

Methodology for Assessing Population and
Ecosystem Level Effects Related to
Intake of Cooling Waters.
Volume 1: Handbook of Methods:
Population Level Techniques

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ABSTRACT

This section is a presentation of various techniques related to assessing power plant induced mortality at the population level. A broad range of techniques is presented through a review of past experience, examples using real or simulated data from power plant studies and new techniques developed for this methodology. This section provides the detailed information needed by an investigator to select those techniques most appropriate for the assessment problem at hand. Guidance is provided for the interpretation of the results of various analyses where appropriate. The major subjects reviewed in this section include techniques for estimating entrainment and impingement mortality, techniques for estimating population size, hydrologically based models for assessment, the use of length data for assessment, the Leslie Matrix Model, population response analysis, including a computer program for reproductive value, stock-recruitment relationships and life cycle models.

EPRI PERSPECTIVE

PROJECT DESCRIPTION

This final report, in the form of a handbook of methods, presents results of the second task (Task B) under Research Project (RP) 876. The task is essentially a presentation of available methods for assessing population and ecosystem effects of impingement upon and entrainment of major fish and invertebrate species. This volume contains detailed documentation of quantitative techniques for population level analyses. A companion report, Volume 2, similarly presents techniques for community level analyses. Another report (EA-1238) describes in more general terms the methods of analysis and provides guidance for interpretation of calculated or measured effects. Task C of the project will demonstrate application of selected aspects of the methodology. The results of Task A were published as EPRI Report EA-1038, Ecosystem Effects of Phytoplankton and Zooplankton Entrainment.

It is recognized that assessment methods are in a stage of rapid development, and that newer concepts, along with those described, will deserve consideration for particular applications. Examples of EPRI support to advance the state of the art include RP878, "Methodology for Assessing Integrated Effects of Multiple Power Stations on a Single Water Body" (EPRI Report EA-1111), and RP1111, "Assessment of Power Plant Induced Mortality on Fish Populations."

PROJECT OBJECTIVE

The goal of this phase of the project was to provide a reasonably complete documentation of methods that may be applied in assessing power plant intake effects on fish and invertebrate populations and associated communities. The reports are to serve primarily as a useful reference to guide an investigator in review of potentially applicable assessment techniques. Discussion would provide insight into the validity of various approaches, necessary assumptions, data requirements, and other important factors, including some rationale for judging the significance of effects.

PROJECT RESULTS

The documentation of methods provided in this task illustrates the great diversity of approaches an investigator may take in conducting an assessment. It is also demonstrated that whatever methods are selected, many assumptions will be required due to inherent limitations in the theoretical constructs and/or data deficiencies. Consequently, the role of the investigator in interpretation of calculated or measured effects is very important and is an integral part of the conceptual framework. A management approach for evaluating the significance of effects is presented in EPRI Report EA-1038, which recognizes the historical role of important species as aquatic resources and the natural resiliency of populations and ecosystems to natural and man-induced mortality.

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SUMMARY

Population Level Techniques includes a range of subjects related to assessing the effects of direct mortality on an individual population of fish or shellfish. Each chapter is a detailed treatment of a subject or techniques which provide basic data input to an assessment of population level effects. The investigator will use the companion "Methodology" to review the available approaches and select those techniques appropriate for the situation under investigation. Many of the discussions contain examples of how data are organized and analyzed. There is also guidance on how to interpret the results of analyses with emphasis on the assumptions and limitations of the techniques.

The estimation of entrainment and impingement mortality (Sections 1.2 and 1.3) is the basic quantification of plant effects which is essential to all assessments at the population level. While these data are basically a count of the number of organisms lost due to entrainment and impingement there are a variety of problems which can complicate and cause errors in the counts and the estimates. Entrainment and impingement mortality are treated separately because these processes involve different life stages and sampling problems. These sections are further divided into approaches for existing plants and proposed intakes, which will lack site-specific data.

The estimation of population size (Section 1.4) for selected species covers a broad range of approaches to providing a quantitative estimate of the source population which is experiencing power plant-induced mortality. The studies required to develop population estimates are site-specific and the techniques presented must be adapted to the special features of each situation. The presentation of various approaches emphasizes their limitations so that the most cost-effective technique can be selected. The bias stemming from unmet assumptions is generally a much greater error factor than the problems caused by low precision. The techniques presented for estimating population size include approaches which rely on different assumptions and, therefore, provide alternative methods for developing estimates which can be compared.

Physical models (Section 1.5) are useful techniques for estimating entrainment effects because the early life stages of fish and shellfish are often under a strong hydrodynamic influence. Where hydrodynamics is established as a dominating factor, a physical transport model can provide both an estimate of entrainment rate and a population cropping rate if data are available for the total range of the early life stages. Physical models are described for rivers, estuaries, and standing waters such as lakes, harbors, and bays.

An alternative approach to assessing entrainment effects is to use length-specific data (Section 1.6) for a population of successive early life stages in conjunction with an estimate of initial abundance. Intensive sampling over the period of spawning and early growth can provide the data for this approach. While costly in terms of the amount of field sampling required, this approach has the advantage of incorporating the population dynamics of entrainable stages rather than relying upon the physical factors in transport models. As with the alternative approaches to estimating adult fish abundance, a comparison of quantitative estimates based on different assumptions can be achieved for entrainable stages.

The Leslie matrix model (Section 1.7) is a method for simulating discrete time changes in abundance and age structure in a population. The model assumes constant rates of age-specific fecundity and survival, which limits most applications to relatively short-term estimates of population level effects. This presentation provides a description of the basic Leslie matrix approach, examples of applications, and possible modifications to extend the time frame for reliable impact estimates.

Population response analysis (Section 1.8) includes techniques for analyzing long-term changes in biological characteristics of fish populations. Biological characteristics such as growth, mortality, abundance trends, and reproduction have been used as indicators of a response by a population to increased mortality. An estimate of the percentage removal of individuals from a population due to plant cropping provides a limited assessment in that different populations may respond quite differently to the same level of removals. The response of the population to removals provides a measure of the ability of the population to tolerate the mortality imposed by entrainment and impingement. In addition, a population response analysis can be a useful approach in the verification of models used to predict impact.

Stock-recruitment relationships (Section 1.9) are a class of population models which represent the number of progeny that attain spawning age as a function of the size of the spawning stock which produced them. These models have been used in fishery management for many years and can be applied to power plant assessment if the long time series of data needed for model development are available. Stock-recruitment relationships are limited in that the environmental and biological factors which play a role in the observed relationship are unknown. The concept of stock-recruitment relationships is presented with guidance for the selection of an appropriate model and a discussion of the limitations of this approach.

Life cycle models (Section 1.10) develop a quantitative description of the life history of a species which is used to produce successive generations and, thereby, carry the population into the future. Power plant mortality is then imposed on the projected population to estimate long-term impact. Life cycle models are species-specific and only the general approach with examples and their limitations are presented. Life cycle models are a recent approach to population level assessment and represent a very complex and costly approach.

Section 1

POPULATION LEVEL TECHNIQUES

1.1 INTRODUCTION

This handbook contains an expansion on various techniques discussed in the methodology (EA1238) which could not be treated in detail because of the lengthy presentation needed. Throughout Section 5.0 of the methodology subsection headings are followed by a number in parentheses which refers to a section of this handbook when a detailed treatment of a subject is available. Subsections of this handbook will stand alone as detailed summaries of particular topics; but for an actual impact assessment problem, it must be used in conjunction with the methodology.

1.2 ESTIMATING ENTRAINMENT CROPPING

An estimate of the total number of individuals killed through entrainment is the quantitative measure of the effects of entrainment that is basic input to impact assessment. Viability studies are an integral part of estimating entrainment cropping because substantial cross-plant survival has been documented. The steps in the design of an entrainment sampling program will involve the selection of appropriate sampling gear, delineation of the sampling season and selection of the sampling design. This section also includes guidance on estimation of entrainment-induced mortality.

1.2.1 Estimation of Entrainment Abundance

1.2.1.1 Delineation of Sampling Season. For some organisms, the sampling season may consist of the entire year; however, many organisms, particularly ichthyoplankters, will appear in the source water body during only some portion of the year. Although it would be economically desirable to limit the sampling season to that period, this is generally not possible. The time of first appearance of the organisms will vary from year-to-year, sometimes by a matter of weeks, as will the length of time they remain in the source water body. This variability results in either a sampling season including stretches of time during which the organisms of interest are not present or a sampling season which fails to include the entire time of presence.

Historical data will be useful in determining an appropriate sampling season. The time of first appearance of the organisms and the length of time that they will be present may sometimes be predicted on the basis of environmental factors such as water temperature. Unfortunately, few sites will have adequate data for this purpose.

In the absence of a reliable predictor of first occurrence, the start of the sampling season may be determined by means of a low-level monitoring program. This program should begin several weeks in advance of the earliest expected date of first appearance and should involve sampling on a regular basis, perhaps every three or four days, at a location and time of day for which relatively high abundances are expected. The full sampling program would be initiated at the first appearance of

the organisms of interest. Of course, this procedure will cause a slight truncation of the sampling season at its onset, since the main sampling program cannot begin until after the first occurrence of organisms. However, the exclusion of a short period of low abundance from the sampling season will have negligible effect on its results.

The length of the sampling season, which may not be predictable, will also vary somewhat from year to year. A sampling design involving the stratification of dates (see below) will provide sufficient flexibility to ensure that the entire period during which the organisms are present is included while minimizing the temporal extent of the program.

1.2.1.2 Selection of Sampling Dates. Existing sampling programs generally follow a two-stage design procedure wherein dates are selected for sampling and samples are subsequently allocated for each selected date. Two-stage procedures are logistically desirable and can also be statistically valid if proper attention is paid to statistical sampling theory when allocating dates and samples within dates. Designs of this sort are alternately referred to as "two-stage" and "subsampling."

The first stage in implementing a subsampling design will be to consider the set of all dates in the sampling season as a statistical population and to utilize an appropriate randomized design in selecting the dates that will actually be sampled. Each selected date is then subsampled; i.e., a randomized design is used to allocate a number of actual water samples to be collected and analyzed on each of the sampling dates. In statistical parlance, the dates are referred to as Primary Sampling Units (PSU) and the actual samples (net or pump samples) are called Secondary Sampling Units (SSU).

The simplest type of subsampling design is obtained when both the PSU's (dates) and SSU's (samples within dates) are selected according to Simple Random Sampling (SRS) designs. An SRS must be used independently in selecting the SSU's for each sampling date. Selection of sampling dates is accomplished by consecutively numbering the dates in the sampling season and using a random number table. For each selected date, the SSU's are numbered consecutively and a random number table used to select several for sampling.

Estimates derived from subsampling designs are subject to two sources of sampling error: variation among dates; and variation among samples within a date. Variability within a date is both spatial and temporal. Variability among dates for an entire sampling season can be quite appreciable, especially for ichthyoplanktonic life-stages. However, existing information about the biological populations being studied can often be used to obtain a modification of the subsampling design to substantially reduce these sources of variability in the estimation of abundance.

Stratification is a device often used for incorporating existing information about a system into a sampling design to improve the precision of estimates without increasing sample size. This process involves the partitioning of the population into disjointed sets of sampling units, called strata. Each sampling unit in the population must be allocated to one, and only one, stratum. Stratification may be applied to either or both of the PSU's (dates) and SSU's (samples). The basic premise in stratification is that existing knowledge about the system can be used to define strata within which the variability among sampling units is of smaller magnitude than the variability among sampling units in the population as a whole.

The stratification of dates will almost always be of benefit in reducing variability, especially when ichthyoplankton are being sampled. A smoothed graph of spot (Leiostomus xanthurus) larval abundances in the Cape Fear Estuary (North Carolina) plotted against time (Figure 1.2-1) illustrates a typical situation. Initially at low levels, abundance gradually peaks, then gradually declines to zero. This graph clearly demonstrates that, within a shorter period of time, perhaps two weeks, the variation among daily mean abundances will be small relative to the variation among daily means for the sampling season as a whole. Hence, all other factors being the same, a design including such a stratification will result in more precise estimates than a design excluding this feature.

The stratification of dates also provides a mechanism for reducing the length of the sampling season. As discussed above, in the case of seasonal organisms, such as ichthyoplankton, the pre-defined sampling season will generally extend beyond that point in time when the organisms last appear in the source water body. In these cases, the disappearance of the organisms from the water body during any date stratum would generally justify the elimination of sampling in subsequent date strata. This can be done without affecting the statistical validity of abundance estimates obtained in the other date strata.

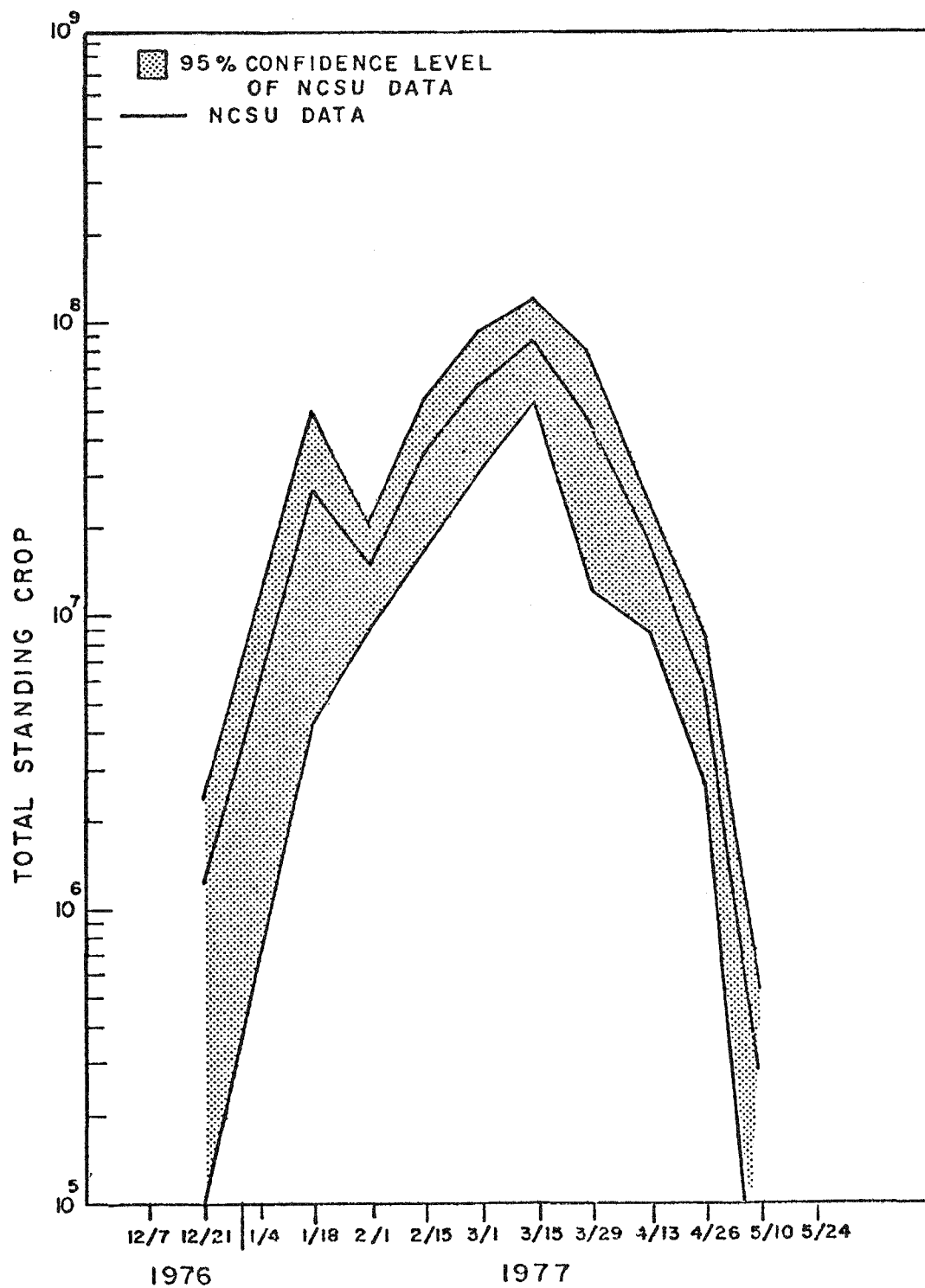


Figure 1.2-1. Total standing crop of spot larvae in the main channel in the Cape Fear Estuary, 1976-1977.

The principal use of entrainment abundance estimates is for comparison to source water body abundance estimates. The precision of such comparisons will be greatly increased if source water body sampling is carried out on the same dates as entrainment sampling. However, valid comparisons may still be made when the two surveys have independently allocated dates according to Stratified Random Sampling designs.

One commonly used alternative to Stratified Random Sampling designs is Systematic Sampling (SYS). Such a design involves sampling once every k days during the sampling season; i.e., every Wednesday ($k=7$). The first sampling date is selected at random from the first k dates of the sampling season. If the first sampling date is randomly selected, no design-related bias will be introduced into estimators of entrainment abundance.

The major shortcoming of Systematic Sampling designs in this setting is that a statistically unbiased estimate of day-to-day variability cannot be obtained. Consequently, it will not be possible to properly quantify the precision of the resulting estimate of total entrainment for the sampling season. However, if the source water body survey is conducted on the same dates as the entrainment study, this shortcoming will not invalidate comparisons between the two.

The use of Stratified Random Sampling designs for the selection of sampling dates is strongly recommended primarily because they provide valid quantification of the standard error of estimates of numbers of organisms entrained during any stratum and during the entire sampling season. The source water body sampling program should also be conducted on the same dates as the entrainment program in order to provide the greatest possible comparative resolution.

If, for whatever reason, a Systematic Sampling plan is used for the selection of dates, it is imperative that the source water body sampling program be conducted on the same dates in order to avoid biased comparisons. However, any generalization of the results of an entrainment/source water body comparison based on systematic sampling will not be fully backed by statistical theory.

1.2.1.3 Within-Date Sampling Design. The selection of a sampling design for each selected date is a less than straightforward matter. Ideally, the entire

volume of water passing through a plant in a 24-hour period should be partitioned into a number of non-overlapping sampling units. An appropriate sampling design would then be used to select those units which would be collected and analyzed. Such a partitioning was described by Weiss (1.2-1); however, sampling designs based on this will present substantial logistical difficulties and be extremely costly. Proper statistical survey design principles should be followed insofar as is practical, although some relaxation of formal statistical requirements is necessary to make entrainment studies feasible.

A stratification of sampling units within each date may be of considerable benefit in improving the precision of abundance estimates. Since sampling units within each date are distributed in both time and space, factors associated with these dimensions should be examined for stratification possibilities. For instance, a stratification could be based on knowledge of the hydrodynamic properties of the intake area, diel behavioral characteristics of the organisms, and known spatial distribution tendencies. Some illustrations of the effective use of prior knowledge in stratification may be of value.

- Stratification in both time and space can be of value in the sampling of organisms which display phototaxic behavior. Such organisms tend to concentrate in the bottom of the water column during the day, and migrate upwards at night. In this case, stratification by photoperiod and depth is clearly appropriate.
- In a marine or estuarine environment, entrainment abundance may vary by tidal phase. Therefore, stratification by tidal phase may be beneficial in such cases.
- For plants having several physically distinct intake areas, each area might constitute a different stratum.

Stratification can serve purposes other than improving the precision of estimates. The strata themselves may be of interest in impact assessment, particularly when mitigating measures are sought. Comparisons between photoperiods, depths, tidal phases, intake units, etc., are frequently desired. Statistically valid comparisons can be made only between strata and generally cannot be made within a stratum. Therefore, it is essential that all comparisons of potential interest be determined in advance so that an appropriate stratification can be specified.

1.2.1.4 Randomization Within Strata. Statistical sampling theory requires that the entire volume of water entering the power plant on a sampling date be divided into non-overlapping sub-volumes called sampling units and that an appropriate randomization scheme be used to determine which of these shall be collected and analyzed. If the date has been stratified, then each sampling unit must be in one, and only one, stratum. Conceptually, this may be achieved by dividing an intake face into squares and dividing the 24-hour period into shorter periods. The sampling units will be the volumes passing through each square during each period of time. A graphic representation of this is given in Figure 1.2-2. Several "boxes" in each stratum are then randomly selected for collection and analysis.

In practice, one would select "box" dimensions compatible with the sampling gear used. For instance, if 1.0 m diameter nets were to be deployed for 15 minutes, the "box" dimensions would be 1.0 m x 1.0 m x 15 minutes. When circular nets are used, although the entire volume in a "box" will not be strained, about 79% of the volume will be strained, resulting in a relatively small measurement error.

At some plants, there may be technical considerations making it difficult, or even impossible, to put the above concept into practice. It will then be necessary to develop a somewhat less formal means of allocating samples within strata. Such procedures will be situation dependent; hence, details shall not be discussed. However, in the selection of a sample allocation scheme, it should be remembered that each stratum is three-dimensional and that the validity of concentration and standard error estimates will depend upon a random scatter of samples throughout the three dimensions. Sampling at a single fixed location in each stratum is not acceptable as no measure of within-stratum spatial variability can be obtained. Similarly, sampling at only one time within a stratum is unacceptable.

1.2.1.5 Computation of Entrainment Abundance. To obtain an unbiased estimate of total entrainment for the sampling season, detailed and complete plant flow data will be necessary, as, without such data, it will not be possible to properly weight the various strata.

At an absolute minimum, the total cooling water flow through the plant during each date stratum must be known. If possible, total flow for each within-date stratum during each date stratum should be known. If this information is not completely available, it may be possible to determine these by making a simple

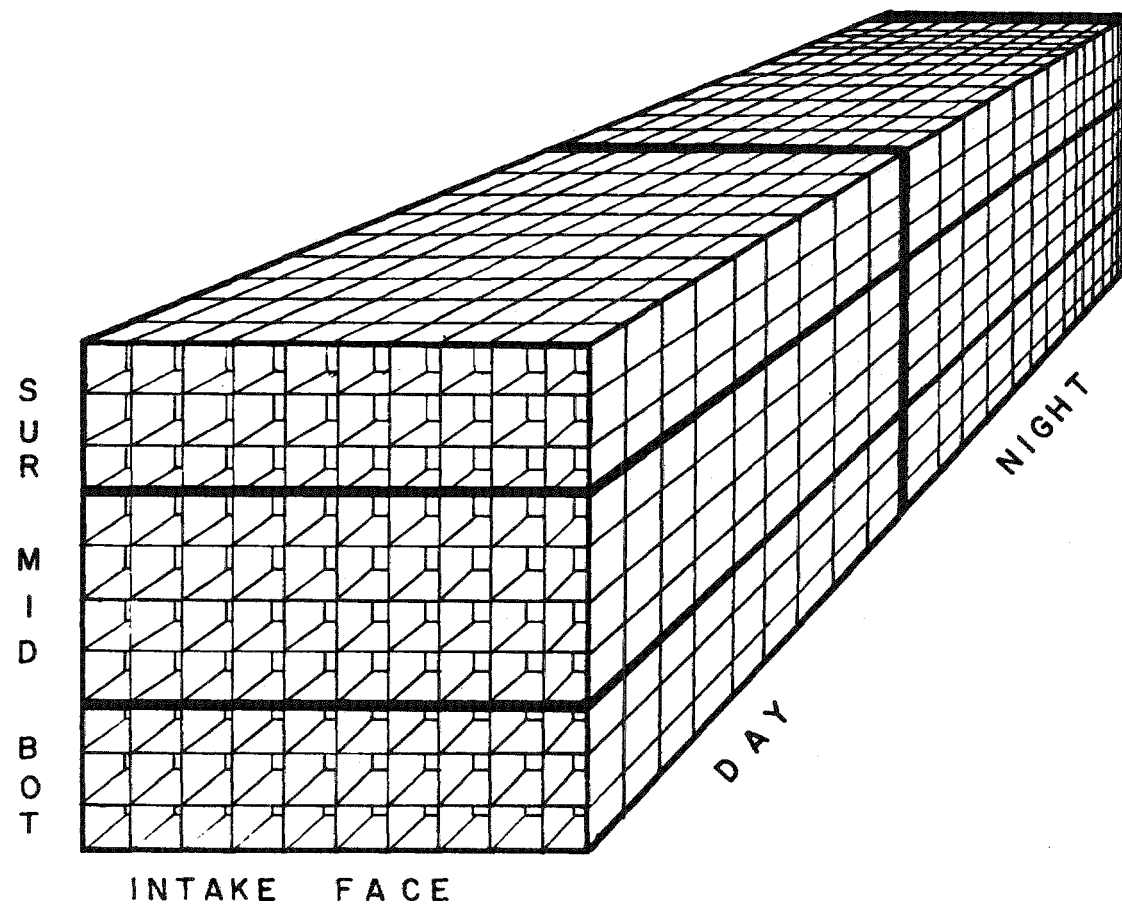


Figure 1.2-2. Graphic Representation of Randomization Within Strata.

assumption. As an example, assume that within-date stratification is by photoperiod (day/night) and depth (surface, mid, bottom) and that plant flow data are available by photoperiod, but not by depth. If it is assumed that, averaged over a date stratum, flow is uniform across the intake face within each photoperiod, then the proportion of the area of the intake face occupied by each depth strata times the total volume for that photoperiod will provide the volume of that stratum.

The following notation will be used:

X_{ijkl} The number of organisms collected in the l th sample from the k th stratum on the j th sampling date of the i th date stratum.

Y_{ijkl} The volume of the sample from which X_{ijkl} was obtained.

Z_{ijkl} X_{ijkl}/Y_{ijkl} (sample concentration).

M_i Number of dates in the i th date stratum.

m_i Number of dates sampled in the i th date stratum.

N_{ijk} Number of sampling units in the k th stratum on the j th sampling date of the i th date stratum.

n_{ijk} Number of sampling units collected from the k th stratum on the j th sampling date of the i th date stratum.

V_{ik} The total volume of water in the k th stratum of the i th date stratum.

I Number of date strata.

K Number of within-date strata.

Sums over an index are denoted by replacing the index by a dot ".". An average over an index will be identified when a bar is placed over the quantity; i.e.,

$$\bar{X}_{ijk.} = \sum_{l=1}^{n_{ijk}} X_{ijkl} \quad (1.2-1)$$

$$\bar{z}_{ijk.} = \left(\sum_{l=1}^{n_{ijk}} z_{ijkl} \right) / n_{ijk} \quad (1.2-2)$$

An estimator of the number of organisms entrained in the kth stratum of the jth sampling date of the ith date stratum is:

$$\hat{T}_{ijk} = \bar{z}_{ijk.} \times V_{ik} / M_i \quad (1.2-3)$$

with variance:

$$\text{Var} (\hat{T}_{ijk.}) = \left[\left(\frac{1}{n_{ijk}} - \frac{1}{N_{ijk}} \right) \sum_{l=1}^{n_{ijk}} (z_{ijkl} - \bar{z}_{ijk.})^2 \right] \times (V_{i.k} / M_i)^2 \quad (1.2-4)$$

The total number of organisms entrained on the jth sampling date of the ith date stratum is estimated by:

$$\hat{T}_{ij} = \sum_{k=1}^K \hat{T}_{ijk} \quad (1.2-5)$$

with variance:

$$\text{Var} (\hat{T}_{ij}) = \sum_{k=1}^K \text{Var} (\hat{T}_{wk}) \quad (1.2-6)$$

An estimate of the total number of organisms entrained in the ith date stratum is:

$$\hat{T}_i = M_i \left(\sum_{j=1}^{m_i} \hat{T}_{ij} \right) / m_i \quad (1.2-7)$$

with approximate variance:

$$\text{Var } (\hat{T}_i) = \left(\frac{1}{m_i} - \frac{1}{M_i} \right) S_A^2 + S_B^2 \quad (1.2-8)$$

where:

$$S_A^2 = \sum_{j=1}^{m_i} (\hat{T}_{ij} - \hat{T}_i)^2 / (m_i - 1) \quad (1.2-9)$$

and

$$S_B^2 = \sum_{j=1}^{m_i} \text{Var } (\hat{T}_{ij}) / m_i \quad (1.2-10)$$

The total number of organisms entrained during the sampling season is estimated by:

$$\hat{T} = \sum_{i=1}^I \hat{T}_i \quad (1.2-11)$$

with approximate variance:

$$\text{Var } (\hat{T}) = \sum_{i=1}^I \text{Var } (\hat{T}_i) \quad (1.2-12)$$

Conversion of the above to concentrations is straightforward. These are obtained by dividing the total by the appropriate volume. The variance of a concentration is obtained by dividing the variance of the corresponding total by the square of the appropriate volume. For example, the concentration of organisms for the i th date stratum is estimated by:

$$\hat{C}_i = \hat{T}_i / V_i. \quad (1.2-13)$$

with variance

$$\text{Var } (\hat{C}_i) = \text{Var } (\hat{T}_i) / V_i^2 \quad (1.2-14)$$

1.2.2 Estimation of Entrainment-Induced Mortality

Many impact studies conservatively assume 100% mortality of entrained organisms without experimental verification. However, entrainment viability studies have demonstrated that organisms survive cross-plant passage, suggesting that an entrainment mortality rate of less than 100% is appropriate. Unfortunately, although it is possible to prove that entrainment mortality is less than 100%, it is not possible, using current sampling methodology, to obtain a consistent estimate of that mortality rate unless a number of presently unverified assumptions are made.

It would at first seem plausible that an observational study, in which the proportion of viable organisms collected in a plant discharge pipe or canal is examined, will provide a means of estimating the entrainment-induced mortality rate. Unfortunately, this approach is not valid for at least two reasons:

- Currently available sampling gears apply mortality-inducing stresses in removing organisms from the water body.
- Sampling methods may not be equally efficient in collecting live and dead organisms. An extreme case will occur when entrainment causes the mechanical disintegration of some organisms to the point that they can no longer be identified.

Several workers have attempted to avoid these difficulties by utilizing a pseudo-experimental procedure in which "control" samples are collected from the intake area and "experimental" samples from the discharge area. The intake samples are presumed to reflect gear-induced mortality and any mortality which may occur prior to entrainment. The most widely used method for the adjustment of discharge mortality estimates for gear-induced mortality is that of Abbott (1.2-2). It is derived as follows:

Define the following symbols:

M_E = entrainment mortality rate

s_I = intake survival rate

s_D = discharge survival rate

Since it is assumed that gear-induced mortality operates independently of entrainment-induced mortality, the rates are multiplicative:

$$s_D = s_I (1 - M_E) \quad (1.2-15)$$

which leads to the formula:

$$M_E = 1 - \frac{s_D}{s_I} \quad (1.2-16)$$

If it is assumed that both intake and discharge samples used in estimating the mortality rates are random in that each entrained, or about to be entrained organism has an equal chance of being collected, then the binomial probability law applies. It can then be shown that the above formula represents the maximum-likelihood estimator of the entrainment-induced mortality rate, with asymptotic variance:

$$\text{Var } (M_E) = \frac{1 - M_E}{N_D} \left(\frac{1}{s_I} + M_E - 1 \right) \quad (1.2-17)$$

where N_D is the number of organisms collected in the discharge sample.

The result stated above assumes that live and dead organisms are collected with equal efficiency. The formula would not be correct if, for instance, the entrainment process resulted in the mechanical destruction of some organisms. This problem was addressed by Marcy (1.2-3) who presents a modification of Eq. 1.2-16:

$$M_E = 1 - \frac{C_D}{C_I} \quad (1.2-18)$$

where C_D and C_I are estimates of the concentrations of live organisms in the intake and discharge, respectively. A formal derivation of this result, along with its sampling error, can be made by assuming that the abundance of live organisms in samples follows a volume-dependent Poisson distribution and that an analogue of condition Eq. 1.2-15 exists.

Let: X_i = number of live organisms in i th intake sample

Y_i = number of live organisms in the i th discharge sample

V_i = volume of the i th intake sample

W_i = volume of the i th discharge sample

N_I = number of intake samples collected

N_D = number of discharge samples collected

The mean concentration of live organisms in the intake has, as its maximum-likelihood estimator:

$$\hat{C}_I = \frac{\sum_{i=1}^{N_I} X_i}{\sum_{i=1}^{N_I} V_i} \quad (1.2-19)$$

The maximum-likelihood estimator of discharge concentration of live organisms is:

$$\hat{C}_D = \frac{\sum_{i=1}^{N_D} Y_i}{\sum_{i=1}^{N_D} W_i} \quad (1.2-20)$$

Marcy assumes that, analogous to Eq. 1.2-15:

$$C_D = (1 - M_E) C_I \quad (1.2-21)$$

Hence, the maximum-likelihood estimator of entrainment mortality is:

$$M_E = 1 - \frac{C_D}{C_I} \quad (1.2-22)$$

with asymptotic variance:

$$\text{Var } (M_E) = \frac{1 - M_E}{n_D \sum_{i=1}^{C_I} W_i} \quad (1.2-23)$$

The key assumption involved in both these methods is that sampling-induced mortality is independent of mortality induced by entrainment related stresses. This assumption is given by Eq. 1.2-15 and 1.2-21 for the Abbott and Marcy estimators, respectively. The independence assumption cannot be examined in situ nor is it believed to have been examined by laboratory experimentation to date.

Conceptual means by which the independence assumption can be violated, and the consequences of such violations were discussed by Weiss and Logan (1.2-4). Possible violations result from two postulates which are intuitively reasonable from a biological viewpoint:

- Mortality induced in an organism by stresses associated with entrainment and sampling is at least partially related to the fitness of that organism; it is not a completely random process in which each organism has an equal chance of survival.
- Organisms are more susceptible to mortality induced by sampling-related stresses if they are first subjected to entrainment.

The first postulate suggests that there is a positive association between the ability of an organism to survive entrainment stresses and its ability to survive collection and handling stresses. The use of the Abbott and Marcy formulas would, if this were the case, result in an underestimate of the entrainment mortality rate. The extreme case would occur if the association were complete; i.e., all organisms which can survive entrainment can also survive collection and handling, and vice versa. In this case, intake and discharge mortality rates would, on the average, be the same, resulting in an estimated entrainment mortality rate of zero.

Some indication of an association between entrainment and collection survival ability is found in a recent entrainment mortality study for larval striped bass (Morone saxatilis) at a Hudson River power plant (1.2-5). In this study, a trash

pump/larval table methodology was used to collect larvae in both the intake and the discharge. This methodology is considered to be state-of-the-art with regard to minimizing collection mortality (1.2-6). Nevertheless, entrainment mortality, as calculated from the Abbott formula, was found to be 8.7%, as opposed to an intake mortality rate (representing collection mortality) of 19.1%. It could be argued that a low stress collection device would not induce mortality in larvae at a rate more than twice that induced by entrainment. This result might be better explained by the postulate of association which suggests that larvae surviving entrainment are better able to survive collection than the population as a whole. Thus, an underestimate of entrainment mortality is obtained.

The second postulate suggests that an organism subjected to entrainment is less likely to survive subsequent collection and handling. Consequently, the use of either the Abbott or Marcy estimator will result in an overestimate of the entrainment mortality rate since the discharge mortality rate will be greater than would be otherwise expected.

The potential for biased estimation of entrainment mortality because of the above postulates is related to the magnitude of the gear-induced mortality. The smaller the gear-induced mortality, the smaller the potential bias of either the Abbott or Marcy estimator.

Another consideration to be addressed in entrainment viability studies is that of latent mortality which is generally evaluated by holding those organisms surviving entrainment for some period of time. Holding should be under ambient conditions, and organisms collected from the intake, which serve as a control are handled in an identical manner. In line with common practice for toxicity assays, a 96-hour holding period seems appropriate for this purpose. Comparisons of the 96-hour survival of organisms that initially survive intake and discharge collection should be made. If a difference is found 96-hour survival rates should be used.

Because of the assumptions of the Abbott and Marcy estimators of entrainment mortality, additional research and development are needed in this area. The development of a no, or very low, mortality collection device would eliminate most problems. An alternative line of potentially useful research would involve an examination of the association between entrainment and gear-induced mortality.

1.3 Estimating Impingement Cropping

An estimate of the total number of individuals killed through contact with the intake screens is the quantitative measure of the effects of impingement. To assess impact it is necessary to estimate total impingement losses by species and in some cases, by life stage, age and size. Impingement surveys have been conducted at many power plants that provide perspective on the sampling problems and indicate what factors may be important in the impingement process. A review of impingement studies by Argonne National Laboratory (1.3-1, 1.3-2, 1.3-3) provides information on the species composition and general magnitude of impingement at a large number of plants in a variety of aquatic environments.

Impingement on intake screens does not invariably result in death for fish and invertebrates, and, in fact, substantial survival rates for some species have been documented. The estimate of impingement effects may, therefore, involve both an estimate of total number collected on the screens and an estimate of mortality if substantial survival is expected. A variety of factors, which must be evaluated at each site, will influence the potential for survival. Important factors known to influence survival include screen operating procedure, screen spray wash pressure, the efficiency of any fish return or salvage systems, and the resistance to mechanical stress of each species.

The presence or absence of impingement sampling data will require different approaches to estimating total plant mortality. Where data are available for the plant under consideration, an estimation procedure will generally be required to determine the total number impinged. At most plants where data presently exist, a subsampling procedure is used to characterize impingement, and the estimation procedure will entail an expansion of existing data to obtain the total impinged.

In those cases where data are lacking, two approaches are available to obtain impingement estimates. The Argonne National Laboratory review indicates that impingement sampling data are available for most major water bodies with power plants. An estimation procedure could be applied to the available data and then extrapolated to the plant or site of interest. The validity of transferring data from one plant to another will depend on the similarity of the sites and the design and operating characteristics of the plants. Experience with existing plants close to each other on the same water body suggests that substantial differences

in the impingement rate may exist although conditions appear to be similar. A careful judgment will be needed in each case where this approach is contemplated.

The second approach in cases where impingement data are lacking for a specific site is to apply the best judgment to those data available on the distribution and abundance of impingeable organisms and the operating characteristics of the intake system to predict total impingement. This approach is primarily applicable to proposed sites, because existing plants can be surveyed to provide a data base. In some cases this approach may be little more than guesswork; however, the experience gained at many operating plants will provide guidance on the likely species composition, seasonality, and size distribution of impinged fish. Unfortunately, the magnitude of impingement, the most important characteristic required for impact assessment, is the most difficult to estimate.

This section will discuss methods for estimating the number of fish impinged (abundance) and post-impingement viability. Estimating impingement from existing data is treated first because data already exist for many plants. There is also a section on the design of impingement surveys for guidance in developing data-gathering programs.

1.3.1 Estimating Abundance When Data Are Available

The objective of all impingement surveys has been to document the species composition, seasonality, size distribution and abundance, as well as other characteristics of the population of fishes and invertebrates collected from intake screens. These surveys generally considered a day (24-hour period) as a sampling unit and employed various sampling schemes to obtain data. In a few cases a complete census was used. Most surveys covered at least one full year because of the potential seasonality of impingement.

The frequency of sampling varied considerably and in some cases was flexible, with increased sampling effort at times of high impingement rate. The 24-hour sampling unit was often stratified into smaller units of time to examine the diel pattern of impingement, and then composited for a unit total. As far as could be determined, most impingement sampling programs were not based on a statistical design nor on information obtained from an impingement pilot study.

The processing of impingement samples has been highly variable, but in most cases total counts by species were made when the number collected was small to moderately large. When samples were large a subsampling scheme was used to estimate the total number by species. Some surveys grouped species into categories such as "minnows," "catfish," and "clupeids." Such groupings will severely limit impact assessment for individual species included, if species data cannot be extracted. Data on the size distribution of impinged fish were collected at many plants.

The existing impingement data have been used at many sites to estimate total impingement. A variety of estimation procedures have been used where a complete census was not available. Because most surveys were not based on a statistical design, ad hoc estimation procedures were used. The following recommendations for estimating impingement apply to both existing data bases and future surveys.

The rate of fish impingement is commonly quantified as the number of fish collected per day or per unit volume of water pumped by the plant. It is recommended that impingement rate be quantified as fish per unit flow because fish per day rates do not reflect changes in the plant flow which could materially affect the number of fish impinged on the intake screens. One hundred fish impinged on a day when only one circulating water pump is operating should not carry the same weight as one hundred fish impinged on a day with all circulating pumps in operation.

Estimates of the total number impinged should be made by species. The total number or weight of all species combined is often presented in reports of monitoring programs, but such totals are of little use in impact assessment. Estimates by age, life stage or size may be needed for assessments. If the need for this information is known in advance of impingement monitoring, analyses of samples can provide the data; however, where such data are lacking, reanalysis of available preserved samples will be the only source of data.

High seasonal variability is a common characteristic of impingement. In those studies based on a statistical design which stratified the year, the estimation procedure should, of course, utilize the preselected strata. In situations where data are available from sampling programs which did not stratify the year, post-sampling stratification should be used whenever possible to improve the accuracy of the estimates.

At plants where impingement monitoring has extended over a series of years, there have been large differences in the total number collected annually by species. A single year of monitoring will be sufficient for assessments where a detailed population dynamics study is not required. However, where comprehensive studies are conducted, impingement sampling must be carried out for the duration of the study. In the case of comprehensive studies, existing data or the first year of impingement monitoring can serve as a pilot study to optimize precision and cost in the monitoring program.

1.3.2 Estimating Post-Impingement Survival

Post-impingement survival has been observed for many species at plants with a variety of intake configurations. While most observations were indicative of survival immediately after removal from intake screens, holding experiments have also established that long-term survival may be high and should therefore be accounted for in impact assessment.

Studies by King et al. (1.3-4) indicated that substantial long-term survival (generally 48-hour holding) of young-of-the-year striped bass, white perch and Atlantic tomcod could be expected at power plants on the Hudson River. High survival of striped bass and white perch was dependent on the mode of screen operation and screen wash water pressure. Tatham et al. (1.3-5) in studies at the Oyster Creek Generating Station found that survival after 48 hours of holding varied widely among species seasonally and for some species may have been dependent on the mode of screen operation. White and Brehmer (1.3-6) reported on the survival of fishes collected on modified traveling screens at the Surry Power Station on the estuarine portions of the James River. High survival was reported for many species on these screens which were designed to remove fish from the screen surface and return them to the water source with a minimum of stress. These studies only investigated short-term survival.

The measurement of post-impingement survival generally involves a series of tests in which fish removed from the intake screens are held in tanks for a preset time interval to observe the rate of die-off and total mortality. Survival tests are not complicated; however, problems have been encountered in such tests and the following recommendations may help investigators avoid problems in the future.

- 1) As for other aspects of population studies, survival studies must be done for individual species.
- 2) Long-term survival is the only measure of survival meaningful for impact assessment. Holding test specimens for 48 to 96 hours is a generally accepted time period for demonstrating long-term survival.
- 3) All tests must include controls to establish the adequacy of test and holding conditions.
- 4) The test procedure should minimize, to the extent possible, stress due to the test and holding procedures. Although controls will indicate problems with test and holding procedures, the potential for clear results will be enhanced if low stress handling and holding are employed.
- 5) Test and control specimens should be held in water from the source where they were obtained.
- 6) Survival tests should be conducted at least seasonally (if the species of interest is present at different seasons) to account for the annual variation in the condition of fish and hence their potential for survival.
- 7) If the screen operating procedure or wash water pressure is changed during an annual cycle, for example, continuous screen operation to remove heavy debris or to prevent freezing, survival tests are needed if the species of interest is present in substantial numbers when conditions change.
- 8) Computation of long-term survival should be based on a total sample from the intake screens which includes both live and dead fish. Live and dead fish are commonly separated during collection and only the live fish held for survival tests. Total mortality, the number of interest in impact assessment, includes both the immediate mortality and the latent mortality which occurs during holding.

The interpretation of survival studies must acknowledge the influence of the protected holding which removes the impingement-stressed fish from exposure to predators and other factors they would experience upon immediate return to the water source. In one study reviewed, the protection was carried a step further with the application of chemicals to test and control fish to prevent disease in the holding containers. The influence of holding cannot be entirely eliminated from the estimate of survival; however, one approach requiring judgment by the investigator is suggested for refining the estimate.

As fish are collected from the intake screens they can be sorted into unstressed, stressed and dead by the investigator, and the unstressed and stressed fish held separately. The unstressed fish would represent those fish having the best chance

for survival following immediate return to the water source. One could assume that stressed fish would have little chance for survival and use the long-term survival of initially unstressed fish as the measure of overall survival. Coutant's (1.3-7) studies with thermally-stressed fish indicate that predators are very sensitive to stressed fish and will selectively prey on them when they are presented simultaneously with unstressed controls.

The actual rate of predation on fish returned after impingement will be very site-specific depending on the abundance of predators and the environmental factors affecting their activity and behavior. An overall survival value based on the long-term survival of all live fish following holding should be considered an overestimate of actual long-term survival. An overall survival value based on the long-term survival of only initially unstressed fish may be somewhat low because some stressed fish would probably recover, particularly when the rate of impingement is high and predators may be saturated with prey. The selection of a survival value should be based on a consideration of local factors and the best judgment of the investigator.

1.3.3 Estimating Abundance When Site-Specific Data Are Not Available

The two approaches available when data are lacking are: 1) the extrapolation of data from an existing plant on the same water body to the site of interest, 2) the development of a best judgment estimate based on existing biological data. Recent impingement literature indicates that efforts to develop a predictive model for estimating impingement have been unsuccessful, primarily due to the large number of factors influencing the impingement rate and the uniqueness of each plant site and its environmental setting. Investigators should understand from the outset that impingement estimates for sites where data are lacking will at best be order of magnitude estimates.

1.3.3.1 Extrapolating data between plant sites. Gross (1.3-8) described an attempt to utilize existing impingement data from the Northport Power Plant on Long Island Sound to estimate impingement at two proposed sites on Long Island Sound which will have intake designs and operating characteristics similar to Northport. Gross compared the estimated annual impingement at Northport with the annual total for Millstone Nuclear Power Station Unit 1 on the opposite shore of Long Island Sound as a test of the transferability of data between plants on the same water body. He found reasonably good agreement between annual totals when adjustments

were made for differences in cooling water flow. In this case, it appears that extrapolating from Northport to the other Long Island Sound sites may be valid when estimates are adjusted for differences in plant flows.

Page et al. (1.3-9) compared the impingement of yellow perch fry and chinook salmon fry at two nearby intakes on the Columbia River. Twice as many yellow perch and thirty times as many chinook salmon were impinged at one intake than at the other. While there were a number of possible explanations for the differences, the evidence available was not sufficient to identify causative factors which could be used to predict impingement. If a new intake were proposed for this reach of the Columbia it is apparent that the existing data could provide only a very rough estimate of expected impingement.

Alevras (1.3-10) utilized impingement data from Indian Point Unit 1 to estimate expected impingement at Indian Point Unit 2 located a few hundred feet upstream on the Hudson River. The intake designs at the two plants are similar, but the flow is approximately three times greater at Unit 2 than at Unit 1. The estimating procedure adjusted for the difference in flows. Unit 2 is now in operation, and the impingement survey data indicate that the preoperational estimate was low by a factor of five when data are adjusted for sampling efficiency (1.3-11). In this instance, transferability of data might be expected to provide a close approximation of total impingement; however, there are obviously significant differences between these intakes not accounted for in the estimating procedure.

It is apparent from the above examples that transferring impingement experience among plant sites on the same water body may not produce reliable estimates of total impingement for use in impact assessment. A factor limiting the accuracy of estimates made by transferring data among plants is the annual variability in the total impingement of each species. At plants where a series of years of impingement surveys are available, high annual variability is a common feature of the data. The Argonne National Laboratory review indicates that many existing impingement data sets are limited to one year or less. In light of the high annual variability observed at some plants, a single year of data may not be representative of an average or typical impingement total. Multiple year data sets are obviously preferred where available.

1.3.3.2 Developing a best judgment estimate. When data are lacking for existing plants, it is particularly difficult to estimate expected impingement levels for a

new site. The value of a reliable estimate for preoperational impact assessment and design purposes has been recognized, but attempts to develop a predictive model have been unsuccessful. Although Murarka (1.3-12) developed a conceptual model to predict impingement, it was impossible to obtain reasonable estimates of model parameters. In another attempt to develop a model, Tetra Tech Inc. (1.3-13) reviewed impingement data for a number of plants to determine if the relationship between plant design or ecological factors and impingement could form the basis for prediction. While some general relationships were found, none was sufficiently strong or consistent to be used for prediction purposes.

Loar et al. (1.3-14) reviewed impingement data from 32 power plants in the Southeastern United States in an attempt to determine what factors were influencing the impingement of threadfin shad (*Dorosoma pretenense*). Data from some plants were limited, but water temperature and overall density of threadfin shad in the water-body were suggested as important factors. This study did not attempt to develop a predictive relationship, but the data were clearly insufficient for that purpose, even though they were for a single species and provided a large amount of impingement and ecological information.

There has been a great deal of interest in the influence of intake velocity and design on the rate of impingement. This interest stems from attempts to modify intake systems to reduce impingement and the attractiveness such simple relationships have for modeling. The predictive model attempts discussed above considered intake velocity and design; however, these factors are treated separately here because they are such commonly discussed aspects of impingement.

Intake velocity as a factor in impingement has often been related to the swimming ability of fish. The generalized concept is that intake velocities below the maximum sustained swimming speed of the species of interest (swim speed is commonly determined for the size range impinged) would permit the fish to escape, and as intake velocities increased the rate of impingement would increase. However, high impingement rates have been recorded at intake velocities below the maximum swim speed of some species, and there is circumstantial evidence that the impingement rate for some species actually declines at high intake velocities (1.3-15). Some species have also been observed to be attracted to the intake flow, and rather than attempt to escape, will follow the flow into the intake. At extremes of intake velocity, its influence may predominate, but extreme conditions such as very low

velocities would generally be impractical for intake systems. The important point is that in the range of velocities practical for intake systems, many factors can influence impingement and, therefore, intake velocity alone is not suitable for predictive purposes or as design criteria for new intakes.

Intake design cannot be analyzed as a factor in specific impingement problems separate from its environmental setting. Gross (1.3-8), for example, points out that the intake of the Northport plant on Long Island Sound violates several design features generally considered desirable for minimizing impingement, yet this intake system impinges fish at a rate far less than many plants with presumably better designs. As with intake velocity, intake design alone is not a useful approach to estimating the magnitude of impingement.

In this review of impingement studies, no basis was found for a predictive model of impingement which could be employed to estimate mortality at a proposed plant site. Therefore, available data on distribution and abundance of the species of interest and the accumulated experience from impingement studies must be used to develop a best judgment of the potential impingement rate. The following information has been extracted from impingement study reports and may aid the investigator in developing impingement estimates.

Clupeiform fishes represent a very large portion of all fishes impinged at U.S. power plants in all major habitat types. Alewife occur in large numbers at most Great Lakes plants, threadfin and gizzard shad rank among the most frequently impinged at many plants on Southeastern reservoirs, blueback herring and alewife are abundant at intakes on coastal rivers in the Northeastern United States, and northern anchovy occur in numbers at coastal plants in California. One should expect substantial impingement of clupeiforms at intakes sited on waterbodies where species of this group occur in abundance.

Among the many factors examined in a search for a predictive base for impingement estimates, water temperature and the relative abundance of fishes in the vicinity of the intake were most consistently related to impingement. These two factors are, in many cases, related because the relative abundance of fishes in certain areas of their habitat is determined by temperature-controlled behavior patterns. Alewife impingement in the Great Lakes, for example, is related to annual onshore movements for spawning which is temperature-related. The movement of migratory

species, which are often impinged in large numbers during periods of active migration, is another example of local abundance related to temperature. The young-of-the-year of blueback herring and alewife are particularly prone to impingement at plants on eastern estuaries during their downstream migration in the fall.

Another frequently observed temperature effect on impingement is peak winter impingement when ambient temperatures are lowest. Threadfin shad are particularly susceptible during the winter months (1.3-14) in those habitats where seasonal low temperatures approach lethal levels for this species. White perch are also very susceptible to impingement during winter in estuarine environments (1.3-16). For some species upwellings in the Great Lakes have been associated with episodes of high impingement, probably the result of dramatic shifts in temperature and the distribution of fish which follow a particular temperature regime.

Eisele and Malaric (1.3-17) have presented a conceptual model of the causal factors related to impingement of gizzard shad. They found temperature changes as a result of warm water recirculation to be a major factor in the occurrence of gizzard shad in the vicinity of intakes at power plants on Lake Erie. However, while temperature changes had a strong influence on local abundance, other factors, primarily turbidity and the condition of fish, were important in the daily rate of impingement. Their conceptual model could be useful for potential plant sites where gizzard shad are likely to occur in abundance.

A measure of the density of fishes by species in the vicinity of the proposed intake location is most useful for estimating impingement when no site-specific or water body impingement data are available. These data, in conjunction with other information such as temperature relationships, will provide the most useful basis for a judgment-type estimate.

Data on density of fishes will be of greatest value when they are as close to the intake site as possible and indicate the likely diel and seasonal patterns of fish abundance in the area. Hydroacoustics (see Section H1.4.6 and the example of density measurements at intakes; also 1.3-18) in conjunction with sampling with a variety of conventional fishery gear are the recommended approach for measurements of density. Hydroacoustics will permit rapid surveying of specific locations at the desired frequency to characterize diel and seasonal variability, and the conventional fishery gear will provide data on species composition and relative abundance.

With data on the density of fish in the vicinity of the intake site as the primary basis, the investigator must now make a qualified best judgment of the annual impingement rate. Whatever corollary information can be gleaned from the literature on other factors which may influence impingement at the site for the species of interest can be used to refine the estimate. It may be possible to obtain impingement survival data from the literature; however, survival can be very site-specific and transferring such data between plants should be done cautiously. Estimates of total annual impingement will be used in further analyses to assess impact on specific fish populations. Because these estimates are very rough, it may be desirable to assume a range of values for total impingement and conduct analyses over the range of values.

1.3.4 Design of Impingement Surveys

When data are not available for operating intakes, surveys can be conducted to characterize impingement and provide estimates of total plant mortality rate. In some cases existing data may be inadequate or limited in duration, thereby requiring new information for impact analyses. The following sections discuss both the practical and statistical aspects of designing a monitoring program to estimate the total number impinged. Section 1.3.2 should be consulted for recommendations on post-impingement survival studies.

1.3.4.1 Practical considerations. The collection of impingement samples takes place under conditions unfamiliar to most field biologists. The biologist will have to utilize power plant equipment to collect specimens and will not have complete control of the operations which influence the efficiency and consistency of the sampling program. Because plant managers are limited in the extent to which they can adjust operating procedures to accommodate sampling programs, plant equipment and operating conditions must be reviewed prior to the design of the sampling program. Sampling programs incompatible with plant operations will produce data of little value due to inefficient collections and gaps in the sampling program. The following discussion covers some of the typical problems of impingement sampling and recommended solutions or adjustments.

1.3.4.1.1 Plant flow rate. As will be discussed below, plant flow information can often be used to improve the precision of impingement estimates; therefore, a reliable daily record of plant flows should be kept for the entire

year. The main circulating water pumps in power plant cooling systems are generally single speed machines with an unvarying design output except for changes in the dynamic head resulting from changes in water level. There are a variety of problems and procedures in power plant operations that can result in downtime for circulating water pumps. Power plant operators typically do not keep records of pump operating times; therefore, estimates of total plant flow are difficult to develop, particularly since plant operation may continue with one or more pumps inoperative, or pumps may continue to operate after the generation process is discontinued. Plant operating time (duration of time the plant is on line) does not provide a reliable estimate of plant flow. It is recommended that a record of operating time for each circulating water pump and service water pump be maintained throughout an impingement study.

Multi-speed pumps have been installed in some plants to provide a lower speed and flow rate to reduce impingement rate on a seasonal basis when plant electrical output can be maintained at a reduced circulating water flow (due to low ambient water temperature). Seasonal changes in plant flow rate must be accounted for in estimates extrapolated from samples collected throughout an annual cycle, and changes in pump flows should be incorporated into the record of pump operating time.

At power plants in cold climates, ice formation on intake screens is prevented by recirculating a portion of the heated discharge water (tempering) to the intake structure upstream of the screens and pumps. This addition of water to the intake reduces the flow from the water source into the intake structure. The lower total flow rate must be accounted for in a work-up of the impingement rate based on a unit of flow. A record of tempering flow should therefore be maintained throughout the study period.

1.3.4.1.2 Screen mesh size. Screen mesh of 3/8 in. woven wire is practically standard at intakes of U.S. power plants. Screen mesh size determines whether an organism is impinged or entrained through the cooling system. Where the impingement impact is evaluated for a number of power plants on the same waterbody, the investigator should substantiate that the standard 3/8 in. mesh is indeed employed at each plant. This consideration is particularly important when impingement rate estimates are extrapolated from a plant with impingement counts to a plant lacking these counts, or when length data are not available to verify that each plant begins cropping at the same life

stage. Population impact assessment is often done separately for different life stages of a species and combined for total impact. It is extremely important that estimated mortality rates for each plant in such an analysis apply to the same life stages.

1.3.4.1.3 Non-operating intake screens. The intake systems of power plants commonly employ a series of intake channels (forebays), each equipped with traveling water screens. Another feature of intake structures is a common trash sluice which collects the debris (and impinged fish) washed from all the screens in the intake structure and discharges to a trash basket or via a return sluice to the waterbody. Samples of impinged fish are often obtained from the trash basket or a collection device in the return sluice. The sample is a composite of fish washed for all operating screens.

When a traveling screen breaks down, the flow is generally permitted to continue through the intake channel as long as the debris load does not exceed the screen's structural capacity. Fish impinged on the non-operating screen will accumulate and may not be removed during the impingement sampling interval. When the screen is repaired, the accumulated fish may be washed from the screen between sampling intervals and, therefore, lost from the original samples. If losses caused by screen breakdown are not accounted for, the total number of fish impinged will be underestimated. It is recommended that a record of screen downtime be maintained along with the pump operating data.

The adjustment of impingement counts for non-operating screens can be made by the calculation of a per screen impingement rate and the application of this rate to the number of non-operating screens. It has been observed at some multi-channel intakes that the per screen impingement rate consistently varies across the intake structure. If the impingement rate by screen is available for an extended period, the adjustment for non-operating screens should utilize the individual screen rates.

1.3.4.1.4 Screen losses. Fish and debris are removed from traveling screens by a wash system consisting of a spray wash manifold mounted behind and spraying through the screen surface. Spray nozzles are known to become misaligned which results in an uncleaned vertical section of screen. Fish on this section of the screen will not be removed and may be dislodged into the

pump flow as the screen rotates. This loss can be detected by the occurrence of fish on the tube sheet on the inlet side of the plant condensers when maintenance inspections are made. This loss is generally small and infrequent, but should be checked during an impingement survey.

1.3.4.1.5 Screen operating procedures. Traveling screens are designed to operate intermittently for the removal of accumulated debris. The screens can be programmed to run automatically for a set interval of time (generally 10 to 15 minutes) when the head differential across the screen, produced by the accumulated debris, reaches a critical level. The frequency of cleaning is dependent upon the rate of debris accumulation when the screens are programmed to run automatically. Traveling screens can also be operated manually, when the frequency of cleaning is determined by a plant maintenance schedule. The screens may be operated continuously when the debris load is very heavy or in an attempt to reduce impingement mortality by minimizing the time fish are impinged against the screen mesh. Continuous operation is not often used, however, because conventional screens are not designed for the excessive wear imposed by continuous operation.

To obtain accurate impingement counts, the screens must be operated in a consistent manner. Automatic screen operation is unacceptable because the collection effort (removal of impinged fish from the screens) will be variable between screens and over time. Constant surveillance of the screens would be needed to record the operating time for each screen. For impingement sampling, the screens should be operated manually. A workable procedure is to wash all screens at the beginning of the sampling period to remove accumulated debris and fish. The screens then remain stationary until the end of the desired sampling interval when each screen is cleaned for an equal time period. For post-impingement viability studies the screens should be operated in the normal manner to measure survival under typical operating conditions.

1.3.4.1.6 Recirculation of dead fish. With the realization that fish can survive impingement if they are removed from the screens quickly and returned to the water body, trash sluices at many plants have been modified and extended to bypass the trash basket and return fish to the water source. Of the fish returned, a high percentage may be dead or will die from injuries and stress

shortly after release. The return systems release the impinged fish in the vicinity of the plant and, therefore, they may be carried by tidal currents or pump flows back to the intake structure. A release of marked dead fish and field observations should indicate the occurrence of recirculation, which may be quantified, and an appropriate adjustment in the estimate of total mortality can be made.

1.3.4.1.7 Bar racks. Intake bar racks prevent large debris from making contact with the traveling screens. The vast majority of fish impinged on power plant intake screens pass through the bar racks. However, occasionally, large fish impinge on the bar racks and may not be accounted for in impingement collections. Bar racks should be visually inspected, and the debris from bar rack cleanings should be examined for the presence of large fish.

1.3.4.2 Impingement sampling design. The validity of assessments made from existing impingement data may be open to question when the data have not resulted from an appropriate statistical sampling design. Regardless of the manner in which existing data have been collected, any additional impingement studies should be performed in a statistically acceptable manner. While the perpetuation of existing sampling approaches may be necessary to ensure year-to-year comparability of results from several types of aquatic studies, this clearly does not apply to impingement surveys.

A key benefit of utilizing statistical sampling designs for impingement surveys will be that the resulting estimators will have known properties (e.g. unbiasedness, consistency) and quantifiable precision. Several different types of design may require vastly different levels of sampling effort in order to obtain results of similar precision. Existing impingement data, when available, will be useful in the development of a sampling design which will minimize the cost of obtaining estimates which meet stated precision requirements.

1.3.4.2.1 Sampling design. Murarka and Bodeau (1.3-19) (see also 1.3-20 for a summary of this work) have reviewed sampling designs for estimating fish impingement and compared designs using simulated and actual impingement data. Although this study is not exhaustive, it provides useful guidance in the

planning of impingement surveys. Following are the results of their work and several recommendations to complete the discussion of impingement sampling design.

Murarka and Bodeau concluded that a stratified systematic random sampling scheme or a stratified random sampling scheme was the most appropriate for estimating impingement. They further recommended that a large number of strata with variable intensity of sampling between strata would greatly improve the precision of estimates for a given level of sampling effort. In those strata with extreme variability, a complete census may be desirable. The Neyman method of optimal allocation (1.3-21, pp. 95-97) provides a means of determining the appropriate sampling intensity for each stratum.

Kumar and Griffith (1.3-22) recommend temporally stratified sampling programs for estimating fish impingement and provide a procedure for selecting strata and optimizing the allocation of samples. Their procedure relies upon an existing impingement data base or corollary data to which impingement is known to be related. Unfortunately, the ad-hoc statistical rule which they apply eliminates, to some extent, the judgment of the investigator in the selection of strata. Their procedure is applied to individual species and in a situation where several species are important, the final sampling design is a compromise of the designs developed for the individual species. If the designs must be compromised, it is probably better for the investigator to use best judgment in selecting strata from the outset.

Kumar and Griffith also provide a procedure for design of temporally stratified sampling programs when existing impingement data are not available. This procedure relies on corollary data which are known to be related to impingement rate. The efforts to develop a predictive model for impingement suggest that the correlation between impingement rate and a large number of potentially controlling variables was generally poor and did not provide a sound basis for a general predictive model. This suggests that the proposed procedure will be of limited value. However, as impingement data are accumulated (see the Argonne reviews, for example) this procedure may be applicable to individual species in some environments.

Both stratified random (STRS) and stratified systematic (STSYS) sampling designs provide relatively good efficiency of estimation. An efficiency-based decision for selecting one of these classes of designs over the other will rarely be possible as the information required to make such a decision (1.3-23, pp. 356-358) will generally be available only for plants having one or more years of impingement census data.

The choice of sampling programs will depend to some extent on the intended use of the data. For the purpose of providing estimates of the total number of fish impinged by species, a stratified random design is preferred over the stratified systematic design. Stratified random designs provide a valid estimate of the standard error of impingement estimates, whereas the validity of a similar quantity from a STSYS is not guaranteed by statistical theory (1.3-23, pp. 369-370). In addition, as will be seen below, plant flow data, obtainable at relatively low cost, can often be used to improve the precision of impingement estimates in a STRS through the use of a ratio or regression estimation. This cannot be accomplished currently with STSYS designs.

The following symbols are required for the subsequent discussion:

- x_{ij} number of fish impinged on the j th sampling date of the i th stratum
- y_{ij} cooling water flow on the j th sampling date of the i th stratum
- n_i number of dates sampled in the i th stratum
- N_i number of dates in the i th stratum
- I the number of strata
- W_i total cooling water flow during the i th stratum

The following statistics are also used:

$$\bar{x}_i = \left(\sum_{j=1}^{n_i} x_{ij} \right) / n_i \quad (1.3-1)$$

$$\bar{y}_i = \left(\sum_{j=1}^{n_i} y_{ij} \right) / n_i \quad (1.3-2)$$

$$s_{x_i}^2 = \sum_{j=1}^{n_i} (x_{ij} - \bar{x}_i)^2 / (n_i - 1) \quad (1.3-3)$$

If, for the moment, plant flow is ignored, then the best available estimator of annual impingement is:

$$\hat{T} = \sum_{i=1}^{n_i} N_i \bar{x}_i \quad (1.3-4)$$

This is an unbiased estimator whose variance is estimated by the quantity:

$$\text{Var}(\hat{T}) = \sum_{i=1}^I N_i^2 \left(\frac{1}{n_i} - \frac{1}{N_i} \right) s_{x_i}^2 \quad (1.3-5)$$

Suppose now that the plant flows for each sampling date (y_{ij}) and each stratum (W_i) were known. These can be used to obtain a consistent estimator of annual impingement:

$$\tilde{T} = \sum_{i=1}^I W_i \tilde{R}_i \quad (1.3-6)$$

which has estimated variance:

$$\text{Var}(\tilde{T}) = \sum_{i=1}^I \left(\frac{1}{n_i} - \frac{1}{N_i} \right) \frac{N_i}{n_i - 1} \sum_{j=1}^{n_i} (x_{ij} - \tilde{R}_i y_{ij})^2 \quad (1.3-7)$$

where R_i is an estimate of impingement on a per unit volume basis:

$$\tilde{R}_i = \bar{x}_i / \bar{y}_i \quad (1.3-8)$$

The above procedure is referred to as ratio estimation (1.3-23, pp. 135-160).

The use of plant flow data in estimating impingement will improve precision only when plant flows vary appreciably between days within strata, and the action causing flow variation also affects impingement rates. Moreover, T will be an unbiased estimator of annual impingement only when the relationship between daily impingement and flow is a zero-intercept straight line for each stratum. This will most often occur when within-stratum flow variations are caused primarily by plant and/or unit shutdowns. In any case, the relationship of impingement to plant flow should be carefully examined before ratio estimation is attempted.

Whenever there appears to be a linear relationship between impingement and flow with an intercept noticeably differing from zero, the technique of regression estimation (1.3-21, pp. 189-205) may prove advantageous. Under these circumstances, regression estimators tend to have less bias and greater precision than ratio estimators.

Murarka and Bodeau (1.3-24) present sampling designs which can be used when the purposes of the program and the control factors are known. The control factors to be specified are coefficient of variation, margin of error, and confidence level. In a companion volume (1.3-24) they provide three computer programs and a user's manual for developing sampling designs and performing the computations needed to obtain estimates of fish impingement.

1.3.4.2.2 Estimation of impingement counts In the preceding discussion of impingement sampling design and estimation, it was assumed that all collected samples have been completely and accurately enumerated. However, it is sometimes unfeasible to completely enumerate samples containing very large numbers of fish. In such cases it will be necessary to obtain an estimate of the true count for each species based on an analysis of a portion of the sample. This may be accomplished by means of statistical sampling designs.

The first step in the process will be to divide the total catch into sampling units. These should be containers into which the fish are placed. A number of these containers are then selected for analysis. The distribution of fish among the containers need not be randomized in any way. However, the selection of containers for analysis should be made in accordance with a simple random sampling plan; i.e., the containers should be numbered consecutively and a random numbers table or generating device used for selection.

The following symbols are used in the following discussion:

- N the number of containers used
- n the number of containers sampled
- x_{ij} the number of individuals of species i in the jth container
- W_{ij} the total biomass of species i in the jth container
- $W_{.j}$ the total biomass of all species in container j
- W_i the total biomass of species i in all N containers
- W. the total biomass of all species in all N containers

An unbiased estimator of the total count of species i is:

$$\hat{x}_i = N \left(\sum_{j=1}^n x_{ij} \right) / n \quad (1.3-9)$$

with standard error:

$$\text{S.E. } (x_i) = N \sqrt{\left(\frac{1}{n} - \frac{1}{N} \right) S_i^2} \quad (1.3-10)$$

where

$$S_i^2 = \sum_{j=1}^n \left(x_{ij} - \frac{x_i}{N} \right)^2 / (n-1) \quad (1.3-11)$$

If there is substantial variation in biomass among containers and the measurement of that biomass can be accomplished at little cost, then another possible estimator of the total count of species i is:

$$\tilde{X}_i = W. \times \hat{R}_i \quad (1.3-12)$$

where

$$\hat{R}_i = \sum_{j=1}^n x_{ij} / \sum_{j=1}^n W_{.j} \quad (1.3-13)$$

The standard error of this estimator is:

$$S.E.(\tilde{X}_i) = \sqrt{\frac{N(N-1)}{n-1} \sum_{j=1}^n (x_{ij} - \hat{R}_i W_{.j})^2} \quad (1.3-14)$$

If it is somehow possible to determine, at reasonable cost, the biomass by species for each of the N containers, then W_i and W_{ij} should be used in place of W and $W_{.j}$ in the above formulae.

