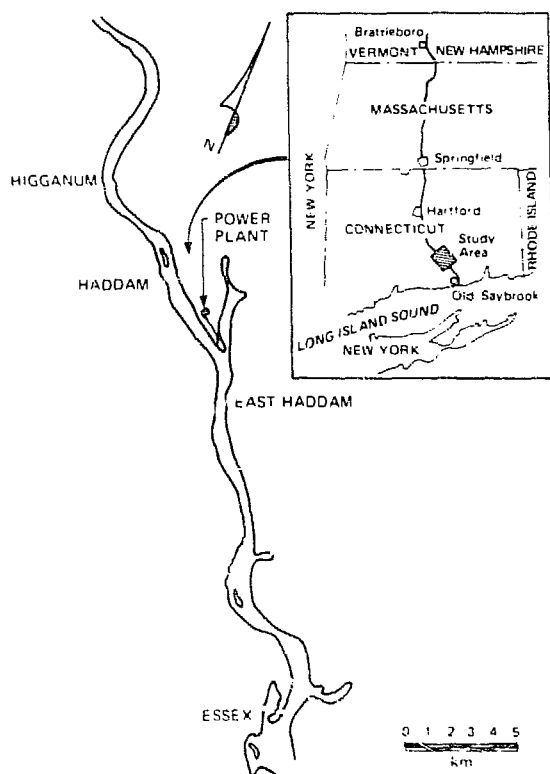


FIG. 1 The lower Connecticut River and its course through the study area in the general vicinity of the Connecticut Yankee Atomic Power Company plant.

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The nuclear unit of Connecticut Yankee is a pressurized-water reactor with a once-through condenser cooling system. The plant began commercial operation in January 1968, at which time it was licensed to operate at 490 Mw(e) gross electrical power. Its design rating was 600 Mw(e), and, since the first refueling in April-May 1969, it has generally maintained this output, with exceptions for repairs and refueling. Since the plant first went into operation, it has generated electricity 73% of the time and has produced more electricity than any other commercial nuclear generating unit in the world.

The river water enters the cooling system at a velocity of 0.3 to 0.6 m/sec and a volume rate of 23.5 m<sup>3</sup>/sec. Passage through the plant requires about 1.5 min, during which time the temperature of the cooling water is raised ~12.5°C at full-power operation. This heated water is discharged over a substantial weir into a 1.6-km canal that enters the river downstream via a wide fan-shaped mouth. During its passage down the canal, the temperature of the thermal effluent drops about 1°C. The canal was constructed to prevent recirculation of heated water through the plant during flood tide.

## THE STUDY

Construction of the Connecticut Yankee plant began in 1963-1964. In considering the application for a dredging permit, the State of Connecticut Water Resources Commission,\* manifested concern as to the effect of plant operation on the biota of the river. The Commission granted the permit, Oct. 21, 1964, with the proviso that an exhaustive study "of the river conditions and environment" begin as soon as practicable and continue for 5 years after the plant went into operation. Two points need mention here: first, the prescribed proviso antedated by several years the national requirements for environmental impact studies, and, second, the company was required to bear the cost of the study.

I was given the opportunity to direct the Connecticut River Ecological Study (CRES), with Lyle M. Thorpe, former director of the Connecticut State Board of Fisheries and Game, as associate.† By the spring of 1965 CRES was underway, and for the most part the key personnel of the staff remained with the study until it was completed in 1973. The final report (Merriman and Thorpe, 1976) is to be published as a monograph of the American Fisheries Society. In addition to the preface, acknowledgements, summary, and conclusions, this report contains 11 separate contributions by eight investigators—the principal sections being those on hydrography, by William A. Boyd; benthos, by Ronald R. Massengill; ichthyofauna, by Barton C. Marcy, Jr.; and American shad, by William C. Leggett.

\*Incorporated into the state Department of Environmental Protection on Oct. 1, 1971.

†Since its inception, CRES has also had the close counsel of a five-man scientific advisory committee, all of whose members were out-of-state and non-utility affiliated.

This paper points out some highlights of CRES as outlined in the summary of the final report. We were fortunate to have 3 years in which to collect basic data before the plant went into operation. As it happened, the 3 years were quite different meteorologically—1965 was the last year of a drought period, 1966 was more normal, and 1967 was unusually wet. There followed a 6-year period of study with the plant in general operation. We feel that during our 9-year investigation we have gone far beyond the "inventory and survey" stage. While we do not claim to have provided a full understanding of all the ramifications of the ecosystem, the interaction of its various elements, or latent synergistic effects, we hope it may be fairly said that, as environmental biologists, we have produced definitive information along the lines ordered by the Connecticut Water Resources Commission (i.e., "to collect relevant data and formulate sound conclusions concerning any changing conditions and their effect"). At the very least, the fundamentals needed for future observation, comparison, and further elaboration are now available.

## SOME RESULTS

The most obvious and immediate concern voiced by the public in 1964 was the welfare of the American shad (*Alosa sapidissima*). This species has historically supported a valuable commercial and an intensive sport fishery in the Connecticut River. The shad is anadromous. It spawns in our latitude mainly in May and June, mostly well upriver from the Connecticut Yankee site (Marcy, 1972). The juvenile fish make their way downstream to the sea from August through October. The marine part of the shad's life history occupies the next 4 (for most males) to 5 (for most females) years. The fish return to the natal stream on a schedule that is highly temperature dependent (Leggett and Whitney, 1972). The concern was how the heated effluent from Connecticut Yankee might affect the annual run of shad in each of three separate phases of its life history: the upstream migration of spawning adults, the return to the sea of the spent fish, and the emigration of juveniles. Might the heated effluent turn out to be a thermal barrier, in part or in whole?

Comprehensive surveys of the three-dimensional extent of the thermal plume under different tidal and seasonal conditions over the years showed that the portion of Connecticut Yankee's effluent plume that retained a temperature 2°C or more above ambient river temperature was largely confined to depths less than 1.2 m. They also showed that, although the plume did extend across the river at the surface at daily intervals under normal- or low-flow conditions, on the average both the up and downstream surface extent of the plume along the east shore was somewhat less than 1 km. We concluded that there are no temperature distributions of the effluent plume that constitute a thermal barrier to the passage of migratory or resident fish, assuming the fish are able to sense the heated plume and to swim under or around it.

To test this assumption, from 1965 through 1973 we tagged ~34,000 adult shad in the lower 3 km of the river with conventional spaghetti darts. We had an annual tag-return rate of over 20%, and there was no significant change in the distribution of returns above and below the Connecticut Yankee plant in the preoperative (1965 to 1967) and postoperative (1968 to 1973) years. In addition, by tracking over 200 adult shad carrying ultrasonic tags in their stomachs, we determined that, in the main, shad follow the natural and dredged river channel, thus effectively avoiding the thermal plume on both the upstream migration and the postspawning return to the sea. Laboratory and field experiments also showed that the juvenile shad avoid potentially lethal temperatures above 30°C and apparently traverse or avoid the Connecticut Yankee effluent on their downstream passage to Long Island Sound.

The balance of the study of the adult American shad involved population trends in the Connecticut River, not only for the immediate period (1965 to 1973) but also for the preceding three decades. We concluded that the Connecticut Yankee thermal discharge has had no meaningful influence on the size or structure of the annual population of adult shad to date.

The entire fish population of the lower Connecticut River is given detailed consideration in the full report. Suffice it to say here that the life-histories, fluctuating abundance, and movements of the major elements of the fish fauna, about 50 species in the lower reaches, were closely scrutinized. Some of the species we had thought of as resident surprised us by the extent of their movements. For example, tagged catfish released in the Haddam Neck area were returned from just below Enfield Dam, 75 km upstream, and from near the mouth of the Connecticut, 25 km downstream in saline waters. Operation of the Connecticut Yankee plant apparently reduced or displaced about 20% of the population of fishes that we originally found in an area of about 150 ha in the general vicinity of the discharge point from the canal. At the same time there was a notable attraction of fishes into the canal; most abundant were two species of catfish, two different shiners, white perch, and carp. The catfish dominated in the winter months, entering the canal when the water dropped to about 20°C and leaving when the discharge temperatures rose above 35°C. The estimates of the annual populations of catfish in the 1.6-km canal ranged from 17,000 to 40,000. A remarkably successful winter-spring rod-and-reel fishery has developed in the discharge canal of the Connecticut Yankee plant; catfish and white perch are most commonly caught, but northern pike weighing up to at least 8 kg have been taken with reasonable frequency.

With regard to entrainment, our studies from 1970 to 1972 showed that all larvae and early juveniles of nine species of fish passing through the plant in June and July were dead by the time they reached the end of the discharge canal. The percentage of mortality caused by mechanical damage alone was determined by sampling

when all the intake pumps were running at capacity and the rest of the plant was shut down for refueling or repairs [i.e., when no heat or biocide (sodium hypochlorite) was being released (Marcy, 1973)]. It turns out that 80% of the total fish mortality is attributable to mechanical damage, and 20% to heat shock and prolonged exposure in the discharge canal to temperatures above 28°C. Further, there was no measurable additional mortality from the injection of the biocide (with intake pumps running, but no heat released). By contrast, mortalities to cladocerans and cyclopoid copepods (the dominant forms of zooplankton) which were attributable to mechanical damage occurring when they passed through the condenser cooling system were negligible. When temperatures exceeded 31°C, however, mortalities of these organisms were essentially 100%. A conservative estimate of the number of nonscreenable young living fish and zooplankton entrained by the intake averaged about 4% of the population that would be passing by the plant under conditions of unidirectional net tidal flow.

The benthos was affected by the Connecticut Yankee plant in two areas. At the intake, high current velocities completely depopulated invertebrates from the substratum in an area of about 0.5 ha when the plant went into operation. Since the plume does not reach the bottom here, the depopulation is due solely to mechanical or scouring action. Further downstream, in the region of the mouth of the discharge canal, an area of approximately 11 ha, mainly in the shallow water on the east side of the river, comes in direct contact with the plume in varying degree depending on river flow, etc. Here there have been obvious changes in the composition of the benthic fauna. The total decrease in abundance and biomass, as well as the changes in percentage composition of the dominant taxa, have been moderate, however, although scattered beds (20 ha) of *Vallisneria* in this general region have entirely disappeared since 1968. In areas still further downstream, beyond the influence of the plume, the population of benthic invertebrates has increased. The 11-ha area in contact with the plume was originally a sand-flat biotope, but, when the power plant began operation, the area was subject to siltation from the canal and soon became a much finer particle biotope. We suggest that the principal effects of Connecticut Yankee power plant on the benthos were caused primarily by changes in the composition of sediment rather than by direct thermal stress.

## DISCUSSION

In summation, we conclude that the thermal discharge from the Connecticut Yankee plant does not appear to exceed the limits of acceptability as judged by any reasonable means. Such ecological alterations as have been observed seem to be of relatively small moment.

It is of interest too, especially at a symposium on thermal ecology, that the most obvious perturbations resulting from the operation of Connecticut Yankee (apart from those in the man-made canal) involve the benthos in the river and that the changes in the benthic fauna appear



to have been wrought by mechanical rather than thermal factors—i.e., by scouring at the intake and siltation in the area adjacent to the mouth of the canal.

Similarly, the mortality of ichthyoplankton by entrainment at Connecticut Yankee appears, in the main, to be the result of mechanical rather than thermal stress. For both benthos and ichthyoplankton, other sources of mechanical disturbance are commonplace. For example, siltation occurs in nature at times of high flow and from such activities by man as dredging of the channel. The ichthyoplankton must surely be subjected to no small amount of mechanical buffeting from commercial shipping on the lower part of the river (over 3.5 million tons in 1970), to say nothing of that from pleasure boat traffic.

In brief, it is unconscionable any longer to hold that thermal effluents *per se* are universally equivalent to environmental degradation. In the larger frame of reference, it matters little whether we talk about thermal or mechanical effects. The fundamental point, so ably put forward by Hawes, Coughlan, and Spencer (1975) with regard to entrainment, is that, since the entire aquatic ecosystem tends to operate with a naturally occurring high mortality rate, "it should not be assumed that a power station is necessarily creating (a significant) unnatural mortality" of specific organisms. "... too little is understood of natality and mortality to allow prediction of effects of an additional mortality through entrainment." This subject is inordinately complex and quite incapable of simple resolution. What we look for, then, are the effects of the power plant(s) on key elements in the ecosystem. To this end, the Connecticut Department of Environmental Protection is

continuing to run an annual inventory of the shad population. The estimate for 1974 shows a substantial increase over 1973, as in fact did that for 1973 over 1972. Clearly we may infer from all the evidence that in this most important respect the operation of Connecticut Yankee has caused no significant adverse changes to date.

## ACKNOWLEDGMENTS

This paper is derived from the work of the active staff of the Connecticut River Ecological Study. It is a pleasure to record my appreciation to all my associates and to the members of the scientific advisory committee. Each of us owes a debt of gratitude to the Connecticut Yankee Atomic Power Company, not only for financial support but also for its policy of allowing the directors and staff complete freedom in research and in the publication of results.

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# Effects of Entrainment Through Oconee Nuclear Station on Carbon-14 Assimilation Rates of Phytoplankton

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## ABSTRACT

Carbon assimilation rates of phytoplankton communities entrained through Oconee Nuclear Station were measured on six dates during 1974. Thermal, mechanical, condenser, and multiple entrainment effects on uptake rates were compared by incubating samples *in vitro* in controlled-temperature water baths. Duplicate light and dark bottles containing water from four cooling-system locations were exposed to temperatures approximating intake and discharge temperatures. The relationships were variable, but exposure of the hypolimnetic intake water at near-discharge temperatures (thermal effect) stimulated primary productivity in four of six experiments. Multiple entrainment and mechanical effects caused no consistent change in assimilation rates.

Phytoplankton are subjected to two major stresses during entrainment at Oconee Nuclear Station: (1) changes in water temperature and (2) mechanical damage in the piping system. Entrainment is defined here as the passage of organisms through a power-plant condenser cooling-water system (Wolman et al., 1972). Our objectives were to determine phytoplankton carbon assimilation rates in the cooling system of Oconee Nuclear Station and to evaluate the effect of the various entrainment stresses on assimilation rates.

In studies at Marshall Steam Station, Lake Norman, North Carolina, entrainment effects on phytoplankton in hypolimnetic intake water temporarily occurred after passage through the cooling system (Smith et al., 1974). Differences in carbon assimilation rates of intake and discharge phytoplankton exposed to intake temperatures were typically less than 20%. As ambient temperatures approached 25°C, less of a stimulatory effect on intake photosynthetic rates due to thermal elevation was noted.

Studies of primary productivity at power plants located on estuaries also indicate a direct relationship between

ambient water temperature, thermal elevations in cooling water, and carbon assimilation rates. When ambient temperatures were below 22°C, a thermal increase of 6 to 7°C stimulated assimilation rates; but, at higher ambient temperatures, the same  $\Delta t$  depressed carbon uptake rates at the Indian River Power Plant, Delaware (Brooks, 1972). In Virginia, with York River water at ambient temperatures between 0 and 10°C, condenser  $\Delta t$ 's up to 14°C stimulated carbon assimilation rates (Warinner and Brehmer, 1966). At water temperatures between 10 and 15°C, a 3°C  $\Delta t$  stimulated rates, but  $\Delta t$ 's of 5.6 and 14°C reduced rates. All  $\Delta t$ 's above 5.4°C depressed rates at ambient temperatures above 15°C. A Patuxent River Estuary study indicated that, at ambient temperatures of 16°C or below, increased carbon uptake rates were observed in cooling water of Chalk Point Power Plant in the presence of an 8°C  $\Delta t$  (Morgan and Stross, 1969). At a  $\Delta t$  of 8°C, inhibition resulted when the ambient water temperature was greater than 20°C.

## SITE DESCRIPTION

Oconee Nuclear Station uses cooling water from Lake Keowee, a warm monomictic, oligotrophic reservoir located in northwestern South Carolina. A physicochemical description of Lake Keowee has been previously summarized (Duke Power Company, 1974). Lake Keowee, which reached full pond level in 1971, has a shoreline of 483 km and a surface area of 7435 ha. Two dams form the reservoir; the Keowee River Dam located on the upper end of Hartwell Reservoir and the Little River Dam located approximately 6 km south of Oconee Nuclear Station. Condenser cooling water is obtained from the Little River section, where it is drawn through a 7-m-high opening below a 21-m-deep surface skimmer wall. Cooling water passes through a 0.8-km intake canal to the cooling system and discharges to the Keowee River section via a 0.5-km discharge cove. Maximum net electric-power generating

Kreh and Derwort: Duke Power Company, Environmental Sciences Unit, Charlotte, North Carolina

capacity of the three-unit station is 2613 Mw(e). Maximum pumping rates reach 128.3 m<sup>3</sup>/sec through the station, with a maximum condenser  $\Delta t$  of 9.6°C. An Amertap system uses continually recirculating 2.5-cm sponge spheres forced through the stainless-steel condenser tubes to scour any growths that impair heat transfer, thus eliminating the need for biocides. Approximate piping-system residence time of entrained water is 5.4 min when all circulating pumps are operating.

In terms of percent composition by number, the green algae (Chlorophyceae) occurred most abundantly in the euphotic zone (average depth, 6 m) of Lake Keowee (Duke Power Company, 1974). Dinoflagellates (Dinophyceae) and diatoms (Bacillariophyceae) comprised the major percentage of the algal biovolume. Uncorrected euphotic-zone chlorophyll *a* concentrations ranged from 0.59 to 3.66 mg/m<sup>3</sup>. Near-bottom chlorophyll concentrations were consistently lower than euphotic-zone concentrations.

Phytoplankton standing crops and chlorophyll *a* levels in the intake canal and near-bottom water immediately lakeside of the skimmer wall were similar. Green algae and diatoms occurred most frequently in the intake canal, but dinoflagellates and diatoms were volumetrically dominant at most times. In the condenser cooling-water system, chlorophyll *a* levels typically changed very little from intake to discharge. The average was 0.56 mg/m<sup>3</sup>.

## MATERIALS AND METHODS

Primary productivity of entrained phytoplankton was studied using the light and dark bottle <sup>14</sup>C assimilation technique, defined as the rate of organic matter synthesis into plant tissue (Steeman Nielsen, 1952; Gurtz and Weiss, 1972). Experiments were designed to show the immediate effects of cooling-system passage on productivity. Effects due to thermal, mechanical, condenser passage, and multiple entrainment stresses were examined (Table 1).

Eight 125-ml reagent bottles (four light and four dark) were filled with water from each of four locations, i.e., intake, Unit 1 precondenser and postcondenser taps, and discharge. An 8.1-liter VanDorn water bottle was used to collect water at the intake and discharge structures. All samples were transported in a light-tight case until they

were inoculated with 0.1 to 0.15 ml of aqueous NaH<sup>14</sup>C<sub>3</sub> (specific activity, 10  $\mu$ Ci/ml). Field determinations of pH and total alkalinity were also performed.

Two pairs of light and dark bottles from each of the four stations were incubated for 3 to 4 hr in each of two controlled-environment chambers. One chamber was maintained at near-intake temperature and the other at near-discharge temperature. A submersible pump circulated (~12 cm deep) water in each chamber. Water temperature was maintained with a heating element and temperature controller. Artificial light (3050  $\pm$  650 lux at incubation water surface) was provided with seven 20-watt fluorescent lamps.

Incubated water samples were vacuum filtered at 0.5 atm through 0.45- $\mu$ m pore-size membrane filters, followed by a distilled-water rinse. The damp membrane filter was placed in 15 ml of scintillation cocktail contained in a glass vial (Gurtz and Weiss, 1972). A liquid scintillation spectrometer provided counts per minute of the filtered radioactive material for each sample. Counter efficiency was determined for each sample (~70%) which was used to convert counts to disintegrations per minute (DPM) (Pugh, 1970). Carbon assimilation rates (expressed as milligrams of carbon assimilated per cubic meter per hour) were derived using the formula (Saunders et al., 1962)

$$\text{mg C m}^{-3} \text{ hr}^{-1} = \frac{r}{R} \times \frac{C}{T} \times I$$

where  $r$  = (DPM rate of light bottle) - (DPM rate of dark bottle)

$R$  = DPM rate inoculated initially to each bottle

$C$  = total available inorganic carbon ( $\text{CO}_2 + \text{HCO}_3^- - \text{CO}_3^{2-}$ ) in mg/m<sup>3</sup>

$T$  = incubation time in hours

$I$  = isotope discrimination factor (1.064)

The hypothesis that mean assimilation rates in various effect comparisons were significantly different from the control sample was tested by a one-way analysis of variance (two-tailed test). When significant differences existed at the  $\alpha = 0.05$  level, Tukey's multiple-range test was performed on the mean rates (Sokal and Rohlf, 1969).

For each study, the lower and higher mean incubation temperature (MIT) simulated the water temperature at the intake and discharge structure, respectively. No thermal elevation occurred across Unit 1 condensers in December (water was pumped though), but samples were incubated at the elevated discharge temperatures resulting from electric power generation in Unit 2 of Oconee Nuclear Station.

## RESULTS

Entrainment studies were made at Oconee Nuclear Station on six dates in 1974. Intake water temperature ranged from 11.0 to 22.2°C. Water temperatures at the discharge structure ranged from 15.1 to 27.8°C (Table 1). The lower MIT for the six experiments ranged between

TABLE 1 Temperature Summary for Entrainment Productivity Studies at Oconee Nuclear Station

| Month<br>(1974) | Surface-water temperature, °C |         |           | Unit 1<br>$\Delta t$ , °C |
|-----------------|-------------------------------|---------|-----------|---------------------------|
|                 | Control<br>station            | Intake* | Discharge |                           |
| Jan.            | 11.5                          | 11.0    | 16.0      | 7.0                       |
| Mar.            | 12.9                          | 11.6    | 15.1      | 8.9                       |
| June            | 24.5                          | 15.6    | 22.8      | 9.8                       |
| July            | 28.0                          | 18.6    | 27.5      | 10.0                      |
| Sept.           | 24.9                          | 22.2    | 27.8      | 8.9                       |
| Dec.            | 12.8                          | 12.8    | 18.9      | 0                         |

\*Lower than control temperatures in summer months because of hypolimnetic withdrawal.

# Interactive Effects of Temperature, Salinity, and Other Factors on Coastal Copepods

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## ABSTRACT

Studies at the power generating plant at Crystal River, Florida, examined the effects of entrainment on the major species of copepods, including *Acartia tonsa*, *Oithona colcarva*, *Paracalanus crassirostris*, and *Euterpina acutifrons*. Experimental field treatments employing circulating-system drift-bottle devices were set up to test the relationships between entrainment mortality and such variables as temperature, salinity, length of exposure to the heated effluent, mechanical damage, seasonal factors, density (numbers per cubic meter) and sex or age class of the various species. Temperature, salinity, and temperature-salinity interactions were seen to be among the most important factors influencing mortality. Although each species exhibited its own response patterns to temperature-salinity effects, as well as to combinations of the other variables, estuarine forms displayed a higher fitness with respect to the stresses of entrainment. Delayed thermal shock is seen as the most important aspect of entrainment which may be influenced by design criteria.

Planktonic organisms are subjected to many environmental stresses that may be actively avoided by the nekton as well as many benthic animals. It is for this reason that many studies have focused on the resistance of planktonic forms to such factors as thermal stress. Recent studies in this area have shown a trend away from the univariate analysis of temperature tolerance and toward a multiple-factor approach. Under many conditions interactions of temperature with salinity, dissolved oxygen, and duration of exposure are as important in shaping the tolerance pattern of a species as the primary response to temperature. Laboratory studies have been performed in an attempt to define the effects of such interactions on the thermal tolerance levels of planktonic larvae of crustaceans (Costlow, Bookhout, and Monroe, 1960; 1962; 1966), molluscs (Kennedy et al., 1974; Lough, 1975), and fishes (Alderdice, 1963; Alderdice and Forrester, 1968). These

studies used multiple regression analysis and response-surface techniques to define the tolerance patterns of the experimental organisms to various environmental factors. The methodology of this approach was reviewed and the biological applications discussed in detail by Alderdice (1972).

A related area of study, prompted by growing concern over industrial thermal pollution, is that of the effects of zooplankton entrainment into the cooling systems of power-generating plants. There are recent reviews of such entrainment studies by Raney, Menzel, and Weller (1973) and Coutant and Pfuderer (1974) and the proceedings of a symposium on the state of the art edited by Jensen (1974). Entrainment studies usually involve on-site field work since conditions experienced by organisms pumped through a power plant cannot be artificially reproduced under laboratory conditions. As a result of limitations on experimental design, statistical treatment of entrainment mortality is generally simpler and less descriptive than that of laboratory thermal-tolerance studies. Zooplankton mortality is usually related by linear regression analysis to the single independent variable of discharge temperature (Icanberry and Adams, 1974) or to the combination of discharge temperature and thermal rise ( $\Delta T$ ) (Davies and Jensen, 1974). Interactions of environmental factors and entrainment effects of the type described by response-surface methodology, for the most part, have not been looked at by entrainment studies.

We report here the results of a 1-year field study designed to provide data for general linear hypothesis (GLH) analysis of entrainment effects on copepods, the dominant zooplankters of the study area. Considered in the entrainment study were such factors as intake and discharge temperature, salinity, mechanical damage, delayed thermal

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changes were included in both assimilation rates, which may unequally influence photosynthesis in the incubated samples. An alternate design to establish a mechanical effect and discount thermal effects would be to examine intake and discharge water at the lower MIT. Annual mean differences in rates from that design showed only a 2% increase in discharge rates. The ability of phytoplankton to recover from the effects of cooling-system passage is also illustrated by comparing intake and discharge rates at the lower MIT. Mechanical effects must be assumed negligible to obtain an indication of recovery from that comparison. The annual mean difference of +2% suggests that short-term stimulation of primary productivity occurred after passage through Oconee Nuclear Station. Thermal elevations in the cooling system ranged from 0 to 10.0°C, which detracts from the reliability of the 2% stimulation.

Numbers of phytoplankton entrained through Oconee Nuclear Station are only a fraction of the concentrations observed from surface water of Lake Keowee because of the use of lower strata water for cooling. Algae are removed from below the euphotic zone, and the mean carbon uptake rates of this lower strata algae are lower ( $<0.1 \text{ mg C m}^{-3} \text{ hr}^{-1}$ ) than surface rates ( $0.29 \text{ mg C m}^{-3} \text{ hr}^{-1}$ ) on the lakeside of the skimmer wall (Rodgers, 1974). Reduced primary productivity is more likely to occur in the intake rather than in the open-lake surface waters. The actual primary productivity depends on the retention time of water in the intake canal. In the cooler months, when the lake is thermally destratified, entrained phytoplankton are returned to the surface waters at slightly higher than ambient surface temperatures. Increased photosynthesis during cooler months would be expected in the discharge cove if intake phytoplankton are light and/or temperature-limited. Slightly higher annual carbon assimilation rates have been reported in the discharge cove (rates were measured in the surface to 9-m water column) at Oconee Nuclear Station than have been reported immediately lakeside of the skimmer wall (Rodgers, 1974).

Our design to examine sources of potential plant effects on intake productivity is limited to short-term alterations. Residual effects of entrainment require measurements of receiving-water productivity. Damage to intake phytoplankton and changes in primary productivity are probably minimal when we consider: (1) the short residence time in

the cooling system followed by rapid mixing in the open lake and (2) the relatively low productivity of the intake water.

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TABLE 2 Sample Comparison Design to Measure Effects of Entrainment on Carbon Assimilation Rates\*

| Effect                           | Sample   | Incubation temperature | Vs.  | Sample    | Incubation temperature |
|----------------------------------|----------|------------------------|------|-----------|------------------------|
| Thermal                          | Intake   | Lower MIT†             | ↔    | Intake    | Higher MIT             |
| Mechanical                       | Intake   | Higher MIT             | ↔    | Discharge | Higher MIT             |
| (pressure, abrasion, turbulence) | Intake‡  | Lower MIT              | ↔ or | Discharge | Lower MIT              |
| Condenser passage                | Precond. | Lower MIT              | ↔    | Postcond. | Higher MIT             |
| Multiple entrainment factors     | Intake   | Lower MIT              | ↔    | Discharge | Higher MIT             |

\* After Brooks, 1972.

†MIT, mean incubation temperature.

‡Alternate design to examine mechanical effects.

TABLE 3 Mean Carbon Assimilation Rates of Phytoplankton Entrained at Oconee Nuclear Station

| Treatment                   | Assimilation rate, mg C m <sup>-3</sup> hr <sup>-1</sup> (mean ± 1 SE)* |             |             |             |              |             | Annual mean |
|-----------------------------|-------------------------------------------------------------------------|-------------|-------------|-------------|--------------|-------------|-------------|
|                             | 1/22/74                                                                 | 3/20/74     | 6/13/74     | 7/16/74     | 9/11/74      | 12/11/74    |             |
| Intake at lower MIT†        | 0.44 ± 0.08                                                             | 0.64 ± 0.06 | 0.47 ± 0.00 | 0.32 ± 0.20 | -0.15 ± 0.00 | 1.00 ± 0.08 | 0.45        |
| Intake at higher MIT        | 0.50 ± 0.05                                                             | 0.80 ± 0.02 | 0.66 ± 0.06 | 0.24 ± 0.06 | 0.12 ± 0.08  | 0.90 ± 0.04 | 0.54        |
| Precondenser at lower MIT   | 0.71 ± 0.23                                                             | 0.38 ± 0.14 | 0.60 ± 0.02 | 0.40 ± 0.08 | 0.44 ± 0.10  | 0.47 ± 0.05 | 0.50        |
| Postcondenser at higher MIT | 0.66 ± 0.04                                                             | 0.45 ± 0.04 | 0.42 ± 0.01 | 0.32 ± 0.08 | 0.23 ± 0.00  | 0.44 ± 0.02 | 0.42        |
| Discharge at lower MIT      | 0.50 ± 0.02                                                             | 0.30 ± 0.02 | 0.62 ± 0.04 | 0.30 ± 0.13 | 0.26 ± 0.09  | 0.79 ± 0.03 | 0.46        |
| Discharge at higher MIT     | 0.35 ± 0.04                                                             | 0.57 ± 0.02 | 0.48 ± 0.11 | 0.55 ± 0.21 | 0.34 ± 0.06  | 0.72 ± 0.06 | 0.50        |

\*The mean was based on two replicates.

†MIT, mean incubation temperature.

water to postcondenser water ranged from -48 to +18%. There were no statistically significant differences in mean assimilation rates of precondenser and postcondenser water (Fig. 1). The annual mean difference in rates from the condenser stations showed a 16% reduction due to condenser passage (Table 3).

### Multiple Entrainment Effects

The combination of thermal elevations, pressure fluctuations, abrasion, and turbulence may influence the photosynthetic activities of phytoplankton in the cooling-water system of Oconee Nuclear Station. An index of the effects of these combined factors on carbon assimilation rates was gained by comparing intake water at the lower MIT to discharge water at the higher MIT (Table 2). Percent differences in rates due to multiple entrainment factors were negative in cooler months (January, March, and December) and positive during warmer months. Percent change in mean photosynthetic rates as related to multiple pumped entrainment effects for the year showed a +11% difference (Table 3). A statistically significant difference in mean uptake rates attributable to multiple entrainment effects occurred in the September experiment only.

### DISCUSSION

In general, short-term effects of thermal elevations on intake phytoplankton productivity in both fresh and brackish water in temperate latitudes indicate stimulation in cooler months but inhibition of photosynthesis during warmer months (Smith et al., 1974; Brooks, 1972; Warinner and Brehmer, 1966; Morgan and Stross, 1969). In one study on the effects of cooling water use on intake phytoplankton withdrawn from the hypolimnetic zone (>20 m) of a reservoir, temporary stimulation of primary productivity occurred as water passed through the cooling system and discharge canal (Smith, Brooks, and Jensen, 1974).

Thermal elevations and multiple entrainment stresses at Oconee Nuclear Station resulted in an average annual stimulation of carbon assimilation rates (20 and 11%, respectively). The variability of replicate data precludes any value judgment of effects of entrainment on assimilation rates. Variation in replicate assimilation rates may be related to differences in NaH<sup>14</sup>CO<sub>3</sub> activity inoculated and to internal interferences in the scintillation process.

Mechanical effects were illustrated by comparing intake and discharge rates observed at the higher MIT. Thermal

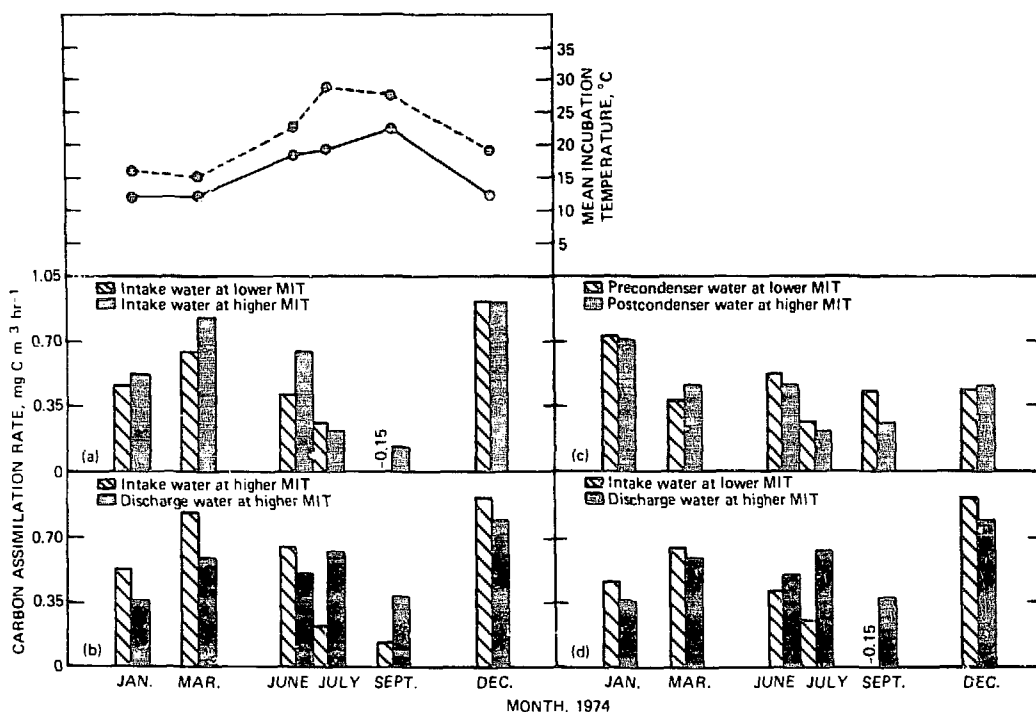


FIG. 1 Monthly differences in carbon assimilation rates due to: (a) thermal elevations of intake water, (b) mechanical effects in the cooling system, (c) condenser passage, and (d) multiple entrainment factors at Oconee Nuclear Station, South Carolina.

12.0°C in January and March to 22.7°C in September (Fig. 1). The higher MIT ranged from 15°C in March to 28.8°C in July.

### Thermal Effects

The effects of temperature elevation on carbon assimilation rates were determined by comparing assimilation rates of intake-water phytoplankton incubated at the lower and higher MIT (Table 2). When intake photosynthetic rates at the two temperatures were compared, increased rates occurred at the higher MIT in four of the six experiments (Fig. 1). Decreased rates at the upper MIT were noted in July and December (−25 to −10%, respectively). In the September experiment, a negative assimilation rate ( $-0.15 \pm 0.0 \text{ mg C m}^{-3} \text{ hr}^{-1}$ ) was recorded in intake water at the lower MIT. The absolute difference between the rate observed in thermally elevated intake water and the negative rate was 180%.

Temperature elevation was not responsible for any significant stimulation ( $p < 0.05$ ) of carbon assimilation rates in any of the six experiments.

On an annual basis, thermal effects at Oconee Nuclear Station caused a 20% increase in photosynthetic uptake of  $^{14}\text{C}$  (Table 3). This increase may be interpreted as the average stimulation of rates when incubation-temperature differences were as low as 3°C.

### Mechanical Effects

Annual mean rates of intake and discharge samples at the higher MIT suggest that mechanical effects on photosynthetic capacity of entrained phytoplankton were negligible (Table 3). Reductions in carbon assimilation rates due to mechanical effects occurred in four of the six comparisons (Fig. 1) and ranged from 20 to 30%. Ambient water temperatures were below 15.7°C when decreased rates were noted. No significant differences occurred among the six experiments with respect to mechanical effects. An annual rate difference of +7% can be attributed to mechanical-factor influences.

Comparing yearly mean rates of intake and discharge water at intake temperatures may also serve as an index of the influence of mechanical factors on phytoplankton physiology. A slight average annual increase in assimilation rates was noted (+2%) for discharge water at the lower MIT (Table 3).

### Effects of Condenser Passage

Phytoplankton passing from the precondenser to postcondenser stations are exposed to rapid temperature rises and mechanical disruption. Our design to examine these effects compared precondenser water at the lower MIT and postcondenser water at the higher MIT (Table 2). The percent differences in photosynthetic rates of precondenser

effects, seasonal effects, density, and sex and age class of each of the major species of copepods. These variables were analyzed to observe how various combinations of factors shaped the response patterns of the entrained organisms. It is hoped that the use of GLH analysis and response-surface techniques will give a clearer picture and a more useful predictive model of what is happening in the field than would a univariate approach.

## STUDY AREA

The entrainment study was conducted at the Florida Power Corporation's Crystal River steam generating plant, which is located between the Withlacoochee and Crystal rivers on the west coast of Florida. The area represents a typical shallow-water middle-salinity estuary found along the Gulf Coast of the state. Annual temperatures generally range from 15 to 30°C, and salinities range from 15 to 30‰. Seasonal freshwater runoff from surrounding terrestrial and marsh areas during rainy seasons cause fluctuations in salinity, detrital content, and organic and inorganic constituents of the water, but the spring-fed rivers of the area provide a constant stabilizing force not found in many northern estuaries.

The Crystal River power plant raises the temperature of the cooling waters passing through it an average of 5.88°C. This thermal increase ( $\Delta T$ ) is quite constant because environmental laws prohibit  $\Delta T$  values above 6.5°C and plant operation efficiencies prevent minimum  $\Delta T$  values much below the mean value. Surface salinity of water going through the plant is generally raised by 1.12‰ because high salinity oceanic water from the bottom is mixed with lower salinity surface waters when the cooling water is taken

from this partially stratified system. The heated effluents are channeled away by a 1-mile-long discharge canal; the water exiting the plant takes approximately 2 hr to traverse the canal during normal operations. During the passage of discharge water down the canal, its temperature remains relatively unchanged until it is released into the open bay area of the estuary.

## METHODS AND MATERIALS

Copepods were collected with a 64- $\mu$ m-mesh 0.5-m plankton net fitted with a digital flowmeter. Samples were taken biweekly from intake and discharge areas from November 1973 through September 1974. The number and length of net tows were adjusted so that approximately equal volumes were sampled from each area and the minimum amount of mortality was caused by collection techniques.

Temperature and salinity measurements were taken at each area during every sampling period. Such factors as dissolved oxygen were not measured because of personnel, equipment, and economic limitations. Data from a concurrent study showed, however, that there was little variation between intake and discharge dissolved oxygen values and that readings never were much less than 6 ppm, a level generally considered far from limiting. It is hoped that future studies will include measurements of such physical parameters so they may also be assessed for relationships to entrainment effects.

Six experimental field treatments were set up to attempt to separate various factors involved in entrainment mortality (Fig. 1). Two copepod populations from the intake area were sampled as controls. The initial control

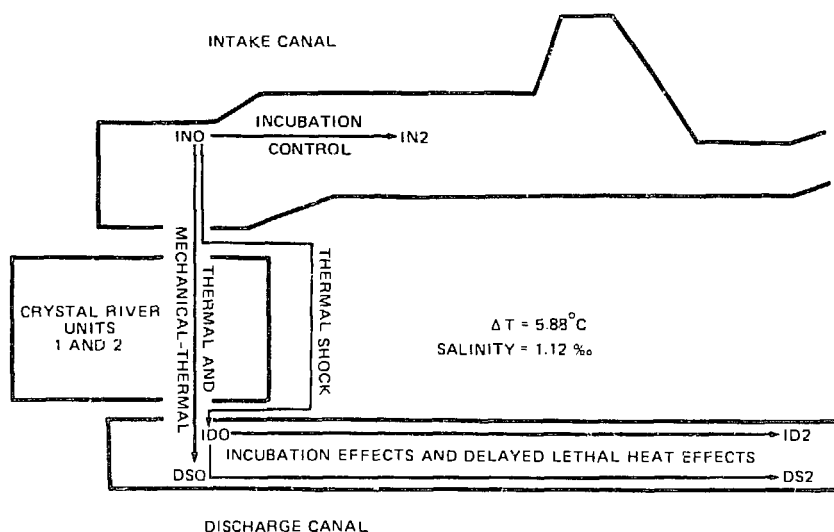


FIG. 1 Schematic diagram of study site indicating designation and purpose of experimental field treatments. Abbreviations are: IN0, initial control; IN2, second intake treatment; ID0, third intake treatment; DS0, discharge sample; ID2, fourth intake treatment; and DS2, second discharge sample. See text for discussion of treatments.