Whenever estimation procedures are used in the place of the complete enumeration of samples, additional sampling variability results. This will be reflected in the precision of estimates of annual impingement. Sampling variability introduced by sample count estimation may result in the need for considerably more sampling dates to attain the required precision than would be necessary if the samples are completely enumerated. Clearly there is an economic trade-off here which should be carefully considered when deciding whether or not to completely enumerate large samples.

An additional point to be considered when sample count estimation is used is the likelihood of detection of rare species. Consider a sample containing 10,000 fish, of which three are of a particular species considered to be "rare and endangered." Had the catch been randomly dispersed among twenty containers, of which three were analyzed, the probability that no specimens of that species would appear in the analysis is about 64%. This is a potentially serious problem when a rare species has been selected for analysis. When it is necessary to sort large samples to enumerate rare species, it will probably be cost effective to enumerate all species at the same time.

1.4 ESTIMATING POPULATION SIZE

1.4.1 Mark-Recapture Methods

Mark-recapture methods represent the most widely applied current techniques for estimating the size of fish populations. Mark-recapture methods were first used in enclosed water bodies by Petersen in 1896 (1.4-1) and by Dahl about ten years later. Applications to ocean fishes started toward the end of the first decade of this century. Statistical techniques for calculating unbiased variance parameters and confidence intervals for mark-recapture estimates have been thoroughly elaborated and appear in comprehensive treatments by DeLury (1.4-2), Chapman (1.4-3), Regier and Robson (1.4-4), Seber (1.4-5), and Ricker (1.4-6).

However, Watt (1.4-7) points out that effort expended by investigators to compute error estimates and confidence limits is of secondary importance to obtaining comparable population estimates from several of the different methods which are discussed throughout Section 1.4 and which rely on quite different assumptions. He goes on to say that in the field of indirect measurement of population size, e.g. mark-recapture methods, catch-per-unit-effort methods, etc., problems caused by low precision are very small by comparison with problems caused by bias stemming from sources such as gear selectivity, non-random sampling, etc., and that often only the latter are worthy of serious consideration. This caveat should be heeded by those who feel that budget limitations restrict the scope of the project to include only one of the methods outlined in the following section.

1.4.1.1 Closed Population: Single Mark-Release

1.4.1.1.1 The Petersen method. A simple and widely used method for estimating N , the number of animals (fish, in this case) in a closed population*, is called the Petersen method after its originator. A sample of n_1 fish is taken from the population, each fish is tagged or marked for future identification and then returned to the population. After allowing sufficient time for marked and unmarked fish to mix, a second sample of n_2 fish is then

*A population which remains unchanged during the period of investigation, i.e., negligibly affected by migration, mortality, and recruitment, is called a closed population. If the population changes because of the effects of one or more of these processes, then the population is termed open (1.4-5).

taken, and it is found that m_2 are marked. It is assumed that the proportion of marked fish in the second sample is a reasonable estimate of the unknown population proportion, i.e., assuming that:

$$m_2/n_2 = n_1/\hat{N}, \text{ an estimate of } N \text{ is } \hat{N} = \frac{n_1 n_2}{m_2} \quad (1.4-1)$$

If $\hat{N}$ is to be a suitable estimate of N , the following assumptions must be satisfied:

- a. The population is closed so that N is constant.
- b. All fish have the same probability of being caught in the first sample (n_1).
- c. The marked fish are as vulnerable to recapture fishing as are the unmarked ones, i.e., marking does not affect catchability.
- d. The marked fish suffer the same natural mortality as the unmarked.
- e. The marked fish do not lose their mark.
- f. All marks are recognized and reported upon recovery in the second sample. In addition, unmarked fish collected during recapture sampling must be accurately reported.
- g. The marked fish become randomly mixed with the unmarked, or the distribution of recapture fishing effort is proportional to the number of fish present in different parts of the water body, i.e., the second sample is a simple random sample.

In order to satisfy the assumption (a) of a closed population, time and space boundaries must be judiciously chosen. For example, it would not be appropriate to collect n_1 and n_2 when recruitment of young, unmarked fish to the n_2 population was a significant factor, since this would result in an overestimate of the initial population. However, corrections for the effect of recruitment can be applied with one of several methods described by Parker (1.4-8), Seber (1.4-5) and Ricker (1.4-6). Note that if natural mortality is

the same for marked and unmarked groups (assumption d), it does not interfere with the accuracy of the population estimate; therefore, the time interval between the collection of n_1 and n_2 is unimportant with respect to natural mortality. Both assumptions (d) and (e) should be verified by comparison of mortality rates of retained marked fish with those of control fish and observation of tag retention.

Spatial boundaries should be chosen to minimize the effects of immigration into and emigration out of the study area during the period between n_1 and n_2 . Spatial boundaries should not present a problem in smaller water bodies where it is economically feasible to sample the entire habitat.

In larger water bodies such as oceans or very large lakes, spatial boundaries of the subject population should be defined by a pilot mark-recapture study or other technique which would indicate whether the population comprised a homogeneous entity distributed throughout the water body, or whether discrete sub-populations existed. Dispersal of the marked segment throughout the water body, for example, would indicate a wide-ranging, relatively homogeneous unit stock, requiring a much wider range of sampling effort than would be necessary for a species which tended to remain localized within a provincial sector of the water body.

A common weakness in mark-recapture studies is the failure to satisfy assumption (g), which most often leads to an underestimate of the true population size. For example, a sampling design which results in a non-random concentration of marking and recapture effort within a single segment of a water body will generally lead to an underestimate of the size of the population residing in the entire water body, if the data from all segments are treated as a whole.

Ricker (1.4-6) gives the following example of a lake population of 20,000 fish, 10,000 in each half. Twice as many traps are set in one half as the other so that, both for marking and for recoveries, one end is sampled twice as much as the other. In a Petersen type experiment, one-fifth of the fish at one lake end and one-tenth of those at the other are marked. Similarly, after mixing of the marked fish with the unmarked, one-fifth and one-tenth, respectively, are taken, and the marked fish among them recorded. Sampling error eliminated, the result is as follows:

		<u>First Half</u>	<u>Second Half</u>	<u>Total</u>
Actual population	(N)	10,000	10,000	20,000
Number marked	(n ₁)	2,000	1,000	3,000
Sample taken	(n ₂)	2,000	1,000	3,000
Recoveries	(m ₂)	400	100	500

If the data are treated as a whole, the estimated population is $3000 \times 3000 / 500 = 18,000$, which is 10% low. This error can be avoided by considering the two halves of the lake separately, calculating the population of each, and adding them together. Thus, if there is any reason to suspect unequal fishing (marking and recapture) effort in two or more parts of a water body, the experiment should be divided into its respective parts, because the error usually makes the result of a common calculation less than the sum of the separate calculations. Although the population estimate for a water body is not biased if either the marking or the subsequent sampling is randomly done, it is advisable to try to make both marking and recapture efforts random.

A similar problem, tantamount to unequal fishing effort, arises when the collection gear used does not have equal fishing power for each age (size) group in the population, i.e., each age group or size stratum exhibits a different vulnerability to the fishing gear. Overton (1.4-9) emphasizes that a stratified Petersen estimate, requiring ancillary data such as age or length in order to assign fish to the various strata, should be used whenever any doubt exists regarding the assumption of a uniform fishing rate over all sub-units of the population. Ricker (1.4-6) agrees that variation in size vulnerability, though commonplace, will not usually be a serious problem since its effects can be minimized by dividing the fish into two or more size groups, i.e., assigning them to strata.

The results of a Petersen mark-recapture study conducted by LMS on the Hudson River are presented in Table 1.4-1 showing the difference between stratified and unstratified Petersen estimates for a multi-age, redbreasted sunfish population. This population was located in a power plant intake pond closed to immigration and emigration by nets placed across an inlet leading to the Hudson River. Provision for a stratified design was

Table 1.4-1
STRATIFIED AND UNSTRATIFIED PETERSEN ESTIMATES (N* AND N)
FOR A RED-BREASTED SUNFISH POPULATION IN BOWLINE POND,
SEPTEMBER 1974

Age Group	Number Marked (n_1)	Number Recaptured (n_2)	Mark Returns (m_2)	N*	$\sqrt{v^*}$	95% Range	$\hat{N}$
0 and 1	102	151	6	2237	745.619	775 to 3698	2567
> 2	<u>113</u>	<u>105</u>	<u>17</u>	<u>671</u>	128.775	<u>419 to 923</u>	<u>698</u>
Totals	215	256	23	2908		1194 4621	3265

$$\text{Unstratified } N^* = \frac{(216)(257)}{24} = 2313 \quad \sqrt{v^*} = 415.280 \quad 95\% \text{ Range } 1499 \text{ to } 3127$$

$$\text{Unstratified } \hat{N} = \frac{(215)(256)}{23} = 2393$$

$$N^* = \frac{(n_1+1)(n_2+1)}{(m_2+1)} \quad (2)$$

$$\hat{N} = \frac{n_1 n_2}{m_2} \quad (1)$$

$$v^* = \frac{(n_1+1)(n_2+1)(n_1-m_2)(n_2-m_2)}{(m_2+1)^2(m_2+2)} \quad (3)$$

necessary because the potential existed for size-selective capture and recapture. The decision to provide for a stratified design had to be made a priori, since all the marked and recaptured fish had to be measured in order to assign them to approximate age groups (strata), an unnecessary step with an unstratified design. As with the previous example, single unstratified estimates are generally lower than the sum of the stratified estimates if the assumption of equal effort across all strata is not fulfilled.

When the assumptions listed earlier are satisfied, the conditional distribution of m_2 (the number of marked fish), given n_1 and n_2 , is the hypergeometric distribution, which is equivalent to the binomial distribution, but accounts for the fact that sampling occurs from a finite population without replacement. Because of the characteristics of this distribution, N is not an unbiased estimator of N , especially in the case of small sample sizes ($n_1 + n_2 \ll N$). Chapman (1.4-10) modified the formula as follows to provide an unbiased estimate in most situations:

$$N^* = \frac{(n_1 + 1)(n_2 + 1)}{(m_2 + 1)} - 1 \quad (1.4-2)$$

Seber (1.4-5) points out that when the number of recaptures (m_2) is greater than or equal to 7, there is 95% confidence that the bias of N^* is negligible. An approximately unbiased estimate of the variance of N^* has been provided by Seber (1.4-11) as

$$v^* = \frac{(n_1 + 1)(n_2 + 1)(n_1 - m_2)(n_2 - m_2)}{(m_2 + 1)^2 (m_2 + 2)} \quad (1.4-3)$$

and an approximate 95% confidence interval for N (as $N \rightarrow \infty$) is given by

$$N^* \pm 1.96 \sqrt{v^*} \quad (1.4-4)$$

Robson and Regier (1.4-12) point out that in many sampling situations, n_1 and n_2 are so small that the recapture sample may not contain any marked members. When this possibility exists, all forms of the Petersen estimate,

whether N or N^* , are biased downward by a factor equal to the probability of getting no recaptures, i.e., the probability that $m_2 = 0$. This bias is approximately $100 e^{-n_1 n_2 / N} \%$ (1.4-10), and thus becomes negligible only when the product of the two sample sizes ($n_1 \times n_2$) exceeds the population size (N) by at least a factor of three or four. If $n_1 n_2 \approx N$, then the Petersen estimator underestimates N by approximately 37% on the average, while if $n_1 n_2 \approx 3 N$ then the bias is reduced to 5%, and for $n_1 n_2 \approx 4 N$, the bias is less than 2%. At the other extreme, where sample sizes are unreasonably small in relation to population size, the bias of Peterson-type estimators approaches 100%. The list of conditions necessary for the validity of the Peterson estimate (discussed previously) should therefore include the following: the product of the two sample sizes n_1 and n_2 must exceed four times the population size N (1.4-12).

Paradoxically, one of the prerequisites to a successful mark-recapture study is an initial educated guess of the population size N , based on the literature, experience, and/or a pilot study. The purpose of the actual experiment may then be regarded as the objective confirmation of or improvement upon this earlier estimate. This step is necessary because the precision of estimation associated with any given sample size depends upon the size of the population from which the sample is drawn.

Once an initial estimate of population size has been made, the level of accuracy (A) and precision ($1 - \alpha$) which will fulfill the goal of the study must be determined. Precision, or $1 - \alpha$, is the probability that the Petersen estimate N will not differ from the true population size by more than 100A%. Three standard levels for α and A are recommended by Robson and Regier (1.4-12):

1 - α = 0.95, A = 0.50, recommended for preliminary surveys where only an approximation of population size is needed

1 - α = 0.95, A = 0.25, recommended for more accurate management work.

1 - α = 0.95, A = 0.10, recommended for careful research into population dynamics.

Population estimates are often separated into age groups or are multiplied or divided by other estimates (e.g., multiplied by mean weight to obtain total biomass, divided by another population estimate to obtain survival). Relatively high precision and accuracy are required of each estimate if the variance of the product or ratio of such estimates is to be reasonably small. In such situations, the highest level (0.95, 0.10) should be routinely used.

Robson and Regier (1.4-12) give charts (Figures 1.4-1 to 1.4-3) for determining n_1 and n_2 from a rough initial estimate of N for the above three levels of α and A . They point out that for the recommended values of n_1 and n_2 , the previously discussed bias is negligible (less than about 1%). To illustrate the use of the figures, suppose that N is estimated to be 1,000,000 and levels of $1 - \alpha = 0.95$ and $A = 0.10$ are required. Then from Figure 1.4-3 we obtain possible values for n_1 and n_2 (or n_2 and n_1 , since the relationships for determining the charts are symmetric), such as (20,000; 18,000), (100,000; 3,500), (800,000; 100) are obtained.

Experiments utilizing markedly unequal sample sizes (either large n_1 and very small n_2 or vice versa) are generally much more costly than if more nearly equal-sized samples are used. Suppose c_1 and c_2 are the costs per fish of catching-marking and catching-examining and f represents the known fixed or overhead costs, so that the total cost of the experiment is:

$$c_T = c_1 n_1 + c_2 n_2 + f \quad (1.4-5)$$

If funds are limited and the accuracy A is predetermined, then an optimum allocation would be to choose n_1 and n_2 which maximize $1 - \alpha$, subject to c_T being fixed. Robson and Regier (1.4-12) show that the solution of this is:

$$\frac{c_1 n_1}{c_2 n_2} = \frac{N - n_2}{N - n_1} \quad (1.4-6)$$

and given estimates of c_1 , c_2 and N , the above equations can be solved for n_1 and n_2 . When n_1 and n_2 are small relative to N , then Eq. 1.4-5

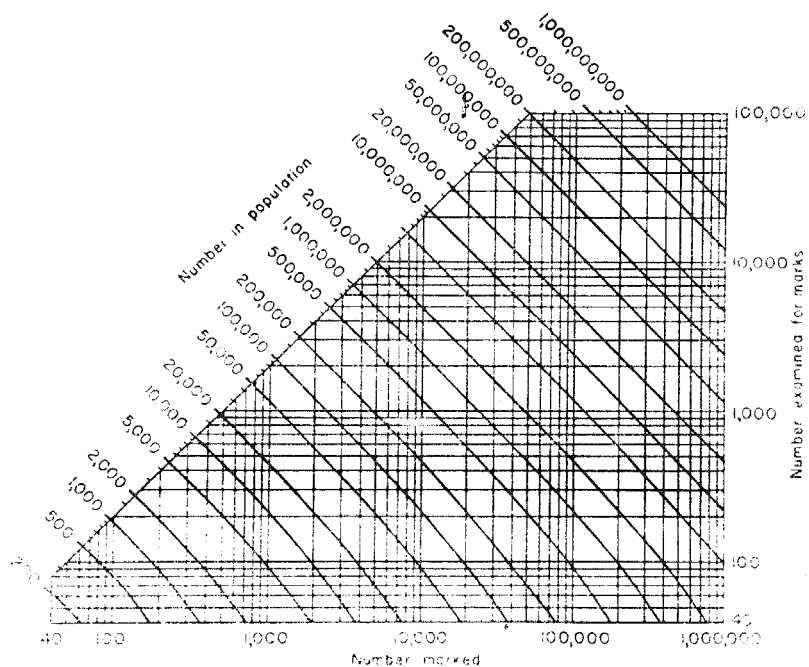


Figure 1.4-1. Sample size when $1-\alpha = 0.95$ and $\rho = 0.50$; recommended for preliminary studies and management surveys. Data based on normal approximation to the hypergeometric distribution.

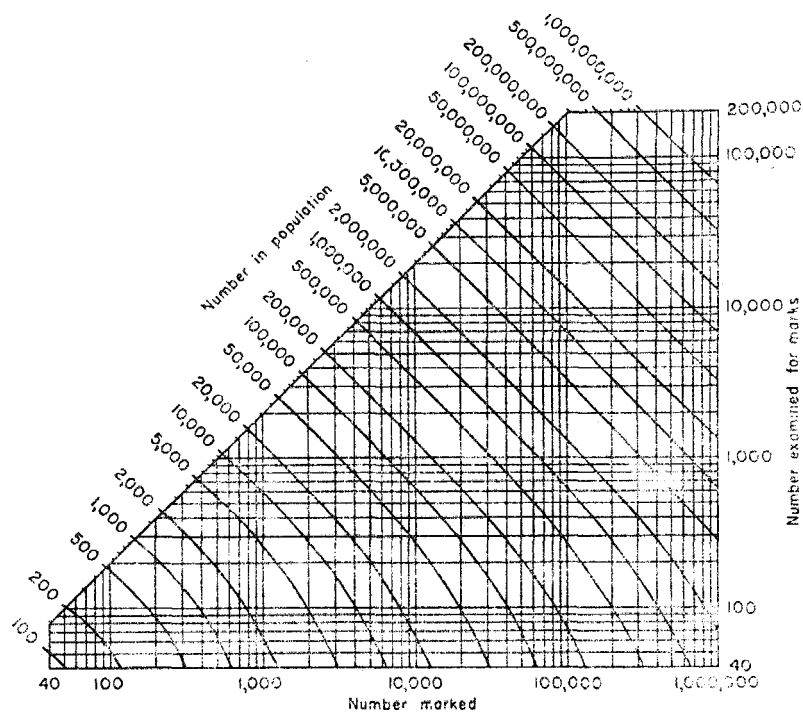


Figure 1.4-2. Sample size when $1-\alpha = 0.95$ and $\rho = 0.25$; recommended for management studies. Data based on normal approximation to the hypergeometric distribution.

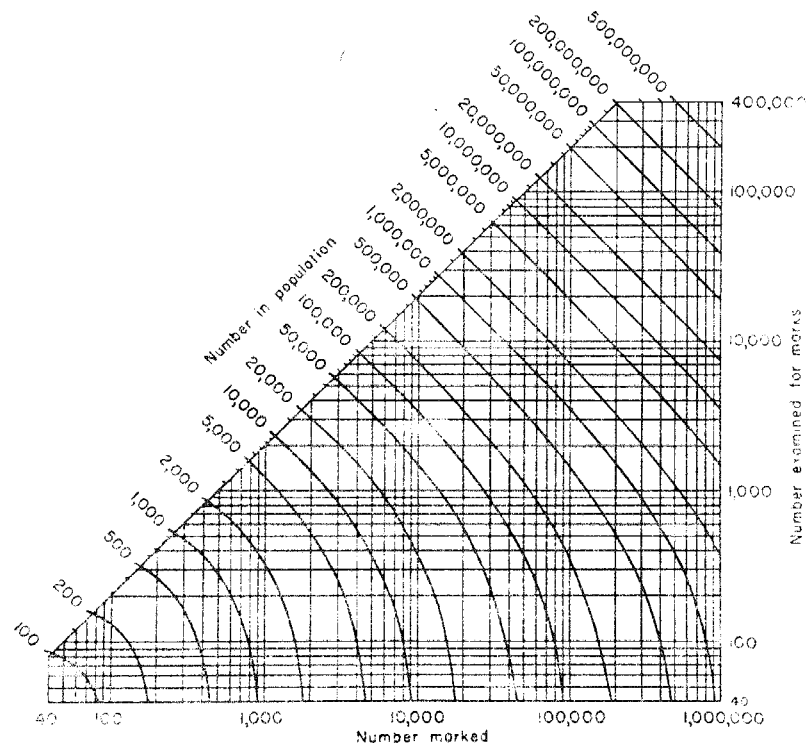


Figure 1.4-3. Sample size when $1-\alpha = 0.95$ and $\rho = 0.10$; recommended for research. Data based on normal approximation to the hypergeometric distribution.

becomes $c_1 n_1 \approx c_2 n_2$, so that an equal division of resources between catching-marking and catching-examining is optimal.

If funds are not restricted, so that both α and A can be fixed in advance, then n_1 and n_2 would be chosen so that c_T is minimized. Given rough estimates of N , c_1 and c_2 , Robson and Regier (1.4-12) show that the solution is obtained by consulting the appropriate curve in Figures 1.4-1 to 1.4-3 and choosing the pair (n_1, n_2) satisfying Eq. 1.4-5.

1.4.1.1.2 Manly-Parr method. Jones (1.4-13) states that it is unlikely that the stipulation that $n_1 \times n_2$ must exceed $4N$ can be met for many fish populations because of effort and cost limitations. A cost-effective large population program would be to direct marking effort at a definable segment of the population, such as a single age class, rather than at the entire population. If that stratum or class is enumerated by a Petersen or other suitable estimate, then the remainder of the population can be quantified by the Manly-Parr (1.4-14) method. Notation for this method is described after Seber (1.4-5) as follows:

- t_i = time when the i th sample is taken
- N_i = total number in the population just before time t_i ;
- n_i = number of fish caught in the i th sample
- p_i = equiprobability that any fish in the population, regardless of class or mark status, will be caught in the i th sample
- C_i = total number of fish within class C of the population just before time t_i ;
- c_i = number of fish in class C caught in the i th sample.

When all fish have the same probability of capture (p_i) in the i th sample, the mean value of n_i (conditional on a fixed N_i) is

$$\bar{n}_i = N_i p_i$$

Therefore, given an estimate $\tilde{p}_i$ of p_i , the estimate

$$\tilde{N}_i = n_i / \tilde{p}_i$$

results.

Here p_i can be estimated by picking out a class of C_i , animals known to be in the population at the time of the i th sample, and then observing the number c_i of this class caught in the i th sample, i.e.

$$\tilde{p}_i = c_i/C_i$$

and therefore

$$\tilde{N}_i = n_i C_i / c_i \quad (1.4-7)$$

Note that N_i is basically a Petersen estimate in which the n_i individuals caught in the i th sample represent the "first" sample (n_1) and the C_i animals represent the "second" sample (n_2).

It is important to realize that this technique is especially dependent upon the assumption that all fish of all classes have an equal probability of capture in each sample. The most likely classification for this type of study is by age groups. Thus, it is necessary to find a sampling gear which collects all age groups with equal efficiency, i.e., a fishing method which is not age (size) selective.

In reality, there is probably no such sampling method, short of poisoning an entire population or draining the water body. A more practical alternative is to sample the population using several gear types, with different fishing characteristics, such as multi-panel gill nets, haul seines, trap nets and otter trawls. Some idea of differences in selectivity is then obtained by examining gear-specific length-frequency distributions for the species of interest. If the length-frequency distributions were identical or entirely overlapping, it could be safely assumed that any of the four gears could be used singly or interchangeably.

However, it is much more likely that each gear would exhibit its own selectivity by collecting different size distributions of fish, perhaps similar to the hypothetical distributions in Figure 1.4-4. Although all of the length distributions in this figure overlap to some extent, it is evident that the

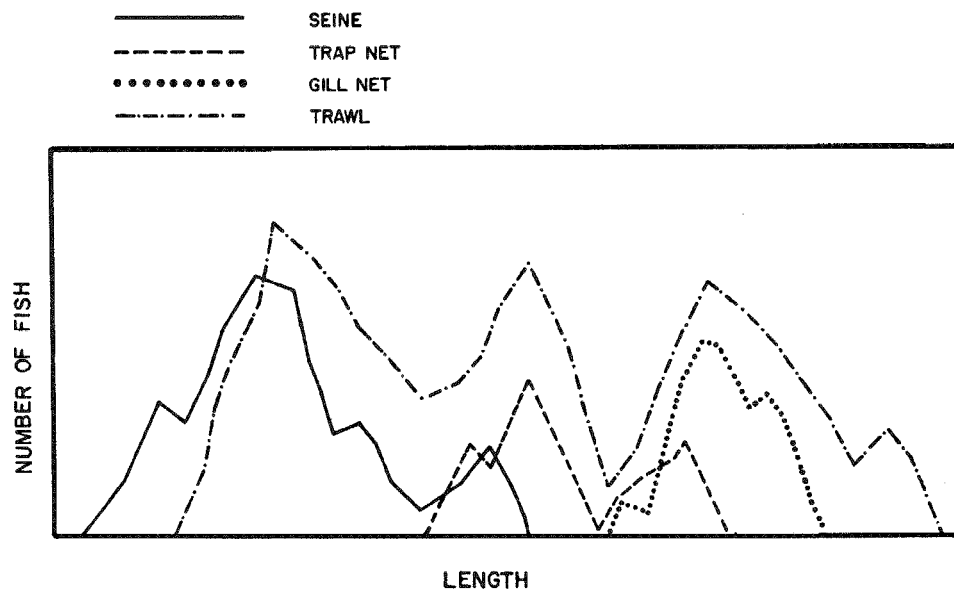


Figure 1.4-4. Hypothetical Length - Frequency Distribution For Four Gear Types

trawl captures a much broader age spectrum of the population than any of the other collecting gears, and it is therefore the gear most likely to collect all age groups with equal efficiency.

Selecting the trawl from other available fishing gears based on examination of length distribution data is a first step in the evaluation of the applicability of the Manly-Parr method. The next step is to test the hypothesis of equal efficiency across age groups by a pilot study in a relatively small water body where a known number of marked fish in each age group have been released. If the rate of fishing, i.e., the percentage recaptures remains the same for each age group, the hypothesis of equal efficiency for the trawl is confirmed. Deviations from the assumption of equal catchability can be quantified and used as correction factors in the full-scale Manly-Parr study. Ricker (1.4-6) states that a rate of fishing independent of age is more likely for a trawl fishery than for collections from gill nets and trap nets, which are notoriously size-selective (1.4-15, 1.4-16, and 1.4-17).

A primary advantage of the Manly-Parr method over the Petersen method is the reduction in the amount of sampling effort (cost). Consider the following scheme for a population of 1,000 fish composed of three age groups, i.e., 500, age 1; 300, age 2; and 200, age 3. With the Petersen method an estimate of N is obtained by marking and releasing 375 fish and recapturing 400 fish, using Robson and Regier's recommended schedule in Figure 1.4-3.

According to the Manly-Parr method, with age-group 3 as the C_1 stratum, it is only necessary to mark and release 100 age-group 3 fish and recapture 150 age-group 3 fish using the same Figure 1.4-3 schedule, which represents 60 to 70% reduction in effort for this particular example. During the recapture efforts, of course, numbers of age-group 1 and 2 fish are also collected and recorded and then used in estimates of N , by Eq. 1.4-7. With the assumption that five trawls were required to collect the requisite 150 age-group 3 fish for the estimate of c_1 , the data and calculations for this hypothetical example are presented in Table 1.4-2.

Although the reliability of the Manly-Parr method depends upon fulfillment of the assumption that all age groups have the same probability of capture in

Table 1.4-2

RECAPTURE DATA FROM A HYPOTHETICAL MANLY-PARR MARK-
RECAPTURE STUDY WHERE FIVE TRAWLS WERE NEEDED TO PROVIDE n_2

Trawl	Number Captured From Marked Structure (c_i)	Recaptures (m_2)	Number Captured From All Age Groups Other Than c_1 (X_i)	Total Number Of Fish From All Age Groups Including c_1 ($n_i = c_i + X_i$)	Total Population Estimates For All Age Groups ($\tilde{N} = n_i c_i / c_i$)
1	30	15	125	155	1033
2	10	5	35	45	900
3	55	25	227	282	1025
4	5	3	22	27	1080
5	<u>50</u>	<u>27</u>	190	240	<u>960</u>
$n_1=100$	$n_2=150$	$m_2=75$		Average $\tilde{N}_{(1,2,3)} = 1000$	

$$\hat{N}_{(3)} = c_1 = \frac{100 \times 150}{75} = 200 \text{ age 3 fish}$$

each sample, the Petersen estimate is not bound by this restriction, if the estimate is stratified by age or size groups.

1.4.1.2 Closed Population: Multiple Marking

1.4.1.2.1 Schnabel Census. A simple extension of the Petersen method (a series of two samples of sizes n_1 and n_2) to a series of s samples of sizes $n_1, n_2, \dots, n_s$ is the so-called Schnabel census (1.4-18). In this method, each sample captured (except the first) is examined for marked members and then every member of the sample is given another mark and returned to the population. Thus, the proportion of marked fish at large is successively increased, creating ever better data for the estimation of N .

Limiting constraints on this sequential approach are the resources available and the requirement that the total experiment be completed over a short enough period of time so that mortality in the population will be negligible. This latter requirement is directly related to the marking system used and is necessary only as long as the same batch-mark is being used on the successive batches to be marked. When numbered tags are used, or a distinctive batch-mark is used on each occasion, including the remarking and release of all mark-recaptures, then multiple recaptures become identifiable, and the assumption of no mortality during the between-sample periods is no longer necessary. The assumptions inherent within the multiple census technique are the same as those outlined for the Petersen method, i.e., assumptions (a) through (g) in Section 1.4.1.1.1.

Notation for multiple mark-recapture methods is as follows:

N = total population size
 s = number of samples
 n_i = size of the i th sample ($i = 1, 2, \dots, s$)
 m_i = number of marked individuals in n_i
 M_i = number of marked individuals in the population just before the i th sample is taken.
 M_{s+1} = total number of marked fish in the population at the end of the experiment, i.e., the total number of different fish caught throughout the experiment.

$$m = \sum_{i=2}^S m_i$$

$$\lambda = \sum_{i=2}^S n_i M_i$$

An approximation to the maximum likelihood estimate of N from multiple censuses is given by Schnabel's short formula:

$$N' = \left(\sum_{i=2}^S n_i M_i \right) / \left(\sum_{i=2}^S m_i \right) = \lambda / m, \quad (1.4-8)$$

which is simply a weighted average of the Petersen estimates $n_i M_i / m_i$. A modification of (Eq. 1.4-8) provided by Chapman (1.4-3, 1.4-19) reduces the bias of N' as an estimator of N:

$$N'' = \lambda / (m+1) \quad (1.4-9)$$

Because (Eq. 1.4-8) and (Eq. 1.4-9) ignore the complications of sampling without replacement, the proportion of marked fish to the total population, i.e., n_i/N and M_i/N must remain less than 0.1 throughout the experiment.

Confidence intervals for N/λ , and hence for N when $m \leq 50$ can be obtained from Chapman's (1.4-20) Poisson Table or Ricker's (1.4-6) Appendix II.

When $m > 50$ and the normal approximation to the Poisson distribution is used, a 95% confidence interval for N can be obtained from the relationship:

$$(N/\lambda) = \frac{2m + 1.96^2 + 1.96 (4m + 1.96^2)^{1/2}}{2m^2} \quad (1.4-10)$$

Where either n_i/N or M_i/N is greater than 0.1 for several samples, so that m will not have a Poisson distribution, the normal approximation to the binomial is appropriate and a 95% confidence interval for N is calculated by

$$(N/\lambda) = \frac{2m + 1.96^2 (1-\delta) + 1.96 \sqrt{(1-\delta) [4m + 1.96^2 (1-\delta)]}}{2m^2} \quad (1.4-11)$$

$$\text{where } \delta = \sum_{i=2}^S n_i M_i^2 / (\lambda N'') \text{ or } N', \text{ depending on the estimate.}$$

1.4.1.2.2 Schumacher - Eschmeyer Method. The Schumacher - Eschmeyer regression method is useful because it is a robust method less sensitive than the Schnabel census to the underlying assumptions of a closed population (constant N) and the random behavior of marked fish. In addition, it provides a graphical method which indicates any marked departures from the underlying assumptions, and the Schumacher-Eschmeyer model should be used in conjunction with other methods discussed here.

Consider a line fitted to values of the proportion of fish marked in each catch (m_i/n_i) plotted against the total number of fish marked previous to each catch (M_i), with the restriction that the line go through the origin; the slope of this line is an estimate of $1/N$. The best estimate of $1/N$ is obtained if each point is weighted by sample size, n_i , and this leads to the estimate

$$1/\tilde{N} = (\sum m_i M_i) / (\sum n_i M_i^2)$$

$$\text{and } \tilde{N} = (\sum n_i M_i^2) / (\sum m_i M_i) \quad (1.4-12)$$

Variance from the regression line of m_i/n_i against M_i is calculated by

$$\tilde{v} = \frac{\sum (m_i^2/n_i) - (\sum m_i M_i)^2 / \sum (n_i M_i^2)}{S-2} \quad (1.4-13)$$

and a 100 (1- α)% confidence interval for N is given by

$$\frac{\sum n_i M_i^2}{\sum m_i M_i \pm t_{[\alpha, s-2]} (\sum n_i M_i^2)^{1/2}} \quad (1.4-14)$$

1.4.1.2.3 Mean Petersen Estimate. Another robust method with regard to departures underlying the Schnabel census is to calculate a modified Petersen estimate of N at each stage of the sampling using

$$N_i^* = \frac{(M_i + 1)(n_i + 1)}{(m_i + 1)} \quad (i = 2, 3, \dots, s) \quad (1.4-15)$$

with variance estimate

$$v_i^* = \frac{(M_i + 1)(n_i + 1)(M_i + m_i)(n_i - m_i)}{(m_i + 1)^2 (m_i + 2)} \quad (1.4-16)$$

Therefore, a natural estimate of N, suggested by Chapman (1.4-20) is the average

$$\bar{N} = \sum N_i^* / (s-1) \quad (1.4-17)$$

Variance can be estimated by either

$$v^* = \sum v_i^* / (s-1)^2 \quad (1.4-18)$$

where each v_i^* is calculated from Eq. 1.4-16, or by

$$\bar{v}[N] = \sum (N_i^* - \bar{N})^2 / (s-1)(s-2) \quad (1.4-19)$$

Variance estimated by Eq. 1.4-18 is approximately unbiased if and only if each v_i^* is almost unbiased, and variance estimated by Eq. 1.4-19 is almost unbiased when the N_i^* have the same mean. When these conditions for unbiasedness are not satisfied, both estimates are conservative because they tend to overestimate the true variance (1.4-5). Confidence intervals (95%) may be calculated using

$$\bar{N} + 1.96 (v^*)^{1/2} \quad (1.4-20)$$

or by

$$\bar{N} + 1.96 (v[\bar{N}])^{1/2} \quad (1.4-21)$$

In any experiment some attempt should be made to determine whether the underlying assumptions have been fulfilled. Seber (1.4-5) suggests that one way to accomplish such an evaluation in a multiple mark-recapture experiment is to obtain several estimates using the models outlined above. General agreement between such estimates and their confidence intervals would indicate that the assumptions underlying the various models are approximately satisfied.

Data from a multiple census taken from Ricker (1.4-6) after Gerking (1.4-21) are shown in Table 1.4-3.

With Schnabel's original (1.4-18) Eq. 1.4-8

$$N' = \left(\sum_{i=2}^s n_i M_i \right) / \left(\sum_{i=2}^s m_i \right) = \lambda / m = 10740/24 = 448$$

Using the formula for the normal approximation to the binomial (Eq. 1.4-11) because, although $m < 50$, M_i/N is greater than 0.1 for most samples so that m will not have a Poisson distribution, the 95% confidence interval for N/λ is:

Table 1.4-3

CAPTURE-RECAPTURE DATA FOR A POPULATION OF SUNFISH

i	n_i	m_i	m_i/n_i	M_i	$n_i M_i$	$m_i M_i$	$n_i M_i^2$	m_i^2/n_i
1	10	-	-	-	-	-	-	-
2	27	0	0	10	270	0	2 700	0
3	17	0	0	37	629	0	23 273	0
4	7	0	0	54	378	0	20 412	0
5	1	0	0	61	61	0	3 721	0
6	5	0	0	62	310	0	19 220	0
7	6	2	0.33	67	402	134	26 934	0.6667
8	15	1	0.07	71	1 065	71	75 615	0.0667
9	9	5	0.56	85	765	425	65 025	2.7778
10	18	5	0.28	89	1 602	445	142 578	1.3889
11	16*	4	0.25	102	1 632	408	166 464	1.0000
12	5	2	0.40	112	560	224	62 720	0.8000
13	7*	2	0.29	115	805	230	92 575	0.5714
14	19	<u>3</u>	0.16	119	<u>2 261</u>	<u>357</u>	<u>269 059</u>	<u>0.4737</u>
Total		24			10 740	2294	970 296	7.7452

*Number of deaths in this sample.

Source: Ricker 1975 after Gerking 1953 (1.4-21).

$$N/10740 = \frac{48 + 1.96^2 (1 - .2017) + 1.96 \sqrt{(1 - .2017) [96 + 1.96^2 (1 - .2017)]}}{1152}$$

$$= \frac{51.0667 + 17.4302}{1152}$$

$$= 0.0292 \text{ to } 0.0595$$

and the confidence interval for N is

$$0.0292\lambda \text{ to } 0.0595\lambda$$

or 314 to 639.

Using the modified Schnabel estimate (Eq. 1.4-9)

$$N'' = \lambda/m+1 = 10740/25 = 430$$

and the 95% confidence interval for N using N'' in Eq. 1.4-11

instead N' , as above, is

$$N/10740 = \frac{48 + 1.96^2 (1 - .2101) + 1.96 \sqrt{(1 - .2101) [96 + 1.96^2 (1 - .2101)]}}{1152}$$

$$= \frac{51.0345 + 17.3355}{1152}$$

$$= 0.0293 \text{ to } 0.0593$$

and the confidence interval for N is

$$0.0293\lambda \text{ to } 0.0593\lambda$$

or 315 to 637

The Schumacher - Eschmeyer estimate using Eq. 1.4-12 is

$$\tilde{N} = 970296/2294 = 423.$$

Variance is calculated by (Eq. 1.4-13) as

$$v = \frac{7.7452 - (2294^2/970296)}{12} = 0.1935$$

and the 95% confidence interval for N is given by Eq. 1.4-14 as

$$\frac{970296}{2294 + 2.179[.1935(970296)]^{1/2}}$$

or 300 to 719.

A plot of the proportion of returned marks in each sample (m_i/n_i) versus the number of fish previously marked (M_i) is shown in Figure 1.4-5. This plot indicates that marking may have affected initial catchability of the fish, since no marked fish appear in catches during the first five days of the 13-day sampling period represented. Seber (1.4-5) states that a good fit of the data to a straight line would lend credence to the assumption that marked and unmarked fish are equally vulnerable to sampling efforts throughout the recapture period (assumption c).

Using another method to help evaluate the population estimates, a mean Petersen estimate $\bar{N}$ was calculated by obtaining a daily estimate N_i^* and its associated variance v_i^* using Eq. 1.4-15 and Eq. 1.4-16, and an average for the period was calculated using Eq. 1.4-17. Variance of $\bar{N}$ was then calculated two different ways using Eq. 1.4-18 and Eq. 1.4-19 (Table 1.4-4). Confidence intervals were obtained using Eq. 1.4-20 and Eq. 1.4-21 as follows:

$$203 \pm 1.96 (1160.99)^{1/2}$$

136 to 270

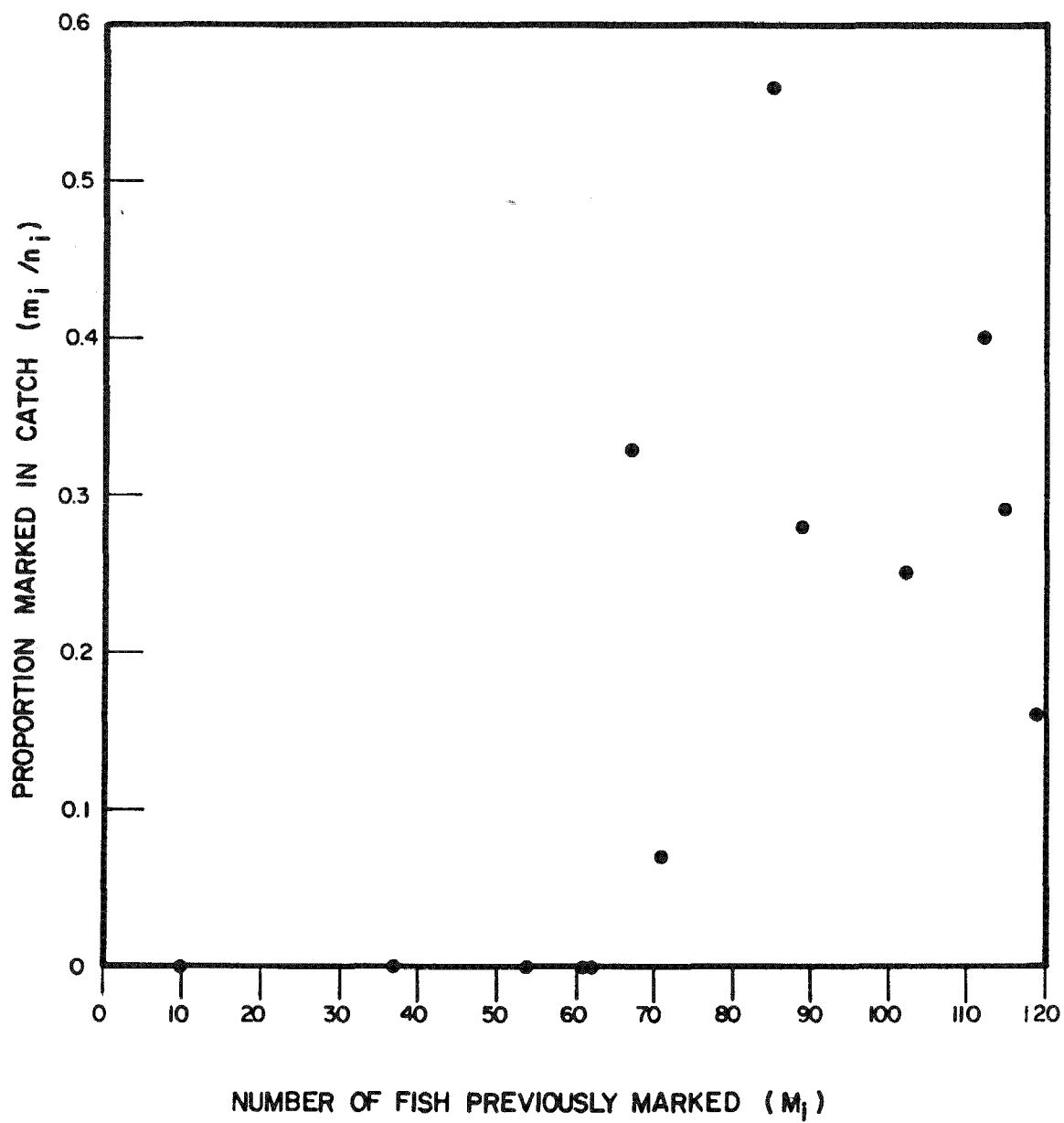


Figure 1.4-5. Schumacher -Eschmeyer Plot of Data on Table 1.4-3

Table 1.4-4
CALCULATION OF MEAN PETERSEN ESTIMATE AND ASSOCIATED
VARIANCE FOR DATA IN TABLE 1.4-3

i	N_i^*	V_i^*
1	NA	-
2	0	-
3	0	-
4	0	-
5	0	-
6	0	-
7	158	3437.78
8	575	94080.00
9	142	1092.06
10	284	7410.00
11	349	13727.84
12	225	6215.00
13	308	14564.44
14	599	55680.00

$$\begin{aligned}\Sigma N_i^* &= 2640 & \Sigma V_i^* &= 196207.12 \\ \bar{N} &= 2640/13 = 203 & & (1.4-17) \\ V^* &= 196207.12/169 = 1160.99 & & (1.4-18) \\ V[\bar{N}] &= 340332/156 = 2181.62 & & (1.4-19)\end{aligned}$$

NA - No marked fish at large.

and

$$203 \pm 1.96 (2181.62)^{1/2}$$

111 to 295.

Population estimates and their associated confidence intervals derived from the above four models are summarized below.

Type of Estimate	N	95% Range
Original Schnabel (1.4-8)	448	314-639 (1.4-11)
Modified Schnabel (1.4-9)	430	315-637 (1.4-11)
Schumacher-Eschmeyer (1.4-12)	423	300-719 (1.4-14)
Mean Petersen (1.4-17)	203	136-270 (1.4-20), 111-295 (1.4-21)

The low estimate provided by the mean Petersen method results from including the five zero values for N_i^* at the beginning of the experiment. This low estimate, in conjunction with the trend observed from Figure 1.4-5, suggests that the assumptions of equal catchability (c) and random mixing of marked fish into the population (g) were not fully satisfied during this phase of the mark-recapture study.

Ricker (1.4-6) states that marked sunfish, when first released, usually swim to the bottom and burrow into the weeds. Such behavior may make them less vulnerable to fixed gear such as traps and gill nets, and more vulnerable to moving gear such as seines or trawls. Assumptions (c) and (g) may have been better satisfied if seines and trawls, rather than traps, had been used as recapture gear in this experiment.

At any rate, the data indicate that the first portion of the experiment should not be used as part of the population estimate. The following are population estimates when the initial six days are treated as a release period only, and recapture effort begins on day 7 (therefore $s = 8$ rather than $s = 14$).

Type of Estimate	N	95% Range
Original Schnabel (1.4-8)	379	269-533 (1.4-11)
Modified Schnabel (1.4-9)	364	270-532 (1.4-11)
Schumacher-Eschmeyer (1.4-12)	393	250-914 (1.4-14)
Mean Petersen (1.4-17)	377	253-501 (1.4-20) 232-522 (1.4-21)

The Schnabel and Schumacher estimates are 7 to 15% lower when the data are treated in this fashion, but the estimates are probably more realistic, as evidenced by the compatibility of the mean Petersen estimate.

1.4.1.3 Open Populations: Multiple Marking

1.4.1.3.1 Jolly-Seber Stochastic Model. The previous section outlines methods applicable to closed populations only, i.e., populations which remain unchanged during the period of investigation. This section presents an extension of the Schnabel method to deal with open populations changing during the investigation because of the effects of dilution (recruitment, immigration) and/or permanent loss (death or emigration).

Again a sequence of s samples of sizes $n_1, n_2, \dots, n_s$ will be considered, but with the added generalization that, for each sample, only R_i of the n_i are marked and returned to the population. This more general model allows for accidental deaths caused by marking and handling, and also includes the case where some of the R_i are zero, as in commercial exploitation where the sample is permanently removed from the population. More detailed treatment of this model than presented here may be found in Jolly (1.4-22) and Seber (1.4-23), as well as in Cormack (1.4-24) and Seber (1.4-5).

Assumptions of the method are as follows (1.4-5):

- a. Every fish in the population, whether marked or unmarked, has the same probability p_i of being caught in the i th sample, given that it is alive and in the population when the sample is taken.
- b. Every marked animal has the same probability ϕ_i of surviving from the i th to $(i+1)$ th sample and of being in the population at the time of the i th sample, given that it is alive and in the population immediately after the i th release ($i=1, 2, \dots, s-1$).
- c. Every animal caught in the i th sample has the same probability, v_i , of being returned to the population: in many experiments $1-v_i$ can be regarded as the probability of accidental death through handling, etc.

- d. Marked animals do not lose their marks and all marks are reported on recovery.
- e. All samples are instantaneous, i.e., sampling time is negligible.

Jolly (1.4-22) states that the population amenable to description by this model is the "single" population which is a population covering an area within whose boundaries the individual fish are free to move and mix with others of their kind, but which is regarded as a single area for which parameters are to be estimated. Excluded by this definition is the situation where the population is split into a number of defined areas (subpopulations), and separate population estimates are required for each area as well as for numbers of fish moving from one area to another. The single population, however, need not be homogeneous, but may consist of different classes (age, sex, etc.) of fish behaving in different ways.

It is evident here, as it was for the closed population models described previously, that knowledge of the biology and behavior of the fish population, in context with the physical aspects of the water body serving as its habitat, must be available as a prerequisite to any successful mark-recapture study. This fact becomes more important with increasing size of the water body and intractability of the population numbers.

The following notation generally follows Jolly (1.4-22) and Seber (1.4-5):

s = number of samples. The range of i is therefore $1 \leq i \leq s$.

N_i = total number in the population when the i th sample is captured. This moment in time will be referred to as time i .

n_i = number captured in the i th sample.

m_i = number of marked fish caught in the i th sample.

r_i = number released from the i th sample after marking.

ϕ_i = probability that a fish alive at the moment of release of the i th sample will survive until the time of capture of the $i+1$ sample (emigration and death are synonymous

for this purpose). The captivity period is assumed to be very short compared with the interval between successive sampling (assumption e).

B_i = number of new fish joining the population in the interval between the i and $i+1$ th samples and alive at time $i+1$.
Definition: $B_0 = N_1$.

Z_i = the number of fish marked before time i which are not caught in the i th sample but are caught subsequently.

R_i = the number of the r_i animals released from the i th sample that are caught subsequently.

N_{ij} = total number in population at time i last captured in j th sample ($1 \leq j \leq i-1$).

n_{ij} = number in i th sample last captured in the j th sample ($1 \leq j \leq i-1$).

a_{ij} = number in the i th sample last caught in the j th sample or before.

Jolly (1.4-22) points out that the capture - recapture problem is solved if two prerequisites for each relevant point in time can be estimated: the proportion, α_i ; and the actual number, M_i , of marked fish in the population. The first of these is readily estimated by finding the proportion of marked fish in a random sample of the population. Therefore,

$$\hat{\alpha}_i = \frac{\hat{M}_i}{N_i} = \frac{m_i}{n_i} \quad (i = 2, 3, \dots, s) \quad (1.4-22)$$

The number of marked fish can be thought of as consisting of the observed number captured, m_i , plus the number not captured, $M_i - m_i$. Immediately after time i there are two groups of marked fish -- the $M_i - m_i$ and the r_i fish just released. Of the former, Z_i are subsequently caught; of the latter R_i are subsequently caught. Therefore, since the chances of recapture are assumed the same for both groups, the proportions $Z_i / (M_i - m_i)$ and R_i / r_i are expected to be approximately equal, and this equality gives

$$\hat{M}_i = \frac{r_i Z_i}{R_i} + m_i \quad (i = 2, 3, \dots, s-1) \quad (1.4-23)$$

With M_i obtained from Eq. 1.4-23 and $\alpha_i = m_i/n_i$ from Eq. 1.4-22, the parameters of ultimate interest, N_i , ϕ_i and B_i are now readily estimated from

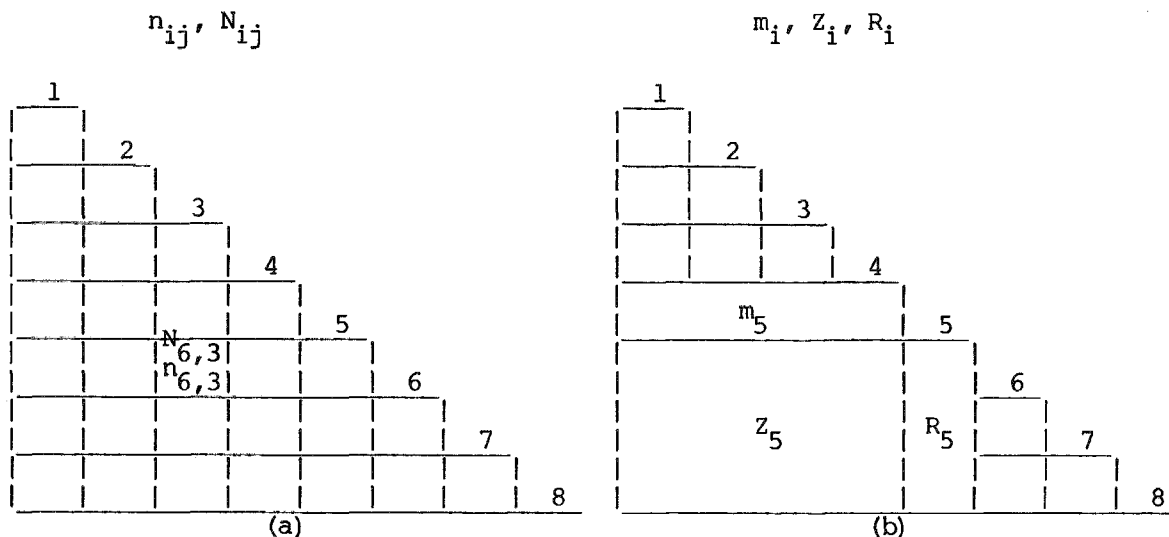
$$\hat{N}_i = \frac{\hat{M}_i}{\hat{\alpha}_i} \quad (i=2,3,\dots,s-1) \quad (1.4-24)$$

$$\hat{\phi}_i = \frac{\hat{M}_{i+1}}{\hat{M}_i - m_i + r_i} \quad (i=1,2,3,\dots,s-2) \quad (1.4-25)$$

$$\hat{B}_i = \hat{N}_{i+1} - \hat{\phi}_i (\hat{N}_i - n_i + r_i) \quad (i=2,3,\dots,s-2) \quad (1.4-26)$$

This method provides direct estimates of N for all sampling times except the first and the last, and ϕ and B are estimated for all time intervals except the first and the last. Thus a minimum of four operation times is required, of which the first involves marking only, the last recovery only, and the others, both marking and recovery.

To retain a clear picture of what is happening to the various marked fish throughout the chain of samples, Jolly (1.4-22) refers us to a table (a) identical to that shown below.



Here, the i th row corresponds to the sample in which n_i fish are captured and r_i released, the n_i subdivided into columns according to the date of their last previous capture. Thus, n_{ij} is entered in row i and column j . Associated with each n_{ij} is the corresponding population number, N_{ij} . Totalling the numbers over the areas indicated in (b) gives the m_i , Z_i and R_i needed in Eq. 1.4-23.

The following example is taken from Jolly (1.4-22). The data consist of 13 successive samples, at alternate 3- and 4- day intervals, of an apple orchard population of female black-kneed capsid, in which dilution both by fresh emergences and immigration was occurring. The data are present in Table 1.4-5, arranged exactly as Table (a) with the addition of columns for n_i and r_i , and a row for R_i . The total of the seventh column, for example, is $R_7 = 108$. Values for m_7 and Z_7 can be obtained by totalling the numbers over the areas indicated in Table 1.4-5, as per example (b), but Jolly (1.4-22) presents a more convenient way of treating the data. The n_{ij} are summed from the left in each row to form the a_{ij} of Table 1.4-5. The total of the $i-1$ th column, omitting the top entry, gives Z_i .

As indicated by the enclosed figures in Table 1.4-5, Z_7 is therefore found by adding all but the top entry of column 6 to give 110. The top entry of column 6 is $m_7 = 112$. Then, from Eq. 1.4-23,

$$\hat{M}_i = \frac{243 \times 110}{108} + 112 = 359.50,$$

and similarly for the other M_i ($i=2,3,\dots,12$). The values for M_i are shown in Table 1.4-5 alongside values for $\alpha_i = m_i/n_i$ from Eq. 1.4-22.

The next six columns of Table 1.4-5 are the estimates N_i , ϕ , and B_i computed from Eq. 1.4-24, 1.4-25 and 1.4-26, followed by their standard errors $[V(\hat{N}_i)]^{1/2}$, $[V(\hat{\phi}_i)]^{1/2}$ and $[V(\hat{B}_i)]^{1/2}$ computed from the following:

$$V(\hat{N}_i) = N_i (N_i - n_i) \left[\frac{M_i - m_i + r_i}{m_i} \left(\frac{1}{R_i} - \frac{1}{r_i} \right) + \frac{1 - \alpha_i}{m_i} \right] + N_i - \sum_{j=0}^{i-1} \frac{N_i^2(j)}{B_j} \quad (1.4-27)$$

Table 1.4-5
BLACK-KNEED CAPSID DATA

Values of n_{ij}

i	n_i	r_i														
1	54	54	1													
2	146	143	10	2												
3	169	164	3	34	3											
4	209	202	5	18	33	4										
5	220	214	2	8	13	30	5									
6	209	207	2	4	8	20	43	6	$m_7=112$							
7	250	$r_7=243$	1	5	5	10	34	56	7							
8	176	175	0	4	0	3	14	19	46	8						
9	172	169	0	2	4	2	11	12	28	51	9					
10	127	126	0	0	1	2	3	5	17	22	34	10				
11	123	120	1	2	3	1	0	4	8	12	16	30	11			
12	120	120	0	1	3	1	1	2	7	4	11	16	26	12	13	
13	142		0	1	0	2	3	3	2	10	9	12	18	35		
$R_i =$			80	70	71	109	101	108	99	70	58	44	35			
			$z_7=110$			R_7										

Values of a_{ij}

	values of s_{ij}												
1													
10	2												
3	37	3											
5	23	56	4										
2	10	23	53	5									
2	6	14	34	77	6								
1	7	12	22	56	112	7							
0	4	4	7	21	40	86	8						
0	2	6	8	19	31	59	110	9					
0	0	1	3	6	11	28	50	84	10				
1	3	6	7	7	11	19	31	47	77	11			
0	1	4	5	6	8	15	19	30	46	72	12		
0	1	1	3	6	9	11	21	30	42	60	95	13	
$z_{i+1} =$	14	57	71	89	121	110	132	121	107	88	60		

α_i	$\hat{M}_i$	$\hat{N}_i$	$\hat{\phi}_i$	$\hat{B}_i$	$[V(\hat{N}_i)]^{1/2}$	$[V(\hat{\phi}_i)]^{1/2}$	$[V(\hat{B}_i)]^{1/2}$	$[V(\hat{N}_i N_i)]^{1/2}$
-	0	-	0.649	-	-	0.114	-	-
0.0685	35.02	511.2	1.015	263.2	151.2	.110	179.2	150.8
.2189	170.54	779.1	0.867	291.8	129.3	.107	137.7	128.9
.2679	258.00	963.0	.564	406.4	140.9	.064	120.2	140.3
.2409	227.73	945.3	.836	96.9	125.5	.075	111.4	124.3
.3684	324.99	882.2	.790	107.0	96.1	.070	74.8	94.4
0.4480	359.50	802.5	0.651	135.7	74.8	0.056	55.6	72.4
.4886	319.33	653.6	.985	-13.8	61.7	.093	52.5	58.9
.6395	402.13	628.8	.686	49.0	61.9	.080	34.2	59.1
.6614	316.45	478.5	.884	84.1	51.8	.120	40.2	48.9
.6260	317.00	506.4	.771	74.5	65.8	.128	41.1	63.7
.6000	277.71	462.8	-	-	70.2	-	-	68.4

Source: Adapted from Jolly (1.4-22).

$$V(\hat{\theta}_i) = \theta_i^2 \left[\frac{(M_{i+1} - m_{i+1}) (M_{i+1} - m_{i+1} + r_{i+1})}{M_{i+1}^2} \left(\frac{1}{R_{i+1}} - \frac{1}{r_{i+1}} \right) + \frac{M_i - m_i}{M_i - m_i + r_i} \left(\frac{1}{R_i} - \frac{1}{r_{i+1}} \right) + \frac{1 - \theta_i}{M} \right] \quad (1.4-28)$$

$$V(\hat{B}_i) = \frac{B_i^2 (M_{i+1} - m_{i+1}) (M_{i+1} - m_{i+1} - r_{i+1})}{M_{i+1}^2} \left(\frac{1}{R_{i+1}} - \frac{1}{r_{i+1}} \right) + \frac{M_i - m_i}{M_i - m_i + r_i} \left[\frac{\theta_i r_i (1 - \alpha_i)^2}{\alpha_i} \right] \left(\frac{1}{R_i} - \frac{1}{r_i} \right) + \frac{(N_i - n_i) (N_{i+1} - B_i) (1 - \alpha_i) (1 - \theta_i)}{M_i - m_i + r_i} + N_{i+1} (N_{i+1} - n_{i+1}) \frac{1 - \alpha_{i+1}}{m_{i+1}} + \theta_i^2 N_i (N_i - n_i) \frac{1 - \alpha_i}{M_i} \quad (1.4-29)$$

The calculation of the variance is straightforward, though tedious if not computerized. For example:

$$\begin{aligned} V(\hat{N}_7) &= 802.5 (802.5 - 250) \left[\frac{359.5 - 112 + 243}{359.5} \left(\frac{1}{108} - \frac{1}{243} \right) + \frac{1 - 448}{112} \right] + N_i - \sum_{j=0}^6 \hat{N}_7^2(j) / \hat{B}_j \\ &= 443381.25 [(1.364) (.005) + .005] \\ &= 5240.8 + N_i - \sum_{j=0}^6 \hat{N}_7^2(j) / B_j \end{aligned}$$

Seber (1.4-5) suggests that inclusion of the quantity

$$N_i - \sum_{j=0}^{i-1} [N_i^2(j) / B_j]$$

does not change the result enough to warrant the extensive calculation effort and that, for practical purposes, $V(\hat{N}_7)$ can remain as 5240.8 and $\sqrt{[V(\hat{N}_7)]} = 72.4$. Technically speaking, however, 72.4 is actually $\sqrt{[V(\hat{N}_1|N_1)]}$, the standard error of N_1 produced by errors of estimation in N_1 . Jolly (1.4-22) states that the values of $[V(\hat{N}_1|N_1)]^{1/2}$ in Table 1.4-5 are only very slightly less than values for $[V(\hat{N}_1)]^{1/2}$, showing that the greater part of the variance in $\hat{N}_1$ is caused by errors in estimation, rather than deviation of the observed from the expected values of $\hat{N}_1$.

1.4.2 Catch-Per-Effort Methods

In a recent international compendium of information on fish population dynamics, Rothschild (1.4-25) states that the most widely practiced contemporary method of estimating the abundance of an exploited stock of fish is to use the catch-per-(unit)-effort* as an index of abundance. Although Gulland (1.4-26) admits that the catch-per-unit-effort will rarely be exactly proportional to the stock density, it is often essential to have some measure of the stock, and the catch-per-unit-effort will nearly always be the best available - better than, for example, total catch.

In the simplest situation, when a single homogeneous population is being fished, and when effort is proportional to rate of fishing, catch-per-unit-effort is proportional to the mean stock present during the time fishing takes place (1.4-6). However, the real situation is usually not this simple and, indeed, most fish populations are heterogeneous in space, and fishing effort is often not proportional to fishing mortality, i.e., the number of fish removed from the population.

This is especially true for commercial fisheries where fishing effort tends to follow fish population density in both space and time, and where increasing efficiency for a given unit of effort through time tends to mask declines in stock abundance as measured by catch-per-unit-effort. The best way to avoid these very prevalent sources of bias in commercial data is to institute a research survey

*The term catch-per-effort is specific to those gears where one effort is equivalent to another effort. An example would be a 100-foot seine which samples the same areas for each haul. Catch-per-unit-effort applies to those fishing gears such as trap nets, gill nets and trawls whose catch varies in proportion to the time sampled. Each catch is adjusted by the number of "units" of effort, i.e., commercial trawl catches are often adjusted to catch per hour of fishing.

program utilizing a stratified random sampling design** covering the entire range of the population in question. In addition, a thorough study of the sampling characteristics of the fishing gear should be conducted, and the same "standardized" gear should be used throughout the study.

Sampling characteristics of fishing gear include such species-specific variables as size selectivity, age selectivity, efficiency (number caught/number present), distribution of catch-per-tow or net set (Poisson, negative binomial), relationship between catch and length of tow or net set, diurnal and seasonal variations in catching efficiency, etc. Such data are necessary to permit investigators to make adjustments in the sampling regime and to work with comparative effort units over the years.

A good example of such a program is the bottom trawl survey conducted by the National Marine Fisheries Service at Woods Hole, Mass., since 1963. This survey, performed in the fall, spring and summer, covers more than 194,250 square kilometers from Cape Hatteras to Nova Scotia, i.e., the entire range of the target fish species. One important goal of this survey is to provide standardized abundance indexes free of bias inherent in commercial fishery statistics for several fish species. Control of such bias includes the use of standardized trawl gear and fishing methods on calibrated fishing vessels, an unbiased method of station selection, and complete records of the weight and length frequency of all fish species in each trawl catch (1.4-27). This example emphasizes the fact that standardized survey techniques had to be instituted to provide less biased data for determining age structure, recruitment, growth and mortality rates, and changes in stock abundance (1.4-28), although abundant commercial catch/effort data exist for all these exploited species.

The fishing gear most frequently used for estimates of both relative and absolute abundance of standing stock in sizeable water bodies is the otter trawl, so called because of the pressure boards used to keep the net mouth flared open while under tow. The trawl is an active (moving) sampler not relying on the activity or movement of the fish for its catch as do gill and trap nets. Trawls are versatile samplers, capable of working the bottom in shallow or deep water; they are not restricted to the near-shore zone as is a haul seine. In addition, trawls can be

**An unstratified or totally random design would assume no knowledge of the vertical or horizontal strata inhabited by the target fish species and would generally be too inefficient for the final design of a survey program.

modified to fish at any depth of the water column. Unlike haul seines, purse seines, gill nets and trap nets, trawls can be fished efficiently in the presence of substantial currents, permitting increased flexibility in the sampling design. Depending on the size of the trawl and the towing vessel, rough bottom topography can present problems for actual bottom fishing, but trawls can be outfitted with rollers to minimize this problem (1.4-28).

The selection of the sampling method used to provide relative or absolute population estimates depends to some extent on the physical characteristics of the water body and the behavior of the species and age groups of interest. For example, abundance estimates for young-of-the-year (age 0) fish which prefer the shoreline habitat of a lake or estuary might be representatively sampled with a haul seine. However, as the size of the water body increases and the ratio of shoreline habitat to total water volume decreases, the assumption of representative sampling with seines becomes more tenuous. Indeed, if the shoreline is the optimum habitat, it may tend to become saturated to a certain density level each year (assuming recruitment is always great enough to saturate this relatively small area), and an abundance index based on seine catch per effort may show very little fluctuation from year to year. If the excess fish from a strong year class reside in the less preferred but adequate offshore habitat, a trawl index of year-class strength may be more informative than a seine index. If an absolute population estimate rather than a catch-per-effort (relative abundance) index is needed, then both seine and trawl data would be necessary in this particular case. One point is very clear, however: active gear such as seines and trawls, whose sample space can be estimated on an areal or volumetric basis, are a necessity for realistic population estimates. This precludes sampling with passive fishing gear such as gill nets and trap nets when an estimate of absolute abundance based on catch-per-effort is required, although indexes of relative abundance are commonly calculated from stationary fishing gear.

1.4.2.1 Relative Abundance. Relative abundance, generally displayed as the number of fish caught per haul or net set or unit time, i.e., the catch-per-unit-effort (C/f), is one of the more useful indicators of the status of a fish stock. Calculated on an annual basis for several years, standardized C/f data collected from the entire range of the population reflect actual stock abundance trends, assuming that the average annual effect of a unit of gear does not systematically change from year to year because of changes in behavior, distribution, and environmental variables. Collected in conjunction with length and/or age frequency analysis of the catch, C/f data can also be used to estimate year-class strength (recruitment),

mortality between age groups (catch-curve analysis), and age-specific growth rates. These latter estimates are influenced by the degree of size-selectivity of the collection methods, and Ricker (1.4-6) points out that trawling with a fine-mesh liner in the cod end is one of the least size-selective sampling methods.

When calculating mean C/f values, care should be taken to ensure that each catch receives equal weight. When fishing effort is the same for each sample, an un-weighted mean can be obtained by dividing the total catch for a time period by the total effort during that period, i.e.:

$$\frac{\sum C_t}{\sum f_t}$$

However, when effort varies among samples this method gives undue weight to catches obtained from the greatest effort. This may be acceptable if the distribution of effort were completely random with respect to the distribution of the fish, but this is often not the case, especially for commercial fisheries where the greatest effort may be expended when fish density is high. In research or survey sampling an opposite trend may appear, i.e., the research crew may shorten the trawl or net-set time during periods of high fish density. In order to avoid possible bias resulting from an association between the level of effort and the level of fish density, mean C/f values should be calculated by

$$\overline{C/f}_t = \frac{(C_1/f_1) + (C_2/f_2) + \dots + (C_n/f_n)}{n} \quad (1.4-30)$$

where

$\overline{C/f}_t$ = average catch-per-unit-effort for time period t.

C_n/f_n = the nth catch-per-unit-effort value.

n = number of samples collected during time period t.

The choice of a time period to be represented by t depends on the frequency of sampling and the expected frequency of change in fish distribution. For most sampling programs of this nature, t is conveniently represented by one month, and the mean annual C/f for, say, 1978 is therefore represented by

$$\overline{C/f}_{1978} = \frac{\sum \overline{C/f}_t}{m} \quad (1.4-31)$$

where m = the number of months for which average C/f values occur. The months used for annual comparisons should, of course, be common to all years.

Gulland (1.4-26) addresses the above considerations when he states that the simple annual C/f value, i.e., the total catch divided by the total recorded effort, is not suitable because its relation to the mean annual abundance varies with the seasonal (monthly) distribution of fishing - if relatively more fishing in one year occurs during the best times of the year, the catch and the C/f will tend to be higher for that year. Therefore the index of abundance for that year may be artificially inflated because of the distribution of fishing effort, and it may bear no relation to stock density.