(IN0) was a population collected and analyzed immediately for mortality by vital-stain bioassay (Dressel, Heinle, and Grote, 1972). In the second intake treatment (IN2), an intake population was subjected to 2 hr of incubation in a circulating-system drift-bottle apparatus of the type described by Gonzalez (1973). This device was placed in the intake waters, and the sample, which was treated for bioassay after the incubation, acted as a control for the delayed thermal-exposure treatments described in the following paragraphs.

A third intake population of copepods (ID0) was placed in a screened PVC container and submerged in the heated effluent at the discharge area until it was brought up to discharge temperatures. The sample was then treated for bioassay, and the mortality above control (IN0) values were assumed to be caused by the effects of immediate thermal shock. A discharge sample (DS0) was also assayed immediately for mortality after having undergone the turbulence, shearing forces, and mechanical damage associated with passage through the condenser system of the power plant. Since the only difference between the ID0 and DS0 populations was that the DS0 was subjected to the mechanical effects of entrainment, the damage caused by the physical factors of plant passage can be assessed by regression analysis.

Delayed thermal effects associated with the amount of time the population spends in the heated effluent of the discharge canal were also observed. An intake population (ID2) and a discharge population (DS2) were placed in a drift-bottle apparatus similar to that used in the IN2 treatment and allowed to drift down the length of the discharge canal for 2 hr. These organisms were exposed to the heated canal waters in the same way that a natural population going through the plant would be. The ID2 population, since it did not experience the mechanical effects of entrainment, might more closely represent a population entrained into the thermal plume by tidal recruitment. The results of these treatments allow the assessment of the delayed thermal effects of entrainment encountered within the thermal plume.

Samples from each field treatment were preserved in 5% formalin and kept cold until counts were made. Live-dead counts, done under a dissecting microscope at 50X, were recorded for each sex and age class of the major species of copepods.

Predictive regression models for entrainment mortality under various conditions were formed by a stepwise multiple regression computer program contained in the library of the Center for Instructional and Research Computing Activities at the University of Florida. The independent variables tested were temperature; salinity; delayed thermal, mechanical, and seasonal effects; and effects associated with the sex and age class and the density (numbers per cubic meter) of the species of copepods being observed. The effects were coded as dummy variables as suggested by Searle (1971). Fourth-order temperature and salinity terms were included in the statistical model.

Preliminary regressions indicated that the increase in the amount of variation explained ( $R^2$ ) ceased to be significant for terms beyond the fourth degree. All reasonable interactions between the independent variables were also tested. The stepwise regression program was of the type described by Lough (1975), and the predictive model selected from the sequence was the one that produced the greatest values of  $R^2$  for the most parsimonious model. More-complex models were observed to make sure that no major increases in  $R^2$  values occurred when various combinations of variables were added.

The regression equation for each species was used to plot response surfaces for various entrainment conditions by use of a three dimensional plotter routine. Predicted mortality values were rounded up to the nearest 10% level so that all trends would be clearly discernible on the graphs.

## RESULTS

Data from the study showed an overall trend for the various species of copepods. The mortalities from the entrained DS2 populations, when plotted against discharge temperatures (Fig. 2), showed a general tendency to remain relatively low up to a certain temperature and then to rise almost exponentially above this point. The only exception to this basic trend was *Labidocera* sp. (Fig. 2c), which exhibited high mortalities throughout the temperature range.

Temperature obviously plays an important role in entrainment effects on the other copepod species of the area, but each species exhibited different responses to temperature, as well as to the other factors observed. Enough data were gathered from experimental observations to analyze more closely the response patterns of the four most abundant copepods of the area: *Oithona* sp., *Acartia tonsa*, *Paracalanus crassirostris*, and *Euterpina acutifrons*. The regression equations describing these patterns were highly significant ( $P < 0.001$ ), and the amounts of variation explained were quite large ( $R^2 > 0.75$ ). The actual predictive equations and associated statistics are contained in Table 1.

### *Oithona*

The category *Oithona* sp. contains at least three species (*O. colcarva*, *O. nana*, and *O. simplex*). Species identification was not feasible for the males and juveniles of this genus under the magnifications used for counting, but *O. colcarva*, is by far the most dominant and can be assumed to make up a majority of the category. *Oithona* sp. are the numerically dominant species of the Crystal River area, at times comprising nearly 80% of the total copepod population. Despite their small size, these cyclopoids probably make up a significant part of the secondary biomass production of the zooplankton of the area.

*Oithona* sp. exhibit the simplest response to entrainment effects (Fig. 3). Female and juvenile *Oithona* [Fig. 3(a)] show low (0 to 20%) mortalities when exposed to

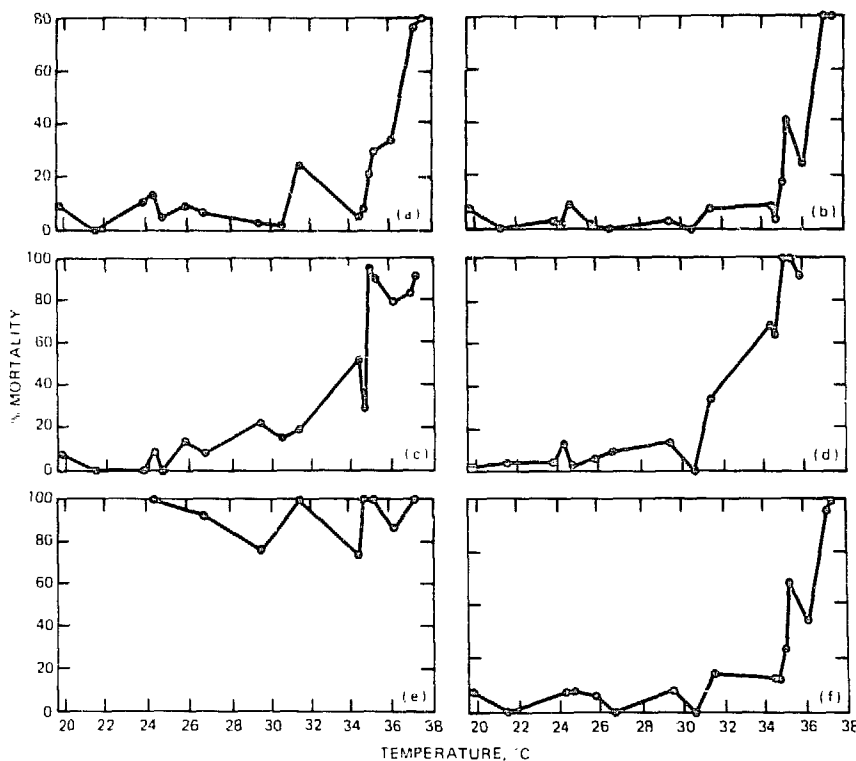


FIG. 2. Entrainment mortality vs. discharge canal temperatures for DS2 populations. (a) *Acartia tonsa*. (b) *Oithona* sp. (c) *Paracalanus crassirostris*. (d) *Euterpina acutifrons*. (e) *Labidocera* sp. (f) Total copepod population.

thermal shock if the discharge temperatures are below 35°C. Higher salinities apparently buffer the effects on mortality of moderately high temperatures (30 to 35°C). Above 35°C, however, mortalities rise exponentially with small increases in temperature regardless of salinity. Male *Oithona* [Fig. 3(b)] exhibit similar patterns but appear to be more sensitive to thermal shock than females or juveniles, having slightly higher mortality values for all conditions except the very lowest temperatures (20 to 22°C).

Extended exposure (2 hr) to the heated effluents encountered in the discharge canal cause male [Fig. 3(c)] and female juvenile [Fig. 3(d)] response patterns to change. Mortalities rise at high temperatures (>32°C), especially with low salinities; the high salinity-high temperature interaction is still somewhat evident. The temperature at which 100% mortality occurs is also lowered because of the delayed thermal effects within the discharge canal.

Mechanical, seasonal, and density-dependent factors were shown to be not significant in relation to entrainment mortality for *Oithona* sp.

#### *Acartia tonsa*

*Acartia tonsa* is probably the major year-round component of secondary production in the zooplankton com-

munity of the Crystal River area. This calanoid comprises 40 to 60% of the total copepod population most of the year and may contribute more biomass than the numerically dominant *Oithona* sp., which are considerably smaller.

The response patterns of male, female, and juvenile *A. tonsa* subjected to thermal shock showed similar trends [Fig. 4(a to c)]. Some interesting temperature-salinity interactions, as expressed by the topography of the response surfaces, were seen for each category at salinities of 19 to 23‰ and temperatures of 20 to 35°C. As for *Oithona* sp., relatively low mortalities are seen for combinations of high salinities and moderately high temperatures (30 to 35°C). The reduction of mortality related to this interaction is greatest in juveniles and least in males. Temperature increases above 35°C, however, are positively correlated with great increases in mortality for the entire range of salinities.

Exposure to delayed thermal effects caused mortalities to rise slightly in all areas except for temperatures less than 35°C and salinities of 20 to 25‰ [Fig. 4(d)]. The temperature at which the exponential rise in mortalities occurs is lowered below 35°C in areas of high salinity, and the range of temperatures at which 100% mortality occurs is broadened.

When mechanical effects are considered in the entrainment conditions [Fig. 4(e)], a rise of 3 to 5%

TABLE 1 Predictive Stepwise Regression Tables

| Environmental variable                 | Regression coefficients* |                      |                                  |                             |
|----------------------------------------|--------------------------|----------------------|----------------------------------|-----------------------------|
|                                        | <i>Oithona</i>           | <i>Acartia tonsa</i> | <i>Paracalanus crassirostris</i> | <i>Euterpina acutifrons</i> |
| Temperature effects                    |                          |                      |                                  |                             |
| Discharge temp.                        | 10.9418                  | 3.1886               |                                  |                             |
| (Discharge temp.) <sup>2</sup>         | 0.6346                   | 0.1974               |                                  |                             |
| (Discharge temp.) <sup>3</sup>         | 0.0161                   | 0.0052               |                                  |                             |
| (Discharge temp.) <sup>4</sup>         | 0.0002                   | 0.0001               |                                  |                             |
| Intake temp.                           | 0.7551                   |                      |                                  | 0.0896                      |
| (Intake temp.) <sup>2</sup>            | 0.0384                   |                      |                                  | 0.0023                      |
| (Intake temp.) <sup>3</sup>            | 0.0006                   |                      |                                  |                             |
| Salinity effects and interactions      |                          |                      |                                  |                             |
| Intake salinity                        |                          | 0.0337               |                                  |                             |
| Intake salinity intake temp.           | 0.0005                   |                      | -0.0005                          |                             |
| Discharge salinity                     |                          | 35.350               |                                  |                             |
| (Discharge salinity) <sup>2</sup>      |                          | 2.1318               |                                  | 0.0981                      |
| (Discharge salinity) <sup>3</sup>      |                          | -0.0567              |                                  | 0.0052                      |
| (Discharge salinity) <sup>4</sup>      |                          | 0.0006               |                                  | -0.0001                     |
| Canal salinity                         |                          |                      | -0.0135                          | 0.0240                      |
| Delayed thermal effects                |                          |                      |                                  |                             |
| Canal temp.                            | -0.0608                  | -0.0783              | -0.0924                          | -0.0452                     |
| Canal dummy                            | 1.0175                   | 1.3415               | 1.8479                           |                             |
| Mechanical effects and interactions    |                          |                      |                                  |                             |
| Mechanical dummy                       |                          | 0.0312               |                                  |                             |
| Mechanical-discharge salinity          |                          |                      |                                  | 0.0328                      |
| Mechanical-intake temp.                |                          |                      | 0.0034                           | -0.0489                     |
| Seasonal effects                       |                          |                      |                                  |                             |
| Spring dummy                           |                          |                      | -0.1157                          | -0.1820                     |
| Population variable groupings          |                          |                      |                                  |                             |
| Males-discharge salinity               |                          |                      |                                  | 0.5608                      |
| Males-mechanical                       |                          | 0.0777               |                                  | 0.1543                      |
| Males-canal                            |                          |                      | 0.1393                           | 0.1266                      |
| Females-(discharge temp.) <sup>2</sup> |                          | 0.0001               |                                  | 0.0003                      |
| Females-discharge salinity             |                          |                      |                                  | 0.3357                      |
| Females-canal                          |                          |                      |                                  | 0.1528                      |

\*For *Oithona* sp.  $R^2 = 0.748$ , T statistic = 44.25; for *A. tonsa*  $R^2 = 0.750$ , T statistic = 30.87; for *P. crassirostris*  $R^2 = 0.760$ , T statistic = 38.97; and for *E. acutifrons*  $R^2 = 0.767$ , T statistic = 9.49. For all species  $P < 0.0001$ .

mortality occurs in all areas except those of optimum conditions (20 to 35°C, 20 to 23‰). The *A. tonsa* male response pattern (not shown) exhibits a similar trend, but increases in mortality are near 10%.

The addition of seasonal effects associated with spring produces a response pattern for *A. tonsa* [Fig. 4(f)] identical to those for other seasons [Fig. 4(e)] except that all mortality values are ~10% higher.

#### *Paracalanus crassirostris*

*Paracalanus crassirostris* is the third most abundant copepod in the Crystal River area. This species is present

year-round and generally comprises 10 to 20% of the total copepod population.

*Paracalanus crassirostris* males [Fig. 5(a)] showed slightly lower mortalities than did the females and juveniles [Fig. 5(b)] when exposed to thermal shock. The response patterns show a gradually increasing rise in mortalities with increases in temperature. A slight temperature-salinity interaction is evident throughout the range of conditions, with higher salinities producing lower mortalities for any given temperature. Mortality values for *P. crassirostris* do not rise much above 70%, even for the highest temperature (40°C) during this short-term exposure to thermal stress.

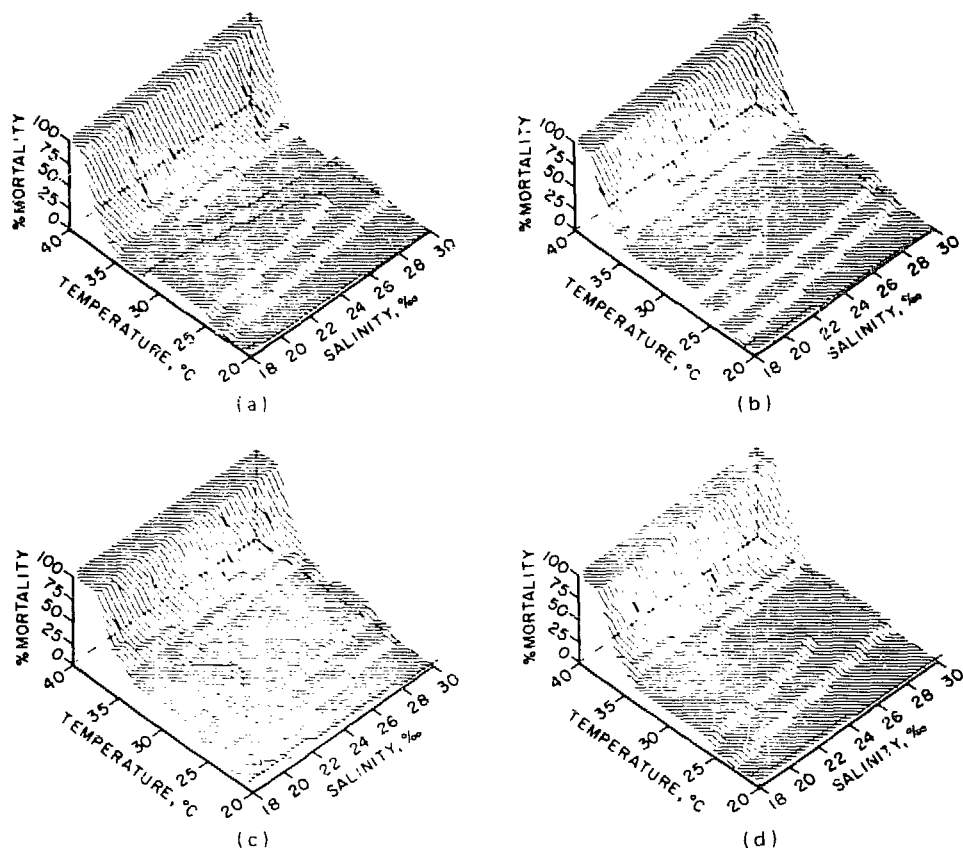


FIG. 3 Response-surface estimation of mortality as a function of discharge temperature and salinity for populations of *Oithona* sp. exposed to various conditions. (a) Female and juvenile and (b) male immediate response to thermal shock. (c) Male and (d) female-juvenile response to extended exposure (2 hr) to elevated temperatures.

Extended exposure to the heated effluent [Fig. 5(c)] causes *P. crassirostris* mortality values to rise rapidly for temperatures above 30°C. For temperatures below this point, higher salinities continue to produce lower mortalities. Mortalities of males subjected to extended exposure were 10 to 15% higher than those of females.

When the mechanical effects of entrainment are included in the *P. crassirostris* response-surface equation [Fig. 5(d)], mortalities are only slightly increased for temperatures below 30°C. Above this point, however, the rise to the 100% mortality level occurs more rapidly and for lower temperatures than would be seen for a population not subjected to the physical stresses involved in passage through the plant [Fig. 5(c)].

*Paracalanus crassirostris* populations observed in the spring [Fig. 5(e)] had mortalities ~10% lower than would be expected, except for the most extreme temperatures (>37°C).

Response patterns for this species are significantly affected by population density at the time of entrainment. When numbers per cubic meter are low [Fig. 6(a)],

entrainment mortalities are generally higher than when the numbers are high [Fig. 6(b)]. This effect is particularly evident for the more lethal conditions, where the temperature at which the 100% mortality level occurs may vary several degrees depending on the density factor.

#### *Euterpina acutifrons*

*Euterpina acutifrons*, a pelagic harpacticoid, is the fourth most abundant species at Crystal River. This species is found throughout the year, with lowest populations in the summer.

In response to short-term thermal stress, male *E. acutifrons* [Fig. 7(a)] are less sensitive to the various temperature and salinity combinations than are females [Fig. 7(b)]. Juveniles [Fig. 7(c)] appear more resistant than adults, except for high salinity. The response surfaces for *E. acutifrons* exhibit a stronger temperature-salinity interaction than do those for *P. crassirostris*. High salinities and low temperatures produce low mortalities, but low salinities and high temperatures produce high mortalities. The effects of this interaction on immediate response to

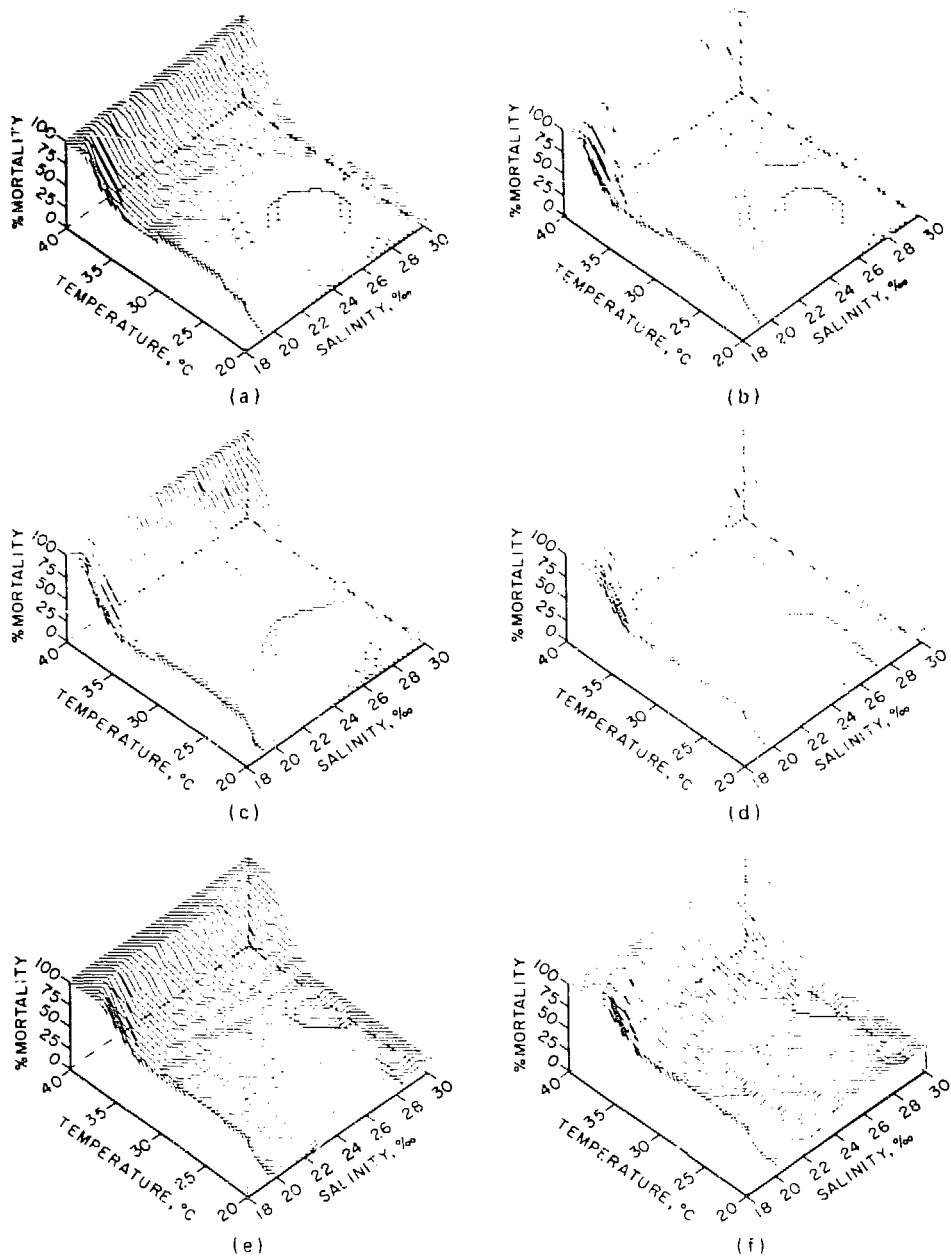


FIG. 4 Response-surface estimations of mortality as a function of discharge temperature and salinity for populations of *Acartia tonsa* exposed to various conditions. (a) Male, (b) female, and (c) juvenile immediate response to thermal shock. (d) Female response to extended exposure to elevated temperatures. (e) Female response to mechanical damage in addition to delayed thermal effects. (f) Female response to the same conditions as e but in spring.

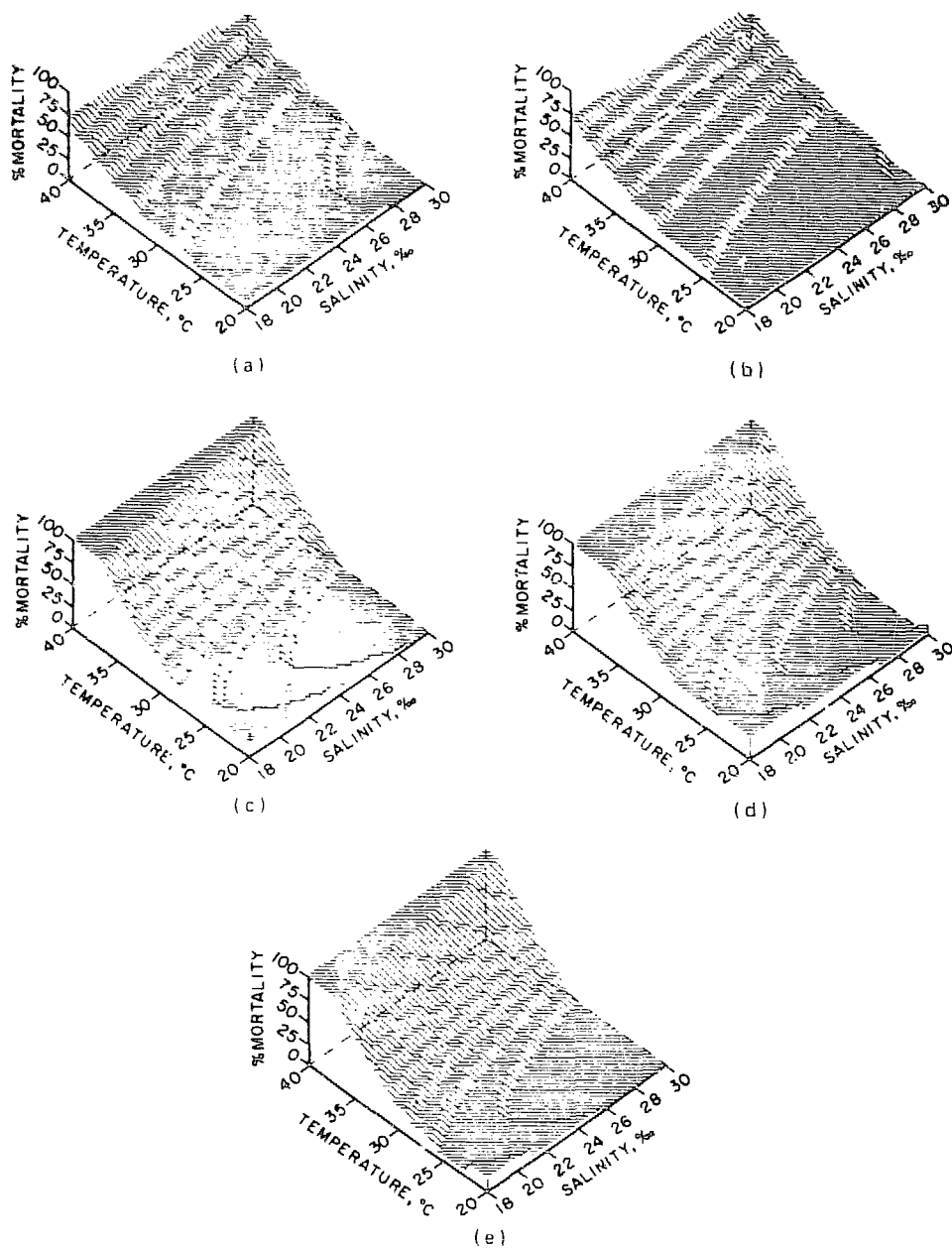


FIG. 5. Response-surface estimations of mortality as a function of discharge temperature and salinity for populations of *Paracalanus crassirostris* exposed to various conditions. (a) Male and (b) female—juvenile immediate response to thermal shock. (c) Female response to extended exposure to elevated temperatures. (d) Female response to mechanical damage in addition to delayed thermal effects. (e) Female response to the same conditions as (d) but in spring.

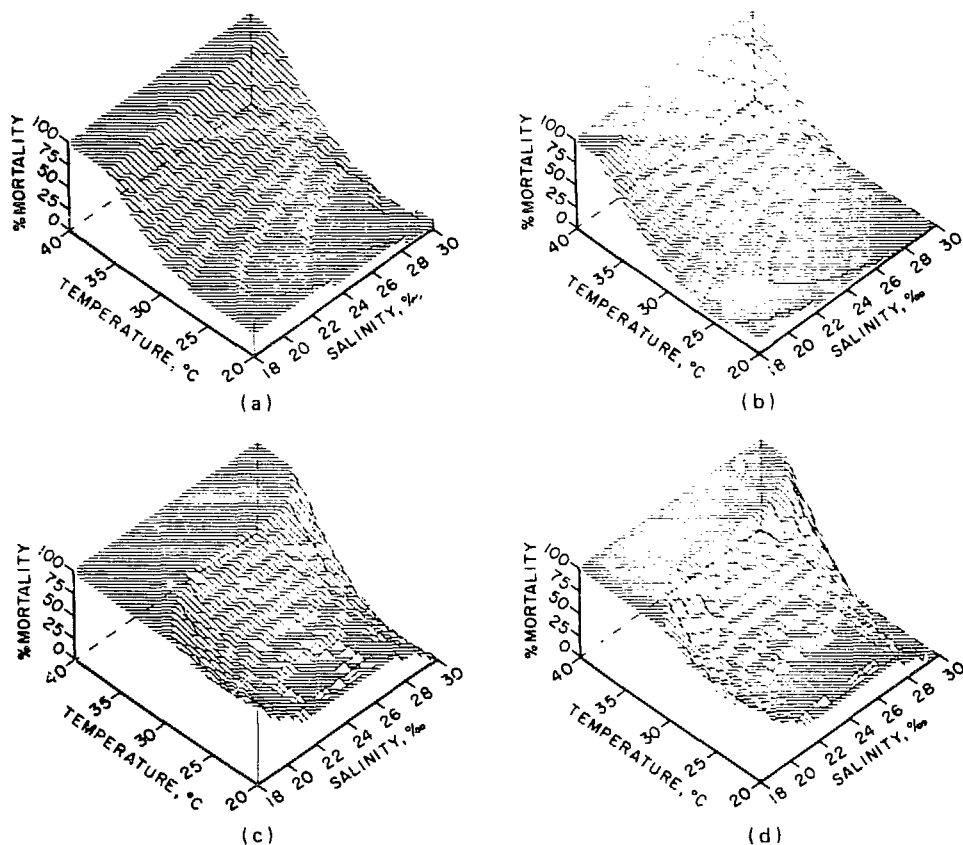


FIG. 6 Response-surface estimations of mortality as a function of discharge temperature and salinity for entrained populations at different densities (numbers/m<sup>3</sup>). (a) Female *Paracalanus crassirostris* at lowest density. (b) Female *P. crassirostris* at highest density. (c) Female *Euterpina acutifrons* at low density. (d) Female *E. acutifrons* at highest density.

thermal shock appear more important than do the effects of either factor considered alone.

When the discharge canal effects are considered [Fig. 7(d)], mortalities rise, especially at high temperatures (>35°C). Male *E. acutifrons* have mortality levels 10 to 15% higher and juveniles have mortality levels 10 to 15% lower than female mortalities. The temperature-salinity interaction remains evident, but extended exposure to the heat makes the factor of temperature relatively more important in its upper range.

Mechanical effects cause mortality values in the response pattern for *E. acutifrons* [Fig. 7(e)] to rise slightly, the effect being especially evident in the area of low temperature and high salinity. Most of the temperature-salinity interaction disappears, and the temperature of the discharge canal becomes the dominant factor in entrainment mortality.

As for *P. crassirostris*, populations of *E. acutifrons* entrained in the spring had significantly lower mortalities than would be expected [Fig. 7(f)]. Except at temperatures

above 36°C, mortalities are 10 to 20% lower throughout the range of conditions.

*Euterpina acutifrons* also showed response-surface changes due to density factors. When there were low numbers of this species per cubic meter of water [Fig. 6(c)], there were higher mortalities, but, when densities were higher, there were lower mortalities [Fig. 6(d)].

## DISCUSSION

### Temperature and Salinity Effects

The response of the various species of copepods to the effects of entrainment show certain similarities and some individual differences. Temperature, salinity, and temperature-salinity interactions are the most important factors in determining the resistance of copepods to thermal stress. All the species appear to have a critical thermal level above which mortality increases exponentially. Below this level, however, the response pattern for each species has a unique configuration shaped by the

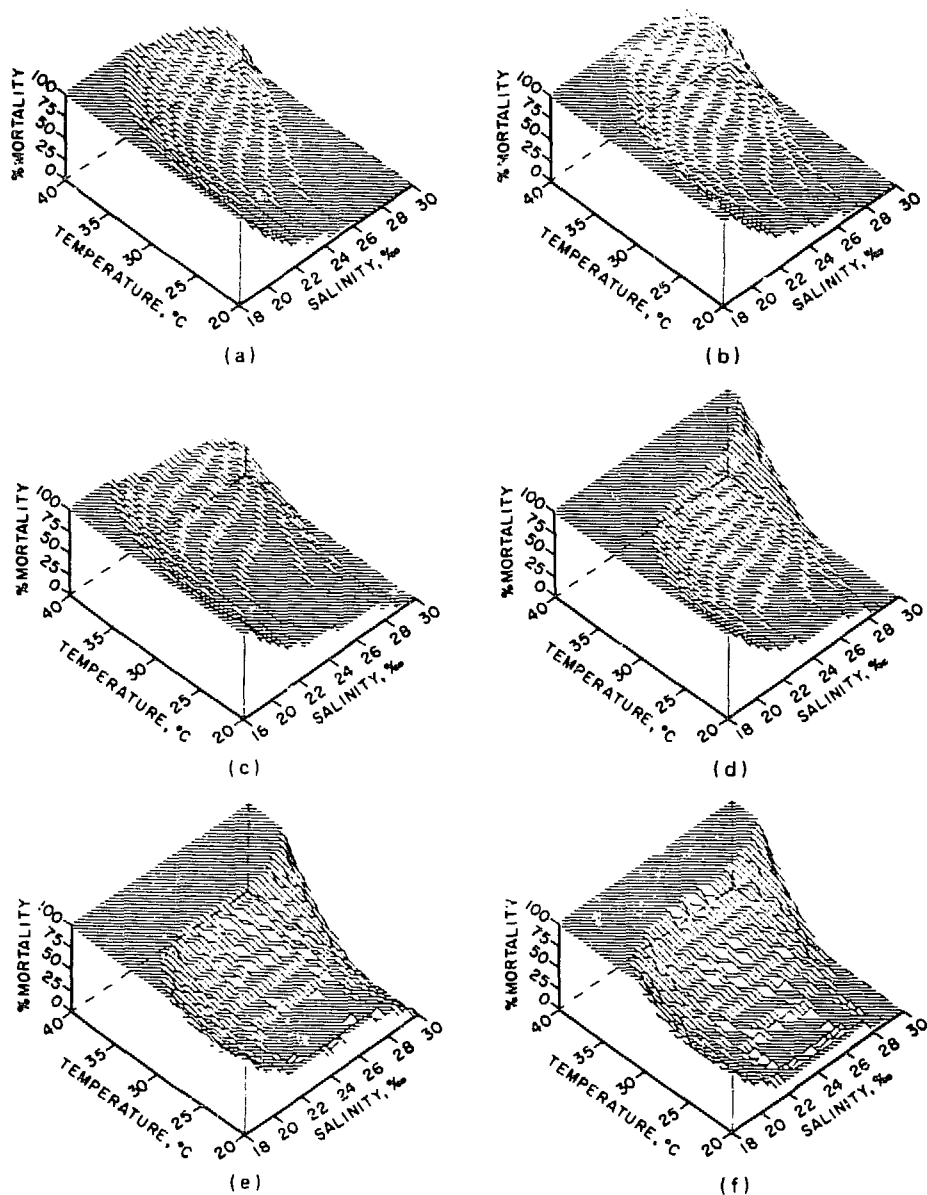


FIG. 7 Response-surface estimations of mortality as a function of discharge temperature and salinity for populations of *Euterpina acutifrons* exposed to various conditions. (a) Male, (b) female, and (c) juvenile immediate response to thermal shock. (d) Female response to extended exposure to elevated temperatures. (e) Female response to mechanical damage in addition to delayed thermal effects. (f) Female response to the same conditions as e but in spring.



ecological requirements and biological characteristics of the species.

When exposed to short periods of thermal shock, *Oithona* sp. show quite low mortalities for temperatures below 35°C. Mortality is most directly related to temperature, but high salinities appear to buffer the effects of moderately high temperatures. The effects of such combinations on poikilotherms was described by Ranarde (1957), Lance (1963), Alderdice (1972), and most extensively by Kinne (1964). Except for this interaction, salinity plays little part in the resistance of *Oithona* sp. to thermal shock. *Oithona* sp., principally *O. colcarva*, are estuarine species which are most abundant in areas with salinities below 31‰ (Grice, 1957).

Another estuarine form, *A. tonsa*, displays a resistance to thermal shock over quite a range of temperatures and salinities. This species is reported to be both euryhaline and eurythermal, but high salinities (>25‰) are considered suboptimal (Barlow, 1955; Riley, 1967; Gonzalez, 1964). The response surface for *A. tonsa* reflects this, with lower salinities allowing complete survival up to 35°C. Higher salinities, which are probably metabolically suboptimal, produce higher mortalities when the species is subjected to thermal shock. A high salinity-high temperature tolerance interaction similar to that seen for *Oithona* sp. is also evident.

*Paracalanus crassirostris* is a coastal species, generally found in areas with salinities of 24.4 to 38.4‰ and temperatures of 10 to 32°C (Grice, 1960). This species demonstrates quite a resistance to the immediate effects of thermal shock, with no mortalities greater than 70%. There is a temperature-salinity interaction; high salinities are associated with lower mortalities across the range of temperatures.

Since *E. acutifrons* was shown by Grice (1957) to have an identical range with *P. crassirostris*, it is not surprising that it shows a temperature-salinity relationship of the type described for *P. crassirostris*. The interaction seems to be more pronounced in affecting the amount of mortality caused by thermal shock.

Two general patterns of response to thermal shock emerge from our studies: (1) low mortalities until very high temperatures are experienced, exemplified by *Acartia* and *Oithona*, and (2) mortalities gradually increasing with temperature but ameliorated by the salinity-temperature interaction, e.g. *Euterpina* and *Paracalanus*. The estuarine forms may be responding as observed because they are preadapted to thermal pulses in such a habitat. In addition, the salinity ranges are well within the natural variation of their environment.

In the second case, the coastal forms are probably experiencing suboptimal salinities at the time of thermal stress. The organism is in better physiological condition to deal with the stress of thermal shock at higher salinity levels. This seems to be a plausible explanation for the presence of a strong temperature-salinity interaction.

The response surfaces are, in effect, the inverse of a fitness surface and map the organism's ecological fitness for various aspects of its environment, whether natural or man induced. We propose that the two apparently different fitness patterns are really the same, but each is centered at a different point along the salinity scale.

### Delayed Thermal Effects

The effects of exposure to the heated effluent for the 2 hr that the entrained population remained in the discharge canal produce observable changes in the response patterns. The high salinity-high temperature interaction for *Oithona* sp. starts to disappear when the species is exposed to thermal stress for a longer period of time. Temperature increases above 35°C produce a greater increase in mortality when this species remains in the heated effluent. The response patterns of *A. tonsa* exhibit similar effects; the high salinity-high temperature interaction is reduced in importance, and mortalities increase rapidly to the 100% level for temperatures above 35°C.

The responses of *P. crassirostris* and *E. acutifrons* to several hours of thermal stress were similar to each other. The initial resistance to thermal shock disappears, and both species show a sensitivity to high temperatures (>33°C) for all salinities. The temperature-salinity interaction remains important for temperatures below 33°C but is overridden by the delayed effect above this temperature. The optimum conditions for resistance to thermal stress center around the combination of 25°C and 30‰ salinity. Males of both species were particularly sensitive under extended exposure to elevated temperatures, showing mortalities 10 to 15% higher than females.

The effects of extended thermal stress seem to be less important for the estuarine species *Oithona* sp. and *A. tonsa* than for the coastal species *P. crassirostris* and *E. acutifrons*. The estuarine species exhibit low mortalities over a wide range of temperature and salinity combinations and show little increase in mortality due to canal effects except at the highest temperatures (>35°C). The coastal species, on the other hand, have low mortalities only in areas of high salinity and low temperatures. The canal effects cause temperatures much above 33°C to become lethal for these species.

### Mechanical Effects

The mortality caused by mechanical damage associated with passage through the power plant seems to depend, at least in part, on the size of the entrained organism. *Labidocera* sp., by far the largest species of the area, exhibit high mortalities, regardless of environmental conditions [Fig. 2(e)]. On the other hand, *Oithona* sp., the smallest copepods in the area, exhibit no significant mechanical damage. The remaining species, which are intermediate in size, undergo some mechanical damage due to entrainment.

Mechanical damage is correlated with a 3 to 5% rise in mortality for *A. tonsa* females and juveniles for all but optimum conditions of temperature and salinity. The males

of this species appear to be more sensitive, exhibiting a rise of 10% mortality. *Paracalanus crassirostris* and *E. acutifrons* also show a 5 to 10% rise for low temperatures, but a temperature-mechanical interaction causes higher mortalities at higher temperatures. This effect is particularly obvious for *E. acutifrons*, for which the temperature-salinity interaction is completely hidden. As a result, temperature now appears to be the major lethal factor.

The temperature-mechanical interaction may result when the entrained organisms come into contact with the hot walls of condenser pipes and thereby encounter more stressful conditions than the temperature of the discharge waters would indicate. Thus the mechanical effects of entrainment may actually have a thermal component that would be important as the water gets hotter. On the other hand, this interaction may result because the thermally stressed organisms are more susceptible to the effects of mechanical damage.

### Seasonal Effects

*Paracalanus crassirostris* and *E. acutifrons* both show lower mortalities than expected in the spring season, but *A. tonsa* shows higher mortalities. The factors that cause such responses are unknown but may be related to phytoplankton blooms or changes in organic and inorganic constituents of the water associated with spring rains. Such factors affect the overall health of the populations, which, in turn, might determine their response to entrainment effects.

### Density Effects

*Paracalanus crassirostris* and *E. acutifrons* show lower mortalities when population numbers are high. This variable may be simply an indirect measurement of the factors determining the general health of the population. High numbers would be found during optimum conditions, and lower entrainment mortalities would be expected during this time when the populations are more robust. Metabolic factors such as hormone levels may also have an effect on response patterns during breeding seasons.

### Age and Sex Effects

Generally, juveniles of a given species show lower mortalities than adults. For *Oithona* sp. and *P. crassirostris*, the difference in response was not significant, but *A. tonsa* and *E. acutifrons* juveniles have lower entrainment mortalities. Jensen et al. (1969) suggest that juvenile poikilotherms may not be as susceptible to thermal death as adults of the same species because they are biochemically less complex. The smaller size of the juveniles may also make them less likely to suffer physical damage in passage through the plant.

Males of a species are usually more sensitive to entrainment effects than females. Even when the males were more resistant to the initial thermal shock, as in the case of *P. crassirostris* and *E. acutifrons*, they were generally more sensitive to mechanical and delayed effects, and thus entrainment mortality was higher. Hormonal and

metabolic differences between males and females may produce differences in resistance to entrainment effects. Woodmansee (1958) explains the low male-to-female sex ratio found in natural populations of *A. tonsa* by suggesting that the males are more susceptible to natural mortality. Sex ratios for all the major species studied show similar trends, suggesting that the males of a species would be the more sensitive to stressful conditions, such as those encountered in entrainment.

### CONCLUSIONS

Using our experimental design and response-surface techniques in a field observational context made possible a detailed examination of factors involved in entrainment mortality. Although the predictive models formed from data taken during this study may not completely reflect all the responses of a natural population, they seem to show the basic reactions of the species to entrainment. These basic reactions support two conclusions: (1) Given that mechanical and thermal shock effects are unavoidable, power plants having prolonged delayed thermal effects will cause significantly increased mortality in entrained planktonic communities. (2) Estuarine copepods are preadapted to many of the stresses imposed by power plants and thus have a higher fitness than do coastal or pelagic forms. This might have a considerable effect on plankton community structure and productivity of estuaries exposed to coastal power plant entrainment. Studies of these aspects are currently in preparation.

### ACKNOWLEDGMENTS

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# Entrainment of Estuarine Zooplankton into a Mid-Atlantic Power Plant: Delayed Effects

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## ABSTRACT

Zooplankton collected from the intake and discharge of the Crane Power Plant, located on a tributary of the Chesapeake Bay, were incubated under continuous-flow ambient and experimental conditions in netted containers. No difference in survival was noted among zooplankton held at ambient water temperatures, those which passed through the plant and were subsequently exposed to 1.0 and 2.0  $\Delta T$ , and the intake zooplankton that were transferred directly to 1.0  $\Delta T$  water. Similarly, no significant effect was observed in a comparison of the resident zooplankton communities in the vicinity of the discharge canal, the intake, and a control station. These data suggest that microzooplankton survival after exposure to condenser cooling-system effects is relatively high for copepod species and population structures tested in these studies. Further attempts to refine evaluations of macrozooplankton, meroplankton, and ichthyoplankton organisms appear to be warranted.

The entrainment of zooplankton into thermal power plant cooling systems creates a potential environmental impact that has only recently come under close scrutiny. Entrainment of zooplankton may result in their death from exposure to increased temperature, rapid pressure change, mechanical abrasion, toxic chemicals, or a combination of any of these factors during passage through a cooling system. Sublethal exposure to any or all of the variables can result, for example, in lowered resistance to predation or increased susceptibility to disease. In addition, such stress may be of ecological importance because it inhibits the reproductive and growth processes. The loss of zooplankton is serious since they are a vital link in the food web and composite ecosystem.

One of the central issues to consider in environmental assessment of the industrial use of natural surface waters is the response of the aquatic biota. The short-term response of zooplankton directly affected by entrainment in power-plant cooling waters has been reasonably well defined by Davies and Jensen (1974), Icanberry and Adams (1974), and others. The question of the long-term effects of

entrainment on zooplankton survival and population structure still remains, however.

The ultimate aim of predictive environmental assessment is to estimate the effect of entrainment of zooplankton on populations endemic to the receiving waters. Few studies to date, however, have combined predictive long-term on-site laboratory studies of partially controlled natural systems with quantitative base-line studies of populations inhabiting the receiving waters of a power plant and surrounding control areas. This study attempts to provide some of this information through in situ culturing of entrained zooplankton populations and comparative base-line sampling. The study also examines the influence of collection techniques and other experimental factors that may affect survival within captive bioassay populations.

## MATERIALS AND METHODS

### Study Area

Entrainment studies were conducted at the Charles P. Crane Power Plant, which is located on the northwest shore of Chesapeake Bay ~20 km (12 miles) north of Baltimore, Md. The power plant consists of two oil-fired units with a total operating capacity of 385 Mw and requires a continuous cooling water flow of 16.4 m<sup>3</sup>/sec. Transit time of cooling water through the power plant is 2 to 3 min, during which its average temperature is increased 5.6 to 6.7°C before it is released into a 0.8-km (0.5-mile) discharge canal. Cooling-water transit time to the receiving waters is approximately 120 min.

The study area, located on the Chesapeake Coastal Plain, can be generally characterized as having shallow, turbid waters overlying unconsolidated bottom sediments composed mostly of sand, gravel, and mud. Seneca and Saltpeter creeks, on either side of the Carroll Island peninsula, are exceedingly shallow, having an average depth

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of 1 to 2 m. Maximum depths in the study area are in a dredged barge channel leading to the power-plant intake on Seneca Creek.

Water quality and exchange volume are dominated primarily by fluctuations in flow from the Susquehanna River at the head of Chesapeake Bay (Schubel, 1972). Tidal fluctuations average 0.5 m, with recorded extremes of 3 m during gale-force winds. Water turbidity is commonly high, reflecting the shallow, unstable nature of the bottom. Salinity ranges from 0 to 5‰, with lowest values observed during periods of high freshwater runoff. Surface pH is slightly basic, and the surface dissolved oxygen is generally near saturation.

### Thermal Bioassays

Zooplankton for thermal bioassays were collected at night from the power-plant intake and discharge by 110 liter/min submersible pumps and were filtered through either No. 20 (aperture ~76  $\mu\text{m}$ ) or No. 10 (aperture ~153  $\mu\text{m}$ ) nets suspended in 120-liter barrels. Zooplankton samples were transferred to 20-liter polyethylene carboys held in Styrofoam chests for transport to the bioassay laboratory. Water circulation and exchange through the carboys were facilitated by two 15- by 20-cm screened portals in each container. In addition, the 76- $\mu\text{m}$  mesh-covered portals were designed to allow concentration of each sample to a 4-liter volume during periodic subsampling.

In the bioassay laboratory each zooplankton sample received one of four experimental treatments. Samples collected from the power-plant intake were either held at ambient temperature as controls or placed into tanks containing water at the discharge temperature (1  $\Delta\text{T}$ ). Zooplankton samples collected from the power-plant discharge were either held at discharge temperature (1  $\Delta\text{T}$ ) or placed into tanks containing water at an elevated temperature of 2  $\Delta\text{T}$  above ambient. The artificially heated tanks were then cooled similarly to normal temperature die-away as measured in the discharge canal. Subsequently, all tanks were allowed to follow normal ambient water-temperature variations for periods of 2 weeks or longer. Each sample population was subsampled and preserved in 80% ethanol at 2-day intervals. Temperature and dissolved oxygen were measured inside and outside each net bottle when subsamples were taken, and once each day all bottles were manually emptied to the 4-liter level to assure complete water exchange.

### Distribution Studies

Zooplankton distribution was determined from samples collected in the intake (Seneca Creek) and discharge (Saltpeter Creek) waters and compared with an offshore station (Channel Marker 1). We sampled all stations each month from August 1974 through February 1975 using a submersible pump in combination with a No. 20 plankton net (aperture ~76  $\mu\text{m}$ ) and towed No. 10 (aperture ~153  $\mu\text{m}$ ) and No. 0 (aperture ~505  $\mu\text{m}$ ) standard 0.5-m plankton nets. To compensate for patchy zooplankton distribution,

we collected pumped samples for 75 min as the boat drifted in the vicinity of each station. In water less than 2 m deep, only subsurface (0.3 m) samples were collected; surface and bottom samples were combined in deeper waters. Net samples were collected by towing at approximately 4 to 6 km/hr in an oblique pattern from bottom to surface. Tows were normally 15 to 60 min long depending on zooplankton densities. Samples were preserved in 80% ethanol for later identification and enumeration.

### Sample Enumeration and Statistical Analysis

Preserved samples were counted in the laboratory by the use of both transmitted- and reflected-light microscopes. A minimum of three 1-ml aliquots were withdrawn from each thoroughly mixed station sample or bioassay subsample with an automatic pipette and were counted in Sedgewick-Rafter cells or petri plates. Sample precision between aliquots was greater than 90%.

In bioassay subsamples only organisms showing little or no decomposition and assumed to have been alive when subsampled were counted. Bacterial decomposition of dead copepods was active at temperatures as low as 2°C. Discernible decomposition within the 2-day subsampling intervals was evident in the fragmentation and disintegration of the copepod abdominal cavity and appendages. When tested, the ethanol method, vital staining (Dressel, Heinle, and Grote, 1972), and motility analysis (Davies and Jensen, 1974) gave comparable results of live-to-dead ratio. Moreover, the ethanol method was preferred because of its simplicity and organism nonspecificity as compared with other methods tested.

Organisms within each sample were identified by species where possible, with a general categorization for nauplii, copepodite, or adult life stages. In counting, life stages were grouped as nauplii (nauplius I to VI), early copepodites (copepodid I to III), late copepodites (copepodid IV to V), and adults.

Population data for monthly thermal bioassays were compiled separately, and all statistical comparisons were performed on data within a particular month. We performed linear regression analyses for each life stage within a specific treatment population by the least-squares method using a natural logarithmic transformation of the population data. Regression coefficients were used as an index of survival within each bioassay population. We made statistical comparisons of regression coefficients between treatments by calculating an analysis of variance separately for each life stage. Mean sums of squares for deviations about each regression (MSE) were calculated for all treatments combined. Criteria for differences between regression coefficients for each treatment were established by the bounds:

Regression coefficient for the intake vs. control treatment,

$$\pm \sqrt{\frac{\text{MSE}}{\sum (X_i - \bar{X})^2}}$$

where  $X$  represents sample dates.

Distribution data are presented as organisms per liter, estimated from the sample counts. We made statistical comparisons of the station distribution data using an *t*-test based on a one-way analysis of variance for each collection technique. Dominant species within each sample population were determined on the basis of percent composition.

## RESULTS

### Thermal Bioassays

To accurately simulate actual thermal conditions, we made several determinations of the temperature profile in the discharge canal and receiving waters. Figure 1 illustrates a representative thermal profile observed in the 1 and 2  $\Delta T$  thermal bioassays that simulate discharge-canal conditions.

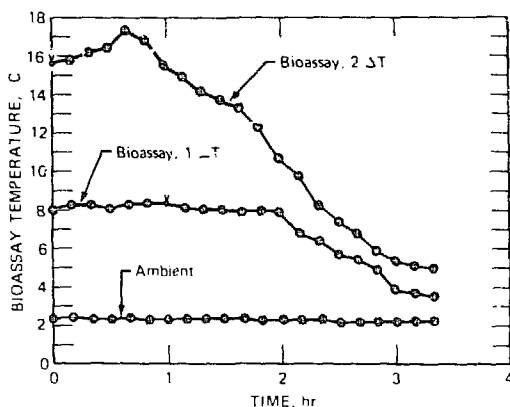


FIG. 1 Typical thermal profiles observed in 1 and 2  $\Delta T$  experimental bioassays. V indicates beginning of bioassay.

In general, bioassay temperature decay to ambient temperature was complete in approximately 2 and 3 hr in the 1 and 2  $\Delta T$  bioassays, respectively. Thermal decay profiles were similar for each monthly bioassay.

### Bioassay Method Evaluation

Several experiments conducted during the course of this research were designed to evaluate the influence of zooplankton collection techniques and abnormally high zooplankton density on survival within our captive bioassay populations. In addition, a series of experiments was undertaken to determine escapement of zooplankton from the bioassay containers and the water exchange patterns between the bioassay containers and the surrounding waters.

Survival of copepodite and adult zooplankton collected with a submersible pump and a standard 0.3-m plankton net was compared. No significant differences were observed in copepodite and adult survival between the pump- and net-collected populations (Davies, unpublished data).

The influence of abnormally high densities of zooplankton within the bioassay containers was of concern.

The initial density in each bioassay container ranged from 15,000 to 35,000 organisms/liter, including nauplii. This is about 170 to 400 times as great as the highest zooplankton density observed in the estuary. A plot of zooplankton survival, as indicated by the least-squares regression coefficient, against the natural logarithm of the mean observed density within each bioassay population collected from the power-plant intake is presented in Fig. 2. An *t*-test was performed on an analysis of variance for each least-squares linear regression. The slope of the regression line for nauplii ( $-0.11$ ) was determined to be significantly different ( $\alpha = 0.05$ ) from 0. Thus increased density appeared to result in either decreased nauplii survival or increased escapement. In these tests, however, density did not appear to significantly affect the survival or escapement of copepodite and adult zooplankton.

Zooplankton escapement from the bioassay containers was evaluated to determine the degree of zooplankton emigration. Population densities were found to be significantly altered by zooplankton escapement, decreasing from  $2.44 \times 10^5$  to  $1.56 \times 10^5$  organisms/liter in 30 min. Samples collected from the prefiltered water surrounding the experimental container were found to contain  $4.69 \times 10^4$  organisms after the 30-min exposure period. Analysis of these data indicates a 36% reduction in population density as a result of escapement.

Nauplii represented 99% of the organisms that escaped. Mean critical width of a sample of nauplii found to pass through the 76- $\mu$ m mesh (107- $\mu$ m diagonal) was 77  $\mu$ m ( $SD = 9$ ). Copepodites were observed to pass through the nets, but the number was negligible. The mean critical width for copepodites was 169  $\mu$ m ( $SD = 31$ ). On the basis of these findings, results presented for each thermal bioassay focus exclusively on copepodite and adult life forms, with major emphasis on copepodite and adult *Eurytemora affinis*.

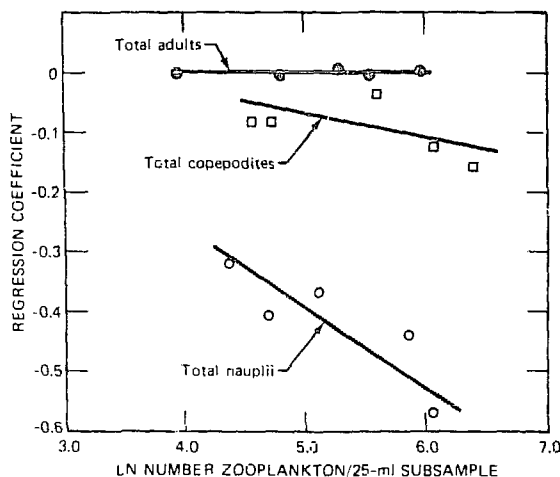


FIG. 2 Relationship between population decrease and mean initial density for each zooplankton life stage within bioassay populations (all species combined).

The degree of water exchange and circulation through the bioassay containers was also examined. Dye dispersion studies, in combination with internal and external temperature measurements during a thermal bioassay, indicated that water exchange in the bioassay containers was minimal. In addition, internal dissolved-oxygen concentrations were observed to decrease substantially over a period of time; this also indicates poor water circulation. To enhance water exchange and to insure adequate oxygen and temperature exchange over the entire observation period, we flushed bioassay containers artificially each day, or more frequently during heat die-away.

### December Bioassay

The bioassay conducted December 19 and observed through January 4 was characterized by an ambient temperature of 4.7°C. Maximum test temperatures in the 1 and 2  $\Delta T$  bioassays were 10.7 and 18.8°C, respectively. Ambient water temperatures during the 18-day posttest observation period averaged 4.8°C and ranged from 4.3 to 6.1°C.

Results of the December bioassay are presented in Table 1 for each treatment. Regression coefficients, derived

### January Bioassay

The bioassay conducted January 9 and observed through January 25 was characterized by an initial ambient temperature of 5.6°C. Maximum test temperatures were 12.2 and 18.1°C in the 1 and 2  $\Delta T$  bioassays, respectively. Ambient water temperatures during the 16-day posttest observation period averaged 6.2°C and ranged from 0.5 to 7.1°C.

Results of the January bioassay are presented in Table 2 for each treatment. Instar analysis was performed on *Eurytemora affinis*, the dominant generic group in the January bioassay population (71%). Differential rates of population decrease were apparent. Early copepodites had the highest rates, and each subsequent stage had progressively lower rates. Mortality trends were similar for each treatment population.

No statistical difference was observed between the regression coefficients for each experimental treatment. It was, however, interesting to note the consistent mortality patterns among the copepodites (MSE, 0.01) and adults (MSE, 0.01) in each treatment population. Mean sums of errors about each treatment regression reflect small deviations between regression coefficients for each treatment population.

TABLE 1 Regression and Correlation Coefficients Calculated by Least-Squares Linear Regression Analysis for December Bioassay Populations

| Life stage                | Intake control         |                         | Discharge 1 $\Delta T$ |                         | Discharge 2 $\Delta T$ |                         | Intake 1 $\Delta T$    |                         |
|---------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|
|                           | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient |
| Total copepodites         | -0.08                  | -0.97                   | -0.08                  | -0.87                   | -0.08                  | -0.89                   | -0.07                  | -0.89                   |
| Total adults              | -0.03                  | -0.87                   | -0.02                  | -0.59                   | -0.02                  | -0.75                   | -0.02                  | -0.38                   |
| <i>Eurytemora affinis</i> |                        |                         |                        |                         |                        |                         |                        |                         |
| Copepodites               | -0.08                  | -0.96                   | -0.08                  | -0.87                   | -0.07                  | -0.88                   | -0.06                  | -0.86                   |
| Adults                    | -0.02                  | -0.71                   | -0.01                  | -0.31                   | -0.03                  | -0.81                   | -0.01                  | -0.20                   |

from least-squares linear regressions, were used as an index of survival within each bioassay population. Linear regressions performed on a natural logarithmic transformation of the data were, in general, indicative of an exponential mortality within the test populations. Differences between copepodite and adult mortality rates were determined to be statistically insignificant.

We performed a statistical comparison of the regression coefficients for each life stage to determine possible treatment effects. Mean sums of errors for deviations about the linear regressions for copepodites (0.04) and adults (0.03) were similar. No significant difference between treatments was determined in the statistical comparison of regression coefficients for each life stage.

*Eurytemora affinis*, the dominant species, which represented 73% of the bioassay population, experienced a mortality pattern similar to that of the total population (Table 1).

### February Bioassay

The bioassay conducted February 13 and observed through March 1 was characterized by an initial ambient temperature of 2.1°C. Maximum test temperatures were 8.2 and 17.2°C in the 1 and 2  $\Delta T$  bioassays, respectively. Ambient water temperature during the 16-day posttest observation period averaged 4.5°C and ranged from 1.1 to 8.1°C.

Results of the February bioassay are presented in Table 3 for each treatment. *Eurytemora affinis* instar analysis reflects the differential rates of population decrease apparent for each life stage. Mortality trends, as indicated by least-squares linear regression coefficients, were similar for each treatment population. The mortality rates for each treatment population were not significantly different from those observed for the intake control population. Mortality, although not significantly different statistically, was consistently higher in the discharge populations than in samples

TABLE 2 Regression and Correlation Coefficients Calculated by Least-Squares Linear Regression Analysis for January Bioassay Populations

| Life stage                | Intake control         |                         | Discharge 1 $\Delta T$ |                         | Discharge 2 $\Delta T$ |                         | Intake 1 $\Delta T$    |                         |
|---------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|
|                           | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient |
| Total copepodites         | 0.04                   | 0.79                    | 0.04                   | 0.81                    | -0.04                  | -0.87                   | -0.03                  | -0.89                   |
| Total adults              | 0.00                   | 0.12                    | 0.02                   | 0.92                    | 0.02                   | -0.75                   | -0.02                  | -0.80                   |
| <i>Eurytemora affinis</i> |                        |                         |                        |                         |                        |                         |                        |                         |
| Early copepodites         | 0.08                   | 0.92                    | 0.07                   | 0.88                    | -0.11                  | -0.95                   | -0.07                  | -0.85                   |
| Late copepodites          | 0.02                   | 0.52                    | 0.02                   | 0.38                    | 0.02                   | 0.71                    | 0.02                   | 0.50                    |
| Adults                    | 0.01                   | 0.24                    | 0.02                   | -0.59                   | -0.01                  | -0.47                   | -0.01                  | -0.51                   |

TABLE 3 Regression and Correlation Coefficients Calculated by Least-Squares Linear Regression Analysis for February Bioassay Populations

| Life stage                | Intake control         |                         | Discharge 1 $\Delta T$ |                         | Discharge 2 $\Delta T$ |                         | Intake 1 $\Delta T$    |                         |
|---------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|
|                           | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient |
| Total copepodites         | 0.12                   | 0.88                    | -0.16                  | -0.85                   | -0.18                  | -0.93                   | -0.13                  | -0.85                   |
| Total adults              | 0.00                   | 0.31                    | -0.05                  | 0.71                    | -0.06                  | -0.82                   | -0.03                  | -0.72                   |
| <i>Eurytemora affinis</i> |                        |                         |                        |                         |                        |                         |                        |                         |
| Early copepodites         | 0.13                   | 0.89                    | 0.16                   | -0.95                   | -0.16                  | -0.99                   | -0.14                  | -0.86                   |
| Late copepodites          | -0.10                  | -0.81                   | 0.07                   | -0.85                   | -0.12                  | -0.97                   | -0.10                  | -0.75                   |
| Adults                    | 0.05                   | 0.50                    | -0.10                  | -0.68                   | -0.09                  | -0.77                   | -0.06                  | -0.70                   |

of populations collected from the intake. In addition, mortality in the discharge 2  $\Delta T$  treatment was higher for each life stage than that observed in the discharge 1  $\Delta T$  population. We should also note that mortality rates observed from the February bioassays were, in general, higher than those observed during the previous months.

### March Bioassay

The bioassay conducted March 13 and observed through March 29 was characterized by an initial ambient temperature of 5.9°C. Maximum test temperatures were 11.8 and 21.6°C in the 1 and 2  $\Delta T$  bioassays, respectively. Ambient water temperatures during the 18-day posttest observation period averaged 8.3°C and ranged from 4.1 to 13.1°C.

Results of the March bioassay are presented in Table 4 for each treatment. The dominant generic group, *Eurytemora affinis*, represented 61% of the experimental population. Differential rates of population decline were apparent for copepodites and adults. Mortality trends were similar for each treatment population. Mortality, although not significantly different statistically, was consistently higher for experimental populations exposed to elevated water temperatures. Mortality rates observed from the March bioassays were higher than those observed during the previous three monthly bioassays.

### Nonthermal Bioassays

During the week of December 8, we had an opportunity to conduct a bioassay on zooplankton that had passed

through the cooling-water system; but had not been exposed to thermal elevations. Zooplankton samples were collected from the intake and discharge on December 12 and were observed through December 29. Ambient water temperature during the 17-day observation period averaged 5.4°C and ranged from 4.1 to 6.5°C.

Results of the 76- $\mu m$  bioassays are presented in Table 5 for each treatment and life stage of *Eurytemora affinis*, the dominant generic group in the bioassay population (82%). Differential rates of population decrease were apparent for each life stage, being highest for copepodites and less for adult organisms. Mortality trends were similar within and between treatment populations. No statistically significant difference between the intake and discharge treatment populations was determined in the comparison of regression coefficients.

### Distribution Studies

Distribution data were compiled on 28 taxonomic groups collected during the course of the studies. Major taxa identified were consistent with those reported by Gore (1973) and Wilson (1932), except for the occurrence of two subdominant harpacticoids, *Tachidius discipes* and *Paronychocamptus huntsmani*.

*Acartia tonsa* and *Eurytemora affinis* were the most persistent and abundant crustacean zooplankton observed in the power-plant receiving waters during the distribution studies. *Acartia tonsa* dominated the August, October, and November samples, comprising 68% of the zooplankton population. During December, January, and February,



TABLE 4 Regression and Correlation Coefficients Calculated by Least-Squares Linear Regression Analysis for March Bioassay Populations

| Life stage                | Intake control         |                         | Discharge 1 $\Delta T$ |                         | Discharge 2 $\Delta T$ |                         | Intake 1 $\Delta T$    |                         |
|---------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|------------------------|-------------------------|
|                           | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient |
| Total copepodites         | -0.10                  | 0.90                    | 0.17                   | 0.87                    | 0.12                   | 0.81                    | 0.17                   | 0.96                    |
| Total adults              | 0.08                   | 0.73                    | 0.11                   | 0.87                    | 0.09                   | 0.80                    | 0.08                   | 0.66                    |
| <i>Eurytemora affinis</i> |                        |                         |                        |                         |                        |                         |                        |                         |
| Copepodites               | 0.21                   | 0.87                    | 0.22                   | 0.84                    | 0.17                   | 0.66                    | 0.23                   | 0.87                    |
| Adults                    | 0.07                   | 0.73                    | 0.15                   | 0.93                    | 0.15                   | 0.84                    | 0.13                   | 0.73                    |

TABLE 5 Regression and Correlation Coefficients Calculated by Least-Squares Linear Regression Analysis for December Nonthermal Bioassay Populations

| Life stage                | Intake                 |                         | Discharge              |                         |
|---------------------------|------------------------|-------------------------|------------------------|-------------------------|
|                           | Regression coefficient | Correlation coefficient | Regression coefficient | Correlation coefficient |
| Total copepodites         | 0.03                   | 0.90                    | 0.08                   | 0.83                    |
| Total adults              | 0.01                   | 0.11                    | 0.09                   | 0.65                    |
| <i>Eurytemora affinis</i> |                        |                         |                        |                         |
| Copepodites               | 0.04                   | 0.44                    | 0.06                   | 0.72                    |
| Adults                    | 0.02                   | 0.19                    | 0.03                   | 0.34                    |

however, *Eurytemora affinis* comprised 80% of the zooplankton population. Age structure of the population sampled with the submersible pump was dominated during the entire study period by nauplii (59%), with copepodite and adult life stages representing 23 and 18% of the population, respectively.

Zooplankton abundance was observed to vary greatly between sampling dates. Population density was highest during February collections and lowest in October. The spatial distribution of zooplankton was variable and patchy, both horizontally and vertically in the water column (Davies, unpublished data).

Combined samples from different dates (Table 6) were used in a one-way analysis of variance to test statistically for differences between stations. The analysis of variance, performed separately on data from each sampling technique, indicated that zooplankton densities were not significantly different between the intake, discharge, and offshore stations. In addition, only slight differences were observed in species percent composition and age structure for populations sampled from each station.

## DISCUSSION

It has long been recognized that planktonic organisms are particularly susceptible to hydraulic entrainment and subsequent exposure to pressure, temperature, and mechanical stresses during passage through the condenser-cooling system of a power plant. To date, however, entrainment research on zooplankton has commonly centered around studies of immediate mortality. Only

recently have attempts been made to assess delayed mortality and sublethal effects of entrainment on estuarine zooplankton. Hence, this research was as much the development of bioassay methodology as the search for data on delayed effects of entrainment on zooplankton.

A comparison of survival of zooplankton collected by a submersible pump and by a standard plankton net showed similar survival for organisms collected by either method. Net collection techniques proved to be generally unacceptable, however, because of great differences in velocity between the intake and discharge water. Therefore the submersible pump was used to collect the 76- $\mu$ m bioassays reported here.

High zooplankton densities within the bioassay containers were necessary because of the demands of periodic subsampling and enumeration. Although desirable, a reduction in densities could not be attained because of the physical limitations of the experimental design.

Recommendations for improving the experimental methodology in future research center on reducing physiological stresses imposed on the captive populations. A shorter sampling time has been suggested to avoid overcrowding during collection. Experimental stress could be further reduced by decreasing population densities through serial culture techniques. Clarke (1937) recommended an experimental population density of 50 copepods/liter in experimental cultures. The retarding effect of starvation, which is a result of high population densities, would also be reduced substantially by decreasing densities. Food availability within each bioassay container could be artificially

TABLE 6 Summary of Distribution Data Showing Organisms/Liter Collected Monthly at Each Station, Grouped by Collection Technique

| Sample date | Collection technique |           |       |                    |           |       |                   |           |      |
|-------------|----------------------|-----------|-------|--------------------|-----------|-------|-------------------|-----------|------|
|             | Pump (73 $\mu$ m)    |           |       | Net (153 $\mu$ m)  |           |       | Net (505 $\mu$ m) |           |      |
|             | Intake               | Discharge | CM1*  | Intake             | Discharge | CM1*  | Intake            | Discharge | CM1* |
| August      | 21.13                | 23.84     | 11.77 |                    |           |       | 0.00              | 0.00      | 0.00 |
| October     | 0.47                 | 1.32      | 1.17  | 0.00               | 0.00      | 0.00  | 0.00              | 0.00      | 0.00 |
| November    | 4.32                 | 0.73      | 2.30  | 0.41               | 2.79      | 0.81  |                   |           |      |
| December    | 10.78                | 6.03      | 1.91  | 2.70               | 0.36      | 0.03  |                   |           |      |
| January     | 16.34                | 14.71     | 15.53 | 9.57               | 9.09      | 9.07  | 0.03              | 0.12      | 0.07 |
| February    | 60.20                | 58.84     | 47.40 | 16.70              | 13.04     | 18.09 | 2.38              | 0.25      | 0.20 |
| Mean        | 18.87                | 17.58     | 13.35 | 5.88               | 5.06      | 5.60  | 0.61              | 0.09      | 0.06 |
| F           |                      | 0.12      |       |                    | 0.02      |       |                   | 0.77      |      |
|             | F(2,15)0.05 = 3.68   |           |       | F(2,12)0.05 = 3.88 |           |       | F(2,9)0.05 = 4.26 |           |      |

\*CM1 = channel marker number 1.

enhanced by adding diatom cultures (Galtsoff, 1937; Mullin and Brooks, 1967). Copepods held under crowded conditions were observed to die off rapidly unless stirring was provided (Clarke, 1937). Our experimental containers were flushed daily but more effective aeration and a continual slow circulation of water could have been achieved by bubbling air into each bioassay container. Aeration of the experimental containers would also eliminate the need to provide flow-through conditions and thus eliminate the possibility of zooplankton escapement.

Microscopic examination of live samples collected from bioassay containers clearly showed intensive bacterial action. Several investigators have suggested the use of antibiotics such as streptomycin sulfate and penicillin G. Mullin and Brooks (1967) found that such treatment did not appear to improve copepod survival, however. In this study antibiotic prophylaxis was not used to avoid interference of bacterial decomposition.

Patterns of age-specific mortality were evident within each experimental population. Mortality rates were greatest for early copepodites and progressively less for late copepodites and adults. We could not resolve whether the observed mortality rates were the result of natural mortality or mortality induced by collection, experimental confinement at abnormally high densities, starvation, age-specific cannibalism or predation, development into the next life stage and perhaps lack of reproductive activity, or decline in fecundity per female. In all likelihood, a combination of physical and biological factors influenced the observed mortality patterns.

Small organisms, such as copepods, which mature rapidly and lay both large and frequent clutches, generally produce young that grow and develop rapidly but are subject to high juvenile mortality. Heinle (1966) found that for *Acartia tonsa*, a calanoid copepod, instantaneous death rates were greatest for nauplii and less for copepodites. Corresponding turnover times estimated for summer populations were 1.94 days for nauplii and 2.54 days for copepodids; this is indicative of age-specific natural mor-

tality (Heinle, 1966). Adult copepods, however, have been shown to live for long periods of time (Zillioux and Wilson, 1966; Mullin and Brooks, 1967).

A high incidence of mortality was observed during the molting process in several laboratory cultures of marine copepods (Mullin and Brooks, 1967; Raymont and Miller, 1962). Whether the higher mortality rates observed for early and late copepodites compared with those for adults were the result of intramolt mortality in these studies cannot be resolved, however.

Recruitment of early life stages into the succeeding developmental phases would account for the relatively higher regression coefficients calculated for the early copepodite life stage. However, from available data we can only speculate on the growth rates of individuals within our experimental populations. Heinle (1966), observing the developmental rates of *Acartia tonsa*, a subdominant calanoid copepod in our experimental populations, estimated a 13-day development time (egg to egg) at 13°C. Zillioux and Wilson (1966) estimated a 21-day development time at 16°C for *Acartia* with a limited diet. The retarding effect of starvation as a result of high population densities has also been shown to influence the rate of development within experimental populations (Heinle, 1966). It seems likely, in perspective with these factors, that little if any recruitment of early copepodite stages can be expected within our experimental populations incubated at temperatures ranging from 2 to 7°C.

It is notable that statistical comparison of the thermal bioassay data failed to identify significant differences between experimental treatments. Mortality estimates were similar for organisms collected from the intake and powerplant discharge. In addition, mortality rates in the monthly bioassays were similar.

Similar results were reported by Icanberry and Adams (1974), who observed no significant mortality difference in adult copepods collected from the intake and discharge of the Potrero Power Plant near San Francisco, Calif. Delayed mortality was observed at 6-hr intervals over a 24-hr period,

June 28 and 29. In contrast, Carpenter, Peck, and Anderson (1974) observed that 70% of the copepods collected in the discharge of the Millstone Nuclear Power Station on Long Island Sound (Waterford, Conn.) had died 5 days after entrainment. Only 10% mortality was observed for copepods collected in the intake during the spring. Copepod mortality was primarily attributed to mechanical and hydraulic stresses encountered during entrainment.

## CONCLUSIONS

Interpretation of bioassay results with regard to assessing a possible entrainment effect has centered on estimations of adult survival. As previously discussed, adult survival appears to be independent of population density, unaffected by intramolt mortality, and probably influenced little by predation within the experimental population. In addition, we wanted to compare our findings with other zooplankton entrainment research, which, to date, has been conducted exclusively on adult zooplankton.

Analysis of adult survival in our experimental system indicated that there was no discernible effect of mechanical or thermal stress as a result of entrainment during December, January, February, or March at the Crane Power Plant. We must observe, however, that during the period of experimental observations the discharge temperature was relatively low compared with the acute lethal temperature for estuarine zooplankton (Lauer et al., 1974). This reduction in thermal stress may account, at least in part, for the observed similarities in zooplankton mortality patterns among treatment populations.

Results of the December nonthermal bioassay showed no significant increase in zooplankton mortality as a result of mechanical damage caused by entrainment in the Crane Power Plant cooling system. These results do not substantiate the hypothesis presented by Carpenter, Peck, and Anderson (1974), Marcy (1974), Miller and Beck (1974), and others, that mechanical stress and not thermal elevation is the primary source of entrainment damage. Miller and Beck (1974) cited unpublished data demonstrating that mechanical damage caused 60% mortality of entrained phytoplankton and zooplankton, with the primary contributor being pumping effects. These investigators also observed an average mortality of 10% before the organisms passed through the circulating water pumps, 50% mortality after passage through the pumps, and 60% after passage through the condenser.

Results of the distribution studies further support the interpretation given the thermal bioassay experiments. In general, no discernible differences were observed in the abundance, species composition, and age structure of zooplankton populations sampled at the power-plant intake or discharge or the offshore station during the 6-month study period. From these data we can conclude either that entrainment and subsequent passage through the power-plant cooling system have little effect on zooplankton survival in the discharge and mixing area of the estuary or that zooplankton populations inhabiting the discharge

receiving waters have a recovery potential capable of compensating for entrainment mortality. In reality, both mechanisms probably contribute to the trophic stability of zooplankton populations in the area.

An additional indicator of the stability of local zooplankton populations inhabiting the receiving-water ecosystem is the high abundance of endemic and migratory fishes. The power-plant receiving waters serve as an important spawning ground for commercially and ecologically important fishes, many of which feed directly on zooplankton.

These data suggest that microzooplankton survival after exposure to condenser cooling-system effects is relatively high for the copepod species and population structures tested in these studies. Since verification of latent effects of entrainment damage is difficult if not impossible to evaluate under open-water field conditions, our studies and techniques represent an attempt to simulate some natural thermal reductions that pump-entrained copepods might experience in downstream mixing areas of estuarine water used for once-through cooling. Observations of pump-entrained copepods suggested that growth continued, as evidenced by molting, and that other biological stresses, such as increased susceptibility to disease, were not conspicuously evident. Further attempts to refine evaluations of macrozooplankton, meroplankton, and ichthyoplankton organisms appear to be warranted.

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# Assessment of Power-Plant Effects on Zooplankton in the Near Field

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## ABSTRACT

With the use of a simple entrainment model that takes into account tidal recirculation, mixing, and once-through kill rate, theoretical expressions are derived for ratios of densities of live organisms at the intake of a power plant to densities in the far field. Experimental data on the abundances of dead and live stages of *Eurytemora affinis* and *Acartia tonsa* are used to compare to model results. A study of the entrainment of zooplankton near the intake of the Morgantown power plant and a concurrent study of the distribution of zooplankton in the Potomac River provided the biological data. Physical information was derived from extensive hydrographic measurements. Comparing experimental data with model predictions showed that radical depletions occurred in the naupliar stages near the plant site which cannot be accounted for by cooling system or delayed entrainment mortalities. These near-field changes are attributed to avoidance reactions and to mortalities in the vicinity of the plant.

The Morgantown power plant uses ~2200 cfs of estuarine Potomac River water for cooling. The cooling water is drawn into the plant from underneath a skimmer wall at depths of 30 to 45 ft. and the thermal discharge is ejected upstream of the intake as a surface jet. Because of the tidal nature of the receiving body, some water that has passed through the cooling system has a positive probability of remaining in the plant vicinity for several tidal cycles. Depending on the buoyancy properties of the heated effluent, discharged water may be recirculated into the cooling system by two mechanisms: (1) mixing by turbulent dispersion, which tends to build up an elevated thermal field near the intake on a tidally averaged, steady-state basis, and (2) direct recirculation of discharged water during a certain phase of the tide; i.e., returning some fraction of the effluent into the intake during a single tidal cycle.

The existence of these mechanisms and their particular magnitudes depend on turbulence, on the vertical stability of the water column, on net nontidal flows that renew the water near the plant by advection, and on the plant's thermal rise. With the curtain-wall configuration, the

deeper, high-salinity intake water could produce sinking plumes if the effluent density is not offset (made buoyant) by the condenser temperature rise. Depending on water-column stratification, therefore, plume behavior may be buoyant or sinking. Sinking plumes tend to increase the probability of recirculation with a deep-water intake and upstream discharge configuration.

The entrainment probability and mortality of zooplankton depend on interactions of river dynamics with plant processes and on plankton behavior. If significant amounts of discharged water remain in the intake area, either by mixing or recirculation, zooplankton density with respect to river areas not under plant influence may decrease. The depletion effect on live zooplankton densities in the near field of the intake depends not only on the amount of recirculation but also on the kill rate during a single passage through the cooling system.

This paper examines zooplankton depletion effects at the intake with respect to areas outside the plant's influence and attempts to resolve whether depletions can be accounted for by entrainment damage alone. We constructed a model that takes into account once-through kill rates, recirculation, and mixing to predict depletions due to entrainment only which should be observed in the vicinity of the intake. Parameters in the model are tuned by hydrographic and temperature measurements. The model predictions are then compared to results obtained from field experiments monitoring live zooplankton densities near the intake and at far-field river stations.

## HYDROGRAPHY

Current measurements were part of the entrainment study in the spring of 1973. At intervals across the river, current meters were strung at various depths along a perpendicular transect near the plant site. Analysis of these measurements showed that the velocity field of the transect was dominated by a semidiurnal tidal component (an ~12.5-hr period) with a 25-hr diurnal variation and minor

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$P'_M$ , probability of mortality weighted for the entire cross section of the river.

$P'_{EN}$ , probability of entrainment weighted for the entire cross section of the river.

$A$ , entire river cross-sectional area.

$A_{LA}$ , cross-sectional area of affected lower-layer region.

When we consider the initial and final stages of the oscillating and advective water mass over a long period of time, the states become dependent only on nontidal flows. Applying mass conservation, we find that the ratio of live organisms in the intermediate field (before combination with recirculated water) to those in the far field is

$$\frac{C_n}{C_i} = \frac{Q_L}{Q_P} \left[ \frac{1 - \nu(1 - \gamma)}{\gamma(1 - \nu)} \right] \times \left( 1 - \exp \left\{ - \frac{Q_P}{Q_L} \left[ \frac{\gamma(1 - \nu)}{1 - \nu(1 - \gamma)} \right] \right\} \right) \quad (1)$$

The ratio of live densities at the intake to those at far field is

$$\frac{C_P}{C_i} = \frac{Q_L}{Q_P} \left[ \gamma \right] \left( 1 - \exp \left\{ - \frac{Q_P}{Q_L} \left[ \frac{\gamma(1 - \nu)}{1 - \nu(1 - \gamma)} \right] \right\} \right) \quad (2)$$

Computation of the ratio between outfall and intake shows consistency with definition:

$$\frac{C_o}{C_P} = 1 - \gamma \quad (3)$$

The probability of mortality, or the fraction of organisms killed passing by the plant in the affected area over all time, is

$$P_M = 1 - \exp \left\{ - \frac{Q_P}{Q_L} \left[ \frac{\gamma(1 - \nu)}{1 - \nu(1 - \gamma)} \right] \right\} \quad (4)$$

The probability of entrainment, or the fraction of available water from the affected area passing through the plant over all time, is

$$P'_{EN} = 1 - \exp \left[ - \frac{Q_P}{Q_L} (1 - \nu) \right] \quad (5)$$

Both expressions 4 and 5 can be weighted for the entire river cross section:

$$P'_{EN} = P_{EN} \frac{A_{LA}}{A} \quad (6)$$

and

$$P'_M = P_M \frac{A_{LA}}{A} \quad (7)$$

The physical configuration for which model expressions have been derived is represented in Fig. 1. These are aerial (Fig. 1a) and cross-sectional (Fig. 1b) views of the river regions modeled. Specified are the far field, in which plant influence is not detectable; the intermediate field, in which plant effects are dispersed over a time scale longer than the

tidal period; and the near or recirculation region, in which a fraction of the discharge water is returned within a tidal period.