The previous discussion of calculation of mean C/f values for use as comparable stock abundance indexes between years must be modified and interpreted in light of definitions of time strata and fulfillment of sampling design goals specific to each program. In general, however, the method of dividing the total catch for a time period by the total effort for that period is unsatisfactory.

Once time strata have been defined, the next step is to stratify the water body into logical strata, some set of which encompasses the population of interest. The logic here involves dividing a heterogeneous population into subpopulations which may be homogeneous. This division is based on knowledge of fish behavior (derived from exploratory fishing and hydroacoustic studies) and natural stratification within the water body, such as depth contours, substrate type, thermoclines, etc. One advantage of stratification is that if each stratum is homogeneous, with population measurements varying little from one sampling unit to another within a stratum, a precise estimate of any stratum mean can be obtained from a small sample in that stratum. These estimates can then be combined into a precise estimate for the whole population (1.4-29).

The relative abundance index for a stratified design is the stratified mean C/f for the set of strata which encompasses the population. The formulae for the stratified mean and its variance are:

$$\overline{C/f}_{st} = \frac{\sum_{i=1}^{\infty} (A_i \overline{C/f}_i)}{A_T} \quad (1.4-32)$$

and

$$v(\overline{C/f}_{st}) = \frac{\sum_{i=1}^{\infty} (A_i^2 S_i^2 / n_i)}{A_T^2} \quad (1.4-33)$$

where A_i = area (or volume) of the i^{th} stratum.

A_T = total area (or volume) of all strata sampled.

$\overline{C/f}_i$ = sample mean catch-per-unit-effort in the i^{th} stratum.

n_i = number of tows in the i^{th} stratum.

S_i^2 = sample variance of catch per unit effort in the i^{th} stratum.

The estimator $\overline{C/f}_{st}$ will be unbiased as long as the stratum means ($\overline{C/f}_i$) are unbiased. Furthermore, variance of the stratified mean will be smaller as compared with simple random sampling, given that abundance is sufficiently different among different strata.

Other than the initial choice of the stratification, which is based mainly on biological considerations, the major design factors to be considered are the sampling strategy within a stratum (e.g. random or systematic), the size and fishing characteristics of the sampling gear, and the duration of the trawl hauls or net sets. Pennington and Grosslein (1.4-27) present an informative discussion on how the distributional properties of the C/f data can be used to maximize sampling efficiency for a large-scale bottom trawl survey. They also discuss the levels of accuracy and

precision for the C/f index of abundance and how these levels can be improved without necessitating the prohibitive expense of increasing the sample size (number of tows).

Although important in the assessment of the distribution, movement and trends in stock abundance through time, estimates of relative abundance do not in themselves permit calculation of direct impact from specific causes such as power plant operation. What are needed for the calculation of cropping impact by power plants are estimates of the absolute abundance of fish in the source water body (as well as estimates of absolute numbers of fish entrained and/or impinged).

1.4.2.2 Absolute Abundance. The method of converting relative abundance to absolute abundance is founded on the basic assumption that C/f is a function of stock density in the area being surveyed and that changes in C/f are directly proportional to changes in density (1.4-30, 1.4-31). In addition to this basic assumption, the method requires that the effective sampling space of the fishing gear be estimated in terms of area or volume and that certain assumptions be made as to the efficiency of the gear.

The effective sampling space of stationary fishing gear such as gill nets is difficult to assess because it requires knowledge of behavioral variables such as fish swim speed, average travel distance per unit time, directional vectors, etc.; therefore, absolute abundance estimates from stationary gear will not be considered further here. Because otter trawls appear to be used more frequently than any other gear-type for standing crop estimates, procedures described here will refer mainly to trawl estimates.

The first step is to define the effective sampling space covered by the gear in an average effort. Sampling space for the trawl is defined as the bottom area or static volume covered by the trawl in a single tow. Thus, it is necessary to know 1) net mouth width under tow (for bottom area calculations) or net mouth area under tow (for volume calculations), 2) trawl speed over the bottom, and 3) duration of tow. Trawl mouth dimensions can best be determined by direct underwater observation and photography while under tow. This was done for the LMS 30-foot otter trawl, for example, and the net mouth width was estimated as 4.72 m and the net mouth area as 2.14 m^2 (headrope height 0.46 m). The trawl speed over the bottom was estimated to

be 54.87 m/min and the average tow duration was 10 minutes. Therefore, the per effort sampling space of the LMS trawl was estimated as:

$$\text{Ground area} = 4.72 \text{ m} \times 54.87 \text{ m/min} \times 10 \text{ min} = 2589.86 \text{ m}^2$$

$$\text{Static volume} = 2.14 \text{ m}^2 \times 54.87 \text{ m/min} \times 10 \text{ min} = 1174.22 \text{ m}^3$$

The term static volume is used here to describe the water volume "carved out" by the trawl, as opposed to actual flow volume moving through the net as it is towed against the prevailing current.

This differentiation is based on the assumption that adult and juvenile fish do not move passively with the current as do planktonic larvae, whose concentrations are calculated using metered flow measurements through the net. In a water body with no prevailing currents, static volume would be synonymous with flow volume.

The next step in converting C/f data into population estimates is to determine the total bottom area or volume in each stratum. The ratio of total area or volume in the stratum to the area or volume sampled by each trawl effort is then used to scale up the mean C/f value for that stratum, i.e.,

$$\overline{P}_i = \overline{C/f}_i (V_i/\overline{v}) \quad (1.4-34)$$

where $\overline{P}_i$ = average standing stock in the ith stratum.

$\overline{C/f}_i$ = average catch per unit effort in the ith stratum.

V_i = total area or volume in the ith stratum.

$\overline{v}$ = average area or volume sampled by each trawl effort.

The total standing stock estimate for the entire region will be the sum of the average standing stock values ($\overline{P}_i$) for the component strata.

This type of population estimate usually underestimates the true population because it does not account for 1) fish residing outside the sampling space of the trawl,

e.g., for a bottom trawl, those fish residing off the bottom and above the vertical extent of the net mouth opening, and 2) gear efficiency less than 100% for those fish residing in the path of the trawl. Alverson and Pereyra (1.4-32) recommend adjusting for such potential underestimates by using a coefficient of vulnerability (c). This coefficient of vulnerability is the product of two coefficients that express 1) vulnerability of those fish that actually come within the influence of the trawl (c_e), commonly known as gear efficiency, and 2) the proportion of the total fish in the column of water above the bottom area swept by the trawl which would come within the trawl's influence (c_a), i.e., an availability term.

Therefore, by utilizing information on gear efficiency from controlled experiments and studying vertical distribution of the fish using hydroacoustic techniques, (Section 1.4.6) estimates of c_e and c_a can be obtained. Eq. 1.4-34 should then be modified to incorporate the efficiency and availability factors as follows:

$$\overline{P}_i = \overline{C/f}_i (V/\overline{v}c) \quad (1.4-35)$$

where $c = c_e c_a$

It is feasible that for some species, such as hogchokers (Trinectes maculatus) and flatfish (Pleuronectiformes) in general, the value for c could be greater than 1.0. For these demersal species c_a is close to 1.0 and c_e may be greater than 1.0, because the trawl doors and their bridles tend to herd these fish into the mouth of the trawl from an area wider than the net mouth itself (1.4-32). However, a value of $c > 1.0$ is very unlikely for any species other than a strict bottom dweller.

A Hudson River study (1.4-33) was used as an example to calculate standing crop estimates from bottom trawl C/f data for white perch (Morone americana) in order to quantify impingement impact for this species. As stated previously, sample space for the trawl was estimated by recording the time for each tow, estimating the bottom velocity, and obtaining towed net dimensions by direct underwater observation and photography. The static volume sampled by an average effort of 10 minutes was $1.17 \times 10^3 \text{ m}^3$.

The strata for which the population was estimated consisted of the volume within the bottom 0.46 m (vertical extent of net mouth opening) of a 32 km segment in front of

each of two power plants, i.e., kilometers 48 through 80 for the Bowline Generating Station (located at kilometer 58) and kilometers 81 through 113 for the Roseton Generating Station (located at kilometer 106). These two strata accounted for about 35 to 45% of the 0.46 m volume segment for the entire, 241 km-long estuary. The volume in the Bowline stratum is approximately $2.69 \times 10^7 \text{ m}^3$ and for the Roseton stratum, $1.80 \times 10^7 \text{ m}^3$.

Recent studies have shown that, as a rule, estimates of fish abundance are biased downward when investigators assume a trawl efficiency of 100%. Loesch et al. (1.4-34) reported trawl efficiencies ranging from 6 to 25%, but it appears that this is an estimate of Alverson and Pereyra's (1.4-32) $c = c_e c_a$ rather than actual efficiency (c_e), since availability (c_a) was also factored into their sampling design (the water depth was apparently substantially greater than the vertical net mouth opening).

Kjelson and Colby (1.4-35) indicated a trawl efficiency of about 20 to 50% for a 6.1 m (20 ft) bottom trawl. This seems to be an actual estimate of c_e because the water depth was only slightly greater than the vertical span of the trawl and the authors make the assumption that the fish were available to the gear at all times ($c_a = 1.0$). Based on Kjelson and Colby (1.4-35) an approximation of $c_e = .30$ to $.50$ was made for the LMS trawl, i.e., the assumption is that the LMS trawl captures 30 to 50% of the white perch present in the path of the trawl.

The calculation of availability (c_a) was based on the assumption that white perch reside only in the bottom 3 m of the water column and that the species-specific vertical distribution of density is fairly uniform within this layer during each trawl sample. The definition of the availability coefficient is the proportion of total fish in the water column above the bottom area swept by the trawl which would come within the trawl's influence. Since the vertical net mouth-opening is 0.46 m, the availability coefficient is calculated as $c_a = 0.46/3 = 0.15$, i.e., 15% of the white perch residing in the bottom 3 m of the water column are available to the bottom trawl. Therefore, the range for $c = c_e c_a$ is from 0.045 to 0.075, and these values were used in Eq. 1.4-35.

The two major assumptions underlying this method of estimating standing crop are:

The estimated standing crop within a 65 km river segment from km (kilometer) 48 to km 113 is used to represent the entire standing crop from km 0 to km 241, i.e., it is assumed that no fish reside north of km 43 or south of km 48.

The concentrations of fish existing within the vertical layer sampled by the trawl, i.e., from bottom to 0.46 m above the bottom, are the same as concentrations existing in the vertical layer from 0.46 to 3 m above the bottom.

The first assumption is conservative, i.e., tends to maximize cropping impact, because it is well known that substantial numbers of white perch are present above and below the delimited segment. If data of the same frequency as used in estimating the 65 km standing crop were available to account for the fish residing in the segments from km 0 to 48 and km 113 to 241, the total standing crop would necessarily increase, thus reducing the estimated impingement cropping rates.

The second assumption may be conservative in that it does not acknowledge any fish which reside in the water column between the surface and 3 m from the bottom. Benthic feeders such as white perch are generally oriented toward the bottom, but hydroacoustic surveys have shown that the number of white perch found in the upper water column is certainly not zero, and that during the winter, white perch are distributed at intermediate depths well off the bottom. However, further investigations should be conducted to verify the assumption of uniform vertical distribution between 0 and 3 m from the bottom. Such investigation would permit a more refined estimate of the availability coefficient, c_a .

Standing crop data based on two different vulnerability coefficients ($c = .045, .075$) are presented separately for each stratum in Table 1.4-6. Total estimated standing crops summed for both strata and associated cropping rates represented by multiplant impingement are shown in Table 1.4-7. The range of 4.2 to 7.0% for the estimated annual cropping rate is, in this example, a direct result of the range chosen for bottom trawl efficiency ($c_e = 30$ to 50%).

1.4.2.3 Fishing Success Approach. This is a catch-per-effort method which is applicable only when a population is fished until enough fish are removed to significantly reduce the C/f , which is considered to be proportional to the stock present. Also, since it is assumed that the size of a sample caught from a population is proportional to the effort put into taking the sample, one unit of sampling effort is assumed to catch a fixed proportion of the population. Therefore, if samples are not replaced, the decline in population size will produce a decline in C/f (1.4-5, 1.4-6). For example, if removal of 1,000,000 fish reduces the C/f by a quarter, the original stock is estimated as $1,000,000/0.25$ or 4,000,000 fish.

Table 1.4-6

ESTIMATED STANDING CROP ESTIMATES FROM BOTTOM TRAWL CATCH
PER UNIT EFFORT DATA FOR HUDSON RIVER WHITE PERCH, 1976

I. BOWLINE STRATUM

Month	Monthly Mean C/f	$\bar{P}_{1B} = C/f(V/\sqrt{c_1})$	$\bar{P}_{2B} = C/f(V/\sqrt{c_2})$
Jan	No Sample	-	-
Feb	29.30	14,940,955	8,964,573
Mar	88.30	45,026,838	27,016,103
Apr	140.99	71,895,062	43,137,037
May	32.00	16,317,767	9,790,660
Jun	57.26	29,198,604	17,519,163
Jul	70.56	35,980,676	21,588,406
Aug	66.41	33,864,466	20,318,680
Sep	19.76	10,076,221	6,045,733
Oct	119.01	60,686,795	36,412,077
Nov	60.61	30,906,871	18,544,122
Dec	64.01	32,640,633	19,584,380

II. ROSETON STRATUM

Month	Monthly Mean C/f	$\bar{P}_{1R} = C/f(V/\sqrt{c_1})$	$\bar{P}_{2R} = C/f(V/\sqrt{c_2})$
Jan	No Sample	-	-
Feb	3.90	1,327,214	796,328
Mar	12.86	4,376,404	2,625,842
Apr	14.94	5,084,251	3,050,551
May	13.38	4,553,366	2,732,019
Jun	7.58	2,579,560	1,547,736
Jul	7.71	2,623,800	1,574,280
Aug	6.47	2,201,814	1,321,089
Sep	19.24	6,547,590	3,928,554
Oct	18.52	6,302,566	3,781,539
Nov	4.15	1,412,292	847,375
Dec	8.51	2,896,049	1,737,630

Constants

$$V_{\text{Bowline}} = 9.50 \times 10^8 \text{ ft}^3$$

$$V_{\text{Roseton}} = 6.34 \times 10^8 \text{ ft}^3$$

$$c_1 = 0.045 \text{ (} c_e = 30\% \text{)}$$

$$c_2 = 0.075 \text{ (} c_e = 50\% \text{)}$$

$$\bar{V} = 4.14 \times 10^4 \text{ ft}^3$$

Table 1.4-7

IMPINGEMENT CROPPING ESTIMATE FOR HUDSON RIVER
WHITE PERCH, 1976

Month	Total Estimated Standing Crop		Multi-Plant Impingement Estimate*	Cropping Estimate Range (%)
	$\bar{P}_{1B} + \bar{P}_{1R}$	$\bar{P}_{2B} + \bar{P}_{2R}$		
Jan	No Sample	No Sample	597,300	No Estimate
Feb	16,268,169	9,760,901	176,300	1.1 - 1.8
Mar	49,403,242	29,641,945	142,800	0.3 - 0.5
Apr	76,979,313	46,187,588	92,800	0.1 - 0.2
May	20,871,113	12,522,679	29,000	0.1 - 0.2
Jun	31,778,164	19,066,899	9,700	0.03 - 0.05
Jul	38,604,476	23,162,686	2,000	0.01 - 0.01
Aug	36,066,280	21,639,769	16,700	0.05 - 0.08
Sep	16,623,811	9,974,287	8,000	0.05 - 0.08
Oct	66,989,361	40,193,616	65,600	0.1 - 0.2
Nov	32,319,163	19,391,497	161,600	0.5 - 0.8
Dec	35,536,682	21,322,010	305,900	0.9 - 1.4
Annual Average	38,312,707	22,987,625		
Annual Total			1,607,700	4.2 - 7.0

*Impingement totals from 3 plants: Bowline, Indian Point, and Roseton.

Other assumptions required by this method are that the population be closed, i.e., unchanged by the effects of migration, mortality, and recruitment during the period of investigation, that the catchability coefficient, q , remain constant within each age or size stratum throughout the entire experiment, and that each unit of effort be independent, i.e., traps, nets or power plants do not compete with each other for fish. DeLury (1.4-2) suggests that the assumption of constant catchability may describe the situation fairly well if records of age or size are kept so that stratification of the experiment is possible. Also, if sampling is intensive enough so that the population is appreciably depleted within a short period of time, the assumptions of a closed population and equal catchability are more likely to be true.

Ricker (1.4-6) states that instead of estimating C/f only at the start and finish of the experiment, a series of estimates is usually made. That is, a number of points are used to determine the rate of decrease of C/f , and hence of the stock, since variables such as weather, which affect vulnerability (q), tend to make single estimates of C/f unreliable for this purpose.

The Leslie method involves plotting C/f against cumulative catch over a period of time (1.4-2, 1.4-36). From the resulting straight line, the initial population (x -axis intercept) and catchability (slope) can be estimated. The following notation is taken from Ricker (1.4-6):

$\hat{N}_0$ = original population size.

N_t = mean population surviving during time interval t .

C_t = catch taken during time interval t .

f_t = fishing effort during time interval t .

C_t/f_t = catch-per-unit-effort during the interval t .

$\overline{C/f}$ = the average of all C_t/f_t values during the entire period.

K_t = cumulative catch up to the start of interval t , plus half of that taken during interval t .

$\bar{K}$ = the average of all K_t values during entire period.

$\hat{q}$ = catchability, i.e., the fraction of the population taken by one unit of fishing effort.

E_t = cumulative fishing effort up to the start of interval t , plus half of that during interval t

For the general case, catch-per-unit-effort during time interval t is, by definition, equal to catchability multiplied by the mean population present during the interval, i.e.,

$$C_t/f_t = qN_t \quad (1.4-36)$$

The population at the time K_t fish have been caught is equal to the original population less K_t :

$$N_t = N_o - K_t \quad (1.4-37)$$

Therefore, from Eq. 1.4-36 and 1.4-37,

$$C_t/f_t = \hat{q} \hat{N}_o - \hat{q} K_t \quad (1.4-38)$$

Eq. 1.4-38 is in the form of a regression equation with a negative slope, q , denoting catchability, and a y-axis intercept which is the product of the original population, $\hat{N}_o$, and the catchability, $\hat{q}$. The x-axis intercept is an estimate of the original population, $\hat{N}_o$, since it represents the cumulative catch if C_t/f_t , and thus the population also, were reduced to zero by fishing.

Confidence limits for the estimate of $\hat{N}_o$ are given by the roots of the quadratic (1.4-2, 1.4-6):

$$\begin{aligned} N^2 \left[\hat{q}^2 - (t_{[\alpha, n-2]}^2 s_{y.x}^2 c_{22}) \right] - 2 \left[\hat{q}^2 \hat{N}_o - (t_{[\alpha, n-2]}^2 s_{y.x}^2 c_{12}) \right] N \\ + \hat{q}^2 \hat{N}_o^2 - (t_{[\alpha, n-2]}^2 s_{y.x}^2 c_{11}) = 0 \end{aligned} \quad (1.4-39)$$

where

$$c_{11} = \sum (K_t)^2 / n \sum (K_t - \bar{K})^2$$

$$c_{12} = \sum (K_t / n \sum (K_t - \bar{K})^2$$

$$c_{22} = 1 / \sum (K_t - \bar{K})^2$$

$t_{[\alpha, n-2]}$ = tabulated t-value at the $1-\alpha$ confidence level
with $n-2$ degrees of freedom.

n = the number of days fishing.

and the estimated variance about the regression is given by

$$s_{x,y}^2 = \frac{\sum y^2 - (\sum xy)^2 / \sum x^2}{n-2} \quad (1.4-40)$$

where

$$y = (C_t / f_t - \bar{C} / \bar{f}) \quad (1.4-41)$$

$$x = (K_t - \bar{K}) \quad (1.4-42)$$

A variation of the Leslie method for obtaining population estimates from the relation of fishing success to the catch already taken is DeLury's method of relating fishing success to the fishing effort already spent. Following the exposition of Ricker (1.4-6), Eq. 1.4-36 can be rewritten as:

$$C_t / f_t = q N_0 (N_t / N_0) \quad (1.4-43)$$

or $\log_e (C_t / f_t) = \log_e (q N_0) + \log_e (N_t / N_0) \quad (1.4-44)$

Where the fraction of the stock taken by a unit of effort is small, e.g. where $q \leq 0.02$, it can be used exponentially to show the fraction of stock remaining after E_t units of effort have been expended, as follows:

$$N_t / N_0 = e^{-q E_t} \quad (1.4-45)$$

Substituting the above in Eq. 1.4-44,

$$\log_e (C_t/f_t) = \log_e (qN_o) - qE_t \quad (1.4-46)$$

Converting to base-10 logarithms Eq. 1.4-46 becomes

$$\log_{10} (C_t/f_t) = \log_{10} (\hat{q}\hat{N}_o) - 0.4343qE_t \quad (1.4-47)$$

A plot of the base-10 logarithm of C/f during each time interval vs cumulative effort up to the middle of the interval, yields a line whose slope is $0.4343q$ and whose y-axis intercept is $\log qN_o$. From these two quantities q and N_o can be estimated.

The following example of the procedure outlined above is based on hypothetical impingement data (Table 1.4-8). January and February were chosen because hydro-acoustic surveys have shown that Hudson River white perch tend to concentrate in one relatively small segment of the river during the winter months. Impingement rates at three power plants located within this segment are much higher during these months because of high fish density and lowered activity level. Also, during this period there is no chance of recruitment and little possibility of immigration and emigration from the area. Natural mortality rates may be low during the winter in the Hudson River due to decreased predation, decreased metabolic rates, high oxygen levels, etc. Because the Hudson River white perch population seems to be the most stable during the winter months, investigators are most likely to satisfy the assumptions of the fishing success method during this season. This may not be true for other water bodies such as Lake Ontario, for example, where various species occasionally sustain large-scale winter die-offs.

The regression equation for C_t/f_t vs K_t values from Table 1.4-8 is:

$$C_t/f_t = \hat{q}\hat{N}_o - \hat{q}K_t = 51.03663 - 0.0000070 K_t$$

and this line is plotted through the data in Figure 1.4-6.

Therefore $q = 0.0000070$ and $N_o = 51.03663/0.0000070 = 7,290,947$, which is the estimated initial population.

Table 1.4-8
HYPOTHETICAL MULTI-PLANT IMPINGEMENT DATA

7-Day Week	C_t^a	f_t^b	E_t	$C_t/2$	K_t	C_t/f_t
1	1,000,000	21,500	10,750	500,000	500,000	46.5
2	950,000	22,000	32,500	475,000	1,475,000	43.2
3	600,000	21,000	54,000	300,000	2,250,000	28.6
4	750,000	20,000	74,500	375,000	2,925,000	37.5
5	550,000	19,500	94,250	275,000	3,575,000	28.2
6	500,000	23,000	115,500	250,000	4,100,000	21.7
7	420,000	22,500	138,250	210,000	4,350,000	18.7
8	350,000	20,000	159,500	175,000	4,735,000	17.5
Average		$\bar{f}=21,187.5$			$\bar{K}_t=2,988,750$	$\bar{C}_f=30.2375$

^a Number of fish impinged at three generating stations during each of eight weeks during January and February.

^b Total number of million-gallon units of flow which passed through all three plants during each week.

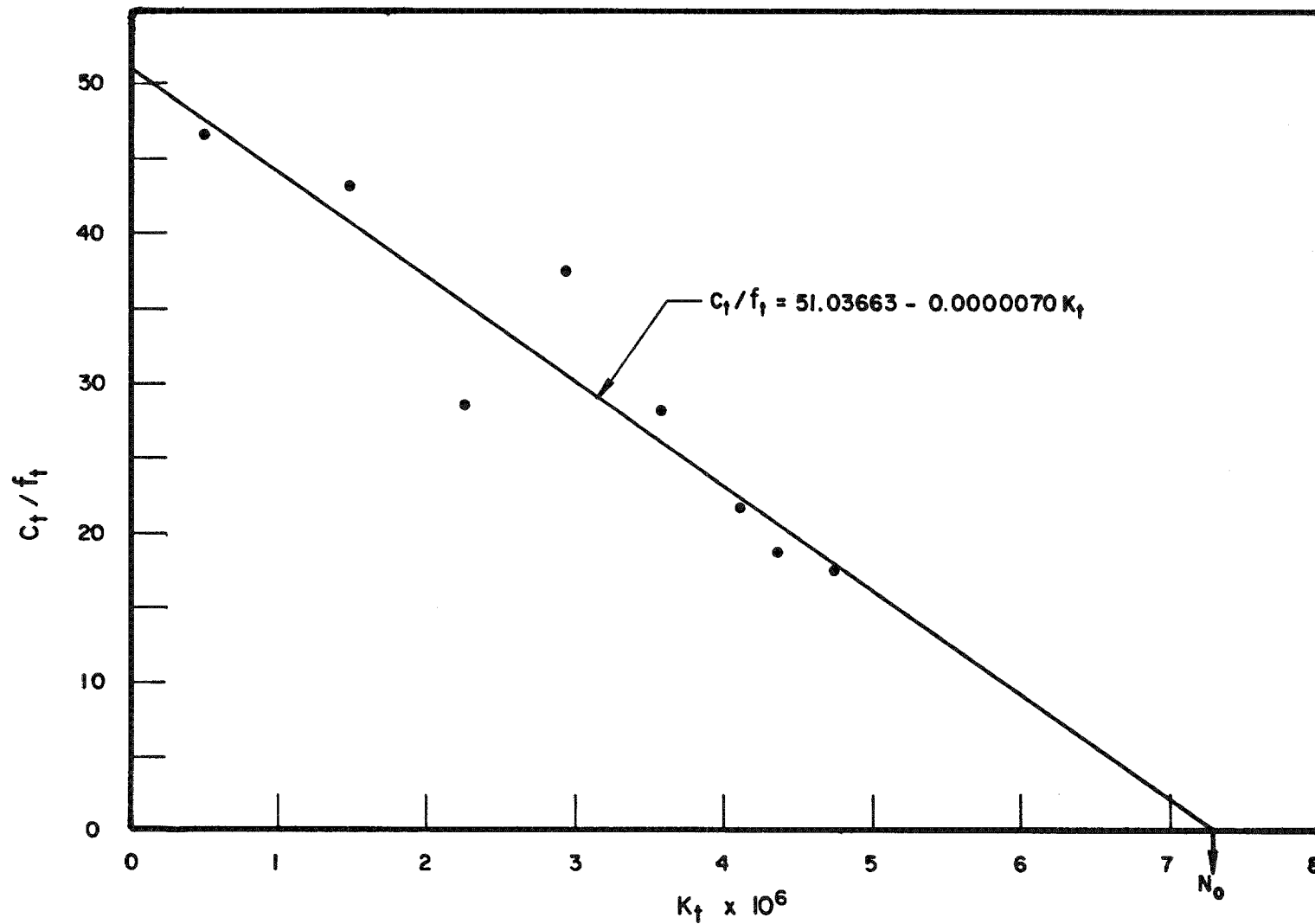


Figure 1.4-6. Catch Per Unit Effort (C_t/f_t) vs Cumulative Impingement Catch (K_t)

The interpretation of the catchability coefficient ($\hat{q}$) is that it is the instantaneous rate of removal of the fish, i.e., 0.000007 fish per million gallons of intake water or per unit of fishing effort. It is synonymous here with instantaneous power plant mortality (P) and total instantaneous mortality (Z), since natural mortality (M) has been assumed to be negligible during the period of the study. Because the average number of effort units per week during the experiment was 21,187.5 (Table 1.4-8), $Z = (21,187.5)(0.000007) = 0.1483$ per week. With the use of the relationship

$$A = 1 - e^{-Z} \quad (1.4-48)$$

which is equivalent to Eq. 5-7 in Section 5.5.2 of the methodology, the percentage removal of fish from the source population is 13.8% per week during this eight-week period.

The 95% confidence interval for $\hat{N}_0$ can be calculated using the following values in Eq. 1.4-39:

$$c_{11} = 2.2420225 \times 10^{13} / [(8)(1.5516088 \times 10^{13})] = 0.18062$$

$$c_{12} = 23,910,000 / [(8)(1.5516088 \times 10^{13})] = 1.926 \times 10^{-7}$$

$$c_{22} = 1 / 1.5516088 \times 10^{13} = 6.4449 \times 10^{-14}$$

$$s_{y.x}^2 = \frac{860.3188 - (1.1659383 \times 10^{16}) / 1.5516088 \times 10^{13}}{6} = 18.147$$

$$t_{[.05,6]} = 2.447$$

so that

$$\begin{aligned} & N^2 \{4.9 \times 10^{-11} - [(5.98781)(18.147)(6.4449 \times 10^{-14})]\} \\ & - 2 \{3.5725764 \times 10^{-4} - [(5.98781)(18.147)(1.926 \times 10^{-7})]\} N \\ & + \{2.6047375 \times 10^3 - [(5.98781)(18.147)(0.18062)]\} = 0 \end{aligned}$$

and

$$\begin{aligned}
N &= \frac{-b + \sqrt{b^2 - 4ac}}{2a} \\
&= \frac{0.000673 + 0.00000045 - 0.0000004343}{0.0000000001} \\
&= 5,475,681 \text{ to } 7,984,319
\end{aligned}$$

Therefore, the 95% confidence range for the estimate $\hat{N}_0 = 5.5 \times 10^6$ to 8.0×10^6 , and this range is not symmetrical with respect to the best estimate.

Using DeLury's method, the regression equation for $\log(C_t/f_t)$ vs E_t values from Table 1.4-8 is

$$\log(C_t/f_t) = \log(qN_0) - 0.4343qE_t = 1.70691 - 0.00000297 E_t$$

and this line is plotted through the data in Fig. 1.4-7. Thus $\hat{q} = 0.0000068$, $qN_0 = 10^{1.70691} = 50.92253$, and $\hat{N}_0 = 7,488,608$. As Ricker (1.4-6) points out, the use of ordinary regression with the DeLury method tends to underestimate q and overestimate N_0 , and the two estimates above differ in the expected direction from the Leslie estimates.

Impingement, like several other fishing methods, is generally selective for a particular age or size segment of the population. Therefore, the source population estimated by the above method represents only the impingeable population, and not the entire population for the species considered. For example, if the impinged fish in Table 1.4-8 were all yearling fish, then $\hat{N}_0$ would be an estimate of the yearling segment of the population only. Thus, the estimate of $\hat{N}_0$ for the entire population could be approximately doubled based on knowledge of the age structure for the Hudson River white perch population. It is helpful, then, to stratify catch data by age groups when using this method.

The information obtained from the catch-per-effort or fishing success method can be maximized if the experiment is conducted simultaneously with a mark-recapture study. The information from these complementary techniques can be used to evaluate the effects of immigration and emigration on catchability and on the initial population estimate. Once the fish are marked and released randomly into the population under investigation, and equal vulnerability of marked and unmarked fish is assumed, the

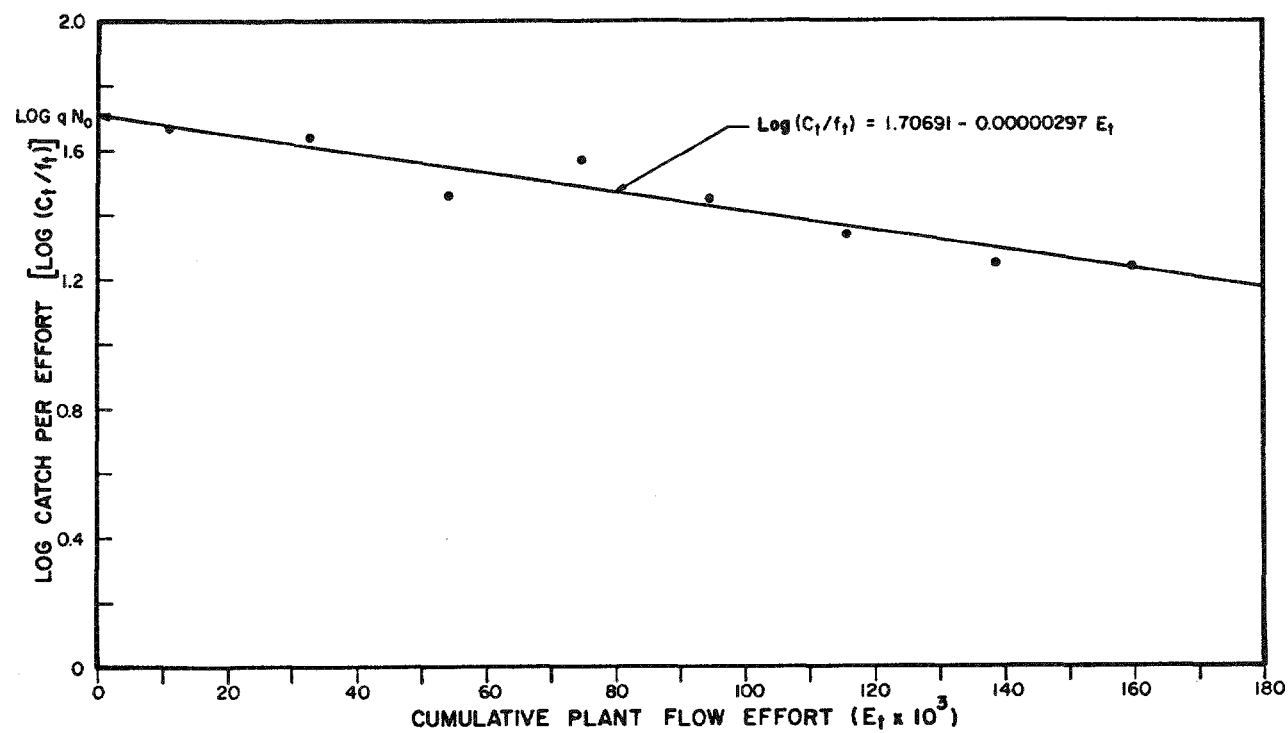


Figure 1.4-7. Log Catch Per Effort ($\log C_t/f_t$) vs Cumulative Plant Flow-Effort (E_t)

trend in catch-per-unit-effort of marked fish is related to the accumulated catch of marked fish in the same way as catch-per-unit-effort from the whole population is related to the cumulative catch from the whole population (1.4-2, 1.4-37). This can be expressed similarly to Eq. 1.4-38 as:

$$C'_t/f_t = q'N'_0 - q'K'_t$$

where

C'_t/f_t = the catch of marked fish per unit effort.

K'_t = the cumulative catch of marked fish.

q' = the apparent catchability of marked fish.

N'_0 = the estimated population of marked fish when $t=0$.

The advantage here is that the number of marked fish at large at the beginning of the experiment is a known value and can be compared with the estimate N'_0 to reflect the influence of those factors tending to remove fish from the population; namely, emigration and tagging mortality. Of course, this latter source of removal should be quantified prior to the beginning of the experiment so that its effects can be eliminated from the analysis.

With the methods devised by Ketchen (1.4-37) the true catchability (q) of the population can be calculated by the apparent catchability of the marked fish (q'). Ketchen (1.4-37) shows that the true catchability of the population is equal to the apparent catchability of the marked fish times the ratio of the estimated number to the actual number of marked fish, i.e.,

$$q = \frac{q'N'_0}{\text{number marked}} \quad (1.4-49)$$

For example, assume that 4000 white perch had been marked and released randomly into the population immediately prior to the beginning of the impingement experiment. The return of marked fish in impingement collections would provide estimates of q' and N'_0 by the same regression techniques used to calculate q and N_0 , namely

Eq. 1.4-38. Regression analysis performed on the hypothetical impingement return data in Table 1.4-9 yields:

$$C'_t/f_t = 0.020678 - 0.0000453K'_t$$

and the true catchability using Eq. 1.4-49 is

$$q = \frac{0.020678}{4000} = 0.0000052,$$

which is lower than the initial estimate of population catchability, $q = 0.000007$. This reflects the fact that the rate of emigration out of the study area was greater than the rate of immigration during the study, and the initial population estimate should be adjusted by

$$N_o = \frac{\hat{q} \hat{N}_o}{q} \quad (1.4-50)$$

The adjusted estimate is thus

$$N_o = \frac{51.03663}{0.0000052} = 9,814,737$$

or an increase of 34.6%. The weekly removal of 13.8% from the source population would now be recalculated as 10.4% Eq. 1.4-47.

1.4.3 Virtual Population Analysis

If fishing is the major cause of mortality for several age groups within a population, much information about the dynamics of the population can be obtained by making an age census of the catch and dividing the fish among their appropriate year classes. The sum of catches from a particular year-class or cohort, during the years it contributes to the fishery, is a minimum estimate of its abundance at the time it was just entering the catchable size range. Similarly, partial sums will

Table 1.4-9

HYPOTHETICAL IMPINGEMENT RETURNS OF MARKED FISH
FROM AN INITIAL POPULATION OF 4,000 FISH

7-Day Week	C'_t	F_t	$C'_t/2$	K'_t	C'_t/F_t
1	300	21500	150	150	0.01395
2	100	22000	50	350	0.00455
3	40	21000	20	420	0.00190
4	10	20000	5	445	0.00050
5	4	19500	2	452	0.00021
6	2	23000	1	455	0.00009
7	0	22500	0	456	0
8	0	20000	0	456	0

give the minimum number of fish from each year class in existence in any calendar year. The total of these sums in a particular year represents the minimum number of catchable fish present in that year (1.4-6). This quantity is known as the utilized stock (1.4-38) or virtual population (1.4-39), and it represents the population present at a given time, with the exception of all fish that have died of causes other than fishing (e.g. natural mortality). Any quantifiable sources of mortality which contribute significantly to the decay of a cohort, e.g., commercial and sport fishing plus power plant impingement on a multiplant basis should be included in the virtual population estimates in order to obtain the best estimates possible from this method.

The following definitions apply:

C_0, C_1, C_2 , etc. = impingement and/or commercial-sport catches in years 0 (base year or initial year of survey), 1, 2, etc.

x_0, x_1, x_2 , etc. = fractional representation of each age t in the catch of a given year, 1 January marking the change-over from one age to the succeeding age. Age 0 begins when young-of-the year fish are fully recruited into catches.

r = greatest age involved.

Ignoring fish which die naturally from age 0 onward, the total stock at the start of a year will be the sum of: that year's catch; the next year's catch diminished by the number of age 0 fish in it (because these were not hatched at the beginning of the base year); the next year's catch not including the age 0 and age 1 fish, and so on. The expression for population size (utilized stock) at the start of age 0 during any one year is:

$$N = C_0 + C_1 (1 - x_0) + C_2 (1 - x_0 - x_1) + \dots + C_r (1 - x_0 - x_1 - \dots - x_{r-1})$$

(1.4-51)

For example, according to the hypothetical impingement data in Table (1.4-10), the total utilized stock, i.e., all age groups present from different year classes in the impingement year 1971 (base year), is:

Table 1.4-10

TOTAL CATCH AND AGE OF THE CATCH OF IMPINGED WHITE PERCH

Collection Year	Total Multi-Plant Impingement Catch (C_t)	Catch By Age*						
		0	1	2	3	4	5	6
1970	1,250,000	300,000	750,000	150,000	30,000	19,000	1,000	0
1971	1,150,000	250,000	790,000	75,000	20,000	10,000	5,000	0
1972	2,350,000	500,000	1,000,000	550,000	<u>150,000</u>	100,000	50,000	0
1973	3,400,000	1,000,000	1,500,000	400,000	<u>300,000</u>	150,000	50,000	0
1974	2,500,000	800,000	1,200,000	250,000	150,000	75,000	25,000	0
1975	1,700,000	400,000	800,000	250,000	150,000	75,000	25,000	0
1976	3,100,000	1,000,000	1,500,000	500,000	70,000	20,000	10,000	0
			(1,077,143)	(310,714)	(124,286)	(64,143)	(23,714)	
				(310,714)	(124,286)	(64,143)	(23,714)	
					(124,286)	(64,143)	(23,714)	
						(64,143)	(23,714)	
						(64,143)	(23,714)	

*Values in parentheses are the average of previous entries, since catches for years later than 1976 would be needed to supply actual data.

$$\begin{aligned}
N_{1971} = & 1,150,000 + (2,350,000) (1 - .212766) + (3,400,000) (1 - .294118 - .441176) \\
& + (2,500,000) (1 - .3200 - .4800 - .1000) + (1,700,000) (1 - .235294 - .470588 \\
& - .147059 - .088235) + (3,100,000) (1 - .322581 - .483871 - .16129 - .022581 \\
& - .006452)
\end{aligned}$$

$$= 1,150,000 + 1,850,000 + 900,000 + 250,000 + 100,001 + 9999 = 4,260,000$$

An equivalent method for finding total utilized stock present in any year is to first arrange stocks by age and brood by summing the values in Table 1.4-10 diagonally from lower right to upper left, i.e., following the cohort by summing age 5 fish collected in 1976, age 4 fish in 1975, age 3 in 1974, etc. through to age 0 fish in 1971. These sums provide the minimum number of survivors of each brood at each age and are shown in Table 1.4-11. The total utilized stock of all ages for each year is then found by summing entries of Table 1.4-11 diagonally from lower left to upper right. Thus, the total utilized stock for 1971 is $1,885,000 + 1,740,000 + 400,000 + 170,000 + 60,000 + 5000 = 4,260,000$, exactly the same value calculated from Eq. 1.4-51. This means that at the start of 1971 there were at least 4,260,000 fish of age 0 and older in the designated water body.

This analysis provides a minimum estimate of population size, V , and an estimate of the biostatistical rate of exploitation, C/V or $u(\max)$, which is always greater than the true rate of exploitation and is therefore conservative. C/V can be calculated for the whole stock or for individual ages separately. The catch of fish of a given age, t , in a given year, divided by the utilized stock of that age at the start of the year is:

$$u(\max)_t = \frac{C_t}{V} \quad (1.4-52)$$

From this a biostatistical rate of fishing can be estimated as:

$$F(\max)_t = -\log_e [1 - u(\max)_t], \quad (1.4-53)$$

where F is the instantaneous rate of fishing (impingement) mortality. An estimate of biostatistical catchability of fish at age t is:

Table 1.4-11

UTILIZED STOCKS OF WHITE PERCH, ARRANGED BY AGE AND BROOD

Hatching Year or Year Class	Age						Total Utilized Stock	Year
	0	1	2	3	4	5		
1965						1,000	3,625,000	1970
1966					24,000	5,000	4,260,000	1971
1967				90,000	60,000	50,000	5,553,714	1972
1968			320,000	170,000	150,000	50,000	5,811,571	1973
1969		1,150,000	400,000	<u>325,000</u>	175,000	25,000	4,748,714	1974
1970	2,040,000	1,740,000	950,000	400,000	100,000	25,000	4,646,571	1975
1971	1,885,000	1,635,000	635,000	235,000	85,000	10,000	5,546,571	1976
1972	2,443,714	1,943,714	443,714	193,714	43,714			
1973	2,607,857	1,607,857	407,857	157,857				
1974	2,312,143	1,512,143	712,143					
1975	2,422,857	2,022,857						
1976	2,600,000							

*Total utilized stock=All age groups that were present from different year classes in that impingement year.

$$q(\max)_t = \frac{F(\max)_t}{f_t} \quad (1.4-54)$$

where f_t is the number of effort units e.g., for impingement, f could be the number of million-gallon flow units during the time the fish were being impinged.

Using a random example, there were 150,000 age 3 white perch impinged during 1972 (Table 1.4-10) out of at least 325,000 present (Table 1.4-11); hence $u(\max)_t = .46$ and $F(\max)_t = .62$. Average values of $u(\max)$ suggest that young-of-the year fish are less vulnerable to impingement than are yearling, two and three-year-old-fish.

<u>Age</u>	<u>C/V or u(max)</u>
0	.25
1	.57
2	.53
3	.50

This might be expected for a species whose young are vulnerable to impingement only during the latter portion of the calendar year (they are subject to entrainment prior to this time), and whose vulnerability decreases with size.

Because the virtual population method is appropriate if fishing mortality comprises a major portion of total mortality for several age groups, it is not useful for analyzing entrainment, since the larval stages are generally characterized by high natural mortality. Although high numbers of larvae may be entrained from a cohort in year 0, relatively few members of this cohort will live to contribute to catches during subsequent years, thus depressing the utilized stock size in relation to entrainment numbers and seriously inflating impact estimates.

It would also be inappropriate to use this analysis if only one or two age groups of a species were impinged, since the biostatistical rate of exploitation, C/V , for the oldest age represented in the catch is always 100%, by definition. The next younger age will also have a considerable bias from the same source (Ricker 1.4-6, 1.4-40). Thus, the average C/V calculated for four and five-year-old white perch in the above example was .65 and 1.00, respectively.

The paradox of this method is that the smaller the ratio between impingement mortality, F , and total mortality, Z , ($=F+M$), i.e., the smaller the contribution of impingement mortality to total mortality, the more C/V or $u(\max)$ overestimates the true exploitation rate, u . This relationship is shown in Figure 1.4-8 which points out that in the simple equilibrium situation where neither natural nor fishing mortality changes with age, $C/V = u$ when impingement represents 100% of the mortality occurring within the population ($F=Z$). When impingement represents only 20% of total mortality, C/V overestimates u by a factor of five.

This relationship of C/V with u may offer a way of placing impingement mortality in perspective by comparing it to total mortality. If an independent estimate of u were obtained, for example, from a mark-recapture program, and C/V values were calculated by virtual population analysis, the resulting points plotted on Figure 1.4-8 would indicate the relationship of impingement mortality to total mortality.

The following analysis is based on: mark-recapture data and impingement counts during 1974; actual estimates of u for Hudson River white perch at three power plants; and hypothetical values for C/V based on the data in Tables 1.4-10 and 1.4-11:

<u>All Age Groups</u>	<u>Age 0</u>
$u = \frac{2,045,385}{41,300,000} = .0496 \text{ (actual data)}$	$u = \frac{1,840,847}{21,000,000} = .0877 \text{ (actual data)}$
$C/V = \frac{2,500,000}{4,748,714} = .5265 \text{ (hypothetical data)}$	$C/V = \frac{800,000}{2,312,143} = .3460 \text{ (hypothetical data)}$
$u: C/V = .09$	$u: C/V = .25$

This analysis suggests that the contribution of impingement mortality to total mortality for Hudson River white perch lies between the range of 9 to 25%, encompassed by the dashed lines in Figure 1.4-8.

If neither fishing nor natural mortality rates vary with age, Ricker (1.4-6, 1.4-40) points out that $C/V:u = Z:F = A:u$, and therefore that $C/V = A$, which is the total annual mortality rate expressed as the number of fish which die during a year,

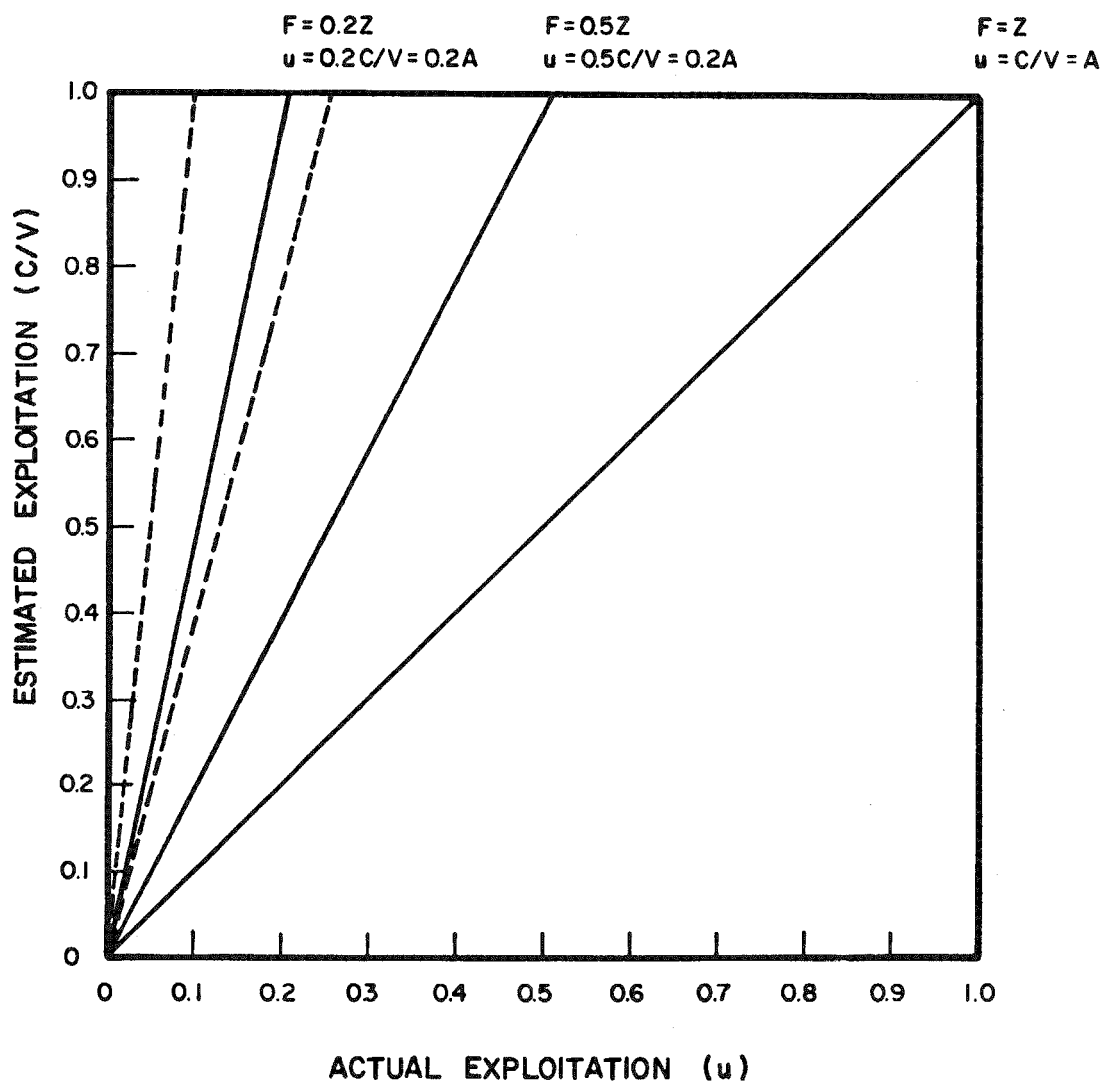


Figure 1.4-8. Relationship between actual exploitation (u) vs estimated exploitation ($C/V = u$ max) when fishing (F) represents varying proportions of a constant total mortality (Z).

divided by the initial numbers. In other words, the biostatistical rate of exploitation, in this case, is equal to the true annual mortality rate, A . However, both fishing and natural mortality commonly change with age, and Ricker (1.4-6, 1.4-40) has analyzed several of these situations as follows.

- When natural mortality, M , increases with age, and the rate of fishing is constant, C/V tends to increase with age whereas u decreases with age. The net result is an increasing discrepancy between C/V and u , so that the former can become two or three times the latter. However, C/V is close to A in such circumstances, being slightly greater than A .
- Within age-groups in which recruitment occurs, i.e., when F is increasing with age, C/V exceeds u by a much greater fraction of u than during a series of years when F does not vary with age. However, during such years C/V is less than A .
- If the rate of fishing, F , and thus the rate of exploitation, u , decrease with age after reaching some maximum, while M is constant, C/V decreases with age but not as rapidly as u does; thus C/V considerably exceeds the total mortality rate, A , over the years concerned.

As a general guideline for deciding if virtual population analysis is applicable to impingement impact assessment it should be pointed out that the older a fish is when caught, the more often it is included in the virtual population computation, i.e., the more utilized stocks this fish will appear in. Therefore, this type of analysis will most likely be of use where several age groups of a fairly long-lived stock are being affected. Because impingement is usually size-selective for the younger age groups, it would be helpful if older age groups from an active commercial and/or sport fishery could be included in the virtual population analysis, so that the usefulness of the relationship between utilized stock and the true stock can be maximized.

1.4.4 Survey-Removal or Change-In-Ratio Method

Populations which can be separated into two identifiable components which are selectively harvested at different rates can be estimated for initial population size. In order to accomplish this, investigators must know the original ratio of the two components, the number harvested from each component, and the final ratio of the two components. Classification into the two categories can be by any observable characteristic such as age, size, sex, color, etc., or even

by species. The survey-removal method has most frequently been applied in game management where for example, males of game birds or deer herds are selectively harvested and classification by sex is not difficult. Relatively little use has been made of the method in fisheries, probably because conditions appropriate for its use do not occur very frequently (1.4-6).

The simplest case for the application of this method would be for a closed population which suffers no natural mortality or any other unaccounted for mortality during the interval between the first and second surveys, i.e., during the time of the kill or harvest. The following definitions and relationships, after Chapman (1.4-41), Lander (1.4-42), and Ricker (1.4-6) are pertinent:

n_1, n_2 = the size of the samples taken at the beginning (survey 1) and the end (survey 2) of the kill period
 x_1, x_2 = number of X items in samples n_1, n_2
 y_1, y_2 = number of Y items in samples n_1, n_2
 $p_1 = x_1/n_1; p_2 = x_2/n_2$
 C_x = number of x-individuals caught during the kill period, i.e., between survey 1 and survey 2
 C_y = number of y-individuals caught during the kill period
 $C_T = C_x + C_y$ total kill or catch

The objective of this analysis is to determine:

N_{1x} = the number of X items at time 1

N_{1y} = the number of Y items at time 1

$N_{1T} = N_{1x} + N_{1y}$ total population size at the beginning of the study (time 1).

Chapman (1.4-41) shows that the change in the proportion of the total population in class X between the first and second surveys, i.e., the change from p_1 to p_2 caused by selective removal, leads to the following maximum likelihood estimators of initial population:

$$N_{1x} = \frac{p_1 (C_x - p_2 C_T)}{p_1 - p_2} \quad (1.4-55)$$

$$N_{1T} = \frac{C_X - p_2 C_T}{p_1 - p_2}$$

(1.4-56)

The value for N_{1Y} may then be obtained by the difference. The most stringent requirements of the survey-removal method are:

1. that the two classes differ sufficiently in vulnerability to cropping so that there is a detectable change in ratio from survey 1 to survey 2.*
2. that the vulnerability to the survey sampling gear for class X relative to class Y remain the same from survey 1 to survey 2.
3. that the vulnerability to the sampling gear be equal for the two classes so that the X:Y ratio in the sampling gear will reflect the true X:Y ratio in the water body.

The second requirement may be the most difficult to satisfy since it prohibits any distributional shifts which would change the proportions of the two classes residing within the sampling area. One way to minimize the possibility of such a distributional artifact would be to sample the entire spectrum of habitats available to the fish under study. If more than one fishing gear-types are used for collection, the proportion of total sampling effort by each gear should remain the same for both surveys.

The above method is applicable only when the assumption is made that no natural mortality occurs during the cropping period. Lander (1.4-42) points out that for a given set of cropping rates and initial populations, the total catch and the catch of X-class individuals will be lower in the presence of concurrent natural mortality than it would be in the absence of such mortality. The initial fraction of total population in the X class, p_1 , is independent of natural mortality, M , and if natural mortality operates at the same rate on both the X and Y populations, i.e., $M_X = M_Y$, then the final fraction of total population in the X class, p_2 , is also independent of natural mortality. Therefore, because of the

*A corollary here is that the cropping rate must be of sufficient magnitude to generate a detectable change in ratio. If no such change in ratio is observed for two unequally vulnerable classes, and if sample sizes and techniques are adequate to permit detection of such a change if it were present, then it may be assumed that the magnitude of cropping during the removal period was too low to generate a significant change in the population and that impact is minimal.

nature of Eqs. (1.4-55) and (1.4-56), estimates of N_{1X} and N_{1T} will be biased low in the presence of natural mortality.

The trend of increasing bias in the estimate of N_{1T} with increasing rates of natural mortality is demonstrated in Table 1.4-12 for an initial population of 1,000,000 fish. The estimate of the initial total population is about 20% low when $M = 0.50$ and approximately 30% low when $M = 0.80$. However, Table 1.4-12 is based on annual rates of instantaneous mortality, and it is assumed that one year elapsed between surveys. The observed bias caused by concurrent natural mortality would be greatly reduced if, for example, only one month elapsed between the first and second surveys.

Careful planning and proper sampling design are essential to a useful survey-removal experiment, and the first step is to determine the minimum number of fish that need to be collected during the survey period to accomplish the study objectives. This minimum design should be based on the supposition that all the underlying assumptions have been satisfied. Paulik and Robson (1.4-43) state that investigators should:

"... first adopt an optimistic frame of mind and see what the minimum size of the experiment would have to be if we were working under ideal conditions. If available funds are inadequate under ideal conditions, they will be even more inadequate under real conditions and the experiment should be either abandoned or refinanced."

For practical purposes we can assume that choosing approximately equal sample sizes for survey one and survey two, i.e., $n_1 = n_2 = n$, represents a nearly optimum allocation of effort, although Seber (1.4-5) points out that this is not exactly true. Paulik and Robson (1.4-43) provide graphs, reproduced here as Figures 1.4-9, 1.4-10, and 1.4-11, which show sample sizes required for population estimates of various levels of accuracy (ϵ) and precision ($1-\alpha$), similar to the graphs provided by Robson and Regier (1.4-12) for Petersen mark-recapture estimates in Section 1.4.1.1.1.

The required sample sizes increase with the required accuracy and precision levels and decrease with the exploitation rate (u) prevailing between n_1 and n_2 , and also with the magnitude of the change in proportion of X-types between the first and the second survey ($\Delta p = p_1 - p_2$). Seber (1.4-5) notes that

Table 1.4-12

HYPOTHETICAL POPULATION ILLUSTRATING THE EFFECT OF
CONCURRENT NATURAL MORTALITY ON POPULATION ESTIMATES WHEN
POWER PLANT MORTALITY AND INITIAL POPULATION ESTIMATES ARE HELD CONSTANT

Quantity	Symbol	Numerical Values		
Initial Total Population	N_{1T}	1,000,000	1,000,000	1,000,000
Fraction of Total Population in X class	p_1	.80	.80	.80
Initial X Population	$N_{1x} = p_1 N_{1T}$	800,000	800,000	800,000
Initial Y Population	$N_{1y} = (1-p_1)N_{1T}$	200,000	200,000	200,000
Instantaneous Rate of Natural Mortality	M	0	0.50	0.80
Instantaneous Power Plant Mortality Rate for X Population	$P_x = -\ln \left[1 - \frac{C_x(p_1 - p_x)}{p_1(C_x - p_2 C_1)} \right]$	0.60	0.60	0.60
Instantaneous Power Plant Mortality Rate for Y Population	$P_y = P_T + \ln \left(\frac{1-p_1}{1-p_2} \right)$	0.20	0.20	0.20
Instantaneous Power Plant Mortality Rate for Total Population	$P_T = P_x + \ln \left(\frac{p_2}{p_1} \right)$	0.506	0.506	0.506
Catch of X Type Fish in Impingement	$C_x = \frac{P_x N_{1x} [1 - e^{-(P_x + M)}]}{P_x + M}$	360,951	291,111	258,310
Catch of Y Type Fish in Impingement	$C_y = \frac{P_y N_{1y} [1 - e^{-(P_y + M)}]}{P_y + M}$	36,254	28,767	25,285
Total Impingement Catch of X and Y Types	$C_T = C_x + C_y$	397,205	319,878	283,595
Natural Deaths of X Type Fish	$D_x = \frac{MN_{1x} [1 - e^{-(P_x + M)}]}{P_x + M}$	0	242,592	344,413
Natural Deaths of Y Type Fish	$D_y = \frac{MN_{1y} [1 - e^{-(P_y + M)}]}{P_y + M}$	0	71,916	101,139
Total Natural Deaths	$D_T = D_x + D_y$	0	314,508	445,552
Final X Population	$N_{2x} = N_{1x} - C_x - D_x$	439,049	266,297	197,277
Final Y Population	$N_{2y} = N_{1y} - C_y - D_y$	163,746	99,317	73,576
Final Total Population	$N_{2T} = N_{2x} + N_{2y}$	602,795	365,614	270,853
Fraction of Final Total Population in X Class	$p_2 = \frac{N_{2x}}{N_{2x} + N_{2y}}$	0.7284	0.7284	0.7284
Estimate of Initial Total Population	$\hat{N}_{1T} = (C_x - p_2 C_T) / (p_1 - p_2)$	1,000,375	811,618	722,617

sample size is much more sensitive to Δp than to u , and states, based on Paulik and Robson (1.4-43), that for Δp values between 0.05 and 0.10, use of the survey-removal method is questionable, and the assumptions upon which the method is based become critical.

According to the hypothetical example with $M = 0$ in Table 1.4-12, $\Delta p = 0.80 - 0.7284 = 0.0716$ and $u = \Delta p / [(C_X/C_T) - p_2] = 0.397$, indicating that survey sample size should be between 3,000 and 4,000 fish for each survey to be 90% confident that the estimate $\hat{N}_{1T}$ will not be in error by more than 25% (Figure 1.4-9). Figures 1.4-10 and 1.4-11 show recommended sample sizes for $\alpha = 0.05$ and for varying levels of ϵ and p_1 . Seber (1.4-5) states that for a fixed Δp , the sample size is insensitive to values of p_1 in the vicinity of 0.50, e.g. 0.40 to 0.60. However, $p_1 = 0.80$ in Table 1.4-12, suggesting that the sample sizes for this study should be in the upper range of those shown in Figure 1.4-11 for $\Delta p = 0.07$, i.e., between 2,000 and 3,000 for $\alpha = 0.05$ and $\epsilon = 0.50$.

As was true for the Petersen mark-recapture method described in Section H1.4.1.1.1, which can be regarded as a special case of the survey-removal or change-in-ratio method (1.4-44), it is evident that one of the prerequisites to the design of a useful survey-removal study is an initial estimate of the expected Δp for a given level of removal through impingement. Otherwise, the choice of where to enter the x-axes in Figures 1.4-9, 1.4-10 and 1.4-11 would be an uninformed guess, and the chosen sample size of the initial survey may be too low to be useful in the study.

To calculate confidence intervals for population estimates from survey-removal studies, Paulik and Robson (1.4-43) show that it is necessary to first calculate the variance of the absolute abundance estimator, $V(N_{1T})$, and the variance of the estimate of the proportion of X-types in survey one, $V(p_1)$, and in survey two, $V(p_2)$. These variances are calculated by the following equations:

$$V(p_1) = \frac{p_1 (1-p_1)}{n_1} \quad (1.4-57)$$

$$V(p_2) = \frac{p_2 (1-p_2)}{n_2} \quad (1.4-58)$$

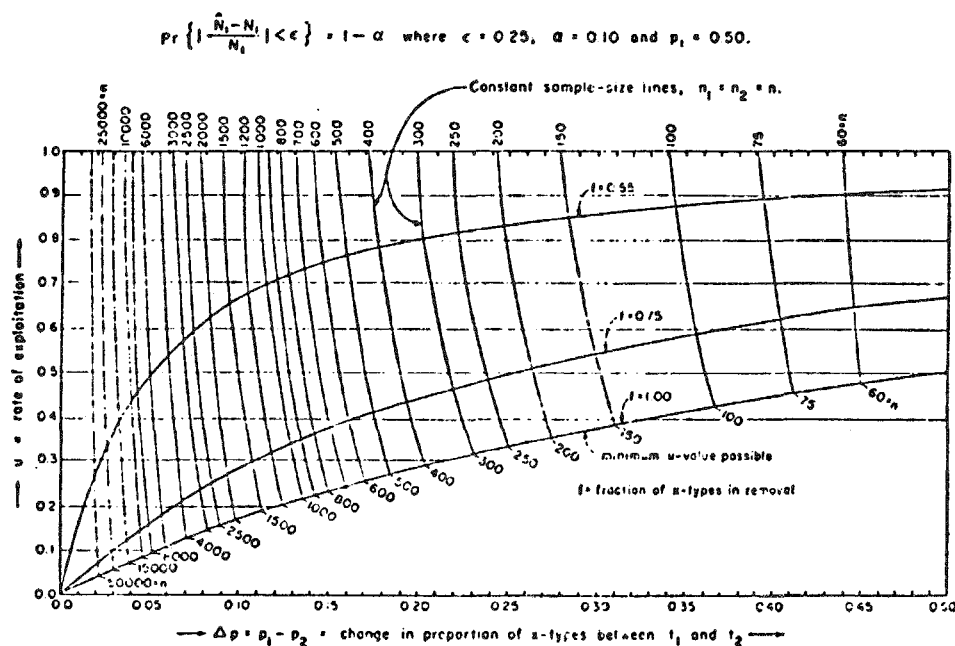


Figure 1.4-9. Sample size required for dichotomy estimates of population abundance with less than 25% error (with probability 0.90) when the initial proportion of x-types is 0.50.

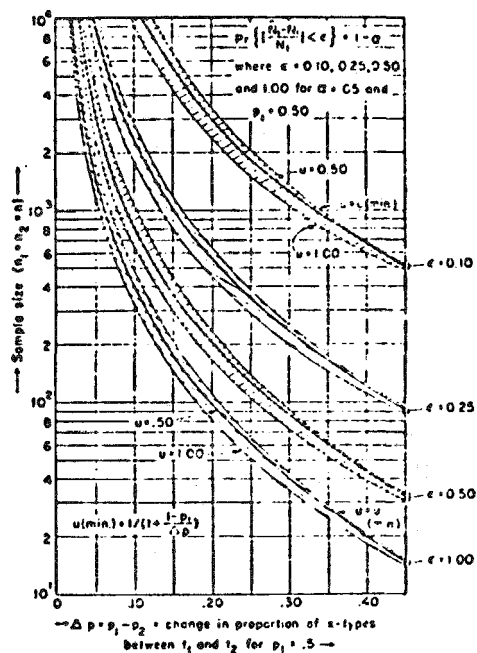


Figure 1.4-10. Sample size required for dichotomy estimates of population abundance with less than 100% error (with probability 0.95) for $\epsilon = 0.10, 0.25, 0.50$, and 1.00 when the initial proportion of x-types is 0.50.

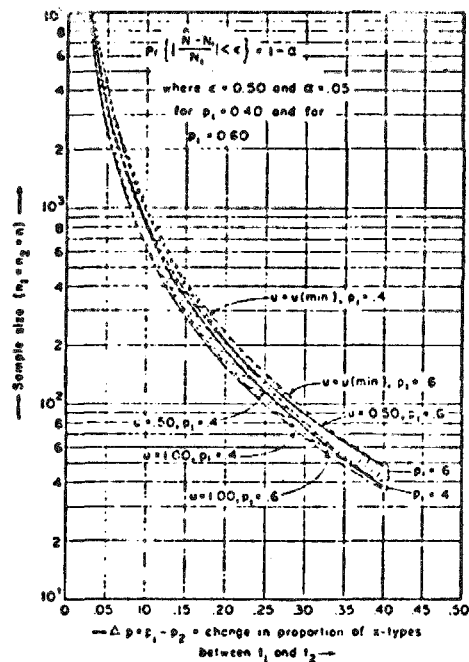


Figure 1.4-11. Sample size required for dichotomy estimates of population abundance with less than 50 % error (with probability 0.95) for the initial proportion of x-types $p_1 = 0.40$, and $p_1 = 0.60$.

$$V(\hat{N}_{1T}) \approx (p_2 - p_1)^{-2} [\hat{N}_{1T}^2 V(p_1) + (\hat{N}_{1T} + C_T)^2 V(p_2) + (1-p_2)^2 V(C_X) + p_2^2 V(C_Y)]$$

(1.4-59)

If C_X and C_Y are known rather than estimated variables, i.e., if an absolute count of all impinged fish of the species under study were made during the period between survey one and survey two, then $V(C_X) = V(C_Y) = 0$, and it is not necessary to include the last two terms of Eq. 1.4-59.

Paulik and Robson (1.4-43) discuss three methods for calculating confidence intervals, but only the simplest, the direct delta-method, will be described here. This method relies on the assumption that N_{1T} is an asymptotically normal variable (1.4-5) i.e., that values of the estimate N_{1T} tend toward a normal distribution as sample size increases. Given this assumption, the standardized normal deviate, $Z = (\hat{N}_{1T} - N_{1T}) / \sigma(N_{1T})$, is also distributed as a normal random variable with mean 0 and variance 1. Confidence intervals are then calculated as:

$$L = \hat{N}_{1T} + Z_{\alpha/2} \hat{\sigma}(N_{1T})$$

(1.4-60)

and

$$U = \hat{N}_{1T} + Z_{(1-\alpha)/2} \hat{\sigma}(N_{1T})$$

(1.4-61)

where $Z_{\alpha/2} = -1.960 \approx -2$ and $Z_{(1-\alpha)/2} \approx 2$ for $\alpha = 0.05$, leading to the maxim that a 95% confidence interval extends two standard deviations on either side of the estimate.

Variance of the estimates in Table 1.4-12, with the use of $n_1 = 3381$ and $n_2 = 2979$ as stated in Section H1.8.3.5 are calculated as follows:

$$V(p_1) = \frac{0.80(1 - 0.80)}{3381} = 0.0000473$$

$$V(p_2) = \frac{0.7284(1 - 0.7284)}{2979} = 0.0000664$$

$$V(\hat{N}_{1T}) \approx (0.7284 - 0.80)^2 [(1,000,375)^2 (0.0000473) + (603,170)^2 (0.0000664)] \\ \approx 1.394556$$

$$\hat{\sigma}(N_{1T}) = \sqrt{1.394556 \times 10^{10}} = 118,091.321$$

Therefore, the lower and upper limits of the 95% confidence interval for N_{1T} are:

$$L = 1,000,375 + [(-2) (118,091.321)] = 764,192$$

and

$$U = 1,000,375 + [(2) (118,091.321)] = 1,236,558$$

Thus, the probability that the interval from 764,192 to 1,236,558 covers N_{1T} is at least 0.95, i.e., $\Pr (L < N_{1T} < U) \geq 1 - \alpha$.