To apply the model to the Morgantown site, we must estimate the parameters  $Q_P/Q_L$ ,  $\nu$ , and  $\gamma$ . After estimating the proper ratio  $Q_P/Q_L$ , we can compute the size of the affected cross-sectional area from the net-flow field obtained from current-meter data. Both  $Q_P/Q_L$  and  $\nu$  can be estimated from intake and river thermal data. Information on once-through kill rates is available from the entrainment study.

Figure 2 presents the temperature records in the river channel (2700 ft from the curtain wall) at depths of 4, 26, and 49 ft and the temperature of the intake at a depth of 40 ft. We can conclude from these patterns that intake temperature does not depend on river temperature at the same depth but results from long-term heating of the water near the plant (a steady component), upon which tidally controlled pulses caused by direct recirculation are superimposed. The suggestion of coincident fluctuations at the intake and in the river on May 10, 1973, does not necessarily mean that river temperatures rise and fall as a result of plant action; it may mean instead that tidal processes control both temperature fluctuations of the river and direct recirculation.

The condensers in the plant cooling system produce a temperature rise of  $\Delta T_c = 5.56^\circ\text{C}$ . By using an appropriate model of thermal addition consistent with the entrainment model, we should be able to arrive at the steady value of long-term temperature rise observed in Fig. 2. Denoting this rise as  $\Delta T_R$ , we find that a reasonable estimate from Fig. 2 is  $\Delta T_R \approx 2^\circ\text{C}$ , or 36% of  $\Delta T_c$ . If we assume that direct recirculation occurs over only one half of a tidal cycle, the  $1^\circ\text{C}$  peaks superimposed on the steady value suggest an estimate of  $\nu \approx 0.1$ , averaged over a whole tidal cycle, or 10% direct recirculation.

A thermal addition model consistent with this entrainment model requires that we neglect cooling by atmospheric exchange and turbulent diffusion. The heat balance for the whole intermediate-field region can be written as:

$$(T_o - T_i)Q_L = \Delta T_c Q_P \quad (8)$$

Identifying the difference between the temperature of water masses entering the intermediate region and that of water advected out of this region as  $\Delta T_R$ , we find that

$$\frac{Q_P}{Q_L} = \frac{\Delta T_c}{\Delta T_R} \approx 0.36 \quad (9)$$

This value of the ratio between dilution and plant flows is only 32% of that estimated by Carter (1973). Since this estimate is based on measured temperature rise at the intake, however, it really results from all the advective and turbulent processes acting on the water masses. By computing the value of  $Q_P/Q_L$  in this manner, we are effectively taking into account turbulent processes in terms of a fictitious advection. This procedure seems reasonable

harmonic and subharmonic components. Analysis of time series revealed irregularities in basic tidal periodicity which were attributable either to instrumental timing errors or to the irregular nature of currents in the area.

The tidal dynamics induce the mixing and recirculation of discharged water. The residence time of effluents near the intake and, therefore, the magnitude of mixing partially depend on the advective flushing rate. Water renewal in the area is controlled by nontidal flows originating in freshwater input and estuarine gravitational convection. To determine the structure of longitudinal nontidal currents, we averaged the components of velocity vectors perpendicular to the transect axis over time. The mean flow structure was found to be typical of a partially mixed estuary—downstream drift in the upper layer and upstream flow in the lower layer. The nontidal velocity in the top layer can be as much as 40% of the peak in the tidal signal or as little as 20% of the peak in the lower layer.

Integration of the longitudinal velocity field yields the following values for nontidal transport:

| Cross-sectional area   | Transport   |
|------------------------|-------------|
| Upper 126,800 sq ft    | -27,600 cfs |
| Lower 59,300 sq ft     | + 9,100 cfs |
| Transect 186,100 sq ft | -18,500 cfs |

Negative values of transport indicate downstream flow. The vertical boundary between layers was found to be between 25 and 30 ft. Because of the deep intake, the source of the 2232-cfs cooling stream is the flow in the lower layer. Since the river is over 2 miles wide at the site, only a portion of the lower-layer transport is involved in cooling the condensers.

Intake-velocity and density-profile measurements made by Obremski (1974) showed that, given vertical salinity differences of 0.1‰, the curtain-wall intake structure will selectively withdraw water from depths below 30 ft. Since this condition persists in almost all seasons, the lower layer, defined by upstream transport, can be considered the source for cooling water. Dye studies conducted over a wide region encompassing the plant site (Carter, 1973) indicated that less than 30% of the 186,000-sq ft cross section of the river is affected measurably by plant circulation. Carter also calculated that the 10°F thermal rise produced by the plant, in combination with lower layer withdrawal, will result in a sinking thermal plume during most of the year. Thus, in modeling depletion of zooplankton densities near the plant, we can consider the lower layer to be both the source for cooling water and the sink for effluent.

## MODELING

The model that estimates density differences of live zooplankton between points near the plant and unaffected areas in the river assumes the following:

1. Live zooplankton density differences between the intake and far-river stations are due solely to entrainment damage.

2. The plant draws and discharges water into the same nontidal layer (lower) flowing upstream.

3. Mixing of effluent waters, which appears as a persistent thermal elevation near the intake, is a result of turbulent dispersion.

4. Direct recirculation of thermal effluent occurs consistently during a fraction of each tidal cycle.

5. Complete cross-stream mixing of the effluent occurs only within a single layer (lower), whose width is a fraction of the width of the river.

These assumptions maximize depletion effects near the intake due to entrainment damage.

In deriving the model we consider a parcel of water oscillating several times past the plant intake over a long period of time, while it also drifts slowly away from the intake structure longitudinally by nontidal advective currents. The lateral dimension times the depth of the oscillating parcel is considered the cross-sectional area of the river affected by the plant; the longitudinal dimension is undetermined. If we allow recirculation to take place from outfall to intake within the same tidal cycle, we produce three characteristic regions with respect to planktonic densities. The far field, which is the source for all entrained organisms, is considered unaffected by plant operations. The near field is the region in which water recirculated directly from the outfall combines with water of intermediate physical and biotic density which has been modified by plant action and mixing over a long period of time. Concentrations of organisms are depleted in the intermediate and near fields to differing extents compared with undisturbed values in the far field.

The following definitions are used in deriving the model:

- $C_p$ , concentration of live organisms at the plant intake.
- $C_i$ , concentration of live organisms measured at locations far away from the plant; used here as the concentration that would exist in the absence of plant influence.
- $C_d$ , concentration of live organisms observable at the plant discharge.
- $C_n$ , concentration of live organisms observed in the intermediate field, i.e., before recirculated water masses combine with ambient (intermediate-field) masses in front of the intake structure.
- $\nu$ , fraction of plant discharge recirculated within a tidal cycle.
- $\gamma$ , fraction of organisms killed by plant action on a single pass through the cooling system.
- $Q_L$ , nontidal lower-layer flow in the affected cross section of the river.
- $Q_p$ , plant circulation.
- $P_M$ , probability of mortality of a single organism passing through the region of plant influence, or the fraction of all organisms killed by plant action in the affected region.
- $P_{EN}$ , probability of entrainment of a water parcel in passage through the affected region.

of temperature rise, we have implicitly accounted for these processes. The remaining process that increases organism residence times is the drawing of water from the interface layer, between upstream and downstream flows, where nontidal velocities approach zero. To maintain an increased depletion, however, the plume is required also to mix back into this layer. Using dye dispersion studies, Carter (1973) showed that there is considerable vertical mixing; however, it is not likely that most of the discharge water would consistently end up in the velocity interface region.

After examining the consequences of inplant kill rate and physical mechanisms on depletion, we concluded that the observed high depletions for nauplii reported in Tables 2 through 4 cannot, in general, be attributed to physical mechanisms, high inplant kills, or delayed mortalities. Alternative explanations must be sought on biological grounds, e.g., sensitivity to near-field effects produced by thermal and chemical discharges.

The different sensitivities of various life-history stages require that rigorous modeling of the inplant and field effects be coupled from one life-history stage to another. Generation and developmental times of zooplankton are close to the residence time of water subject to entrainment. Such coupling by population dynamics could explain only significant adult depletions when both earlier stages simultaneously show some significant depletions. However, high naupliar depletions occur even when the adult population is unaffected at the 95% level (spring, postchlorination) and when copepodite densities are enriched at the intake in comparison with all far-field values. It seems that thermal build-up in near-field waters and/or plume entrainment are sufficient to cause high naupliar depletions outside the plant cooling system.

The two major types of mechanisms responsible for depletions greater than those expected from entrainment mortality alone are near-field mortalities of unentrained organisms and avoidance by vertical redistribution. If mortalities do occur in the near field, mixing and population growth act to counter depletions. Thus we would be forced to conclude that, in the time it takes to reach the adult stage, mixing alone compensates for near-field mortalities in adults since nauplii are radically depleted. This leads immediately to a contradiction, however, because, even when adults and copepodites are not depleted, nauplii are not restored by mixing. Therefore we have to consider both the mortality and the avoidance mechanisms, together with increased predation on affected organisms.

The literature contains evidence for near-field depletion of zooplankton. Churchill and Wojtalik (1969), Arnold (1972), and Carpenter, Peck, and Anderson (1974) showed lowered zooplankton population densities in the vicinity of operating facilities. Estuarine circulation patterns provide a basis for believing that the effective exposure times of unentrained organisms to the excess temperatures of the mixing zone is close to a tidal period. If we assume classic response mechanisms to time, temperature, and dose (Coutant, 1970), mixing-zone exposure effects may out-

weigh inplant entrainment mortalities in producing near-field depletions. Of course, the sensitivities of various organisms and their life-history stages differ.

An apparent avoidance mechanism might result from immediate and long-term recirculation of the effluent. Carpenter, Peck, and Anderson (1974) showed that the sinking rate of stressed copepods is two to three times the rate observed for animals without a stress history. Thus stressed organisms (organisms previously unentrained but that have been mixed into recirculating discharge water) may have a tendency to accumulate in layers below the intake depth. Since these layers were not sampled, an apparent depletion would result. Significant vertical avoidance responses to temperature increases have also been noted by Gehrs (1974b).

Organisms subjected to thermal and chemical discharges may display reduced reproductive capacities; this would immediately reduce the abundance of nauplii. Mihursky (1966) and Heinle (1969) identified thermal and chlorine stresses as the principal causes of loss in reproductive potential. In fact, the reduction of nauplii may originate in the exposure of eggs attached to gravid females.

Seemingly sublethal elevations in temperature in the mixing zone may cause significant reductions in populations when temperatures rise near the upper natural limit for a given species. Since the measured depletions in early stages of *E. affinis* and *A. tonsa* occurred in late spring and summer, respectively, we may be observing an acceleration in the natural succession pattern near the plant.

Chlorine addition to the cooling water dramatically increases inplant copepod mortalities, especially in nauplii and copepodite stages (Bongers, Polgar, and Krainak, 1976). The effects of chlorine residuals on survival in the early life-history stages and on the reproductive success of adults might also contribute to reduced abundances in the mixing zone. The 48-hr median tolerance limit values for *A. tonsa* were reported to be less than 5 ppb by Roberts et al. (1975). Brungs (1973) indicated that safe values for total residual chlorine should be 2 ppb or less for the protection of most aquatic organisms, and Gehrs (1974a) found the hatchability of carp eggs to be significantly reduced after exposure to chlorine levels as low as 1 ppb. Harmful levels of chlorine residuals (estimated to be in the range of 1 to 10 ppb) might have been present in the mixing-zone area when chlorine was applied to the cooling water. In nauplii, however, no significant increases were observed in near-field depletion with chlorination. Our data are clearly not adequate to evaluate chlorine effects in the near field, but inplant measurements did show significant increases in naupliar and copepodite mortalities with chlorination.

From the model calculation using 100% once-through kill, we estimated that the plant impacts 2% of the zooplankton population by entrainment damage. Large intake depletions of nauplii showed apparent effects in excess of those expected from entrainment alone. Older zooplankton stages were present in low numbers, and no



TABLE 3 Near-Field to Far-Field Ratios of Live Organisms\*†

| Experimental condition             | Developmental stage |                |                 |                 |                 |                 |
|------------------------------------|---------------------|----------------|-----------------|-----------------|-----------------|-----------------|
|                                    | Adults              |                | Copepodites     |                 | Nauplii         |                 |
|                                    | S2'/St.2            | S2'/All        | S2'/St.2        | S2'/All         | S2'/St.2        | S2'/All         |
| <i>Eurytemora affinis</i> (Spring) |                     |                |                 |                 |                 |                 |
| Prechlorination                    | 1.07<br>(NS)        | 0.80<br>(NS)   | 1.30<br>(NS)    | 0.75<br>(NS)    | 0.07<br>(<0.01) | 0.74<br>(<0.01) |
| Chlorination                       | 0.96<br>(NS)        | 0.73<br>(NS)   | 0.47<br>(<0.01) | 0.66<br>(0.05)  | 0.09<br>(<0.01) | 0.18<br>(<0.01) |
| Postchlorination                   | 1.43<br>(NS)        | 1.53<br>(NS)   | 5.93<br>(<0.01) | 4.93<br>(<0.01) | 0.08<br>(<0.01) | 0.10<br>(<0.01) |
| <i>Acartia tonsa</i> (Summer)      |                     |                |                 |                 |                 |                 |
| Prechlorination                    | 0.39<br>(0.01)      | 0.64<br>(NS)   | 0.78<br>(NS)    | 0.62<br>(0.10)  | 0.28<br>(<0.01) | 0.29<br>(<0.01) |
| Chlorination                       | 1.27<br>(NS)        | 0.62<br>(0.10) | 0.23<br>(<0.01) | 0.32<br>(<0.01) | 0.08<br>(<0.01) | 0.13<br>(<0.01) |
| Postchlorination                   |                     |                |                 |                 |                 |                 |

\*Comparisons are intake (S2') to nearest station (St.2) and intake to all far-field stations combined (All). The ratio estimates are derived from the contrasts of ANOVA tests on log-transformed data.

†Numbers in parentheses indicate levels of significance different from 1.0 or whether depletion or increase is significant. NS = not significant.

TABLE 4 Near-Field to Far-Field Ratios of Live Organisms\*†

| Experimental condition             | Developmental stage |                 |                 |                 |                 |                 |
|------------------------------------|---------------------|-----------------|-----------------|-----------------|-----------------|-----------------|
|                                    | Adults              |                 | Copepodites     |                 | Nauplii         |                 |
|                                    | S2'/St.1            | S2'/St.3        | S2'/St.1        | S2'/St.3        | S2'/St.1        | S2'/St.3        |
| <i>Eurytemora affinis</i> (Spring) |                     |                 |                 |                 |                 |                 |
| Prechlorination                    | 0.48<br>(<0.01)     | 0.95<br>(NS)    | 0.33<br>(<0.01) | 0.99<br>(NS)    | 0.05<br>(<0.01) | 0.05<br>(<0.01) |
| Chlorination                       | 0.57<br>(<0.05)     | 0.69<br>(0.05)  | 0.74<br>(NS)    | 0.83<br>(NS)    | 0.37<br>(<0.01) | 0.17<br>(<0.01) |
| Postchlorination                   | 0.78<br>(NS)        | 0.34<br>(0.10)  | 3.93<br>(<0.01) | 1.66<br>(<0.01) | 0.11<br>(<0.01) | 0.11<br>(<0.01) |
| <i>Acartia tonsa</i> (Summer)      |                     |                 |                 |                 |                 |                 |
| Prechlorination                    | 1.51<br>(NS)        | 0.47<br>(<0.10) | 0.40<br>(<0.01) | 0.77<br>(NS)    | 0.26<br>(<0.01) | 0.32<br>(<0.01) |
| Chlorination                       | 0.47<br>(<0.01)     | 0.54<br>(0.10)  | 0.36<br>(<0.01) | 0.40<br>(0.01)  | 0.28<br>(<0.01) | 0.09<br>(<0.01) |
| Postchlorination                   |                     |                 |                 |                 |                 |                 |

\*Comparisons are intake (S2') to farthest river stations (St.1 and St.3). The ratio estimates are derived from the contrasts of ANOVA tests on log-transformed data.

†Numbers in parentheses indicate levels of significance different from 1.0 or whether depletion or increase is significant. NS = not significant.

on the residence time of organisms in the impact area and that this residence time is the same as that for water in the vicinity of the plant. Since vertical behavior is diurnal and current variation is tidally dominated, however, populations may maintain their longitudinal position in the estuary as a result of beat-frequency behavior in the longitudinal mo-

tion of organisms. This phenomenon tends to increase residence time occasionally and to magnify depletion.

Generally, vertical, lateral, and longitudinal turbulent mixing tend to decrease the apparent depletion effect by mixing in organisms from the far field. By tuning the model to the proper value of advective-to-plant flow on the basis

condition and life stage. The first method tried used the arithmetic average of densities in the 24 samples taken in front of the curtain wall to represent the intake value and the arithmetic average of densities in the 48 samples taken at station 2 in the river to represent the far-field value. The densities were computed as numbers per 20 liters. Since 19-liter samples were taken at the intake, the numbers in these samples were multiplied by the factor 20/19. The ratios of these averages are shown in Table 2. Depletions of up to 96% at the intake are indicated. However, since it is extremely difficult to determine whether these ratios are significantly different from 1, this method was abandoned in favor of computing the ratios using geometric means to allow computation of confidence levels.

TABLE 2 Averaged Near-Field (Area S2') to Far-Field (Station 2) Ratios of Live Organisms

| Experimental condition             | Developmental stage |             |         |
|------------------------------------|---------------------|-------------|---------|
|                                    | Adults              | Copepodites | Nauplii |
| <i>Eurytemora affinis</i> (Spring) |                     |             |         |
| Prechlorination                    | 0.72                | 0.45        | 0.04    |
| Chlorination                       | 0.32                | 0.26        | 0.06    |
| Postchlorination                   | 0.50                | 0.61        | 0.05    |
| <i>Acartia tonsa</i> (Summer)      |                     |             |         |
| Prechlorination                    | 0.17                | 0.63        | 0.29    |
| Chlorination                       | 0.79                | 0.20        | 0.10    |
| Postchlorination                   |                     |             |         |

Computing the ratio of geometric means is equivalent to calculating the difference of the means of logarithms of the numbers used to compute the means in the numerator and denominator of the ratio.

$$\ln \frac{(X_1 \cdot X_2 \cdot \dots \cdot X_n)^{1/n}}{(Y_1 \cdot Y_2 \cdot \dots \cdot Y_m)^{1/m}} \approx \frac{1}{n} \sum_{i=1}^n \ln X_i - \frac{1}{m} \sum_{j=1}^m \ln Y_j \quad (10)$$

The densities at each station were transformed using the natural logarithm. If a data point was zero, the transformation was defined as zero.

A one-way analysis of variance was performed to test for differences in means among stations. Then the difference of the mean of each of the three river stations from the mean of the intake (S2') was computed, along with the difference of the average of the means at the three river stations (All) and the mean at the intake (S2'). The ratios in Tables 3 and 4 were obtained by raising  $e$  to these differences, i.e., by untransforming the differences. The tables display these ratios for each experimental condition and developmental stage. A significance level in the table is

the probability that the difference of the means is zero (or equivalently that the ratio is 1) and that a value greater than or equal to the absolute value of the difference occurs. The live density ratios displayed in Table 3 represent comparisons of intake values with the closest far-field station (station 2) and with all three river far-field stations combined. Significance levels of the ratios different from 1 are also shown. Table 4 presents similar comparisons of the intake with each of the two farthest river stations, station 1 and station 3.

Model predictions based on measured physical data indicate that depletions at the intake should not exceed ~20% of far-field values if entrainment damage is responsible for all kills. This means that the experimentally measured ratios should not have values below 0.8. On the basis of significance levels of computed ratios, such damage should not be detectable from zooplankton measurements because values between 1.0 and 0.6 are not significantly different from 1.0, or 0% depletion. At least 40% depletion has to occur to be detectable.

Examining Table 3, we could conclude that nauplii are significantly depleted under all conditions with no significant change in depletion with chlorination. Copepodites are depleted by chlorination only and are, in one instance, enriched near the intake. Finally, adults are not significantly depleted under any of the experimental conditions.

Considering the comparisons made in Table 4 also, we must revise our interpretation. The only clear result is that very high depletions of nauplii occur near the intake under all conditions in comparison with all river far-field stations. The data for nauplii contained few zeros and also large numbers; this leads to consistency in comparing river stations to the intake. The data for adults and copepodites were weaker in these respects, resulting in the variation in estimates of the ratios from station to station. These observations demonstrate the need for samples of larger volumes, so that each sample contains non-zero observations.

## DISCUSSION AND CONCLUSIONS

Model calculations have shown that, regardless of the magnitude of the once-through kill rate, there should be no measurable near-field depletion. Therefore, because of entrainment damage alone, it seems that latent or delayed effects on organisms that have already passed through the cooling system cannot account for the observed depletion.

The behavior of the density ratios of live nauplii at the intake to the far field seems to be a consequence of near-field plant effects on organisms that have not been entrained. Since these near-field effects are not considered in the model, model calculations indicate no significant near-field depletions. The various assumptions made in model formulation must be reconsidered in light of the large near-field depletions; thus we are seeking explanations in neglected factors.

The assumption of uniform vertical distribution in the model implies that vertical behavior will not have an effect

TABLE 1 Calculations for the Tuned Model\* ( $Q_p/Q_L = 0.36$ )

| $\gamma$ | $\nu$ | $P_{EN}$ | $P_M$ | $P'_{EN}$ | $P'_M$ | $C_n/C_i$ | $C_p/C_i$ |
|----------|-------|----------|-------|-----------|--------|-----------|-----------|
| 0.1      | 0.0   | 0.302    | 0.035 | 0.021     | 0.003  | 0.982     | 0.982     |
| 0.3      | 0.0   | 0.302    | 0.102 | 0.021     | 0.007  | 0.948     | 0.948     |
| 0.5      | 0.0   | 0.302    | 0.165 | 0.021     | 0.012  | 0.915     | 0.915     |
| 0.8      | 0.0   | 0.302    | 0.250 | 0.021     | 0.018  | 0.869     | 0.869     |
| 1.0      | 0.0   | 0.302    | 0.302 | 0.021     | 0.021  | 0.840     | 0.840     |
| 0.1      | 0.1   | 0.277    | 0.035 | 0.020     | 0.002  | 0.982     | 0.972     |
| 0.2      | 0.1   | 0.277    | 0.068 | 0.020     | 0.005  | 0.966     | 0.945     |
| 0.3      | 0.1   | 0.277    | 0.099 | 0.020     | 0.007  | 0.950     | 0.919     |
| 0.4      | 0.1   | 0.277    | 0.129 | 0.020     | 0.009  | 0.934     | 0.894     |
| 0.5      | 0.1   | 0.277    | 0.157 | 0.020     | 0.011  | 0.919     | 0.871     |
| 0.6      | 0.1   | 0.277    | 0.183 | 0.020     | 0.013  | 0.905     | 0.849     |
| 0.7      | 0.1   | 0.277    | 0.208 | 0.020     | 0.015  | 0.892     | 0.827     |
| 0.8      | 0.1   | 0.277    | 0.232 | 0.020     | 0.016  | 0.879     | 0.807     |
| 0.9      | 0.1   | 0.277    | 0.255 | 0.020     | 0.018  | 0.866     | 0.787     |
| 1.0      | 0.1   | 0.277    | 0.277 | 0.020     | 0.020  | 0.854     | 0.769     |
| 0.1      | 0.3   | 0.223    | 0.034 | 0.016     | 0.002  | 0.983     | 0.943     |
| 0.3      | 0.3   | 0.223    | 0.091 | 0.016     | 0.006  | 0.954     | 0.845     |
| 0.5      | 0.3   | 0.223    | 0.138 | 0.016     | 0.010  | 0.929     | 0.765     |
| 0.8      | 0.3   | 0.223    | 0.193 | 0.016     | 0.014  | 0.900     | 0.670     |
| 1.0      | 0.3   | 0.223    | 0.223 | 0.016     | 0.016  | 0.884     | 0.619     |
| 0.1      | 0.6   | 0.134    | 0.031 | 0.010     | 0.002  | 0.985     | 0.856     |
| 0.3      | 0.6   | 0.134    | 0.072 | 0.010     | 0.005  | 0.964     | 0.665     |
| 0.5      | 0.6   | 0.134    | 0.098 | 0.010     | 0.007  | 0.950     | 0.543     |
| 0.8      | 0.6   | 0.134    | 0.123 | 0.010     | 0.009  | 0.937     | 0.426     |
| 1.0      | 0.6   | 0.134    | 0.134 | 0.010     | 0.010  | 0.931     | 0.373     |

\* $\gamma$  is once-through kill rate;  $\nu$  is the fraction of immediate recirculation;  $P_{EN}$  and  $P_M$  are probabilities of entrainment and mortality, respectively;  $P'_{EN}$  and  $P'_M$  are probabilities of entrainment and mortality, respectively, weighted for the entire river section;  $C_p$ ,  $C_n$ , and  $C_i$  are the densities of live organisms at the plant intake, the intermediate field and the far field, respectively.

Because conditions were changed at weekly intervals and because zooplankton patchiness effects were amplified by a lack of sampling stratification by tidal phase, analysis of variance (ANOVA) tests were not powerful enough to show plant effects on the river distributions. Despite this limitation, the vertical distribution of nauplii seemed to be significantly influenced by chlorination over a large segment of the river. The population variations over the 3-week measurement period and the large-scale patchiness in the river prevented a clear detection of a zooplankton density gradient induced by plant action.

Having derived an entrainment model that maximizes the apparent depletion of organisms near the plant intake structure in comparison with far-field densities of live organisms, we can now compare abundances of live organisms at the intake to those at the river stations to gain knowledge of possible population depletions due to plant operations. We have already tuned the model by analyzing the long-term and short-term thermal rise at the intake. From this analysis, we obtained values for two parameters, the ratio of plant flow to dilution flow and the direct fraction of discharge water recirculated. Table 1 presented

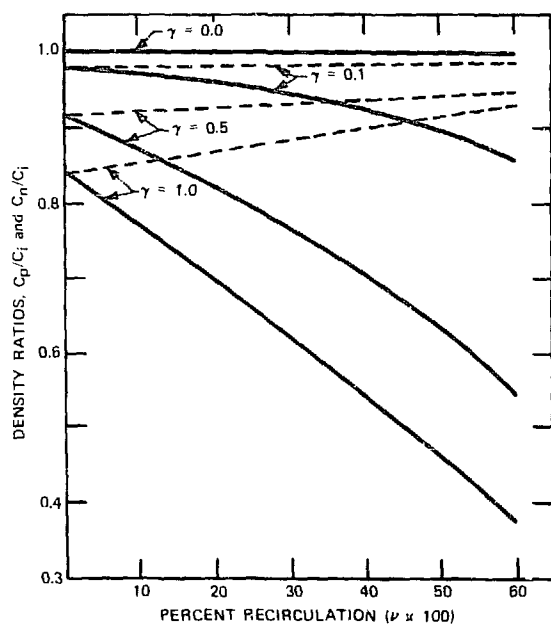


FIG. 3 Theoretical live-organism density ratios for  $Q_p/Q_L = 0.36$  and for intake (near field) vs. far field and intermediate field vs. far field at various values of once-through kill rate ( $\gamma$ ) and direct recirculation ( $\nu$ ). —,  $C_p/C_i$  at intake. ----,  $C_n/C_i$  in the intermediate field.

the results of model calculations over a range of once-through kill rates from 10 to 100%. On the basis of these calculations, we would expect to see a depletion of incoming live organisms in comparison with far-field values of ~20% averaged over a tidal cycle if all organisms passing through the plant were destroyed.

To compare this result with experimental data, we must estimate the ratios of live organisms in the intake to live organisms in the far field. Pooling the inplant data over the 24-hr study period averages the results for approximately two tidal cycles and also averages out diurnal movement effects. To arrive at a conservative estimate of depletion from the data, we computed the plankton densities in the river on the basis of average densities for day and night over the entire water column. This approach tends to minimize the depletion estimate because vertical distribution maxima occurring frequently at the intake depth are averaged out. When we average over the entire water column, we underestimate the river values to which intake values are then compared. As the results show, in many instances higher near-field depletions are found from the observational data than are predicted by the model, even though the method of depletion calculations tends to minimize plant effects.

The ratio of the density of live organisms at the intake to that in the far field was computed for each experimental

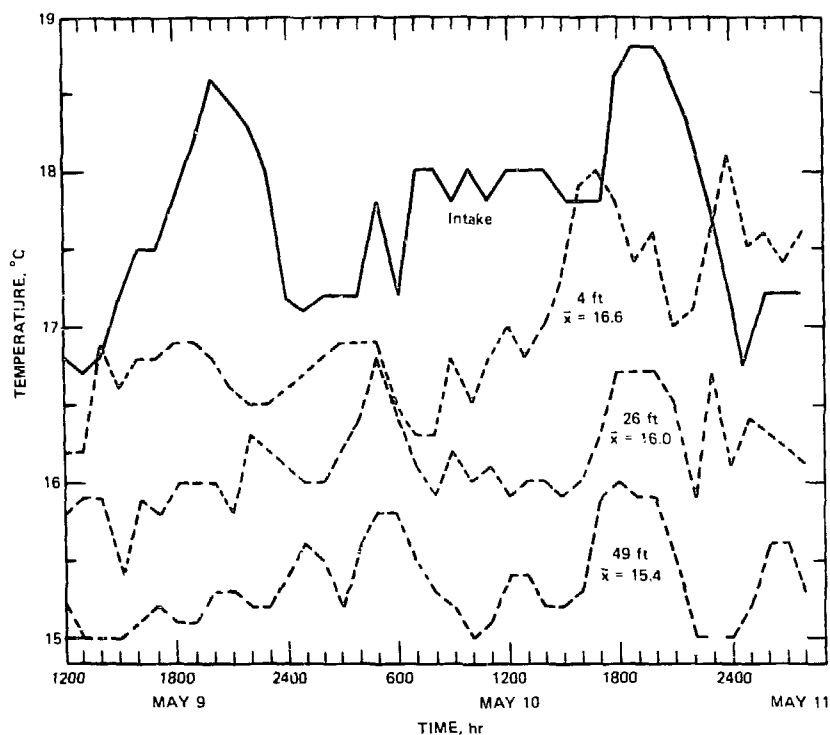


FIG. 2 Temperature histories in the river channel 3000 ft from the intake at depths of 4, 26, and 49 ft and at the intake at a depth of 40 ft.

## COMPARISONS OF MODEL RESULTS WITH EXPERIMENTAL DATA

### Intake Zooplankton Sampling

The inplant entrainment study involved sampling at various points along the cooling system. Here we are concerned only with samples taken in front of the curtain wall at the intake depth (area S2'). Samples were obtained by pumping triplicate 19-liter samples every 3 hr for a 24-hr period for three experimental conditions: prechlorination, chlorination (6000 lbs/unit/day), and postchlorination. After each condition was maintained for approximately a week, the 24-hr sampling period began. Each sample was stained immediately with a vital stain (Dressel, Heinle, and Grote, 1972). The samples were then concentrated down to a small volume, and the organisms were counted. Large numbers required subsampling for counts. Each zooplankton was categorized by species and developmental stage (adult, copepodite, or nauplius) and classified by use of the staining criterion as live or dead. The spring samples were dominated by *Eurytemora affinis*, and *Acartia tonsa* dominated summer collections. Subsequent calculations and statistics deal with the various stages and viability of *E. affinis* for spring results and *A. tonsa* for summer results.

### River Zooplankton Sampling

Concurrently with each 24-hr sampling period inplant, zooplankton collections were made in three river-channel areas. All three sampling locations were within a tidal excursion of the plant. Station 1 was approximately 2 miles upstream; station 2, in the river channel opposite the plant intake (approximately 3000 ft from the intake); and station 3, approximately 1.5 miles downstream of the plant site. Samples were taken twice a day, at noon and midnight, at each station. Three depths, 5, 25, and 50 ft, were sampled each time, with eight replicate samples of 20 liters each being pumped from each depth at each location.

The samples were stained, and counts were made from subsamples of 50-ml concentrates according to viability, species, and developmental stage.

### Results

Using the intake and river data, we can evaluate whether any effects of plant mortality are apparent in the river itself. An initial hypothesis was that plant effects are most likely to be detected at station 2, which is nearest the plant.

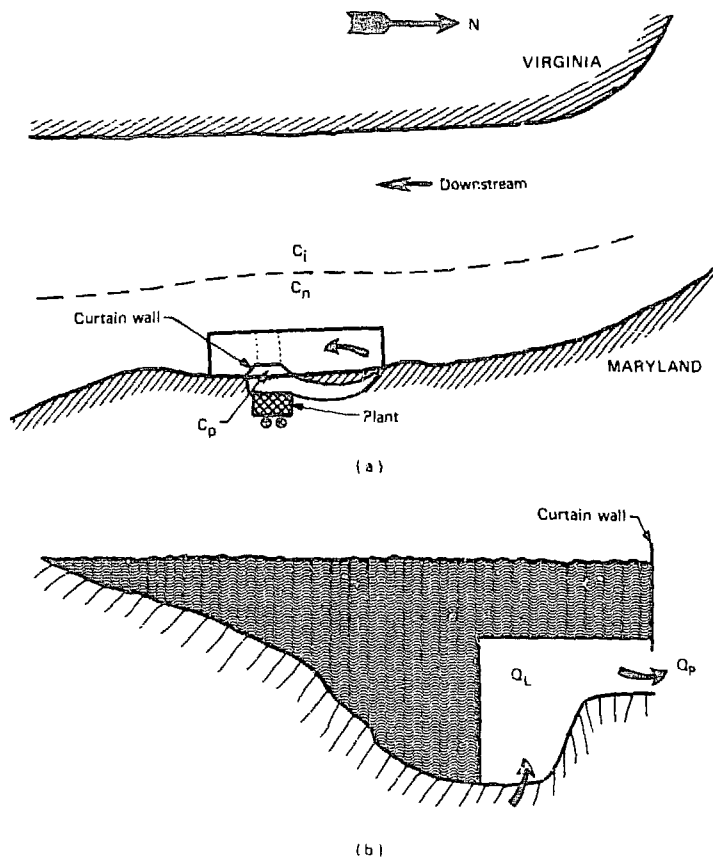


FIG. 1 (a) Aerial schematic representation of modeled plant and river regions.  $C_i$ , far-field live density.  $C_n$ , intermediate-field live density,  $C_p$ , near-field density. (b) Cross-sectional view of the river region near the plant in relation to the intake curtain wall.  $Q_L$ , nontidal transport of lower layer in upstream direction.  $Q_p$ , plant circulation.

since the difference between the processes can be considered one of spatial scale.

This value of the estimated ratio fixes the dilution flow at 6200 cfs, or approximately two-thirds of the net lower-layer transport. The region of plant influence, therefore, extends to roughly the midchannel, approximately 3000 ft from the curtain wall. The lower-layer cross-sectional area involved in plant circulation is 13,200 sq ft; in comparison, the total river cross-sectional area is 186,100 sq ft.

Table 1 presents model calculations based on the estimated ratio  $Q_p/Q_L = 0.36$  and on various parametric values of once-through kill rate and immediate recirculation rate within a tidal cycle. An immediate observation is that, with total once-through kill and 60% immediate recirculation, the number of live organisms at the intake would still be nearly 40% of the number in the far field.

The essential features of the model calculations presented graphically in Fig. 3 are that:

1. The live organisms at the intake decrease with respect to the far-field concentrations as immediate recirculation increases.
2. The greater the recirculation is, the less the intermediate field,  $C_n$ , is affected, even though live organism levels keep decreasing at the intake.
3. Probability of mortality for the entire river section decreases with increasing recirculation and decreasing once-through kill (Table 1).
4. On the basis of processes weighted for the entire cross section of the river, less than 2% of the planktonic populations transported past the plant are killed by plant entrainment. Note that under realistic operating conditions a depletion of 23% at the intake, with respect to far field, will be produced by an overall kill rate of 2%.

# Effects of Power-Plant Cooling-Water Entrainment on the Drifting Macroinvertebrates of the Wabash River (Indiana)

James R. King  
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## ABSTRACT

The drifting macrobenthic community of the Wabash River was quantitatively sampled at the intake and discharge of the Wabash Generating Station from June to November 1973. Weekly samples were dominated by the insects *Potamya flava* and Chironomidae. These and other abundant taxa were examined for preentrainment and postentrainment viability and physical condition. The mean percent survival was lowest for *Baetis* spp. and *Stenonema* spp. Deterioration of physical condition during entrainment was also greatest for *Baetis* spp. Other abundant taxa (*P. flava*, *Tricorythodes* sp., *Pentagenia vittigera*, *Hydropsyche* spp., and Chironomidae) apparently suffered little overall mortality or condition loss during entrainment. For most taxa differences in viability and physical condition between intake and discharge were greatest during periods of high discharge temperature (high  $\Delta T$ ).

The entrainment of aquatic organisms into once-through cooling water systems of thermal power plants is a major environmental concern. During passage through the condenser and subsequent exposure to temperature elevation, mechanical abrasion, and chemicals, certain types of organisms (including phytoplankton, zooplankton, planktonic eggs, and larvae of fish and invertebrates) can experience physiological shock that may cause death or may induce less obvious secondary effects that ultimately affect the survival of individuals or populations. The magnitude of the effect depends on a number of factors, including the proportion of water from the receiving body diverted through the condensers, the organism's susceptibility and acclimatized temperature, the degree of temperature rise, and the duration of exposure to elevated temperatures.

Several on-site evaluations of entrainment effects have been published. Most of these have been concerned primarily with zooplankton (Davies and Jensen, 1974), phytoplankton (Brooks, 1974) or ichthyoplankton (Coutant, 1971). Reviews of thermal effects on aquatic organisms (Coutant and Talmage, 1975) have few references to studies

of pumped-entrainment effects on freshwater macroinvertebrates. Early work, such as that done by Markowski (1959; 1960), seldom provided more than descriptive information.

Ginn, Waller, and Lauer (1974) detected a statistically significant difference between survival of an amphipod, *Gammarus* sp., at the intake station and survival at the discharge station of the Indian Point plant on the Hudson River, N. Y. Mean percentage survival during temperature changes from 7.1 to 8.3°C and ambient temperatures of 24.9 to 26.0°C was above 90% at the two intake and the two discharge stations. *Gammarus* sp. were also found to be significantly more abundant in the entrained waters during darkness.

Marcy (1975) reported that field and laboratory studies of thermal tolerances and the thermal component causing inner-plant mortalities have been emphasized in the past. Almost all the studies were concerned with phytoplankton and zooplankton.

Most of the existing information on tolerance temperatures of aquatic organisms is based on 48- or 96-hr exposures. These data have little relevance to exposure times of 2 min or less experienced by organisms at the Wabash Generating Station.

According to Ginn, Waller, and Lauer (1974) temperature bioassays indicate that the thermal tolerance of *Gammarus* sp. depends on exposure time and ambient temperature. Reports concerning the tolerance of aquatic insects to temperature extremes include laboratory and field studies by Sage (1974) on the caddis fly, *Hydropsyche* sp.; by MacPhee and Brusven (1973) on the caddis fly, *Dicosmoecus* sp., and on the stonefly *Pteronarcys californica*; and by Martin and Gentry (1974) on dragonfly nymphs, *Libellula* sp. Other data compiled from the literature on lethal temperature limits for freshwater

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consistent near-field effects could be detected. Near-field effects could considerably increase the impact if the observed depletions result from mortality outside the cooling system rather than from avoidance. Determining the total impact clearly requires more information on the mechanisms governing near-field depletion and the extent of the depletion plumes.

## ACKNOWLEDGMENTS

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invertebrates are reported by Bush, Welsh, and Mar (1972). The literature search indicated that temperature tolerances have been determined for only a small percentage of the total number of freshwater invertebrate species. Bush, Welsh, and Mar reported on 94 species, but about 8500 described species of freshwater invertebrates (excluding protozoa and parasitic classes) exist (Pennak, 1953).

Studies undertaken to determine the effects of heated water on drift activity of stream benthos (Wojtalik and Waters, 1970) and secondary production modifications due to thermal load (McNaught and Fenlon, 1972) have also provided intriguing data for biologists studying the impact of heated effluents on water resources.

The principal objectives of our study were to document the composition and relative abundance of the drifting macroinvertebrates in the Wabash River and to examine the short-term effects on these organisms of the combined entrainment effects at the Wabash Generating Station, a plant using once-through cooling.