1.4.5 Estimating Population Size With the Use of Catch Equation

Although the catch equation is basic to many estimation procedures in fisheries, it is demanding because it contains three unknown parameters, i.e., the population size, the catchability coefficient, (Section 1.4.2.3 and 1.8.3.4.3 Population Response Analysis), and the instantaneous natural mortality rate, and two observed or directly estimated statistics, catch and effort.

The catch equation is generally written as:

$$C_t = N_{t_0} [1 - e^{-(M + qf)}] \frac{qf}{M + qf} \quad (1.4-62)$$

where

C_t = the catch during a time period t (day, month, year, season)

N_{t_0} = the number of individuals in the population at the beginning of the time period.

M = the instantaneous rate of natural mortality over the time period t

q = catchability coefficient defined as the proportion of the population caught by one unit of effort, i.e., the instantaneous rate of fishing mortality associated with one unit of effort

f = the total number of effort units applied during the time period

qf = P = the instantaneous rate of fishing (plant) mortality over the time period t.

Dividing both sides of Eq. 1.4-62 by f as follows enables the catch equation to be written in terms of catch-per-effort:

$$C_t/f = N_{t_0} [1 - e^{-(M + qf)}] \frac{q}{M + qf} \quad (1.4-63)$$

Murphy (1.4-45) has expressed the catch equation as follows: "The catch during a specified time period (C_t) is equal to the population at the beginning of the time period (N_{t_0}), multiplied by the fraction of the population that dies during the period $[1 - e^{-(M + qf)}]$, and, finally, multiplied by the fraction of the total deaths ascribable to the fishery $\left(\frac{qf}{M + qf}\right)$."

Solving for population size, Eqs. 1.4-62 and 1.4-63 can be written as:

$$N_{t_0} = \frac{C_t}{[1 - e^{-(M + qf)}] \left(\frac{qf}{M + qf}\right)} \quad (1.4-64)$$

and

$$N_{t_0} = \frac{C_{t/f}}{[1 - e^{-(M + qf)}] \left(\frac{q}{M + qf}\right)} \quad (1.4-65)$$

The main prerequisite to satisfy the requirements of the catch equation is to have a reliable record of catch and effort for each time period analyzed. If natural mortality and catchability are the same for each age group in the catch,

then total catch of all age groups combined can be used for C_t . If not, then age-specific values of C_t should be used for each regime of M and/or q .

Some of the methods used to estimate M are described in Section 1.8.3, and some of those for q may be found in Sections 1.4.2.3, 1.4.3 and 1.8.3.4. If an estimate of P is available, then there is no need to evaluate q , since $P = qf$. One way to obtain an average estimate of q would be to calculate impingement rates during those season or years for which population estimates were available. For example, if an average impingement rate of two fish per million gallons of plant flow were calculated during a period when the average source population numbered 21,000,000, the estimate for the catchability coefficient, i.e., the proportion of the total population captured by a single, million-gallon unit of effort, would be:

$$q = \frac{2}{21,000,000} = .0000001$$

A more convenient unit of effort may be one million cubic meters of flow, whereby the above coefficient would convert to:

$$q = \frac{2/.003785}{21,000,000} = .000025$$

As a hypothetical example, average values for q and M were 0.0000001 and 0.84, respectively, for the years when impingement rates, population estimates, and catch curve data were available. Then, for the year when 2,000,000 young-of-the-year fish were impinged with a total of 750,000 million-gallon units of effort expended, the initial population of young fish at the beginning of the impingement season, calculated by Eq. 1.4-64 would be:

$$N_{t_0} = \frac{2,000,000}{(0.599483) \left(\frac{0.075}{0.915} \right)} = 40,701,738$$

1.4.6 Hydroacoustic Assessment of Fish Populations

1.4.6.1 Introduction. The problem of determining the numbers of fish within a specified region has been made easier with the use of underwater sound or sonar.

Sonar systems are said to be active when they produce a sound which propagates through the water, strikes a target, and has its returning echo received by a hydrophone. Passive sonar, on the other hand, is a listening device that depends on sound produced by the target. In either case, once sound impinges on the listening device it is converted into an electrical signal. This output is amplified and processed and displayed in a variety of ways. An understanding of the usefulness of hydroacoustic (underwater sound) techniques in determining the quantity or biomass of fish depends on the knowledge of each step in the production, transport through the medium, deflection or reflection by the target, and reception and processing of sound.

Echo sounders have been used since the early 1940's to locate or track fishes with hydroacoustic gear that usually resulted in a strip chart of electrical signals burned into a paraffin coated paper (1.4-46). Since that time many new systems are available for use by commercial fishermen. These units have been useful for detecting the presence or absence of fishes, but attempts to relate the strip chart counts to the abundance of fish have been of dubious value principally because their variance is usually from two to five times the mean value. Sometimes the variances can be on an order of magnitude greater than the estimate.

With the advent of submarine warfare and the necessity of submarine detection, the precision and accuracy of hydroacoustic instrumentation have been greatly increased and an understanding of the phenomenon of sound transmission in water developed (1.4-47). However, the adaptation of this technology to fishery science has been slow and often applied in a unscientific and misleading manner.

Descriptions of the behavior of sound, the instrumentation necessary for quantification, and a general review of past research are given to illuminate the hydroacoustic technique.

The hydroacoustical equipment (Echo Sounder or Fathometer) produces an electrical signal of a specific carrier frequency, amplitude and duration at the terminals of a transducer. The transducer converts this electrical signal into a dynamic pressure wave of a specific frequency, amplitude, and duration. Transduction, the conversion of sound into electricity is usually accomplished through the ability of certain crystals to develop an electric charge when stressed (piezoelectric). The counterpart occurs when a magnetic field produces a change in the shape of certain substances (magnetostriction). The resulting hydroacoustical

signal, called a pulse, is projected from the transducer and spreads spherically in water. The hydroacoustical energy radiated by the transducer is greatest on its axis (acoustic) and varies off the axis as a function of the transducer's directivity. The directivity function is the relationship of the intensity of the sound directly in the center of the sound beam to the intensity at some distance from the center (usually to the half-power beam width or the 3 dB points).

When this projected pulse encounters an object whose acoustical properties are different from those of water (that is, an impedance mismatch), a portion of the hydroacoustical energy is reflected or scattered back toward the source. This reflected (or back scattered) energy also spreads spherically and varies as a function of the directivity of the object. When the transducer receives the reflected energy, an electrical signal is produced and is handled in various ways to indicate the presence of a target processed. In modern "fish-counting" machines, this signal is generally recorded on magnetic tape or converted into a visual display by the activation of an electrical stylus which produces a mark on electrosensitive paper.

Whether or not an individual signal is recorded depends on the amplitude of the hydroacoustical echo signal and on the choice of an acceptable amplitude on the part of the researcher as well as on the ambient noise level. By increasing the sensitivity (thus decreasing the amplitude needed to stimulate the recorder), a researcher can record the presence of small objects in the water column. When ambient noise level is of sufficient magnitude, it can mask the presence of targets.

A brief generalized description of the equipment used for hydroacoustic work is given to aid the reader in understanding the more detailed mathematical presentation that follows.

A function generator is used to produce an electrical signal, usually a square pulse, sine wave. This must be amplified in order to overcome the resistance and capacitance of the transducer crystals and to produce a pulse of sufficient magnitude to travel to and from the target. A high quality linear amplifier is required for this. The sound is projected with a transducer and can be received by the same transducer or by a hydrophone. The electrical analogue signal must then be amplified so that it can be observed and recorded. Again, a high quality linear amplifier is required. Once amplified, the signal may be observed as a deflection of

the base line of a cathode ray tube, printed on a strip chart and/or recorded on magnetic tape.

1.4.6.2 Hydroacoustic Formula. The above discussion was a simplified view of the hydroacoustic phenomena. Urick (1.4-48) presents the basic echo sounding equation for a single target.

$$V_{rms}^2 = P_o^2 \cdot K^2 \cdot \frac{\alpha}{4\pi} \cdot \frac{1}{R^4} e^{-2\alpha R} \cdot G^2(0, \phi) \quad (1.4-66)$$

where

V_{rms} = root mean squared voltage produced at transducer terminals

K = transducer voltage response (volts $\text{cm}^{-2}/\text{dyne}$)

P_o = rms pressure at unit distance - source level (dyne/cm^2)

$G(0, \phi)$ = transducer directivity function

$(0, \phi)$ = directional spherical coordinates relative to reference coordinates fixed at transducer.

σ = ratio of power scattered in direction of transducer per unit solid angle to the incident intensity at target.

α = path attenuation loss due to combine effects of scattering and absorption

R = range to target (meters)

It is sometimes more convenient to use an auxiliary variable, TS, in decibels because the acoustic cross section is not easily calculated for fish (see section on individual fish target strength). The relationship of TS to acoustic cross section is:

$$TS = 10 \log_{10} \frac{\sigma}{4\pi} \quad (1.4-67)$$

and thus Eq. 1.4-66 becomes

$$TS = 10 \log_{10} \frac{R^4 e^{-2\alpha R}}{P_o^2 K^2 G^2 (0, \phi)} \quad V_{rms}^2 \quad \text{or} \quad \frac{V_{pp}^2}{8} \quad \text{where } \begin{matrix} \text{pp is voltage peak} \\ \text{to peak} \end{matrix} \quad (1.4-68)$$

The quantity TS is called the equivalent (plane wave) target strength (1.4-49) and refers to the echo produced by an object in the path of a plane acoustic wave.

Eq. 1.4-68 shows that in order to count individual targets (multiple targets are represented by another equation presented below), several of the parameters of the instrumentation either must be measured directly or experiments performed to quantify them, e.g. K^2 , P_o^2 , and $G^2 (0, \phi)$. G^2 is the directional characteristic of the transducer on a three dimensional basis. The analytical expressions of the directional characteristics for a particular transducer are not usually sufficient, and direct measurements are necessary for their verification. K^2 , the transducer voltage response, and P_o^2 , the source level, must also be verified by direct measurement.

The target strength (TS) is considered herein since this is the parameter which will result in the hydroacoustic equipment's recording or not recording a fish. That is, based on intuition or direct observations of the target strength of fish, the investigator chooses a target strength which he will accept and count as a fish. If all the other equipment and hydroacoustical variables were known with their appropriate statistical limits (confidence intervals) and if the target strength of fish were known, then the counting of individual fish would be a relatively easy task. However, the usual case is that fish target strength is not known, and the confidence intervals on the instrumentation employed are large. Even if the accurate attainment of these parameters is possible, there are still other factors to be considered. An important consideration in the counting of individual fish is that there must be a certain minimum distance between targets (fish) in order for the hydroacoustic apparatus to discern them and count them separately. As Suomala (1.4-50) points out, the minimum separation or distance between two hydroacoustically identical targets, with pulsed hydroacoustical transmissions employed, is defined by the following expression:

$$d = \left[R_1^2 R_2^2 \phi^2 + \left(\frac{ct}{2} \right)^2 \right]^{1/2} \quad (1.4-69)$$

where

- C = velocity of propagation of sound in sea water
- t = time interval of the transmitted hydroacoustical pulse (pulse length)
- R_1, R_2 = range to the insonified targets referenced to the hydroacoustical transducer
- ϕ = angle between the targets referenced to the hydroacoustical transducer (see Figure 1.4-12 for the coordinate reference angles of the transducer)

When the distance between individual targets is assumed to be great enough to allow for discrimination, then there are still other practical considerations that must be recognized:

1. The maximum received echo signal from a single insonified target can vary by a factor of from two to three, from the absolute maximum, depending upon the location of the target relative to the axis of symmetry of the transducer directivity function (1.4-49, 1.4-50, and 1.4-51). In other words, the same size fish can produce a received echo signal which may be three times greater when the fish is on the acoustical axis than when off the axis.
2. The absolute maximum received echo signal from a single insonified fish target with an air bladder, can vary by a factor of from two to four depending upon a small difference in the pitch attitude (approximately 6 to 9 degrees) of the target relative to the insonifying pressure wave (1.4-52). This work was performed on marine commercially important fish, i.e., herring, cod and dogfish. Because fish attitude in the wild is not easily determined and because estimates of fish abundance would depend on the average attitude of the insonified fish, this factor must be considered in quantification exercises.
3. The minimum distance between a target to be counted and the bottom of the aquatic habitat must be known to be greater than the resolution capability of the hydroacoustical apparatus and received echo signal processing technique (1.4-52).

The foregoing discussion pertains to the recognition of single fish and deals only with the properties of sound in water and backscattering from targets. Once the echo signal arrives at the transducer and is sufficient to stimulate

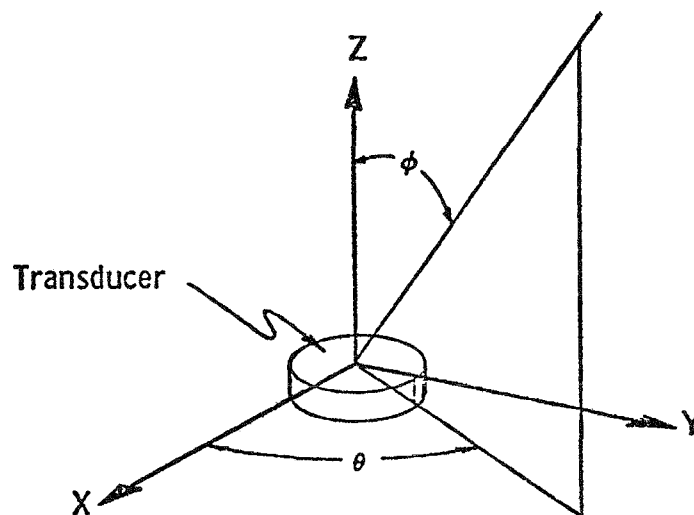


Figure 1.4-12. Transducer directional coordinate reference.

a response, then the hardware employed to store and process the signal becomes important. For the moment a consideration of the electrical apparatus used for storage and processing will be deferred until a presentation of the backscattering from multiple targets is presented.

1.4.6.3 Multiple Targets. It has been shown by Lozow and Suomala (1.4.49)* that the estimated density (ρ) of an aggregation of randomly distributed targets is:

$$\rho = \frac{\bar{R}^2 e^{-2\alpha\bar{R}}}{P_o^2 \frac{\bar{\sigma}}{4\pi} \frac{Ct}{2} \psi_{\infty}} \cdot \bar{p}^2 \quad (1.4-70)$$

where

$\bar{R}$ = average range to targets

α = sound attenuation loss due to the combined effects of scattering and absorption

$\bar{p}^2$ = randomly varying received sound pressure

P_o^2 = transmitted acoustic source level

$\frac{\sigma}{4\pi}$ = average target strength

C = sound propagation velocity

t = time interval of transmitted pulse

ψ_{∞} = integrated function of transducer directivity

Typically, data gathered during hydroacoustic surveys is in the form of voltages at the transducer terminals. The randomly varying sound pressure can be written in terms of a randomly varying voltage as follows.

*The reader is referred to pages 1 through 4 of this publication wherein the assumption that fish are distributed in a given volume of water in a random fashion is suggested, and the distribution is assumed to obey the Poisson Probability Law.

$$P_{rms}^2 = \frac{(V_{rms}^2)}{K^2} \quad (1.4-71)$$

where

K^2 = receiving voltage response of the hydroacoustical instrument

Hamilton et al. (1.4-53) find it convenient to introduce a new term, M to represent the size of the target aggregation in terms of speed of sound and pulse length.

$$M = \frac{\delta}{Ct/2} \quad (1.4-72)$$

The geometry of the hydroacoustic transducer relative to the targets of interest is shown in Figure 1.4-13.

The average density of targets, $\bar{\rho}$, can then be defined as

$$\bar{\rho} = \frac{1}{M} \sum_{N=1}^M \rho_N \quad (1.4-73)$$

With these definitions and the substitution of Eq. 1.4-71 and 1.4-73 into 1.4-70, the following estimate of target density is attained:

$$\bar{\rho} = \frac{\bar{R}^2}{8 P_o^2} \frac{e^{-2\alpha\bar{R}}}{\frac{\sigma}{4\pi} K^2 \delta \psi_{\infty}} \sum_{N=1}^M \left(\tilde{V}_{PP}^2 \right)_N \quad (1.4-74)$$

If all of the parameters of Eq. 1.4-74 were known, it would be possible to derive an estimate of the abundance of targets present. This, however, is seldom the case and in particular the average target strength of the fish is rarely known. Target strength varies proportionally with the length of the fish and differs among species. Thus, for groups of fish composed of different sizes or species, the absolute abundance cannot be obtained. Then, an indirect approach is taken to overcome the problem of unknown average target strength.

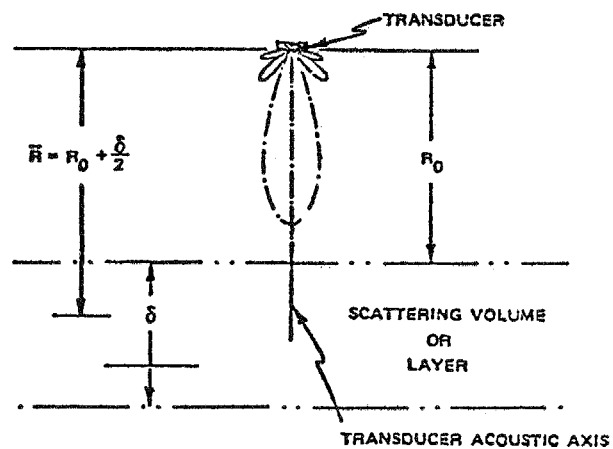


Figure 1.4-13. The geometry of a hydroacoustical transducer in relation to the targets of interest.

The average target strength combined with the density can be set equal to a new term, VB, (volume backscatter), and we obtain:

$$VB = \rho \frac{\sigma}{4\pi} = \frac{\bar{R}^2 e^{-2\alpha R}}{8 P_o^2 K^2 \delta \psi_{\infty}} \sum_{N=1}^M \left(\bar{V}_{PP}^2 \right)_N \quad (1.4-75)$$

Volume backscatter can be obtained without the knowledge of the target strength of the fishes. This results in a relative estimate of the acoustic energy returning from the insonified targets.

Even if it is possible to identify all of the parameters in Eq. 1.4-74, of course with the exception of the peak-to-peak voltage which results from actual measurements, there are still other factors to be considered, such as what characteristics of fish schools produce echoes of a particular magnitude. Lozow and Suomala (1.4-54) state that there are two components to the received echo signal. They are: 1) the incoherent component and 2) the coherent component.

1. The Incoherent Component. If the average target distribution is uniform, serial estimates of the target density will exhibit significant variations, and the arithmetic average of the serial estimations can be shown to be proportional to the number of insonified targets. This proportionality is consistent if the average distance between targets is greater than the wavelength of the transmitted hydroacoustical signal.
2. The Coherent Component. If the average target distribution is uniform and the average distance between the targets is less than the wavelength of the transmitted carrier signal, or if the number of targets in the region determined by one quarter of this wavelength is large, it can be shown that a single estimation of target density is proportional to the square of the number of insonified targets. When these conditions are met then the scattering of sound is termed coherent and will predominate in the received echo signal.

Lozow and Suomala (1.4-54) present an equation that can be used to estimate the effects of incoherent and coherent components of the signal. Depending upon the true density of targets, the received signal can be interpreted as ranging from directly proportional to the number of targets to the square of the number of targets. The latter may result, when aggregations of fish are encountered, in significant overestimates of the numbers of fish present.

1.4.6.4 Instrumentation Calibration. As shown above, there are many instrumentation constants to be determined if reliable abundance estimates are expected. These include:

1. Transducer resonant frequency and impedance alignment
2. Transmitter carrier frequency
3. Transmitter power
4. Transmitted sound pressure (source level)
5. Transducer receiving sensitivity (voltage response)
6. Equipment receiving sensitivity (voltage response)
7. Equipment and ambient noise (during actual sampling)
8. Transducer directivity function

Unless the value of each of these parameters is known, attempts at determining abundance of targets or biomass will not be successful.

The first three of these parameters can be measured in the laboratory with standard electrical engineering techniques. Four through seven, however, need to be measured in the field and require a rigorous measurement program to ensure that the instrumentation is functioning consistently at a specified level. The eighth parameter should be determined in the laboratory and verified with field observations.

The easiest method to determine source level and receiving sensitivity is to suspend a machined sphere, with known backscattering characteristics and target strength, in the center of the acoustic beam. The magnitude of the returning signal is then determined and Eq. 1.4-68 set equal to receiving sensitivity (P_o^2) times source level (K^2). Since the target strength of the sphere is known and since the sphere is suspended on the acoustic axis, the directivity function is one, and the equation is solvable. Of course, this exercise should be repeated often, and confidence intervals calculated. Additional calibration methods for determining $P_o^2 K^2$ are given by Urick (1.4-47).

The seventh instrumentation calibration is required because sound produced by wind, waves, and internal movement of water masses has the potential for interfering with the discriminative ability of the hydroacoustic instrumentation. Ambient noise of a frequency within the band width limitation of the transducer and of sufficient intensity will mask returning echos from targets of interest. Therefore, the

level of ambient noise in the area to be surveyed should be determined before the frequency to be used is chosen.

One additional liability associated with hydroacoustic sensing is that some researchers find it necessary to convert the raw received signal from the analogue to a digital form. Also, because the tape recorder needed to record at kilohertz frequencies is expensive, the signal is often heterodyned (frequency-reduced) down to a frequency that can be recorded on a less expensive tape recorder, of lower fidelity. Orłowski (1.4-55) has shown that changing the frequency of an echo pulse from 38 to 12 kHz results in significant information loss (from 15 to 40%), and yet most hydroacoustic research is conducted with frequency-shifted data. When frequency-shifted data are to be used, the investigator must also determine the information lost during the process.

1.4.6.5 Uses of and Problems with Hydroacoustics. Hydroacoustics has been used in a variety of ways in fisheries science, including research into the movement of salmon ascending streams, the distributional ecology of lake and ocean fishes, pelagic stock estimates, comparisons of relative abundance near power plants and in control areas to determine the consequences of cooling water discharges, determinations of bottom vegetation distribution, studies of deep-sea scattering layers, and as a measure of zooplankton populations and communities. The first four of these categories are briefly reviewed to illustrate the appropriate application of hydroacoustics to fisheries science.

1.4.6.5.1 Studies of movement. Hydroacoustics can be used to monitor fish movement either passively or actively. In the passive system, an ultrasonic transmitter is attached to the fish, and receiver hydrophone stations are established in the area being studied. If two stations are set up the position of the fish can be determined sequentially. With the addition of a magnetic tape recorder, the position of the fish can be monitored automatically. This has been done successfully by Osburn et al. (1.4-56), Thorson et al. (1.4-57), Pincock et al. (1.4-58), Hawkins et al. (1.4-59), and Holland et al. (1.4-50). This procedure is limited to narrow waterways where the fish must pass within receiving distance of the hydrophones.

A second or active method of determining movements of fish has been used to document the one-way travel of fishes as they ascend rivers for spawning

(1.4-61, 1.4-62, 1.4-63). The acoustic transducer is positioned nearshore and directed horizontally. The beam width of the transducer must be narrow so that it does not intersect the bottom or surface. This procedure can result in counts of individual fish if, and only if, they pass through the beam individually or are separated from one another as specified in Eq. 1.4-69. When the distance separating the fish is small, the echo from the fish will arrive at the transducer during the same time interval (pulse length) and the pulse recorded as one. This pulse will, however, be slightly longer than the transmitted pulse and hence identifiable as being produced by more than one target. In order to convert these latter signals into abundance estimates, the target strength of the fish is needed. Additionally, species identification can be assumed or verified with the occasional use of capture gear.

1.4.6.5.2 Distributional ecology of fishes. Hydroacoustic assessment techniques have been used to determine the distribution of fishes in lakes, rivers and the oceans (see, for instance, 1.4-64, 1.4-65, 1.4-66, 1.4-67, 1.4-68, 1.4-69). This type of study depends on the knowledge of the fish species present and hence the use of traditional fisheries gear. It can be quite useful in impact assessment for defining the population boundaries needed to estimate population size. Behavior of the fish can be accurately described with this technique. For example, Hergenrader and Hasler (1.4-70) studied the diel behavior of yellow perch under the ice by scanning schools of fish during the day and night. They demonstrated that schools of fish become less compact at night and also showed that seasonal onshore movement of yellow perch occurred.

1.4.6.5.3 Pelagic stock estimates. Hydroacoustics has also been used to estimate the biomass and sometimes the actual abundance of pelagic fishes in fresh and salt water (1.4-66, 1.4-67, 1.4-68, 1.4-71, 1.4-72, 1.4-73, 1.4-74, 1.4-75, 1.4-76, 1.4-77, 1.4-78, 1.4 79, 1.4-80). By and large, the most outstanding problems with these studies can be summarized into four categories. These are:

1. non-rigorous approach (no or little calibration or verification)
2. different instrumentation is used by almost every investigator
3. a priori signal processing before storage of data onto magnetic tape
4. no target strength data for species of interest.

A series of calibration exercises is required to assure the consistent operation of the instrumentation and the values of instrumentation constants. Without this information, attempts at statistical manipulations of hydroacoustic data to develop abundance estimates will yield only relative values. In fact, the state of the science is such that even when these procedures are carried out, the variance associated with the best estimate will be from +25% to a factor of two with a probability confidence of 95% (1.4-81).

The second shortcoming of current hydroacoustic research in pelagic stock estimates is that each investigator or group of collaborators finds it necessary to adopt a "new system." Thus, the comparison of results among these various systems is dubious because of the many sources of error which can arise. What is needed are inter-calibration exercises wherein the differences and similarities of the various systems can be ascertained.

The third problem is even more difficult to overcome because it involves economics and investigator prejudices. The recording instrument needed to handle the raw incoming data for frequencies normally used in fresh and salt water is expensive (\$15,000 to \$20,000). Therefore, every investigator mentioned above, with the exception of Yudanov et al. (1.4-81), has altered the incoming signal before it is recorded. The most typical alterations include heterodyning and digitization.

As Yundanov et al. (1.4-81) points out the indiscriminate use of filtering, exponential amplification, frequency shifting and other signal conditioning techniques to simplify handling, recording and/or processing of the received hydroacoustical echo signals may introduce considerable uncertainty in the assessment of the true information content of the data. Subsequent processing of these modified data, without demonstrable verification of fidelity, tacitly assumes that the true nature of all the critical parameters relating to the hydroacoustical environment is accurately known and the final conditioned data retain all the necessary components of information. Further, it is assumed that the knowledge of the accuracy of the critical parameters is considerably better than the expected accuracy of the final result. Orłowski (1.4-55) illustrates a typical result of filtering, exponential amplification, frequency shifting and narrow band recording of a signal of known characteristics.

A joint research effort between the USA and USSR, ongoing since 1974 (1.4-81) has avoided the above problems by the direct recording of the raw hydroacoustical signal. These techniques also have been applied to impact assessment on the Great Lakes (1.4-69).

Finally, when groups of targets are encountered and the returning echos are not resolvable as individuals, then the average target strength of the insonified targets must be known in order to convert the volume backscatter data into abundance estimates.

Attempts at determining target strength have met with limited success because the properties of fish which result in sound reflection are not well understood. For instances, Nakken and Olsen (1.4-82) show that the target strength of fish, approximately 40 cm long with an air bladder, exhibit a variation of nearly a factor of two about a mean value of - 30.5 dB. Since it is speculated that about 80% of the echo from a fish is from the swim bladder, then it follows that changes in swim bladder size or density will affect echo returns.

Fish have been shown to take as long as several days to re-equilibrate the swim bladder once a change in pressure or depth has occurred. In addition, fish moving in a water current continuously readjust swim bladder volume (1.4-83, 1.4-84). Experiments conducted on living fish by Olsen (1.4-85), Nakken and Olsen (1.4-82), Haslett (1.4-86) and Olsen et al. (1.4-87) have shown that the orientation of the fish, such as head up or head down, can cause significant changes in target strength. And by orienting the fish in a variety of directions, Nakken and Olsen (1.4-82) have shown as much as a 5 dB difference. Generally, target strength increases with fish length, although the spread of data is large and depends upon the fishes' preceding behavior. Finally, the echo from groups of restricted fish (in cages) is seen to decrease over time suggesting that a period of time is required for the fish to give a reliable echo (1.4-80).

Love (1.4-88, 1.4-89, 1.4-90, 1.4-91) has conducted considerable experimental work on the maximum side-aspect and dorsal-aspect target strength of anesthetized fish. The maximum side-aspect target strength is needed for quantification when sideways-looking sonar is employed, while the maximum

dorsal-aspect target strength is required for quantification with downward-looking sonar. One of the more important results of measurements of fish target strength to date is discussed below.

It has been determined that practically every case of interest to the fishing industry concerns interference effects. That is, a fish may be considered to be a finite object with acoustic impedance surrounded by a boundary which differs from the medium of water. When sound strikes the fish, some energy is reflected and some is transmitted. After the sound travels through the fish it strikes the second boundary and a portion is transmitted and a portion is reflected. This process continues until a steady state is reached. The reflected sound waves add vectorially, and the amplitude of the returning signal may be increased or decreased depending upon the thickness of the fish and the wave length of the incident wave. If a sphere is used to approximate the process as it occurs with a fish one or more of the following may apply. When the sphere is large compared to the acoustic wavelength, the echo originates from specular reflection and where the sound strikes at right angles to the surface, coherent reflected waves result. When the size of the sphere is similar to the wave length interference effects arise as described above. When the sphere is small compared to a wave length, scattering takes place, and the acoustic cross-section of a sphere of radius a is proportional to a^6/λ^4 where λ is the wave-length. This region is called the region of Rayleigh scattering after Lord Rayleigh who derived the solution.

The reflection of sound from a fish will consist of sound originating from some portion of the above three processes. An additional complication arises when the fish possesses a gas-filled air bladder which also offers an impedance mismatch and may resonate at a particular frequency. When the air bladder resonates the amplitude of the echo will be greatly increased over that predicted with Rayleigh scattering. Figure 1.4-14 presents the theoretical relationship between the ratios of acoustic backscattering to geometric cross-sections and of the circumference of the target to the wavelength. The relationship is linear on a log-log plot up to the point when the circumference and wavelength are near to, or are equal. In this region, the resultant echo from a sphere depends on the precise relationship between the circumference and wave length, and, for a fish, this result cannot be predicted a priori (1.4-90). After reviewing most of the literature dealing with the measurement of fish target strength, Love (1.4-90) found that most of the work is conducted

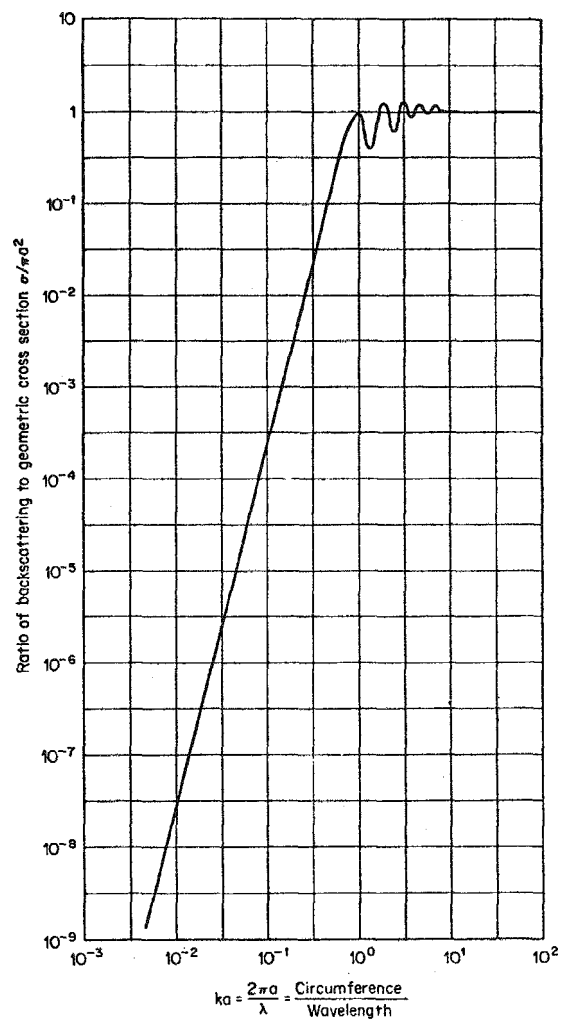


Figure 1.4-14. Ratio of acoustic to geometric cross sections of a fixed, rigid sphere of radius a . The target strength equals $10 \log 4 \pi \sigma$ (from 1.4-47).

in this region (i.e., fish of sizes from 10 cm to 150 cm and sonar frequencies from 10 kHz to 200 kHz).

1.4.6.5.4 The role of hydroacoustics in impact assessment. Hydroacoustics has application to the impact assessment process in providing data on the distribution and abundance of source waterbody populations. In the establishment of boundary conditions for some species, hydroacoustics will permit a rapid means of studying distribution and density. Limits on distribution and natural gradients in density could be used along with other population data to define the limits of a study area.

Although density estimates may have wide confidence intervals, they could provide an initial estimate of stock size that would be input to study designs. An estimate of stock size and some knowledge of distribution and behavior would permit an estimate of the level of effort required to perform more detailed population studies. For example, if a mark/recapture population estimate is contemplated, the initial density estimate from hydroacoustics would provide the initial estimate of population size needed for developing a schedule of marking and recapture, which has a major bearing on the use of such studies (see Section 1.4.1). In addition, the distribution and density data would indicate how to stratify the sampling area for greatest efficiency. This early indication of the magnitude and distribution of the stock may show that some population studies would not be feasible within a given budget and time schedule.

Estimates of absolute abundance by a catch-per-effort technique require a knowledge of the availability of the stock to the fishery gear (see Section 1.4.2). By defining the distribution of the stock of interest, the catch/effort data can be corrected for that portion of the stock unavailable to the sampling gear.

As pointed out in Section 1.3 it is difficult to estimate a potential impingement rate at a proposed intake site. Studies at existing intakes have established that the dynamics of distribution and abundance are important factors in explaining the magnitude of impingement at many sites. Hydroacoustic gear could be employed to survey fish distribution and abundance in the near vicinity of proposed intake sites (1.4-69). These data

may permit refinement of impingement estimates based on changing diel and seasonal patterns in distribution and abundance.

Hydroacoustics can be used to determine the limits of distribution for a population within a water body or segment of the water body of interest. In very large bodies, a specific sampling design is needed to assure adequate coverage and nonbiased sampling. If proper procedures are followed as outlined herein, then it is also possible to derive density estimates, although the confidence intervals may be quite wide in relation to the mean, depending on the operating characteristics of the instrumentation, the spacing of the fish, and the fishes' behavior both before and during insonification.

Studies in the vicinity of operating steam electric generating stations have focused on determining the distribution and abundance of fish in relation to the discharge of heated water. For instance, Kelso and Minns (1.4-92) investigated the pelagic fishes in the vicinity of the Pickering Generating Station in Lake Ontario. They used an automated digitizer to convert the output of the transducer terminals into a digital count. Unfortunately, because instrumentation was not calibrated, their results are of limited value. Zankel et al. (1.4-93) used hydroacoustics to examine the size and distribution of the 1974 striped bass spawning stock in the Potomac River in conjunction with entrainment and impingement surveys at the Morgantown Steam Electric Station. They estimated that the spawning stock of bass was approximately one million fish. Two technical problems are apparent in their procedures. First, they heterodyned the hydroacoustic signal from 105 kHz to 7 kHz without presenting any information on what effect this had on the information content of the data. Second, an amplifier drift may have caused errors in individual signals of as much as 1.5 dB or a factor of 1.4. Considering these shortcomings, the estimated abundance of striped bass is likely to vary by a factor of at least 1.4.

The applicability of hydroacoustics to impact assessment can be best illustrated with an example. Wyman et al. (1.4-69) used hydroacoustic methods in assessing the distribution and abundance of fishes around three power plants on Lake Ontario. These researchers used methods developed by Yudanov et al. (1.4-81) and included rigorous instrumentation calibration in the field, and the recording of the raw received output of the transducer at real time.

Hydroacoustical instrumentation was used in the analysis of the effects of the type of discharge from steam electric power plants on the distribution and abundance of Great Lakes fishes. Canadian power plants utilize onshore discharge canals, while some U.S. plants use offshore submerged diffusers. Ontario Hydro was interested in determining if the effects of these discharges on fish distribution and abundance could be measured and if a difference was detected between discharge and control areas and if the effect was the same for the two discharge types. In addition, because the Lennox Power Plant in the Bay of Quinte operates in the vicinity of a yellow perch spawning ground, it was important to estimate the density of these fishes.

Three power plants on the shores of Lake Ontario were chosen for these studies; the Oswego Steam Station, Oswego, New York; the Lennox Steam Station, Bath, Ontario; and the Pickering Nuclear Power Plant in Pickering, Ontario. Experimental and control areas were established at each plant. Two types of hydroacoustic surveys were conducted at each plant during the day and night, monthly from April through November 1977. The first was a series of eight 1.6-km long transects perpendicular to shore (Figure 1.4-15). Five of these transects were centered on the discharge in the experimental area and three were in the control area. The second survey was a series of four, 3-min long transects (boat speed approximately 2 m sec^{-1}) parallel to shore conducted directly over the discharge and along similar depth contours in the control areas. The hydroacoustic gear employed was standard electrical engineering instrumentation and a Honeywell 5600-C instrumentation tape recorder as recommended by Yudanov et al. (1.4-81). Three kinds of transducers were used (Figure 1.4-16). A wide beam (40 degree half-power beam width) was used to survey the shallow water along the transects parallel to shore, a narrower beam (17 x 25 degree half-power beam width) was used along the transects perpendicular to shore, and an even narrower beam was used to determine the target strength of individual fish.

Full calibration procedures were conducted throughout the survey period and these resulted in a 95% confidence interval statistic of from 35 to 75%.

$$\text{Confidence Interval Statistic} = \frac{\bar{X}}{\frac{\text{S.D.} \times \text{Z score}}{\sqrt{N}}}$$

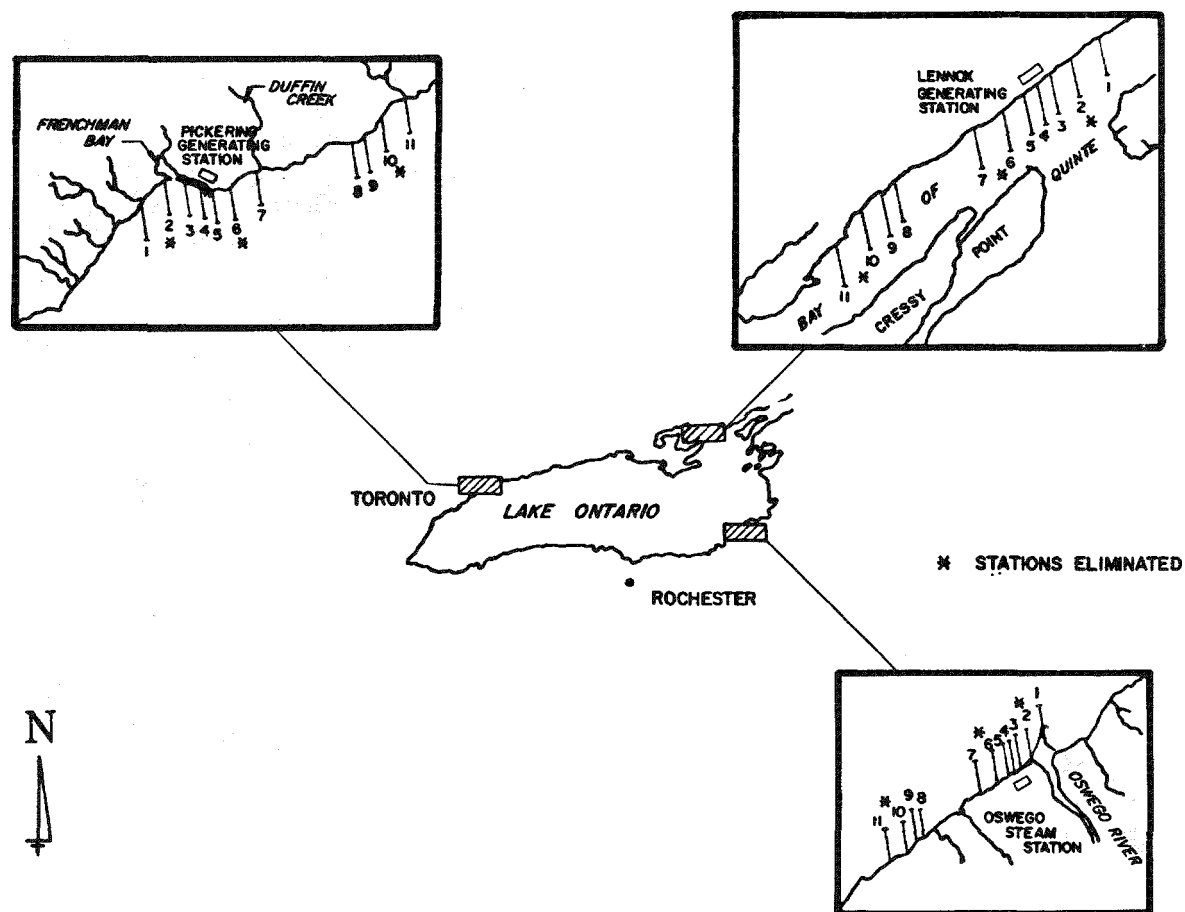


Figure 1.4-15. Location of the Hydroacoustic Transects at the Three Power Plants.

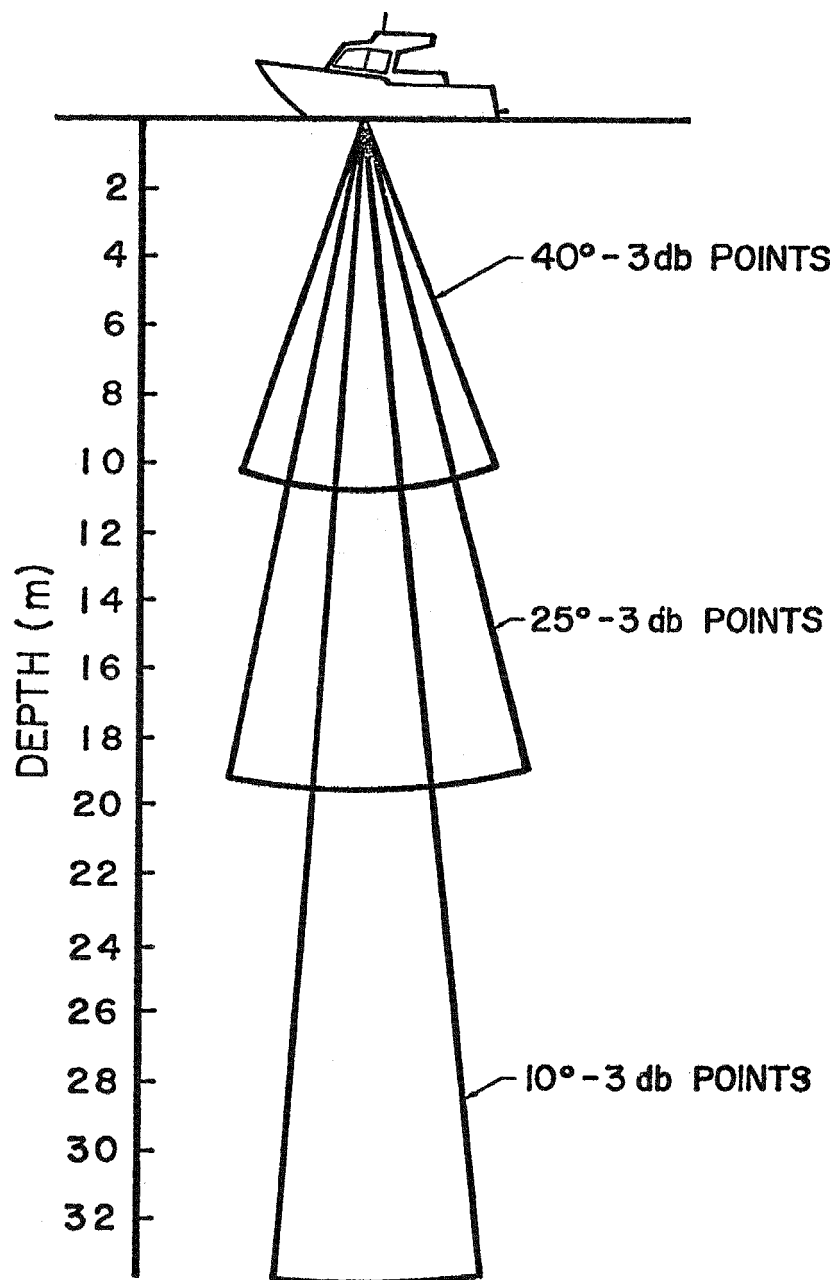


Figure 1.4-16. Beam angles of transducers used in Lake Ontario studies.

where: $\bar{X}$ - mean of sample values
 S.D. - standard deviation
 Z - Z score for 95% of normal distribution
 N - number of samples

That is, the variance associated with the operation of the equipment is between 35 and 75% of the mean.

In gill netting surveys conducted in Lake Ontario with approximately 128, 12-hr net sets per month, the confidence interval was from 120 to 200% of the mean. Thus, the hydroacoustic method yields a more precise confidence interval with less effort. Netting, however, is still necessary for species and size identifications.

During the survey period, fish were collected at the nearshore end of each perpendicular transect with trap nets and in the vicinity of the parallel transects with gill nets. In addition, underwater observations were made in the vicinity of the parallel transects during the day and night by divers. These data were used to confirm the species identification and size of fish present.

Three kinds of data resulted from the hydroacoustics survey. First, single targets were identifiable as sound pulses of the same pulse duration as the sound emitted by the transducer (i.e., 10 to 13 cycles in 0.25 msec). These data were simply counted by depth-time intervals for each transect. In addition the density of fish based on the single target data was calculated for each power plant by month. Second, multiple targets were those whose sound envelope was of longer duration than 13 cycles and 0.25 msec. Multiple target data were converted into a volume backscatter parameter according to the following formula:

$$VBP = \frac{R^2}{\delta} V_{pp}^2 \quad (1.4-76)$$

where R = range to the center of the aggregation
 V_{pp} = peak-to-peak voltage returned from the aggregation
 δ = depth of the aggregation, i.e., the distance from the top of the aggregation to its bottom.

This was done because only relative comparisons were to be made between control and experimental transects and among power plants, and hence, all instrumentation constants were dropped from Eq. 1.4-75. These data were summed per transect and statistical tests were applied to the summed transect volume backscatter parameter.

During the survey period (April through November 1977), the Oswego plant operated at approximately 50% of maximum output, Lennox varied between 1 and 15% while Pickering operated near full capacity.

Graphs of the average numbers of single targets observed per transect for control and experimental areas and the average of the volume backscatter parameter are shown in Figures 1.4-17 through 1.4-19. These data are presented as a comparison of single target and volume backscatter during the day and night. The resulting picture is a complex one. It should be remembered that it is the behavior of the fishes which determines the relationship between single target abundance and volume backscatter. That is, when the fish are tightly grouped and deep in the water, volume backscatter will dominate. Conversely when fish are loosely grouped or near to the surface, single targets will dominate. In between these two extremes, a range of relationships will occur.

There are several consistent trends among power plants (Figures 1.4-17 through 1.4-19). First, nighttime surveys revealed two to ten times the number of single targets as did daytime surveys. Secondly, during the day at each plant single targets ranged between 25 and 100 per transect for April through June. During August, September and October and during both the day and night, single target abundance increased, probably because of the recruitment of young-of-the-year fishes to the hydroacoustic gear. At Oswego, scuba divers observed schools of young alewives and rainbow smelt during these months (Figure 1.4-20). At Lennox, yellow perch were observed by divers during the day and night. During the day, yellow perch were observed singly or in schools from one to several meters from the bottom. At night, the yellow perch were on or near the bottom and motionless. As with Oswego, schools of alewives and rainbow smelt were observed at Lennox and Pickering during the night (Figure 1.4-21 and 1.4-22). The third pattern that emerges is that volume backscatter, i.e., the sonic energy returning from aggregated fishes, was low or nonexistent for all

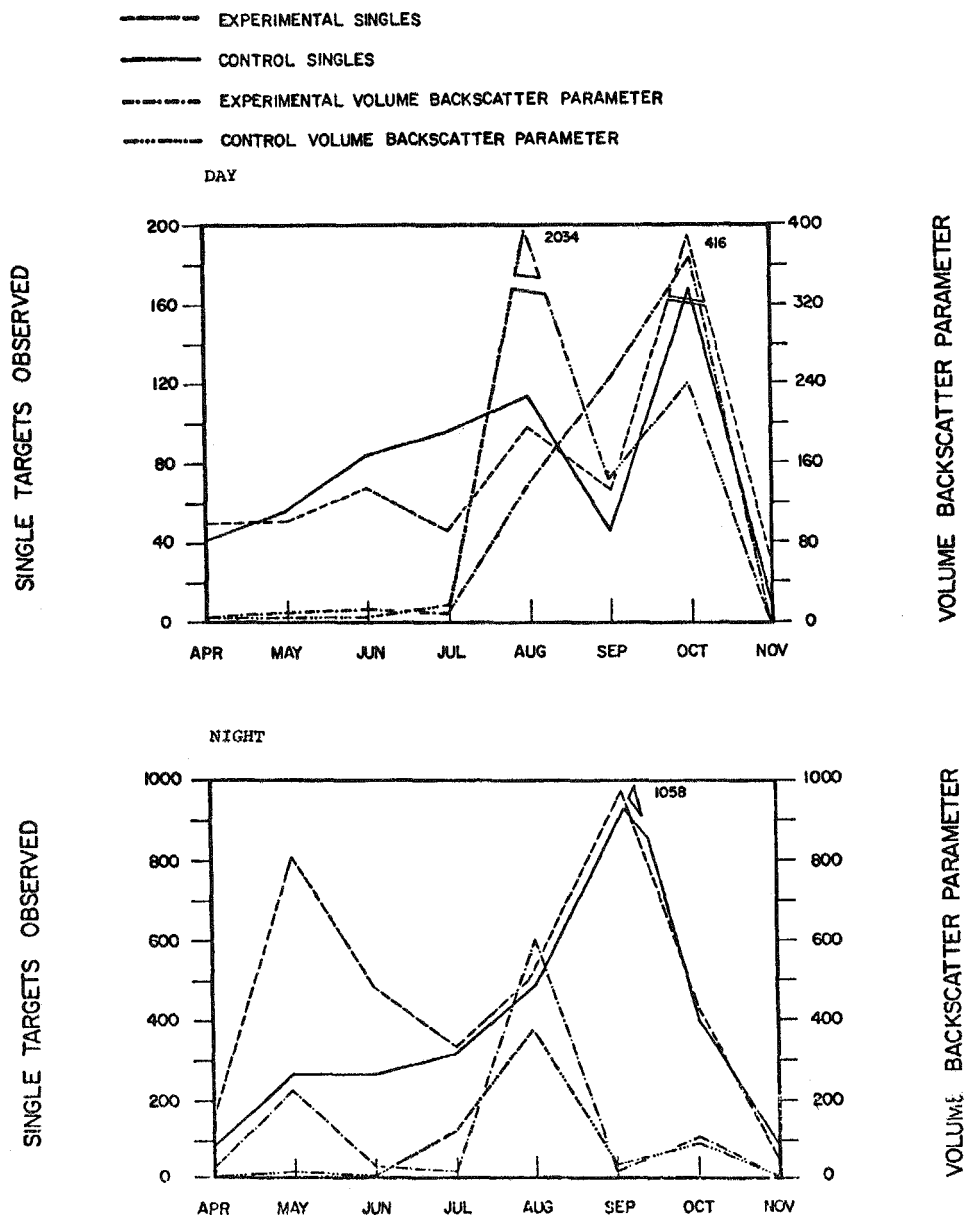


Figure 1.4-17. Summary of single targets and volume backscatter parameters by month for control and experimental areas at the Oswego Power Plant, 1977.

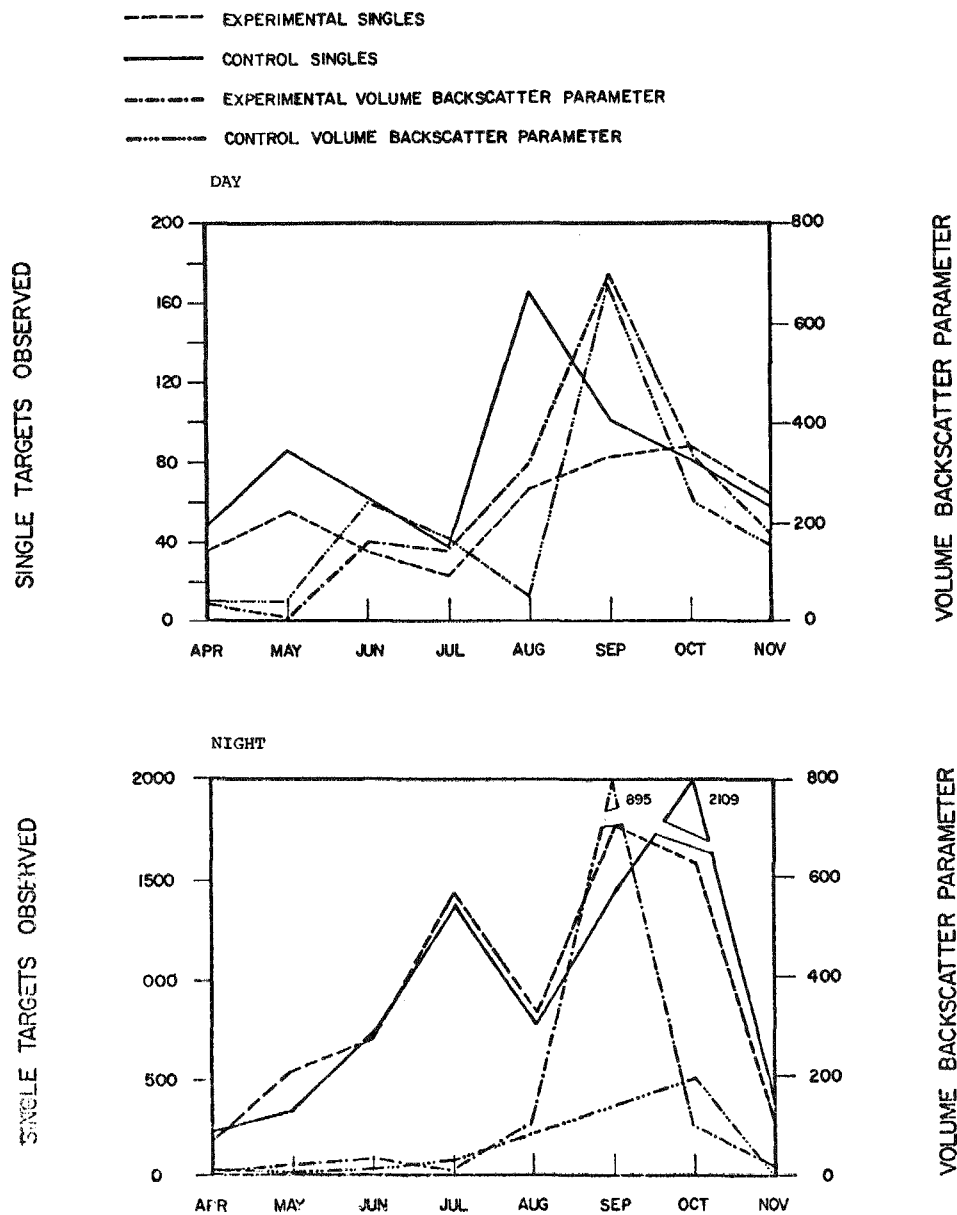


Figure 1.4-18. Summary of single targets and volume backscatter parameters by month for control and experimental areas at the Lennox Power Plant, 1977.

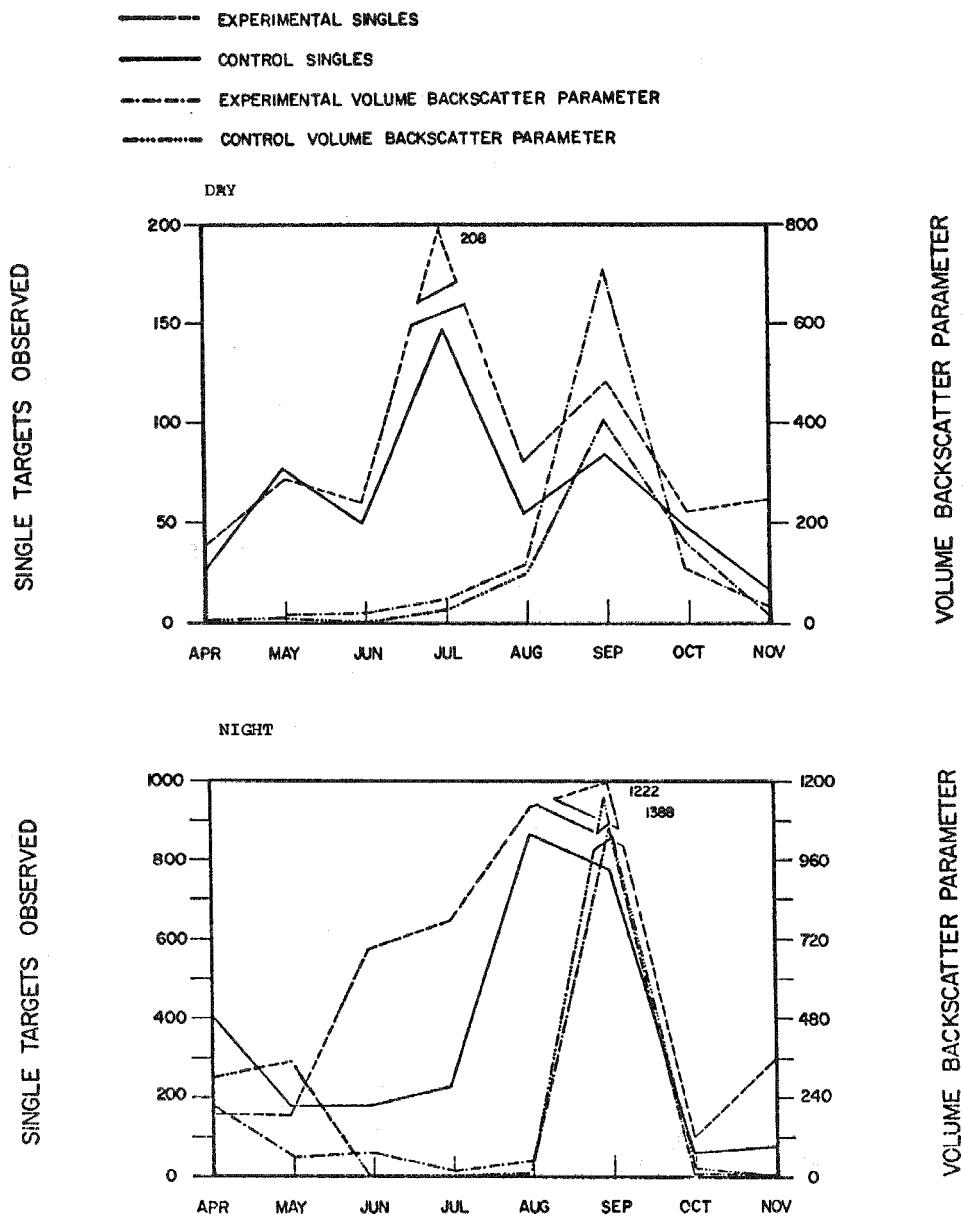


Figure 1.4-19. Summary of single targets and volume backscatter parameters by month for control and experimental areas at the Pickering Power Plant, 1977.

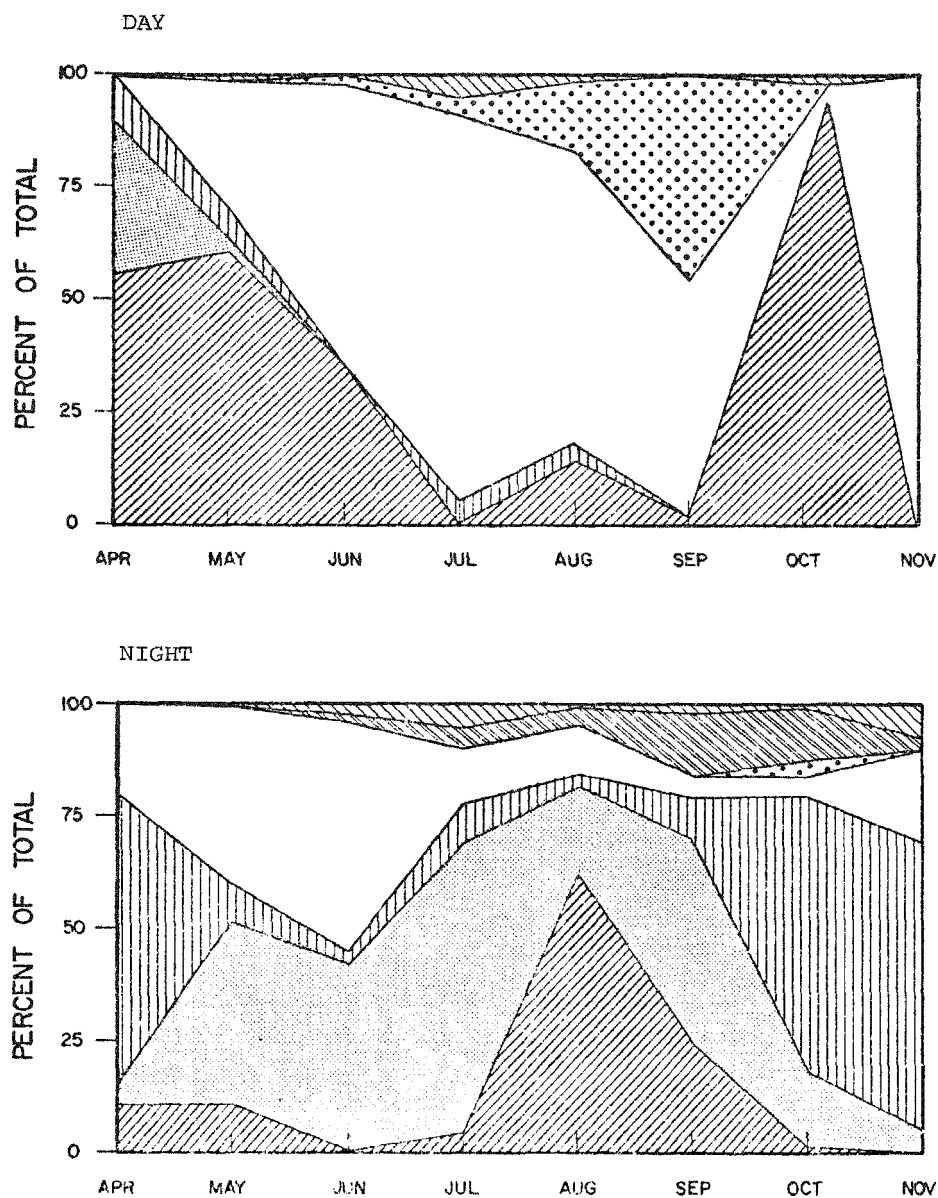


Figure 1.4-20. Fish community composition based on scuba diver observations at the Oswego Power Plant, 1977.

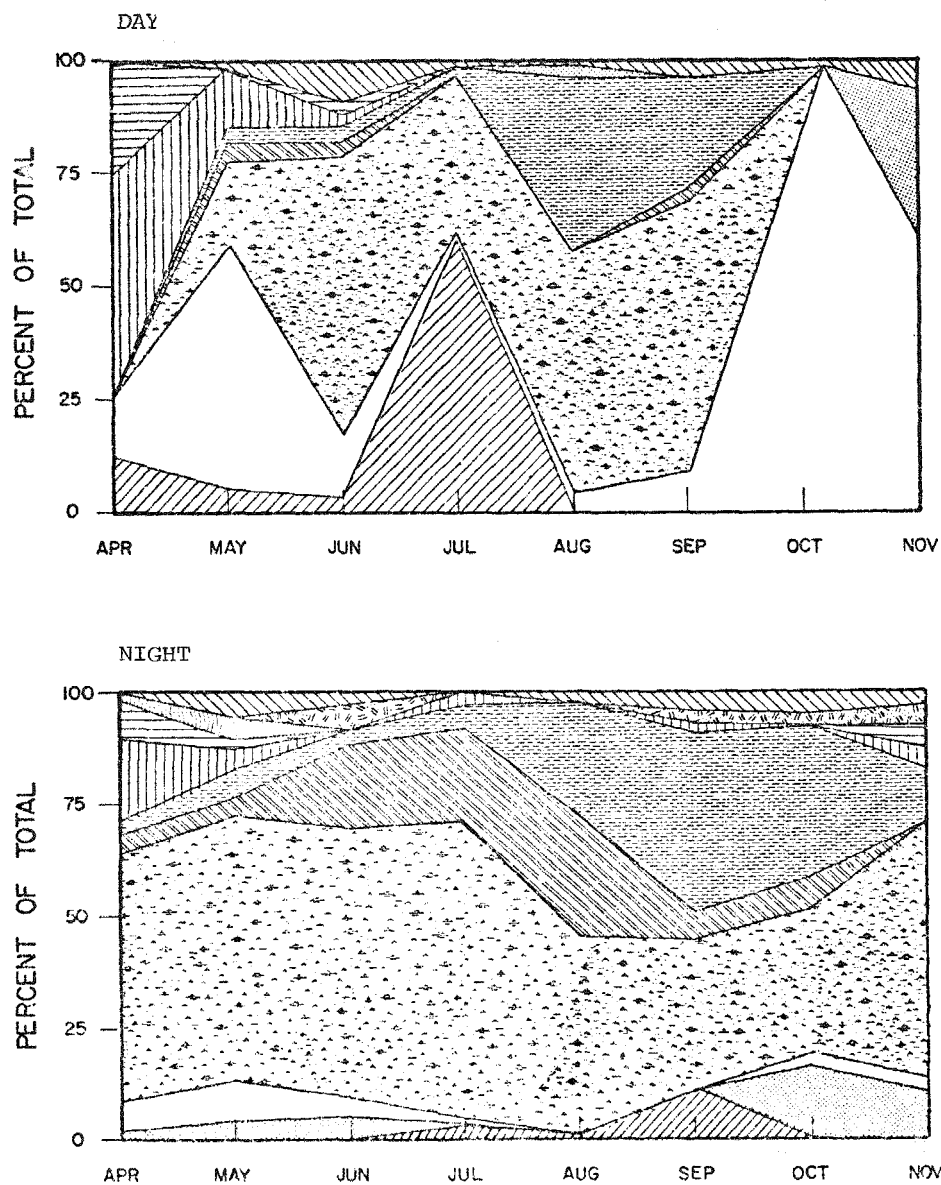


Figure 1.4-21. Fish community composition based on scuba diver observations at the Lennox Power Plant, 1977.

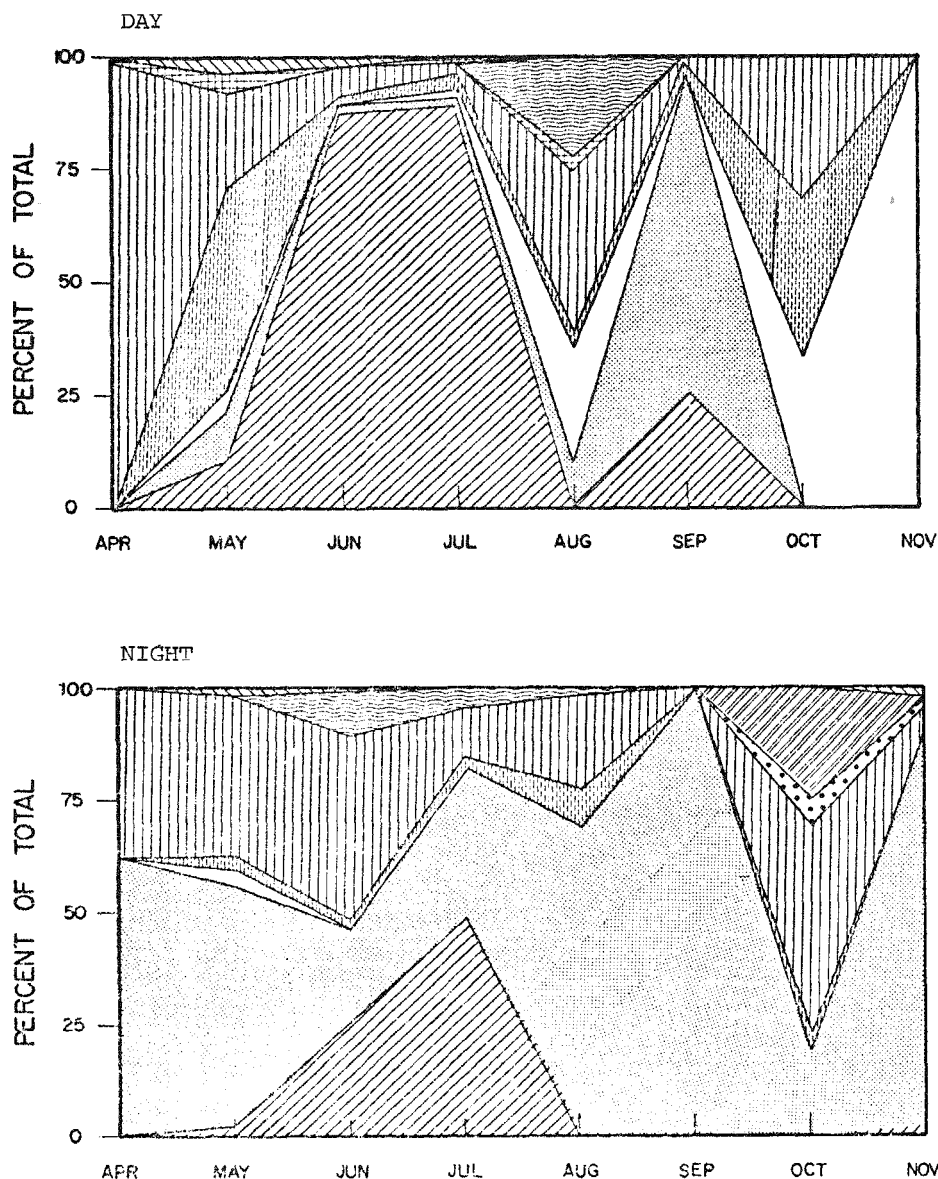


Figure 1.4-22. Fish community composition based on scuba diver observations at the Pickering Power Plant, 1977.

plants during the day and night from April to June. After June the volume backscatter gradually increased and peaked from August to October. The exceptions were during the night in April and May at Pickering and during the night in May at Oswego. During these months, spawning rainbow smelt were observed at night at Pickering and spawning alewives at Oswego; hence, the volume backscatter observed during these times can be attributed to schools of aggregated rainbow smelt or alewives. The remaining peaks of volume backscatter all occurred between August and October and coincided with the appearance of young-of-the year alewives and rainbow smelt at each site.