## MATERIALS AND METHODS

All samples were collected from June 1 to Nov. 2, 1973, at the Wabash Generating Station on the west bank of the Wabash River, 1.6 km north of Terre Haute, Ind. When the plant is operating at peak capacity (880 Mw), its six coal-fired units require a cooling-water flow of 30 m<sup>3</sup>/sec (475,500 gpm). Water enters the condenser cooling system through three shoreline intakes and is discharged less than 60 sec later into a 120-m concrete canal. At river discharges of less than 230 m<sup>3</sup>/sec, the discharge plume is separated from the river by a 79-m steel-reinforced concrete cell structure.

Duplicate samples were collected once each week at 2- to 3-hr intervals over a 24-hr period. Collections were made from the intake bay for units 5 and 6 (station 1) and from the discharge-cell structure (station 2). Conical plankton nets with a mouth opening 50 cm in diameter and a mesh size of 550  $\mu$ m were used. Torpedo-type digital flowmeters, attached within the nets by cables, recorded water flow. The volume of water sampled was calculated from these readings.

Sampling time was normally 4 min. At the intake the net was submerged and allowed to drift in the water column approximately 10 m in front of the intake grates. The net fluctuated in depth from just beneath the surface to mid-depth or slightly below mid-depth (3 m). Discharge samples were collected from just beneath the surface, about 6 m from the opening of the discharge canal. At times considerable effort was required to keep the net submerged in the turbulent discharge waters.

A mercury thermometer accurate to  $\pm 0.4^{\circ}\text{C}$  provided temperature readings at both stations. During the 5-month sampling period, we made a conscious effort to avoid sampling during periods of chlorination. Chlorine residual was measured in the discharge canal on Sept. 13, 1973, while chlorination of units 5 and 6 was under way, but no detectable residual above 0.2 ppm was recorded.

Organisms collected from both stations and not retained for condition analysis were immediately preserved in 70% isopropyl alcohol. Samples taken during the period immediately after sunset were kept for condition analysis and were not preserved until each sample had been sorted. Sorting was completed within 1 hr after collection. Live organisms were separated from dead organisms in the field laboratory on the basis of the following criteria: Those swimming, crawling, showing gill movement, or moving when prodded were considered live; those showing no vital signs and no response to gentle probing were considered dead. Later all organisms were identified to the lowest possible taxon given time available. The presorted sunset samples were then analyzed further for physical condition of each individual. Each organism was assigned to one of the following categories of physical condition based on appearance:

Good, all gills, appendages, and antennae present.

Fair, one or two gills or appendages or one antenna missing or otherwise slightly damaged.

Poor, two or more gills or appendages missing or visibly broken or otherwise damaged.

All collected organisms were classified as live or dead and categorized as to condition by the same person throughout the study period. Damage, if any, sustained by the organisms during sampling was assumed to be equal among replicates and among stations.

## RESULTS

Wabash River temperatures measured at the intake ranged from 11.5 $^{\circ}\text{C}$  (Nov. 1) to 28.5 $^{\circ}\text{C}$  (Aug. 8). The highest  $\Delta T$  (11.0 $^{\circ}\text{C}$ ) measured during the study period occurred Sept. 27, 1973. The highest discharge temperature and the largest  $\Delta T$  range did not occur simultaneously; 37.5 $^{\circ}\text{C}$  was recorded at the discharge on Aug. 30.

At the maximum possible pumping rate (30 m<sup>3</sup>/sec), the Wabash Plant circulated 10.8% of the total river discharge through the condensers in July, 11.4% in August, 28.3% in September, and 24.8% in October. From the beginning of the study period until Sept. 13, 1973, the Wabash Plant operated all six units. From this date to the end of the study period, only three units were operating; these included units 5 and 6, where the entrainment samples were taken.

The eight most abundant taxa, representing ~90% of all individuals, were caddis flies, *Potamya flava* and *Hydropsyche* spp.; mayflies, *Baetis* spp., *Tricorythodes* sp., *Stenonema* spp., and *Pentagenia vittigera*; and the dipteran groups Chironomidae and *Chaoborus* spp.

Figure 1 summarizes the mean intake and discharge densities of the eight abundant insect taxa. Each point on the figure represents the mean of approximately 12 samples taken over a 24-hr period. High river stages prohibited access to the discharge station during the first part of the study period; thus discharge samples were not collected at that time. Discharge densities were higher than intake



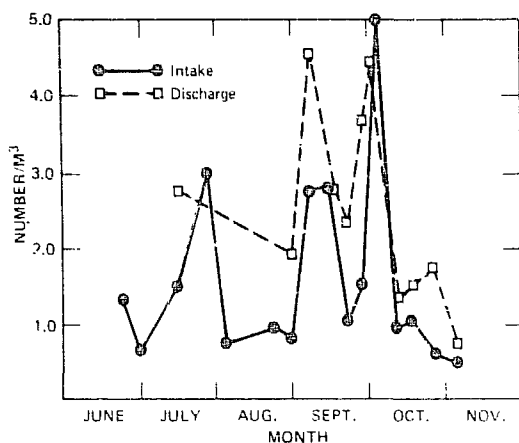


FIG. 1 Mean density of all abundant insects at the intake and discharge of the Wabash Generating Station, 1973.

densities during most sampling periods. Discharge samples generally consisted of three times as much water as intake samples during each sampling period since the discharge velocity was higher than intake velocity. *Potamyia flava* and/or Chironomidae dominated species composition at both intake and discharge in all sampling periods except June, when *Tricorythodes*, *Baetis*, and *Hydropsyche orris* were more numerous.

Survival data for selected abundant insect taxa observed in samples from the intake and discharge of the Wabash Station are summarized in Table 1. *Baetis* spp., *Stenonema* spp., *Pentagenia vittigera*, *Hydropsyche* spp., and Chironomidae all display fewer live animals at the discharge than

at the intake. *Potamyia flava* abundance at the discharge was more than twice that at the intake; this suggests that intake sampling may have been selective for a larger percentage of dead individuals than discharge sampling (waters were more thoroughly mixed at the discharge).

*Baetis* spp. exhibited the largest difference in survival percentages between intake and discharge (69.5%). The difference was most pronounced during periods of high  $\Delta T$  and high discharge temperature. Other species, with the possible exception of *Stenonema*, showed little overall reduction in survival due to high  $\Delta T$  or high effluent temperatures (Table 1).

Results of the analysis of physical condition for abundant insect species is presented in Table 2. Most species displayed little deterioration of physical condition overall. In samples of *Baetis* spp. at the discharge, however, 60.1% fewer animals were in good condition than in samples at the intake. High temperature and high  $\Delta T$  apparently caused the greatest deterioration of physical condition to *Baetis* sp. More detailed analysis of the results of entrainment effects is presented by King (1974).

## DISCUSSION

When estimating abundance or viability of the drifting macrobenthic community from a small number of representative samples, we must select the sampling method, location, depth, frequency, and duration carefully.

The distribution of organisms affects sampling procedures. Single-depth samples were collected assuming that a random horizontal and vertical distribution of organisms existed in the water column. Since a greater density of organisms was observed at the discharge, this was evidently

TABLE 1 Mean Percentage of Survival of Selected Species of Insects at the Intake and Discharge of the Wabash Station\*

| Species and location        | $\Delta T$ Range, °C |             | Discharge temp., °C |             | All samples |
|-----------------------------|----------------------|-------------|---------------------|-------------|-------------|
|                             | 5.5-8.0              | 8.1-11.0    | 19.5-31.5           | 31.6-37.5   |             |
| <i>Baetis</i> spp.          |                      |             |                     |             |             |
| Intake                      | 77.1 (48)            | 77.8 (45)   | 78.4 (51)           | 76.2 (42)   | 85.4 (199)  |
| Discharge                   | 24.0 (150)           | 11.4 (264)  | 54.2 (72)           | 7.9 (342)   | 15.9 (414)  |
| <i>Stenonema</i> spp.       |                      |             |                     |             |             |
| Intake                      | 81.8 (11)            | 100.0 (9)   | 84.6 (13)           | 100.0 (17)  | 95.9 (98)   |
| Discharge                   | 58.3 (24)            | 85.7 (21)   | 76.0 (25)           | 65.0 (20)   | 71.1 (45)   |
| <i>Pentagenia vittigera</i> |                      |             |                     |             |             |
| Intake                      | 93.5 (31)            | 100.0 (34)  | 100.0 (28)          | 94.6 (37)   | 91.3 (73)   |
| Discharge                   | 96.5 (85)            | 88.9 (126)  | 97.9 (146)          | 78.5 (65)   | 91.9 (211)  |
| <i>Hydropsyche</i> spp.     |                      |             |                     |             |             |
| Intake                      | 69.5 (95)            | 80.0 (40)   | 59.6 (89)           | 97.8 (46)   | 82.4 (222)  |
| Discharge                   | 61.2 (188)           | 71.1 (142)  | 61.5 (143)          | 68.4 (187)  | 65.5 (336)  |
| <i>Potamyia flava</i>       |                      |             |                     |             |             |
| Intake                      | 24.3 (300)           | 91.8 (208)  | 40.4 (394)          | 92.1 (114)  | 75.8 (1042) |
| Discharge                   | 87.0 (947)           | 80.6 (1270) | 85.6 (1104)         | 80.9 (1115) | 83.3 (2217) |
| Chironomidae                |                      |             |                     |             |             |
| Intake                      | 91.3 (46)            | 92.3 (118)  | 95.5 (112)          | 84.6 (52)   | 92.4 (172)  |
| Discharge                   | 84.1 (44)            | 94.2 (104)  | 91.1 (101)          | 91.5 (17)   | 91.2 (148)  |

\*Data are for low and high  $\Delta T$  ranges, low and high discharge temperature ranges, and all samples combined. Numbers in parentheses are totals of organisms sampled.

not the case. One reason for the difference may be that the intake for units 5 and 6 entrained fewer organisms than either of the other two intake structures, which were located 45 and 90 m upstream. This possibility was not tested, however.

Another reason for the difference in observed intake-discharge densities may be that subsurface sampling at the intake was not the best estimate of the true entrainment density. Mancini and Gammon (1974) conducted a subsequent program to test this theory. Their data indicate that most of the drifting macrobenthos in the Wabash River are suspended near the river substrate. This suggests that we missed some of the organisms by sampling at one depth. Passage through the cooling system probably mixed the water sufficiently to distribute the organisms more equally throughout the water column and resulted in higher observed discharge densities. Multidepth sampling at each intake and at the discharge would be likely to provide more accurate numerical density estimates. More detailed analysis of the multidepth sampling conducted on the main stem of the Wabash River in the vicinity of the Wabash Plant, including results of vertical and horizontal drift stratification, will be published in the future.

When nets are used to sample organisms, water velocity through the nets must be considered since increased velocity may increase sampling mortality. This is particularly important in entrainment studies since most often there is a large difference between the water velocity at the intake and that at the discharge. We selected 4 min as the sampling time to minimize potential sampling damage. We would have collected more organisms using a longer sampling time, but the potential for damage to the organisms would have increased, particularly at the dis-

charge, where water velocities through the net were approximately three times as great as at the intake.

The results of this study also indicate that tolerance of the rigors of entrainment into this cooling-water system varies among species and among members of the same species. Any measurable effects depend on the organism's susceptibility to entrainment, its physical condition, its life stage, and other factors, such as the ambient temperature of the river water, the degree of temperature rise during entrainment, and the amount of mechanical and chemical stress to which an organism may be subjected during condenser passage.

Our study of the Wabash Generating Station indicates that most organisms subject to entrainment are not homogeneously distributed in the water column. We observed that the distributional patterns of particular species change according to diel patterns. The diel periodicity in the drift demonstrated by the Trichoptera and Ephemeroptera of the Wabash River is consistent with trends noted by other investigators (Waters, 1972). Mancini (1974) further details the occurrence of diel periodicity in these and other taxa in the Wabash River. This type of information is necessary if we are to be able to predict when and how many of each species will be subject to entrainment, what percentage of animals will be killed, and what the effect on the drifting macrobenthic community will be.

Although chlorine toxicity can be eliminated from consideration, the observed mortalities may be the result of some undetermined factor(s) acting either individually or synergistically with temperature. It was apparent from our study that large numbers of *Baetis* spp. and *Stenonema* spp. are killed while passing through the condenser cooling

TABLE 2. Mean Percentage of All Animals in Good Physical Condition at the Intake and Discharge of the Wabash Station\*

| Species and location        | $\Delta T$ Range, °C |     |       |      | Discharge temp., °C |           | All samples |
|-----------------------------|----------------------|-----|-------|------|---------------------|-----------|-------------|
|                             | 5.5                  | 8.0 | 8.1   | 11.0 | 19.5-31.5           | 31.6-37.5 |             |
| <i>Baetis</i> spp.          |                      |     |       |      |                     |           |             |
| Intake                      | 81.3                 | 48  | 95.6  | 45   | 86.0                | 50        | 93.0 (199)  |
| Discharge                   | 48.0                 | 150 | 24.2  | 264  | 91.7                | 72        | 32.9 (414)  |
| <i>Stenonema</i> spp.       |                      |     |       |      |                     |           |             |
| Intake                      | 90.9                 | 11  | 73.7  | 19   | 92.3                | 13        | 69.4 (98)   |
| Discharge                   | 58.3                 | 24  | 81.0  | 21   | 84.0                | 25        | 68.9 (45)   |
| <i>Pentagenia vittigera</i> |                      |     |       |      |                     |           |             |
| Intake                      | 100.0                | 31  | 97.1  | 34   | 100.0               | 28        | 98.6 (73)   |
| Discharge                   | 98.8                 | 85  | 98.4  | 126  | 99.3                | 146       | 98.6 (211)  |
| <i>Hydropsyche</i> spp.     |                      |     |       |      |                     |           |             |
| Intake                      | 85.3                 | 95  | 100.0 | 140  | 84.3                | 89        | 93.7 (222)  |
| Discharge                   | 91.0                 | 188 | 99.3  | 142  | 96.5                | 143       | 94.5 (330)  |
| <i>Polynebia flava</i>      |                      |     |       |      |                     |           |             |
| Intake                      | 67.7                 | 300 | 99.0  | 208  | 87.8                | 391       | 90.2 (1042) |
| Discharge                   | 97.7                 | 947 | 99.9  | 1270 | 99.1                | 1164      | 99.0 (2217) |

\*Data are for low and high  $\Delta T$  ranges, low and high discharge temperature ranges, and all samples combined. Numbers in parentheses are totals of organisms sampled.

not the case. One reason for the difference may be that the intake for units 5 and 6 entrained fewer organisms than either of the other two intake structures, which were located 45 and 90 m upstream. This possibility was not tested, however.

Another reason for the difference in observed intake-discharge densities may be that subsurface sampling at the intake was not the best estimate of the true entrainment density. Mancini and Gammon (1974) conducted a subsequent program to test this theory. Their data indicate that most of the drifting macrobenthos in the Wabash River are suspended near the river substrate. This suggests that we missed some of the organisms by sampling at one depth. Passage through the cooling system probably mixed the water sufficiently to distribute the organisms more equally throughout the water column and resulted in higher observed discharge densities. Multidepth sampling at each intake and at the discharge would be likely to provide more accurate numerical density estimates. More-detailed analysis of the multidepth sampling conducted on the main stem of the Wabash River in the vicinity of the Wabash Plant, including results of vertical and horizontal drift stratification, will be published in the future.

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charge, where water velocities through the net were approximately three times as great as at the intake.

The results of this study also indicate that tolerance of the rigors of entrainment into this cooling-water system varies among species and among members of the same species. Any measurable effects depend on the organism's susceptibility to entrainment, its physical condition, its life stage, and other factors, such as the ambient temperature of the river water, the degree of temperature rise during entrainment, and the amount of mechanical and chemical stress to which an organism may be subjected during condenser passage.

Our study of the Wabash Generating Station indicates that most organisms subject to entrainment are not homogeneously distributed in the water column. We observed that the distributional patterns of particular species change according to diel patterns. The diel periodicity in the drift demonstrated by the Trichoptera and Ephemeroptera of the Wabash River is consistent with trends noted by other investigators (Waters, 1972). Mancini (1974) further details the occurrence of diel periodicity in these and other taxa in the Wabash River. This type of information is necessary if we are to be able to predict when and how many of each species will be subject to entrainment, what percentage of animals will be killed, and what the effect on the drifting macrobenthic community will be.

Although chlorine toxicity can be eliminated from consideration, the observed mortalities may be the result of some undetermined factor(s) acting either individually or synergistically with temperature. It was apparent from our study that large numbers of *Baetis* spp. and *Stenonema* spp. are killed while passing through the condenser cooling

TABLE 2. Mean Percentage of All Animals in Good Physical Condition at the Intake and Discharge of the Wabash Station\*

| Species and location        | $\Delta T$ Range, °C |             | Discharge temp., °C |             | All samples |
|-----------------------------|----------------------|-------------|---------------------|-------------|-------------|
|                             | 5.5-8.0              | 8.1-11.0    | 19.5-31.5           | 31.6-37.5   |             |
| <i>Baetis</i> spp.          |                      |             |                     |             |             |
| Intake                      | 81.3 (48)            | 95.6 (45)   | 86.0 (50)           | 84.3 (51)   | 93.0 (199)  |
| Discharge                   | 48.0 (150)           | 24.2 (264)  | 91.7 (72)           | 20.5 (342)  | 32.9 (414)  |
| <i>Stenonema</i> spp.       |                      |             |                     |             |             |
| Intake                      | 90.9 (11)            | 73.7 (19)   | 92.3 (13)           | 70.6 (17)   | 69.4 (98)   |
| Discharge                   | 58.3 (24)            | 81.0 (21)   | 84.0 (25)           | 50.0 (20)   | 68.9 (45)   |
| <i>Pentagenia vittigera</i> |                      |             |                     |             |             |
| Intake                      | 100.0 (31)           | 97.1 (34)   | 100.0 (28)          | 94.6 (37)   | 98.6 (73)   |
| Discharge                   | 98.8 (85)            | 98.4 (126)  | 99.3 (146)          | 96.9 (65)   | 98.6 (211)  |
| <i>Hydropsyche</i> spp.     |                      |             |                     |             |             |
| Intake                      | 85.3 (95)            | 100.0 (40)  | 84.3 (89)           | 100.0 (46)  | 93.7 (222)  |
| Discharge                   | 91.0 (188)           | 99.3 (142)  | 96.5 (143)          | 93.0 (187)  | 94.5 (330)  |
| <i>Potamyia flava</i>       |                      |             |                     |             |             |
| Intake                      | 67.7 (300)           | 99.0 (208)  | 87.8 (394)          | 98.2 (114)  | 90.2 (1042) |
| Discharge                   | 97.7 (947)           | 99.9 (1270) | 99.1 (1104)         | 98.7 (1115) | 99.0 (2217) |

\*Data are for low and high  $\Delta T$  ranges, low and high discharge temperature ranges, and all samples combined. Numbers in parentheses are totals of organisms sampled.

system. *Baetis* spp., *Stenonema* spp., and *Hydropsyche* spp. also show deterioration of physical condition, which probably results from mechanical or pressure conditions experienced during entrainment.

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# An Evaluation of the Impingement of Fishes at Four Duke Power Company Steam-Generating Facilities

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## ABSTRACT

Fish impingement data were collected at four generating facilities representing river intake, epilimnetic reservoir intake, and hypolimnetic reservoir (skimmer-wall) intake designs. The study sites included both traveling and stationary intake-screen systems. At all sites fewer species were represented in impingement samples than were collected during concurrent field investigations in the study areas. Significant differences in impingement rates among seasons were established and were caused primarily by large numbers of threadfin shad impinged during the coldest months. Intake velocities did not determine relative rates of impingement at the sites studied. Certain characteristics of intake design appeared to influence impingement rates significantly, however. Skimmer walls, present at two sites, did not provide a deterrent to fish impingement.

A continuous supply of condenser cooling water is required in the operation of steam electric stations using once-through cooling systems. Losses due to impingement represent a potentially significant impact on existing fisheries, depending on the number of fish involved and the species affected. The impingement of fish on the intake screens of power stations presents problems in terms of operating efficiency as well as from a biological standpoint (Edsall and Yocom, 1972). Losses due to impingement do not necessarily represent significant losses to resident fish populations, however (Landry and Strawn, 1974).

Our study, initiated in the spring of 1974, was designed to monitor impingement rates. Data were analyzed in an effort to explain some of the causative factors involved in the impingement of fish and to provide an indication of overall effects on resident fisheries. Fish-impingement data were collected at four generating facilities operated by Duke Power Company. The sites include an epilimnetic reservoir intake design (Allen Steam Station), two hypolimnetic reservoir (skimmer-wall) intake designs (Marshall Steam Station and Oconee Nuclear Station), and a river

intake design (Buck Steam Station). Condenser cooling at each of these sites is accomplished with a once-through open-loop system.

## DESCRIPTION OF STUDY AREAS

### Allen Steam Station

Allen Steam Station, an epilimnetic reservoir intake design located in Gaston County, N. C., draws condenser-cooling water from the northern end of Lake Wylie, a 5040-ha impoundment. This five-unit coal-burning facility began operating in 1957 and has a total capacity of 1140 Mw(e). The intake structure, comprised of 10 pumps with a maximum total flow of 35 m<sup>3</sup>/sec, is located at the back of a small cove of ~1.5 ha. Intake velocities are indicated in Table 1.

An extensive study of the Lake Wylie fishery was conducted from September 1973 to August 1974 (Industrial Bio-Test Laboratories, Inc., 1974). A total of 40 species of fishes representing 10 families were collected with gill nets and electrofishing gear. Collections were comprised primarily of the families Clupeidae, Cyprinidae, Catostomidae, Ictaluridae, and Centrarchidae. Important forage species include gizzard and threadfin shad and several cyprinids. Creel census data indicate that Lake Wylie supports a better-than-average sport fishery; major species were crappie, bluegill, largemouth bass, and catfish (Industrial Bio-Test Laboratories, Inc., 1974).

### Marshall Steam Station and Oconee Nuclear Station

Marshall Steam Station is located on Lake Norman in Catawba County, N. C.; Oconee Nuclear Station is located on Lake Keowee in Oconee County, S. C. Each is a hypolimnetic reservoir (skimmer-wall) intake design. At both sites condenser cooling water is drawn from an intake

Edwards, Hunt, Miller, and Sevic: Duke Power Company, Environmental Sciences Unit, Charlotte, North Carolina.

NRD

TABLE 1 Intake Velocities (cm/sec)  
at Four Duke Power Company  
Generating Facilities

| Pumping mode           | Full pond | Maximum drawdown |
|------------------------|-----------|------------------|
| Allen Steam Station    |           |                  |
| Units 1 and 2          |           |                  |
| 1 pump                 | 4.57      | 7.62             |
| 2 pumps                | 7.62      | 12.50            |
| Units 3, 4, and 5      |           |                  |
| 1 pump                 | 6.70      | 11.28            |
| 2 pumps                | 11.28     | 18.90            |
| Marshall Steam Station |           |                  |
| Units 1 and 2          |           |                  |
| 1 pump                 | 7.32      | 13.41            |
| 2 pumps                | 10.67     | 18.90            |
| Units 3 and 4          |           |                  |
| 1 pump                 | 5.49      | 9.75             |
| 2 pumps                | 9.14      | 16.15            |
| 3 pumps                | 11.89     | 21.03            |
| Buck Steam Station     |           |                  |
| Units 1 and 2          |           |                  |
| 1 pump                 | 79.25     | 79.25            |
| 2 pumps                | 73.76     | 73.76            |
| Units 3, 4, 5, and 6   |           |                  |
| 1 pump                 | 50.60     | 82.90            |
| 2 pumps                | 38.10     | 61.57            |
| Oconee Nuclear Station |           |                  |
| Units 1, 2, and 3      |           |                  |
| 2 pumps                | 36.27     | 51.51            |
| 3 pumps                | 31.70     | 45.11            |
| 4 pumps                | 26.80     | 38.05            |

cove separated from the main body of the lake by a skimmer wall, which is a solid concrete structure extending from the surface to a depth of 20 m, well below the depth of the summer thermocline. During the summer, low-temperature hypolimnetic water is drawn into the cove and provides efficient condenser cooling. At both Marshall and Oconee, intake structures are located at the extreme back of the intake coves; water flow is from the skimmer wall to the intake structure.

Marshall Steam Station is a four-unit coal-burning facility with an output capacity of 2025 Mw(e). The total capacity of the 10 intake pumps is 65 m<sup>3</sup>/sec. The Marshall intake includes 16 stationary intake screens, and the skimmer wall separates the intake cove (~81 ha) from the main body of Lake Norman (Fig. 1).

The fishery of Lake Norman has been investigated by the North Carolina Wildlife Resources Commission (Miller and DeMont, 1974) and the Environmental Sciences Unit at Duke Power Company (Duke Power Company, 1974). Forty species of fishes representing 10 families were reported in these two studies. Lake Norman supports several species of Cyprinidae, Catostomidae, Ictaluridae,

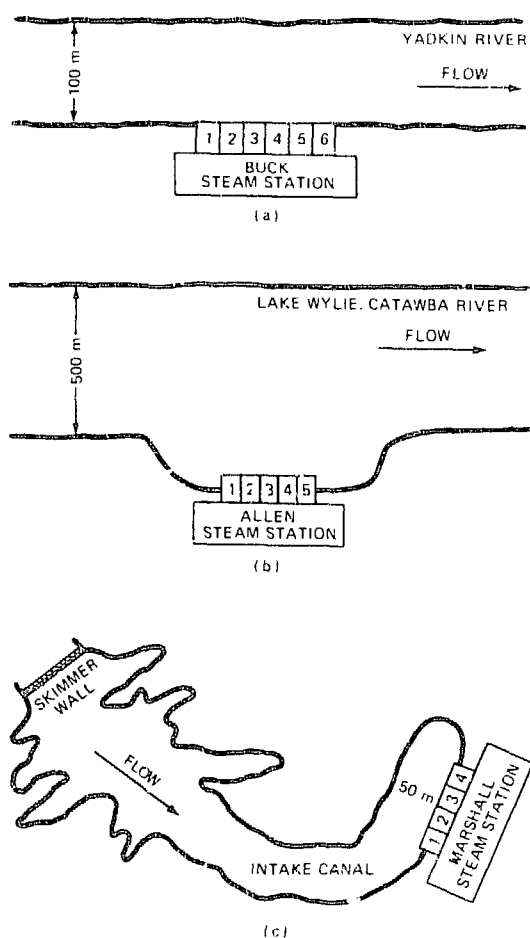


FIG. 1 Intake structure locations at three power-plant sites. (a) river intake design. (b) epilimnetic reservoir intake design. (c) hypolimnetic reservoir (skimmer-wall) intake design.

and Centrarchidae. Valued sport species are sunfish, crappie, largemouth bass, white bass, and striped bass. Threadfin shad have been introduced and have become well established as a forage species. Cove rotenone data collected during the summers of 1973 and 1974 provided a mean biomass estimate of ~123 kg/ha.

Oconee Nuclear Station has three generating units with a total output capacity of 2625 Mw(e). The intake structure includes 24 stationary intake screens and 12 intake pumps with a total capacity of 128 m<sup>3</sup>/sec. The intake cove at Oconee (~25 ha) is similar to the Marshall intake cove shown in Fig. 1.

Since June 1972, the Southeast Reservoir Investigations, Bureau of Sport Fisheries and Wildlife, has been investigating the fishery of Lake Keowee in connection with the operation of Oconee Nuclear Station (Southeast Reservoir Investigations, 1972). Electrofishing and gill-net

data have accounted for 41 species representing 9 families. Centrarchidae, Ictaluridae, and Catostomidae made up the majority of the total catch. A creel census conducted between July and September 1973 indicated that largemouth bass, bluegill, and crappie are important sport fishes. Yearly estimates of fish biomass from cove rotenone data from 1972 to 1974 have averaged  $\sim 67.4$  kg/ha. Threadfin shad were stocked in February of 1974 and apparently have become established as a forage fish.

### Buck Steam Station

Buck Steam Station, a river intake design located in Rowan County, N. C., draws condenser-cooling water from the Yadkin River near its mouth at High Rock Lake. A six-unit coal-burning facility, Buck began operation in 1926 and has a total capacity of 488 Mw(e).

The intake structure, which is made up of 10 pumps capable of drawing water at a maximum rate of  $21 \text{ m}^3/\text{sec}$ , is located on the shoreline of the river and parallel to the river flow (Fig. 1). Intake velocities at Buck are shown in Table 1.

An environmental investigation of the Yadkin River, including a field study of the river's fishery, has been conducted by Duke Power Company (1975). Forty species representing 11 families were collected from October 1973 to September 1974. Collections from all stations sampled were composed primarily of the families Cyprinidae, Ictaluridae, Centrarchidae, and Clupeidae. Important sport fishes in the area include several species of centrarchids, as well as white and channel catfish. Abundant forage species are gizzard shad, satinfish shiner, and whitefin shiner. In contrast to other sites, threadfin shad are not abundant in the area.

## MATERIALS AND METHODS

Duke Power Company steam stations have two basic types of intake structures. Marshall Steam Station and Oconee Nuclear Station have stationary intake screens, which must be removed with a crane for each inspection. Allen and Buck Steam Stations have traveling screens made up of several panels mounted on a vertical conveyor-belt arrangement. Both traveling and stationary intake screens have 1-cm mesh and are located behind vertical trash racks with bars at 7.6-cm intervals.

At sites having stationary intake screens, because of the difficulty and time involved, one screen from each unit (25% of the total screen area) was removed and inspected at 2 week intervals. After each inspection screens were cleaned thoroughly and replaced; each inspection represented the total impingement per screen occurring over a 2-week period. Fishes were identified, measured, and counted. When exceptionally large numbers of fishes were collected, total numbers were estimated from a subsample.

At traveling screen sites, one screen associated with each generating unit was inspected regularly twice each week. During the normal screen-washing procedure, water jets are directed at screens as they are rotated. Debris and

impinged fishes are washed off into a sluiceway that flows into the river or reservoir. For our inspection screens were rotated, washed, and left in place for 24 hr. They were then rotated and washed again, and fishes and debris were collected in a wire-mesh catch basket placed in the sluiceway. Fishes collected were preserved in 10% formalin and brought back to the laboratory, where they were identified, measured, and counted.

At Allen Steam Station impingement studies were conducted for 1 year by Industrial Bio-Test Laboratories, Inc. (Krueger, Rice, and Otto, 1974). Three 8-hr subsamples were taken over a 24-hr sample period every 2 weeks. All screens were sampled on each sample date.

At some steam stations, when river conditions and heavy accumulation of debris have necessitated early cleaning of intake screens, total sample periods have been less than 24 hr. An effort was made at all stations to be as consistent as possible in the actual screen sampling and the total time screens were left stationary. Occasionally alternate screens were sampled because of operating difficulties or required repairs.

Fish impingement has been expressed as total seasonal impingement by species (fish per screen per day) and estimated impingement rates (total daily impingement). Estimated rates were derived by extrapolating collected data (fish per representative intake screen per sample period). Fishes that were unidentifiable because of decomposition were listed as such; otherwise, identifications were made to lowest possible taxon.

Intake-approach velocities were calculated from total pump flow and actual area of intake-screen openings. Intake temperatures were monitored hourly at each station with a thermocouple or a resistance temperature device.

To satisfy required assumptions of analysis of variance, we transformed raw data collected at Marshall Steam Station using common logarithms. A two-way analysis of variance (ANOVA) (Sokal and Rohlf, 1969) was then used to test for differences in impingement rates between units and seasons. Duncan's multiple range test was used to determine relative differences between units (Steel and Torrie, 1960).

## RESULTS

### Allen Steam Station

At Allen Steam Station, 55,762 fishes representing 19 taxonomic groups (18 species plus 1 genus) were collected from October 1973 through September 1974 (Table 2). Gizzard shad, threadfin shad, channel catfish, and bluegill were collected most consistently and were present in samples during each month of the study. Threadfin shad accounted for the majority of fishes collected (54,971) and comprised 98.6% of the total catch. Of these, 91.8% were collected from November 1973 through January 1974.

As the data in Table 2 indicate, rates of fish impingement at Allen were lowest in spring and summer and highest in fall and winter. The estimated number of fish impinged per screen per day, calculated on a seasonal basis, ranged from 13 in spring 1974 to 349 in fall 1973. This

TABLE 2 Seasonal Summary of Fish Impingement at Allen Steam Station\*

| Common name                    | Scientific name            | Oct.—Dec. 1973 | Jan.—Mar. 1974 | Apr.—June 1974 | Jul.—Sept. 1974 |
|--------------------------------|----------------------------|----------------|----------------|----------------|-----------------|
| Gizzard shad                   | <i>Dorosoma cepedianum</i> | 59             | 26             | 43             | 70              |
| Threadfin shad                 | <i>Dorosoma petenense</i>  | 31,295         | 20,219         | 836            | 2621            |
| Channel catfish                | <i>Ictalurus punctatus</i> | 7              | 45             | 20             | 12              |
| Bluegill                       | <i>Lepomis macrochirus</i> | 31             | 24             | 168            | 89              |
| Miscellaneous†                 |                            | 20             | 34             | 71             | 72              |
| Total number of fish collected |                            | 31,412         | 20,348         | 1138           | 2864            |
| Estimated fish/screen/day      |                            | 349            | 226            | 13             | 32              |
| Estimated total fish/day       |                            | 5,235          | 3,390          | 195            | 480             |

\* Fish per screen per day and total fish per day were determined from the number of fish impinged on all intake screens. All screens were sampled from October 1973 to August 1974 by Industrial Bio-Test Laboratories, Inc. Total impingement in September 1974 was estimated from samples collected from five representative screens by Duke Power Company, Environmental Sciences Unit (total number of screens at Allen is 15).

† Miscellaneous fish, listed from most to least frequently impinged, are: white catfish, *Ictalurus catus*; yellow perch, *Perca flavescens*; pumpkinseed, *Lepomis gibbosus*; warmouth, *Lepomis gulosus*; redbreast, *Lepomis auritus*; catfish, *Ictalurus* sp.; redear, *Lepomis microlophus*; silvery minnow, *Hybognathus nuchalis*; brown bullhead, *Ictalurus nebulosus*; golden shiner, *Notemigonus crysoleucas*; mosquitofish, *Gambusia affinis*; white bass, *Morone chrysops*; satinfish shiner, *Notropis analostanus*; black crappie, *Pomoxis nigromaculatus*; and largemouth bass, *Micropterus salmoides*.

corresponds to a range in total daily impingement of 195 to 5235 fishes for spring and fall, respectively. An estimated 898,911 fishes were impinged at Allen throughout the study period.

### Oconee Nuclear Station

At Oconee Nuclear Station, 241,697 fishes, including 12 taxonomic groups (9 species and 3 genera), were collected on representative screens from July 1974 through May 1975 (Table 3). Threadfin shad, stocked in February 1974, were not represented in the samples to any extent until November 1974. Prior to this, yellow perch and bluegill comprised the majority of identifiable fish (22.2% and 72.6%, respectively). By December threadfin shad had become the dominant species collected and remained so through winter 1975. Overall, threadfin shad accounted for 49.3% of the total identifiable fish collected. Yellow perch, bluegill, and threadfin (4 to 6 cm long and weighing ~2.1 g each) were the most commonly impinged fishes at Oconee.

Impingement rates were lower in spring and summer than in fall and winter. The estimated number of fish impinged per screen per day ranged from 15 in summer 1974 to 393 in winter 1975. This represents an estimated total daily impingement rate of 372 in summer 1974 and 9425 in winter 1975. An estimated total of 1,064,262 fishes were impinged at Oconee throughout the study period.

### Marshall Steam Station

Impingement data collected at Marshall Steam Station from April 1974 through March 1975 are presented in Table 4. Fishes collected at Marshall have been categorized into 14 groups, including 10 species and 4 genera. A total of 974,896 fishes were collected on representative screens throughout the study period. Impingement rates at Marshall

were highest in fall 1974 and winter 1975 (1518 and 1108 fish/screen/day, respectively). An estimated 54 fish/screen/day were collected in spring 1974, and 6 fish/screen/day were collected in summer 1974. Total fish impinged per day ranged from 96 in summer 1974 to 24,090 in fall 1974. An estimated 3,769,300 fishes were impinged at Marshall throughout the study period. Approximately 99% of this total was made up of threadfin shad and *Dorosoma* sp. The size range of the *Dorosoma* suggests that they also were threadfin shad. The majority of the shad were collected in November and December 1974 and January 1975.

Analysis of variance indicated that significant differences existed in the number of fishes collected among units ( $P < 0.01$ ) and among winter, spring, and summer seasons ( $P < 0.01$ ). Fall data were not included in statistical analyses because unit 3 intake pumps did not operate for several days. Duncan's multiple range test has shown that the mean number of fishes collected at each unit is significantly different from means at all other units.

### Buck Steam Station

Impingement rates were low at Buck Steam Station (Table 5) despite relatively high intake velocities (Table 1). A total of 467 fishes was collected from representative screens from July 1974 through June 1975. Fishes impinged at Buck, identified to lowest possible taxa, comprise 7 species and 2 genera groups.

Estimated total daily impingement at Buck was less than 1 fish/day in both fall and winter 1974. The estimated total fish collected per day was highest in summer 1974 (42.5 fish/day) and intermediate in spring 1975 (1.3 fish/day). Fish mortalities occurring upstream are the suspected cause of relatively high numbers of fish collected during the summer period.



TABLE 3 Seasonal Summary of Fish Impingement at Oconee Nuclear Station\*

| Common name                    | Scientific name            | July–Sept. 1974 | Oct.–Dec. 1974 | Jan.–Mar. 1975 | Apr.–May 1975 |
|--------------------------------|----------------------------|-----------------|----------------|----------------|---------------|
| Threadfin shad                 | <i>Dorosoma petenense</i>  | 0               | 9,949          | 107,865        | 1426          |
| Bluegill                       | <i>Lepomis macrochirus</i> | 2749            | 183            | 365            | 82            |
| Yellow perch                   | <i>Perca flavescens</i>    | 757             | 182            | 4,164          | 874           |
| Unidentifiable                 |                            | 5005            | 3,157          | 99,653         | 5215          |
| Miscellaneous†                 |                            | 24              | 26             | 14             | 7             |
| Total number of fish collected |                            | 8535            | 13,497         | 212,061        | 7604          |
| Estimated fish/screen/day      |                            | 15              | 24             | 393            | 28            |
| Estimated total fish/day       |                            | 372             | 587            | 9,425          | 676           |

\*Fish per screen per day and total fish per day were estimated from the number of fish impinged on six representative intake screens; total number of screens at Oconee is 24.

†Miscellaneous fish, listed from most to least frequently impinged, are: gizzard shad, *Dorosoma cepedianum*; crappie, *Pomoxis* sp.; white crappie, *Pomoxis annularis*; catfish, *Ictalurus* sp.; pumpkinseed, *Lepomis gibbosus*; warmouth, *Lepomis gulosus*; carp, *Cyprinus carpio*; largemouth bass, *Micropterus salmoides*; and trout, *Salmo* sp.

TABLE 4 Seasonal Summary of Fish Impingement at Marshall Steam Station\*

| Common name                    | Scientific name            | Apr.–June 1974 | July–Sept. 1974 | Oct.–Dec. 1974 | Jan.–Mar. 1975 |
|--------------------------------|----------------------------|----------------|-----------------|----------------|----------------|
| Gizzard shad                   | <i>Dorosoma cepedianum</i> | 1,108          | 49              | 69             | 40             |
| Threadfin shad                 | <i>Dorosoma petenense</i>  | 1,995          | 1879            | 181,109        | 393,623        |
| Shad                           | <i>Dorosoma</i> sp.        | 6,780          | 70              | 377,419        | 0              |
| Catfish                        | <i>Ictalurus</i> sp.       | 4              | 99              | 22             | 0              |
| Yellow perch                   | <i>Perca flavescens</i>    | 722            | 19              | 42             | 10             |
| Unidentifiable                 |                            | 4,414          | 22              | 17             | 5,356          |
| Miscellaneous†                 |                            | 17             | 6               | 3              | 2              |
| Total number of fish collected |                            | 15,040         | 2144            | 558,681        | 399,031        |
| Estimated fish/screen/day      |                            | 54             | 6               | 1,518          | 1,108          |
| Estimated total fish/day       |                            | 864            | 96              | 24,090         | 17,728         |

\*Fish per screen per day and total fish per day were estimated from the number of fish impinged on four representative intake screens; total number of screens at Marshall is 16.

†Miscellaneous fish, listed from most to least frequently impinged, are: minnows, *Notropis* sp.; black crappie, *Pomoxis nigromaculatus*; white crappie, *Pomoxis annularis*; largemouth bass, *Micropterus salmoides*; bluegill, *Lepomis macrochirus*; sunfish, *Lepomis* sp.; striped bass, *Morone saxatilis*; carp, *Cyprinus carpio*; and golden shiner, *Notemigonus crysoleucas*.

TABLE 5 Seasonal Summary of Fish Impingement at Buck Steam Station\*

| Common name                    | Scientific name            | July–Sept. 1974 | Oct.–Dec. 1974 | Jan.–Mar. 1975 | Apr.–June 1975 |
|--------------------------------|----------------------------|-----------------|----------------|----------------|----------------|
| Gizzard shad                   | <i>Dorosoma cepedianum</i> | 391             | 0              | 1              | 0              |
| Threadfin shad                 | <i>Dorosoma petenense</i>  | 1               | 0              | 0              | 10             |
| Shad                           | <i>Dorosoma</i> sp.        | 33              | 0              | 0              | 2              |
| Miscellaneous†                 |                            | 14              | 0              | 1              | 14             |
| Total number of fish collected |                            | 439             | 0              | 2              | 26             |
| Estimated fish/screen/day      |                            | 4.25            | 0              | <0.1           | 0.13           |
| Estimated total fish/day       |                            | 42.5            | 0              | <0.1           | 1.3            |

\*Fish per screen per day and total fish per day were estimated from the number of fish impinged on four representative intake screens; total number of screens at Buck is 14.

†Miscellaneous fish, listed from most to least frequently impinged, are: white catfish, *Ictalurus catus*; bluegill, *Lepomis macrochirus*; catfish, *Ictalurus* sp.; channel catfish, *Ictalurus punctatus*; brown bullhead, *Ictalurus nebulosus*; and black crappie, *Pomoxis nigromaculatus*.

An estimated 4069 fishes were impinged at Buck throughout the study period. Gizzard shad, threadfin shad, and *Dorosoma* sp. made up 93.8% of all fish collected.

## DISCUSSION

### Intake Velocity

Intake velocities vary considerably among the stations at which impingement has been monitored (Table 1). Velocities fluctuate with water level and the number of intake pumps in operation. Ranges of possible intake velocities for Marshall and Allen Steam Stations are similar and are the lowest ranges of the four sites. Intake velocities at Oconee Nuclear Station are intermediate, and those at Buck Steam Station are the highest encountered in the study.

Impingement rates at these sites cannot be explained on the basis of intake velocity. Despite the high intake velocities at Buck, the rate of fish impingement at this site was lowest. The highest number of fish were collected at Marshall and Allen, where velocities were relatively low.

Average intake velocities throughout the study period were calculated for each of the four generating units at Marshall Steam Station, and corresponding impingement rates were determined (Fig. 2). Velocities were calculated on the basis of the number of pumps in operation; constant water level (full pond) was assumed. Although intake velocities are similar for each of the four units at Marshall, significant differences in rates of impingement occurred.

At Allen the highest numbers of fish were collected at units 1 and 5, even though these units represent the lowest and highest velocities, respectively, at this station. These data indicate that impingement rates vary significantly and that differences occur independent of intake velocities.

### Intake Structure Design

The intake design at Buck Steam Station may be a major factor contributing to low impingement rates at that site. The intake structure is parallel to and relatively flush with the river bank (Fig. 1). This design reduces the possibility of eddy currents, which may concentrate fish and subsequently trap them in the intake (U. S. Environmental Protection Agency, 1973).

Impingement of both threadfin shad and all other species combined is considerably higher at Marshall than at Buck despite Marshall's relatively low intake velocities. It has been established that variation in impingement rates among units 1 to 4 at Marshall is apparently not related to intake velocity. The shape of the intake cove probably causes the high impingement rates at unit 4 and the consecutively decreasing rates at units 3, 2, and 1 (Fig. 2). As shown in Fig. 1, the intake structure at Marshall is located at the extreme back of a cove. Currents create eddies next to screens associated with unit 4, and fish apparently concentrate in these eddy currents and become susceptible to impingement.

Variations in impingement rates among units at Allen were also observed. These have been attributed, in part, to

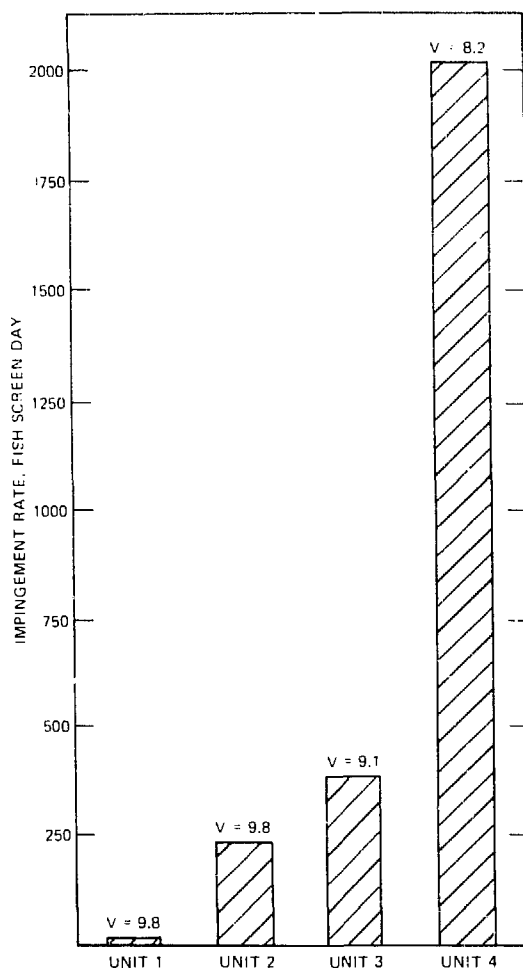


FIG. 2 Impingement rate and mean daily velocity (V), in cm/sec, for the four units at Marshall Steam Station from Apr. 30, 1974, through Feb. 4, 1975.

the relative position of intake screens (Fig. 1). Consistently higher numbers of fish were collected at units 1 and 5, probably because of the extreme upstream and downstream position of these units. It is likely that eddy currents on either side of the intake structure attract fish, which subsequently become impinged on the nearest intake screens (those associated with units 1 and 5).

### Resident Fish Populations

Field fishery investigations in the vicinities of Allen, Buck, and Marshall Steam Stations and Oconee Nuclear Station have indicated that, even where impingement rates are low, the areas support substantial fish populations. In all cases, many species reported in the field were not collected on intake screens.

At Allen, 24 species were collected in the area during a field study on Lake Wylie; 16 of these were collected from the traveling screens at the intake. Silvery minnows and mosquito fish, which were collected on intake screens, were not represented in field collections, and eight species were collected in the field which were absent from screen samples. At Lake Keowee, 33 species collected in the field were not collected on intake screens at Oconee: all species impinged were represented in field samples. Of the 40 species reported from Lake Norman, only 10 were impinged at Marshall. At Buck, 100 fishes were collected immediately above and below the intake structure in ~40 min on a single day in February; these included 16 species representing 7 families. There were no fish impinged at Buck in February, however, and only 7 species were impinged throughout the entire study.

That the species collected on intake screens were not represented in numbers proportional to the field samples suggests further that susceptibility to impingement is, to some degree, species specific. Data collected from the intake cove at Marshall provide an example. Electrofishing samples were collected in the intake cove on Feb. 21, 1975, in an effort to compare the numbers and species of fishes impinged with those present in the intake cove. Samples collected in small inlets above the intake structure provided 192 fishes, representing 12 species and 5 families. Sport fishes, including centrarchids, percids, and percichthyids, made up 39.5% of the catch. Threadfin shad accounted for 52.7% of all fish collected; all others were carp. Impingement data collected at Marshall in February indicate a total of 166,115 fishes impinged on four representative intake screens. Threadfin shad accounted for 99.9% (166,095) of the total; others included gizzard shad and one white catfish.

### Temperature

Data collected at Allen and Marshall Steam Stations and Oconee Nuclear Station have shown seasonal trends in total numbers of fish impinged. Impingement rates have generally been relatively high in late fall and winter and low throughout the rest of the year. This trend resulted from the collection of large numbers of threadfin shad when temperatures were low.

The relationship between temperature, the impingement of shad, and the impingement of all other species at Marshall is shown in Fig. 3. Threadfin shad were collected at relatively high rates beginning in November, when water temperatures dropped to ~13°C. Threadfin shad are known to be susceptible to low temperatures, and mortalities have been reported below 9°C (Strawn, 1963). Data presented here indicate that threadfin shad may become stressed at temperatures considerably higher than 9°C and are then susceptible to impingement because of their inability to resist intake currents. At Marshall, the impingement of species other than threadfin shad appears to be unrelated to temperature. Other species impinged, which have wider

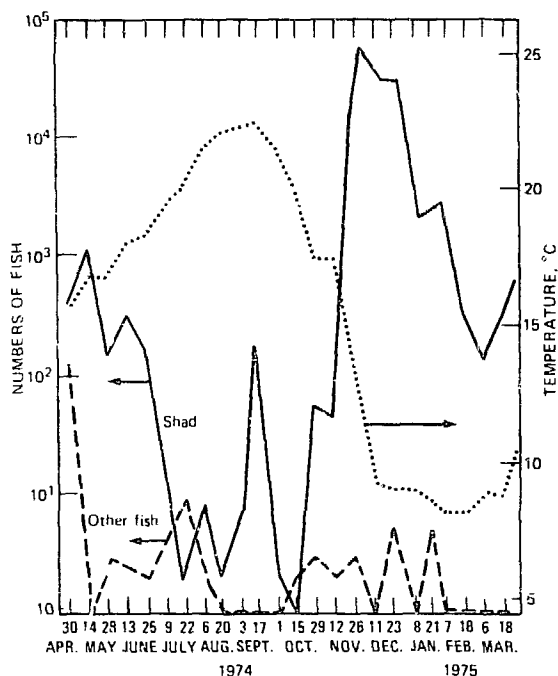


FIG. 3 Fish impingement at Marshall Steam Station, in mean number of fish per screen per day, relative to water temperature. Shad —, Other species, ---, Temperature, . . . .

ranges of thermal tolerance, are apparently not stressed by the ambient winter temperatures of Lake Norman.

A similar situation existed at Allen Steam Station. Of the total number of threadfin shad impinged there (98.8% of total fishes collected), 95.4% of the mortalities were attributed to low winter temperatures.

After being introduced into Lake Keowee for the first time in February 1974, threadfin shad made up the majority of the fishes impinged at Oconee Nuclear Station in the winter of 1974–1975. Low winter temperatures are suspected as the cause.

The impingement of threadfin shad at these sites does not appear to be a major cause of mortality but is rather an indication of natural mortality of the species.

Summarizing the major points discussed, we conclude:

1. Threadfin shad made up the majority of fish collected on intake screens. Sport fishes and other valuable species were considerably less susceptible to impingement.
2. In all cases the number of fish species impinged at specific sites was fewer than the number collected during concurrent field investigations in the study area.

3. Overall, relatively large numbers of fish were collected on intake screens during late fall or winter. This was caused by the susceptibility of threadfin shad to low temperature. The majority of the threadfin shad were apparently stressed or dead before they were collected on intake screens.

4. Intake velocity was not a determining factor concerning relative rates of impingement among the sites discussed.

5. Intake design, specifically the position of the intake structure relative to the cooling-water source, significantly affected impingement rates.

6. The skimmer walls at Marshall Steam Station and Oconee Nuclear Station have not provided a deterrent to the impingement of fish.

7. In view of the species and numbers of fish affected, it is doubtful that impingement at these sites has had significant impact on resident fish populations.

## ACKNOWLEDGMENTS

This research would not have been possible without the cooperation of the superintendents and station personnel at Duke Power Company's Allen, Buck, and Marshall Steam Stations and Oconee Nuclear Station. We appreciate the constructive reviews offered by W. D. Adair and L. L. Olmsted and the preparation of the final manuscript by Starr McCorkle.

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# Preliminary Assessment of Larval Fish Entrainment, Colbert Steam Plant, Tennessee River

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## ABSTRACT

Condenser cooling water for Colbert Steam Plant (1396 Mw, fossil fueled) is pumped from the Tennessee River, heated about 8°C, and discharged into a creek channel that flows back into the river ~1.6 km below the plant. In 1972 and 1973 the larvae of at least 13 species of fish were entrained by this plant: shad (gizzard and threadfin) and freshwater drum comprised 90 and 6%, respectively, of the total number entrained. Peak density of entrainment (1.5/m<sup>3</sup>) occurred in June. Assessment of entrainment was complicated since four to six species apparently spawned in the discharge channel in phase with their spawning in the Tennessee River. The overall ecological impact was considered minor since this plant uses only ~3% of the river flow for condenser cooling.

Nonscreenable young fishes entrained in the condenser cooling system of a steam electric generating station are subjected to mechanical and thermal stresses and possibly to toxic chemicals. The potential damage caused by entrainment has been reviewed in detail by Coutant (1970; 1971). Initially the primary concern was thermal shock, but a more recent study (Marcy, 1973) of thermal, mechanical, and biocidal factors at an operating power plant indicated that mechanical damage is also significant. Published entrainment studies to date have dealt primarily with anadromous species in relation to steam electric stations on coastal rivers (Coutant, 1971; Marcy, 1973; Kerr, 1953) as opposed to inland rivers and reservoirs.

Until recently fishery studies at fossil-fueled electric-generating stations on the Tennessee River involved only adult fish populations and were limited to intermittent surveys to determine species composition, relative abundance, and sport fishing in heated discharge basins and canals (Dryer and Benson, 1956; Tennessee Valley Authority, 1969; 1973). Although other entrainment studies have been initiated, this study was the first evaluation of fish entrainment at a fossil-fueled plant in the Tennessee Valley Authority (TVA) system. The primary objective was to determine species composition and relative densities of larval fishes entrained by an operating power

plant on the Tennessee River, and the secondary objective was to estimate mortality rates of the species entrained.

## MATERIALS AND METHODS

Two modified plankton nets were used to collect larval fish: (1) a framed stationary net for filtering larvae from the condenser cooling-water system within the plant and (2) a towed net for sampling near the intake forebay and in the heated discharge channel. The 0.79-mm-mesh stationary net was cone shaped, 0.5 m in diameter, and 1.0 m long. The collecting bucket was a 15-cm section of 7.6-cm-diameter PVC pipe threaded on the posterior end and fitted with a cap. The net was suspended in a cylindrical steel frame, and both frame and net were suspended vertically in a 0.75-m<sup>3</sup> galvanized tank mounted on a wheeled platform. Attachment to the frame prevented the net from collapsing in the tank as the condenser water flowed through the net and out over the top of the tank. The tank served as a retention reservoir for the collected specimens and also provided a means of estimating total volume sampled. The platform facilitated positioning the tank and net under a 10-cm valved drain from which the water was filtered. One of these drains, ~1 m above floor level, was located on the condenser inlet and outlet pipes for each unit.

Gear for towed net collections, modified slightly from that described by Netsch, Houser, and Voge (1971), included an enlarged cone-shaped 0.79-mm-mesh plankton net 1.0 m in diameter and 3.1 m long, a planing-type depressor and wire-angle indicator for deep tows, a stanchion-and-winch assembly mounted in a 6.1-m aluminum boat, and a flowmeter. The flowmeter was mounted on a retractable bracket that was lowered before each tow. The boat, powered by a 65-hp outboard motor, was equipped with a tachometer and timer to standardize speed and duration of tow.

Stationary-net collections were initiated on an exploratory basis on May 10 and 26, 1972. Samples were collected weekly from June through early August. A 2-hr sample was collected simultaneously from the condenser inlet and

outlet piping (immediately before and after condenser passage) for unit 1. Total volume was  $\sim 150 \text{ m}^3$  for each sample, but it varied somewhat since the valve opening was not exactly the same for each sample. Water pressure was too great to open the 10-cm valve completely. All samples were collected from the condenser cooling water for unit 1. Each sample was examined for live specimens and then preserved in 5% formalin; counting and identification were completed in the laboratory. Mean number of fish larvae per cubic meter was calculated and then extrapolated to number per  $100 \text{ m}^3$ .

Stationary-net collections were discontinued in 1973 so that we could sample in the discharge channel and in the reservoir adjacent to the intake forebay. In 1973 samples were collected weekly at three locations from March 28 to June 29. At station A, in the river channel adjacent to the intake forebay, duplicate 5-min tows were made at the surface and at 5 m, the approximate depth at which the intake forebay connects to the river channel (Fig. 1). At stations B and C, in the heated discharge, only duplicate surface tows were made since submerged objects prevented deeper tows. Duration of tows at stations B and C was based on preset distances. Towing speed (6.7 m/sec) was the same for all stations. Preservation and counting of the samples were the same as described for the stationary-net samples. The following sources were consulted for larval fish identification: May and Gasaway (1967), Meyer (1970), Nelson (1968), Purkert (1961), Taber (1969), Shelton (1972), and Wenn and Grinstead (1971).

To estimate the number of fish larvae entrained in relation to total river flow, we calculated the volume of river flow and the volume of the condensing water used for monthly intervals during the period samples were collected. Volume of river flow past the plant was computed by the Hydraulic Data Branch of TVA. Volume of condensing water was computed from the pumping capacity for each of the five units and the plant operating records for each unit during that interval.

## STUDY AREA

Colbert Steam Plant, constructed and operated by TVA, is located on Pickwick Reservoir in northwest Alabama (Fig. 1). It is a five-unit fossil-fueled plant with a total electrical generating capacity of 1396 Mw. The intake structure is approximately 50 m from the main channel of the Tennessee River. Connection to the river is via an excavated channel or basin that is  $\sim 3.5 \text{ m}$  deep where it intersects the river channel and 8.0 m deep at the intake structure. Depth of the reservoir at this location is 9.0 to 10.0 m at normal reservoir elevations. Since the edge of the excavated area does not extend to the bottom of the river channel, the condenser cooling water is apparently drawn from the upper 3.5 m of the water column passing the plant. The volume of cooling water used ( $56 \text{ m}^3/\text{sec}$  at peak load) is generally less than 5% of the river flow (mean river flow is  $1513 \text{ m}^3/\text{sec}$ ).

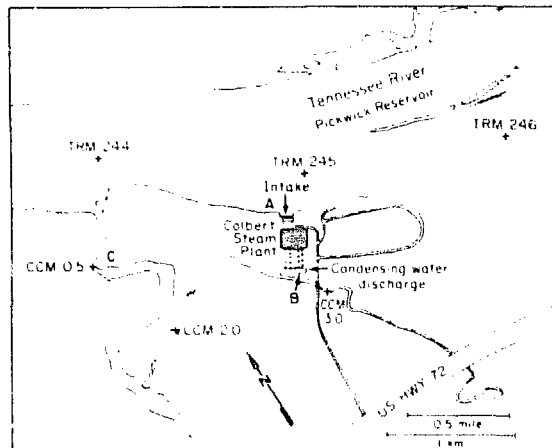


FIG. 1 Map of lower Cane Creek, which receives the heated effluent from Colbert Steam Plant. CCM, Cane Creek mile; TRM, Tennessee River mile. A, B, and C designate sampling stations.

Water for condenser cooling passes through 1-cm<sup>2</sup>-mesh traveling screens, mixed-flow circulating pumps, and 1.5-m piping to the condensers. From the condensers the water moves to a series of five concrete conduits that discharge into an oblong basin  $\sim 700 \text{ m}$  in circumference which was dredged to join an existing creek channel. Velocities approach 2.2 m/sec, and travel time from the intake pumps to the discharge basin ranges from 174 sec (for units 1 to 4) to 193 sec for unit 5. The heated effluent ( $\Delta T$  5 to 8°C) then passes from the discharge basin down the lower 4.5 km of Cane Creek before returning to the Tennessee River 1.6 km below the plant intake. Time of travel back to the river has been calculated to be  $\sim 2.5$  to 3 hr, depending on plant load and reservoir elevation. Mean velocity in the discharge channel is 0.4 m/sec. Velocity is quite variable, however, because of variations in depth, width, substrate material, and upstream runoff: it approaches 2.0 m/sec in some areas. Upstream runoff has only a minor influence; e.g., mean daily volume, calculated from an upstream gauging station for March–June 1973, was less than 1% of the cooling-water discharge.

Pickwick Reservoir, a mainstream impoundment of the Tennessee River, covers parts of Alabama, Mississippi, and Tennessee. The reservoir, formed by Pickwick Landing Dam and bounded upstream by Wilson Dam, is part of the 1046-km navigation system of the Tennessee River. Length of shoreline is 798 km, and the reservoir is 85 km long, with a maximum width of 2.4 km. Width is about 1.0 km at the plant site, which is 24 km downstream from Wilson Dam. The habitat at the plant site is essentially riverine. Surface area is 17,449 ha at summer elevation; volume at this elevation (126 m above sea level) is  $1.35 \times 10^9 \text{ m}^3$ . At normal winter elevation (124 m above sea level), volume is  $8.4 \times 10^8 \text{ m}^3$ . Flushing rates at these volumes are approximately 10 and 6 days, respectively. Ambient water tem-

perature ranges from 4.0 to 31.0°C, depending on seasonal climatic conditions. Thermal stratification does not occur.

The fish fauna of Pickwick Reservoir is composed primarily of typical warm-water species (Lagler, 1956). However, sauger (*Stizostedion canadense*) and walleye (*Stizostedion vitreum*), which are present, have been considered cool-water or "lukewarm" species in relation to thermal requirements. Cove rotenone samples (Tennessee Valley Authority, 1972) in 1971 yielded a mean standing stock biomass of 502 kg/ha, consisting of 48 species. Dominant species by weight were gizzard shad (*Dorosoma cepedianum*), 58%; threadfin shad (*Dorosoma petenense*), 13%; smallmouth buffalo (*Ictalurus bubalus*), 7%; carp (*Cyprinus carpio*), 6%; and freshwater drum (*Aplodinotus grunniens*) and channel catfish (*Ictalurus punctatus*), each 4%. Combined weight of sauger, largemouth bass (*Micropterus salmoides*), and smallmouth bass (*Micropterus dolomieu*) was 2.5%.

## RESULTS

A total of 887 postlarval fish (yolk sac absorbed) was collected from the condenser inlet and outlet piping within the plant from May 10 to August 9, 1972 (Table 1). Although there appeared to be only one species of *Lepomis* and *Notropis* represented, two species of *Dorosoma* and *Morone* were probably present. Shelton (1972) reported

TABLE 1. Species Composition of Larval Fishes Collected at Colbert Steam Plant, 1972-1973

| Species or taxa               | Common name*         | Sampling station† |   |   |   |
|-------------------------------|----------------------|-------------------|---|---|---|
|                               |                      | A                 | B | C | D |
| <i>Polyodon spathula</i>      | Paddlefish           | X                 |   |   |   |
| <i>Dorosoma cepedianum</i>    | Gizzard shad         | X                 | X | X | X |
| <i>Dorosoma petenense</i>     | Threadfin shad       | X                 | X | X | X |
| <i>Hiodon tergisus</i>        | Mooneye              | X                 | X |   |   |
| <i>Cyprinus carpio</i>        | Carp                 | X                 | X | X |   |
| Cyprinidae                    | Minnows              | X                 | X | X | X |
| Catostomidae                  | Suckers              | X                 | X | X |   |
| <i>Moxostoma</i> sp.          | Redhorse             | X                 |   | X |   |
| <i>Ictalurus natalis</i>      | Yellow bullhead      | X                 | X |   |   |
| <i>Ictalurus punctatus</i>    | Channel catfish      | X                 |   | X |   |
| <i>Morone</i> sp.             | White or yellow bass | X                 | X | X | X |
| <i>Lepomis</i> sp.            | Sunfish              | X                 | X | X | X |
| <i>Micropterus dolomieu</i>   | Smallmouth bass      | X                 |   |   |   |
| <i>Micropterus salmoides</i>  | Largemouth bass      | X                 |   | X |   |
| <i>Pomoxis</i> sp.            | Crappie              | X                 | X | X |   |
| <i>Percina caprodes</i>       | Logperch             | X                 | X | X |   |
| <i>Stizostedion canadense</i> | Sauger               | X                 |   | X |   |
| <i>Stizostedion vitreum</i>   | Walleye              | X                 |   |   |   |
| <i>Aplodinotus grunniens</i>  | Freshwater drum      | X                 | X | X | X |

\*Common names are as listed in *A List of Common and Scientific Names of Fishes from the United States and Canada*, American Fisheries Society, Special Publication No. 6.

†Station A (river channel near intake forebay), station B (discharge basin), and station C (lower discharge channel) were sampled in 1973. Station D (within the plant) was sampled in 1972.

that it is practically impossible to separate postlarval gizzard and threadfin shad (~6 to 18 mm total length) and noted that the spawning periods and habitat of these species overlap. A similar problem has been encountered in identifying white and yellow bass larvae. All four species are common in Pickwick Reservoir. Shad (6 to 18 mm total length) made up 62% of the total catch, and freshwater drum (5 to 12 mm total length) comprised ~6%. The remaining 2% include *Lepomis* (5 to 7 mm total length), *Morone* (5.5 to 11 mm total length), *Notropis* (7 to 9 mm total length), and one channel catfish (17 mm total length). Peak numbers entrained were on June 13 (1.2/m<sup>3</sup>) and June 22 (2.0/m<sup>3</sup>), and 98% of these were shad. Relative densities of fish larvae in the five samples collected from July 5 to August 9 ranged from 0.05/m<sup>3</sup> to 0.1/m<sup>3</sup>. Only two live specimens were noted, and both were collected from the outlet side of the condenser.

Towed-net samples collected from three stations in 1973 yielded 22,185 young fish. Pro larvae and postlarvae fish, less than 23 mm total length, comprised ~98% of the specimens. Percent composition of the dominant species or taxa (Table 2) was: shad, 87%; freshwater drum, 6%; catostomids, 2%; and carp, 1%. At least 19 species (10 families) were collected at station A, 13 at station B, and 15 at station C (Table 1). All species collected in the heated effluent (B and C) were present in the samples from the river channel adjacent to the intake; the reverse was not true, however. Species collected only from the river channel included paddlefish, smallmouth bass, and walleye. The occurrence of the larvae of these three species was considered rare since only two paddlefish (April 11 and May 4), one smallmouth bass (May 16), and one walleye (April 11) were captured. A single specimen of three other species occurred only once in the heated effluent: yellow bullhead, station B, June 6; channel catfish, station C, June 29; and sauger, station C, April 18. At station A, a total of 74 sauger (6.0 to 6.5 mm total length) were collected on three sampling dates from April 4 to 18. Since species composition and numbers of larvae collected at the surface and at a depth of 5 m at station A were similar, the samples were combined to determine mean densities at this station.

Comparison of mean relative densities of selected species at each sampling station in 1973 indicated that on various sampling dates *Dorosoma* sp., *Cyprinus carpio*, and *Morone* sp. were more abundant in the heated discharge channel than near the intake. For example, on May 16 the mean density of carp at station C was 12.3/100 m<sup>3</sup>, but at the intake (station A) it was only 0.2/100 m<sup>3</sup>; on the same date mean density of *Morone* sp. was 10/100 m<sup>3</sup> at station C and 0.5/100 m<sup>3</sup> at station A.

Total estimated numbers of larvae entrained in June (the only period with almost equal sampling effort) were similar for 1972 and 1973 (Table 3). The percentage of cooling water used declined during the sampling period for 1973 because of above average river flow.

TABLE 2 Mean Relative Density (Number/100 m<sup>3</sup>) of Dominant Larval Fish Collected from the River Channel Near the Condenser Water Intake (A), from the Discharge Basin (B), and from the Lower Discharge Canal (C), Colbert Steam Plant, 1973

| Taxa                          | March |      |     | April |      |      | May   |       |       | June |       |       |
|-------------------------------|-------|------|-----|-------|------|------|-------|-------|-------|------|-------|-------|
|                               | A     | B    | C   | A     | B    | C    | A     | B     | C     | A    | B     | C     |
| <i>Dorosoma</i> sp.           |       |      |     | 0.4   |      | 2.2  | 97.1  | 143.3 | 99.9  | 33.2 | 126.4 | 109.8 |
| <i>Hiodon tergisus</i>        |       |      |     | 0.5   | 0.4  |      | 0.1   |       |       |      |       |       |
| <i>Cyprinus carpio</i>        |       |      |     | 0.1   | 0.3  | 1.8  | 0.1   | 0.3   | 5.7   | 1.2  | 0.7   | 0.6   |
| Catostomidae                  | 1.2   | 2.1  | 4.1 | 6.2   | 2.2  | 0.8  | 0.8   | 0.2   | 1.2   | 0.1  |       | 0.8   |
| <i>Morone</i> sp.             | 0.1   | 0.3  | 0.6 | 0.1   |      |      | 0.6   | 0.7   | 2.4   | 0.1  | 0.1   | 0.5   |
| <i>Lepomis</i> sp.            |       |      |     |       |      |      |       |       |       | 0.1  | 0.2   | 1.7   |
| <i>Micropterus salmoides</i>  |       |      |     |       |      | 0.01 | 0.01  |       | 0.2   |      |       | 0.1   |
| <i>Pomoxis</i> sp.            |       |      |     |       |      | 0.03 | 0.2   | 0.4   | 0.1   |      | 0.1   |       |
| <i>Percina caprodes</i>       | 0.3   | 0.1  | 0.2 | 0.4   | 0.7  | 0.4  | 0.6   | 1.5   | 0.9   | 0.1  | 0.2   | 0.2   |
| <i>Stizostedion canadense</i> |       |      |     | 1.3   |      | 0.03 |       |       |       |      |       |       |
| <i>Aplodinotus grunniens</i>  |       |      |     |       |      |      | 5.8   | 4.8   | 0.7   | 16.3 | 14.1  | 4.3   |
| Total/100 m <sup>3</sup>      | 1.6   | 2.5  | 4.9 | 9.9   | 3.6  | 5.3  | 105.3 | 151.2 | 111.1 | 59.1 | 141.8 | 118   |
| Number of samples             | 1     | 1    | 1   | 4     | 4    | 4    | 5     | 5     | 5     | 4    | 4     | 4     |
| Temperature, °C               |       |      |     |       |      |      |       |       |       |      |       |       |
| Minimum                       | 12.2  | 16.7 |     | 12.2  | 18.6 |      | 18.0  | 22.5  |       | 23.0 | 27.0  |       |
| Maximum                       |       |      |     | 16.2  | 21.4 |      | 21.4  | 27.0  |       | 26.8 | 32.2  |       |
| Mean                          |       |      |     | 13.9  | 19.9 |      | 19.5  | 24.6  |       | 24.9 | 30.1  |       |

TABLE 3 Estimates of Numbers of Larval Fish Entrained in Relation to Volume of River Flow and Volume of Condenser Cooling Water, Colbert Steam Plant, 1972-1973

| Date            | Larvae/100 m <sup>3</sup> | Species                      | Percent of total number entrained | Volume of condensing water, m <sup>3</sup> | Percent of river flow |
|-----------------|---------------------------|------------------------------|-----------------------------------|--------------------------------------------|-----------------------|
| 1972*           |                           |                              |                                   |                                            |                       |
| May 10-26       | 20                        | <i>Dorosoma</i> sp.          | 83                                | 5.9 × 10 <sup>7</sup>                      | 2.0                   |
|                 |                           | <i>Aplodinotus grunniens</i> | 10                                |                                            |                       |
| June 1-30       | 110                       | <i>Dorosoma</i> sp.          | 97                                | 9.1 × 10 <sup>7</sup>                      | 3.3                   |
| July 1-Aug. 9   | 6.1                       | <i>Dorosoma</i> sp.          | 53                                | 14.5 × 10 <sup>7</sup>                     | 3.5                   |
|                 |                           | <i>Aplodinotus grunniens</i> | 43                                |                                            |                       |
| 1973            |                           |                              |                                   |                                            |                       |
| Mar. 28-Apr. 30 | 4.0                       | Catostomids                  | 57                                | 11.4 × 10 <sup>7</sup>                     | 1.7                   |
|                 |                           | <i>Percina caprodes</i>      | 16                                |                                            |                       |
|                 |                           | <i>Morone</i> sp.            | 11                                |                                            |                       |
| May 1-31        | 139                       | <i>Dorosoma</i> sp.          | 95                                | 12.4 × 10 <sup>7</sup>                     | 2.0                   |
|                 |                           | <i>Aplodinotus grunniens</i> | 3                                 |                                            |                       |
| June 1-30       | 153                       | <i>Dorosoma</i> sp.          | 88                                | 9.8 × 10 <sup>7</sup>                      | 1.6                   |
|                 |                           | <i>Aplodinotus grunniens</i> | 10                                |                                            |                       |

\*Based on samples collected from cooling system within the plant in 1972 and on samples from the discharge basin in 1973.

## DISCUSSION

On the basis of a study of the Connecticut Yankee Nuclear Plant (Marcy, 1973), in which 80% of the total mortality of entrained young fish was attributed to mechanical damage and 20% to heat shock and prolonged exposure at temperatures above 28°C in a 1.83-km discharge canal, larval fish entrained by Colbert Steam Plant probably incurred mortality from mechanical damage and possibly thermal effects. Although thermal effects would not necessarily occur at 28°C in the heated discharge at

Colbert, thermal resistance varies according to species and acclimation temperature. It has been emphasized, however, (Coutant, 1970) that long discharge canals compound the temperature effect that is a function of exposure time and temperature above the incipient lethal level. Of the dominant species entrained in this study, shad (*Dorosoma* sp.) appear to be most vulnerable during May and June (Table 3). On the basis of the thermal regime for this period (Table 2) and the thermal-resistance data for gizzard shad, however, it appears that, even when we consider the



additional 180-min exposure in the 4.5 km of lower Cane Creek, thermal resistance of gizzard shad would not be exceeded. Hart (1952) reported that juvenile gizzard shad acclimated at 25°C could withstand 34°C for ~300 min. Although similar data are not available for threadfin shad (*Dorosoma petenense*), a study by Hubbs and Bryan (1974) of the maximum incubation temperature (34°C) for this species indicated that the upper thermal resistance of larval threadfin shad is similar to that of gizzard shad. Thermal resistance data for freshwater drum, the second most abundant species entrained, was not available in the literature.

Sauger, relatively abundant in mainstream reservoirs of the Tennessee River, is apparently one of the most temperature intolerant species (Smith and Koenst, 1975). In view of when sauger would be vulnerable to entrainment at Colbert Steam Plant (primarily April), thermal resistance of the species would not be exceeded. Juvenile sauger acclimated at 12°C had a median survival time of 260 min at 27°C (Smith and Koenst, 1975); and 2-day-old walleye fry, which have thermal requirements very similar to those of sauger, acclimated at 11°C had a 78% survival rate after exposure for 72 hr to 21°C. From late March through April the ambient temperature of Pickwick Reservoir ranged from 12.2 to 16.2°C, and the respective effluent temperatures at Colbert Steam Plant ranged from 16.7 to 21.4°C.

Regardless of the mortality rate experienced by entrained larval fish, the overall ecological impact will depend on the proportion of the total volume of receiving water diverted through the condensers (Coutant, 1971). Also, Marcy (1973) noted that any estimate of the effect of entrainment on mortality should take into account the natural mortalities during specific life stages. The natural survival rate for prolific species (such as shad and drum in this study) may be as low as 0.001%. On the basis of these observations, even assuming 100% mortality of entrained fish, overall effects of entrainment appear to be insignificant at Colbert Steam Plant, which uses only ~3% of the river flow.

Mortality rates due to entrainment could not be determined by the sampling methods used in this study. Although additional sampling within the plant at more than one unit (after pump and condenser passage) may provide a better estimate of mechanical damage, thermal effects would ultimately have to be evaluated in the long discharge channel by stationary nets (fast-towed nets are generally not suitable for collecting live larvae). Also in the towed-net samples in this study, because of heavy concentrations of zooplankton (primarily *Leptodora*), it was often impossible in a field inspection even to locate fish larvae.

Another aspect which confounds mortality estimates at this plant is that the discharge channel is a spawning area for some species, e.g., carp, shad, white bass, and yellow bass. Although sample size in this study and the inherent variability of larval fish distribution (Carlson and McCann, 1969; Nohle, 1968) may have accounted for density differences between the intake samples and samples from

the discharge channel (station C in particular), the collection (Wrenn, 1974) of ripe adults of these species indicated that spawning within the discharge channel accounted for the higher numbers of larvae of these species.

With regard to the sampling problems within the plant and in the discharge channel, the best estimate of entrainment at Colbert Steam Plant could probably be made by sampling with stationary nets in front of the intake and with stationary or towed nets along cross-sectional transects upstream and downstream from the plant to determine if certain species were being entrained in a higher proportion than the densities occurring in the river.

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# The Incidence and Effect of Once-Through Cooling on Young-of-the-Year Fishes at Long Point Bay, Lake Erie: A Preliminary Assessment

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## ABSTRACT

Every 2 weeks, from June 7 to Aug. 21, 1974, larval and young juvenile fish were sampled in the forebay, outfall gates, and effluent channel of the Nanticoke Generating Station located on Long Point Bay, Lake Erie, Ontario, Canada. With the plant operating at 25% maximum capacity, from 7708 to 134,820 larvae of six species were entrained every 24 hr. At peak plant output, estimates were from 30,800 to 539,000 fish entrained per 24 hr. This equals an hourly removal of 2.9% of the study-area water ( $22.4 \text{ by } 10^6 \text{ m}^3$ ), and most of the organisms contained therein.

Smelt (*Osmerus mordax*), constituting 95% of the total number of larvae entrained, were taken during all sampling hours. Mottled sculpin (*Cottus bairdi*), carp (*Cyprinus carpio*), alewife (*Alosa pseudoharengus*), and gizzard shad (*Dorosoma cepedianum*) made up 4% of the catch and were captured only at night. Yellow perch (*Perca flavescens*) comprised the remaining 1% and were taken in dawn and dusk samples.

Of all the fish passing through the condenser, only 1 to 3.7% survived. However, on the basis of the density of living smelt larvae in the forebay and effluent channel, 84% were killed by the time they had reached the effluent channel via the condenser and the tempering water. Separating the lethality on the basis of volume of water used by the two systems, 49.5% of the total plant mortality was due to mechanical injuries and only 34.5% due to condenser-passage shock.

The impact of this entrainment loss to the harvestable stock, not including loss of spawners, would be 43 tonnes annually, assuming the survival from larvae to recruit in nature is 1%. This loss is equal to 12% of the average (1972 to 1974) annual catch of one trawl fisherman operating in the eastern basin of Lake Erie.

At present more than 90% of all generating stations on the Great Lakes cool their steam condensers by pumping large quantities of water through the condenser tubes then back into the lake (once-through cooling). All fish larvae in the entrained water are exposed to high heat, pressure, and mechanical abrasion. Investigations at selected stations along the east coast of the United States (Marcy, 1975) have shown an estimated larval fish loss of  $0.3 \text{ to } 166$  million larvae per day per site. Few data are available to describe the magnitude of the problem in the Great Lakes. Edsall and Yocom (1972) estimated that entrainment of three species at the Cook and Palisades generating stations on Lake Michigan could be as high as  $4 \times 10^6$  fish/24 hr, of which at least 50% would die from heat shock alone. According to Neison Thomas, U. S. Environmental Protection Agency, Grosse Ile Laboratory, from April to August 1974 an estimated 300 million larvae and 50,000 kg of young-of-the-year yellow perch were entrained at the Detroit Edison Plant on Lake Erie, near Monroe, Mich.

The entrainment study at the Nanticoke Generating Station attempted to answer the following questions:

1. What species and how many larval fish are entrained annually?
2. When diurnally and seasonally are these larvae entrained?
3. What percentage of the entrained larvae are killed?
4. What are the causes of mortality among entrained fish?
5. How does the larval fish loss by entrainment affect recruitment into the fishery?
6. How can entrainment and the associated mortality be reduced?

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## STUDY AREA

The Nanticoke Generating Station, located on the northeastern shore of Long Point Bay in eastern Lake Erie (Fig. 1), presently produces 200 to 900 Mw(e)/hr. The plant withdraws  $1320 \text{ m}^3$  of water per minute for condenser and effluent cooling. At peak output in 1978, the plant will produce 4400 Mw(e) and will withdraw water

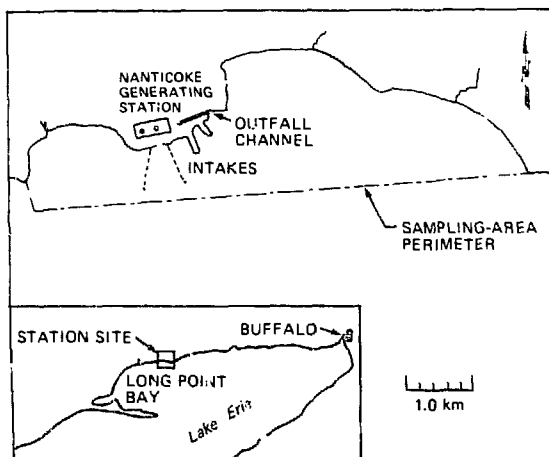


FIG. 1 Location of the Nanticoke Generating Station.

from the lake at a rate of  $11,090 \text{ m}^3/\text{min}$ —the equivalent of 3.3% of the total flow of all tributaries entering Lake Erie.

Water is taken into the station through two tunnels, each 8.3 m in diameter, buried in the lake bottom (Fig. 2). The intake structures are located 650 m offshore in 10 m of water with the intake ports raised 2.5 m above the lake bottom. The flow into the ports is horizontal, at a present maximum of 0.076 m/sec. From the intakes the water flows into the forebay, where 41% of the total volume is diverted through the condenser; the remainder (tempering water) bypasses the condenser and is pumped directly into the discharge canal to reduce effluent temperatures.

The condenser water passes through a 1.27-cm-mesh traveling screen at 0.122 m/sec before entering the condenser. This coolant is then pumped through many 2.5-cm-diameter condenser tubes, reaching a mean velocity of 2.2 m/sec and a  $\Delta T$  of 18 to  $20^\circ\text{C}$ . Passage through the condenser requires 35 to 40 sec. Downstream of the condenser, the cooling water is piped into a main underground conduit where it joins the hot water from a second generating unit. This water flows over an exit wier into the outfall channel. Total time course from screen well to wier is 2.5 min.