Three-way analyses of variances (ANOVA) were conducted separately for day and night on the single target data collected along the perpendicular transects and were compared between experimental control areas, among the plants, and among sampling months to determine whether the abundance of fish was different between control and experimental areas (Table 1.4-13).

During the day, there were two significant interactions, one among power plants and experimental and control areas and the second among power plant and night comparisons. Follow-up, two-way ANOVAS were used to determine the cause of these interactions. The first interaction is attributed to the observation at Lennox of significantly more fish in the control area than in the experimental area in the daytime; no such daytime difference was detected for the other two power plants.

The second interaction, among power plants and months, was ascribed to the occurrence of peak abundance during different months at the three power plants. At Oswego, peak daytime abundance occurred during October, at Pickering it took place during July, and at Lennox during August.

During the night, a significant three-way interaction occurred, i. e., plants differed significantly in fish abundance observed. Significantly greater abundances were noted in either the experimental area or in the control area depending on plant and month (Table 1.4-14).

Two significant differences were found when the control and experimental areas were compared for the day and night for each power plant. During the day at Lennox, significantly more targets were observed in the control area

Table 1.4-13

STATISTICAL ANALYSIS OF HYDROACOUSTIC SURVEYS
 ALONG PERPENDICULAR TRANSECTS DURING THE DAY
 AT THE THREE POWER PLANTS, 1977

Analysis of Variance (Log Transformed)				
Source	d.f.	SS	MS	F
Plant	2	0.087 144	0.043 572	0.945 NS
Exp/Cont	1	0.000 206	0.000 206	0.004 NS
Months	7	4.997 362	0.713 909	15.482 **
Plant x Exp/Cont	2	0.572 637	0.286 319	6.209 **
Plant x Month	14	5.655 040	0.403 931	8.760 **
Exp/Cont x Month	7	0.527 221	0.075 317	1.633 NS
Plant x Exp/Cont x Month	14	0.602 088	0.043 006	0.933 NS
Error	144	6.640 086	0.046 112	
Total	191	19.983 024		

NS - Not significant

** - Significant at $\alpha = .01$

Table 1.4-14

STATISTICAL ANALYSIS OF HYDROACOUSTIC SURVEYS
ALONG PERPENDICULAR TRANSECTS DURING THE NIGHT
AT THE THREE POWER PLANTS, 1977

Analysis of Variance (Log Transformed)				
Source	d.f.	SS	MS	F
Plant	2	6.353 773	3.176 887	79.361 **
Exp/Cont	1	0.236 284	0.236 284	5.903 *
Months	7	14.239 400	2.034 200	50.816 **
Plant x Exp/Cont	2	0.205 74	0.102 872	2.570 NS
Plant x Month	14	8.190 611	0.585 044	14.615 **
Exp/Cont x Month	7	0.436 612	0.062 373	1.558 NS
Plant x Exp/Cont x Month	14	1.388 450	0.099 175	2.477 **
Error	144	5.764 465	0.040 031	
Total	191	38.677 532		

Bonferroni Procedure ($\alpha = 0.05$)

Plant: Largest Lennox Oswego Pickering Smallest

* - Significant at $\alpha = .05$

** - Significant at $\alpha = .01$

NS - Not significant

than in the experimental area. On the other hand, at night, more targets were observed in the experimental area than in the control area at Pickering. The difference at Lennox was primarily because of the presence of a large number of yellow perch spawning west of the plant in May and June and subsequent recruitment of young yellow perch of sufficient size to be sensed by the hydroacoustic gear during September and October. The difference at Pickering occurred during June and July when alewives were spawning.

These perpendicular transects were arranged to cover a wide area in the vicinity of the power plants. In general, consistent differences between control and experimental transects in the distribution of individual fishes attributed to the presence of the discharge were not identified. An examination of only transect 4, the discharge transect, identified several noteworthy events. At both Pickering and Oswego, at night, during May to June and September to October, more fish were observed near the discharge than in the control area suggesting that during the spring and fall, more fish reside near the plant discharge than in the control areas, and during the summer these fishes withdraw from the discharge areas.

The second parameter, multiple targets, calculated from the hydroacoustic surveys along the perpendicular transect was a volume backscatter parameter (VBP). For each group of targets observed, the VBP was calculated, and these results were summed up for each transect. This parameter is equivalent to a relative integrated sonic energy returning from grouped fishes. Four comparisons of this parameter were made at each plant. These were 1) the mean of the control transects was compared to the mean of the experimental transects for day surveys; 2) the means of the transects were compared for night; 3) the discharge transect (transect 4) VBP was compared with the average control value for day; and 4) the discharge VBP was compared to the control area for night.

These analyses were conducted on log transformed data and with t-tests for means (Table 1.4-15). No significant differences in fish aggregations as measured by VBP were detected among any of the comparisons. It should be remembered that the volume backscatter data appeared to represent mainly young-of-the-year fishes, especially from August to October, and that the individual target data better represent the distribution and abundance of larger fishes.

Table 1.4-15

COMPARISON OF THE VOLUME BACKSCATTER PARAMETER BETWEEN
CONTROL AND EXPERIMENTAL AREAS AND BETWEEN
THE CONTROL AREA AND THE DISCHARGE TRANSECT

Location	Area	Day			Night		
		Mean	S.D.	t-value	Mean	S.D.	t-value
Oswego	Control	1.145	1.254		1.265	0.912	
	Experimental	0.930	1.239	0.276	1.572	0.849	0.554
	Discharge	0.948	0.993	0.277	1.587	1.005	0.530
Lennox	Control	2.045	0.473		1.329	1.390	
	Experimental	1.999	0.703	0.121	0.617	0.980	0.118
	Discharge	1.916	0.933	0.276	0.906	1.075	0.764
Pickering	Control	1.136	0.944		1.513	1.262	
	Experimental	1.316	1.017	0.290	1.929	0.579	0.671
	Discharge	0.802	1.177	0.498	1.692	0.960	0.256

Thus, these analyses seem to indicate that the distribution of young-of-the-year fishes in the far-field are not perturbed by a thermal discharge.

The single target data for the day and night survey conducted along the four transects parallel to shore in the immediate vicinity of the discharge and in control areas were compared with t-tests of means (Figure 1.4-23). Data from these transects give the near-field distribution of fishes in the waters around the discharges, while the data from the perpendicular transects present a far-field view.

At Oswego and Pickering both day and night surveys revealed significantly more single targets in the vicinity of the discharge than in the respective control areas. At Lennox this difference was only detected during the night.

That the parallel and not the perpendicular transects revealed a significant difference in the abundance of fish is probably attributable to the survey methodology. The parallel transect surveys were conducted with the broad-beam transducers (i.e, 40 degree half-power beam width) while the perpendicular transects were surveyed with the narrow beam transducer (17 x 25 degrees). The broad beam transducer covers a wide area more effectively in shallow water. Thus, the parallel transects which ran sideways through the flowing Pickering discharge were of sufficient resolution to detect the differences in abundance in the near field, and the perpendicular transects were not. It appears that in the nearshore, the data illustrate the diel behavior of the littoral fishes, while the offshore data represent the diel behavior of the pelagic fishes.

The depth distribution of the single targets observed with hydroacoustics was plotted for three months for day and night and for averages of all transects in control and experimental areas and for the discharge transect 4 alone (Figure 1.4-24). These data were compared statistically and in general showed that near the discharges of the two active plants, the fish were distributed in deeper water than in areas away from the discharge.

The last distributional pattern documented with the hydroacoustic method was the location of fish as a function of distance offshore during the day and

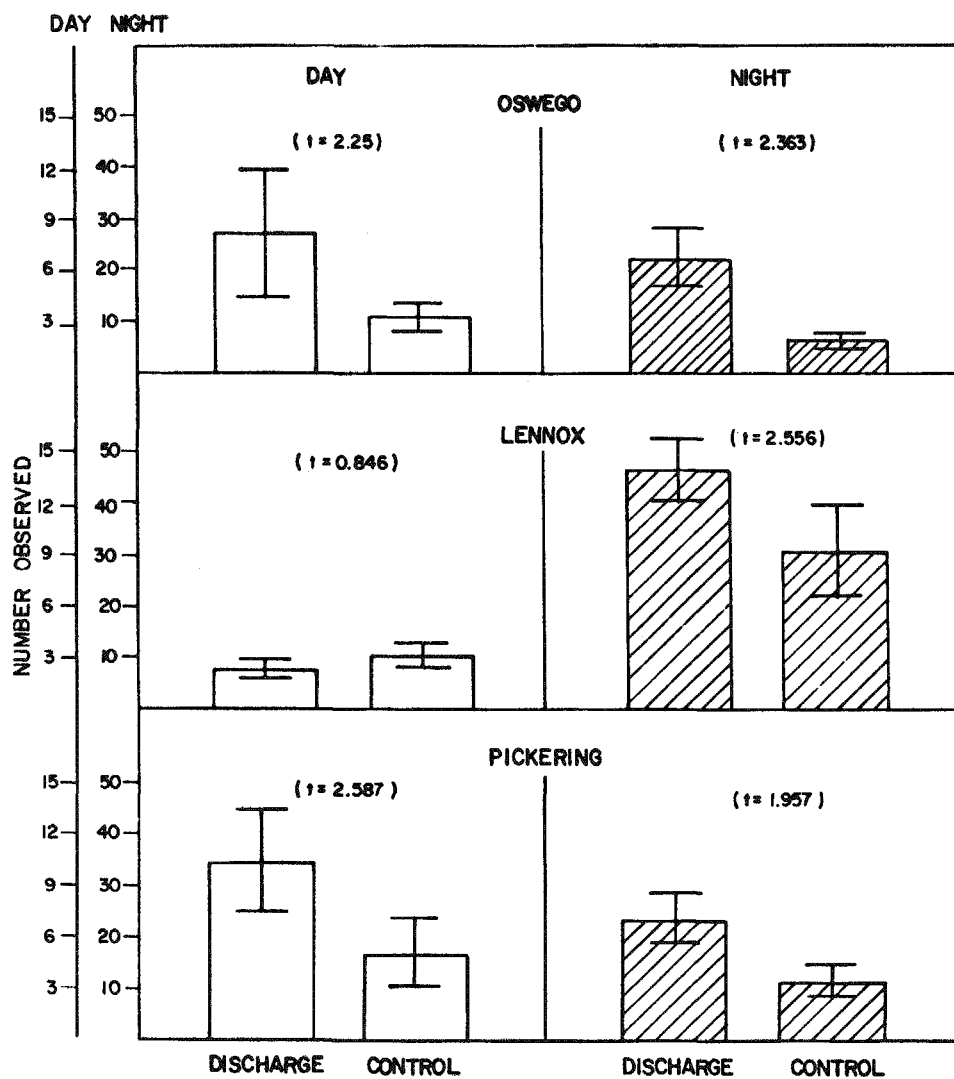


Figure 1.4-23. Total number of individuals observed by scuba divers along parallel transects at the three power plants, 1977.

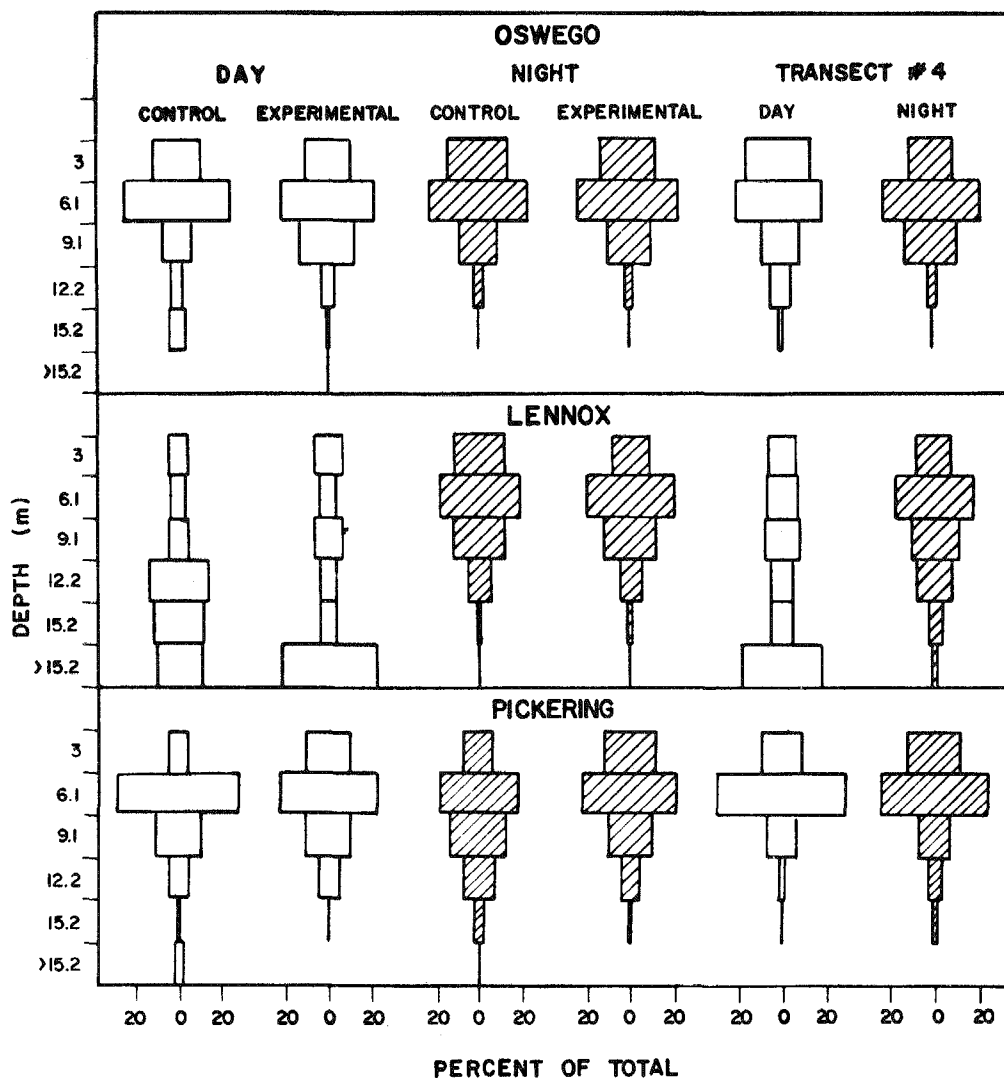


Figure 1.4-24. Depth distribution of single targets at the three power plants, April-June, 1977.

night in control and experimental areas (Figure 1.4-25 through 1.4-27). The same general trend of greater numbers of fish at night compared to the day is present at each plant. Individual plant differences, however, were present when comparisons were made between control and experimental areas. At Oswego, the distribution of fish is skewed toward shore in the experimental area during both day and night (Figure 1.4-25), whereas in the control area, this is not the case. During the day, at Oswego, the greatest percentage of fish was observed in the distance offshore interval corresponding to the location of the discharge.

At Lennox, only the night data in the experimental area suggest that the fish distribution is skewed toward shore (Figure 1.4-26). Little difference is detectable between control and experimental areas during the day. During most of the year, the Lennox Power Plant was discharging a relatively small quantity of water.

The Pickering data do not reveal a dramatic difference in the distribution of fish between control and experimental areas (Figure 1.4-27), although during the day there is a slight tendency for more fish to be located nearer shore in the experimental area than in the control area.

Thus, it is only at Oswego that the distribution of fish appears to have been different between control and experimental areas, and this difference was most pronounced in the interval within which the discharge is located.

An estimate of the density of fish per 1000 m³ was made for control and experimental areas at each plant to illustrate the usefulness of hydroacoustics in assessing 1) the potential for impingement losses at proposed sites and 2) the size of resident fish populations for comparison to impingement and entrainment cropping.

Single target data for each transect were converted to numbers per 1000 m³ by dividing by the volume of water insonified during the hydroacoustic surveys (Tables 1.4-16 through 1.4-18). These data were compared to the gill net catch-per-effort data for the same time period, and a ratio of gill net catch effort⁻¹ to the hydroacoustic density estimate was calculated for each plant (Table 1.4-19).

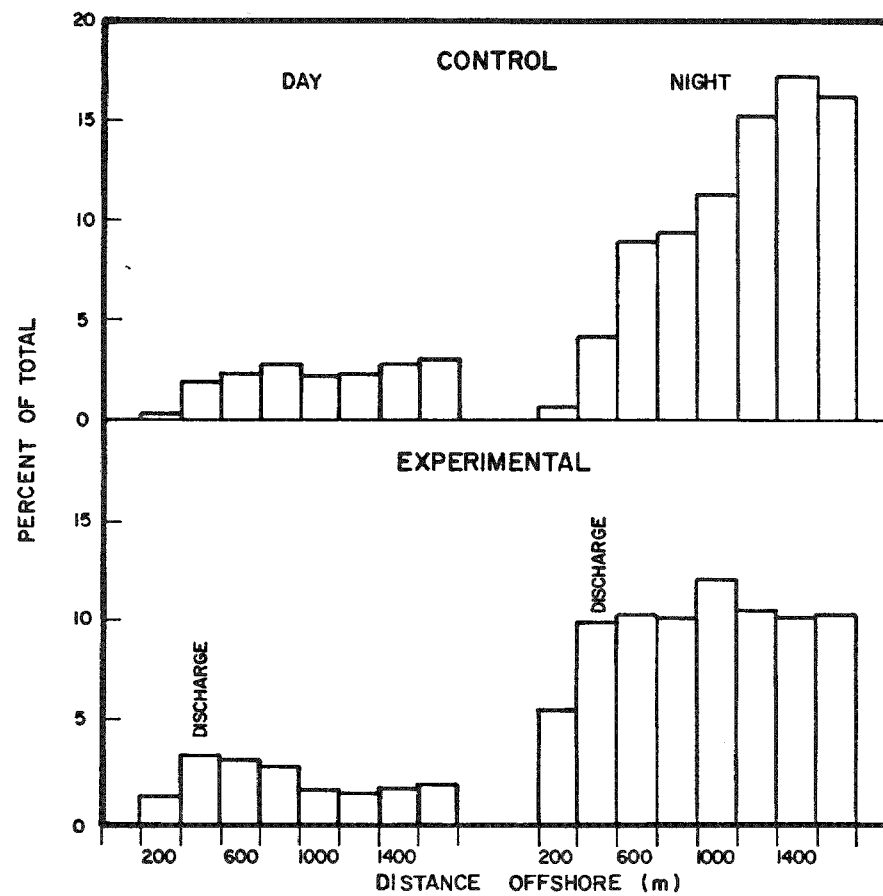


Figure 1.4-25. Comparison of the distribution of fish as a function of distance offshore between the control and experimental areas at the Oswego Power Plant, 1977.

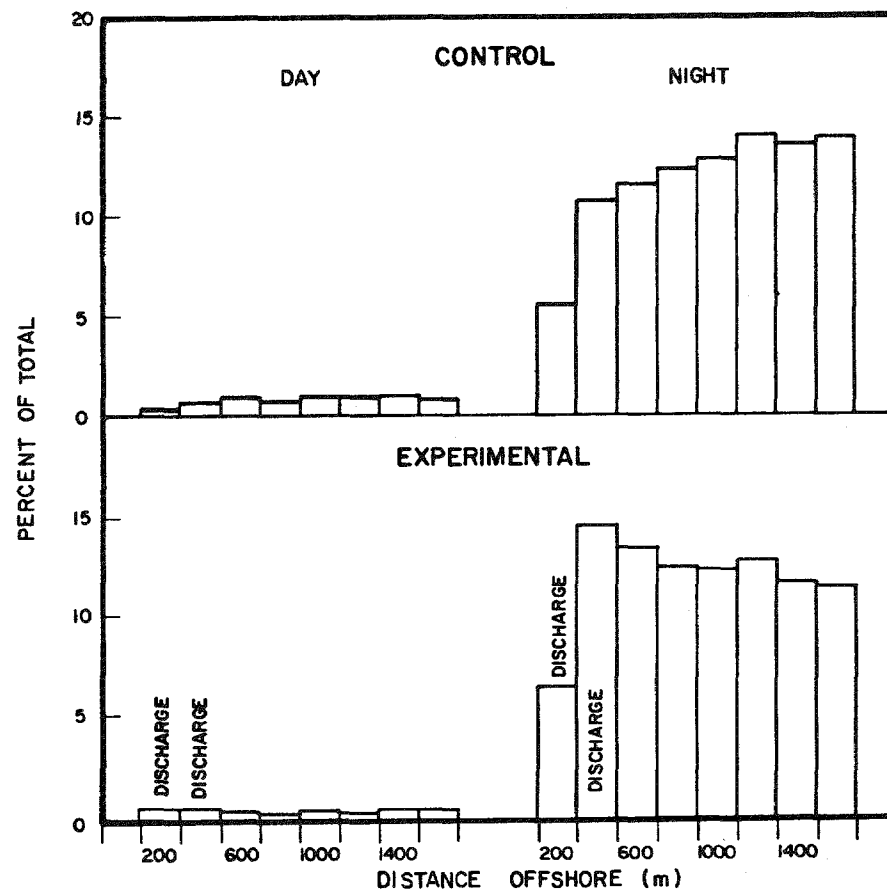


Figure 1.4-26. Comparison of the distribution of fish as a function of distance offshore between the control and experimental areas at the Lennox Power Plant, 1977.

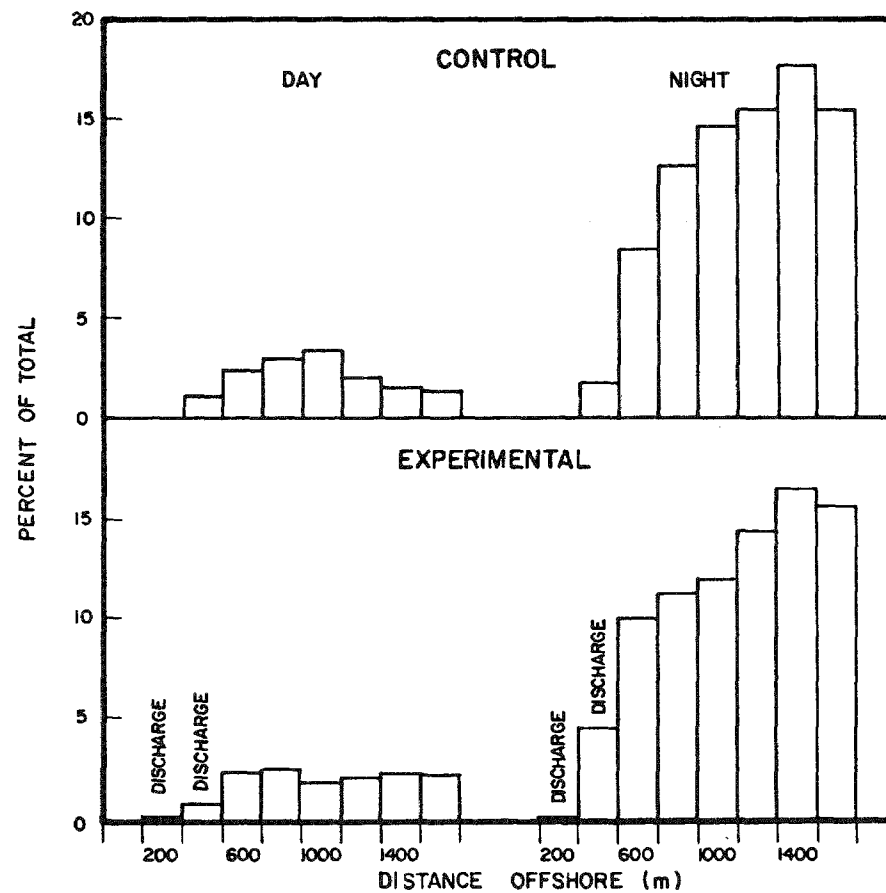


Figure 1.4-27. Comparison of the distribution of fish as a function of distance offshore between the control and experimental areas at the Pickering Power Plant, 1977.

Table 1.4-16

SUMMARY OF THE ESTIMATED ABUNDANCE OF FISH*
 BASED ON HYDROACOUSTIC SURVEYS IN THE OSWEGO STEAM STATION VICINITY, 1977

Month	Control		Experimental	
	Day	Night	Day	Night
Apr	1.03 (0.24)	1.88 (0.56)	1.24 (0.73)	3.24 (2.07)
May	1.41 (0.10)	6.56 (0.48)	1.29 (0.36)	20.02 (13.11)
Jun	2.09 (0.77)	6.48 (0.86)	1.71 (0.36)	12.16 (3.61)
Jul	2.38 (0.92)	7.75 (2.17)	1.17 (0.42)	7.86 (2.39)
Aug	2.86 (0.77)	11.98 (8.05)	2.43 (0.69)	12.27 (4.87)
Sep	1.16 (0.24)	26.45 (11.82)	1.66 (1.10)	24.66 (4.39)
Oct	4.22 (2.45)	9.80 (4.71)	10.41 (4.63)	10.95 (3.45)
Nov	0.33 (0.06)	1.91 (0.45)	0.79 (0.23)	1.21 (0.51)
Mean	1.94 (1.15)	9.10 (7.32)	2.59 (2.99)	11.54 (7.37)

*Number of fish per 1000 m³; numbers within parentheses indicate standard deviation.

Table 1.4-17

SUMMARY OF THE ESTIMATED ABUNDANCE OF FISH*
 BASED ON HYDROACOUSTIC SURVEYS IN THE LENNOX GENERATING STATION VICINITY, 1977

Month	Control		Experimental	
	Day	Night	Day	Night
Apr	1.23 (0.24)	5.77 (0.93)	0.88 (0.28)	4.59 (1.26)
May	2.17 (1.11)	8.53 (1.78)	1.40 (1.00)	12.93 (6.12)
Jun	1.56 (0.29)	18.16 (3.05)	0.88 (0.18)	17.63 (7.68)
Jul	0.97 (0.59)	34.13 (0.20)	0.61 (0.11)	16.24 (3.93)
Aug	4.41 (0.59)	19.48 (6.71)	1.69 (0.91)	19.60 (12.94)
Sep	2.58 (1.24)	36.01 (3.53)	2.09 (0.64)	44.21 (13.10)
Oct	2.03 (0.28)	52.73 (3.81)	2.23 (0.25)	39.55 (10.38)
Nov	1.48 (0.19)	9.98 (5.15)	1.54 (0.80)	7.93 (0.86)
Mean	2.02 (0.94)	23.09 (15.36)	1.42 (0.55)	20.34 (13.33)

*Number of fish per 1000 m³; numbers within parentheses indicate standard deviation.

Table 1.4-18

SUMMARY OF THE ESTIMATED ABUNDANCE OF FISH*
 BASED ON HYDROACOUSTIC SURVEYS IN THE PICKERING GENERATING STATION VICINITY, 1977

Month	Control		Experimental	
	Day	Night	Day	Night
Apr	0.68 (0.36)	10.13 (3.50)	0.97 (0.47)	4.11 (1.45)
May	1.98 (0.87)	4.36 (0.58)	2.32 (0.75)	3.92 (2.62)
Jun	1.30 (0.28)	4.26 (0.53)	1.52 (0.53)	14.35 (7.68)
Jul	3.68 (2.13)	5.73 (0.65)	5.22 (1.27)	16.24 (3.93)
Aug	1.39 (0.26)	21.69 (3.37)	2.01 (0.66)	23.27 (9.22)
Sep	2.13 (0.40)	19.67 (3.93)	3.08 (1.36)	30.57 (7.46)
Oct	1.21 (0.57)	1.42 (0.38)	1.40 (0.67)	2.57 (1.44)
Nov	0.43 (0.16)	2.20 (0.38)	1.57 (2.47)	7.36 (5.31)
Mean	1.60 (0.95)	8.68 (7.36)	2.26 (1.27)	12.80 (9.54)

*Number of fish per 1000 m³; numbers within parentheses indicate standard deviation.

Table 1.4-19

COMPARISON OF MEAN CATCH PER EFFORT FOR GILL NETS AND MEAN HYDROACOUSTIC DENSITY ESTIMATES FOR EXPERIMENTAL AND CONTROL AREAS (SINGLE TARGETS ONLY)

Plant	Gill Net Catch Per Effort	Hydroacoustic Density Estimate Fish Per (10^3 m^3)	Ratio
Oswego	31.81	6.30	5.04
Lennox	57.15	11.72	4.88
Pickering	48.79	6.34	7.70

The ratio for Oswego was 5.04 and for Lennox 4.88; this represents a 97% agreement between the two independent sampling techniques. The ratio of gill net catch to single target density at Pickering was 7.70 which is higher than either the Lennox or Oswego ratio. This is attributed to the large numbers of brown bullhead collected in gill nets at Pickering and their presumed relatively low representation in hydroacoustic signals because of their benthic nature. Therefore, with two independent sampling methods, the relative abundance of fishes shows substantial agreement.

The density numbers given in Tables 1.4-16 through 1.4-18 can also be used to estimate a standing stock of fish in the water body segments surveyed. This can be done by an estimation of the total water volume available and the assumption of an equal distribution of fish, or by the use of depth distribution and distance offshore data, algorithms can be calculated to represent these distributions and then the total abundance calculated by the multiplication of the fish density times the water volume times the various distribution factors.

1.4.6.6 Summary. As long as good engineering practices are adhered to, and a priori assumptions are tested, and the effects of the violation of these assumptions quantified, or better yet if no a priori assumptions are made and the hydroacoustic data recorded in its raw analogue form, then the method holds promise as a tool in the evaluation of fisheries.

This section presented a review of the uses of hydroacoustics and a description of the development of the quantification methodology. It was demonstrated that the distribution of fishes can be documented both spatially and temporally. Changes in distribution of Great Lake fishes were observed and correlated with known behavioral characteristics of the dominant fishes (e.g. day - night and seasonal differences).

Fish abundance or biomass can be determined although the confidence intervals on such estimates are wide. Methods are presented for the counting of individual targets and for the estimation of volume backscatter from groups of fishes. Both of these types of data are amenable to statistical evaluations, and several significant differences were demonstrated for fish distribution in the vicinities of three power plants on Lake Ontario.

Density estimates can be made and standing stocks calculated to be used to estimate potential impingement losses or to contrast with the impingement losses as one form of impact evaluation. In each case, the hydroacoustics should be accompanied by netting or some other method to document the species present and their sizes.

In order to convert the aggregation data into density estimates, the average target strength of the insonified fishes must be known. This can be accomplished by measurements of the target strength of representative individuals, and much more work is needed in this area.

1.5 PHYSICAL MODELS FOR OBTAINING ESTIMATES OF ENTRAINMENT EFFECTS

1.5.1. Introduction

Entrainment involves the interaction of physical factors (e.g. plant hydraulics, intake vicinity, geometry, hydrology, etc.), and biological factors (e.g. distribution of organisms relative to the intake structure, avoidance capability, etc.). Thus, physical measurements play a major role in the prediction of the fractional reduction in abundance of organisms due to entrainment mortality. This section addresses the role of physical measurements in assessing entrainment impact through the construction of several simple entrainment models. In each model, emphasis is placed on the physical parameters of water body flow and mixing, and plant flow and recirculation. River, estuary and lake/bay examples are given. In addition, the models show how certain simple biological parameters, including plant cropping (cross-plant organism mortality) and organism spatial distribution, interact with the preceding physical parameters.

The object is to demonstrate the model construction methodology, to provide a series of basic models, and to show the relative significance of various physical and biological parameters in entrainment estimation. The methodology provides some of the tools, and the basic models provide the building blocks. Together, they are useful in the construction of more complex, site-specific or waterbody-specific models. The evaluation of the relative significance of various parameters shows the relative priorities that must be placed on the measurement of each in an information gathering program.

1.5.2. River Models

1.5.2.1 River Model #1: Uniform Upriver Organism Concentration, No Recirculation. Consider Figure 1.5-1. A single generating station is located on the west bank of a freshwater, non-tidal river.

River flow rate is Q cubic feet per second (cfs). Plant withdrawal rate is Q_p cfs. Live planktonic organisms, assumed to be drifting with the river flow, are present in the water column. The live organism density, or concentration, in numbers per unit volume of water (e.g., per thousand cubic meter [tcm], per thousand cubic foot [tcf] or other appropriate volume unit) is designated generally as C . Initial organism concentration is the concentration upstream of, and assumed to be unaffected by the plant and is designated C_o .

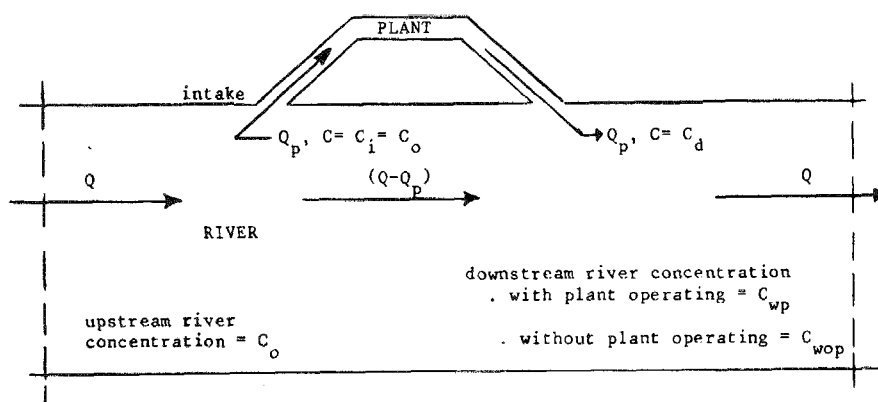


Figure 1.5-1. River Based Intake-Discharge System

Downstream river concentration varies and partly depends on plant construction and operating conditions (flow, power level, pump-pipe-condenser configuration). The downstream concentration variable is designated C_{wp} when water is being withdrawn and C_{wop} when no water is being withdrawn.

Plant concentration at the intake is designated C_i and is assumed to be equal to the upstream river concentration C_o . This is tantamount to assuming uniform concentration of organisms across the upstream cross-section, given the earlier assumption that the concentration in the waters immediately upstream of the plant intake is unaffected by the plant.

The discharge concentration is designated C_d and in general can be expected to be less than the intake concentration C_o by an amount determined by the entrainment kill.

To measure the short-term population impact due to entrainment, the downstream river concentration with and without the plant operating must be compared. To do this, an inventory equation on organism numbers between the upstream and downstream river sections must be written. This is equivalent to writing a material balance on organism biomass between these two sections, the material balance in turn being the application of the law of conservation of mass over a fixed volume of the river.

The basic inventory equation over a fixed region in space (here, the river volume between the upstream and downstream sections) is written:

$$\text{INPUT} - \text{OUTPUT} + \text{PRODUCTION} - \text{LOSSES} = \text{ACCUMULATION.}$$

(1.5-1)

Eq. 1.5-1 simply inventories, or accounts for, all of the quantity under study, which in any way passes through or appears within the fixed volume under study during the period of interest.

Each of the terms in Eq. 1.5-1 can be written as a total over a finite period of time or as a rate. For example, if the statement on plankton numbers is written over a finite time period, Δt , then "input" is simply the total number of plankton organisms passing through the upstream section* into the study reach (the river volume between the upstream and downstream sections - designated as V).

In the case under study, this total would be written $QC_o \Delta t$. The product QC_o yields the rate, or number per unit time, at which organisms move across the upstream boundary into river reach "V."

Multiplying this quantity by the time increment yields the total number input to the system.

Alternatively, each term in the inventory statement can be written as a rate, applicable at any point in time. Expressed as a rate, the input term for the above case is simply QC_o .

The rate approach is generally preferable, particularly when developing a general model, since it avoids problems of imprecise product representation when one or more variables are time-dependent; e.g., if either Q or C_o or both are time variable in the above example, the product is more properly written

$$\overline{Q_p C_o} \Delta t \quad \text{or} \quad \int_t^{t+\Delta t} Q_p(t) C_o(t) dt \quad (1.5-2)$$

rather than simply $Q_p C_o \Delta t$. Furthermore, the rate approach leads more quickly to the development of the ordinary or partial differential equations that describe

*In general, the input term would include organisms entering "V" from any direction. In the case under study, the organisms are planktonic, assumed to flow with the river, which in turn is non-tidal and therefore unidirectional.

the system and lends itself immediately to "steady state" analyses of the system. The rate approach is used exclusively throughout this section.

Apply Eq. 1.5-1 to the system in Figure 1.5-1. Choose the system boundaries so that the intake and discharge canals and the plant are within the system. In so doing, Q_p , the plant flow, does not appear in either the input or output terms since it does not cross system boundaries. The terms in the inventory equation (Eq. 1.5-1) are evaluated as follows:

Input:	$Q C_o$	
Output:	$Q C_{wp}$	(plant operating)
Production:	0	(assume no new plankton generated within "V")
Losses:	$Q_p (C_o - C_d)$	(cropping by entrainment)
Accumulation:	0	(assume steady state)

For the general case, the accumulation term is simply the net result of taking inventory and simply registers whether input and production outstrips, equals, or trails output and losses. It is expressed mathematically as the time rate of change of the total number of organisms resident in the system at any time and is written $d(\bar{C}V)/dt$, in which $\bar{C}$ is the average concentration of organisms within the volume V at time t . Steady state defines the condition for which all parameters are time-independent, so at steady state the time derivative is zero. It simply says that when the system is at steady state, the concentration of organisms within the system is neither increasing or decreasing or that accumulation is zero.

Substitution of the foregoing quantitative expressions in Eq. 1.5-1 yields:

$$QC_o - QC_{wp} - Q_p(C_o - C_d) = 0 \quad (1.5-3)$$

Solve Eq. 1.5-3 for the downriver concentration C_{wp} and obtain:

$$C_{wp} = C_o \left[1 - \frac{Q_p}{Q} \left(1 - \frac{C_d}{C_o} \right) \right] \quad (1.5-4)$$

Now recognize that C_{wop} , the downriver concentration when the plant is not operating is simply equal to C_o , the upriver concentration, since no other

mechanisms have been provided within the river reach "V" to impact C_o . For example, generation and natural mortality have both been neglected, the justification being that the reach is too small to expect significant changes due to these causes. With plant impact defined as the fractional reduction in downriver organism density with the plant operating by comparison to no operation, it can be stated:

$$\text{IMPACT} = \frac{C_{wop} - C_{wp}}{C_{wop}} = 1 - \frac{C_{wp}}{C_{wop}} \quad (1.5-5)$$

Substitution of Eq. 1.5-4 into Eq. 1.5-5, recognizing C_{wop} is given by C_o , yields:

$$\text{IMPACT} = \frac{Q_p}{Q} \left(1 - \frac{C_d}{C_o} \right) \quad (1.5-6)$$

Thus, for the assumptions made, the impact, in terms of fractional reduction in downriver organism population, is seen to be directly proportional to the product of a physical parameter, the plant to river flow ratio (Q_p/Q), and a biological parameter, the cross-plant mortality rate ($1 - C_d/C_o$).

The cross-plant mortality rate is usually obtained through measurement of the concentration of live organisms in the intake and discharge. Since physical destruction of organisms during plant passage may occur, it is important that studies be designed to account for cross-plant changes in both viability and abundance. Guidelines for the design and interpretation of such studies appear in Section 1.2. Cross-plant mortality rates are highly taxa specific, may vary with sex and/or size of organisms, and often depend on plant operating characteristics (e.g., ΔT , use of biocides) and ambient temperature. Thus, where information on cross-plant mortality is lacking for a given species, it is best to take a conservative stance and assume total mortality of entrained organisms. If impact estimates obtained with 100% mortality assumed are found to be potentially serious, a study to estimate cross-plant mortality may then be undertaken. For the remainder of this section, cross-plant mortality rate is designated f_c . Note that when f_c is set equal to one in Eq. 1.5-6 (corresponding to 100% mortality), entrainment impact exactly equals the plant to river flow ratio.

To summarize, the assumptions under which the statement of impact given by Eq. 1.5-6 applies are:

1. Uniform concentration across the upstream river cross section
2. No local recirculation of discharge organisms back to intake
3. Intake concentration unaffected by withdrawal
4. No generation or natural mortality
5. No compensation
6. All organisms are neutrally buoyant. They drift with the river flow and exhibit no other motion or behavior.
7. River velocity is uniform over each river cross-section. No density currents or other circulation patterns exist.

The impact of assumptions 1 through 5 is explored below in several river and estuary models.

1.5.2.2 River Model #2: Distributed Upriver Concentration, No Recirculation.

River plankton concentrations are rarely fully uniform across the river cross-section. They often vary laterally as well as vertically, sometimes in a steady fashion and other times quite randomly. This can be the result of localized spawning activity, bottom feeding preferences, sunlight preference, etc. The implications of non-uniform organism concentration are explored in the following model.

Consider Figure 1.5-2. The plant is seen to withdraw its required Q_p from a fraction of the river's cross-section proportional to Q_p/Q . This assumes

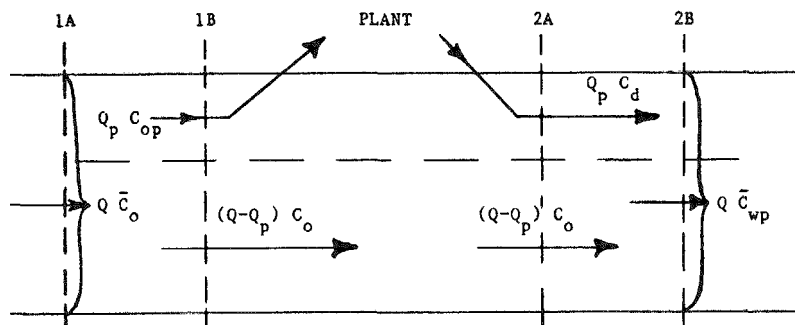


Figure 1.5-2. Withdrawal concentration not equal to upriver average concentrations.

uniform river velocity over the whole cross-section. For more complex, non-uniform flows, the zone of the river from which the intake water is taken can be obtained by flow net analysis.

River concentration inside the withdrawal zone is designated C_{op} and outside it, C_o . The intake concentration is still assumed to be unaffected by the withdrawal, and with the exception of the relaxation of the assumption of uniform upriver concentration, all the other listed assumptions apply. Hence, the analysis is similar to that for the previous situation.

As before to determine the overall impact, the average downriver concentration for the case of the plant operating, $\bar{C}_{wp}$, must be compared to its counterpart under no operation. The latter is clearly the average upriver concentration, since no change is assumed to occur between upriver and downriver sections under the no operation condition.

Applying the inventory equation between sections 1A and 1B in Figure 1.5-2 equates the average upriver organism number flow to its components inside and outside the withdrawal zone:

$$Q\bar{C}_o - Q_p C_{op} - (Q - Q_p) C_o = 0 \quad (1.5-7)$$

Solution of equation for $\bar{C}_o$, the average upriver concentration, yields:

$$\bar{C}_o = \frac{Q_p C_{op} + (Q - Q_p) C_o}{Q} \quad (1.5-8)$$

Likewise, applying Eq. 1.5-1 between sections 2A and 2B in Figure 1.5-2 equates the average downriver organisms number flow to its plant and remaining river flow components:

$$Q_p C_d + (Q - Q_p) C_o = Q\bar{C}_{wp} \quad (1.5-9)$$

Solution of Eq. 1.5-9 for $\bar{C}_{wp}$, the average downriver concentration when the plant is operating, yields:

$$\bar{C}_{wp} = \frac{Q_p C_d + (Q - Q_p) C_o}{Q} \quad (1.5-10)$$

Note that this result is identical to that for $\bar{C}_o$ in Eq. 1.5-8 when the plant is not operating, since, in that situation, there is no entrainment loss and $C_d = C_{op}$. More precisely, when Q_p is not diverted through the plant, it is simply that portion of the river's total flow Q that is carrying the concentration C_{op} .

Use the definition of impact given by Eq. 1.5-5, recognize that $C_{wop} = \bar{C}_o$ and obtain:

$$\text{IMPACT} = \frac{f_c}{1 + (Q/Q_p - 1) C_o/C_{op}} \quad (1.5-11)$$

Here f_c , the cross-plant mortality rate, is equal to $(1 - C_d/C_{op})$ as described above. Eq. 1.5-11 reduces to simpler cases as it should. Where $f_c = 0$, impact is zero, as required. Where the upriver concentration is uniform, $C_o = C_{op}$ and Eq. 1.5-11 reduces to Eq. 1.5-6.

Referring to Eq. 1.5-6, note that, in that simple case of uniform upriver concentrations, given a fixed f_c , impact is directly proportional to Q_p/Q . In Eq. 1.5-11, when $C_o/C_{op} > 1$, the concentration in the withdrawal zone is less than the average upriver concentration, and impact should be less than would be obtained by Eq. 1.5-6. Similarly, where $C_o/C_{op} < 1$, the withdrawal zone concentration is greater than the average, and the impact should be greater than would be estimated using Eq. 1.5-6.

To show that Eq. 1.5-11 yields the behavior described above, rearrange it to the following form:

$$\text{IMPACT} = \frac{(Q_p/Q) f_c}{\left(\frac{Q_p}{Q}\right) + \left(1 - \frac{Q_p}{Q}\right) \frac{C_o}{C_{op}}} \quad (1.5-12)$$

The quantity $[Q_p/Q + (1 - Q_p/Q)]$ is always unity and each of its components, Q_p/Q and $(1 - Q_p/Q)$, are always positive. The latter is so since $Q_p < Q$, or else

the plant would drain the river dry or the assumption of no recirculation would be invalid. Thus, when $C_o > C_{op}$, the denominator of the right side of Eq. 1.5-12, $[Q_p/Q + (1-Q_p/Q) C_o/C_{op}]$, must always be greater than unity. The Eq. 1.5-12 impact will be less than the value of impact given by Eq. 1.5-6. Similar reasoning shows that for $C_o < C_{op}$, the denominator is less than unity and the Eq. 1.5-12 impact is greater than that obtained by Eq. 1.5-6.

This model is useful in estimating impact where the upriver organism concentration is not uniform. In the frequent situation of random and alternating behavior; i.e., values of C_o/C_{op} varying between zero and infinity obtained for a number of samples taken over the period of entrainment vulnerability, the calculation can be used on a per sampling day basis and an average impact over the full period of vulnerability estimated.

Once again, the physical variable, namely the plant to river flow ratio, is seen to appear in concert with biological variables, in this case the plant cropping factor and the upriver organism distribution. The significance of the flow ratio can be obtained by viewing the behavior of impact with changing flow ratio either graphically or analytically. Figure 1.5-3 shows the behavior graphically.

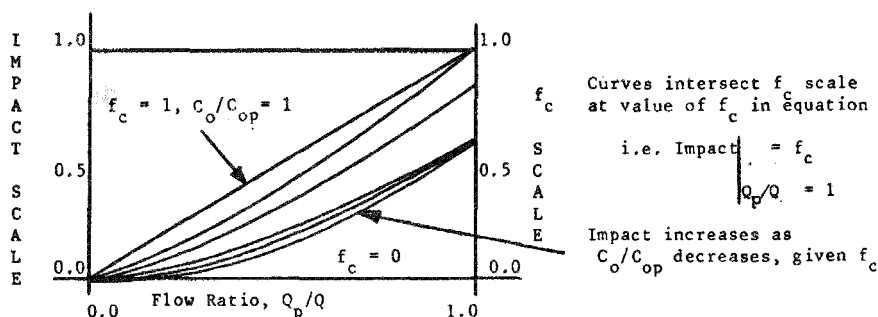


Figure 1.5-3. Graphic Representation of Equation 1.5-11.

The effect of a change in the flow ratio on impact is obtained analytically by writing the partial derivative of impact with respect to flow ratio. From Eq. 1.5-11, obtain:

$$\frac{\partial I}{\partial (Q_p/Q)} = \frac{f_c (C_o/C_{op}) (Q/Q_p)^2}{[1 - (Q/Q_p - 1) C_o/C_{op}]^2} \quad (1.5-13)$$

As C_o approaches C_{op} , this rate of change approaches a constant value, namely f_c . For such a situation, impact is essentially a linear function of flow ratio with slope f_c , as would have been expected by examination of Eq. 1.5-6. As the difference between C_o and C_{op} increases, the linear dependence on flow ratio fades, and the curvilinear behavior shown in Figure 1.5-3 takes over.

The interdependent role of the various physical and biological variables on the impact can be seen quite clearly by examination of Eq. 1.5-11. For a situation of relatively high plant flow ($Q_p/Q \rightarrow 1$), impact approaches the actual level of cross-plant mortality, unless C_o/C_{op} is also quite large. This simply means that when the plant is withdrawing a large portion of the river's flow, unless the organism concentration in that portion is very small by comparison to organism density in the bypassed water, impact will, for all practical purposes, be close to the level of cross-plant mortality. On the other hand, for relatively low plant flow ($Q_p/Q \rightarrow 0$), the overall impact on the river will be substantially lower than the level of cross-plant cropping, unless C_o/C_{op} is quite small. This occurs when the plant is withdrawing only a small portion of the river; unless most of the river's organism population is concentrated in that portion, the overall impact will be small, the actual plant cropping effect being dampened considerably due to the relatively low use of the river resource.

1.5.2.3 River Model #3: Uniform Upriver Concentration with Local Recirculation.

Consider Figure 1.5-4. A fraction r of the discharge flow Q_p recirculates back into the intake withdrawal zone, and, along with $Q_p(1-r)$ of new water, makes up the plant flow Q_p .

Apply the inventory equation to the river system bounded by sections A and B in Figure 1.5-4. Section A represents the inflow of the partitioned waters: $Q_p(1-r)$, the net contribution of the discharge to the river, and $Q - Q_p(1-r)$, the portion of the river not withdrawn. Section B represents outflow of the mixture of the above inflows. Whether they actually mix is immaterial: C_{wp} is the average downriver concentration. Whether uniform due to lateral mixing after plant discharge or whether simply the computed average, it has the same value. This application of Eq. 1.5-1 yields:

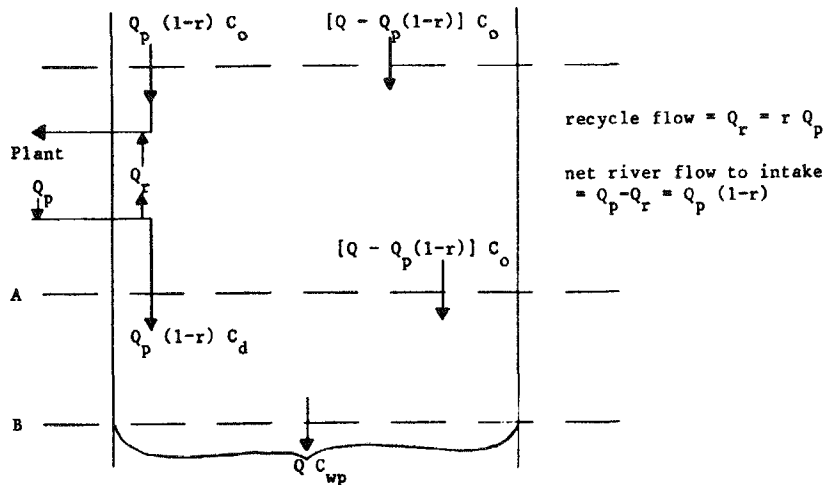


Figure 1.5-4. Local recirculation of a fraction of the discharge back to the intake.

$$C_{wp} = \left[1 - \frac{Q_p}{Q} (1 - r)\right] C_o + \frac{Q_p}{Q} (1 - r) C_d \quad (1.5-14)$$

The intake flow receives contributions from both upstream waters, $Q_p (1-r) C_o$, and the recycled flow from the discharge, $Q_p r C_d$. Thus,

$$Q_p C_i = (1 - r) Q_p C_o + r Q_p C_d \quad (1.5-15)$$

By definition, the cross-plant mortality rate, f_c is given by the expression:

$$f_c = \frac{Q_p C_i - Q_p C_d}{Q_p C_i} \quad (1.5-16)$$

Substitution into Eq. 1.5-15 of $Q_p C_i$ in terms of its components given in eq. 1.5-14 yields:

$$f_c = \frac{1 - C_d/C_o}{1 + (C_d/C_o)(r/1-r)} \quad (1.5-17)$$

which may be rearranged to yield:

$$C_d = \frac{C_o (1 - f_c)}{1 + f_c \left(\frac{r}{1-r}\right)} \quad (1.5-18)$$

Note that when recirculation is absent ($r=0$), discharge concentration is equal to the unaffected upstream concentration multiplied by the cross-plant survival rate, as expected. To find the downriver organism concentration with the plant operating, replace C_d in Eq. 1.5-14 by the right-hand side of Eq. 1.5-18 to yield:

$$C_{wp} = C_o \left[1 - \frac{Q_p}{Q} (1-r) \left(\frac{f_c}{1-r + rf_c} \right) \right] \quad (1.5-19)$$

In accordance with Eq. 1.5-5 and the fact that $C_{wop} = C_o$, the impact statement is:

$$\text{IMPACT} = \frac{(Q_p)}{Q} \frac{[f_c (1-r)]}{1-r + rf_c} \quad (1.5-20)$$

Comparison of this result to Eq. 1.5-6 shows that for a fixed level of cross-plant mortality f_c , and fixed plant flow, Q_p , the impact for some degree of recirculation is smaller than that for no recirculation. This result is as expected, for when recirculation occurs, the concentration of organisms seen by the intake is reduced by the discharge contribution to the intake flow. This comparison would not be quite correct if the cross-plant mortality rate were dependent on the degree of recirculation. This could occur if the probability of organism death during entrainment were greater for an organism that had already survived one or more passages through the plant than for an organism passing through for the first time.

1.5.2.4 River Model #4: Distributed Upriver Concentration with Local Recirculation. In this model, the individual considerations developed in the first three models are grouped together. Consider Figure 1.5-5.

Use the approach developed above for River Model #2. The average upriver concentration, C_o , is similar to that given by Eq. 1.5-8, but now recognize that the total river flow Q has components within and without the withdrawal zone equal to $Q_p(1-r)$ and $Q - Q_p(1-r)$, respectively. Evaluation of Eq. 1.5-1 over the system bounded by Sections 1A and 1B yields:

$$Q\bar{C}_o = Q_p (1-r) C_{op} + [Q - Q_p (1-r)] C_o \quad (1.5-21)$$

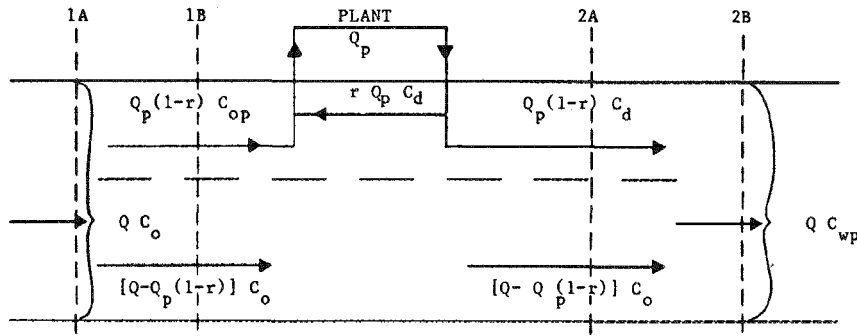


Figure 1.5-5. Local recirculation, withdrawal concentration not equal to upriver average concentration.

Similarly, evaluation of Eq. 1.5-1 between Sections 2A and 2B yields:

$$Q \bar{C}_{wp} = \frac{Q_p (1-r) (1-f_c) C_{op}}{1 + f_c \frac{r}{(1-r)}} + [Q - Q_p (1-r)] C_o \quad (1.5-22)$$

Note that the first term on the right side of Eq. 1.5-22 is obtained by using the definition of C_d given by Eq. 1.5-18. Since C_{wop} equals $\bar{C}_o$, divide Eq. 1.5-22 by Eq. 1.5-21 to obtain $\bar{C}_{wp}/\bar{C}_{wop}$. Use the impact definition given by Eq. 1.5-5 and obtain:

$$\text{IMPACT} = \frac{\left(\frac{C_{op}}{C_o} \right) \left(\frac{Q_p}{Q} \right) \left(\frac{f_c (1-r)}{1-r + r f_c} \right)}{1 + \left(\frac{Q_p}{Q} \right) (1-r) \left[\left(\frac{C_{op}}{C_o} \right) - 1 \right]} \quad (1.5-23)$$

For $C_{op} = C_o$, the uniform concentration case, Eq. 1.5-23 reduces to Eq. 1.5-20. If, further, recirculation is zero, the result reduces to Eq. 1.5-6. If $C_{op} \neq C_o$, but $r = 0$, Eq. 1.5-23 reduces to Eq. 1.5-11.

Thus Eq. 1.5-23 contains all of the impact solutions for steady-state conditions in unidirectional rivers in which the upriver concentration is unaffected by the plant, and in which the parameters influencing plant effect on downriver concentration include plant flow, river flow, plant cropping, local recirculation of plant discharge water, and upriver organism concentrations. Eq. 1.5-23 places each of these variables in its proper quantitative position as far as its influence on impact is concerned.

The following numerical example provides tangible application of the foregoing. Consider Figure 1.5-6.

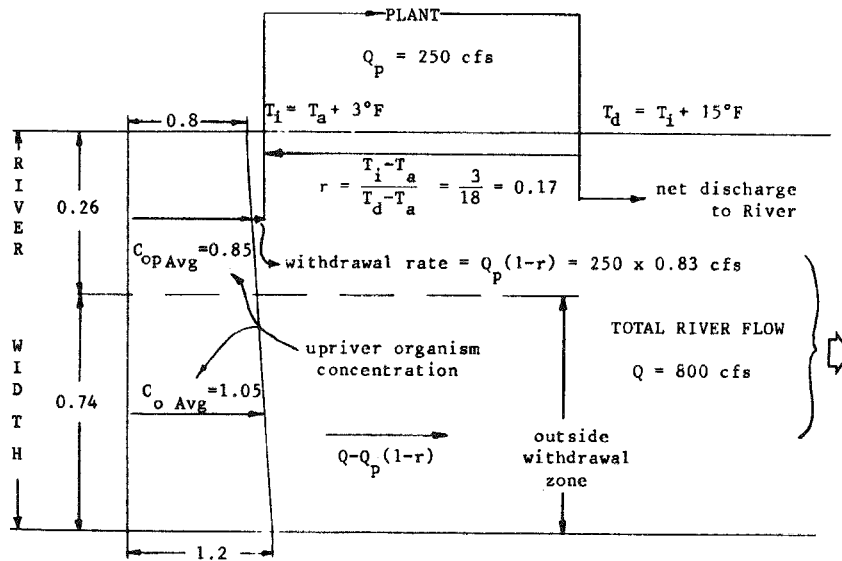


Figure 1.5-6. Schematic for example of general river model (Eq. 1.5-23).

A 200-MW, oil-fired generating station is located on a unidirectional fresh-water river, whose average flow during the period of the year when fish larvae are vulnerable to entrainment is 800 cfs. Plant flow is 250 cfs and plant T is 15°F . Because of the closeness of intake and discharge, local recirculation of a portion of the plant flow is suspected. Temperature measurements are made to verify this, and the intake temperature is found to be 3°F higher than the upriver ambient temperature. Spawning activity is known to be concentrated upriver on the opposite shore, and plankton tows show that the larval concentration varies linearly across the river, from a low of 20% below the riverwide average concentration on the plant side of the river to a high of 20% above the average on the opposite shore. Larval table data (1.5-1) show an intake sample mortality of 30% and a discharge sample mortality of 60%.

To solve this problem, begin by computing the recirculation ratio, which can be shown to be given by:

$$r = \frac{\text{recycled flow}}{\text{total plant flow}} = \frac{T_i - T_a}{T_d - T_a} \quad (1.5-24)$$

in which T_i = plant intake temperature
 T_d = plant discharge temperature
 T_a = upriver ambient temperature

For this example, $r = \frac{3}{15 + 3} = 0.17$.

Next, compute the fraction of the river's flow that actually "sees" the plant. This is given by $Q_p(1-r)/Q$ and is $250 \times 0.83/800$ or 0.26. Assume the river depth and velocity are fairly uniform. The zone of withdrawal, in terms of the fraction of the river's width from which the river's contribution to the plant flow, $Q_p(1-r)$, is drawn, will then also be 0.26.

Use this result to compute the C_{op}/C_o ratio. Since the larval density varies between 0.8 and 1.2 of the area-wide average, the mean relative density inside the withdrawal zone will be $0.5 [0.8 + (0.8 + 0.26 (1.2-0.8))]$ or 0.85. Similarly, the mean relative density outside the zone of withdrawal will be $0.5 [1.2 + (1.2-0.74 (1.2-0.8))]$ or 1.05. Thus $C_{op}/C_o = 0.85/1.05$ or 0.81.

Larval mortality due to plant cropping can be shown to be obtained as follows:

$$f_c = \frac{\text{Intake Survival} - \text{Discharge Survival}}{\text{Intake Survival}}$$

This assumes that sampling apparatus, catch velocities, and in general, all aspects of the sampling are identical in intake and discharge. When this is not the case (the normal situation), correction factors can be applied as appropriate, provided data are developed under a sufficient number of conditions to make such factors meaningful (1.5-1). For this problem:

$$f_c = \frac{(1 - 0.3) - (1 - 0.6)}{(1 - 0.3)} = 0.43$$

Summarizing, the relevant ratios are:

$$\begin{array}{ll} Q_p/Q = 0.31 & C_{op}/C_o = 0.81 \\ r = 0.17 & f_c = 0.43 \\ (1 - r) = 0.83 & (Q_p/Q)(1 - r) = 0.26 \end{array}$$

Substitution of the foregoing into the various impact expressions yields the following estimates of impact:

Condition	Equation	Percentage Impact	
		$f_c = 0.43$	$f_c = 1.0$
Recirculation, distributed concentration	1.5-23	11	22
No recirculation, distributed concentration	1.5-11	11	26*
Recirculation, uniform concentration	1.5-20	12	26
No recirculation, uniform concentration	1.5- 6	13	31

*Zone of withdrawal becomes equal to Q_p/Q or 31% of the river's width. Accordingly, C_{op}/C_o is recomputed but still rounds to be 0.81.

These results clearly show that for the particular set of numerical values that apply to the case under study, recirculation and concentration distribution have less influence on the impact than do the plant to river ratio and the cropping factor. When cropping is complete ($f_c = 1$), recirculation and distributed concentration effect more change in the raw flow impact (31%), than they do in the case of incomplete cropping ($f_c = 0.43$).

However, other examples could be constructed which would more clearly point out the importance of estimating the cross-sectional distribution of organisms. As an extreme example, consider the possibility that all organisms are concentrated within the intake zone of withdrawal. In this case, impact could be as high as 100% reduction in downriver concentration, depending on the values of f_c and r . In contrast, if all organisms were located outside the intake withdrawal region, no impact would occur, regardless of the values of other parameters. This is an important consideration when deriving initial estimates. No recirculation and a cross-plant mortality rate of 100% can be assumed to obtain a conservative impact estimate, if additional information on these factors is lacking. However, to assume uniform distribution of organisms in the upriver cross section may not be conservative, as pointed out by the extreme example above. Thus, the minimum information required to derive a conservative estimate of entrainment impact is a measure of the relative concentration of organisms within and outside of the intake withdrawal zone.

1.5.3 Estuary Models. A major feature of most tidal river and estuarine areas is that the estuary organism concentration in the vicinity of the plant is not clearly characterized by a separate upriver "unaffected" concentration and a downriver "affected" concentration. Because of the large scale mixing and tidal exchange induced by tidal oscillation, density currents and fluctuating tidal turbulence, the concentration in the vicinity of the plant, both above and below the intake/discharge location, is influenced by the plant operation.

1.5.3.1 Estuary Model #1: Complete Mixing in the Vicinity of the Plant. An elementary view of this situation is developed below.

Consider Figure 1.5-7 which represents a situation in which mixing in the vicinity of the plant is sufficiently intense to make the concentration which the intake "sees" completely influenced by plant withdrawal and subsequent cropping. This is tantamount to "global" recirculation or complete mixing in front of the plant, so that the intake concentration cannot be distinguished in terms of components of "new" river water and associated organism levels and locally recirculated discharge flow and organism levels. All that is known is that the intake can be expected to "see" an organism concentration level that is also seen immediately upstream and downstream of the plant.

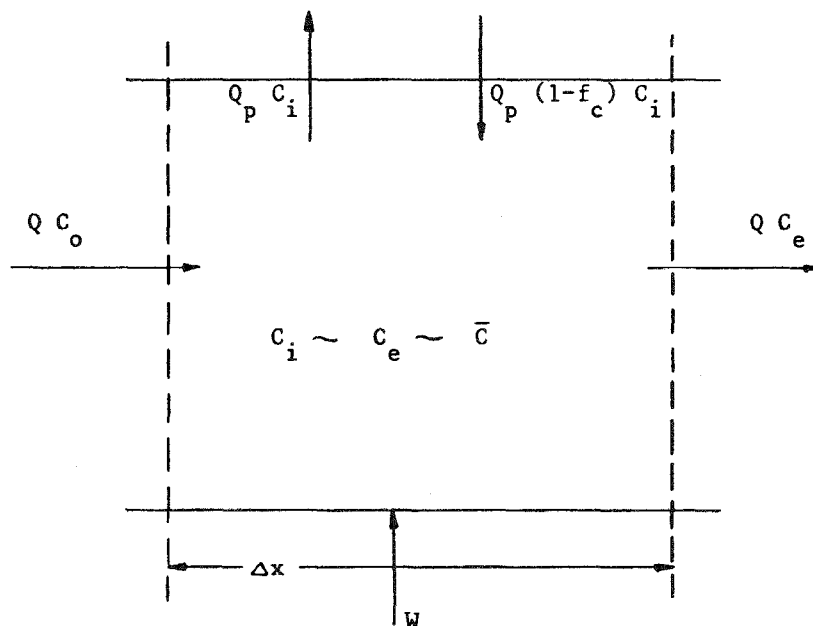


Figure 1.5-7. Schematic of organism concentration behavior under well-mixed conditions in the vicinity of an intake/discharge system.

This situation represents the opposite extreme of River Model #1, above, in which the entire load entering the plant was derived from upstream waters unaffected by plant operations. Intermediate situations include the local recirculation schemes of River Models #3 and 4 and also the superposition of such local recirculation on the completely mixed model being considered here. The latter case is discussed later under Lake Models.

Apply the steady-state inventory equation to organism numbers over the system bounded by sections A and B in Figure 1.5-6. This yields:

$$QC_o + W - Q_p C_i + Q_p (1-f_c) C_i - QC_e = 0 \quad (1.5-25)$$

in which: QC_o = contribution of new organisms from unaffected upstream waters

W = generation (spawn) of organisms within the zone of plant influence

C_i = concentration withdrawn by the plant

Assume the extent of river encompassing the intake/discharge system is completely mixed. The concentration over the extent of the river that is completely mixed is equal to C . Further, the concentration leaving this zone, C_e , is also equal to $\bar{C}$, since the zone is assumed to be completely mixing. This provides a means of relating C_e , the net downriver organism concentration, to the concentration $\bar{C}$ seen by the plant. Eq. 1.5-25 can now be solved to yield:

$$C_e = \bar{C} = C_{wp} = \frac{QC_i + W}{Q + Q_p f_c} \quad (1.5-26)$$

Actually, the assumption of complete mixing is never completely fulfilled, so, over the small but finite river reach Δx in Figure 1.5-7, $C_i \neq C_e$ in the precise sense. Since the mixing is intense, both C_i and C_e tend to be approximated by the average concentration over the reach, $\bar{C}$. As the reach length becomes increasingly smaller, this approximation becomes more exact; i.e., the complete mixing assumption is closer to reality. In the limit, as $\Delta x \rightarrow 0$, the average concentration in the infinitesimally small segment, as well as that leaving it by whatever means, i.e., C_i and C_e , approach a single value. Thus, the best view of Eq. 1.5-24 is that it applies at the river section bisecting the intake-discharge location.