The tempering water is pumped directly from the forebay into the effluent channel 50 m downstream of the wier, where it mixes with the heated water discharged from

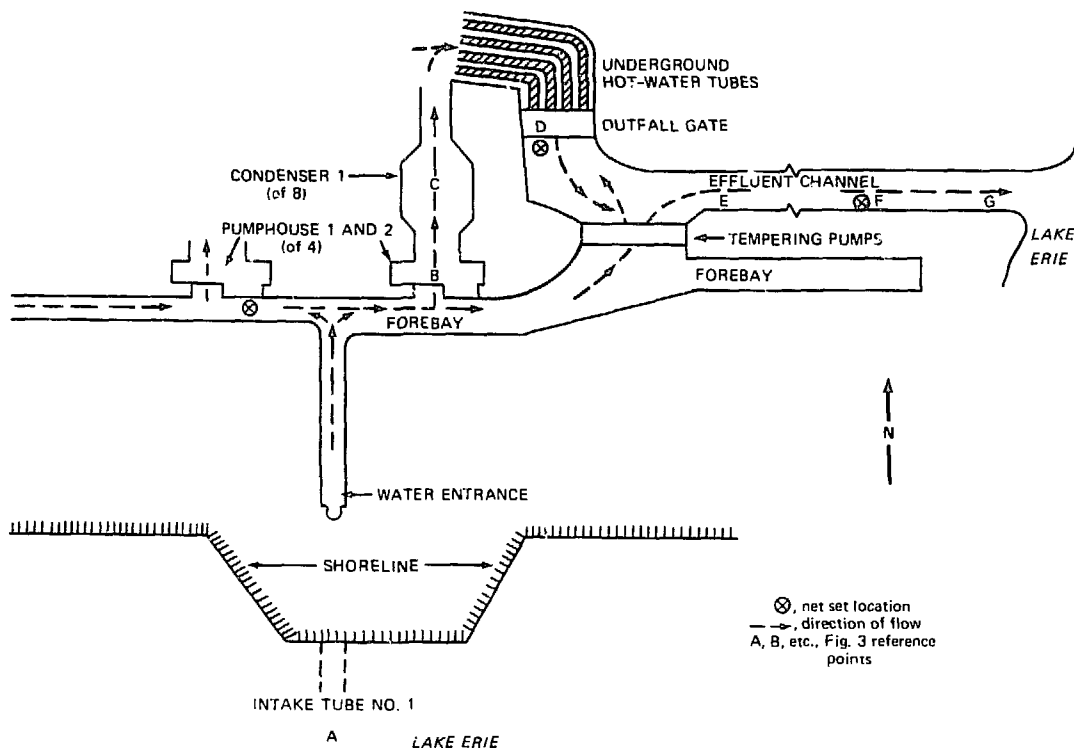


FIG. 2 The condenser cooling system of the Nanticoke Generating Station. X indicates sampling stations.

the condensers. The effluent channel is 15.25 m wide, 9.5 m deep, and 550 m long. At present generation levels, the water velocity averages 0.78 m/sec, and passage from the outfall gate to the lake requires from 8 to 12 min. Ambient lake temperatures, when fish larvae were being entrained (May through August), ranged from 12 to 22°C. There were no biocides used during the sampling season.

## MATERIALS AND METHODS

Six in-plant sampling series were completed, two of which occurred during periods when no heat was being added. Every 2 weeks, from June 7 to Aug. 20, samples were taken in the forebay, immediately below the outfall gates, and in the effluent channel (one cycle) (Figs. 2 and 3). Each series consisted of 24 samples in which each station was sampled, in replicate, during four light periods: day, 08:00 to 18:00; dusk, 18:00 to 20:00; night, 21:00 to 05:00; and dawn, 05:00 to 07:00.

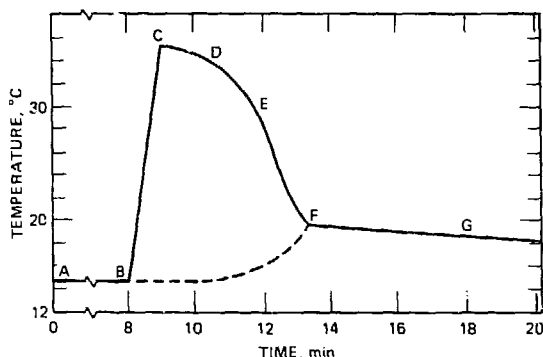


FIG. 3 Thermal experience of larval fish passing through the Nanticoke Generating Station cooling system during June and July 1974. Symbols A to G correspond to locations shown in Fig. 2.

Forebay and outfall-gate samples were taken with 0.5-m-diameter 782- $\mu$ m-mesh plankton nets having a 1 : 6 diameter-to-length ratio, reducing the effective mesh opening to one-half that of the standard 1 : 3 ratio net. The larger than usual mesh size ensured nearly 100% filtering efficiency; smaller mesh nets could not be controlled under the tremendous pressure of the water current. Each net was equipped with a 6000-ml plexiglass cod-end bucket. The net used in the effluent channel was also 0.5 m in diameter but was suspended in a frame similar in design to that used by Lewis et al. (1970). Each replicate sample was taken by fishing one net for one-half hour.

Fish captured in the nets were separated into those living and those dead, then sorted by species and measured. The fins and bodies of all fish were examined for signs of mechanical damage. During heat addition, fish killed while passing through the condenser were frequently contorted and were an opaque white color which distinguished them from those killed by other means (mechanical death). All live fish found in outfall samples were held in the sampling

buckets at effluent-channel temperatures from 0.05 to 10 hr so that latent mortality could be observed.

Concurrent with in-plant samples, 1-m-net tows were made in the lake near the station intakes (Fig. 2) to determine the percentage of all larval fish in the study area lost to entrainment.

Hourly records of water volume used; intake, condenser, and discharge water temperatures; and megawatt production were provided by the generating station. Point measurements of intake and discharge temperatures and water flow were taken during each sampling cycle.

Densities of larval and juvenile fish in each plant sampling area were calculated by dividing the number of fish captured by the volume of water passing through the net. The net was assumed to be 100% efficient.

Numbers of larval and juvenile fish in each plant sampling location were calculated from the following formula:

$$Lu = (D_{sm}^3 \times V_S)u$$

where  $Lu$  equals the total number of larvae entrained per 24 hr for a given number of generating units;  $D_{sm}^3$  equals the density of larval fish per cubic meter at any location; and  $V_S$  equals the volume of water passing by that station during 24 hr.

Monthly entrainment estimates were made by multiplying  $Lu$  by 25, the average number of days the station was in operation.

To estimate the percentage of the total larval population vulnerable to entrainment per hour, I calculated the larval density in the 0.6- by 7-km study area (Fig. 1) surrounding the intakes, then applied this number to the following formula:

$$E_T = \frac{(D_L m^3 \times V_e)}{(D_L m^3 \times V_L)} = \frac{N_e}{N_L} \times 100$$

where  $E_T$  = percentage of the total larval population in the study area vulnerable to entrainment per hour

$D_L m^3$  = mean density of larval fish in the lake surrounding the intakes

$V_e$  = volume of water entrained per hour

$V_L$  = volume of water in the study area per hour

$N_e$  = estimate of the total number of larval fish lost to the intake per hour based on lake densities

$N_L$  = total number of larvae in the study area at any given time

Estimates of survival at the in-plant sampling stations were made only during series 4 (July 22 to 26), because other series produced too few fish. Values were obtained by dividing the total number of living larvae per cubic meter at a station downstream of the intakes by the number of living larvae per cubic meter in the forebay samples. Marcy (1973) used a similar method to obtain larval fish survival and, in turn, mortality estimates at various points in the once-through cooling circuit. All mean values were weighted by the number of hours per diel period as well as the month-to-month variation in species density in the lake.

## RESULTS AND DISCUSSION

Of nine species captured in all sample locations, only smelt (*Osmerus mordax*), yellow perch (*Perca flavescens*), alewife (*Alosa pseudoharengus*), mottled sculpin (*Cottus bairdi*), gizzard shad (*Dorosoma cepedianum*) and carp (*Cyprinus carpio*) were taken in the forebay and outfall-gate locations (Table 1). Young-of-the-year white sucker (*Catostomus commersoni*), emerald shiner (*Notropis atherinoides*), and white bass (*Morone chrysops*), although very rarely (<1%) taken in the forebay, were consistently

were taken at dawn, dusk, and at night. Other workers (Edsall and Yocom, 1972; Marcy, 1973) reported similar diel fluctuations in the composition of the samples of entrained young-of-the-year fishes.

The presence or absence of light appears to be a critical parameter in abating larval fish entrainment. Pavlov (1969) conducted a study where he exposed larval (8 to 16 mm) fish to varying conditions of entrainment. By manipulating the light intensity and angle, he reduced entrainment by 40% in five of eight tests. During the last three experiments,

TABLE 1 Mean Density of Young-of-the-Year Fishes Sampled in the 0.6- by 7-km Area Surrounding the Nanticoke Generating Station Intakes, 1974

| Species                                                                 | Mean fish density/1000 m <sup>3</sup> |            |            |            |            |            |
|-------------------------------------------------------------------------|---------------------------------------|------------|------------|------------|------------|------------|
|                                                                         | June 7-12                             | June 21-27 | July 10-12 | July 22-26 | Aug. 10-13 | Aug. 19-20 |
| Alewife                                                                 | 0.11                                  | 0.80       | 1.95       | 3.50       | 3.75       | 0.40       |
| Gizzard shad                                                            | 0.08                                  | 2.10       | 4.33       | 8.00       | 9.58       | 0          |
| Rainbow smelt                                                           | 14.10                                 | 13.00      | 8.80       | 5.50       | 5.80       | 0          |
| Carp and emerald shiner                                                 | 1.10                                  | 0          | 7.20       | 0          | 2.50       | 2.90       |
| White sucker and shorthead redhorse ( <i>Moxostoma macrolepidotum</i> ) | 1.13                                  | 0.23       | 0          | 0          | 0          | 0          |
| Trout-perch ( <i>Percopsis omiscomaycus</i> )                           | 0.25                                  | 0          | 4.05       | 0          | 0          | 0          |
| White bass                                                              | 0.04                                  | 0          | 0          | 0.31       | 0.30       | 0.66       |
| Log perch ( <i>Percina caprodes</i> )                                   | 0.73                                  | 0.88       | 0.83       | 0.43       | 0          | 0          |
| Yellow perch                                                            | 4.41                                  | 0.53       | 0.10       | 0          | 0          | 0          |
| Mottled sculpin                                                         | 0                                     | 2.17       | 2.90       | 0          | 0          | 0          |

captured in the effluent channel, indicating that they had not been entrained at the intakes. Other than eggs, the smallest fish sampled were the yellow perch (6 mm) and the largest were smelt (42 mm). Developmental stages ranged from yolk-sac stage in perch to young juvenile for smelt and shad. Smelt made up 95% of the 987 larval and juvenile fish captured.

There were marked diel variations in the entrainment samples. Smelt, sculpin, alewife, and shad were captured mostly at night, whereas yellow perch and white bass were taken at dawn and dusk periods. More than 90% of all fish

in addition to lighting, Pavlov placed artificial reference points, such as tree branches, around the intakes and reduced entrainment by an amazing 88% over natural rates.

Seasonally, the susceptibility of larval fish to entrainment was directly proportional to their density at the intake (Table 2); e.g., in early June, when yellow perch became free swimming and moved away from shore, they appeared in the forebay samples.

During 1974 the daily estimated larval and juvenile entrainment for two generating units ranged from 7708 to 134,820 fish/24 hr (Table 2). At peak plant operation

TABLE 2 Estimated\* Number of Young-of-the-Year Fish Entrained at the Nanticoke Generating Station in 1974

| Species         | Date       | Total length, mm | Dawn and dusk |                             | Night      |                             | Day        |                             | Average No. per 24 hr |
|-----------------|------------|------------------|---------------|-----------------------------|------------|-----------------------------|------------|-----------------------------|-----------------------|
|                 |            |                  | No. per hr    | No. per 1000 m <sup>3</sup> | No. per hr | No. per 1000 m <sup>3</sup> | No. per hr | No. per 1000 m <sup>3</sup> |                       |
| Rainbow smelt   | June 7-12  | 7-10             | 751           | 3.50                        | 188        | 0.89                        | 0          |                             | 6,198                 |
|                 | 21-27      | 10-18            | 0             |                             | 663        | 3.90                        | 0          |                             | 5,967                 |
|                 | July 10-12 | 18-38            | 45            | 0.13                        | 6,588      | 21.01                       | 157        | 0.53                        | 61,132                |
|                 | 22-26      | 22-39            | 3,266         | 14.58                       | 13,214     | 52.43                       | 664        | 3.41                        | 133,276               |
|                 | Aug. 19-20 | 29-42            | 565           | 2.79                        | 1,182      | 5.84                        | 173        | 0.85                        | 13,748                |
| Yellow perch    | June 7-12  | 6                | 0             |                             | 168        | 0.84                        | 0          |                             | 1,512                 |
| Alewife         | June 21-27 | 9-13             | 0             |                             | 363        | 2.75                        | 0.16       | 2.58                        | 2,812                 |
|                 | July 22-26 | 16               | 0             |                             | 59         | 0.23                        | 0          |                             | 472                   |
| Mottled sculpin | June 21-26 | 7-8              | 0             |                             | 234        | 1.37                        | 0          |                             | 2,106                 |
|                 | July 10-12 | 8                | 0             |                             | 75         | 0.23                        | 0          |                             | 600                   |
| Gizzard shad    | July 22-26 | 22-25            | 0             |                             | 134        | 0.53                        | 0          |                             | 1,072                 |

\*Estimates were based on the samples taken in the station forebay.

(eight units) these values may increase to 30,832 and 539,280 fish/24 hr during May–June and July–August, respectively. For comparative purposes, I calculated the maximum number of young-of-the-year fish that could be entrained per hour. Using a mean density of 0.027 fish/m<sup>3</sup> ( $D_L$  m<sup>3</sup>) and a maximum flow of 665,453 m<sup>3</sup>/hr, 17,967 fish ( $N_e$ ) would be entrained each hour. This equals a removal of 2.9% of the total study-area population per hour. In 34.48 hr a volume equivalent to that in the entire study area would be drawn through the plant, entraining a maximum of (0.027 fish/m<sup>3</sup> × 22.4 × 10<sup>6</sup> m<sup>3</sup> water) 604,800 larval and juvenile fishes.

Entrainment mortalities were separated into those caused by mechanical injuries and those resulting from heat shock. Thirty percent of the fish taken in the forebay were dead (Table 3). Over one-half of these deaths were attributable to netting injuries and another 45% to mechanical abrasions and pressure changes inflicted during the 8-min passage from intakes to forebay. A very small percentage, 1 or 2%, were dead before entrainment. Although mortalities as high as 50% (Marcy, 1973) have been recorded for river intake samples, it seems unlikely that values in lentic waters could be comparable. Dead larvae and juveniles either float to the surface or sink to the bottom and are not transported very far in a lake situation. Since the intakes are well off the bottom (2.5 m), they would not pull in these larvae. Also, the semidecayed bodies of larval fish, dead for a number of days, are easily distinguishable from recent mortalities.

During the July 22 to 26 sampling period, the total mortality from forebay to effluent channel was 84%. Considering that 59% of the total volume of water drawn into the plant was pumped directly into the effluent channel, bypassing the condensers, a reduction in the total (effluent channel) mortality by only 12% over condenser heat death (Table 3) pointed to mechanical injuries as a major lethal factor. I separated mechanical from condenser death as a proportion, according to the volume of water passing through the tempering and condenser systems. Of the total mortality, 49.5% was due to mechanical effects and 34.5% due to condenser shock.

A regression of smelt percent mortality, due to mechanical effects vs. their length, was highly negatively correlated ( $r = -0.943$ ). With each millimeter increase in length over 13 mm (the lowest datum used), the chances of death via mechanical damage was reduced by 3% (Fig. 4). These results, although in opposition with other studies, support the physiological observations that larvae, once beyond the critical developmental period, can withstand jostling, abrasion, and related problems the older or more developed they become.

With a cross condenser  $\Delta T$  of 20°C (SD = 4°C), all (96.3%) fish but the mottled sculpin died within 2 hr of condenser passage (Table 3). The reactions of the heat-shocked fish were typical (Coutant, 1970) for fish exposed to high temperatures. The few live smelt emerging from the outfall gates were in shock, exhibiting a loss of equilibrium

and total disorientation. Stressed larvae were rapidly consumed by adult white bass residing in the outfall channel (personal observation; see Coutant, 1970).

Time-course exposure data of Mischuk (1972) and Brunder (1972) for smelt and yellow perch, respectively, suggested that a 100-sec exposure to temperatures above 37°C would be 100% lethal for both species. These levels were common on passage through the Nanticoke Generating Station (Table 4). Schubel (1975) indicated that a  $\Delta T$  greater than 19°C for more than 30 sec was lethal to all fish he had exposed and that at a  $\Delta T$  of 15°C, fish were severely traumatized and highly vulnerable to predators.

The extended thermal exposure in the effluent channel may actually increase the chances of mortality due to heat shock. Dean (1973) showed that thermal exposure was a dose response based on acclimation temperature. He investigated this by exposing fish larvae to various thermal regimes. The species tested could withstand instant exposure to 30°C from an acclimation temperature of 15°C, followed by a 2°/min temperature drop. However, the same species showed a 60 to 100% mortality when exposed to either a gradual 1°/min temperature increase to 30°C followed by a rapid return to the acclimation temperature or a gradual increase followed by a gradual decrease. The Nanticoke station has a slow cooling regime; initial cooling is at 5°/min followed by a <1°/min rate for a time course of 8 to 12 min (Fig. 3). On the basis of Dean's (1973) work, all larvae exposed for these times, at a  $\Delta T$  of from 7 to 17°C, would be near death on entering the lake.

It is unrealistic to assume that all the entrained fish would have survived to a recruitable size. Therefore I applied a general 1% survival rate (from larvae to harvestable recruit) to the 17,967 fish/hr entrainment estimate, obtaining an actual loss to recruitment of 179 fish/hr. The total actual removal during the 120-day larval presence would be 515,520 fish. Were these all commercially

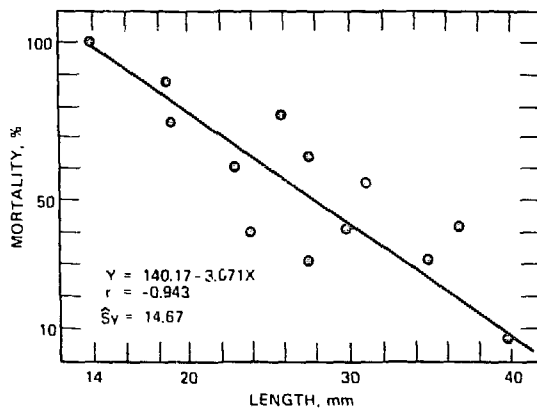


FIG. 4 Relation between length and mortality of entrained young-of-the-year smelt exposed only to ambient lake temperatures during passage through the Nanticoke Generating Station. (Data fitted by least squares regression.)  $S_y$  = standard error of estimate.

TABLE 3 Estimated Percentage Mortality of Smelt (22 to 39 mm) Sampled at Three Locations in the Nanticoke Generating Station Cooling System, July 22 to 26, 1974

| Diel period   | Forebay<br>(4060.8 m <sup>3</sup> sampled/24 hr) |                 |                              | Outfall gate<br>(3865.6 m <sup>3</sup> sampled/24 hr) |                 |                              |                                         | Effluent channel<br>(8022.0 m <sup>3</sup> sampled/24 hr) |                 |                              |                                                    |
|---------------|--------------------------------------------------|-----------------|------------------------------|-------------------------------------------------------|-----------------|------------------------------|-----------------------------------------|-----------------------------------------------------------|-----------------|------------------------------|----------------------------------------------------|
|               | No.<br>captured                                  | Percent<br>dead | No.<br>living/m <sup>3</sup> | No.<br>captured                                       | Percent<br>dead | No.<br>living/m <sup>3</sup> | Forebay to outfall<br>percent mortality | No.<br>captured                                           | Percent<br>dead | No.<br>living/m <sup>3</sup> | Percent total mortality<br>(condenser plus bypass) |
| Dawn and dusk | 16                                               | 31.2            | 0.0027                       | 23                                                    | 100.0           | 0.0000                       | 100.0                                   | 0                                                         |                 |                              |                                                    |
| Nocturnal     | 449                                              | 30.1            | 0.0773                       | 93                                                    | 88.2            | 0.0028                       | 96.3                                    | 181                                                       | 46.4            | 0.0120                       | 84.0                                               |
| Daylight      | 1                                                | 0.0             | 0.0002                       | 3                                                     | 100.0           | 0.0000                       | 100.0                                   | 6                                                         | 100.0           | 0.0000                       | 100.0                                              |
| Total         | 466                                              | 30.0            |                              | 119                                                   | 90.7            |                              |                                         | 187                                                       | 47.5            |                              |                                                    |

TABLE 4 Relation Between Water Temperature and Percentage Mortality of Young Fishes Sampled from Three Locations in the Nanticoke Station Cooling System, June to August 1974

| Diel period    | June 7-12 |                 |              | June 21-27 |                 |              | July 10-12 |                 |              | July 22-25 |                 |              | Aug. 8-13 |                 |              | Aug. 19-21 |                 |              |
|----------------|-----------|-----------------|--------------|------------|-----------------|--------------|------------|-----------------|--------------|------------|-----------------|--------------|-----------|-----------------|--------------|------------|-----------------|--------------|
|                | N         | Percent<br>dead | Temp.,<br>°C | N          | Percent<br>dead | Temp.,<br>°C | N          | Percent<br>dead | Temp.,<br>°C | N          | Percent<br>dead | Temp.,<br>°C | N         | Percent<br>dead | Temp.,<br>°C | N          | Percent<br>dead | Temp.,<br>°C |
| Dawn and dusk  |           |                 |              |            |                 |              |            |                 |              |            |                 |              |           |                 |              |            |                 |              |
| Forebay        | 3         | 100             | 13           | 0          |                 | 14           | 1          | 100             | 16           | 16         | 31              | 19           | 0         |                 | 18           | 1          | 100             | 22           |
| Outfall gates  | 5         | 100             | 35           | 0          |                 | 35           | 0          |                 | 37           | 23         | 100             | 38           | 0         |                 | 18           | 7          | 14              | 22           |
| Effluent canal | 6         | 100             | 20           | 0          |                 | 15           | 1          | 100             | 18           | 0          | 100             | 21           | 0         |                 | 18           | 9          | 22              | 22           |
| Night          |           |                 |              |            |                 |              |            |                 |              |            |                 |              |           |                 |              |            |                 |              |
| Forebay        | 17        | 100             | 16           | 1          | 100             | 14           | 5          | 40              | 16           | 431        | 27              | 19           | 0         |                 | 18           | 3          | 33              | 22           |
| Outfall gates  | 3         | 100             | 34           | 7          | 84              | 35           | 8          | 87              | 37           | 94         | 100             | 38           | 6         | 83              | 18           | 9          | 33              | 22           |
| Effluent canal | 10        | 70              | 21           | 5          | 100             | 15           | 73         | 90              | 18           | 189        | 49              | 21           | 0         |                 | 18           | 23         | 30              | 22           |
| Day            |           |                 |              |            |                 |              |            |                 |              |            |                 |              |           |                 |              |            |                 |              |
| Forebay        | 3         | 0               | 15           | 0          |                 | 14           | 0          |                 | 16           | 1          | 0               | 19           | 0         |                 | 18           | 0          |                 | 22           |
| Outfall gates  | 0         |                 | 35           | 1          | 100             | 35           | 0          |                 | 37           | 3          | 100             | 38           | 1         | 100             | 18           | 2          | 0               | 22           |
| Effluent canal | 0         |                 | 21           | 5          | 100             | 15           | 1          | 100             | 18           | 6          | 100             | 21           | 0         |                 | 18           | 1          | 0               | 22           |



valuable fish, e.g., smelt, at 12 fish/kg, they would equal 43 tonnes of fish lost by entrainment to that year class. This figure is equal to 12% of the catch of a single eastern-basin Lake Erie trawl fisherman (based on 1972-1974 records). The retail value of such a catch, at 70¢/kg (1974 prices) would be \$30,000.

Only one study to date has estimated the reduction in a fish population after many years of plant operation. Hess, Sissenwine, and Saila (1975) simulated the impact of three decades of entrainment on flounder (*Pseudopleuronectes americanus*) in Long Island Sound, New York. They calculated a 1% annual recruitment loss and a 6% decrease in the population after 35 years of plant operation.

Nanticoke's long-term impact has yet to be determined; however, its immediate influence on the 1974 smelt year class was not alarming. My estimates must be viewed with reservation since I assumed small year to year fluctuations in such variables as spawning success, fish distribution, and hatching success when, in fact, they could vary by 50% annually. Once the plant is in full operation, a clearer evaluation of its total impact on the fish stocks will be possible.

The three overriding parameters controlling power plant young-of-the-year fish losses are intake locations, the volume of water drawn into the generating station, and outfall port design.

Shore and river intakes, at either surface or bottom locations, remove water from the highly productive littoral zone and thus have a far greater rate of fish entrainment than near-bottom offshore intake ports. Most larval and juvenile fish exhibit vertical phototactic movements, spending up to 18 hr of their diel cycle in the bottom and surface layers and only 6 hr migrating between the two areas (personal observation). Surface or bottom intakes are thus situated in strata of high larval and juvenile fish densities. The location of intake ports one-third the distance off the bottom would aid in reducing larval entrainment.

The fish loss could also be reduced by decreasing the total volume of coolant water used (Marcy, 1975; Schubel, 1975). As most entrained larvae are killed by mechanical and heat shock, reducing the total volume of coolant used would expose fewer organisms to the once-through cooling process. On the basis of the mortality calculation in Table 3, if total entrainment mortality is proportional to the total volume of water withdrawn, the fish loss at the Nanticoke Generating Station could be reduced 59% by simply eliminating the tempering (effluent cooling) water system. Great care must be taken to carefully evaluate the benefits of this method. The discharge of very hot effluent into a thermally sensitive area, i.e., an estuary or spawning river, may be far more detrimental than the death of millions of entrained larval fish.

Finally, replacing the long effluent outfall channel with short discharge ports could reduce the thermal trauma described by Dean (1973). This would allow any surviving larvae to recover from the condenser passage and hopefully elude receiving-water predators.

## ACKNOWLEDGMENTS

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# Survival of Juvenile Fishes Receiving Thermal and Mechanical Stresses in a Simulated Power-Plant Condenser

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## ABSTRACT

Six species of larval fish and one species each of frog, tadpole, and zooplankton were subjected to fluid-induced stresses (i.e., turbulence, pressure, and vacuum) developed in the condenser portion (excluding the pumps) of a steam power plant. Results are presented for three species (*Morone saxatilis*, *Lepomis macrochirus*, and *Daphnia magna*). Results for all larval fish species were similar. Fluid-induced stresses were examined separately and in combination with thermal stresses. Fluid-induced stresses were generally not fatal to larval fish at velocities up to 5.8 m/sec. Sublethal effects were observed for *Cyprinus carpio*. *Daphnia magna* had 10 to 30% mortality at 4.2 m/sec loop water velocity. When larval *Morone saxatilis* were exposed to combined thermal and fluid-induced stresses, the resulting mortalities were equivalent to those from thermal stresses alone.

Larval fish and other planktonic organisms that are caught in the cooling-water intake current and pass through the condenser of a steam electric-power plant are subjected to a variety of stresses. Thermal stress resulting from heat released by the condenser is the most commonly recognized (Coutant, 1970), but there are also mechanical stresses resulting from physical contact of the organism with the inlet screen, pump impeller, condenser tubing, etc., and often chemical stresses from chlorine and other chemical additions. Also, there are a variety of stresses that can be grouped under the heading fluid-induced; these include the shear and buffeting of being in a turbulent field (Cox and Mason, 1971; Hinze 1955), rapid acceleration and deceleration as the water moves through the pump and condenser, pressurization by the pump, and exposure to vacuum as a result of fluid-dynamic-design parameters. These stresses often act together or in close sequence. Many of them can be lethal and, because of the large amounts of cooling water involved, can exert a significant impact on the environment.

Investigations of the responses of organisms to these stresses at operating power stations, while desirable, are inadequate for several reasons. Utility companies generally cannot allow experimental manipulation of such operating

parameters as flow rates and temperatures while maintaining electrical production. Intakes, condensers, and discharges were not designed for biological sampling and often preclude sampling at locations critical to proper measurements. Organism densities are often too low or too patchy in natural waters to provide adequate numbers of the right species for experiments. The investigator usually cannot extend his experimental range from high effects to effect thresholds within the constraints of existing engineering designs.

If the objective is to determine the survival thresholds for various physical and chemical parameters operating in a condenser system (with the intent of modifying engineering design or operation), a controllable experimental apparatus is necessary. In laboratory investigations organisms may be exposed to either individual stress or combinations of stresses. The advantage of experiments involving multiple-stress exposure is that synergistic effects of combined stresses can be assessed.

We conducted an experiment in which a variety of aquatic organisms (Table 1) were exposed to most of the stresses generated in one segment of a condenser cooling-water system, i.e., a simulated condenser tube. Six species of fish were tested (three at high thermal conditions), and one species of frog tadpole and one zooplankton species were tested. Mortality results of only three species (striped bass, bluegill and *Daphnia*) are presented here. Mortality results of all other larval fish species tested are similar. Sublethal results for carp are discussed. A topical report in which all results are described in detail is forthcoming.

## EQUIPMENT AND PROCEDURES

Organisms were added to a recirculating water loop (Fig. 1) in which they were forced through a heat-exchanger tube typical of a steam power plant. The organisms did not go through the pump, however, but only

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TABLE 1 Summary of Experiments

| Species                                              | Size, mm                     | Minimum and maximum water velocity in tube,* m/sec | Thermal stresses involved |        | Results                          |
|------------------------------------------------------|------------------------------|----------------------------------------------------|---------------------------|--------|----------------------------------|
|                                                      |                              |                                                    | Sublethal                 | Lethal |                                  |
| Striped bass<br>( <i>Morone saxatilis</i> )          | 4-6                          | 2.1, 5.8                                           | X                         | X      | See Figs. 3 and 4 and text       |
| White bass<br>( <i>Morone chrysops</i> )†            | 6-9                          | 2.1, 5.8                                           | X                         | X      | ‡                                |
| Bluegill<br>( <i>Lepomis macrochirus</i> )           | 10-12                        | 2.1, 4.6                                           | X                         |        | See Table 2 and text             |
|                                                      | 25-30                        | 2.1, 4.6                                           | X                         |        | See Table 2 and text             |
| Carp<br>( <i>Cyprinus carpio</i> )                   | 6-10                         | 2.1, 5.8                                           | X                         |        | ‡ and text for sublethal effects |
| Brook silversides<br>( <i>Labidesthes sicculus</i> ) | 16-22                        | 2.1, 5.8                                           | X                         |        | ‡                                |
| Mosquito fish<br>( <i>Gambusia affinis</i> )         | 6-30                         | 2.1, 5.8                                           | X                         | X      | ‡                                |
| Frog tadpoles<br>( <i>Rana catesbeiana</i> )         |                              | 2.1, 5.8                                           | X                         |        | ‡                                |
| Zooplankton<br>( <i>Daphnia magna</i> )              | 1.5-2<br>(largest available) | 2.1, 4.6                                           | X                         |        | See Table 3 and text             |

\* Fluid-induced stresses:

| Water velocity, m/sec | Stresses imposed on organism |             |                              |
|-----------------------|------------------------------|-------------|------------------------------|
|                       | Positive pressure, atm       | Vacuum, atm | Water velocity               |
| 2.1                   | 0.3                          | ~0.5        | Typical of steam power plant |
| 4.6                   | 1.2                          | None        | 2.1 x typical                |
| 5.8                   | 2.0                          | None        | 2.7 x typical                |

† Wild larvae collected during white bass spawning season; identification tentative.

‡ As noted in the text, fluid-induced stresses generated by our loop, either alone or combined with lethal thermal stresses, resulted in no more mortalities to larval fish than did thermal stress alone. Because of space limitations, a detailed discussion with results for all species cannot be included here. A topical report in which all results will be described in detail is forthcoming.

through the tube and a collecting device. We sought primarily to identify gross mortality, but sublethal effects were also noted and recorded.

The loop consisted of a pump, flow-rate-control station, loop cooler, fish-addition station, 12 m of heat-exchanger tubing, and fish-recovery station. Except for small amounts of plastic in gaskets, valve packing, and bellows, all materials exposed to the water were stainless steel or glass. The plankton net in the recovery station was nylon. The pump was a canned-motor type so that lubricants could not contaminate the water.

The loop operated as follows (letters refer to Fig. 1): While the pump was on and the bypass valve was open, the fish-addition station (A) could be isolated and opened and a sample of fish or other organisms gently poured in. When the isolation valves were opened and the bypass valve closed, the fish were forced through the experimental tube (B). After passing through the tube, they were recovered with a plankton net (C) suspended in a 200-liter drum. The fish were added to the loop downstream of the pump and recovered upstream. They did not go through the pump because it was not typical of a power-plant cooling-water pump.

Most Tennessee Valley Authority steam power plants have condenser tubes that are 22 mm in outside diameter (OD) and range in length from 9 to 18 m, with most at the shorter end of the range. Thus the 12 m of 22-mm tubing installed in our loop was representative of a typical system. Typical coolant-water velocity in a condenser tube is 2.1 m/sec. Our loop had water velocity capability up to 5.8 m/sec. The U-shaped heat-exchanger tube was mounted vertically so that a vacuum could be developed near the top of the inverted U. Vacuum occurs in the condensers of a number of power plants. The approach and discharge lines to the heat-exchanger tube were 50-mm inside diameter (ID) glass pipe. Thus the tube inlet and outlet configurations had a sudden contraction and expansion, respectively (velocity in the tube was 7.2 x velocity in glass pipe), partially simulating the power-plant condenser headers.

In our experiments the lowest velocity at which fish were sent through the tube was 2.1 m/sec and the highest was 5.8 m/sec. At 2.1 m/sec, a vacuum of about 0.5 atm was developed at the top of the U tube (Fig. 2); this reflected its vertical orientation. At 5.8 m/sec, the fish were subjected to a positive pressure of about 2 atm gauge at the fish-addition station; this was high enough to suppress the

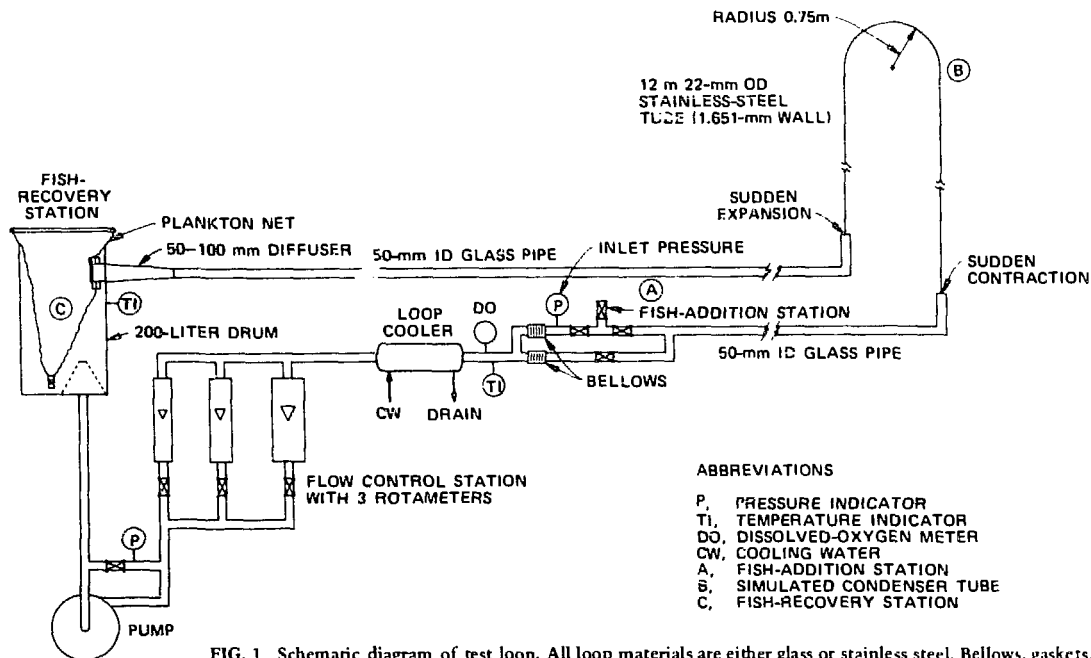


FIG. 1 Schematic diagram of test loop. All loop materials are either glass or stainless steel. Bellows, gaskets, and valve seals are plastic, and a canned-motor pump is used.

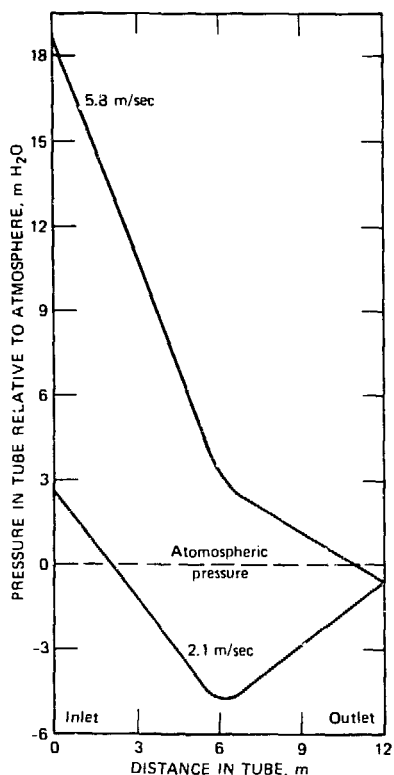


FIG. 2 Pressure in tube relative to atmosphere (m H<sub>2</sub>O).

vacuum in the U tube. The turbulent-energy dissipation rate, a measure of the amount and intensity of turbulence, was  $0.00132 \text{ cal sec}^{-1} \text{ g}^{-1}$  at  $2.1 \text{ m/sec}$  and  $0.0217 \text{ cal sec}^{-1} \text{ g}^{-1}$  at  $5.8 \text{ m/sec}$ .

The loop cooler worked against heat added to the system by the pump to control loop temperature. Precision was  $\pm 0.1^\circ\text{C}$  at high lethal temperatures. Several experiments were conducted when the loop temperature was higher than the organism's acclimation temperature. Thus organisms received a thermal shock when they were placed in the loop, as well as mechanical and fluid-induced stresses in passing through the loop.

The loop was normally filled with well or pond water, which was aerated by bubbling air into the collection drum for 1 to 2 hr before the first run. Aeration continued at night when the loop was shut down. The measured dissolved oxygen content was always 95 to 100% of saturation.

The extent of testing of each species was determined largely by its availability. If only a few larvae were available, they were exposed only to fluid-induced stresses. If large numbers were available, the organisms were exposed first to fluid-induced stresses alone and then to thermal stresses superimposed on the fluid-induced stresses. Sample sizes ranged from 10 to 100 fish, depending on availability, size, and other considerations. Our experiments were conducted using only small organisms believed to represent sizes capable of being entrained at power stations. All the fish were less than 30 mm in length, and most were less than about 12 mm. Carp were tested the day after hatching.

Most fish were raised at the Aquatic Ecology Laboratory at Oak Ridge National Laboratory. After acclimation at a predetermined temperature, they were transported to the loop, about 8 miles away. For low-temperature runs this transfer presented no problems, but, for high-temperature runs, where acclimation temperature was more critical, the holding (and thus acclimation) temperature tended to drift during the daily trip and also during temporary storage. Small changes in acclimation temperature should not affect our conclusions, however, since only relative mortalities are used in the analysis of results.

Two kinds of controls were used. In the first (barrel controls), the fish were subjected to handling stresses, principally during collection from the loop. For this sampling fish were poured into the collecting barrel from a height of about 150 mm and recovered with the plankton net. Usually a barrel control was taken after each run, with the water in the loop and barrel at the same temperature as during the run. For the second control (absolute control), the fish were simply transferred very gently from the transport bucket into a beaker of loop water. The only stresses to which these fish were subjected were those involved in being transported to the loop and living in small groups in a laboratory rather than in their natural environment or previous rearing conditions.

In thermal-shock experiments in the loop, the length of thermal exposure was controlled to either 5 or 6 min. Time zero was taken when the fish were poured from a beaker into the hotter fish-addition station (Fig. 1). At the end of the exposure time, the bucket at the bottom of the collection net, which by this time contained the fish, was lowered into a beaker of water at ambient temperature. Barrel controls for the thermal-stress runs were conducted similarly. Time zero was when the sample was poured into the hotter barrel; at the end of the exposure time, the sample was retrieved in the same manner as in the experimental runs. For the thermal-shock experiments, the loop temperature was chosen so that part, but not all, of the sample would be killed during the exposure.