Without the plant operating, a similar application of the inventory equation yields:

$$C_{wop} = \frac{QC_i + W}{Q} \quad (1.5-27)$$

In accordance with Eq. 1.5-5, the impact for this configuration is:

$$\text{IMPACT} = \frac{f_c(Q_p/Q)}{1 + f_c(Q_p/Q)} \quad (1.5-28)$$

Comparison to Eq. 1.5-5 shows a substantially lower impact can be realized if the intake withdraws a mixed river concentration as described above, rather than withdrawing a previously wholly unaffected resource. Physically, what occurs in the well mixed situation is that each successive Q_p aliquot draws fewer total numbers than its precursor because the available level drops off. Concentration drawdown occurs and at steady state the withdrawal concentration is lower than it would be if the intake continued to see a wholly new source.

A good view of what is occurring here by comparison to the system described by Eq. 1.5-5 can be obtained by considering $Q_p = Q$ and $f_c = 1$. For this situation, Eq. 1.5-5 yields a 100% impact. The intake takes everything and crops it all, and the downriver concentration is zero. For the well-mixed situation described by Eq. 1.5-28, the impact value is only 50%. The downriver concentration is one-half the upriver value because the river flow out of the well-mixed reach in Figure 1.5-7 has the same influence as Q_p . Both draw continuously at the same rate on the same organism concentration level.

This idealized model can be employed to represent either a unidirectional river, wherein the mixing in the vicinity of the plant is so intense as to be unable to be distinguished between unaffected upstream contribution to the intake and the affected contribution, or to an estuary, where the view is on a tidal average basis; i.e., the parameters represent average behavior over the tidal cycle. Commentary on the care that must be taken in using tidal averages appears further on.

Notice that, in this model, the impact definition obviates the need to know the generation level, W , or the upriver contraction, C_i . Further, depending on

the type of interaction that exists between the estuary's organisms and its hydrodynamics, it may be possible in some instances to use the same model when downriver sources also exist.

The foregoing example of a maximum of 50% impact for the model described by Eq. 1.5-28 should not be interpreted to mean estuarine impact cannot exceed this level. The model assumes no feedback of the downriver organism flow, once this leaves the system. Return upriver movement of downriver moving organisms can sometimes occur, depending on estuarine circulation levels and organism response to this phenomenon.

1.5.3.2 Estuary Model #2: Tidal Averaged Behavior with Longitudinal Dispersion. One-dimensional, steady state models in which mixing and exchange from tidal oscillations, turbulence, and salinity-induced density currents have been incorporated in an overall longitudinal dispersion coefficient, have found much use in prediction of water quality behavior (1.5-2, 1.5-3, and 1.5-4). With particular reference to power plants, these methods have been employed to predict estuarine temperature distributions resulting from condenser waste heat discharges (1.5-5, 1.5-6, and 1.5-7).

That approach is applied below to track neutrally buoyant organism behavior in an estuary. The organisms are assumed to move with the tide and be subject to the mechanisms of tidal turbulence and exchange much as would small, neutrally buoyant particles of dye. Behavior is average behavior over the tidal cycle. Development of the nature of tidal average concentrations and associated parameters is given in Lawler (1.5-3). Additional considerations must be given when organisms, such as many marine and estuarine fish larvae, have ancillary motions; i.e., are not completely subject to the estuary's hydrodynamics.

Consider Figure 1.5-8. Organisms are being generated in the near vicinity of the plant, at the rate W numbers per day. They are vulnerable up to some age T , and the level of vulnerability is assumed to be relatively constant over the life stage. The spawn or generation is going on long enough so that the system time constant (a function of system variables) is less than the period of the spawn which, in turn, is less than the age to non-vulnerability. This means that everything in the system remains vulnerable and that steady state has been reached.

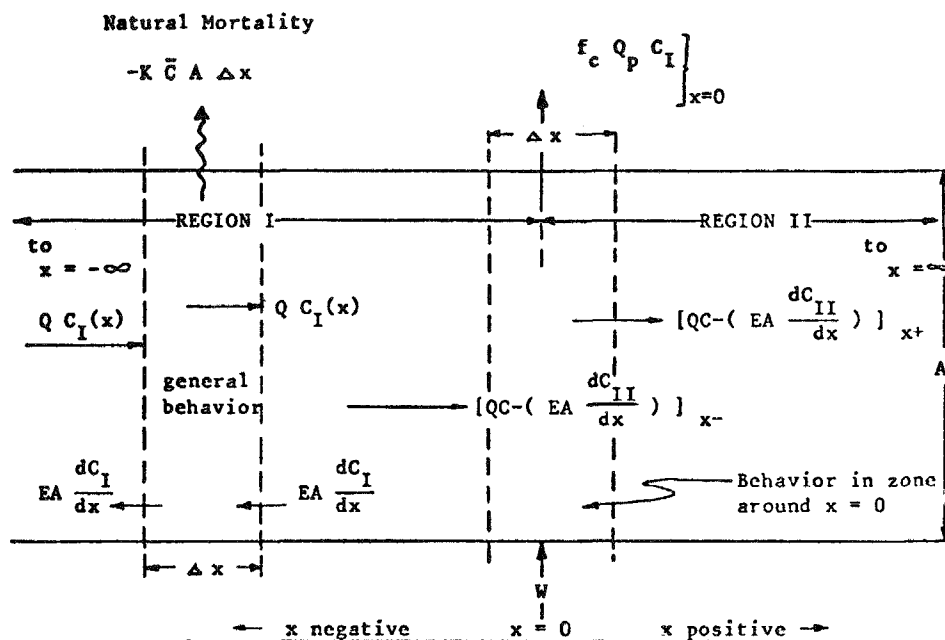


Figure 1.5-8. Schematic for one-dimensional, steady-state estuary model of organism generation, transport, entrainment and decay.

A brief summary of the relevant mathematics is given next. Substantially more discussion, derivation, and justification of these mathematics in terms of the physics of estuaries can be found in the above references, particularly Lawler (1.5-3) and QLM (1.5-7).

The defining differential equation for the system shown in Figure 1.5-8 is:

$$E \frac{d^2C}{dx^2} - U \frac{dC}{dx} - KC = 0 \quad (1.5-29)$$

in which: C = organism concentration at any x , No./1000³m
 x = distance variable along longitudinal axis of the estuary, miles
 E = longitudinal dispersion coefficient, mile²/day
 U = net freshwater runoff velocity, miles/day
 K = organism natural mortality rate, day⁻¹
 A = cross sectional area.

The boundary conditions for this system are:

$$\text{BC \#1} \quad C_I = 0 \quad \text{at } x = -\infty$$

$$\text{BC \#2} \quad C_{II} = 0 \quad \text{at } x = \infty$$

$$\text{BC \#3} \quad C_I = C_{II} \quad \text{at } x = 0$$

$$\text{BC \#4} \quad W - [f_c Q_p C_I] \Big|_{x=0} = AE \frac{dC_I}{dx} - \frac{dC_{II}}{dx} \Big|_{x=0}$$

Eq. 1.5-29 is obtained by applying Eq. 1.5-1 over the finite element Δx in Figure 1.5-8 and obtaining the limit as $\Delta x \rightarrow 0$. The estuary is assumed to be infinite in extent, which simply means the organism concentration level for organisms generated at $x = 0$ will drop to zero before reaching either end of the estuary. The first two boundary conditions express this.

Boundary conditions 3 and 4 simply recognize behavior in the near vicinity of the intake/discharge system ($x = 0$). Condition 3 says the concentration will be single valued at $x = 0$. Condition 4 is a statement of Eq. 1.5-1 applied across the plane of intake/discharge.

Four conditions are necessary because Eq. 1.5-29 applies to both the region upstream of the intake/discharge plane (Region I) and the region downstream of this plane (Region II) but does not apply across this plane. In other words, E , U , and K terms in Eq. 1.5-29 state that organism flux by dispersion, fresh-water flow, and natural mortality occurs everywhere throughout the estuary. Generation (W) and plant withdrawal ($Q_p f_c$) occur only at one point and hence do not appear in the differential equation which generally applies over all points in a given region. Thus, the location $x = 0$ is a boundary, and two conditions, one for each region, are necessary at $x = 0$.

The solution of Eq. 1.5-29 subject to the four boundary conditions, for the situation of no operation (either f_c or $Q_p = 0$), is:

$$\left. \begin{matrix} C_I \\ C \\ C_{II} \end{matrix} \right\}_{wop} = \frac{W \exp [(U/2E) (1 \pm \sqrt{1 + 4KE/U^2})] x}{Q \sqrt{1 + 4KE/U^2}} \quad (1.5-30)$$

For the plant operating, the analogous result is:

$$\left. \begin{matrix} C_I \\ C \\ C_{II} \end{matrix} \right\}_{wp} = \frac{W \exp [(U/2E) (1 + \sqrt{1 + 4KE/U^2})] \times}{f_c Q_p + Q \sqrt{1 + 4KE/U^2}} \quad (1.5-31)$$

Assuming no compensation in the survival rate K , i.e., $K_{wp} = K_{wop}$, impact in accord with Eq. 1.5-5, is:

$$IMPACT = \frac{f_c \left(\frac{Q_p}{Q} \right) \left(\frac{1}{\sqrt{1 + 4KE/U^2}} \right)}{1 + f_c \left(\frac{Q_p}{Q} \right) \left(\frac{1}{\sqrt{1 + 4KE/U^2}} \right)} \quad (1.5-32)$$

For either f_c or $Q_p = 0$, this reduces to zero impact, as it should. For $E = 0$, Eq. 1.5-28 is obtained. This simply recognizes that the system reduces to that of Figure 1.5-7 when organisms are moved away from the vicinity of the plant by freshwater flow only. The complete mixing assumption is inherent in the use of one-dimensional analysis and the assignment of BC #3 and #4.

For a comparable Q_p/Q ratio, estuaries will usually show less impact than the analogous freshwater river. This is partially due to the complete mixing behavior at the intake/discharge location and partially due to the dispersive effect.

For the same plant and river flows described above, suppose the river is estuarine, with a cross-sectional area of 30,000 ft², a longitudinal dispersion coefficient of 8 square miles per day and an organism decay rate of 10% per day. This yields:

$$\begin{aligned} E &= 8 \text{ sq mi/day} \\ U &= 800 \text{ cfs}/30,000 \text{ sf} = 0.0267 \text{ ft/sec} \\ &\quad 0.44 \text{ mile/day} \\ K &= 0.1 \text{ day}^{-1} \\ 4KE/U^2 &= 16.5 \\ 1/\sqrt{1 + 4KE/U^2} &= 0.24 \end{aligned}$$

Substitution in Eq. 1.5-32, using $f_c = 0.43$ and $Q_p/Q = 0.31$ as previously, yields an impact value of 0.33 or 3%. By comparison, the River Model #1 result, obtained by Eq. 1.5-6, was previously found to be 13%.

The results for various comparisons of Eq. 1.5-6, 1.5-28 and 1.5-32 are given below to demonstrate the role of f_c , Q_p/Q , E and the complete mixing assumption on impact, as defined by Equation 1.5-5.

Condition	Equation	Percentage Impact	
		$f_c = 0.43$	$f_c = 1.0$
Estuary, $E \neq 0$	1.5-32	3	7
Estuary, $E = 0$			
or	1.5-28	12	24
Well Mixed River			
River*	1.5-6	13	31

*For effect of recirculation and organism distribution see summary table to earlier example (Figure 1.5-6).

Note that when $f_c(Q_p/Q)$ is small, say on the order of 0.1, there is little difference between Eq. 1.5-6 and 1.5-28. On the other hand, as it becomes larger, here seen at the level of 0.3, the difference is more substantial. At the limit of the comparison (a value of 1.0), Eq. 1.5-6 yields a 100% impact, whereas Eq. 1.5-28, a 50% impact. Explanation of this is given above after Eq. 1.5-28. In reality, Q_p can exceed Q but the model leading to Eq. 1.5-6 is limited. In that model, Q_p cannot exceed Q , else the plant drains the river. Thus, even for rivers, particularly where Q_p/Q is relatively large, and dams, on-stream reservoirs, forebays or the like exist to provide sufficient plant water (situations where recirculation tends to be global), Eq. 1.5-28 may be more appropriate than Eq. 1.5-6.

1.5.2.3 Estuary Model #3: Model 2 with Compensation. The notion of impact adopted in this study is simply that of local and direct change; i.e., what change is effected in the population or concentration of organisms as it moves downriver past the plant? It addresses neither the question of offsetting influences on this population, the question of extent and rate of recovery of such, nor the question of significance; i.e., what does it all mean with respect to the overall ecosystem of which it is a part?

An early step in assessing such local and direct impacts is to determine whether compensatory mechanisms exist which can be expected to offset, at least to some extent, the initial concentration change.

One means by which such compensation can occur is by a decrease in the natural mortality rate in the presence of population reduction, thus allowing those organisms which survive the initial cropping a better chance of surviving natural predation, disease, etc., than they would have in the presence of denser populations. Examination of most equations of predation, competition, etc., show them to be density-dependent, with increased unit mortality rates among the prey and predators or competitors as populations increase.

Eq. 1.5-30 and 1.5-31 can be employed to show how compensation, assuming it exists, acts to offset initial impacts. Designate the unit natural mortality rates, in Eq. 1.5-30 and 1.5-31, as K_{wop} and K_{wp} , respectively. Consider these to be functions of C , the organism concentration, such that $K_{wop} > K_{wp}$, since $C_{wop} > C_{wp}$. This simply recognizes that as concentration increases, the unit mortality rate also increases. Thus, natural survival of C_{wp} will be higher than natural survival of C_{wop} .*

Under these conditions, application of Eq. 1.5.5 to Eq. 1.5-30 and 1.5-31 to determine impact will yield a more complex result, since the terms containing K no longer simply cancel. For example, the decay coefficient in the "x" direction is no longer the same for both C_{wp} and C_{wop} , so the impact expression will now include variation in x . The result is:

$$\text{IMPACT} = \frac{f_c\left(\frac{Q_p}{Q}\right) + F(K_{wp}) - F(K_{wop}) \exp \{+[F(K_{wop})]x\}}{f_c\left(\frac{Q_p}{Q}\right) + F(K_{wp})} \quad (1.5-33)$$

in which: $F(K) = \sqrt{1 + 4KE/U^2}$

*Strictly speaking, if K is a function of C , Eq. 1.5-29 is non-linear and the solution is more complex than given by Eq. 1.5-30 and 1.5-31 (Lawler, personal files). However, only small changes in K are being investigated herein, so that the procedure which follows is quite valid.

The graphical behavior of Eq. 1.5-30 and of Eq. 1.5-31, the latter for both density-dependent and density-independent K , is shown on Figure 1.5-9. Note that the maximum concentration of C_{wp} for the density-dependent case; i.e., $C_{wp} + x = 0$, is larger than its density-independent counterpart, and further that the longitudinal decay rate of the latter is faster than that of the former.

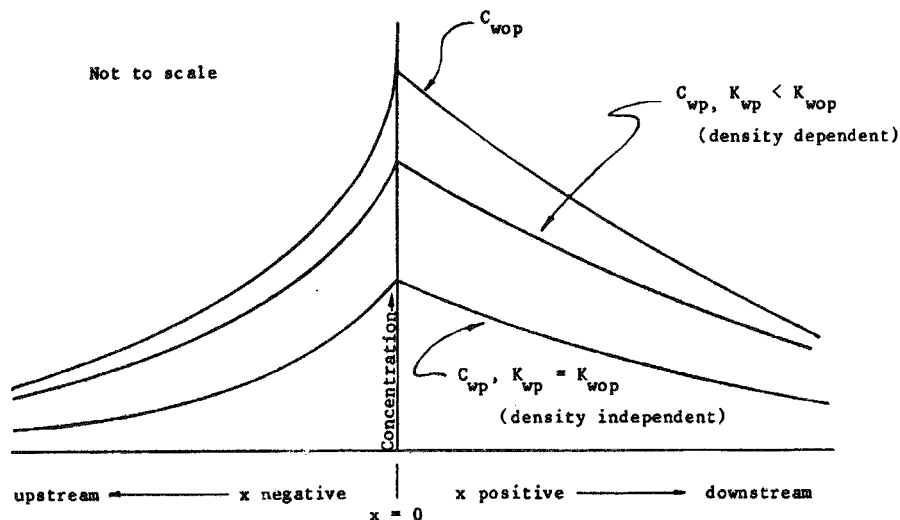


Figure 1.5-9. Relative behavior of density-dependent and density-independent mortality.

To illustrate the quantitative effect of compensation, calculate the maximum impact for the same conditions as used in illustrating Eq. 1.5-32, but consider a 5% decrease from the K_{wp} rate to account for compensation.

The maximum impact will occur at $x = 0$ (see Eq. 1.5-33 and Figure 1.5-9) so the expression to be evaluated is Eq. 1.5-33 at $x = 0$, or:

$$\text{IMPACT (max.)} = \frac{f_c \left(\frac{Q_p}{Q} \right) + F(K_{wp}) - F(K_{wop})}{f_c \left(\frac{Q_p}{Q} \right) + F(K_{wp})} \quad (1.5-34)$$

in which: $f_c (Q_p/Q) = 0.43 \times 0.31 = 0.13$

$$F(K_{wp}) = \sqrt{1 + 4K_{wp} \cdot E/U^2} = (1 + 16.5 \times 0.95)^{1/2} = 4.08$$

$$F(K_{wop}) = \sqrt{1 + 4K_{wop} \cdot E/U^2} = (1 + 16.50)^{1/2} = 4.18$$

This yields a maximum impact value of 0.7%, by comparison to the 3% obtained for the no compensation case (Eq. 1.5-32). This illustrates that, at least in estuaries where the mixing phenomenon makes the decay rate operative at all points, relatively small changes in unit natural mortality rates appear to be able to substantially offset losses due to cropping. This appears to make sense, particularly in that entrainment would appear to be purely a cropping phenomenon, with no deleterious changes imposed on the habitat (water or bottom substrate).

Equations similar to Eq. 1.5-30 and 1.5-31 can also be derived for the various river models, all of which will contain multipliers of the form $\exp(-K/U)x$ to describe the downriver decay of organism concentration due to natural mortality. The same approach described above may be used to evaluate possible compensation effects.

1.5.4 Lake, Harbor and Bay Models

The completely mixed notion is an adequate starting point to discuss models of enclosed or semi-enclosed water bodies whose volume to throughput ratio, or residence time, is relatively large by comparison to rivers or estuaries.

Consider Figure 1.5-10. Freshwater runoff, Q , and associated organisms, C_o , enter the water body from outside, W organisms per day are generated, and organisms leave the water body in the effluent stream at a concentration C_e . Organisms die at the unit decay rate K , which is considered to be operative over the entire water body volume V . Estuary average concentration is $\bar{C}$.

Application of Eq. 1.5-1 to this system yields:

$$QC_i - QC_e + W - \int_V K(c) C(x,y,z,t) dv = d\bar{C}V/dt \quad (1.5-35)$$

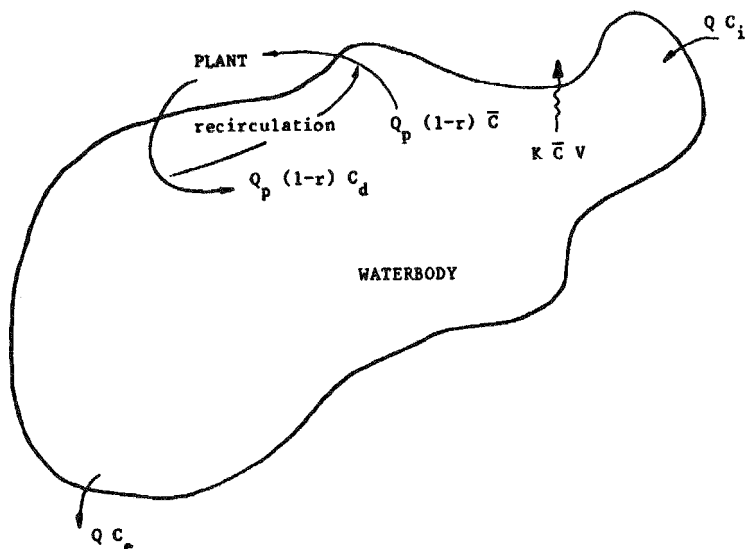


Figure 1.5-10. Schematic for Lake, Harbor, and Bay Models

Eq. 1.5-35 is a reasonably general expression for overall waterbody behavior. It is time variable, considers volume fluctuations with time, and recognizes spatially variable concentration and density-dependent mortality rates. When the volume is relatively constant and either the spatial variation is small or K is not a function of C , it reduces to:

$$QC_i - QC_e + W - K\bar{C}V = \frac{Vd\bar{C}}{dt} \quad (1.5-36)$$

The case of small spatial variation means the water body is essentially completely mixed and $C_e = \bar{C}$. For this case, Eq. 1.5-36 is rewritten:

$$QC_i + W - (Q + KV) \bar{C} = V \frac{d\bar{C}}{dt} \quad (1.5-37)$$

A whole host of time variable water quality and organism behavior can be described by Eq. 1.5-37. So long as K is not density-dependent (it can be time variable) and W , if a function of C , is a linear one, Eq. 1.5-37 can be solved very directly in closed form (1.5-2). Even if non-linear, numerical solutions can be obtained relatively easily.

This discussion here is limited to the steady-state form of Eq. 1.5-37. This is:

$$QC_i + W - (Q + KV) \bar{C} = 0 \quad (1.5-38)$$

Rearrange to solve C and obtain:

$$\bar{C} = \bar{C}_{wop} = \frac{W + QC_i}{Q + KV} \quad (1.5-39)$$

Now, consider plant operations. Include local recirculation in a manner identical to the treatment of this factor under River Model #3. The steady-state form of Eq. 1.5-1 becomes:

$$QC_i - Q\bar{C} - Q_p (1-r) \bar{C} + Q_p (1-r) C_d + W - KCV = 0 \quad (1.5-40)$$

Evaluate C_d in terms of f_c and $\bar{C}$, in accordance with Eq. 1.5-19, and then solve Eq. 1.5-40 for $\bar{C}$ to obtain:

$$\bar{C}_{wp} = \frac{W + QC_i}{Q + KV + Q_p (1-r) \left[\frac{f_c}{1-r + rf_c} \right]} \quad (1.5-41)$$

Apply Eq. 1.5-5 to Eq. 1.5-39 and 1.5-41 to obtain:

$$IMPACT = \frac{1}{1 + \frac{(Q + KV) (1-r + rf_c)}{Q_p (1-r) f_c}} \quad (1.5-42)$$

Note the striking comparison of this form to Eq. 1.5-28. If local recirculation were to be included in the model leading to Eq. 1.5-28, f_c in Eq. 1.5-28 would take on the same "r" multipliers that appear in Eq. 1.5-42 (this can be seen by observation of Eq. 1.5-5 and 1.5-22). Actually, the model leading to Eq. 1.5-28, with local recirculation included, will yield Eq. 1.5-42 if written over the finite volume under x in Figure 1.5-7, and if a decay rate K is operative within that volume.

Applications of Eq. 1.5-42 and the model depicted in Figure 1.5-10 usually result in the quantity KV being large by comparison to Q . KV/Q_p is often very substantially greater than unity so that impacts are often negligible.

1.5.5. Summary

A number of idealized models for rivers, estuaries and embayments have been presented. The techniques are applicable to the construction of more complex schemes that may be necessary for site-specific applications. Emphasis has been placed on the role of the physical control parameters, with some indication of how these interact with biological parameters. Impact is presented in terms of the immediate and direct changes that may be seen in the water body after entrainment. With the exception of a few remarks on the role of compensation, no attempt is made herein to assess the significance of these impacts on the ecosystem. The approach is considered to constitute "Initial Analysis," the first step an investigator might consider in placing a given entrainment situation into a quantitative perspective.

1.6 USE OF LENGTH DATA TO ASSESS ENTRAINMENT/IMPINGEMENT EFFECTS

1.6.1 Introduction

In the previous section several idealized models were presented for the estimation of entrainment impact. These models emphasized the role of physical measurements and required a minimum of (biological) sampling for implementation. Since many of the physical measurements discussed are relatively easy to obtain, such models are extremely useful for gaining a quantitative perspective of impact during the early assessment stages. However, to obtain more precise estimates of entrainment impact at a particular site, considerably more information may be needed, especially where the hydrodynamics are complex and simple assumptions (such as complete mixing in plant vicinity and steady-state conditions) are not valid.

In addition, some of the biological complexities that enter into the estimation of entrainment impact were only briefly addressed in construction of the models. For example, in some of the estuary and lake/bay models, the unit natural mortality rate (K) appeared in the impact equation. One of the previously discussed problems was the possibility that the magnitude of this rate may vary with plant operating conditions, and biological factors. When spawning occurs over a prolonged time period, and if (as is likely) different natural mortality rates are associated with each life history stage, there is the additional difficulty that generation and natural mortality cannot simply be described by constant rate values.

Behavioral plasticity of entrainable ichthyoplankton presents another possible source of error. For example, spot (Leiostomus xanthurus) in the Cape Fear Estuary have been reported to show strong diel vertical migration and interaction with tidal flow, which allow them to select and stay within a specific portion of the estuary. This ability has been observed in individuals as small as 8 mm standard length. Croaker (Micropogon undulatus) in the same estuary show similar capacity at standard lengths as small as 5 mm (1.6-1). Thus, the assumption that ichthyoplankton can be treated as non-motile, passively drifting particles may not always be valid. The increased mobility and habitat selection of older individuals make impingement estimation through the use of simple physical models even less realistic.

1.6.2 General Technique

The following is a general method for assessing the fractional reduction in cohort abundance due to plant operation during the first year of life. Unlike the previous methods which stressed physical measurements, biological measurements are emphasized herein. This method represents a combination and modification of several techniques and has been designed to include some of the biological complexities discussed above. Factors which may be accounted for in this analysis are prolonged duration of spawning, behavioral plasticity of larvae, and differential growth and natural mortality rates associated with various larval life stages. A major disadvantage of this technique is the high cost of the large amount of sampling required for implementation.

The method is applicable to the analysis of an entire closed system. A closed system is one in which, for the segment of the population under study, no immigration into or emigration out of the study area occurs. The implications of this assumption not being met are discussed below.

Data requirements include estimates of numbers entrained and impinged, and absolute abundance of the source waterbody population, each stratified temporally, spatially, and by length category. Source waterbody abundance estimates may be obtained through catch-per-effort methods designed to measure absolute density (numbers per-unit-volume) of organisms. This necessitates use of a gear which captures individuals of all lengths (within the population segment of interest) with equal efficiency, and for which there is no associated gear avoidance by individuals. However, gear avoidance appears to be an inescapable phenomenon, and is generally believed to intensify with increasing organism size. Thus, either a gear efficiency study to adjust for such avoidance may be undertaken, or alternatively, a gear may be chosen which minimizes such avoidance; at any rate, limitations of data obtained must be recognized.

Mark-recapture techniques (Section 1.4) provide alternative methods for estimating population abundance. However, the fragility of larvae may often lead to the violation of one of the basic assumptions underlying the use of such techniques: that marked and unmarked organisms have equal probability of survival. Sampling designs requiring multiple markings and recaptures are even less tenable for this reason.

If organisms are not uniformly distributed throughout the water body, the length frequency distribution for a given date should be adjusted by weighting the distributions found at particular sampling stations by the relative density of organisms found there. Also, sampling frequency and the size of length intervals should be chosen to assure that individuals make the transition from one length category to the next between sampling periods. In addition to some preliminary knowledge of growth rates, this may require setting uneven length intervals and/or varying sampling frequency with time. A priori estimates of growth rates may be obtained through a literature search, a pilot field study, or laboratory experiments. Finally, newly "hatched" individuals must be distinguishable from others on the basis of length (i.e. the range of newly hatched larval lengths must be small).

Given that the assumptions outlined above have been met and the required information gathered, it is now possible to obtain estimates of the following:

1. "initial" year-class abundance (eggs, or newly hatched larvae, as appropriate)
2. total number of individuals attaining each length category, L_i
3. length-specific growth rates
4. total mortality rate between length categories
5. length-specific power plant exploitation rates
6. length-specific and total annual fractional reduction in abundance due to power plant operation

The analyses utilized in obtaining items (1) through (4) are adaptations of techniques presented by Hackney and Webb (1.6-2), Hackney (1.6-3) and, to a lesser extent, Pearcy (1.6-4). Items (5) and (6) are obtained through application of techniques presented in Section 5.5.2, adapted from Ricker (1.6-5).

Estimates of length-specific growth and mortality rates are desirable because they provide a greater degree of understanding of larval dynamics and hence allow for more precise impact estimates than could otherwise be obtained. In general, the expected trend is increased probability of survival with increasing length. The mechanism believed to be responsible for this trend is size-selective predation on larvae. As larvae grow, mobility increases, and hence, the ability to avoid predators is improved. As an example, this mechanism has been proposed as an explanation for increased survival of winter flounder larvae (Pseudopleuronectes americanus) with increasing length. Laboratory experiments showed that larger larvae were better able to avoid capture by the medusoid Sarsia tubulosa than

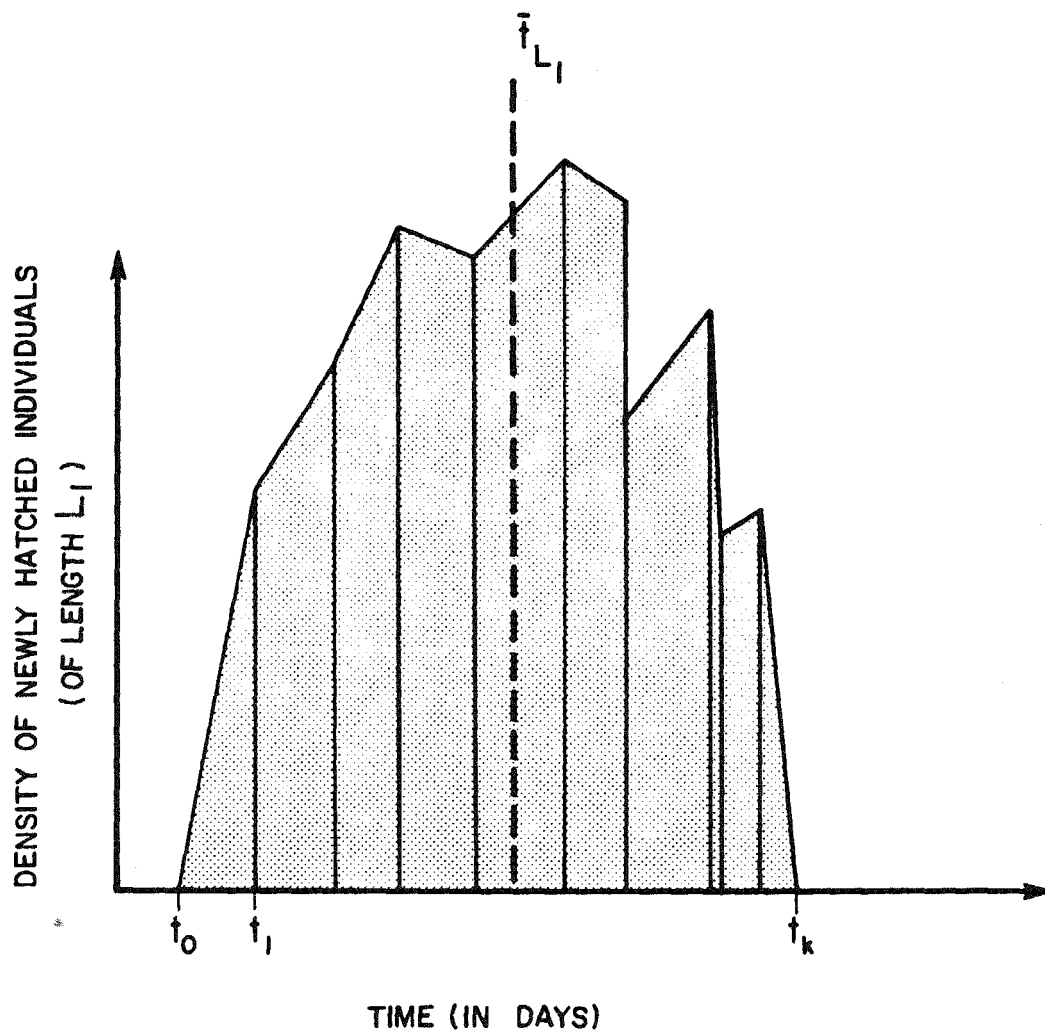
smaller larvae. S. tubulosa is believed to be the major predator of winter flounder larvae (1.6-4). Another predation mechanism that could account for increased survival of larvae with increasing length is: the larger the larva, the smaller the number and kinds of predators able to capture and handle it, and hence, the greater the expected survival.

"Initial" year-class abundance, for the purpose of this analysis, is the abundance of newly-spawned individuals, measured as the (cumulative) number of individuals of the earliest life stage vulnerable to entrainment or impingement. Thus, if entrainment of eggs occurs, "initial" abundance would be measured as the total number of eggs spawned. Estimates of total spawn may be obtained through measurements of egg density in the source water body. Alternatively, it is possible to obtain indirect estimates of total spawn through estimates of the size of the spawning stock, mean fecundity, and egg-retention rate.

For some species, "initial" abundance may be measured as cumulative abundance of a particular length category. For example, species which spawn demersal eggs may first become vulnerable to entrainment in the larval "swim-up" stage following hatching, e.g. larval crappie (1.6-3). The analysis below is described for a hypothetical population whose members are first entrained as post-hatch larvae. However, it is easily modified for egg entrainment.

The term "length category" is used loosely in subsequent analyses. Individuals of some species may metamorphose (e.g. from yolk-sac to post-yolk-sac form) without any concurrent change in size. Thus, where appropriate, the term "life stage" may be substituted for length category.

The length interval for newly hatched larvae is denoted L_i . Estimates of total numbers of newly hatched larvae by sampling date may be plotted as in Figure 1.6-1. Recall that sampling frequency has been chosen so that post-hatch larvae counted on one date are independent of those counted on previous or subsequent dates. Abundance on a given sampling date, (denoted N_{L_i} , t_i) represents the daily hatching rate on day t_i . The areas under the curve in Figure 1.6-2 thus provide an estimate of total number of individuals hatched. This quantity may be found by using the trapezoid summation method as indicated in Figure 1.6-3, and is denoted N_{L_i} . In addition, the average hatch date may be found as the mean time at which an individual hatched and is indicated on Figure 1.6-3 as $\bar{t}_{L_i}$. More precisely,



$\square = N_{L_I}$

Figure 1.6-1. Densities of newly hatched individuals on a per sampling basis for a hypothetical population.

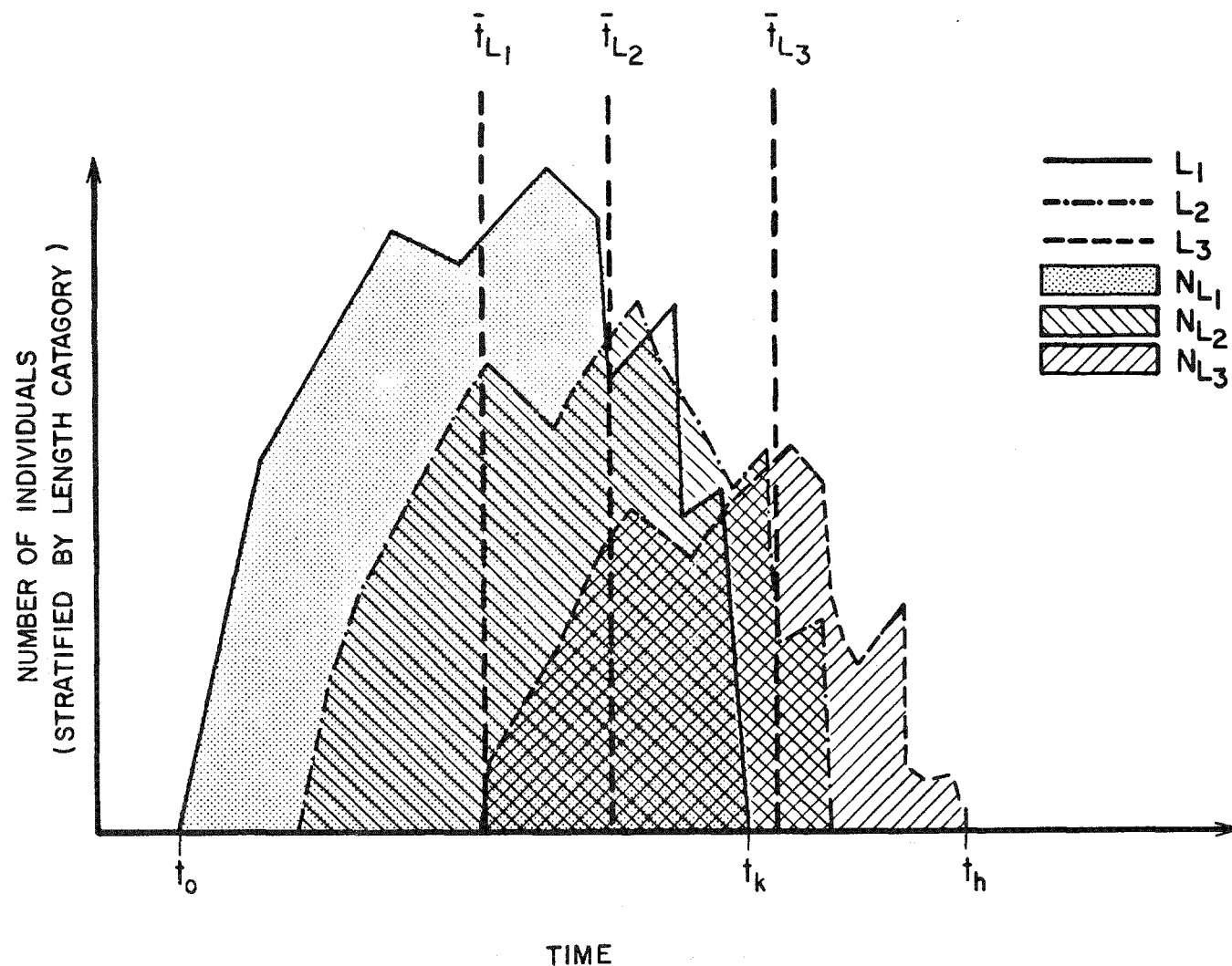


Figure 1.6-2. Number of individuals through time, stratified by length, for three selected length categories. Total number of individuals attaining length L_i is given by the area under each curve, N_{L_i} . Mean time at which an individual attains length L_i is denoted $\bar{t}_{L_i}$.

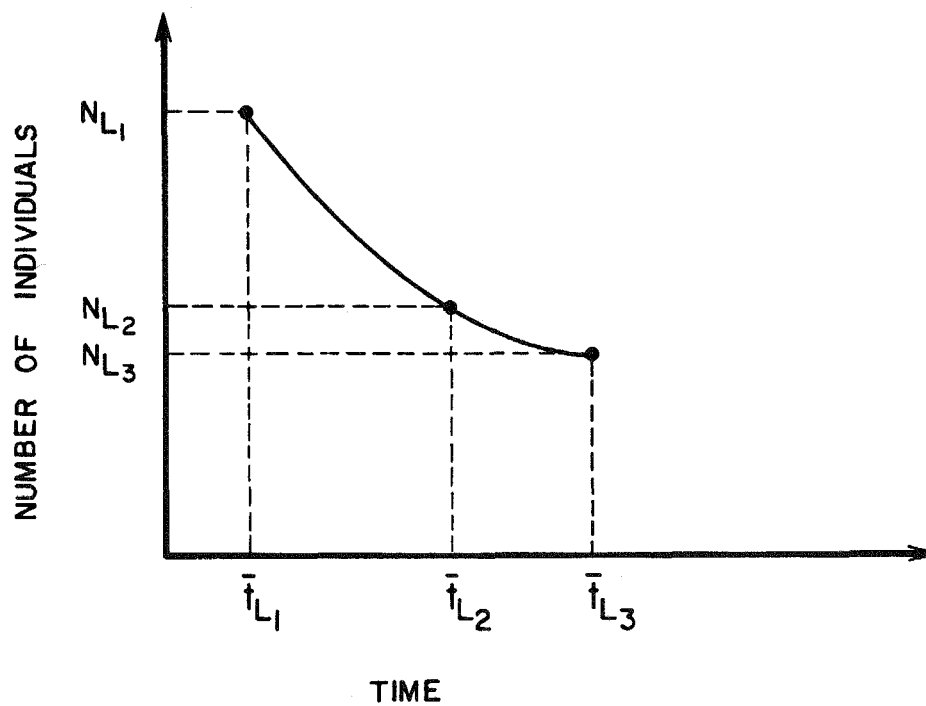


Figure 1.6-3. Rearrangement of data in Figure 1.6-2 presenting larval population behavior as following that of the "average" individual, e.g., all individuals present at time $\bar{t}_{L_i}$ are shown as belonging to category L_i .

$$\bar{t}_{L_i} = \frac{\sum_{i=0}^k (N_{L_i, t_i}) (t_i)}{\sum_{i=0}^k (N_{L_i, t_i})} \quad (1.6-1)$$

The method outlined above may be used repeatedly to obtain, for each length category L_i , total number of individuals attaining length L_i , (N_{L_i}) and the mean date on which an individual attains length L_i , ($\bar{t}_{L_i}$). Figure 1.6-2 illustrates this for three consecutive length categories of a typical population.

It is now possible to calculate the average time it takes an individual in length category L_i to attain length L_{i+1} , as $(\bar{t}_{L_{i+1}} - \bar{t}_{L_i})$, denoted T_{L_i} . In addition, since total numbers of individuals attaining each length category are now known, the total mortality rate, denoted A_{L_i} , between length intervals L_i and L_{i+1} is calculated as:

$$A_{L_i} = 1 - \frac{N_{L_{i+1}}}{N_{L_i}} \quad (1.6-2)$$

This completes the description of the estimating procedure for listed items (1) through (4).

It is now necessary to integrate information obtained in the source water body larval population with estimates of numbers entrained and impinged. One way to do this would begin with rearrangement of the time scale for the source waterbody population data, so that the population's behavior is presented as mimicking that of the "average" individual. Thus, the frequency distribution of numbers of individuals in a specific length interval L_i through time is "collapsed" to a point, representing total number of individuals in that length interval at time E_{L_i} . This procedure is illustrated by Figure 1.6-3, which is a rearrangement of the data presented in Figure 1.6-2 in the manner just described. In addition, it is assumed that loss of individuals during the time period $(\bar{t}_{L_i}, \bar{t}_{L_{i+1}})$ occurs in negative exponential fashion. This is equivalent to the standard assumption that:

$$\frac{dN_{L_i}(t)}{dt} = k_{L_i} N_{L_i}(t) \quad \begin{cases} t = t_{L_{i+1}} \\ t = t_{L_i} \end{cases} \quad (1.6-3)$$

where:

$N_{L_i}(t)$ = number of individuals in length category i at time t
 k_{L_i} = (constant) instantaneous total mortality rate applicable during the time interval $(\bar{t}_{L_i}, \bar{t}_{L_{i+1}})$ (equivalent to time of the length interval (L_i, L_{i+1})).

This assumption is consistent with the frequent observation of concave curves when N_{L_i} is plotted against $\bar{t}_{L_i}$.

All length-specific information can now be considered as time-specific, and the analysis can proceed as outlined in Section 5.5.2 of the methodology. Rather than review this analysis, its use will be illustrated by example.

Table 1.6-1 provides data for a hypothetical population during the first twelve days of life following hatching. Total mortality rate, A_{L_i} , plant exploitation rate, U_{L_i} and conditional power plant mortality rate, M_{L_i} , are calculated as in Section 5.5.2 of the methodology, with the exception that they are computed on a per-length category (rather than actual time) basis. For this example, total fractional reduction due to plant operation prior to attaining 9.0 mm is given by:

$$M = 1 - \prod_{i=1}^5 (1 - M_{L_i}) = 0.00167 \quad (1.6-4)$$

Various modifications of this basic approach may be made, depending on the particular species under study. For example, Hackney and Webb (1.6-2) found that the expression:

$$\frac{dN(t)}{dt} = -Z N(t) \quad (1.6-5)$$

was sufficient to predict larval abundance of crappie during the entire period of vulnerability. Whereas, Percy (1.6-4) found an expression of the form:

Table 1.6-1

DATA FOR A HYPOTHETICAL LARVAL POPULATION DURING THE FIRST TWELVE DAYS OF LIFE

Age in Days	Length Interval (Standard Length in mm)	Numbers Attaining Length Interval (N_{L_i})	Numbers Entrained or Impinged	Total Mortality Rate (A_{L_i})	Plant Exploitation Rate (u_{L_i})	Conditional Power-Plant Mortality Rate (m_{L_i})
0.0	$L_1 = (4.0-4.9)$	1,232,015,000	500,000	.291	.0004	.00047
1.7	$L_2 = (5.0-5.9)$	873,151,467	458,763	.195	.0005	.00056
4.2	$L_3 = (6.0-6.9)$	702,820,813	212,061	.153	.0003	.00033
8.1	$L_4 = (7.0-7.9)$	595,381,652	100,032	.123	.0002	.00021
10.0	$L_5 = (8.0-8.9)$	522,129,550	57,168	.062	.0001	.00010
12.2	$L_6 = (9.0-9.9)$	489,673,987				

$$\frac{dN(t)}{dt} = -[a - bl(t)] N(t) \quad (1.6-6)$$

where:

$l(t)$ = length at time t .

to be in better agreement with data observed for winter flounder.

In addition, since data on length and time to attain a specific length are obtained, it may be possible to derive growth equations applicable for describing larval growth. Such expressions are generally not of the form characteristic of adult fish. If data are available for several years, and if information on key environmental factors has also been recorded, it may be possible to derive expression for larval growth and mortality as functions of density and the environmental factors. Such expressions would prove invaluable for assessing long-term impact, as they would represent the first step in the quantification of compensatory ability. Unfortunately, because of characteristically high levels of environmental (and biological) variation, this would require a very long time series of data.

Some limitations of the analysis described above should be noted. First, recall that average growth and mortality rates were obtained, and thus, much of the natural variation in these rates due to factors other than length has not been accounted for. For example, if spring spawning occurs over a prolonged period, larvae hatched later in the season may have a better chance of survival (e.g. due to increased temperature) than those which hatched earlier. Also, the averaging procedure does not allow for the analysis of inter-length class competition. These factors may be accounted for through more intensive sampling and aging procedures. However, costs of such analyses will often be prohibitive.

Also note that several fairly stringent assumptions were made regarding the source waterbody population. If the assumption of a closed population is not met, migration rates must be accounted for. This may be done in a variety of ways. For example, Pearcy (1.6-4) computed differential seaward movement of various water strata, as well as vertical length-frequency distribution of larvae, to estimate net seaward translocation rates.

1.7 THE LESLIE MATRIX MODEL

1.7.1 Introduction

The Leslie matrix model, first presented by Leslie (1.7-1, 1.7-2), is a convenient method by which the discrete time change (usually in units of one year) in population abundance and age structure may be simulated. The population matrix elements include age-specific fecundity and survival rates, which are presumed to be constants in the basic Leslie matrix.

The model has limited utility in modeling long-term impact because of the assumption of time-invariant (and hence, density-independent) rates. However, modifications of the basic model may be made which extend the time span for prediction. In addition, in certain appropriate instances, the basic approach can provide useful information. The discussion below includes a description of the basic Leslie matrix approach, applications, and possible modifications.

1.7.2 Basic Form of the Leslie Matrix Model

First, terms are defined:

Let $n_i(t)$ = the number of individuals in age class i at time t ,

Age classes are segregated by year class, and time is measured in discrete one-year units. An "oldest age" must then be chosen. This age need not be the actual oldest age extant in the population, but rather, may be an age old enough so that the number of individuals generally attaining that age and their reproductive contribution are insignificantly small in relation to the rest of the population. If k represents the number of age classes, then the population can be represented at any time t in vector form, where the i th element of the vector is $n_{i-1}(t)^*$. This is termed the population vector and denoted $\tilde{N}(t)$. Thus:

$$\tilde{N}(t) = \begin{pmatrix} n_0(t) \\ n_1(t) \\ n_2(t) \\ \vdots \\ \vdots \\ n_{k-2}(t) \\ n_{k-1}(t) \end{pmatrix}$$

*This is because, by convention, the first age class is usually called the "0th" age class; thus the first vector element is $n_0(t)$.

Now assume that age-specific survival and fecundity are time invariant.

Let p_i represent the probability that an individual of age i survives till age $i+1$. Let q_i be the probability of death for an individual in the i th age class (i.e., $q_i = 1 - p_i$). Similarly, let f_i equal the number of 0-age offspring produced by an individual of age i . Note that since f_i is defined per individual without regard to sex:

$$f_i = f_{f_i} \cdot f_{m_i} \cdot E_{f_i} \quad (1.7-1)$$

where:

f_{f_i} = the fraction of i year olds which are female

f_{m_i} = the fraction of i year old females that are mature

E_{f_i} = the average number of eggs produced per mature female

As seen by Eq. 1.7-1 the assumption that all f_i are time invariant is equivalent to the assumption that for each age class, sex ratio, fecundity, and average state of maturity are constant. To further simplify, assume that all reproduction occurs at one, well-defined season. Finally, define l_i as the probability that a newly-born individual survives till age i (e.g. $l_4 = p_0 p_1 p_2 p_3$; $l_0 = 1$).

Now, it is of interest to find the population vector at time $t+1$, given the population vector at time t , and the age-specific mortality and fecundity rates.

To do this, note that by definition:

$$n_i(t+1) = n_{i-1}(t) p_{i-1} \quad (1.7-2)$$

Also, if spawning is assumed to occur immediately prior to time $t+1$ then:

$$n_0(t+1) = \sum_{i=1}^{k-1} n_i(t+1) f_i \quad (1.7-3)$$

and, substituting the right-hand side of Eq. 1.7-2 for $n_i(t+1)$ in Eq. 1.7-3 gives:

$$n_0(t+1) = \sum_{i=1}^{k-1} n_{i-1}(t) p_{i-1} f_i \quad (1.7-4)$$

and hence:

$$\tilde{N}(t+1) = \begin{pmatrix} \sum_{i=1}^{k-1} n_{i-1}(t) p_{i-1} f_i \\ n_0(t) p_0 \\ n_1(t) p_1 \\ \vdots \\ n_{k-2}(t) p_{k-2} \end{pmatrix}$$

Now note that this result could have been obtained through multiplication* of the following matrices:

$$\begin{pmatrix} F_0 & F_1 & \cdot & \cdot & \cdot & F_{k-2} & F_{k-1} \\ p_0 & 0 & \cdot & \cdot & \cdot & 0 & 0 \\ 0 & p_1 & & & & \cdot & \cdot \\ \cdot & 0 & & & & \cdot & \cdot \\ \cdot & \cdot & & & & \cdot & \cdot \\ 0 & 0 & \cdot & \cdot & \cdot & p_{k-2} & 0 \end{pmatrix} \begin{pmatrix} n_0(t) \\ \cdot \\ \cdot \\ \cdot \\ \cdot \\ n_{k-1}(t) \end{pmatrix}$$

where

$$F_i = p_i f_{i+1}$$

With the left-hand matrix denoted as M , this can more simply be written as:

$$\tilde{N}(t+1) = M \tilde{N}(t) \quad (1.7-5)$$

* The reader unfamiliar with matrices and basic matrix operations will find a clear discussion with a biological orientation in Searle (1.7-3).

The matrix M , with constant values for age-specific survival and fecundity is known as the Leslie matrix. There are modifications of the Leslie matrix, useful when reproduction occurs at several times in the year, or is spread over a portion of the year. In addition, it may often be simpler to consider only females and to apply the matrix to this sex for which fecundity is clearly defined. The present form, appropriate for seasonal spawners, is the one usually encountered in the fisheries literature. Note that:

$$\tilde{N}(t) = M \tilde{N}(t-1) = \dots = M^k \tilde{N}(t-k)$$

through repeated application of Eq. 1.7-5. Thus, for a population with time invariant age-specific survival and fecundities, the Leslie matrix provides the basic algorithm for obtaining the population vector at any time, given some initial population vector. In addition, this approach is useful for simulating the course of population abundance through time. This is illustrated in the following examples.

Example 1

Consider a hypothetical population whose members live for a maximum of three years, and the following information is available:

Age (i)	p_i	f_i	$n_i(0)$	l_i
0	0.05	0.00	0	1.000
1	0.90	18.60	10,000	0.050
2	0.00	1.55	0	0.045

The corresponding Leslie matrix for this population is:

$$M_1 = \begin{pmatrix} 0.93 & 1.40 & 0.00 \\ 0.05 & 0.00 & 0.00 \\ 0.00 & 0.90 & 0.00 \end{pmatrix}$$

and the initial population vector is:

$$\tilde{N}(0) = \begin{pmatrix} 0 \\ 10,000 \\ 0 \end{pmatrix}$$

To follow the population vector through time, recall that, by Eq. 1.7-5:

$$\tilde{N}(t+1) = M_1 \tilde{N}(t), \text{ hence:}$$

$$\tilde{N}(1) = M_1 \tilde{N}(0) = \begin{pmatrix} 14,000 \\ 0 \\ 9,000 \end{pmatrix}$$

$$\text{and setting } N_{TOT}(t) = \sum_{i=0}^2 n_i(t), \text{ gives:}$$

$$N_{TOT}(0) = 10,000; \quad N_{TOT}(1) = 14,900$$

Continuing as above:

$$\tilde{N}(2) = M_1 \tilde{N}(1) = \begin{pmatrix} 13,020 \\ 700 \\ 0 \end{pmatrix}; \quad N_{TOT}(2) = 13,720$$

$$\tilde{N}(3) = M_1 \tilde{N}(2) = \begin{pmatrix} 13,088.6 \\ 651.0 \\ 630.0 \end{pmatrix}; \quad N_{TOT}(3) = 14,369.6$$

$$\tilde{N}(4) = M_1 \tilde{N}(3) = \begin{pmatrix} 13,083.80 \\ 654.43 \\ 585.90 \end{pmatrix}; \quad N_{TOT}(4) = 14,324.13$$

..... and so on.

Example 2:

Given the following information:

Age (i)	p_i	f_i	$n_i(0)$	l_i
0	0.05	0	5000	1.00
1	0.20	0	800	0.05
2	0.00	100	120	0.01

The corresponding Leslie matrix is:

$$M_2 = \begin{pmatrix} 0.00 & 20.0 & 0.00 \\ 0.05 & 0.0 & 0.00 \\ 0.00 & 0.2 & 0.00 \end{pmatrix} \text{ and}$$

$$\tilde{N}(0) = \begin{pmatrix} 5,000 \\ 800 \\ 120 \end{pmatrix} \quad N_{TOT}(0) = 5,920$$

The population is followed for four more consecutive years as before:

$$\begin{aligned} \tilde{N}(1) &= M \tilde{N}(0) \\ &= \begin{pmatrix} 16,000 \\ 250 \\ 24 \end{pmatrix} ; \quad N_{TOT}(1) = 16,274 \end{aligned}$$

$$\tilde{N}(2) = \begin{pmatrix} 5,000 \\ 800 \\ 50 \end{pmatrix} ; \quad N_{TOT}(2) = 5,850$$

$$\tilde{N}(3) = \begin{pmatrix} 16,000 \\ 250 \\ 160 \end{pmatrix} ; \quad N_{TOT}(3) = 16,410$$

$$\tilde{N}(4) = \begin{pmatrix} 5,000 \\ 800 \\ 50 \end{pmatrix} ; \quad N_{TOT}(4) = 5,850$$

A plot of total number of individuals through time for the two populations is presented in Figure 1.7-1. Note that both age distribution and total abundance oscillate in varying degrees for both populations. It should now be asked whether these oscillations will damp out in time and whether a stable age distribution and/or stable population level be reached.

To add mathematical formalism to these questions, first note that a stable age distribution is one where the relative proportion of individuals in each age class remains constant, or equivalently, one for which the population vector at a particular time is simply a constant multiple of the population vector at the immediately preceding time. Thus, a stable age distribution exists only if there exists a scalar (i.e, constant) λ , and a population vector $\tilde{N}_t$ such that:

$$\tilde{N}(t+1) = M \tilde{N}(t) = \lambda \tilde{N}(t) \quad (1.7-6)$$

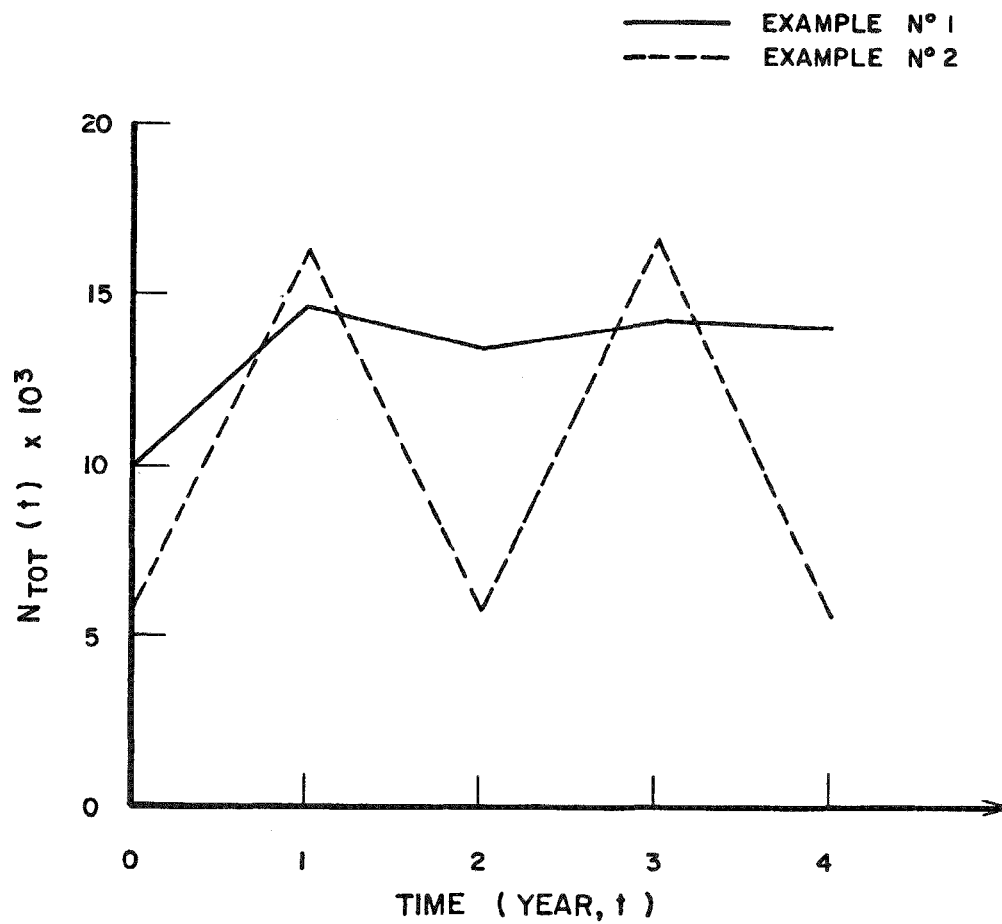


Figure 1.7-1. Plot of $N_{TOT}(t)$ vs t for the two hypothetical population discussed.

It can be shown (1.7-3) that any scalar which satisfies Eq. 1.7-6 is one which also satisfies the equation*:

$$|M - \lambda I| = 0 \quad (1.7-7)$$

Eq. 1.7-7 is called the characteristic equation. For a matrix with n rows and n columns, the characteristic equation is a polynomial in λ of degree equal to or less than n . In the present case, the $k \times k$ Leslie matrix generally yields a polynomial of degree $k-1$. Hence, it has $k-1$ solutions, say, $\lambda_1, \lambda_2, \dots, \lambda_{k-1}$. The λ_i 's are known as the latent roots (or eigenvalues) of the matrix M . However, it can also be shown that only one of the $k-1$ latent roots is positive and real valued (1.7-4), and hence only one λ_i is applicable for a biologically feasible population (i.e. there cannot be a negative or imaginary numbers of individuals in an age class). Denote this latent root as R . If:

$$|R| > |\lambda_i| \text{ for all } \lambda_i \neq R. \quad (1.7-8)$$

then the population will converge on a stable age distribution, and R is termed the dominant latent root.

Thus, to find the stable age distribution corresponding to a population with Leslie matrix M , first solve the characteristic equation to find all λ_i , (and hence R). If R also meets condition Eq. 1.7-8 proceed to obtain the stable age distribution vector as follows.

First recall that R is a solution of Eq. 1.7-6 so that:

$$n_i(t) = R n_i(t-1) = R^2 n_i(t-2) = \dots = R^i n_i(t-i) \quad (1.7-9)$$

So, for example:

$$n_0(t-i) = n_0(t) R^{-i} \quad (1.7-10)$$

*The symbol $|A|$ takes on different meanings, depending on the form of A . If A is a matrix, $|A|$ represents the determinant of A . If A is a complex number, (i.e. $A = a + ib$), then $|A| = a^2 + b^2$ and is often called the modulus of A . Finally, if A is a real number, $|A|$ denotes the absolute value of A , namely $\sqrt{A^2}$.

Since survival probabilities are presumed to be time invariant, it follows that:

$$\begin{aligned} n_i(t) &= n_{i-1}(t-1) p_{i-1} = n_{i-2}(t-2) p_{i-2} p_{i-1} = \dots = n_0(t-i) p_{i-1} \dots p_0 \\ &= n_0(t-i) l_i \end{aligned} \quad (1.7-11)$$

Substitution of the right-hand side of Eq. 1.7-10 for $n_0(t-i)$ in Eq. 1.7-11 yields:

$$n_i(t) = n_0(t) R^{-i} l_i \quad (1.7-12)$$

So at stable age distribution, the proportion of individuals in the i th age class is given by:

$$C_i = \frac{n_i(t)}{\sum_i n_i(t)} = \frac{n_0(t) R^{-i} l_i}{\sum_i n_0(t) R^{-i} l_i} = \frac{R^{-i} l_i}{\sum_i R^{-i} l_i} \quad (1.7-13)$$

For a population that has achieved stable age distribution, note that:

$$\frac{N_{TOT}(t+1)}{N_{TOT}(t)} = R \quad (1.7-14)$$

Thus at stable age distribution, R is the annual geometric growth rate. Note that:

$R > 1$ implies that the population will increase indefinitely

$R < 1$ implies the population will decrease to 0

$R = 1$ implies equilibrium conditions

It is seen that a stable population abundance will be reached only if $R = 1$.

Calculation of the stable age distribution vector and corresponding annual geometric growth rate are illustrated for the previous examples.

Example 1

Recall that $M_1 = \begin{pmatrix} 0.93 & 1.40 & 0.00 \\ 0.05 & 0.00 & 0.00 \\ 0.00 & 0.90 & 0.00 \end{pmatrix}$

To find the eigenvalues of M_1 , all λ_i which satisfy must be found:

$$0 = |M_1 - \lambda_i I| = \begin{vmatrix} 0.93 - \lambda_i & 1.40 & 0.00 \\ 0.05 & -\lambda_i & 0.00 \\ 0.00 & 0.90 & -\lambda_i \end{vmatrix}$$
$$= \begin{vmatrix} 0.93 - \lambda_i & 1.40 \\ 0.05 & -\lambda_i \end{vmatrix} = \lambda_i^2 - 0.93\lambda_i - 0.07$$

The two solutions of the characteristic equation are:

$$\lambda_1 = \frac{0.93 + 1.07}{2} = 1.00$$

and

$$\lambda_2 = \frac{0.93 - 1.07}{2} = -0.07$$

Since λ_1 is positive and real valued, $\lambda_1 = R$

Note that condition Eq. 1.7-8 is satisfied, that is:

$$|R| = |\lambda_1| = 1.00 > |\lambda_2| = 0.07;$$

Hence, a stable age distribution will be attained by this population. Also note that since $R = 1$, a stable population level will also be reached. The proportion of individuals in age class i is given by Eq. 1.7-13:

$$C_i = \frac{R^{-i} l_i}{\sum_i R^{-i} l_i}, \text{ and since } R = 1$$

$$C_i = \frac{l_i}{\sum_i l_i}$$

Therefore:

$$C_0 = 0.913 \quad C_1 = 0.046 \quad C_2 = 0.041$$

and the stable population vector is proportional to:

$$C = \begin{pmatrix} 0.913 \\ 0.046 \\ 0.041 \end{pmatrix}$$

Example 2

The Leslie matrix corresponding to this population is:

$$M_2 = \begin{pmatrix} 0.00 & 20.00 & 0.00 \\ 0.05 & 0.00 & 0.00 \\ 0.00 & 0.20 & 0.00 \end{pmatrix}$$

Solution of the characteristic equation yields:

$$0 = |M_2 - \lambda_i I| = \lambda_i^2 - 1$$

The eigenvalues of M_2 are:

$$\lambda_1 = 1 \text{ and } \lambda_2 = -1; \text{ hence } R = 1$$

However,

$$|R| = 1 = |\lambda_2|$$

Hence, condition (Eq. 1.7-8) is not satisfied, and no stable age distribution will be reached. Returning to the first part of example 2 shows that the population vector will continue to oscillate between:

$$\begin{pmatrix} 5,000 \\ 800 \\ 50 \end{pmatrix} \quad \text{and} \quad \begin{pmatrix} 16,000 \\ 250 \\ 160 \end{pmatrix}$$

in alternate years. This type of oscillation is often called a stable limit cycle of period 2.

1.7.3 Limitations and Applicability of the Leslie Matrix Approach

The major limitation of the Leslie matrix approach should now be apparent. In the assumption that time invariant (and hence, density-independent) values for age-specific survival and fecundity, the Leslie matrix approach implies a linear relationship between stock and recruitment at stable age distribution. Section H1.9 indicates that such a relationship is biologically unrealistic; thus, the utility of this approach may be questioned. However, there appear to be several situations when the basic model has some utility, and these are explored below.

Some of the populations under study may be perceived as fluctuating about some stable, equilibrium level. As long as the fluctuations are small, the range of population densities extant among years will be small, and the evocation of density-dependent responses will not be expected. For such populations, the use of constant survival and fecundity rates will approximate real population behavior.

One use of the Leslie matrix algorithm for species near equilibrium is the calculation of y-o-y survival probability, p_0 . As, for example, Vaughan and Saila (1.7-5) have shown, if age-specific (average) survival and fecundity rates are known for the adult stock, the p_0 can be calculated as:

$$p_0 = \frac{1}{f_1 + \sum_{i=1}^{k-1} \left[f_{i+1} \left(\prod_{j=1}^i p_j \right) \right]} \quad (1.7-15)$$

When $R = 1$

More generally:

$$p_0 = \frac{R}{f_1 + \sum_{i=1}^{k-1} \left[\frac{f_{i+1}}{R_i} \left(\prod_{j=1}^i p_j \right) \right]} \quad (1.7-16)$$

This estimate of p_0 requires a fair amount of information (namely f_i and p_i for each adult age class), and the accuracy of the estimate depends on the accuracy of the components of Eq. 1.7-14 and the validity of assuming an equilibrium condition. However, as Vaughan and Saila (1.7-5) show in their example on Atlantic bluefin tuna, an estimate of p_0 derived in this manner, where only rough estimates of adult mortality and fecundity rates are available, is preferable to the assumption that y-o-y natural mortality is on a par with adult mortality especially for highly fecund species. Most preferred, of course, are direct estimates of p_0 , obtained from a carefully designed sampling program. However, the procedure described above, where applicable, is useful for obtaining an indirect estimate of p_0 when a direct one is not available.

A modification of this technique would be to assume that adult mortality and fecundity vary randomly with a given distribution about mean values. Distributions may be determined empirically or through theoretical analyses. It is then possible to calculate confidence limits for the mean value of p_0 obtained through successive calculation. Vaughan (1.7-6) presents a method for obtaining confidence limits of $M_0 = -\log_e(p_0)$, which examines adult survival as a function of fishing effort and utilizes Monte Carlo simulation techniques.

Although the above only considers estimation of p_0 , the Leslie matrix algorithm could be used to find p_i or f_i for any age class, given information on all other parameters, and a fixed value of R assumed. The rationale for presenting the procedure for estimation of p_0 is that this parameter is generally the most difficult to estimate directly.

Another potentially useful application of Leslie matrices for populations near equilibrium is the study of relative population stability in response to proportional changes in y-o-y survival. Horst (1.7-7) empirically determined that " ...the more closely the eigenvalues of the Leslie matrix trace a circle of radius equal to the modulus of the maximal eigenvalue (i.e. dominant latent root), the less R changes, and thus the less the population size changes, in response to perturbations."

Perturbations considered are fractional reductions in the value of p_0 . He thus proposed an index of relative population instability, denoted here* as S , to be:

$$S = \frac{1}{k} \sum_{i=0}^{k-1} (|R| - |\lambda_i|) \quad (1.7-17)$$

Where:

k = number of age classes in the population

λ_i = the i th eigenvalue (latent root) of the Leslie matrix

R = the real, positive maximal eigenvalue of the Leslie matrix.

This index applies only when R satisfies Eq. 1.7-8 as described above. For $R = 1$, S can only take on values between 0.0 and 1.0. The closer S is to 0.0, the less sensitive R is to a given fractional reduction in p_0 , hence the more stable the population. Horst calculated this index for populations of winter flounder, cunner, menhaden and Atlantic silverside using data obtained from various literature sources. The relative sensitivities of these four species to the same fractional reduction in p_0 obtained by this analysis were found to be consistent with the theory of r and K selection, as first proposed by MacArthur and Wilson (1.7-8). It was suggested that in selecting species for study, consideration should be given to the sensitivity of populations to perturbations, of which S provides one measure as well as to the expected magnitude of the perturbation (i.e. the incremental impact).