Generally, from 90 to 95% of the fish were recovered from the loop in these experiments. Unrecovered fish were usually stuck to the plankton net or sometimes lost in the loop piping. Computations of the percent mortality were based on the number of fish recovered. All samples were maintained at  $\sim 20^{\circ}\text{C}$  for a few days after their run and were observed to determine latent mortalities.

Some modifications of the general procedures were necessary to accommodate each species. Most of these changes are noted, along with results, in the following section.

Considerable difficulty was experienced in conducting experiments with the zooplankton, *Daphnia magna*, because the water used in the loop affected its survival. Clear, fresh well water was necessary to achieve good survival. The loop was originally built with copper alloys in some of its components, and these apparently released small amounts of copper in the solution. Since *Daphnia* are quite

susceptible to copper poisoning, the copper alloys had to be eliminated. Also, the zooplankton had to be handled with much more care than the larval fish. For example, in the seemingly trivial process of pulling the plankton net out of the barrel, the net had to be removed extremely slowly (over several minutes) and washed down continuously to avoid handling mortalities.

## RESULTS

Results of experiments with bluegill are shown in Table 2. The fish were segregated by size; those run on July 30 averaged 2.5 times longer than those tested on July 29. Because of a limited supply, bluegill were subjected to fluid-induced stresses only. There were no mortalities of fish of either size immediately after passage through the loop at two velocities. Observations of latent mortalities were hampered by massive die-offs attributed to inadequacy of the holding facilities. For 2 days after the runs on July 29, only 2% of one sample (one fish) died. On the third day, however, large numbers of bluegill started dying in some samples. These fish were not fed. The larger fish run on July 30 were fed brine shrimp the next morning (July 31). On the following day they started dying in large numbers.

Because striped bass are commercially important, and since sufficient larvae were available, tests were conducted at both sublethal and lethal thermal conditions (Coutant and Kedl, 1975). At the time of the tests, larvae were about 2 weeks old, were 4 to 6 mm in length, and were acclimated at  $21^{\circ}\text{C}$ . Sample sizes ranged from 40 to 100 fish.

The results for thermally nonlethal runs for striped bass passing through the simulated heat-exchanger tube at 2.1 (Fig. 3a) and 5.6 m/sec (Fig. 3b) show that the immediate mortality (i.e., the fraction dying within an hour after passage through the loop) was less than 5%. These fish died continuously with time and at a statistically significant rate, which was typical for many of the striped bass samples we held for long-term surveillance. Results of three absolute control samples taken from the same container of fish as these runs (Fig. 3c) show that the controls died at about the same rate as the stressed fish. Actually, Fig. 3b indicates that the latent mortality of fish passing through the tube at the highest velocity was less than that of the absolute controls. We prefer not to conclude that this exposure was actually beneficial to the fish. Comparing the results shown in Fig. 3, we concluded that these striped bass experienced no latent mortalities over and above those of the absolute and barrel controls, within the precision of the experiment.

Typical results with striped bass when the loop temperature was in a lethal range are shown in Fig. 4. Loop temperatures were  $31.0$  and  $31.9^{\circ}\text{C}$ , and the exposure time was 6 min. In both cases the velocity through the tube was 5.8 m/sec. Immediate mortalities were similar for loop-run samples and barrel controls, and the rate of subsequent death for loop-run samples and barrel controls was similar. The higher mortalities in Fig. 4b are caused by the higher loop temperature. Although absolute controls are shown

TABLE 2 Summary of Selected Runs with Bluegill\*  
Subjected to Fluid-Induced Stresses Only

| Run number | Conditions       |     | Total mortality,<br>% dead fish/hr after test |       |                  |
|------------|------------------|-----|-----------------------------------------------|-------|------------------|
| 7/29/1     | Absolute control | 0/1 | 0/22                                          | 0/46  | 0/70             |
| 7/29/2     | Absolute control | 0/1 | 0/22                                          | 0/46  | Not recorded     |
| 7/29/3     | Barrel control   | 0/1 | 0/22                                          | 0/46  | 20/70            |
| 7/29/4     | Barrel control   | 0/1 | 0/22                                          | 0/46  | 12/70            |
| 7/29/5     | 2.1 m/sec, tube  | 0/1 | 0/20                                          | 0/44  | 0/68             |
| 7/29/6     | 2.1 m/sec, tube  | 0/1 | 2/20                                          | 2/44  | 2/68             |
| 7/29/7     | 4.6 m/sec, tube  | 0/1 | 0/20                                          | 0/44  | 2/68             |
| 7/29/8     | 4.6 m/sec, tube  | 0/1 | 0/20                                          | 0/44  | 2/68             |
| 7/30/1     | Absolute control | 0/1 | 0/22                                          | 45/46 | 60/70            |
| 7/30/2     | Absolute control | 0/1 | 0/22                                          | 15/46 | 45/70            |
| 7/30/3     | Barrel control   | 0/1 | 0/22                                          | 5/46  | Discarded sample |
| 7/30/4     | Barrel control   | 0/1 | 0/22                                          | 25/46 | 40/70            |
| 7/30/5     | 2.1 m/sec, tube  | 0/1 | 0/22                                          | 15/46 | 15/70            |
| 7/30/6     | 2.1 m/sec, tube  | 0/1 | 0/22                                          | 15/46 | 25/70            |
| 7/30/7     | 4.6 m/sec, tube  | 0/1 | 0/22                                          | 5/42  | 15/70            |
| 7/30/8     | 4.6 m/sec, tube  | 0/1 | 0/22                                          | 20/46 | 25/70            |

\*Fish in the 7/29 runs were 10 to 12 mm in length; 50 fish were tested in each run. Those in the 7/30 runs were 25 to 30 mm in length, and 20 were tested in each run. The fish in the 7/29 runs were not fed; those in the 7/30 runs were fed after 22 hr.

for both temperatures, they cannot be used for comparison because they were not subjected to thermal shock. For striped bass the mortalities resulting from fluid-induced stresses generated in the loop superimposed on lethal thermal stress were not significantly greater than the mortalities induced by the thermal stress itself, within the precision of the experiments.

Results of experiments with *Daphnia magna* are shown in Table 3. These experiments were conducted with the loop at 25 to 30°C so that the zooplankton were not subjected to significant thermal shock. Sample size for each run was 10 of the largest individuals (1.5 to 2.0 mm) in the culture. Zooplankton were passed through the simulated heat-exchanger tube at velocities of 2.1 and 4.6 m/sec. Typically, mortalities occurred for about a day after the run, even in the absolute controls. Table 3 shows two mortality columns for each velocity, one for immediate mortality and one for total mortality 1 day later. The samples were kept for about 3 days for surveillance. There were no significant additional mortalities after the first day. From the table we can make the following observations:

1. The average immediate mortalities for barrel controls were significant (11 and 3%) and were not very different from the average immediate mortalities for loop-run samples (8 and 5%). Fish forced through the tube, at either velocity, seemed to have no significant additional immediate mortalities above those resulting from the process of retrieval with the plankton net (barrel controls).

2. The average mortality of all samples and controls had increased significantly by the next day.

3. Although the scatter prevents a definitive conclusion, at 2.1 m/sec there seems to be no additional latent mortality of the run samples over either the absolute or barrel controls after 1 day.

4. At 4.6 m/sec, latent mortality for loop-run plankton seems to be considerably higher than that for either the absolute or barrel controls after 1 day. The differences seem too great to be attributed to data scatter. We concluded that significant mortalities may be experienced by *Daphnia magna* at 4.6 m/sec in the heat-exchanger tube. This mortality, estimated at 10 to 30%, occurs over a period of time that may be as long as 1 day.

Some sublethal effects were observed, but only observations on carp are presented here. Carp removed from the loop were very lethargic and displayed a notable loss of activity. This was observed when two petri dishes, one containing fish that had gone through the loop and the other containing controls, were placed side by side. When someone passed a hand over the dishes to make shadows, the fish that had been through the loop reacted and swam rather slowly compared with the controls, which darted around rapidly in a fright response. By the next morning the loop-run fish had completely recovered. This differential behavior after nonthermal runs was not observed in other fish species tested.

Results of experiments with other larval fish species tested (Table 1) were generally the same as for the species reported here. Fluid-induced and mechanical stresses imposed on larval fish by our loop were only minimally lethal (less than ~5%) at velocities up to 5.8 m/sec. We also

TABLE 3 Summary of Results with Zooplankton (*Daphnia magna*)

| Condition         | Velocity in cube         |                                |                                   |                          |                                |                                   |
|-------------------|--------------------------|--------------------------------|-----------------------------------|--------------------------|--------------------------------|-----------------------------------|
|                   | 2.1 m/sec                |                                |                                   | 4.6 m/sec                |                                |                                   |
|                   | Total number of plankton | Average immediate mortality, % | Average mortality after 1 day,* % | Total number of plankton | Average immediate mortality, % | Average mortality after 1 day,* % |
| Loop-run samples  | 80                       | 8                              | 16                                | 60                       | 5                              | 31                                |
| Barrel controls   | 80                       | 11                             | 21                                | 60                       | 3                              | 8                                 |
| Absolute controls | 30                       | 0 (by definition)              | 14                                | 20                       | 0 (by definition)              | 5                                 |

\*Very few additional mortalities occurred after 2 additional days of surveillance.

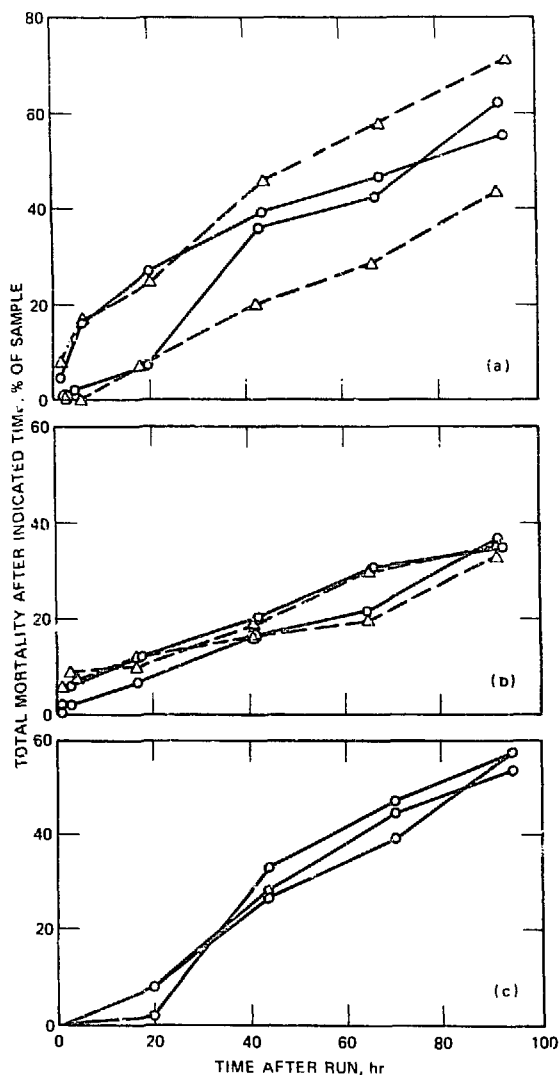


FIG. 3 Percent mortality of larval striped bass tested at a nonlethal temperature. (a) Loop velocity, 2.1 m/sec, low  $\Delta T$ .  $\circ$ — $\circ$ , tests 2 and 4.  $\triangle$ — $\triangle$ , barrel control samples 3 and 5. (b) Loop velocity, 5.6 m/sec, low  $\Delta T$ .  $\circ$ — $\circ$ , tests 6 and 8.  $\triangle$ — $\triangle$ , barrel control samples 7 and 9. (c) Absolute control samples 1, 2, and 3.

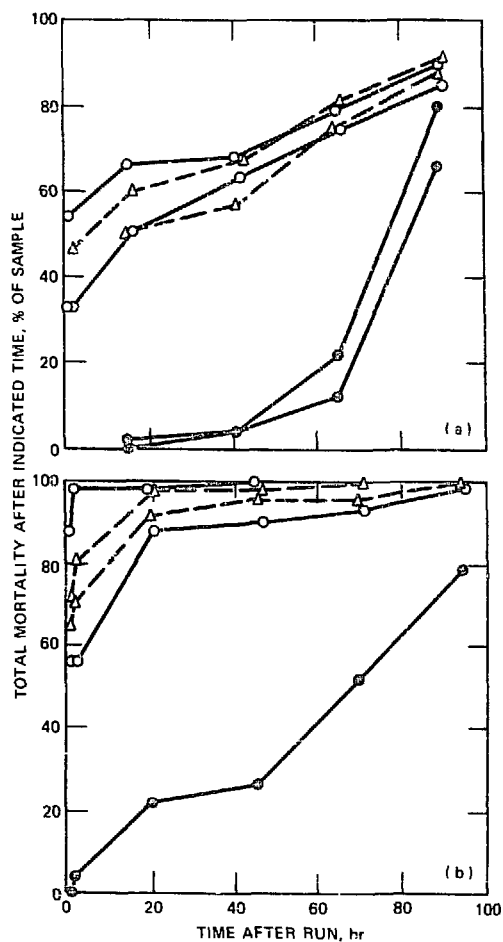


FIG. 4 Percent mortality of larval striped bass tested at a lethal temperature and a loop velocity of 5.8 m/sec. (a) Loop temperature, 31.0°C.  $\circ$ — $\circ$ , tests 8 and 10, high  $\Delta T$ .  $\triangle$ — $\triangle$ , barrel control samples 9 and 11, high  $\Delta T$ .  $\circ$ — $\circ$ , absolute control samples 12 and 13, no  $\Delta T$ . (b) Loop temperature, 31.9°C.  $\circ$ — $\circ$ , tests 3 and 7, high  $\Delta T$ .  $\triangle$ — $\triangle$ , barrel control samples 4 and 6, high  $\Delta T$ .  $\circ$ — $\circ$ , absolute control sample 5, no  $\Delta T$ .

found that for the larval fish tested, when lethal thermal stresses were superimposed on fluid-induced stresses, the resulting mortalities were equivalent to those for thermal stress alone.

## DISCUSSION

These experiments suggest minimal mortalities to the species tested when they were exposed to fluid-induced stresses generated in a condenser tube typical of that in a steam power plant. Additional mortalities beyond those of the controls were generally <5%. Our results substantiate earlier work by Kerr (1953), who passed striped bass through condenser tubes at the Contra Costa, Calif., power plant. We had anticipated greater mortalities with the small and seemingly fragile larvae. Studies by Morgan and coworkers (1973) defining tolerable shear levels for striped bass eggs and larvae seem to be relevant to our studies. They are not directly comparable, however, since their experiments used idealized laminar flow, which is quite different from the highly turbulent flow in a condenser tube. Thermal resistance of organisms passed through the loop or handled as barrel controls was similar to that shown in laboratory bioassays by us and others (Kelly and Chadwick, 1971).

Care must be exercised in extrapolating these results to the condenser of a real power plant. In a real situation the larvae may experience stresses that are greater than or other stresses in addition to those simulated here; e.g., cavitating conditions may exist in a power plant. We simulated a vacuum of about 0.5 atm, but this is not equivalent to cavitation. In addition, the vacuum profile simulated by this experiment was peaked (inverse of Fig. 2), and the maximum of 0.5 atm lasted only momentarily. If the vacuum lasted for a longer period of time or was higher than we simulated, greater mortality might occur to the larval fish than was the case in our simulation.

Our conclusions differ from results obtained at several operating power stations (Beck and Lackie, 1974). Part of this discrepancy suggests a hypothesis for further experimentation (one also suggested by Beck and Lackie's

review); namely, that much observed physical damage in power-plant cooling systems may be derived from the pump rather than from fluid stresses in the condenser.

## ACKNOWLEDGMENTS

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## MINERAL CYCLING IN SOUTHEASTERN ECOSYSTEMS

*Edited by Fred C. Howell, John B. Gentry, and Michael H. Smith  
Savannah River Ecology Laboratory  
Proceedings of a Symposium held May 1-4, 1974, Augusta, Georgia*

This symposium dealt with the modeling of ecosystems and the cycling of natural and man-made elements in terrestrial, freshwater, and marine environments. The 66 papers included in the proceedings reflect in many ways the state of the art in mineral-cycling research. Most of the studies reported document and compare elemental concentrations in various ecosystem compartments, with emphasis on verification of techniques, documentation of data with statistically determined confidence intervals, interpretation of results, and synthesis leading to development of concepts for this area of ecology. Although the primary emphasis is on mineral cycling in the southeastern United States, reports of work in other geographic areas are included.

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## THERMAL ECOLOGY

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Savannah River Ecology Laboratory*

Proceedings of the first symposium on the environmental effects of heated effluents, held at Augusta, Georgia May 3-5, 1973

The release of vast quantities of heated water from both fossil-fueled and nuclear power plants raises questions regarding the impact of thermal effluents on natural ecosystems. Contributions from thermal research programs are discussed by representatives from industry, government, and academia in the 55 papers in this volume. The papers are divided into nine sections: physiochemical phenomena, physiological ecology, thermal tolerance and adaptation, population ecology, behavioral ecology, community ecology, productivity, diversity, and models.

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# Program

## Second Thermal Ecology Symposium

APRIL 2 - 5, 1975  
EXECUTIVE HOUSE AUGUSTA  
AUGUSTA, GEORGIA

*Sponsored by:*

SAVANNAH RIVER ECOLOGY LABORATORY

INSTITUTE OF ECOLOGY  
UNIVERSITY OF GEORGIA

UNITED STATES ENERGY RESEARCH AND  
DEVELOPMENT ADMINISTRATION

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## SCHEDULE OF EVENTS

### Wednesday, April 2

### Executive House Augusta

|                         |                                                            |           |
|-------------------------|------------------------------------------------------------|-----------|
| 8:00 a.m. - 5:30 p.m.   | Registration Desk Open                                     | The Foyer |
| 10:00 a.m. - 12:00 noon | Annual Meeting of the Society of Power Industry Biologists | Grove I   |
| 2:00 p.m. - 5:00 p.m.   | <b>OPENING SESSION</b>                                     | Grove I   |

"An overview of thermal ecology at the individual, population and ecosystem levels of biological organization"

Chairman: F. B. Golley  
University of Georgia

**Featured Speakers:**

Victor Hutchison,  
University of Oklahoma

J. Whitfield Gibbons  
Savannah River Ecology Laboratory

John Cairns, Jr.  
Virginia Polytechnic and State University

6:30 p.m.

**BANQUET**

**EMBASSY ROOM**  
(Executive House  
Convention Center  
738 Ellis St.)

**Featured Speaker:**

Eugene P. Odum, Director  
Institute of Ecology  
University of Georgia

Address: "The Power Park Concept  
Ameliorating Man's Disorder with  
Nature's Order"

### Thursday, April 3

|                        |                                                     |           |
|------------------------|-----------------------------------------------------|-----------|
| 8:30 a.m. - 5:00 p.m.  | Registration Desk Open                              | The Foyer |
| 8:30 a.m. - 11:30 a.m. | <b>SESSION I-A<br/>THERMAL ECOLOGY—<br/>GENERAL</b> | Grove I   |

Chairman: J. P. Giesy, Jr.  
Savannah River Ecology  
Laboratory

D. C. Adriano  
Savannah River Ecology  
Laboratory

8:30 a.m. - 11:45 a.m.

**SESSION I-B  
THERMAL TOLERANCE  
AND ADAPTATION**

Grove II

Chairman: W. Adair  
Duke Power Company

J. Clugston  
University of Georgia

2:00 p.m. - 5:15 p.m.

**SESSION II-A  
THERMAL ECOLOGY—  
GENERAL**

Grove I

Chairman: R. V. Dimock, Jr.  
Wake Forest University

M. J. Schneider  
Battelle Northwest  
Laboratory

2:00 p.m. - 4:45 p.m.

**SESSION II-B  
BEHAVIORAL ECOLOGY**

Grove II

Chairman: H. Eure  
Wake Forest University

J. R. Spotila  
State University College  
at Buffalo

7:30 p.m. - 10:00 p.m.

**SPECIAL PROGRAM**

Grove I

"Environmental programs of several  
power companies in the Southeastern  
United States"

Chairman: Loren Jensen  
Ecological Analysts, Inc.

**Panel Members:**

William Adair  
Duke Power Company

Terry Arnold  
Alabama Power Company

Donald Stevenson  
Carolina Power and Light Company

Karen Wilson  
Florida Power Corporation

Robert Woodall  
Georgia Power Company

William Wrenn  
Tennessee Valley Authority

An open discussion between panel members  
and the audience will follow brief presentations.

**Saturday, April 5    FIELD TRIP – Savannah River Plant**

9:00 a.m. - 4:30 p.m.

A field trip is scheduled to various Savannah River Ecology Laboratory environmental research areas on the Savannah River Plant. Inquire at registration desk for information (field clothes are suggested).

The field trip will begin at 9:00 a.m. A shuttle bus will transport all participants to the site; it will depart from the Executive House Augusta and return to the hotel at approximately 4:30 p.m. Sack lunches will be provided.

Sites on the Savannah River Plant to be visited include artificial streams, natural streams, thermal streams, cypress swamp, Par Pond Reservoir and the Thermal Effects Laboratory of the Savannah River Laboratory (Dupont).

**Wednesday, April 2**

**OPENING SESSION —**

"An overview of thermal ecology at the individual, populations and ecosystem levels of biological organization"

Grove I

Chairman: F. B. Golley  
University of Georgia

2:00 p.m.

Welcoming Address: F. B. Golley  
University of Georgia

2:15

Victor Hutchison, University of Oklahoma

3:00

Break

3:15

J. Whitfield Gibbons, Savannah River Ecology Laboratory

4:00

John Cairns, Jr., Virginia Polytechnic and State University

**Thursday, April 3**

**SESSION I-A THERMAL ECOLOGY—  
GENERAL**

Grove I

Chairman: J. P. Giesy, Jr.  
Savannah River Ecology Laboratory

8:30 a.m.

- (1) Biological perspectives in regulation of marine cooling water use. D. C. Miller, Environmental Protection Agency

8:45

- (2) An analysis of the EPA requirements for heated waste water discharge — A site specific case history evaluation. J. Cairns, Jr., D. S. Cherry, K. L. Dickson and J. R. Stauffer, Virginia Polytechnic Institute and State University

9:00

- (3) Practicability of quantitative thermal assessment strategies. J. B. Buffington, R. K. Sharma and I. P. Murarka, Argonne National Laboratory

9:15

- (4) Thermal effluents from a power plant into Card Sound, Florida: Effects before, during and after. A. Thorhaug, S. D. Bach, M. Josselyn, R. F. Hixon and I. M. Brook, University of Miami

9:30

- (5) The Connecticut River ecological study — A summary. D. Merriman, Yale University

9:45

- (6) Thermal effluent in coastal rivers, a contribution to the locations of power plants. R. C. Baird and J. A. Watson, University of South Florida and Geo-Marine, Inc.

Chairman: D. C. Adriano

Savannah River Ecology Laboratory

10:00

- (7) Postoperational impact of the Palisades Nuclear Power Plant, R. S. Benda, Aquinas College

- 10:15 (8) Thermal effects at the Point Beach Nuclear Power Station. R. V. Harmsworth, Limnetics, Inc.
- 10:30 (9) A study of the physical and biological effects of thermal discharge on La Cygne Lake. F. B. Cross, R. W. Drenner, D. G. Gilman, W. J. O'Brien, J. R. Triplett, J. Wagner, G. P. Willhite, Y. S. Yu, University of Kansas
- 11:00 (10) Local environmental effects of a natural draft cooling tower plume. A. L. Tate, Emory University
- 11:15 (11) Water quality evaluation of the Mississippi River near the Quad-Cities Station during diffuser pipe discharge. T. D. Larker, Industrial Bio-Test Laboratories, Inc.

Thursday, April 3

**SESSION I-B THERMAL TOLERANCE AND ADAPTATION**

Grove II

Chairman: W. Adair  
Duke Power Company

- 8:30 (12) Sources of variation in estimating thermal tolerances of dragonfly nymphs from thermal and natural environments. W. J. Martin, C. T. Garten, Jr. and J. B. Gentry, Birmingham-Southern College and Savannah River Ecology Laboratory
- 8:45 (13) Thermal tolerance of dragonfly nymphs from thermal and natural environments. C. T. Garten, Jr. and J. B. Gentry, Savannah River Ecology Laboratory
- 9:00 (14) Chronic *in situ* exposure of the bay scallop to power plant effluents. J. F. Studt and N. J. Blake, University of South Florida
- 9:15 (15) Thermal tolerance of the adult Asiatic clam *Corbicula manilensis* (Mollusca:Bivalvia). J. S. Mattice and L. L. Dye, Oak Ridge National Laboratory.
- 9:30 (16) Temperature tolerance of freshwater fishes. S. E. Peterson and R. M. Schutsky, Ichthyological Associates, Inc.
- 9:45 (17) Thermal tolerance of mosquitofish (*Gambusia affinis*) from habitats differentially heat stressed. W. Holland, M. H. Smith and S. P. Gaffney, Jr. Vanderbilt University and Savannah River Ecology Laboratory

Chairman: J. Clugston  
University of Georgia

- 10:00 (18) Thermal tolerances of two *Notropis* Minnows. R. W. McFarlane, B. C. Moore and S. E. Williams. Savannah River Ecology Laboratory, Birmingham-Southern College and Savannah River Ecology Laboratory
- 10:15 (19) The effect of acclimation on the temperature tolerance of young muskellunge fry (*Esox masquinongy*). K. Hassan and J. R. Spotila. State University College at Buffalo
- 10:30 (20) Effects of acclimation temperature and water hardness on the thermal tolerance of bluegill. J. C. Murphy, C. T. Garten Jr. and M. H. Smith. Savannah River Ecology Laboratory
- 10:45 (21) Survival of juvenile fishes receiving thermal and mechanical stresses in a simulated power plant condenser. R. J. Kedl and C. C. Coutant. Oak Ridge National Laboratory
- 11:00 (22) Acute cold shock resistance of gizzard shad, *Dorosoma cepedianum*. D. K. Cox. Oak Ridge National Laboratory
- 11:15 (23) The effect of cold shock on the vulnerability of young bluegill (*Lepomis macrochirus*) to predation. W. R. Wolters and C. C. Coutant. Oak Ridge National Laboratory
- 11:30 (24) Further studies of cold shock effects on susceptibility of young channel catfish to predation. C. C. Coutant, D. K. Cox and K. W. Moored Jr. Oak Ridge National Laboratory

Thursday, April 3

## SESSION II-A THERMAL ECOLOGY - GENERAL

Grove 1

Chairman: R. V. Dimock, Jr.  
Wake Forest University

- 2:00 p.m. (25) Thermal ecology of phytoplankton in a desert reservoir. R. D. Olsen and M. R. Sommerfeld. Argonne National Laboratory and Arizona State University
- 2:15 (26) Studies of diurnal temperature and stratifications in lake systems in Chambal River. P. R. Kamath, R. P. Gurp, T. A. Sebastian and B. Dube, Bhabha Atomic Research Centre, India
- 2:30 (27) The effect of suspended sediment on the influx of radiant energy to reservoirs. F. R. Schiebe, J. C. Ritchie and J. R. McHenry. USDA Sedimentation Laboratory

- 2:45 (28) Effects of hypolimnetic withdrawal and Oconee Nuclear Station on water chemistry. G. Hoecker, R. Caccia and F. Abernathy, Duke Power Company
- 3:00 (29) Biological implications of chlorine in thermal discharges: A review. R. M. Block and G. R. Helz, University of Maryland
- 3:15 (30) Calculation of impact of entrainment by power plants. M. Lahlberg, NUS Corporation  
Chairman: M. J. Schneider  
Battelle Northwest Laboratory
- 3:30 (31) Variability of entrainment of organisms in an estuarine power plant. B. J. Copeland, R. Hodsden, W. S. Birkhead and J. M. Miller, North Carolina State University
- 3:45 (32) Entrainment of estuarine zooplankton into a mid-Atlantic power plant-delayed and sublethal effects. R. M. Davies, L. D. Jensen and C. H. Hanson, The Johns Hopkins University, Ecological Analysts, Inc., and The Johns Hopkins University
- 4:00 (33) Assessment of near field manifestations of power-plant entrainment damage on zooplankton. T. Polgar, L. Bongers and G. Krainak, Martin Marietta Corporation
- 4:15 (34) The effects of pumped-entrainment on estuarine macroplankton. J. A. Mihursky, J. E. Cooper, L. Lubbers and E. W. Gordan University of Maryland
- 4:30 (35) Abundance of insect drift in the Wabash River (Indiana) and power plant entrainment effects. J. R. King and E. R. Mancini, Sargent and Lundy Engineers and University of Louisville
- 4:45 (36) Preliminary assessment of larval fish entrainment, Colbert Steam Plant, Tennessee River. W. B. Wrenn, Tennessee Valley Authority
- 5:00 (37) Fish impingement at nine Duke Power Company steam generating facilities. T. J. Edwards, B. Hunt, L. E. Miller and J. J. Sevic, Duke Power Company

Thursday, April 3

# **SESSION II-B BEHAVIORAL ECOLOGY**

Grove II

Chairman: H. Eure  
Wake Forest University

- 2:00 p.m. (38) The fish communities in two reservoirs receiving heated effluents. J. P. Clugston, University of Georgia

- 2:15 (39) Largemouth bass under conditions of extreme thermal stress. J. R. Siler and J. P. Clugston, Duke Power Company and University of Georgia
- 2:30 (40) Food habits of bluegill and largemouth bass at two thermal regimes. S. R. Johnson, Duke Power Company
- 2:45 (41) The behavior of largemouth bass (*Micropterus salmoides*) in a reservoir receiving a heated effluent. S. P. Dupont, University of Georgia
- 3:00 (42) Thermal preferenda and behavioral thermoregulation in teleost fishes. W. W. Reynolds and M. E. Casterlin, The Pennsylvania State University.
- 3:15 (43) Behavioral thermal regulation of rainbow trout in a horizontal temperature gradient. R. W. McCauley, Wilfrid Laurier University, Canada  
Chairman: J. R. Spotila  
State University College at Buffalo
- 3:30 (44) Behavioral thermoregulation responses of bluegill (*Lepomis macrochirus*) exposed to various rates of temperature change. T. L. Beitinger, Argonne National Laboratory
- 3:45 (45) Daily ration, temperature selection and activity of bluegill, *Lepomis macrochirus*. W. E. Stuntz and J. J. Magnuson, University of Wisconsin Madison
- 4:00 (46) Temperature preference and movement of fish in relation to a long heated discharge channel. W. B. Wrenn, Tennessee Valley Authority
- 4:15 (47) The incidence and effect of condensor passage on Long Point Bay, Lake Erie larval fishes. G. Teleki, Ministry of Natural Resources, Canada
- 4:30 (48) Response of largemouth bass to thermal effluent from Oconee Nuclear Power Plant. J. A. Chappell, Clemson University

**Thursday, April 3**

**SPECIAL PROGRAM**

("Environmental programs of several power companies in the Southeastern United States")

Grove I  
7:30 - 10:00 p.m.

Chairman: Loren Jensen  
Ecological Analysts, Inc.

**Friday, April 4**

**SESSION III-A COMMUNITY ECOLOGY AND DIVERSITY**

Grove I

Chairman: R. R. Sharitz  
Savannah River Ecology Laboratory



- 8:30 a.m. (49) Distribution and abundance of submerged macrophytes, including *Myriophyllum spicatum* L. (Angiospermae), in a reactor cooling reservoir. J. B. Grace and L. J. Tilly, Clemson University and Savannah River Laboratory
- 8:45 (50) Thermal ecology of marine plants. R. F. van Tine, University of Florida
- 9:00 (51) The benthic community of a thermally altered area of Tampa Bay, Florida. N. J. Blake, L. J. Doyle and T. E. Pyle, University of South Florida
- 9:15 (52) Distribution and abundance of benthos in a lake receiving heated effluent. C. F. Sigmon, G. L. Ryals and B. R. Ingram, Clemson University
- 9:30 (53) Effect of thermal effluent upon the Rotifera of Lake Wabamun, Alberta. J. P. K. Horkan and D. N. Gallup, University of Alberta, Canada
- 9:45 (54) The effect of thermal effluents on stream periphyton. W. M. Rizze and R. R. Sharitz, Virginia Institute of Marine Science and Savannah River Ecology Laboratory
- Chairman: S. A. Spigarelli  
Argonne National Laboratory
- 10:00 (55) Effects of thermal constancy and seasonal temperature displacement on community structure of stream macro-invertebrates. J. V. Ward, Colorado State University
- 10:15 (56) Influence of thermal loading on the ecology of intertidal algae and on the growth and dynamics of *Ascophyllum nodosum*. R. L. Vadas, M. Keser and P. C. Rusanowski, University of Maine
- 10:30 (57) Diversity of aerial Trichoptera in thermally altered habitats on the Savannah River Plant. F. G. Howell and P. D. Hartfield, University of Southern Mississippi
- 10:45 (58) Macrobenthos diversity in confluent natural and thermal discharge streams. F. G. Howell and V. J. Fitzgerald, University of Southern Mississippi and The University of Chicago
- 11:00 (59) Relationships between genic and species diversity among fish communities. J. Greene, N. L. Cashion, D. W. Kaufman and M. H. Smith, Furman University of Arkansas and Savannah River Ecology Laboratory

- 11:15 (60) Fish diversity in adjacent natural, thermal and post-thermal streams. R. W. McBarlane, Savannah River Ecology Laboratory
- 11:30 (61) Some effects of a power plant discharge on estuarine epibenthic fauna. J. S. Young and A. B. Frame, National Marine Fisheries Service
- Friday, April 4** **SESSION III-B PHYSIOLOGICAL ECOLOGY**
- Grove II Chairman: D. Paine  
Savannah River Ecology Laboratory
- 8:30 a.m. (62) Growth of Salmonid fishes from heated and unheated areas of Lake Michigan as measured by RNA-DNA ratios. S. A. Spigarelli and D. W. Smith, Argonne National Laboratory
- 8:45 (63) Temperature as a growth accelerator in the spot, *Leiostomus xanthurus*. J. C. White, Jr., R. A. Hammond, N. H. Wooding, Jr. and M. L. Brehmer, Virginia Electric and Power Company
- 9:00 (64) Growth rates of subadult largemouth bass, 24-35.5°C. C. C. Coutant and D. K. Cox, Oak Ridge National Laboratory
- 9:15 (65) Growth-temperature response of striped bass, *Morone saxatilis*. D. K. Cox, Oak Ridge National Laboratory
- 9:30 (66) The effect of temperature and food availability on the growth of postlarval pinfish. D. S. Peters, M. T. Boyd and J. C. DeVane, Jr. Atlantic Estuarine Fisheries Center
- 9:45 (67) Effects of thermal effluents on growth and reproduction of the mosquitofish, *Gambusia affinis*. L. Goldner, University of Rhode Island
- 10:00 (68) Asymmetry in centrarchid fishes inhabiting thermally stressed environments. L. Ames, Savannah River Ecology Laboratory
- Chairman: J. B. Gentry  
Savannah River Ecology Laboratory
- 10:15 (69) Thermal effects of power plants on fish eggs and larvae. J. R. Schubel and T. S. Y. Koo, State University of New York and Chesapeake Biological Laboratory
- 10:30 (70) Thermal effluents, fish and gas bubble disease in southwestern Lake Michigan. R. G. Otto, Industrial Bio-Test Laboratories, Inc.

- 10:45 (71) Effects of a thermal effluent on the gonadal development and spawning of the American oyster. J. C. Tinsman, D. L. Maurer and S. G. Tinsman, University of Delaware
- 11:00 (72) Effect of temperature on growth, mortality and production of adult *Lymnaea ohrussa* Say (Mollusca:Gastropoda). J. S. Mattice, Oak Ridge National Laboratory
- 11:15 (73) Effects of constant and cycling temperature regimes on the pelagic larval development of a brachyuran crab. A. N. Sastry, University of Rhode Island
- 11:30 (74) The effects of exposure to time-temperature changes simulating power plant condenser entrainment on oxygen consumption patterns of *Gammarus* sp. and *Palaemonetes* spp. D. T. Burton, S. L. Margrey and L. B. Richardson, Academy of Natural Sciences of Philadelphia
- 11:45 (75) Temperature-time relationships for survival of embryos and larvae of the bay scallop *Argopecten irradians* (Mollusca:Bivalvia). W. H. Roosenburg, M. Castagna, W. P. Duggan, R. C. Carney and J. A. Mihursky University of Maryland, Virginia Institute of Marine Science and University of Maryland

#### Friday, April 4

#### SESSION IV-A POPULATIONS ECOLOGY AND MODELS

Grove 1

Chairman: F. G. Howell

University of Southern Mississippi

- 2:00 p.m. (76) Growth of aquatic herbs along a thermal gradient. E. J. Christy and R. R. Sharitz, University of Georgia and Savannah River Ecology Laboratory
- 2:15 (77) Responses of two species of cattail to thermal effluents. R. R. Sharitz, Savannah River Ecology Laboratory
- 2:30 (78) Effects of thermal loading on the growth and ecology of a northern population of *Spartina alterniflora*. R. L. Vadas, M. Keser, P. C. Rusanowski and B. R. Larson, University of Maine
- 2:45 (79) The rooting and early growth of mangrove seedling from thermally stressed trees. M. D. Banus, and S. E. Kolehmainen, Puerto Rico Nuclear Center
- 3:00 (80) Seasonal distributions and abundance of Ohio River fishes at the J. M. Stuart Electric Generating Station. C. O. Yoder and J. R. Gammon, DePauw University

- 3:15 (81) Distribution and abundance of fish populations in the middle Wabash River. T. C. Teppen and J. R. Gammon, DePauw University  
Chairman: J. E. Pinder, III  
Savannah River Ecology Laboratory
- 3:30 (82) The seasonal distribution of *Haplocytheridea setipunctata* in a thermally altered area of Tampa Bay, Florida. C. D. Stiles and N. J. Blake, University of South Florida
- 3:45 (83) Recolonization of the *Bithynia tentaculata* (Linnaeus) Gastropoda, in the affected zone by hot water overflows from Gentilly Nuclear Plant, G. Vaillancourt and R. Couture, University of Quebec, Canada
- 4:00 (84) The population biology of *Proteocephalus ambloplitis* (Cestoidea: Proteocephalidea) from largemouth bass living in a heated reservoir. H. Eure, Wake Forest University
- 4:15 (85) Effects of thermal loading on parasite populations in mosquitofish, *Gambusia affinis*. J. M. Aho, J. W. Gibbons and G. W. Esch, Wake Forest University and Savannah River Ecology Laboratory
- 4:30 (86) Measurements and models for a shallow estuarine ecosystem receiving thermal addition at Crystal River, Fla. W. H. B. Smith, University of Florida
- 4:45 (87) Potential influence of thermal effluents on the production and water use efficiency of mangrove species in south Florida. P. C. Miller, San Diego State University
- 5:00 (88) A simulation model of a thermal community. J. Sprague and R. Wiegert, University of Georgia

#### Friday, April 4

#### SESSION IV-B SYNERGISTIC EFFECTS AND PRODUCTIVITY

Grove II

Chairman: K. W. McLeod  
Savannah River Ecology Laboratory

- 2:00 p.m. (89) Some effects of temperature, chlorine and copper on the survival and growth of the coon stripe shrimp, *Pandalus danae*. C. I. Gibson, T. O. Thatcher and C. W. Apts, Battelle-Northwest Marine Research Laboratory
- 2:15 (90) Bioassays on the combined effects of chlorine, heavy metals and temperature on fish and fish food organisms. E. G. Wolf, M. J. Schneider and T. O. Thatcher, Battelle-Northwest Research Laboratory

- 2:30 (91) Interaction of temperature and salinity on crab metabolism. K. L. Hollifield and R. V. Dimock, Jr., Wake Forest University
- 2:45 (92) Synergistic effects of daylength and temperature on aquatic insects. B. R. Ingram, Clemson University
- 3:00 (93) Synergistic thermal and air pollution effects on a mangrove forest. M. I. Canoy, Puerto Rico Nuclear Center
- 3:15 (94) Oyster reefs at Crystal River, Florida and their adaptation to thermal plumes. M. E. Lehman, University of Florida
- Chairman: L. J. Tilly  
Savannah River Laboratory
- 3:30 (95) BOD experiments on the effect of thermal pollution in rivers and estuaries. H. Caspers, Institut für Hydrobiologie und Fischereiwissenschaft, West Germany
- 3:45 (96) Ecosystems developed in the intake and discharge canals of a coastal power plant at Crystal River, Florida. W. M. Kemp, University of Florida
- 4:00 (97) Effects of entrainment through Oconee Nuclear Station upon phytoplanktonic carbon-14 assimilation. T. V. Kreh, J. Derwort and M. Marcus, Duke Power Company.
- 4:15 (98) The effects of a channelized thermal effluent on entrained planktonic algae, bacteria and zooplankton in the Ohio River. M. C. Miller, G. R. Hater, J. P. Reed and T. Federle. University of Cincinnati
- 4:30 (99) Zooplankton motility and standing crops at a power plant utilizing an intake surface skimmer wall on Lake Keowee, South Carolina. J. S. Carter, W. Horton and G. Johnston, Duke Power Company

**Saturday, April 5**

**FIELD TRIP — Savannah River Plant**

9:00 a.m. - 4:30 p.m.

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