Saila and Lorda (1.7-9) argue that excluding of compensatory phenomena in the basic Leslie matrix model is not unduly prohibitive for impact assessment. Their argument is based on two lines of reasoning:

1. The left-hand ascending limb of both the Ricker and Beverton-Holt stock-recruitment curves (see Section 1.9) may be approximated by a straight line. The ascending limb covers the range of stock densities of primary environmental concern; hence, the Leslie Matrix approach is applicable for studying the effects of plant impact at moderate stock densities.

* Horst (1.7-7) denoted his stability measure as I . Since I has been previously used to represent short-term impact (Section 5.5.2 of the methodology), S has been substituted to avoid confusion.

2. The basic Leslie matrix model has limited utility for long-term prediction. However, most environmental studies are concerned with population responses on a relatively short-time scale (e.g. 20 years for Saila and Lord's study). Hence, the Leslie model seems appropriate if not extended too long in time.

Although moderate to low stock densities represent the range of greatest environmental concern, particular populations under study may beat much higher abundance levels. It does not seem appropriate to use a model restricted in validity to the study of moderate stock densities for such populations. Moreover, for both the Ricker and Beverton-Holt curves, the left-hand ascending limb will generally occur above the replacement line, because, at moderate stock densities, an unexploited population will generally be increasing (i.e., annual geometric growth rate greater than 1.0). Thus, one or more of the p_i and/or f_i corresponding to equilibrium for a virgin stock must be smaller than the maximum rates possible.

However, many applications of the Leslie matrix model have used calibration techniques which force the dominant latent root to take on the value 1.0. In an equilibrium (rather than increasing) condition, the models predict fairly steady population decline, eventually leading to extinction, from any level of plant cropping. In contrast, a formerly unexploited equilibrium population would be expected to decline initially in response to plant cropping. Following a reduction in stock abundance, density-dependent responses in the parameters p_i and/or f_i would occur, and the population would stabilize at a new reduced level at which the rate of exploitation would balance the "surplus" rate of increase. Decline to extinction would only occur if exploitation exceeded the maximum potential "surplus" rate of increase.

Populations in equilibrium with an existing exploitation (e.g., commercial or sports fishery) will eventually stabilize at a new level when a new source of exploitation (entrainment and/or impingement) is added as long as the combined exploitation rate does not exceed the maximum to which the population can respond. It is clear that a Leslie matrix approach, further constrained to represent an equilibrium population, will not provide a realistic model for most populations. In addition, the Leslie matrix provides no clues on the maximum potential of populations to absorb impacts, nor does it predict the reduced stable stock level. However, it can provide information on the degree of change in survival and/or fecundity necessary for stabilization.

Saila and Lordas' argument that, since environmental studies are concerned with relatively short-term effects, Leslie matrices are appropriate if not extended too far in time, is also questionable. For example, 20 years may, in fact, be long enough to predict substantial reduction with a density-independent model and to evoke compensatory response mechanisms, depending on life-history characteristics of the particular population being studied. For such (relatively) short-term predictions, Leslie matrix results are more properly interpreted as providing the maximum possible level of reduction that could be expected to occur. This might be useful for extremely short-term prediction. Given the generally short time frame for assessment, it might be decided to allow plant operation and continue monitoring if the "worst-case" five-year reduction (as provided by the model) was predicted to be small. However, the longer the time period considered, the less useful the upper bound of reduction. In addition, the appropriate time period for extended predictions depends on the life of the particular plant being considered, which, in some cases, will exceed 35 years. Moreover, if replacement plants or additional plants are to be built in the future, a prediction of impact may be needed over a much greater time period. Thus, although the basic Leslie matrix model may provide useful insights, its limitations must be recognized in making inferences based on model results. In addition, it appears that models which incorporate density-dependent phenomena are needed for long-term prediction in many instances. The next section addresses the construction and limitations of such models.

1.8 POPULATION RESPONSE ANALYSIS

1.8.1 Introduction

Much of the effort in impact assessment typically focuses on attaining estimates of cropping mortality through a comparison of the number of fish entrained/impinged at power plants with estimates of the size of the total population residing in the source water body. While realistic estimates of cropping help to provide a picture of the actual magnitude of removals from a population, these estimates do not describe the actual response of the population to a given rate of removal of individuals. That is, the "impact" of such removals is not known from the cropping rate alone. In the final analysis, the biological response of the fish population is the key to the objective assessment of impact, and population response analysis is an important factor in the verification phase of models used to predict such impact. Population response parameters discussed in this section are growth, mortality, age structure, trends in abundance, trends in recruitment, fecundity and age at sexual maturity.

1.8.2 Somatic Growth

Density-dependent growth, especially in the early (first year or two) life stages, is one of the more powerful compensatory mechanisms existing within fish populations (1.8-1). Unlike most vertebrates, fish continue to grow throughout their life, and their growth is much more plastic and labile than growth in other animals, often reflecting changes in water temperature and competition level (1.8-2). The relationship between growth and population density is most often negative and there is abundant documentation of increased growth rates concomitant with increased exploitation rates (1.8-3, 1.8-4, 1.8-5, 1.8-6, 1.8-7). Growth is therefore a population response parameter which may serve as an indicator of the intensity of exploitation represented by entrainment and impingement and the capacity for increased exploitation in the event of power plant expansion. In addition, comparison of various aspects of growth may provide insight into size-selective mortality and population condition.

Information on fish growth is available from two basic sources: (1) actual mean length (or weight) of fish from age group y present in the population at time t ; (2) back calculation of length at annulus y from scales of fish surviving from time t to time $t+x$, where x is the amount of time elapsed between time t and the time when the fish is captured and scales are taken.

Each year of growth for a given fish is represented on the scales of that fish by an annulus, or winter ring, similar to the concentric rings used to determine the age of trees. By convention, the last annulus is synonymous with the outer edge of the scale after 1 January of each year, since that is where the annulus will form when growth resumes in the spring. This assumes that we are dealing with temperate-zone fishes whose growth ceases or slows down considerably during the winter season. This methodology is often not applicable to warm-water, tropical species which observe a year-round growing season.

The distance from the center or focus of the fish scale to each annulus is related to the growth of the fish during the growing season preceding that annulus. The amount of growth of a fish in each year of its life prior to capture is calculated by the equation

$$L' = C + [(S'/S) (L-C)] \quad (1.8-1)$$

where

L' = the length of fish when annulus y was formed

S' = the length of scale radius to annulus y

S = the length of total scale radius

L = the length of fish at the time of capture

C = the correction factor representing the y -intercept in the body-scale length regression

The age-growth formula is used in an iterative way to accumulate information from all fish analyzed during different collection years. Data grouping is conducted so that fish from the same year class or hatching year will maintain their integrity as contributors to a data point. In this way, a cumulative data file is built, and each succeeding year's data can be added. The following table (1.8-1) demonstrates this method of data grouping (according to only three years of collection: 1971, 1972, 1973):

Table 1.8-1

DATA GROUPING FOR SCALE ANALYSES IN SUCCESSIVE YEARS

Year Class or Hatching Year	Annulus At Which Fish Length Is Calculated			
	1	2	3	4
1973	No fish have annulus			
1972	1+cap* 1973			
1971	1+cap 1972 2+cap 1973	2+cap 1973		
1970	1+cap 1971 2+cap 1972 3+cap 1973	2+cap 1972 3+cap 1973		
1969	2+cap 1971 3+cap 1972 4+cap 1973	3+cap 1972 4+cap 1973	3+cap 1972 4+cap 1973	
1968	3+cap 1971 4+cap 1972 5+cap 1973	3+cap 1971 4+cap 1972 5+cap 1973	3+cap 1971 4+cap 1972 5+cap 1973	4+cap 1972 5+cap 1973

*1+cap 1973 denotes a fish which was one year old in January 1973 and was captured sometime during the course of 1973 sampling. Additional terms follow this example.

One of the assumptions that permit the use of scales to determine the length of a fish at successive years of its life is that the annual increment in scale length maintains a constant ratio with the annual increment in body length throughout the life of the fish (1.8-8). However, this assumption is not true for most fish, and for many species, the scales tend to grow proportionally faster than the fish itself. One effect of this decreasing ratio between body length and scale length is that the back-calculated body length for any preceding period decreases with increasing fish size, an occurrence known as Lee's phenomenon.

Another effect of this decreasing ratio is that a positive y-intercept or correction factor (C) results from a regression of body length on scale length. A constant ratio would result in the regression line's passing through the 0,0 intercept. However, the use of a y-intercept value permits correction for this changing body-scale length ratio so that the simple proportion formula, i.e., $L' = (S'/S)L$, is applicable for back calculation of growth. This correction factor is not morphologically significant and does not represent the length of the fish when scales are first formed (1.8-9, 1.8-10).

The relationship between fish growth and scale growth prior to first annulus formation is irrelevant to the problem of back calculating to the first annulus, and therefore only yearling and older fish should be used to calculate correction factors. The regression of body length on scale length should then be checked for linearity because, as Schuck (1.8-9) states, if the line of best fit is actually curvilinear or composed of a series of two or more straight lines along successive sections of the total range of body-scale length values, then the proportional method will not yield accurate results unless multiple y-intercept values are used to correct for nonlinearity.

The decision on whether the body-scale length relationship is linear often requires the investigator's judgment. This is especially true for large sample sizes where the degrees of freedom are so great that a test for deviation from linearity through analysis of variance (ANOVA) almost always results in a significant F-value denoting nonlinearity, i.e., the power of the test increases to a point where it is practically impossible to satisfy the conditions for linearity. In the event of a statistically significant deviation from linearity, the easiest way to evaluate the biological significance (1.8-11) of this deviation is to examine plots of the grouped and ungrouped body-scale length data to determine whether any obvious trend of nonlinearity exists.

Data from scale analyses on Hudson River white perch collected from 1971 through 1977 provide a detailed example of this procedure and its ramifications. A plot of the ungrouped data for 4,442 white perch is shown in Figure 1.8-1, and a more concise plot of the same data grouped as means for 10-mm body size intervals is shown in Figure 1.8-2. Neither plot demonstrates an obvious trend of nonlinearity, although a test for linearity (Table 1.8-2) shows that the body-scale length relationship is significantly nonlinear at $\alpha = .01$, i.e., the computed F-value of 1.364 is greater than the critical tabular F-value of 1.00 for >120 degrees of freedom (1.8-12).

If the relationship is treated as linear, the following body-scale length regression equation is obtained:

$$BL = 32.61 + 46.95 \text{ TSL} \quad (n = 4442) \quad (1.8-2)$$

where BL is body length and TSL is total scale length in millimeters, and the correction factor is 32.61. The back-calculation data matrix with this single

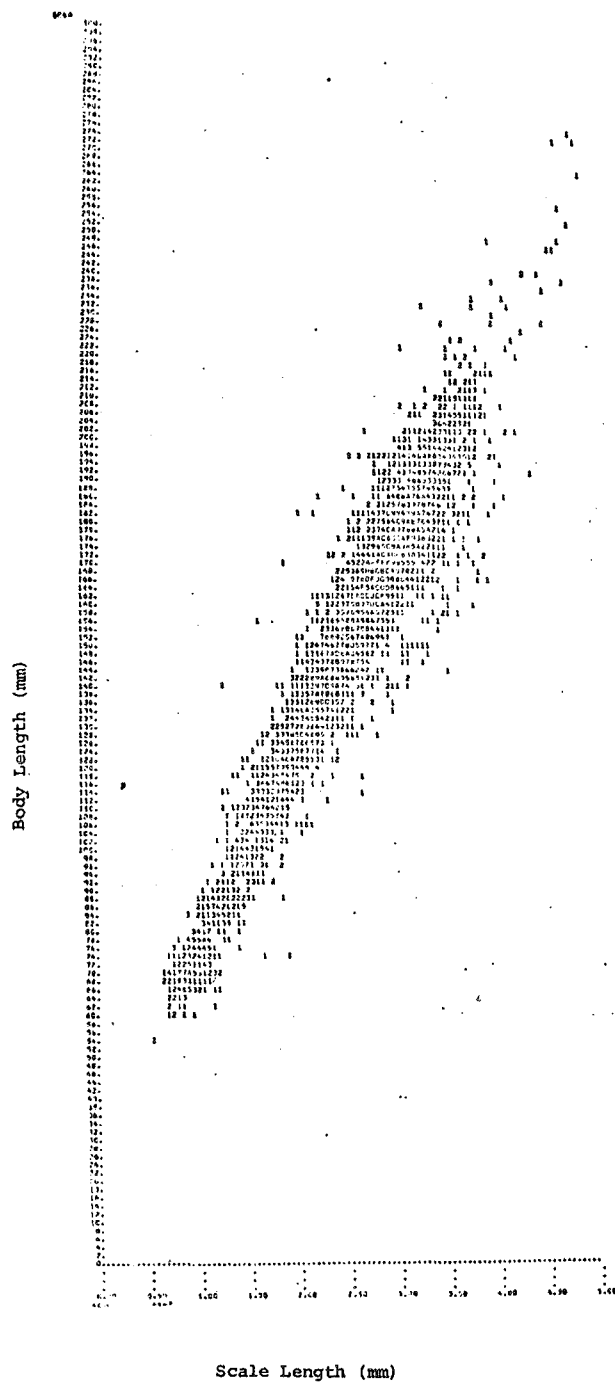


Figure 1.8-1. Relationship between length and scale length for 4,442 Hudson River white perch.

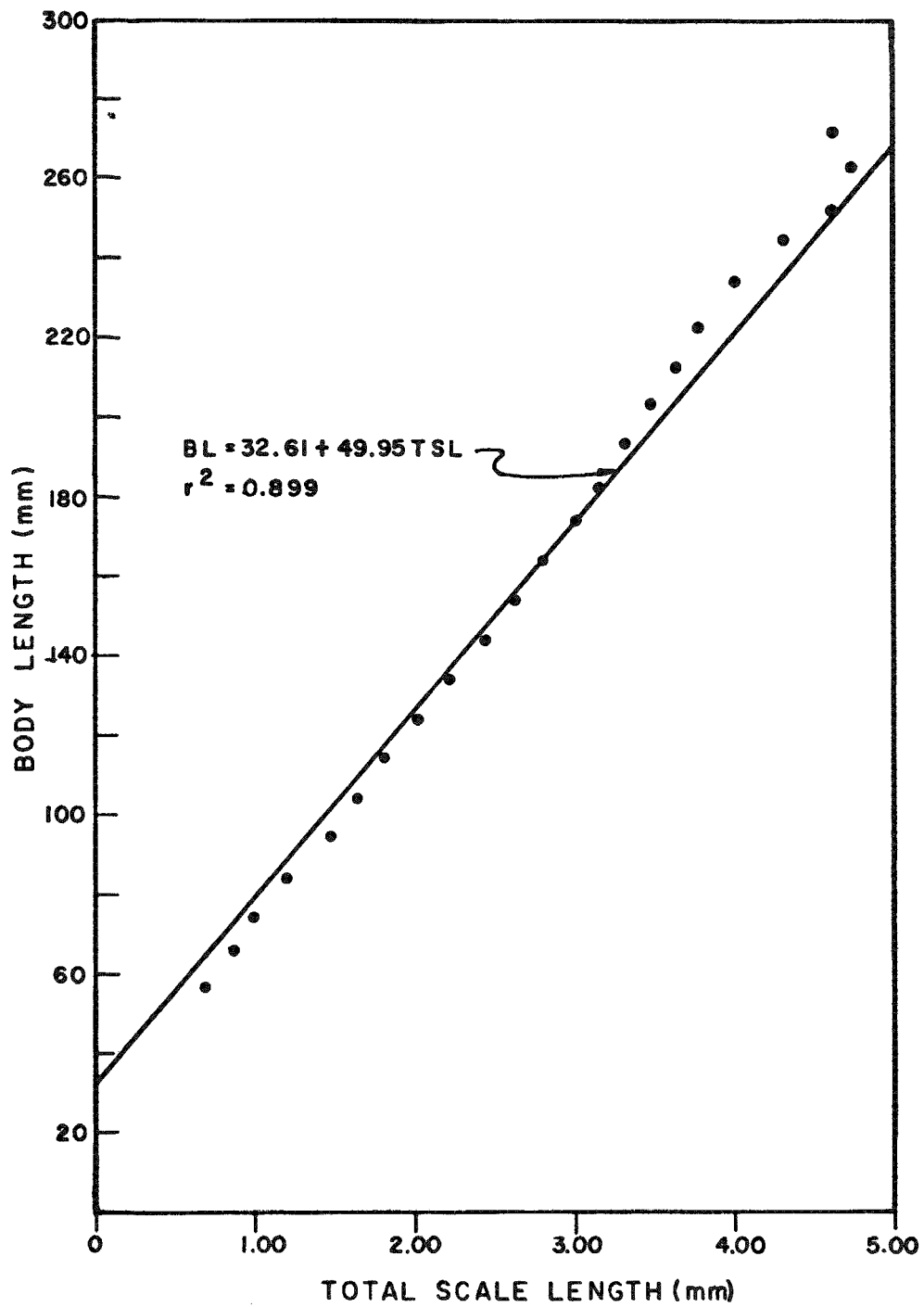


Figure 1.8-2. Relationship between body length and scale length for Hudson River white perch grouped in 10-mm intervals.

Table 1.8-2

TEST FOR LINEARITY IN THE RELATIONSHIP BETWEEN
 BODY LENGTH AND SCALE LENGTH FOR 4,442
 HUDSON RIVER WHITE PERCH

Source	d.f.	SS	MS	F
Scale Lengths	1711	5.0421×10^6		
Linear Reg.	1	4.7929×10^6	4.7929×10^6	4.49×10^4 **
Dev. from Lin.	1710	0.2492×10^6	1.4573×10^2	1.364**
Error	2730	0.2917×10^6	1.0685×10^2	
Total	4441	5.3338×10^6		

** - Significant at $\alpha = .01$

BL = $32.61 + 46.95 \text{ TSL}$

($R^2 = 0.899$)

correction factor employed is shown in Table 1.8-3 for male white perch and in Table 1.8-4 for females.

If we follow Schuck's (1.8-9) advice and assume that the relationship is best expressed by a series of straight lines, i.e., one for each age group, we obtain the following equations:

<u>Age Group</u>	<u>Regression Equation</u>	<u>Sample Size</u>
1	BL = 33.37 + 42.95 TSL	(n = 740)
2	BL = 56.85 + 35.19 TSL	(n = 1004)
3	BL = 77.85 + 30.19 TSL	(n = 1124)
4	BL = 96.62 + 26.31 TSL	(n = 900)
>5	BL = 78.81 + 34.22 TSL	(n = 674)

The back calculation data with multiple correction factors employed are found in Tables 1.8-5 and 1.8-6. A comparison plot of growth with the two methods shows that multiple correction factors result in greater growth rates (Figure 1.8-3).

In this particular case, the decision was to treat the body-scale length relationship as a linear one and to use the single correction factor of 32.6 mm for the back calculation of growth. This decision was accomplished by examination of the data plots in Figures 1.8-1 and 1.8-2 and by comparison of actual mean lengths at capture with predicted lengths based on back calculation analysis. If the interpreted annuli serve as true indicators of age and growth, and if the collection of analyzed fish was distributed fairly evenly throughout the year, the mean length for each age group at capture should be greater than the mean back-calculated length at the last formed annulus. Examination of the data in Tables 1.8-3 through 1.8-6 shows that the single-intercept method satisfies this criterion much better than the multiple-intercept method.

We can now compare back-calculated lengths at the first annulus to the actual mean length of young-of-the-year fish measured at the end of the growing season. With unbiased collection methods and scale analysis techniques assumed, the mean length of fish at the first annulus and the mean length at the end of the growing season should be identical except for the presence of size-selective mortality. Ricker (1.8-13, 1.8-14) points out that when there is size-selective mortality within a year class, the true (individual) growth rate of the fish differs from the apparent or population growth rate, and it is necessary to distinguish between them.

Table 1.8-3

STANDARD DEVIATION AND STANDARD ERROR FOR THE AVERAGE CALCULATED GROWTH
OF WHITE PERCH MALES WITH SINGLE INTERCEPTS

*****			*****														*****	
* * * * *	* * * * *	* * * * *	* * * * *	CALCULATED TOTAL LENGTH AT END OF YEAR														* * * * *
* * * * *			* * * * *			CORRECTION FACTORS											* * * * *	
* * * * *			* * * * *			WHITE PERCH MALES CAUGHT FROM 1971 TO 1977											* * * * *	
*****			*****														*****	
* * * * *	* * * * *	* * * * *	* * * * *	1	2	3	4	5	6	7	8	9	10	11	12	13	14	
*****			*****														*****	
76	53	97.3 (I)	66.9															
75	127	133.1 (II)	76.7	117.5														
74	167	157.9 (III)	74.2	126.6	153.2													
73	293	173.5 (IV)	77.2	126.4	154.8	171.7												
72	218	184.2 (V)	75.7	128.2	157.5	172.1	183.8											
71	434	194.2 (VI)	78.2	124.3	155.2	173.6	185.3	194.8										
70	400	206.5 (VII)	79.7	133.4	156.9	171.0	183.6	193.9	206.7									
69	740	209.4 (VIII)	78.6	131.7	158.7	172.1	180.3	192.2	196.3	221.0								
68	73	0.0 (IX)	78.3	128.7	155.8	170.9	182.1	188.2	205.6	194.5	0.0							
67	34	0.0 (X)	80.1	132.6	157.6	171.9	186.4	198.6	222.0	0.0	0.0	0.0						
66	13	0.0 (XI)	80.6	132.3	156.6	170.4	182.6	196.7	212.4	207.7	0.0	0.0	0.0	0.0				
65	2	0.0 (XII)	79.4	138.2	157.8	168.9	182.7	194.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0			
GRAND AVERAGE			77.5	128.3	156.4	171.9	183.4	193.9	204.0	209.4	0.0	0.0	0.0	0.0				
SAMPLE SIZE			2063	1778	1281	777	325	117	22	8	0	0	0	0				
STANDARD DEVIATION			9.0	12.6	11.3	10.8	11.2	12.6	20.2	19.2	-0.0	-0.0	-0.0	-0.0				
AVERAGE ANNUAL INCREMENT			77.5	50.8	28.1	15.5	11.5	10.5	10.1	5.4	0.0	0.0	0.0	0.0				
CONFIDENCE INTERVAL OF THE MEAN			77.1-127.7-155.8-171.2-182.2-191.6-195.4-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-				
			77.9 128.9 157.0 172.7 184.7 196.3 212.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0				

Table 1.8-4

STANDARD DEVIATION AND STANDARD ERROR FOR THE AVERAGE CALCULATED GROWTH
OF WHITE PERCH FEMALES WITH SINGLE INTERCEPTS

* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *
* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *
* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *
* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *	* * *
76	63	99.2 (I)	70.1											
75	140	135.2 (II)	79.7	118.7										
74	175	159.3 (III)	76.7	128.8	154.6									
73	282	177.5 (IV)	80.2	128.7	157.9	174.6								
72	225	189.7 (V)	77.6	131.1	160.9	176.7	187.8							
71	514	201.4 (VI)	80.2	125.8	158.7	177.8	188.9	200.2						
70	462	211.3 (VII)	80.8	134.9	157.9	175.5	191.1	199.6	224.5					
69	259	216.4 (VIII)	81.9	135.6	163.2	177.4	189.4	204.5	208.9	212.6				
68	96	250.5 (IX)	81.3	132.0	161.2	177.4	191.9	205.9	217.4	224.9	273.0			
67	33	0.0 (X)	80.4	133.6	159.9	176.7	189.8	200.5	214.0	0.0	0.0	0.0		
66	11	0.0 (XI)	85.2	131.5	158.8	173.5	184.5	196.9	206.3	223.0	0.0	0.0	0.0	
65	8	0.0 (XII)	80.2	123.4	156.6	172.5	188.5	201.4	214.6	224.8	243.0	0.0	0.0	0.0
64	2	0.0 (XIII)	79.1	112.4	141.2	153.9	167.7	182.0	199.3	0.0	0.0	0.0	0.0	0.0
GRAND AVERAGE			79.7	130.5	159.3	176.6	189.3	201.2	212.3	221.9	250.5	0.0	0.0	0.0
SAMPLE SIZE			227C	1906	1410	794	349	112	36	15	4	0	0	0
STANDARD DEVIATION			8.7	12.9	12.4	12.7	13.9	18.8	20.9	23.9	37.3	-0.0	-0.0	-0.0
AVERAGE ANNUAL INCREMENT			79.7	50.7	28.9	17.2	12.7	11.9	11.1	9.6	28.6	0.0	0.0	0.0
CONFIDENCE INTERVAL OF THE MEAN			79.4-129.9-158.7-175.7-187.8-197.6-205.3-209.6-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-
			80.1	131.0	160.0	177.5	190.8	204.7	219.2	234.3	0.0	0.0	0.0	0.0

STANDARD DEVIATION AND STANDARD ERROR FOR THE AVERAGE CALCULATED GROWTH
OF WHITE PERCH MALES WITH MULTIPLE INTERCEPTS

76	53	97.3 (I)		67.1															
75	127	133.1 (II)		77.0	120.2														
74	167	157.9 (III)		74.6	130.7	155.0													
73	293	173.5 (IV)		77.6	131.2	158.0	173.0												
72	218	184.2 (V)		76.1	134.5	162.7	175.2	184.3											
71	434	194.2 (VI)		78.7	130.0	160.1	177.4	188.7	195.0										
70	409	206.5 (VII)		80.1	138.9	161.0	174.1	185.2	194.0	206.7									
69	240	209.4 (VIII)		79.1	138.4	163.3	174.3	181.7	189.5	197.4	221.0								
68	73	0.0 (IX)		78.8	136.8	162.5	174.6	183.8	189.6	206.9	194.5	0.0							
67	34	0.0 (X)		80.6	141.0	165.9	177.7	188.1	198.7	222.0	0.0	0.0	C.0						
66	13	0.0 (XI)		81.1	141.7	167.4	180.3	186.4	198.8	213.6	207.7	0.0	0.0	0.0	0.0				
65	2	0.0 (XII)		79.9	146.7	168.1	179.C	186.0	194.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GRAND AVERAGE				77.9	134.0	160.9	175.1	184.9	194.6	204.9	209.4	0.0	0.0	0.0	0.0				
SAMPLE SIZE				2063	1778	1281	777	325	117	22	8	0	0	0	0				
STANDARD DEVIATION				9.0	12.7	11.8	11.3	11.2	12.7	20.2	19.2	-0.0	-0.0	-0.0	-0.0				
AVERAGE ANNUAL INCREMENT				77.9	56.1	26.9	14.2	9.7	9.8	10.3	4.5	0.0	0.0	0.0	0.0				
CONFIDENCE INTERVAL OF THE MEAN				77.5-133.4-	160.2-	174.3-	183.6-	192.3-	196.3-	0.0-	0.0-	0.0-	0.0-	0.0-	0.0-				
				78.3	134.6	161.6	176.C	186.1	197.0	213.5	0.0	0.0	0.0	0.0	0.0				

STANDARD DEVIATION AND STANDARD ERROR FOR THE AVERAGE CALCULATED GROWTH
OF WHITE PERCH FEMALES WITH MULTIPLE INTERCEPTS

1-234

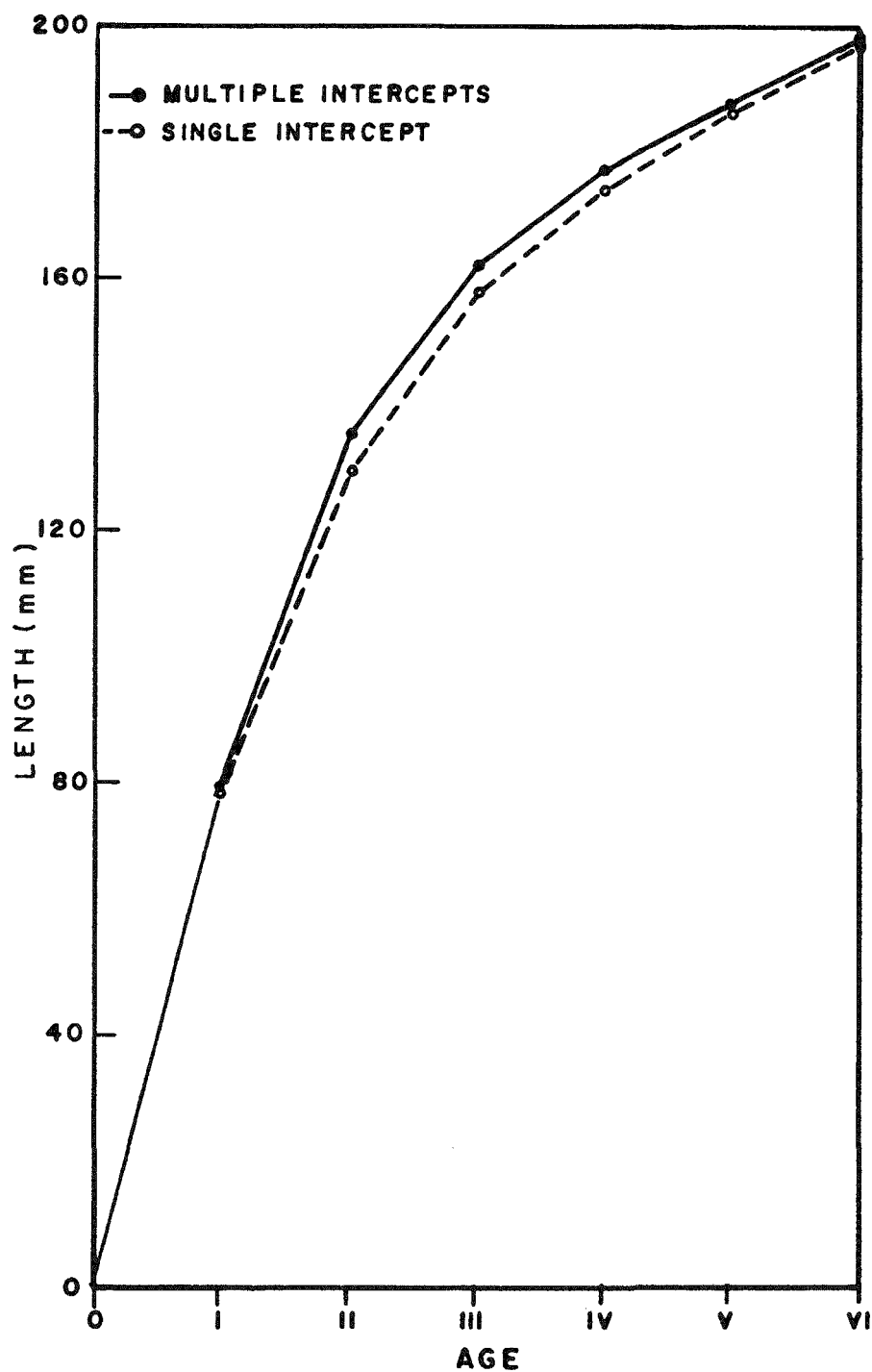


Figure 1.8-3. Calculated growth for Hudson River white perch with various correction factors employed.

The population growth rate is obtained simply by the comparison of the mean size of surviving fish at successive ages. But if the smaller fish of a year class die more frequently than the larger, for example, the mean size of the survivors is greater than it would be in the absence of size-selective mortality.

Scale analysis and the resultant back-calculated lengths to the first annulus are based on the survivors of the young-of-the-year cohorts which were measured at the end of their first growing season. Thus, the back-calculated lengths for male and female white perch plotted in Figure 1.8-4 are based on fish from one to eight or nine years old found in Table 1.8-5 and 1.8-6, whereas the actual mean lengths plotted in Figure 1.8-4 represent young (age 0) white perch alive in November of each year. The difference between the November means and the back-calculated means serves as an index of size-selective mortality and suggests that mortality is selective for the smaller fish. That is, the mean length of young fish alive in November of each year is less than the mean length of survivors in later years as represented by back-calculation to the first annulus.

The data in Figure 1.8-4 further indicate that some aspect of size-selective mortality changed from the pre-operative period of 1971-1972 to the post-operative period of 1973-1976, when the largest Hudson River power plants became operational. Size-selective mortality occurs to some extent during all years from 1971 through 1976, but it appears that the time of occurrence for such mortality may have changed. During pre-operational years, the smallest fish of the 1971 and 1972 cohorts were still present during November, and the bulk of the size-selective mortality occurred sometime subsequently in the life of these cohorts, perhaps during the first winter. During post-operational years, the bulk of the selective mortality occurred prior to November, and therefore the difference between actual mean November length and back-calculated length is substantially less during the post-operational period of 1973-1976. If this hypothesis is true, then the change in the time of occurrence for size-selective mortality may be caused by the greatly increased magnitude of entrainment during June through October of the post-operational years as compared to the pre-operational period of 1971-1972. In other words, because entrainment is probably selective for the smallest young-of-the-year fish, increased summer and early fall entrainment during the post-operational years would tend to eliminate the smallest fish from a cohort before November when they are measured. Were it not for entrainment, these smaller fish would be eliminated after November, as indicated by the 1971 and 1972

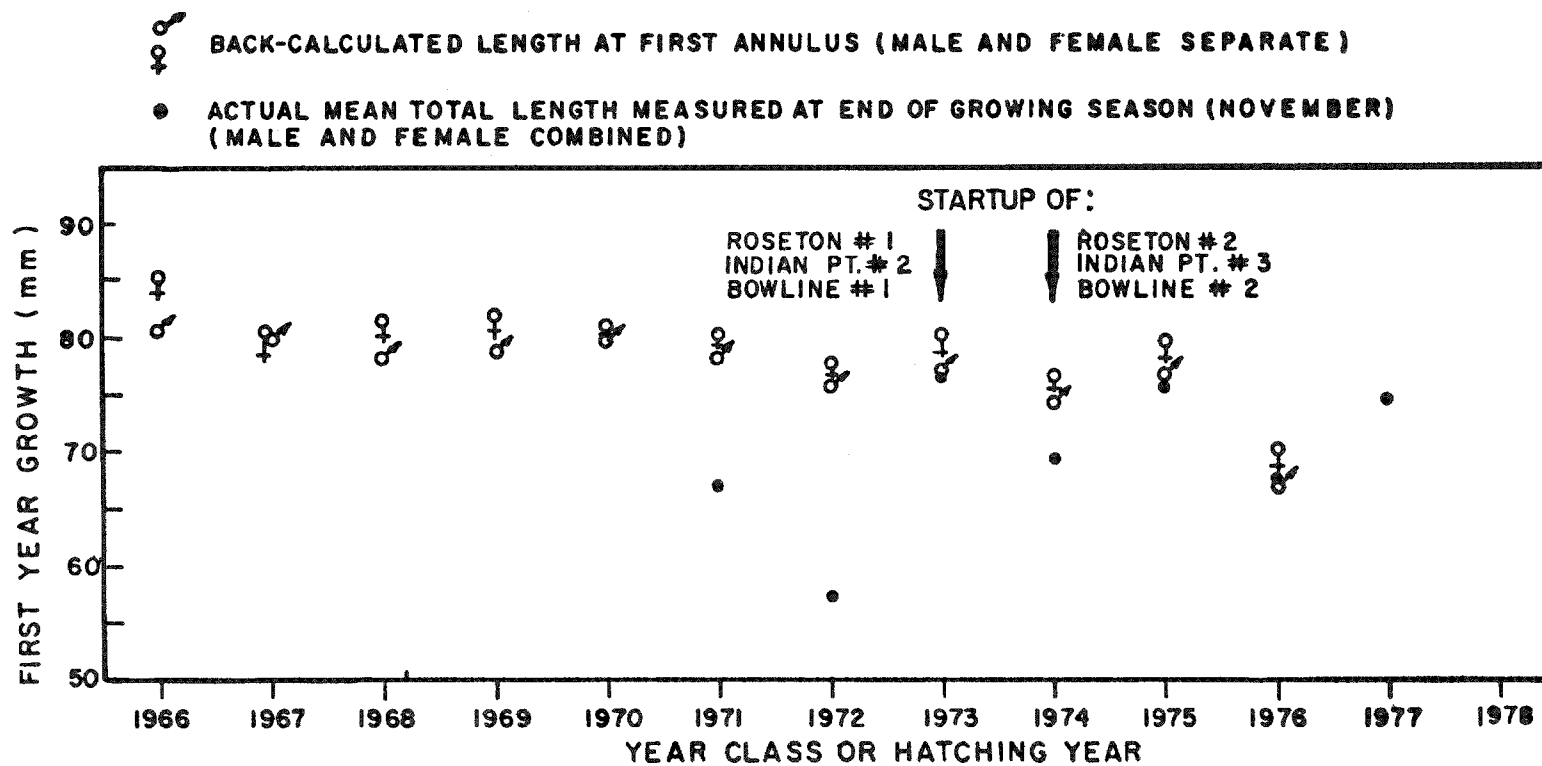


Figure 1.8-4. First-Year Growth for Hudson River White Perch Collected During 1971-1977.

data. Given this hypothesis, a positive correlation between the ratio of mean November length and calculated length at first annulus vs summer/fall (June-October) entrainment flows should be observed. The data in Table 1.8-7 tend to confirm this ($r = 0.804$, significant at $\alpha = 0.10$), and it is possible that a concept of plant (entrainment) impact may be based on differences in size-selective mortality observed between pre- and post-operational periods.

An alternate and equally tenable hypothesis is that the greater the length attained by young fish by the end of November, the better they survive throughout the life of the cohort. That is, size-selective mortality after November becomes less effective with increasing November length, and therefore the ratio of mean November length and calculated length at first annulus approaches unity as November length increases. The data in Table 1.8-7 and the resulting correlation coefficient of $r = 0.847$ (significant at $\alpha = 0.05$) support this hypothesis. Thus, we have the intuitively attractive proposition that size-selective sources of mortality are directed toward smaller white perch in the Hudson River, but the greater the growth during the first year of life, the less effective is this size-selective mortality after November. These two hypotheses are not entirely independent or mutually exclusive, and additional years of data are needed to properly evaluate them.

Further evidence of size-selective mortality is provided by the back calculation data in Tables 1.8-3 and 1.8-4. Ricker (1.8-13, 1.8-14) shows that when population growth rates (G_x) differ from the true growth rates (G), size-selective mortality is indicated. Specifically, when the true growth rates are less than the population growth rates, the selective mortality is directed against the smaller fish.

The population growth rate is obtained by the comparison of the mean size of surviving fish at successive ages; it is calculated in Table 1.8-8 as the difference in natural logs of the lengths from the uppermost diagonal in Tables 1.8-3 and 1.8-4. The true growth rate is calculated as the difference in natural logs of the lengths from the last pair of entries of each line in Table 1.8-3 and 1.8-4. Ricker (1.8-13) states that this is the most recent and most direct information on the rate of growth of the surviving fish of a year class in successive years. Table 1.8-8 shows that population growth rates are all greater than true growth rates, reflecting selective mortality against the smaller fish.

Table 1.8-7

CORRELATIONS FOR THE ACTUAL/CALCULATED MEAN LENGTH RATIO WITH
NOVEMBER MEAN LENGTH AND PLANT ENTRAINMENT FLOWS

Year	(1) Ratio of November Mean Length to First Annulus Length	(2) November Mean Length (mm)	(3) June-October Plant Flows (cfs)
1971	0.84	66.9	0
1972	0.74	57.1	23
1973	0.97	76.5	1704
1974	0.91	69.3	3391
1975	0.96	75.7	3734
1976	0.98	67.2	3956

$r_{1,3} = 0.804$ Significant at $\alpha = 0.10$
 $r_{1,2} = 0.847$ Significant at $\alpha = 0.05$

Table 1.8-8

COMPUTATION OF GROWTH RATES FOR HUDSON RIVER WHITE PERCH
COLLECTED DURING 1971-1977

I. MALE

Age Interval	Population Growth (G_x)		Mean Individual Growth (G)	
	Length ^a Increment (mm)	Instantaneous ^b Growth Rate	Length ^c Increment (mm)	Instantaneous ^b Growth Rate
1-2	66.9 - 117.5	0.563	76.7 - 117.5	0.426
2-3	117.5 - 153.2	0.265	126.6 - 153.2	0.191
3-4	153.2 - 171.7	0.114	154.8 - 171.7	0.104
4-5	171.7 - 183.8	0.068	172.1 - 183.8	0.066
5-6	183.8 - 194.8	0.058	185.3 - 194.8	0.050

II. FEMALE

1-2	70.1 - 118.7	0.527	79.7 - 118.7	0.398
2-3	118.7 - 154.6	0.264	128.8 - 154.6	0.183
3-4	154.6 - 174.6	0.122	157.9 - 174.6	0.101
4-5	174.6 - 187.8	0.073	176.7 - 187.8	0.061
5-6	187.8 - 200.2	0.064	188.9 - 200.2	0.058

^aThe length increments are from the uppermost diagonal in Tables 1.8-3 and 1.8-4.

^bInstantaneous growth in length = $\log_e L_2 - \log_e L_1$.

^cThe length increments are from the last pair of entries of each line in Tables 1.8-3 and 1.8-4.

The hypothesis of selective mortality is further supported by the appearance of a negative Lee's phenomenon when the back-calculated lengths at the first annulus are plotted (Figure 1.8-4). Such a trend results when the back-calculated growth in earlier years tends to increase with the age of the fish being calculated, and Ricker (1.8-13) states categorically that a negative Lee's phenomenon can occur only when mortality decreases with length, i.e., when mortality is selective for the smaller fish.

It is important to establish whether a relationship between size and mortality exists, since such a relationship may be integral to the analysis of fluctuations in year-class strength. For example, LMS has recently shown that the average water temperature during July and August is an important factor in determining the mean length attained by young white perch at the end of their first growing season. Thus, if power plant mortality is size-selective, then the potential exists for higher impact to occur during those years when summer temperatures are least favorable for growth.

1.8.3 Mortality

Whenever an investigation is conducted on the dynamics of a fish population (or, in fact, of any biological population), it is imperative to have a working estimate of the prevailing mortality rates (or complementary survival rates) within that population. This is especially important in the investigation of the dynamics of death and survival with respect to power plant impact.

Changes in mortality rate with age may provide insight into the relative magnitude of sources of death. For example, a high rate of mortality for fish between age 0 and 1 compared to older fish may indicate the presence of a size-selective cropping mechanism. On the other hand, a sharp increase for the oldest fish in a mortality curve would suggest that the fish were living long enough to approach the end of their life span and then dying from old age.

With regard to the attainment of mortality data, Beverton and Holt (1.8-15) state:

"In general, the estimation of mortality parameters makes a greater demand on the sampled data, with regard to both their extent and reliability, than does the analysis of any other factor; and it is probably true to say that if an adequate sampling system can be devised for this purpose it will almost certainly provide satisfactory measures of other parameters."

It is convenient for our purposes to divide total mortality into two components: 1) mortality due to plant operation, P , and 2) natural mortality, M , resulting from all other causes including fishing. Various aspects of these two sources of mortality are described in Section 5.2.2 of the methodology.

1.8.3.1 Catch Curves

Perhaps the most widely used method for estimating mortality is catch curve analysis, where the natural logarithms of the number of fish caught at each age are plotted against age, and the total mortality coefficient, A , is estimated from the slope of the descending straight line. In a catch curve, the fish comprising the various age groups are from different year classes, so that to use it to estimate mortality, the effects of fluctuations in year-class strength have to be minimized. Ricker (1.8-14) and Beverton and Holt (1.8-15) state that when mortality rates are stable and the population is in equilibrium, catch curves from several years can be combined so that fluctuations in year-class strength are averaged out.

An alternative but more exacting method is cohort analysis, where the fish comprising the various age groups are from the same year class collected in succeeding years. This method requires that a year class or cohort be followed through time from birth to death. The first method is analogous to a vertical life table and the second, a horizontal life table. Krebs (1.8-16) points out that the mortality trends from the two types of analysis ".... will be identical if and only if the environment does not change from year to year and the population is at equilibrium."

A basic assumption in catch curve analysis is that all age groups for which mortality is computed are equally vulnerable to the sampling gear, i.e., that each age group is captured in proportion to its true abundance. This is probably not true for certain gear such as gill nets of a single mesh size and small seines which tend to capture only the younger fish. For many species, large trawls with a fine-mesh cod end liner appear to be a very suitable gear for capturing both young and old fish in a representative fashion.

It is evident that all fish to be used in a catch curve must be assigned an age. Most survey programs, however, do not provide a random, representative sample of

aged fish, but do often provide a random sample of fish measured for length. Thus an age-length key can be constructed to assign ages based on length. In using an age-length key, Ricker (1.8-14) cautions that "... the fish used for age determination must be taken from the same stock, during the same season, and using gear having the same selective properties as that used to take the length-frequency samples. Above all, an age-length key cannot be applied to length samples of any year except the one from which it was derived...." Generally it is more profitable to place all available resources into increasing the size of the age sample rather than implementing a massive length sampling program.

If age group 0 fish are to be included in the catch curve, then data should be utilized only from that period when young-of-the-year fish are present and fully recruited to the sampling gear. Otherwise the abundance of older fish will be overestimated in relation to the 0-age fish. An example of an age-length key for Hudson River white perch collected during August-December 1972 is shown in Table 1.8-9.

Because catches from each station are often subsampled for length measurements rather than measuring all fish in a catch, some adjustment must be made so that the age-length key reflects the actual number caught, rather than simply the number measured. The following example should suffice to illustrate where adjustments must be made. Consider that 1,000 fish were caught at shallow Station A, 90% of which were young-of-the-year fish and 10% were adult fish. One hundred of these fish were selected randomly for measurement, resulting in an age composition of 90 young and 10 adults. At Station B, a deeper station, 100 fish were caught 90% of which were adult and 10% young fish. In this case, the sample is small enough so that the entire 100 were measured. If the two samples of measured fish were simply combined without adjustment to reflect the total fish caught, it would appear that young fish were as abundant as adult fish, (100 of each), and the resulting mortality estimate would be zero. The adjusted number of fish in each age group collected at a given station on a given date (N_x) can be calculated by

$$N_x = \sum \left(N_L \cdot P_x \cdot \frac{C/F}{M} \right) \quad (1.8-3)$$

Table 1.8-9
AGE-LENGTH KEY FOR WHITE PERCH COLLECTED
DURING 1972

Length Interval (cm)	Size Class (cm) Represented by Length Interval	Fish Actually Aged	Allocation into Age Groups Using Scale Analysis Data (Jul-Dec Only)							Bowline Vicinity Adjusted Trawl Catch (Aug-Dec Only)	Calculated Age Representation For Measured Fish						
			0	1	2	3	4	5	6		0	1	2	3	4	5	6
0.5- 1.4	1																
1.5- 2.4	2								1.4	1.4							
2.5- 3.4	3								19.4	19.4							
3.5- 4.4	4								59.8	59.8							
4.5- 5.4	5								415.7	415.7							
5.5- 6.4	6								422.8	422.8							
6.5- 7.4	7								159.0	159.0							
7.5- 8.4	8								19.9	19.9							
8.5- 9.4	9	18	10						19.2		19.2						
9.5-10.4	10	42	40	2					143.5		136.7	6.8					
10.5-11.4	11	36	36						475.9		475.9						
11.5-12.4	12	21	18	3					184.0		157.7	26.3					
12.5-13.4	13	26	3	23					38.9		4.5	34.4					
13.5-14.4	14	46	2	41	2	1			35.3		1.5	31.5	1.5	0.8			
14.5-15.4	15	44		35	9				55.5			44.1	11.4				
15.5-16.4	16	42		22	15	5			85.6			44.8	30.6	10.2			
16.5-17.4	17	57		13	38	6			88.1			20.1	58.7	9.3			
17.5-18.4	18	28		3	15	9	1		59.8		6.4	32.0	19.2		2.1		
18.5-19.4	19	8			2	4	2		32.2				8.1	16.1	8.1		
19.5-20.4	20	4			2		1	1	11.3				5.6		2.8	2.8	
20.5-21.4	21	1				1			2.2					2.2			
21.5-22.4	22																
22.5-23.4	23																
23.5-24.4	24																
Total									2329.5	1098.0	795.5	214.4	147.9	57.8	13.0	2.8	
Log _e										7.0012	6.67897	5.36784	4.99654	4.05699	2.56495	1.02962	
Percent										47.1	34.1	9.2	6.3	2.5	0.6	0.1	

where

N_L = the number of fish measured that were in length interval L

P_x = the fraction of x-year-old fish in length interval L

C/f = catch per 10 minutes at the station

M = total number of fish measured from the station

The adjusted numbers representing the actual relationship between abundances of young and adult fish are shown in Table 1.8-10, along with the calculated mortality for this hypothetical situation.

Actual catch curve data for nine cohorts of white perch collected from the Hudson River during 1971 to 1976, excluding 1974, are presented in Figure 1.8-5 and Table 1.8-11. The year 1974 was excluded because catches from all gears and all stations were combined prior to measurement for length, making it impossible to extract trawl data for the catch curve analysis. This is a good example of sample treatment which saves money but loses information.

Catch curves drawn from commercial catches typically have the appearance of the first curve in Figure 1.8-5, where the ascending left limb and the dome represent age groups which are incompletely captured by the gear used to take the sample. It is impossible to find out anything definite about the actual mortality rate during the years covered by the left limb and dome of the curve because sampling of the population is not random (1.8-14). However, if experimental gear is employed, such as the 30-foot lined otter trawl used in this study, the representative part of the curve is pushed back to the earliest age groups, providing information on early mortality rates and fluctuations in year-class strength.

Examination of each annual catch curve in Figure 1.8-5 provides insight into changes in year-class strength because the irregularities in the curves are due to fluctuations in recruitment from year class to year class. The 1971 catch curve, for example, indicates that the strength of the 1970 year class was poor compared to that of the 1969 and 1971 year classes. The overall trend presented in Figure 1.8-5 indicates that white perch year class strength since 1967 has fluctuated on an alternate year basis, and no unidirectional trend of increasing or decreasing recruitment is discernible.

Table 1.8-10

METHOD FOR ADJUSTING NUMBER OF FISH MEASURED TO REFLECT ACTUAL
NUMBER CAUGHT

STATION B					
M = 100		C/f = 100 fish/10 min		$\frac{C/f}{M} = 1.0$	
Length Interval	P _{young} *	P _{adult} *	N _L	N _{young}	N _{adult}
-	1.00		4	4	
-	1.00		6	6	
-		1.00	40		40
-		1.00	30		30
-		1.00	20		20
Adjusted Total (N _x) ₁				10	90

STATION A					
M = 100		c/f = 1000 fish/10 min		$\frac{C/f}{M} = 10.0$	
Length Interval	P _{young} *	P _{adult} *	N _L	N _{young}	N _{adult}
-	1.00		40	400	
-	1.00		50	500	
-		1.00	4		40
-		1.00	3		30
-		1.00	3		30
Adjusted Total (N _x) ₁				900	100

Combined total for both stations: 910 young; 190 adults.

Percent of total for combined stations: 82.7% young; 17.3% adults.

Mortality between young and adult = $\frac{910-190}{910} = 79.1\%$.

*From age-length key.

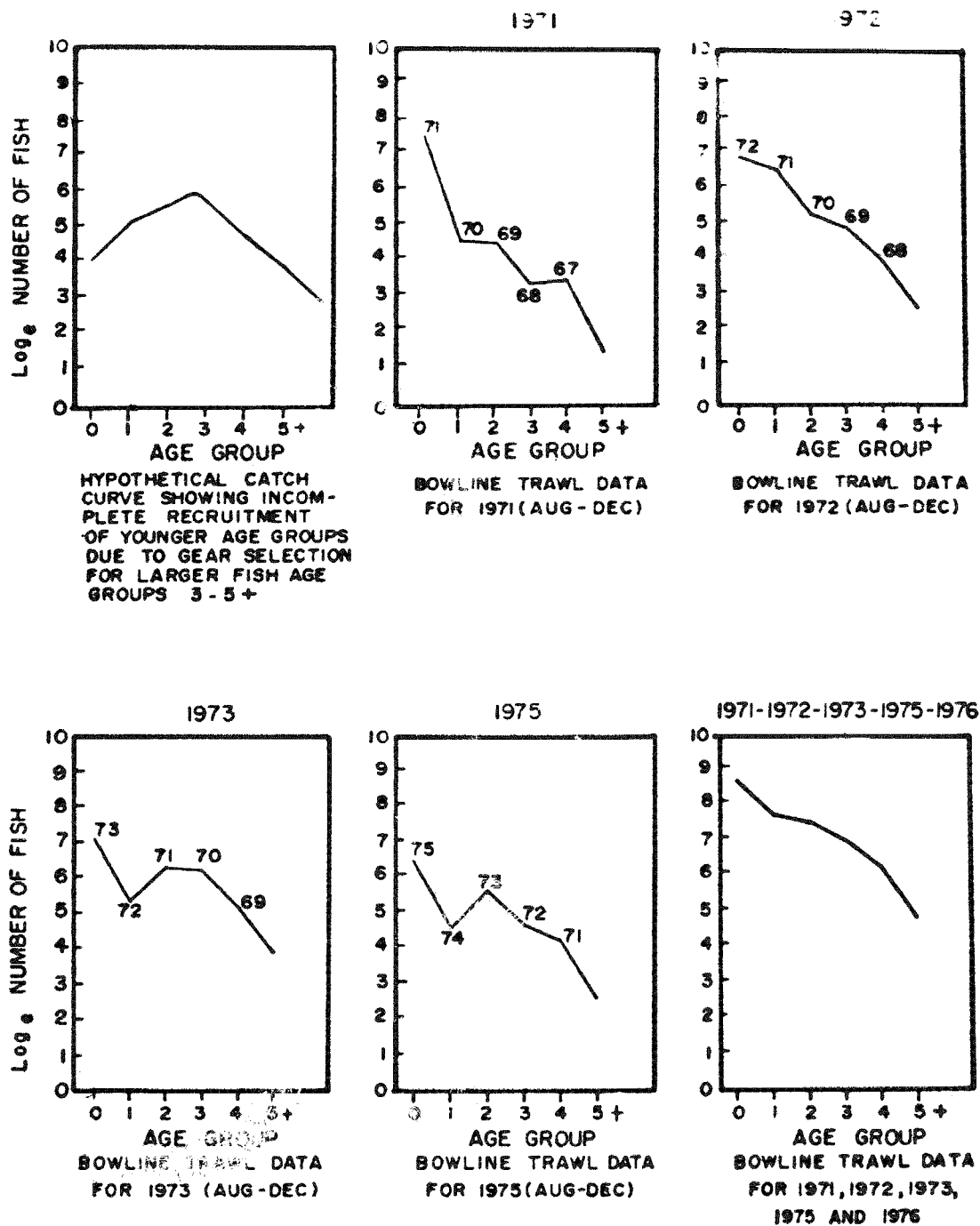


Figure 1.8-5. Catch-curve data for white perch collected by bottom trawls in the Bowline vicinity, 1971-1973, 1975-1976.

Table 1.8-11
 CATCH-CURVE DATA AND CALCULATED TOTAL ANNUAL MORTALITY RATES
 FOR HUDSON RIVER WHITE PERCH COLLECTED BY BOTTOM TRAWL
 1971-1973, 1975-1976

Cohorts Represented	Age Group	Number of Fish	Percent of Total	Proportional Mortality Between Age Groups (A)	Total Instantaneous Mortality Rate (Z)*
1976 + 75 + 73 + 72 + 71	0	5516	50.9	0.6222	0.9734
1975 + 74 + 72 + 71 + 70	1	2084	19.2	0.2111	0.2371
1974 + 73 + 71 + 70 + 69	2	1644	15.2	0.3625	0.4502
1973 + 72 + 70 + 69 + 68	3	1048	9.7	0.5849	0.8792
1972 + 71 + 69 + 68 + 67	4	435	4.0	0.7241	1.2877
	>5	120	1.1		
	Total	10847	Average	A = 0.5010 A = $1 - e^{-Z} = 0.5349$	Z = 0.7655

*Z = $-\log_e(1-A)$.

Changes in the total annual mortality rate with age (Table 1.8-11) show that relatively high mortality ($A = 0.62$) exists for age 0 fish between the August-December period used to assess young-of-the-year abundance and the August-December period of the next year, which is used to assess abundance of yearling fish. This suggests the existence of a mortality source for younger fish which operates sometime during December through August. The bulk of such mortality may probably occur during the winter when white perch impingement, which is definitely selective for the smallest fish, reaches its highest levels. After the first year of life, changes in mortality rates for white perch generally follow the schedule demonstrated by Ricker (1.8-17) and discussed by Beverton and Holt (1.8-1) for several little-exploited fish populations, where mortality rates increased with age in all cases.

The necessity of mortality estimates in impact assessment can best be illustrated by an example of impingement of Hudson River white perch. When the total annual mortality rate (A) has been established, the total instantaneous rate (Z) can be calculated by the relationship $Z = -\log_e (1-A)$. The next step is to calculate an exploitation rate (u), where

$$u = \frac{\text{number of fish impinged}}{\text{initial population size}}$$

The number of fish impinged is based on estimated total counts at intake screens, and the initial population size is provided by a Petersen mark-recapture estimate (Section 1.4.1) for the entire river population. Exploitation rates for the 1974 year class alone and for all age groups combined are shown in Tables 1.8-12 and 1.8-13. Once A , Z , and u are known, the remainder of the parameters necessary for the calculation of impact (m) can be obtained via the relationships shown in Table 1.8-14.

Impingement impact, expressed by m , is defined as the annual reduction of the population due to plant cropping in the absence of other sources of mortality. In other words, it is the reduction of an initial population due solely to plant cropping during a particular year, as if there were no concurrent and competing sources of mortality present.

The use of m , the conditional plant mortality rate, rather than the exploitation rate, u , as an estimate of impact is not intuitively obvious and reflects the fact

Table 1.8-12

ESTIMATED IMPINGEMENT RATES AND CALCULATED EXPLOITATION RATES FOR
THE 1974 YEAR-CLASS OF WHITE PERCH AT THREE HUDSON RIVER GENERATING STATIONS
JULY 1974 - JUNE 1975

Power Plant River Mile	Annual Impingement Estimates ^a	Hudson River Population Estimate ^b	Exploitation Rate (u)	Contribution to Multiplant Exploitation $(\frac{u_n}{u_{Total}} \times 100)$
37	425,739		$u_1 = 0.0203$	23.1%
43	1,368,285	21,000,000	$u_3 = 0.0652$	74.3%
66	46,823		$u_5 = 0.0022$	2.5%
Total	1,840,847		$u_{Total} = 0.0877$	

^aEstimated number of 1974 year-class fish impinged = 90% of total annual impingement for all age groups.

^bPeterson estimate of young white perch population present in the Hudson River during October 1974.

Table 1.8-13

ESTIMATED IMPINGEMENT RATES AND CALCULATED EXPLOITATION RATES FOR
ALL AGE GROUPS OF WHITE PERCH AT THREE HUDSON RIVER GENERATING STATIONS
JULY 1974 - JUNE 1975

Power Plant River Mile	Annual Impingement Estimates	Hudson River Population Estimate*	Exploitation Rate (u)	Contribution to Multiplant Exploitation $(\frac{u_n}{u_{Total}} \times 100)$
37	473,043		$u_1 = 0.0115$	23.2%
43	1,520,317	41,300,000	$u_3 = 0.0368$	74.2%
66	52,025		$u_5 = 0.0013$	2.6%
Total	2,045,385		$u_{Total} = 0.0496$	

*Population of all age groups estimated from catch-curve data which show that young (age group 0) fish comprise 50.9% of the entire population.

Table 1.8-14

NUMERICAL RELATIONSHIPS BETWEEN IMPINGEMENT AND NATURAL MORTALITY
FOR A RICKER TYPE 2 FISHERY

Subdivisions of Mortality	Symbol	Relationships	Multiplant Values (Per Year)	
			All Age Groups	Age 0
I. Instantaneous Mortality Rates				
1. Total (coefficient of decrease)	Z	$Z = P + M$	0.7655	0.9734
2. From impingement (rate of impingement)	P	$P = \frac{Zu}{A}$	0.0710	0.1372
3. From natural causes	M	$M = Z - P$	0.6945	0.8362
II. Annual Conditional Mortality Rates				
1. From impingement (without concurrent natural mortality)	m	$m = 1 - e^{-P}$	0.0685	0.1282
2. From natural causes (without concurrent impingement mortality)	n	$n = 1 - e^{-M}$	0.5007	0.5666
III. Annual Expectation of Death (actual mortality rates)				
1. Total	A	$A = 1 - e^{-Z} = u + v$	0.5349	0.6222
2. From impingement (rate of exploitation)	u	$u = \frac{PA}{Z}$	0.0496	0.0877
3. From natural causes (with concurrent impingement mortality)	v	$v = \frac{MA}{Z}$	0.4853	0.5345

Implied from above:

$$A = m + n - mn$$

$$\frac{Z}{A} = \frac{P}{u} = \frac{M}{v}$$

that impingement and natural mortality compete for some of the same fish. That is, impingement appears to conform more closely to a Ricker "Type 2" fishery in which fishing (power plant) mortality and natural mortality operate concurrently, than it does to a "Type 1" fishery where $m = u$, and there is no natural mortality occurring during the fishing (impingement) season. Additional discussion of the interrelationships among mortality rates may be found in Section 5.2.2 of the methodology.

The competing aspects of a Type 2 fishery may be seen in Table 1.8-15, where theoretical natural mortality in the absence of impingement, n , is greater than actual natural mortality when impingement is present, v , pointing to the fact that impingement and natural death are competing sources of mortality. In the young-of-the-year situation 9.1×10^6 fish survive when there is no impingement, and 7.9×10^6 survive when impingement is added to the system. Multi-plant impact, then, is the percent reduction of those fish which would have survived if only natural mortality were operative within the population, i.e., $m_0 = 12.8\%$. In this case $m_{0,5+} = 6.8\%$ for the mixed cohort approach. These are maximized or conservative estimates because of the assumption that there is no survival associated with the impingement process and there is no compensatory response via decreased natural mortality rate or increased reproductive rate.

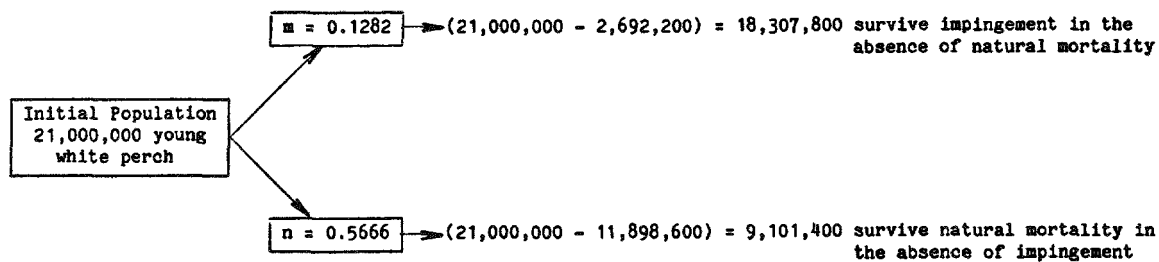
1.8.3.2 Successive Density Method. Mortality estimates can be made with information from successive sampling throughout the range of a non-uniformly distributed population, or from successive sampling within a localized segment of a spatially homogeneous population, assuming no immigration, emigration, or recruitment during the sampling period. This approach is especially useful for larval and early juvenile populations subject to high natural and/or plant-induced mortality and which are not amenable to alternative methodology using standard mark-recapture techniques. The successive density method may be viewed as a type of catch curve procedure, since it permits comparison of the number of animals alive at successive ages, where age is measured in terms of weeks or months, rather than years.

Very few published field studies of larval/early juvenile mortality are available, and Cushing (1.8-18) states that the best measure of larval mortality has been made by Percy (1.8-19), who utilized the successive density method in a study of the winter flounder (Pseudopleuronectes americanus) in Connecticut's Mystic River estuary.

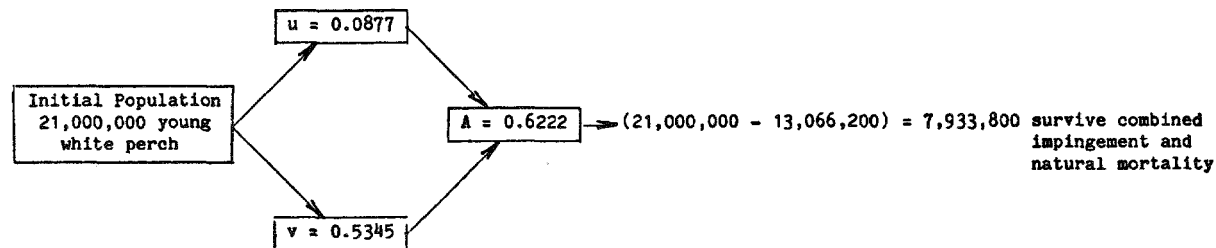
Table 1.8-15

NUMERICAL RELATIONSHIPS BETWEEN IMPINGEMENT AND NATURAL MORTALITY
IN A TYPE 2 FISHERY

1. SYSTEM WHERE ONLY IMPINGEMENT OR ONLY NATURAL MORTALITY IS OPERATIVE



2. SYSTEM WHERE BOTH IMPINGEMENT AND NATURAL MORTALITY ARE OPERATIVE (COMPETING SOURCES OF MORTALITY)



$$m_o = \frac{9,101,400 - 7,933,800}{9,101,400} = \frac{1,167,600}{9,101,400} = 0.1283^* = 12.8\%$$

$$m_{o,5+} = \frac{20,621,090 - 19,208,630}{20,621,090} = \frac{1,412,460}{20,621,090} = 0.0685 = 6.8\%$$

*This should equal m of 0.1282. The slight difference is due to round-off error in the computations.

More recently this method was used to study larval and early juvenile mortality for Atlantic tomcod (Microgadus tomcod) in the Hudson River estuary (1.8-20). Successive sampling was conducted weekly over the entire range of the larval population after recruitment from hatching was complete. The plotted data (Figure 1.8-6) for April and May 1975 showed a larval/early juvenile mortality of 4.6% per day which, when applied to the entire first year of life, amounted to an annual mortality, A, of 99.99% for the population. This is consistent with age-distribution data for Hudson River tomcod which indicate the presence of very few fish that have lived beyond their first year of growth.

This method can be used with either average densities (C/f) or actual population estimates through time as input to the equation:

$$\log_e \left(\frac{N_o}{N_t} \right) = (P + M)t \quad (1.8-4)$$

In this case, actual population numbers were calculated for each sample date by a variation of Eq. 1.4-34 in Section 1.4.2.2. That is, the average larval concentration (number of larvae/1000 m³) from each river stratum was multiplied by the water volume (number of 1000 m³ segments) in each stratum. The numbers of larvae in each stratum were then summed over all strata to obtain estimates of the entire estuarine population for each sample date.

As for most types of mortality estimates, an assumption to be met is that all age groups be equally vulnerable to the collecting gear. In a study of this type the assumption of equal vulnerability is often confounded by increasing avoidance of the sampling gear and/or emigration out of the sampling area as larvae grow into juveniles with greater swimming ability. It was possible to conduct this sampling program within a manageable framework because the population was well defined spatially and remained in a fairly localized (approximately 30-mile long) segment of the estuary during the sampling period. Efforts were made on each sampling date to cover both tails of the longitudinal distribution of the population.

A series of simultaneous replicate tows with nets of 571 μ and 3000 μ mesh size was conducted to evaluate avoidance. Because of different collection efficiencies for the two mesh sizes it was assumed that avoidance would be made evident by

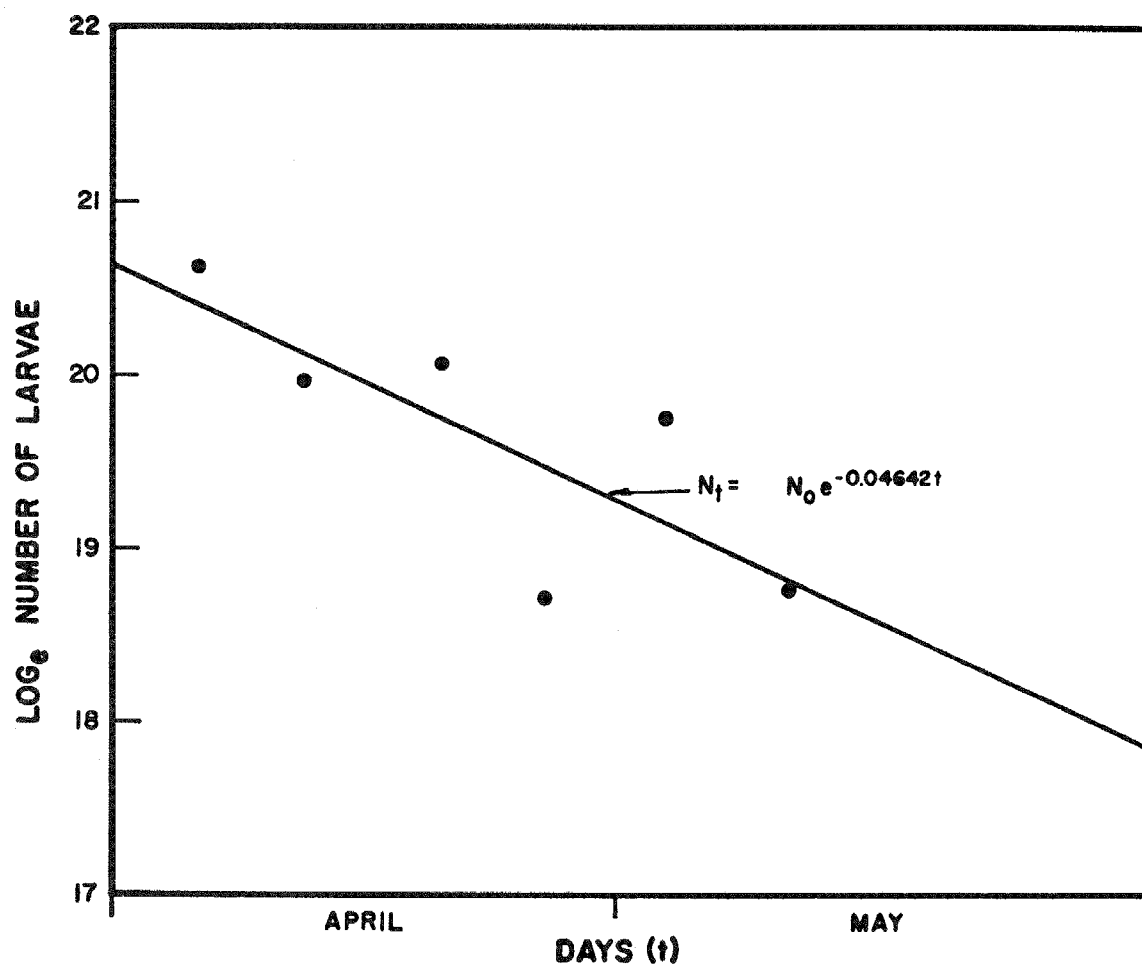


Figure 1.8-6. Calculation of mortality for Atlantic Tomcod larvae and early juveniles using successive population estimates, 1975.

differences in larval concentration and/or mean length of larvae. Data collected by the two nets were almost identical, and it was concluded that net avoidance did not significantly influence population and mortality estimates in this study.

1.8.3.3 Estimating Mortality from Growth Data. Estimates of total mortality (Z) can be gleaned from knowledge of growth rates combined with average length at recruitment and average length of fish in the catches by the following equation derived by Beverton and Holt (1.8-15):

$$Z = \frac{K(L_{\infty} - \bar{l})}{\bar{l} - l'} \quad (1.8-5)$$

where

- l' = the smallest length of fish that are fully represented in catch samples
- $\bar{l}$ = the average length computed from l' upward
- $*L_{\infty}$ = mean asymptotic (ultimate) length for a species
- $*K$ = rate at which a species approaches its asymptotic length

The von Bertalanffy growth equation parameters, L_{∞} and K , can be estimated by a Walford (1.8-21) plot of length at age $t + 1$ (l_{t+1}) against length one year earlier (l_t). The slope of this line is Ford's (1.8-22) growth coefficient (k), which is equal to e^{-K} , and therefore $K = -\log_e k$. The y-axis intercept is $L_{\infty}(1-k)$, from which L_{∞} can be calculated. The asymptotic length, L_{∞} , is also the length (measured on the abscissa) at which the line $l_{t+1} = L_{\infty}(1-k) + kl_t$ cuts the 45 degree diagonal from the origin. An example, from Hudson River white perch age and growth data is shown in Figure 1.8-7, where $K = 0.393$ and $L_{\infty} = 222.76$.

The data in Figure 1.8-7 can also be used to provide a check on gear selectivity. Ricker (1.8-14) points out that if the collection gear selects for the larger of the younger fish, the left end of the line will be depressed, i.e., the points will tend to fall below the regression line. Ricker (1.8-14) stresses that younger age groups which fall below the Walford line established by the older fish should be

*Both L_{∞} and K are parameters of the von Bertalanffy growth equation which expresses length (l) at age (t) by

$$l_t = L_{\infty} [1 - e^{-K(t-t_0)}]$$

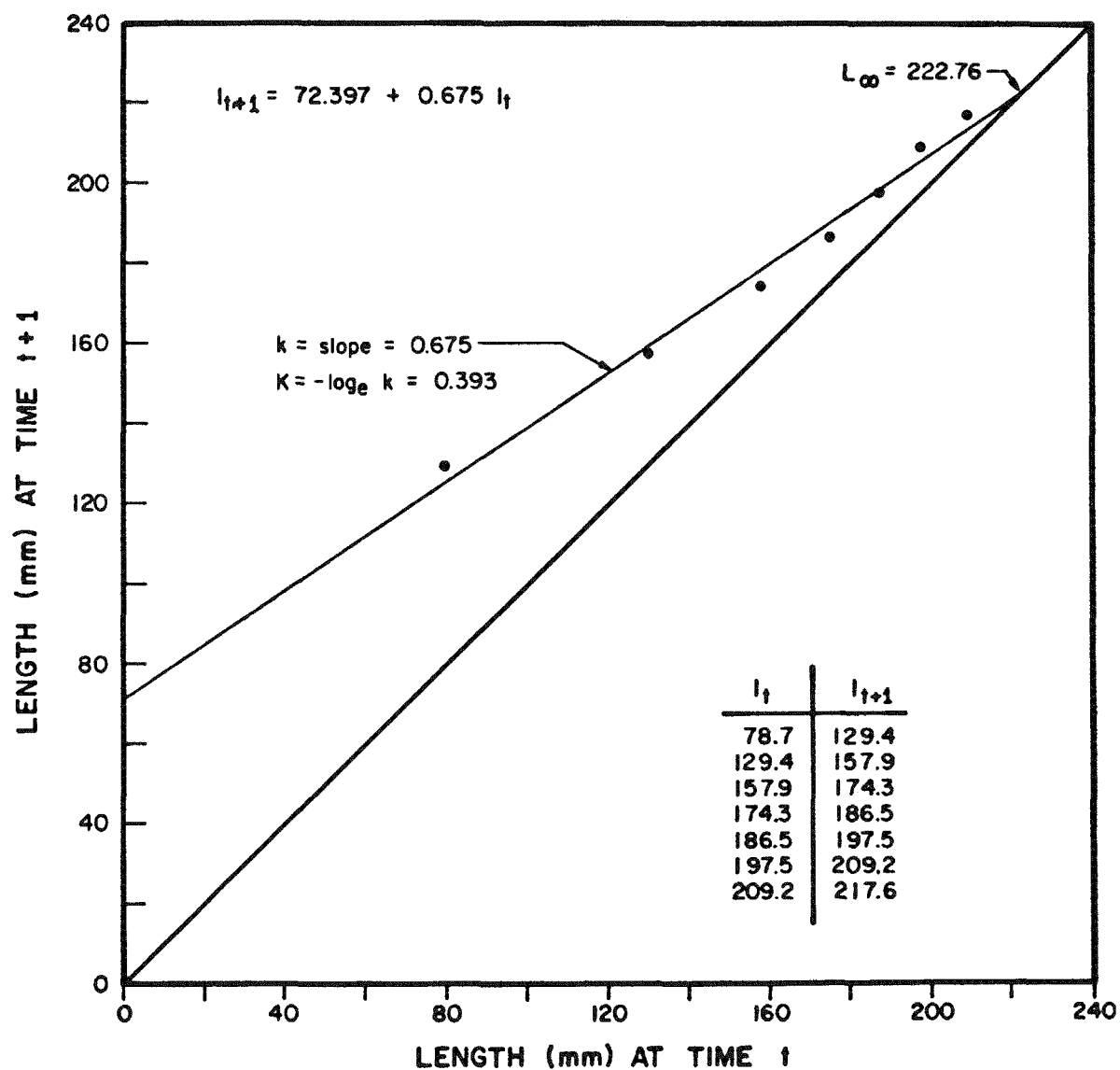


Figure 1.8-7. Mean length at age combined as weighted means for male and female white perch, 1971-1977 (data from Table 1.8-12 and 1.8-13).

rejected from the analysis. Examination of the data in Figure 1.8-7 reveals no such trend, indicating that the trawls and seines used for this study collected a representative sample of the younger fish.

Using Eq. 1.8-5 and estimates of $\bar{L} = 119.8$ mm (based on July-December length-frequency data for 1974, 1975 and 1977 trawl and seine catches) and $l' = 51$ mm, we have

$$Z = \frac{0.393 (222.8 - 119.8)}{119.8 - 51} = 0.59$$

which is substantially lower than the estimate of $Z = 0.77$ obtained from the catch curve (Table 1.8-11).

Ricker (1.8-23) states that this method can be useful when other methods are not available, or as a rough check on other methods, but that estimates of mortality based on average size of the fish must be regarded as inferior to those obtained from a catch curve. One of the assumptions of this method is that the average size of the fish in a catch decreases as fishing mortality increases. This is the typical phenomenon of juvenescence discussed by Thompson and Bell (1.8-24) and Ricker (1.8-25). Beverton and Holt's (1.8-15) equation for predicting the mean length of the fish in a catch also show that mean length decreases as mortality increases, that is:

$$\bar{L} = L_{\infty} \left[1 - \frac{F + M}{F + M + K} e^{-K(t' - t_0)} \right] \quad (1.8-6)$$

where

F = fishing mortality

M = natural mortality

t' = the youngest age fully represented in catches

t_0 = the hypothetical age at which fish length would be zero.

With the above equation, mean length decreases from 60.0 cm to 50.4 cm for constant values of $L_{\infty} = 180.1$, $K = 0.07$, $t' = 3.5$, and $t_0 = 0.01$ when mortality ($F + M$) increases from 0.4 to 0.8, for example. This situation applies to a population

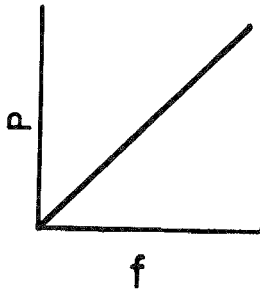
exploited by fishing which is selective for the older, larger segment of the population. However, the use of this method where only entrainment and impingement are operative within the population, and there is no commercial fishery, may lead to depressed estimates of mortality because of the possible size-selective aspects of plant cropping discussed in Section H1.8.2. If entrainment/impingement is selective for the smallest fish, and if this selective mortality is substantial enough to increase the value for $\bar{I}$ calculated from trawl and seine data, then the mortality estimates calculated from Eq. 1.8-5 would be biased downward.

1.8.3.4 Separating Total Mortality (Z) Into Two Components: Natural Mortality (M) And Power Plant Mortality (P). Sampling for age or length structure was used in the preceding sections to obtain estimates of the total instantaneous mortality coefficient, Z. However, the only way mortality could be divided into its two components, P and M, was to calculate P on the basis of the actual knowledge of impingement (or entrainment) and standing crop values and to obtain M by subtraction. This section outlines methods employed to obtain separate estimates of P and M when standing crop estimates are not available.

The requirements and assumptions to be fulfilled so that these methods are applicable as follows (1.8-14, 1.8-15).

1. Fishing effort (equated here with plant flow) is sufficiently great to remove at least a moderately large fraction of the stock in a year.
2. The amount of effective fishing effort must vary considerably between years or with the age of the fish, i.e., it must vary enough to produce measurable changes in estimates of Z.
3. Natural mortality, M, remains constant and independent of fluctuations in P, both between years and between age groups.
4. Plant mortality, P, is directly proportional to plant intake flow or plant "fishing intensity," f.

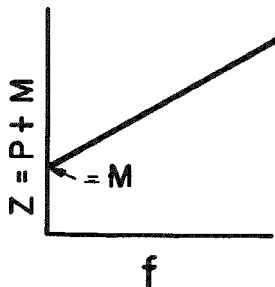
If the last assumption is to be fulfilled the following relationships must hold:



where:

$$P = qf, \quad (1.8-7)$$

q being the slope of the line representing the catchability coefficient,
and



where:

$$(P + M) = qf + M \quad (1.8-8)$$

Thus, the quantity $(P + M)$ is linearly related to f , the constant term being M . The more nearly proportional the relation between plant flow and total mortality, the smaller M must be in relation to P , and vice versa. The procedure for estimating the two unknown parameters of Eq. 1.8-8, q and M , and thus separating total mortality into its components, generally follows the form of analyzing the relationship between changes in f and changes in Z .

1.8.3.4.1 Plant flow (fishing effort) stabilized at two different levels in time. This method was first developed by Silliman (1.8-26) for the California sardine fishery. The necessary requirement here is that the plant intake flow shall have been roughly stable for two periods, each long enough for the corresponding total mortality coefficients, Z , to have been estimated from catch data as described in Sections 1.8.3.1 or 1.8.3.2. In addition, of course, the plant flows for the two periods must be known and expressed in standardized units. The computations are as follows (Ricker 1.8-14)

$$P_1 + M = Z_1 \quad (1.8-9)$$

$$P_2 + M = Z_2 \quad (1.8-10)$$

$$P_1/P_2 = f_1/f_2 \quad (1.8-11)$$

The right-hand sides of these equations are known, so they can be solved directly for P_1 , P_2 , and M ; also $q = P_1/f_1 = P_2/f_2$.

This method could be used, for example, in a hypothetical situation where the average annual plant flow during 1960 through 1972 (f_1), was 1000 cfs and the average annual flow from 1973 through 1978 (f_2) was 4000 cfs. The total mortality coefficient for both periods was estimated by catch curve techniques (Section 1.8.3.1) as, say, $Z_1 = 0.60$ for 1960 through 1972 and $Z_2 = 0.80$ for 1973 through 1978. With the knowledge that $f_1/f_2 = 0.25$, then, Eq. 1.8-11 yields $P_1 = 0.25 P_2$. Further, using Eq. 1.8-9 and 1.8-10, $0.25 P_2 + M = 0.60$ and $P_2 + M = 0.80$. Subtracting, we have $0.75 P_2 = 0.20$, or $P_2 = 0.27$. Thus $P_1 = 0.07$, and instantaneous natural mortality for both periods is $M = 0.53$.

1.8.3.4.2 Plant flow (fishing effort) varying continuously with time. Many fisheries and power plant monitoring programs are characterized by several years of recorded catch at age and effort data which are used only to describe general population trends. Here, a method is delineated for the use of such data to estimate the natural mortality rate and the catchability coefficient for a population. The method was presented by Beverton (1.8-27) and Beverton and Holt (1.8-15, 1.8-1) and modified to a simplified linear formula by Paloheimo (1.8-28) with the following notation and definitions:

j	subscript referring to the j^{th} age group, where $j = 0, 1, 2$, etc.
n	number of years
i	subscript referring to year i , where $i = 1, \dots, n$
C_{ji}	number of j -year-old fish caught during the year i

f_i	total effective effort during year i
q	catchability coefficient defined as the fraction of the population caught by one unit of effort
M	instantaneous natural mortality rate
$F_i = qf_i = P_i$	instantaneous fishing (power plant) mortality during the year i
$\hat{q}, \hat{M}$	estimates of q, M

The basic data for estimating mortality rates with this method are pairs of values C_{ji}, f_i . These data must be available for at least two age groups and for several years; the number of years required for a reliable estimation depends on the accuracy of the age-specific catch and effort data. Paloheimo's linear formula is:

$$\log_e \frac{C_{1i}/f_i}{C_{2i+1}/f_{i+1}} = q [1/2 (f_i + f_{i+1})] + M \quad (1.8-12)$$

Making $\log_e \frac{C_{1i}/f_i}{C_{2i+1}/f_{i+1}} = Z_i$ and $1/2 (f_i + f_{i+1}) = \bar{f}_i$, least squares regression

estimates for q and M from Eq. 1.8-12 are now given by:

$$\hat{q} = \frac{(n-1) (\sum \bar{f}_i Z_i) - (\sum \bar{f}_i) (\sum Z_i)}{(n-1) (\sum \bar{f}_i^2) - (\sum \bar{f}_i)^2} \quad (1.8-13)$$

and

$$\hat{M} = \frac{(\sum \bar{f}_i^2) (\sum Z_i) - (\sum \bar{f}_i) (\sum \bar{f}_i Z_i)}{(n-1) (\sum \bar{f}_i^2) - (\sum \bar{f}_i)^2} \quad (1.8-14)$$

where n is the number of years for which data are available.

Eq. (1.8-12) is a linear equation of the form

$$Z_i = M + q\bar{f}_i$$

where q equals the slope and M equals the intercept of the regression line at zero effort.

This method is applicable only in those situations where fishing or power plant mortality is a significant proportion of total mortality and where the level of effort has varied enough to produce a positive slope, q . If levels of effort, e.g., power plant flows, have changed substantially during a long-term study without generating a positive slope when Z_i values are plotted against $\bar{f}_i$ values, then this is prima facie evidence that plant flows do not represent a significant cropping impact - at least to the life stages examined within the scope of this analysis. Because of changes in q due to weather, hydrographic conditions, etc., and the systematic variations in q associated with the age and size of the fish, it is obvious that catchability is not constant but a variable, and any estimate of q is only an estimate of its average value (1.8-28).

An example of the linear method is presented in Tables 1.8-16 and 1.8-17 based on ten years of hypothetical impingement data. The catches of one- and two-year-old fish were generated by the catch equations shown at the bottom of Table 1.8-16, and constant values for q and M of 0.1 and 0.7, respectively. Power plant mortality can now be calculated as $P = q\bar{f}_i$ and, in this example, ranges from $P = 0.1 \times 0.45 = 0.045$ to $P = 0.1 \times 1.625 = 0.16$.

1.8.3.4.3 Catchability and plant mortality varying with the age of fish. The above method assumes, among other things, equal catchability or vulnerability to the fishing method for each age group of concern. However, in general, vulnerability of a species to impingement decreases with the age and size of the fish present in the vicinity of the intake. Also, the above method is based on the supposition that changes in P are caused by changes in plant flow, f ; however the same principle can be used to separate P and M if P changes in a known way with the age of fish, even though f may be effectively constant (1.8-15). In this case, the age composition of an impingement catch, for example, differs from that of the population, and to obtain comparable abundance indexes the catch of each age group must be adjusted to that of one particular age group taken as a reference.

Following the methodology and notation of Beverton and Holt (1.8-15) and using the youngest age group, v , present representatively in the catch as reference,

Table 1.8-16

HYPOTHETICAL IMPINGEMENT CATCHES FOR TWO SUCCESSIVE
AGE GROUPS OVER A 10-YEAR PERIOD

Year	Effort ^a	Catch ^b	
		Yearlings (C_{1_i})	2-Year Olds ($C_{2_{i+1}}$)
1	0.40	28,264	
2	0.50	35,176	16,783
3	0.75	52,190	24,653
4	0.50	35,176	16,206
5	1.50	101,044	47,730
6	1.75	116,628	49,848
7	1.00	68,834	28,694
8	0.50	35,176	15,805
9	0.75	52,190	24,653
10	1.50	101,044	46,552

^aTotal number ($\times 10^6$) of million-gallon units of flow through all plants during each year.

^bValues generated from $N_0 = 1,000,000$ recruits each year with $M = 0.7$ and $q \cong 0.1$ by catch equations

$$C_{1_i} = N_0 \left[1 - e^{-(M+qf_i)} \right] \frac{qf_i}{M+qf_i}$$

$$C_{2_{i+1}} = \left\{ N_0 e^{-(M+qf_i)} \left[1 - e^{-(M+qf_{i+1})} \right] \right\} \frac{qf_{i+1}}{M+qf_{i+1}}$$

Table 1.8-17

DATA IN TABLE 1.8-16 CONVERTED INTO z_i AND $\bar{f}_i$ VALUES FOR
REGRESSION ANALYSISⁱ

$z_i = \log_e \frac{C_{1i}/f_i}{C_{2i+1}/f_{i+1}}$	$\bar{f}_i = 1/2(f_i + f_{i+1})$
0.74437	0.450
0.76093	0.625
0.76404	0.625
0.79342	1.000
0.86073	1.625
0.84269	1.375
0.77822	0.750
0.76093	0.625
0.80747	1.125
$\sum z_i = 7.11280$	$\sum \bar{f}_i = 8.20$
$\sum z_i^2 = 5.63400$	$(\sum \bar{f}_i^2) = 8.73375$
	$(\sum \bar{f}_i)^2 = 67.24$
$\sum \bar{f}_i z_i = 6.60653$	
$(\sum \bar{f}_i)(\sum z_i) = 58.32496$	
$\hat{q} = \frac{(n-1)(\sum \bar{f}_i z_i) - (\sum \bar{f}_i)(\sum z_i)}{(n-1)(\sum \bar{f}_i^2) - (\sum \bar{f}_i)^2} = 0.09977$	
$M = \frac{(\sum \bar{f}_i^2)(\sum z_i) - (\sum \bar{f}_i)(\sum \bar{f}_i z_i)}{(n-1)(\sum \bar{f}_i^2) - (\sum \bar{f}_i)^2} = 0.69941$	
$n = 10 \text{ years}$	

the associated plant mortality is P_v and that for any other age group $v + r$ can be written as $Y_{v+r} \cdot P_v$, where Y_{v+r} is the efficiency of impingement on any group $v + r$ relative to age group v . The factor Y_{v+r} is used to adjust the index of abundance for age group $v + r$ obtained from the catch of that age group, so that it bears the same relation to true abundance as does the index for age group v . Thus, if the true abundances of age groups v and $v + r$ are N_v and N_{v+r} , and the unadjusted catch values are N'_v and N'_{v+r} we have the relationships

$$N'_v = kN_v \quad (1.8-15)$$

and

$$\frac{N'_{v+r}}{Y_{v+r}} = kN_{v+r} \quad (1.8-16)$$

For any pair of age groups $v+r$ and $v+r+1$, Eq. 1.8-4 therefore becomes:

$$\log_e \left[\frac{Y_{v+r+1} N'_{v+r}}{Y_{v+r} N'_{v+r+1}} \right] + Q = (Y_{v+r} \cdot P_v) + M, \quad (1.8-17)$$

where the entire left-hand side of the equation is an estimate of total instantaneous mortality ($Z = P + M$). To estimate P_v and M from this equation, a minimum of three and preferably four age groups with differing selection factors are required, since the number of data points for the regression analysis is one less than the number of age groups.

The "correction term" Q is necessary because the log abundance ratio alone does not provide an estimate of the total mortality coefficient, now that P varies with age. The magnitude of Q increases with the magnitude of the change in P between age groups. Beverton and Holt (1.8-15) derive Q in their Appendix D, p. 72, and it is expressed as:

$$Q = \log_e \left[\frac{\left(\frac{Y_{v+r} \cdot P_v + M}{Y_{v+r+1} \cdot P_v + M} \right) \left\{ 1 - e^{-\left(\frac{Y_{v+r+1} \cdot P_v - M}{Y_{v+r} \cdot P_v - M} \right)} \right\}}{\left(\frac{Y_{v+r+1} \cdot P_v + M}{Y_{v+r} \cdot P_v + M} \right) \left\{ 1 - e^{-\left(\frac{Y_{v+r} \cdot P_v - M}{Y_{v+r+1} \cdot P_v - M} \right)} \right\}} \right] \quad (1.8-18)$$

The correction term ignored for the moment, the log ratios of abundance for adjacent age groups from the left hand side of Eq. 1.8-17 are plotted against the different values of Y_{v+r} for each pair to obtain the first estimates of P_v (slope) and M (intercept). These initial estimates are then used to compute approximate values of Q for each pair of age groups, which can then be added to the corresponding log abundance ratios and the whole of the left side of Eq. 1.8-17 replotted against Y_{v+r} . This gives more exact estimates of P_v and M , and the process is repeated until there is no appreciable difference in successive estimates of these parameters. Ricker (1.8-14) states that the third or fourth fitting is likely to be the last that will be at all rewarding.

A hypothetical, single-year impingement catch separated by age groups with their associated relative efficiency values is shown in Table 1.8-18. Regression of $\log_e \left[\frac{Y_{v+r+1}}{Y_{v+r}} \cdot \frac{N'_{v+r}}{N'_{v+r+1}} \right]$ vs Y_{v+r} provided the initial values of $P_v = 0.366$ and $M = 0.672$ shown for iteration number 1 below. These values were then used in Eq. 1.8-18 to obtain approximate values of Q which were used in iteration number 2 to provide refined estimates of P_v and M . This process was continued until there were no substantive changes after iteration 3:

Iteration No.	P_v	M
1	0.366	0.672
2	0.397	0.698
3	0.400	0.700
4	0.400	0.700

Plots of the initial variates and the final adjusted variates are shown in Figure 1.8-8. Based on the impingement catch in Table 1.8-18 the instantaneous power plant mortality for age group 0 is $P_0 = 0.40$, for age group 1 is $P_1 = 0.75 P_0 = 0.30$, for age group 2 is $P_2 = 0.25 P_0 = 0.10$, and for age group 3 is $P_3 = 0.10 P_0 = 0.04$.

The most critical aspect of this method is to obtain reliable estimates of the differential selection factors Y_{v+r} , because the accuracy of estimates P_v and M depends on these relative efficiency factors. One way to accomplish this is to compare specific catch values for impingement collections to catch

Table 1.8-18

HYPOTHETICAL IMPINGEMENT CATCH COMPOSED OF FOUR AGE GROUPS
WITH VARYING CATCHABILITY

Age Group (v+r)	Number Impinged*	Relative Efficiency of Impingement	$\log_e \left(\frac{Y_{v+r+1} \cdot N'_{v+r}}{Y_{v+r} \cdot N'_{v+r+1}} \right)$
0	$N'_0 = 242,592$	$Y_0 = 1.00$	
1	$N'_1 = 63,124$	$Y_1 = 0.75$	1.05860
2	$N'_2 = 8,429$	$Y_2 = 0.25$	0.914811
3	$N'_3 = 1,555$	$Y_3 = 0.10$	0.773912

*Values generated from catch equations similar to Table 1.8-16.

$N_0 = 1,000,000$ age-0 recruits each year.

$qf = 0.4$

$M = 0.7$

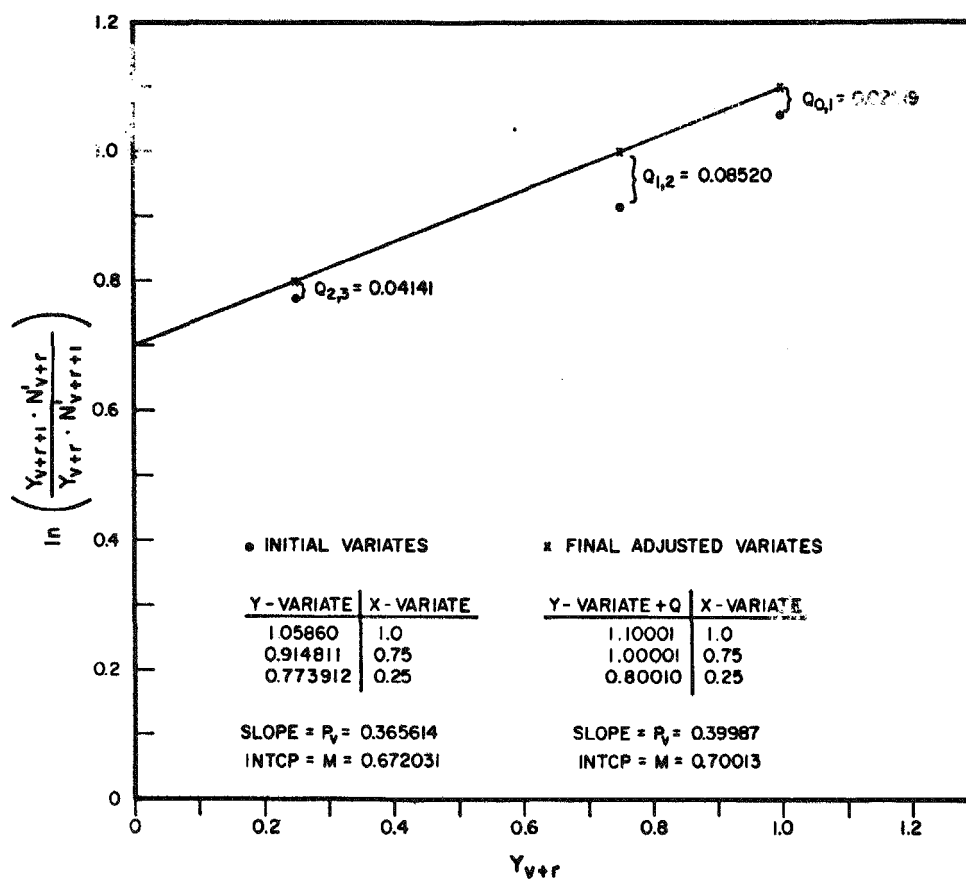


Figure 1.8-8. Plot of Initial Variates and Final Adjusted Variates.

values for a gear fishing the same population and which is known to catch all the age groups in question with equal efficiency. The particular units in which the fishing effort of the calibrating gear is expressed do not have to be the same as impingement units, e.g., trawl catches may be expressed in catch per unit time and impingement catches as catch per unit volume, or total catches alone may be used. If the catches per unit effort of age groups v and $v+r$ by impingement are C/f_v and C/f_{v+r} , while those of the calibrating gear are C/f'_v and C/f'_{v+r} , then the selection factor Y_{v+r} defining the ratio P_{v+r}/P_v for impingement is given by the expression:

$$Y_{v+r} = \left\{ \frac{C/f_{v+r}}{C/f_v} \right\} / \left\{ \frac{C/f'_{v+r}}{C/f'_v} \right\} \quad (1.8-19)$$

For example, using the data in Table 1.8-18 and hypothetical catch per ten minute effort values for young ($C/f = 57.7$) and yearling ($C/f = 20.0$) fish in trawl collections during the same time period and in the same vicinity as the plant, we can calculate a relative efficiency for impingement of yearlings as:

$$Y_{v+r} = \frac{\left\{ \frac{63,124}{242,592} \right\}}{\left\{ \frac{20.0}{57.7} \right\}} = 0.75$$

Perhaps the most useful way to obtain the required selection factors is to utilize a mark-recapture experiment which avoids the difficulty of having to rely on one particular gear as being truly unselective. In addition, a mark-recapture experiment, if properly conducted according to the guidelines in Section 1.4.1, should provide investigators with a working estimate of the exploitation rate (the term u in Table 1.8-14) for each age group. The procedure here is to set up an experimental "subpopulation" of a known number of marked fish from each age group and then to monitor impingement closely enough to collect all marked fish impinged. The hypothetical data below provide an example:

Age Group	0	1	2	3
Number Marked (M)	1000	800	600	400
Number of Returns (R)	700	420	105	28
Exploitation Rate $u = \frac{R}{M}$	0.700	0.525	0.175	0.070
Differential Selection Factors	1.00	0.75	0.25	0.10

$$y_{v+r} = \frac{u_{v+r}}{u_v}$$

(1.8-20)

The differential selection factors could be attained from a very localized marking program, i. e., marking could be restricted to a relatively small segment of the water body in the vicinity of the power plant. However, if the plan were to use the data to provide exploitation rates as well, then the experiment would necessarily be much broader in scope, and the marked fish would have to be distributed randomly among the population of interest. The level of marking effort would then have to be much greater to ensure enough returns to provide adequate data for selection factor analysis.

1.8.3.5 Estimation Of Power Plant Mortality (P) by The Survey-Removal Method.

The survey-removal or change-in-ratio method for estimating population size was described, with notation, in Section 1.4.4. This same method can be used to obtain estimates of fishing or power plant mortality. Lander (1.8-29) shows that estimates of fishing rates (power plant mortality rates) are independent of the natural mortality rate, M , when natural mortality operates at the same rate on both the X and Y populations, i.e., $M_x = M_y$. This fact was illustrated in Table 1.4-12, Section 1.4.4, which showed that the following equations could be used to estimate power plant mortality on segment X (P_x), segment Y (P_y), and the total population (P_T):

$$P_x = -\log_e \left[1 - \frac{C_x (p_1 - p_2)}{p_1 (C_x - p_2 C_T)} \right] \quad (1.8-21)$$

$$P_y = P_T + \log_e \left(\frac{1 - p_1}{1 - p_2} \right) \quad (1.8-22)$$

$$P_T = P_x + \log_e \left(\frac{p_2}{p_1} \right) \quad (1.8-23)$$

where

C_x = catch (removal) of x -type fish

C_T = catch (removal) of $x + y$ -type fish

p_1 = fraction of total population in X -class during the first (survey 1)

p_2 = fraction of total population in X -class during the last (survey 2)

Chapman and Murphy (1.8-30) and Seber (1.8-31) show that when removals are fairly continuous during the period between the first and last surveys and when it is assumed that $M_x = M_y$ and that each of the two classes x and y are equally vulnerable to the sampling gear,

then

$$P_x = \frac{\left[\left(\frac{C_x}{n_{1x}} \right) / \left(\frac{C_y}{n_{1y}} \right) \right] \cdot \log_e \left[\left(\frac{n_{2x}}{n_{1x}} \right) / \left(\frac{n_{2y}}{n_{1y}} \right) \right]}{1 + 1/2 \log_e \left[\left(\frac{n_{2x}}{n_{1x}} \right) / \left(\frac{n_{2y}}{n_{1y}} \right) \right] - \left[\left(\frac{C_x}{n_{1x}} \right) / \left(\frac{C_y}{n_{1y}} \right) \right]} \quad (1.8-24)$$

and

$$P_y = P_x + \log_e \left[\left(\frac{n_{2x}}{n_{1x}} \right) / \left(\frac{n_{2y}}{n_{1y}} \right) \right] \quad (1.8-25)$$

where

n_{1x} = the number of X-class fish collected during survey one

n_{2x} = the number of X-class fish collected during survey two

n_{1y} = the number of Y-class fish collected during survey one

n_{2y} = the number of Y-class fish collected during survey two

For example, with the hypothetical population data in Table 1.4-12, Section 1.4.4, estimates of $P_x = 0.5997$ and $P_y = 0.19999$ are calculated from Eqs. 1.8-24 and 1.8-25. Using Eqs. 1.8-24 and 1.8-25 and assigning hypothetical values of $n_{1x} + n_{1y} = 3381$ and $n_{2x} + n_{2y} = 2979$, and since $P_1 = 0.80$ and $P_2 = 0.7284$ from Table 1.4-12 we have:

$$\begin{aligned} P_x &= \frac{\left[\left(\frac{360,951}{2,705} \right) / \left(\frac{36,254}{676} \right) \right] \cdot \log_e \left[\left(\frac{2,170}{2,705} \right) / \left(\frac{809}{676} \right) \right]}{1 + 1/2 \log_e \left[\left(\frac{2,170}{2,705} \right) / \left(\frac{809}{676} \right) \right] - \left[\left(\frac{360,951}{2,705} \right) / \left(\frac{36,254}{676} \right) \right]} \\ &= \frac{-0.995201}{-1.688113} \\ &= 0.5895 \end{aligned}$$

and

$$P_y = 0.5895 + \log_e \left[\left(\frac{2,170}{2,705} \right) / \left(\frac{809}{676} \right) \right]$$
$$= 0.1895$$

The method for estimating the variance of these estimates of P_x and P_y is described by Chapman and Murphy (1.8-30) and Seber (1.8-31).

Both Lander (1.8-29) and Chapman and Murphy (1.8-30) point out that bias in the estimate of P_x accruing from the violation of the assumption that $M_x = M_y$ will not be appreciable if M_x and M_y are not too widely different and are small relative to P_x .

1.8.4 Abundance Trends

If power plant cropping were exerting a substantial impact upon a fish population an obvious expectation is that the population would decline to a new, lower equilibrium level, or that it would decline to extinction where compensatory response was overwhelmed. Thus, one of the most direct and pragmatic methods of assessing the viability or resiliency of a population is to monitor abundance trends in the face of increased cropping to see if population size decreases or remains essentially the same.

Relative abundance as explained in Section 1.4.2.1, rather than absolute abundance, may be adequate to describe year-to-year trends in population size, assuming that annual average C/f values are proportional to the average population present each year. This requires that the survey program be properly designed so that the annual proportion of the total population captured by a sampling gear remains effectively constant from year to year, i.e., does not vary substantially because of changes in factors such as weather or the behavior and distribution of the fish (1.8-32). To ensure that the above assumption is satisfied, the survey should incorporate the entire range of the target population, including sampling from all major habitats or strata within the range; the program should also be consistent with respect to collection gear, stations, months and time of day sampled each year.

Since a single collection gear cannot fulfill the requirement of sampling from all major habitats, abundance indexes should be based on more than one type of sampling gear. To formulate a composite index of abundance from two or more sampling gears, absolute abundance values must be used rather than relative abundance in the form of catch per unit effort (C/f) values. The reason for this is that C/f values from beach seines and bottom trawls, for example, cannot simply be combined into a meaningful abundance index without a consideration of the actual standing crop represented by the C/f value from each gear (or habitat). In other words, to combine information from both gears, the C/f value for seines must be weighted by a factor representing the proportion of the total population residing in the shore zone, and the trawl C/f value must be similarly weighted by that proportion residing in the offshore zone.

To estimate these proportions, absolute standing crop values must be determined by the methods described in Section 1.4.2.2. Of course, if the absolute standing crop is estimated, the need for relative abundance values is negated. It should be kept in mind that standing crop estimates should be adjusted by the collection efficiency factor for each gear type, and therefore gear efficiency studies should be conducted in conjunction with absolute abundance studies.

If both sampling gears (or all gears where more than two were used) showed the same trend over several years, one gear, perhaps the most costly one to employ, could be discarded. Then relative abundance would be a sufficient indicator of population trends. However, the multi-gear approach should be utilized for several years to ensure that year-to-year distributional shifts from, for example, the shore zone to the deeper water zone in response to a warming trend, do not affect the annual abundance index for each gear.

An important consideration in the monitoring of abundance trends is that the C/f values or standing crop estimates be age-specific, at least to the point of separating young-of-the-year from older fish. Watt (1.8-33) says, rather categorically: "Any estimate that pools data on all age groups will typically be so meaningless as to be worthless." The problem with combining all age groups, especially for a long-lived stock, is that the abundance index loses sensitivity and tends to indicate that the stock is more stable than it is in reality.

A vital link within the life cycle of a self-perpetuating fish stock is the appearance each year of young-of-the-year fish. Estimation of the relative

magnitude of this appearance, commonly known as recruitment, is a key factor in monitoring abundance trends. Secular trends in recruitment reflect all the parameters interacting within the framework of a dynamic population, i.e., it is the end product of such basic factors as birth, death and growth. Annual analysis of young-of-the-year recruitment should be an integral part of any study to assess the effects of entrainment and impingement. These sources of mortality are, almost by definition, selective for the youngest fish within a population, and therefore a commonsense approach dictates monitoring annual recruitment of those species most vulnerable to intake effects. It is easy to age young-of-the-year fish compared to older fish, and fish can often be assigned to age group 0 based on length alone, since there is usually little size overlap between ages 0 and 1 for fish species with discrete annual spawning periods.

A thought-provoking method for examining trends represented by changes in population size was presented by Tanner (1.8-34), who analyzed growth rates of populations (differentiated from somatic body growth discussed in Section 1.8.2) with respect to their density. The objective of his study was to determine if changes in the size of animal populations were determined solely by environmental variations or if the density of the population itself affects such changes.

The change in numbers of a population with respect to time, dN/dt , equals rN , where N is the number in the population and r is the rate of change per unit population.

thus

$$rN = \frac{dN}{dt} \quad (1.8-26)$$

and

$$r = \frac{dN}{dt} / N \approx \frac{\Delta N / N}{\Delta t} \quad (1.8-27)$$

When $t = 1$, which is true if the population was censused at regular intervals, then

$$r_t = \frac{N_{t+1} - N_t}{N_t} = \frac{N_{t+1}}{N_t} - 1 \quad (1.8-28)$$

Tanner (1.8-34) shows that a record of a population, $N_0, N_1, \dots, N_t, \dots$, can be used to calculate the population's rate of growth for each interval by Eq. 1.8-28, and that the relation between population growth rate and population density may be determined statistically by correlation analysis if the following three conditions are met:

1. N_t and N_{t+1} are correlated, i.e., are significantly different from a series of random numbers.
2. the population was censused at regular intervals so that $\Delta t = 1$.
3. the estimates or indexes of abundance reflect the actual density of the population.

Tanner (1.8-34) examined 71 animal species (composed of many separate populations) in this manner and, because the correlations between r_t and density were largely negative, he concluded that most animal species were governed by density-dependent factors, i.e., that the growth rate of populations is usually a decreasing function of density. He went on to state that this helps to explain the relative stability of animal populations, which never continue to increase at rates their fertility would allow and which rarely decrease to extinction. This conclusion is relevant to the subject of population response analysis since it embodies the concept that direct mortality at power plant intakes tends to be offset by the very nature of the fish populations involved.

To examine this premise in more detail, the 1971-1978 C/f records were analyzed for four Hudson River fish species, specifically Atlantic tomcod, hogchoker, white perch and striped bass (Table 1.8-19). The computation of a correlation coefficient requires that the two variables be linearly related. Plots of r_t vs C/f_t in Figures 1.8-9 through 1.8-12 suggest that the relationship of these two variables conforms more closely to a negative exponential than a linear relationship. Therefore, the C/f values were transformed to natural logarithms* prior to correlation analysis.

Results of the analyses in Table 1.8-19 show that the correlation coefficients are all negative, although the only value significantly ($\alpha = 0.05$) different from

*If the r_t values are log transformed, the resulting equation will describe a line which does not drop below the zero-value for population growth - an unrealistic situation since r_t must assume both positive and negative values.

Table 1.8-19

CORRELATION ANALYSIS OF POPULATION GROWTH RATE (r_t)
VS THE NATURAL LOG OF BOTTOM TRAWL CATCH PER UNIT EFFORT (C/f_t)
FOR FOUR HUDSON RIVER FISH SPECIES

Species	Year(t)	C/f_t	$\log_e C/f_t$	$r_t = \frac{C/f_{t+1}}{C/f_t} - 1$	Correlation Coefficient
Atlantic tomcod	1971	29.4	3.3810	4.1667	$r = -0.739^*$
	1972	151.9	5.0232	-0.5721	
	1973	65.0	4.1744	1.9415	
	1974	191.2	5.2533	-0.6841	
	1975	60.4	4.1010	0.2169	
	1976	73.5	4.2973	0.4082	
	1977	43.5	3.7728	0.1356	
	1978	49.4	3.9000		
Hogchoker	1971	6.3	1.8405	11.5397	$r = -0.951^{**}$
	1972	79.0	4.3694	-0.4772	
	1973	41.3	3.7209	0.7724	
	1974	73.2	4.2932	1.0068	
	1975	146.9	4.9900	-0.3009	
	1976	102.7	4.6318	0.1052	
	1977	113.5	4.7318	-0.2167	
	1978	88.9	4.4875		
White perch (All age groups combined)	1971	145.7	4.9815	0.0213	$r = -0.622^*$
	1972	148.8	5.0026	-0.2339	
	1973	114.0	4.7362	0.0281	
	1974	117.2	4.7639	-0.1186	
	1975	103.3	4.6376	-0.2197	
	1976	80.6	4.3895	-0.0906	
	1977	73.3	4.2946	1.4379	
	1978	178.7	5.1857		
White perch (Young-of-the- year only)	1971	213.5	5.3636	-0.4496	$r = -0.741^*$
	1972	117.5	4.7664	-0.6698	
	1973	38.8	3.6584	1.5876	
	1974	100.4	4.6092	-0.4721	
	1975	53.0	3.9703	-0.1453	
	1976	45.3	3.8133	0.3355	
	1977	30.1	3.4045	2.0033	
	1978	90.4	4.5042		
Striped bass (Young-of-the- year only)	1971	194.1	5.2684	-0.8393	$r = -0.663^*$
	1972	31.2	3.4404	-0.2724	
	1973	22.7	3.1224	2.7048	
	1974	84.1	4.4320	-0.4031	
	1975	50.2	3.9160	-0.9124	
	1976	4.4	1.4816	0.9318	
	1977	8.5	2.1401	4.4588	
	1978	46.4	3.8373		

*Not significant at $\alpha = 0.05$ (critical value = 0.754, df = 5).

**Significant at $\alpha = 0.05$.

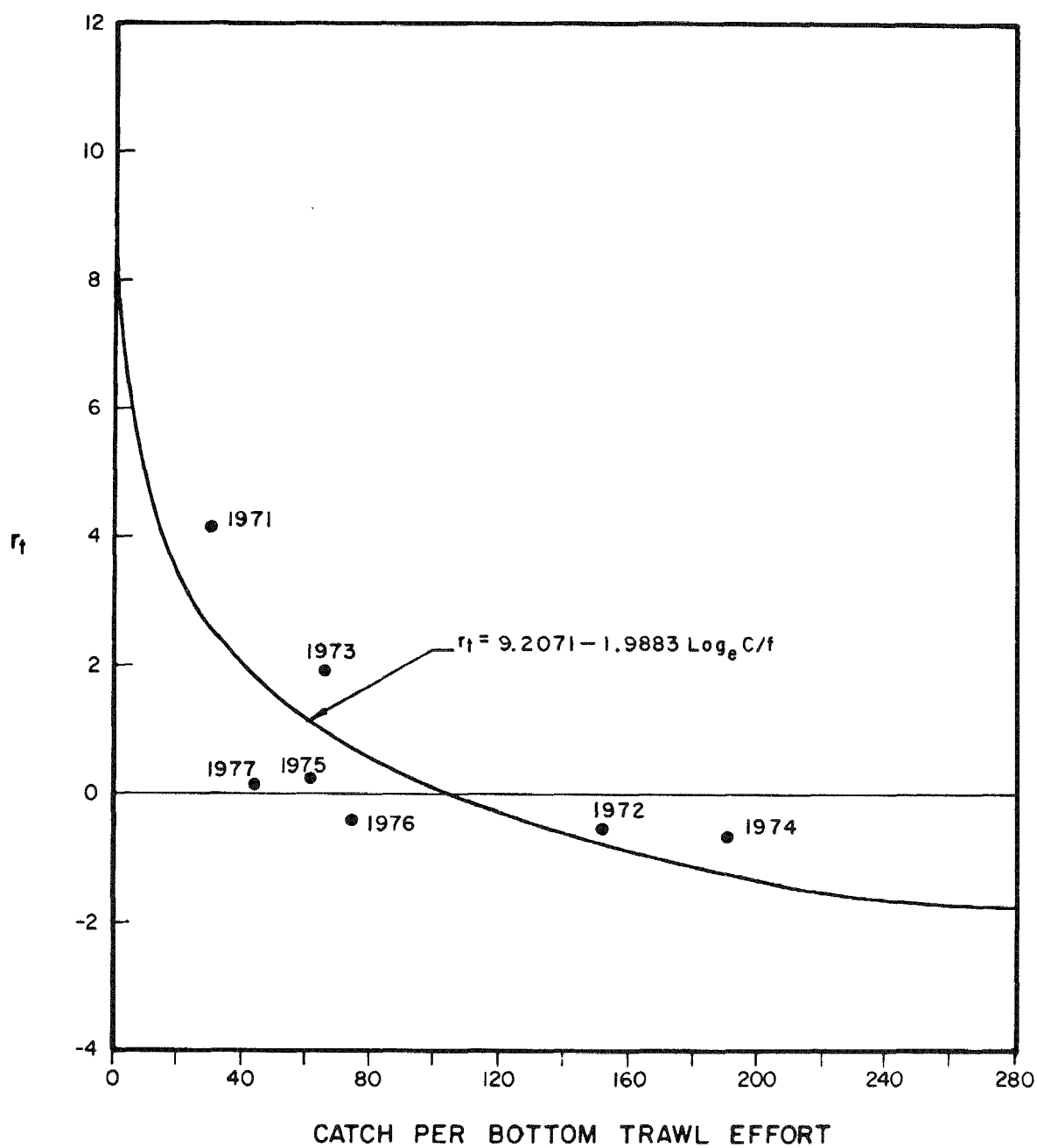


Figure 1.8-9. Atlantic Tomcod Population Growth, 1971-1978.

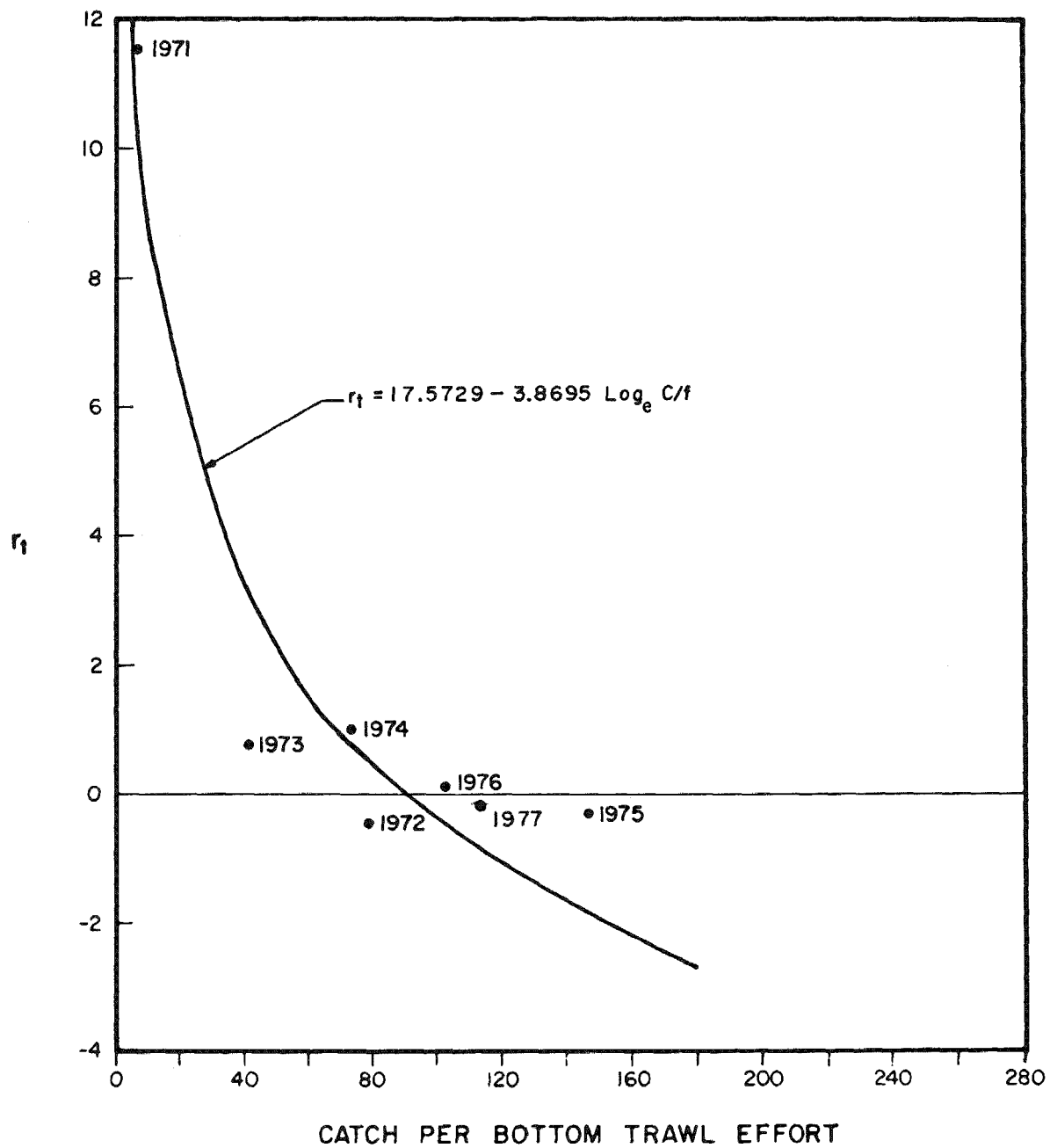


Figure 1.8-10. Hogchoker Population Growth, 1971-1978.

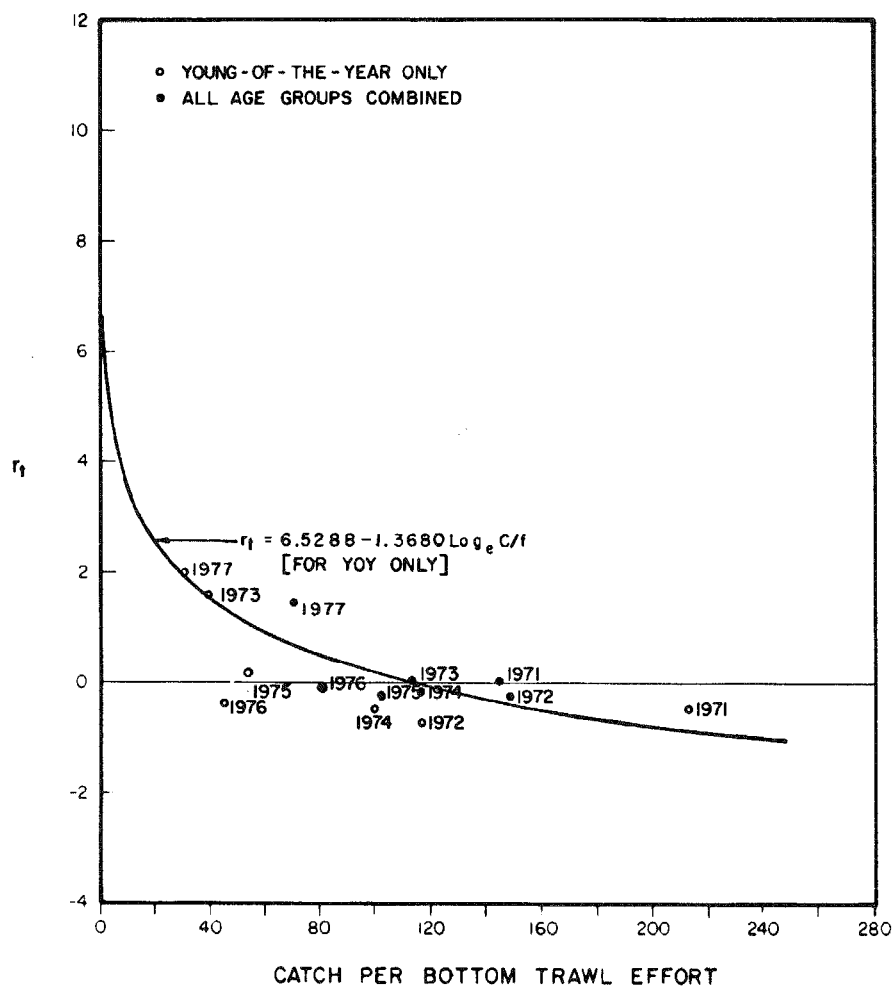


Figure 1.8-11. White perch population growth for young-of-the-year only and all age groups combined, 1971-1978.

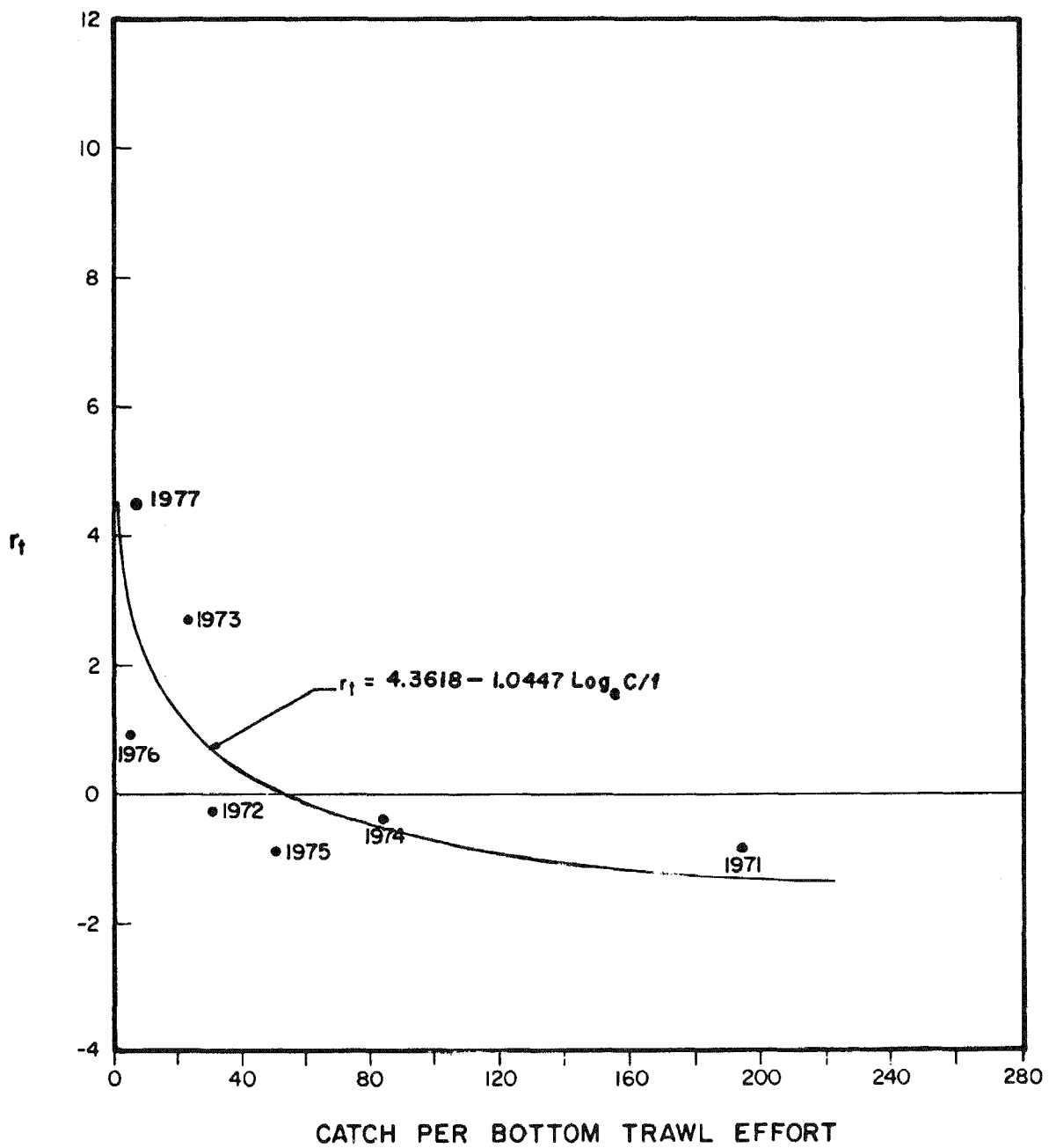


Figure 1.8-12. Striped Bass population growth for young-of-the year only, 1971-1978.

zero is for hogchokers ($r = -0.951$). The analyses are somewhat restricted because each has only five degrees of freedom ($df = n-2$), but they are sufficient to demonstrate that the relationships between population growth rate and density are in the direction expected for self-regulating populations.

1.8.5 Reproduction

Fish populations, like most persistent biological populations, are characterized by oscillations around a semi-stable equilibrium abundance. These oscillations are driven by the complex interaction of biotic and abiotic checks and balances influencing the rates of death and birth within the population. These two phenomena, death and birth, form the very basis of population regulation, and all of the mechanisms commonly associated with such regulation are effective only to the degree that they are able to influence death and birth rates.

Population size for all animal species generally remains remarkably stable considering environmental variation and the reproductive potential for unbounded growth. Such stability results from the fact that, on the average, each single female must replace herself with one and only one female during her lifetime. As Slobodkin (1.8-35) says:

"If this were not the case, the size of animal populations would be changing permanently and strikingly at a much greater rate than any existing evidence indicates."

Thus it may be concluded that, over the long-term, an established, persistent population neither grows nor declines unidirectionally, but that it oscillates around a mean value or equilibrium abundance. Whether the frequency of these oscillations is short or long depends somewhat upon the time perception of the viewer and the generation time for the population.

This conclusion can be summarized by the concept of net reproductive rate or replacement rate (R_0), which is the average number of female offspring produced by each female during her entire lifetime. For a species with discrete, nonoverlapping generations, R_0 represents the exact amount by which the population increases each generation (1.8-36). The net reproductive rate is calculated by multiplying the fraction of females surviving to each age (L_x) by the average number of female offspring produced per female at that age (M_x), and summing all the $L_x M_x$ values for the entire lifespan, i.e.:

$$R_0 = \sum_{x=0}^{\infty} L_x M_x, \quad (1.8-29)$$

where x denotes age group.

In this way the birth rate is tempered by survival to each age. If survival were total, R_0 would merely be the sum of M_x over all ages, or the total number of female offspring that would be produced by an average female in the absence of mortality, a quantity known as the gross reproductive rate (GRR). An R_0 value greater than unity implies that the population is increasing, $R_0 < 1$ implies that the population is decreasing, and $R_0 = 1$ implies that the population replaces itself precisely each generation. For this latter situation to exist as a constant rather than an average condition, a fixed schedule of survival and fertility must be maintained, and the population would remain at a constant size with no oscillations. In nature, however, oscillatory behavior is the rule rather than the exception, and such oscillations are the result of alternating values of R_0 greater than and less than unity, caused by changes in the L_x and M_x schedules. Thus $R_0 = 1$ is always an average or equilibrium condition rather than an absolute value.

Assuming that survival rates remain constant, there are two reproductive parameters which can affect the M_x schedule for temperate-zone* fish populations: one is fecundity or total ovarian egg count, and the other is age at sexual maturity, i.e., age at first reproduction. These two parameters are not entirely independent from each other since changes in age at sexual maturity have the effect of changing total population fecundity. However, within the context of this discussion, individual age- or size-specific fecundity may be considered to be independent from changes in age at maturity. Often, the most profound effects on birth rate are achieved by changing the age at first reproduction rather than changing age-specific fecundity (1.8-16).

Both fecundity and age at maturity may serve as population response parameters since both are more closely associated with size than with age (1.8-37, 18-38,

*In addition to changing fecundity and age at maturity, tropical-zone fishes may have the capability of altering M_x schedules by increasing or decreasing the number of spawnings within a year.

1.8-39) and are thereby mediated by density-dependent rates of growth (Section 1.8.2). In general, a population characterized by high growth rates, early onset of sexual maturity, and high fecundity would be considered to have less compensatory reserve, i.e., would be less resistant to cropping impact than a population of the same species with low growth rates, later onset of maturity, and relatively low fecundity (1.8-7).

In the real world, the above assumption of a constant survival (L_x) schedule is most unlikely, meaning that individual or population fecundity does not typically act in a direct fashion on the fertility or M_x schedule. Krebs (1.8-16) states that these two aspects of reproduction must be distinguished. Fertility is an actual level of population performance based on the numbers born and surviving to a given age, whereas fecundity is the potential level of performance based on numbers of eggs present. Fertility is never proportional to fecundity throughout the range of fecundity values that can be generated by a population, and this concept is elaborated in Section 1.9 on stock and recruitment relationships.

Therefore, it is emphasized here that increases in fecundity resulting from higher ovarian egg counts and/or an increased proportion of younger age groups reaching sexual maturity are meant to serve merely as an index of population response to increased cropping rates, and not as a measure of compensation for such increased rates. In other words, if individual and population fecundity remained unchanged in a population before and after a new level of cropping were instituted, then this would suggest that the increase in cropping was not a sufficient stimulus to evoke a classical population response. It is of course true that such information would be more suggestive of negligible impact if the population were characterized by low fecundity and low growth for the species. Increased fecundity alone, however, in the face of changing mortality arising from both density-dependent and density-independent sources, provides little or no information concerning fertility, which is the actual number of young surviving, or the capability of the population to offset higher rates of cropping. Such information would have to be gleaned from data used to construct a stock (fecundity) - recruitment (fertility) relationship for the population (Section 1.9).

1.8.5.1 Fecundity. Two somewhat different categories for the use of age-specific or size-specific fecundity data are: 1) to provide estimates of total spawning production for input to various models, and 2) to provide an indicator of population response to cropping or environmental changes. In order to serve as a rough

approximation to the real world, a spawning production estimate based on a single-year fecundity study may suffice. But for fecundity to serve as an indicator of population response, a multi-year study, often requiring a large expenditure of man-hours, is necessary to document the variability or the stability of population fecundity.

Although a spawning production estimate may be derived from a single-year study, the fact remains that the estimate should be developed from data on eggs actually spawned, thus accounting for egg retention and reabsorption after spawning. In a study of Connecticut River blueback herring (*Alosa aestivalis*), Loesch (1.8-40) reported that total ovarian egg counts ranged from 45,800 to 349,700, but that the range for retained eggs after spawning was 9,300 to 107,600, which represents a substantial percentage of the original estimate for unspawned ovaries. Kissel (1.8-41) reported similar results for Connecticut River alewives (*Alosa pseudoharengus*), and Holsapple and Foster (1.8-42) acknowledged the difficulty of obtaining fecundity estimates for Hudson River white perch because of egg retention. Therefore, any study of egg production prior to spawning should be followed up by a study of retention after spawning.

Because it is expensive and tedious to make actual total counts of eggs present in both ovaries, most fecundity studies rely on indirect counts from subsampling procedures, using gravimetric or volumetric methods. However, all indirect methods have some sampling error associated with them, and the magnitude of this error should be evaluated and used in assessing the degree of confidence in the fecundity estimates. One of the most common indirect methods is to count the number of eggs in an accurately weighed subsample of an ovary and then to multiply the number of eggs per subsample weight times the weight of the entire ovary (1.8-40, 1.8-41, 1.8-43, 1.8-44, 1.8-45, 1.8-46, 1.8-47). Loesch (1.8-40) found that the difference between the actual total number of eggs and the estimated total using anterior, center, and posterior sections of blueback herring ovaries ranged from -14 to +16%. Kissel (1.8-41), using only center sections, shows that the average estimate was 14 percent greater than the actual total count of eggs for alewife ovaries.

The best way to avoid such error is to rely on direct counts rather than estimates. This may be accomplished manually for species such as brook trout (1.8-48) and bullheads (1.8-49), for example, which are characterized by relatively low fecundity (<3,000 eggs). For highly fecund species the only practicable method

for obtaining direct counts of total fecundity is to utilize an automatic counting device such as that described by Boyar and Clifford (1.8-50) and Perkins and Anthony (1.8-51). This device is used at the U.S. Bureau of Commercial Fisheries Biological Laboratory, W. Boothbay Harbor, Maine, and it has the capacity to count 10,000 eggs averaging 0.9 mm in diameter in five minutes with an accuracy of 95%; larger eggs take slightly longer to count.

Because of the great variability typically observed for the number of eggs in fish of the same length, weight, and age, it is essential that the data be treated statistically in order to quantify differences in fecundity between different populations or different years, and to provide information on the precision of the fecundity estimates. As Baganel (1.8-52) advises, the first step is to plot fecundity against length or any other parameter of interest such as weight, age, and ovary weight. Fecundity is generally more closely related to length than to age or body weight (1.8-46, 1.8-53), and the relationship between fecundity and length is most often a power curve of the form:

$$F = aL^b \quad (1.8-30)$$

where F = fecundity, L = fish length and a and b are a constant and an exponent derived from the data. The curve may be linearized by a logarithmic transformation to:

$$\log F = \log a + b \log L. \quad (1.8-31)$$

In addition to log-log relationships, fecundity vs length may also be linearized by a semi-log transformation of the type:

$$\log F = \log a + bL \quad (1.8-32)$$

as used by Sheri and Power (1.8-46).

Both log-log and semi-log plots should be examined prior to deciding on the transformation to be used. As a general rule of thumb, the smaller the length range analyzed compared to the actual length range for the sexually mature members of the species, the less likely the true nature of the relationship between fecundity and length will be revealed.

Linearization through a logarithmic transformation permits standard statistical procedures to be used in analysis and acts to stabilize the variance throughout the range of the regression line, since large fish tend to be more variable in their fecundity than small ones and log transformation affects large values more than small ones. The actual value of the slope b is usually slightly greater than 3 when fecundity is plotted against length, although the observed range for several fish species is $b = 2.34$ to 5.28 (1.8-53).

1.8.5.2 Age at Sexual Maturity. Determination of the age or size at sexual maturity is relatively straightforward compared to fecundity analyses. The primary objective of a maturity study is to determine the proportion of mature fish from each age or size group. Perhaps the easiest way to determine maturity is to collect fish immediately prior to the spawning season when sexual products are the most obvious. Sexual maturity may then be determined by slight pressure on the abdomen, causing the extrusion of milt and roe from sexually mature males and females.

A more exact method, if sacrificing the fish is of no concern, is to dissect the fish and visually examine the gonads. Generalized classifications of maturity stages may be found in Kesteven (1.8-54), Mansueti (1.8-38), and Nikolsky (1.8-55), but if the fish are collected just prior to spawning there should be no problem in differentiating sexually mature from immature fish.

1.8.5.3 Reproductive Value. Power plant entrainment and impingement, in the vast majority of cases, are selective for young, sexually immature fish, whereas sport and commercial fisheries generally focus on the older, sexually mature members of a population. One way to lend perspective to the effects of cropping a given number of fish through entrainment and impingement is to compare the analogous effects of harvesting the same number of older fish via the sport and commercial fishery. Intuitively it is generally acknowledged that the elimination of a single pre-spawning adult is more important to the population than the removal of a single young-of-the-year fish, but the presentation of such an argument is often only semi-quantitative at best.

One way to quantify this idea of the relative worth or importance of individuals of a specific age within a population is to utilize the concept of reproductive value as originally defined (1.8-56) and subsequently elaborated on (1.8-57) by Sir

Ronald Fisher. Reproductive value can be described as the worth of an individual in terms of the number of offspring it is destined to contribute to the next generation (1.8-36). In other words, the elimination from the population of a given number of individuals which have relatively low reproductive value will make little difference in the expected future magnitude of the stock as contrasted with the elimination of the same number of individuals which have considerably higher reproductive values. As with life table data, it is conventional to present reproductive value in terms of female parents and female offspring, and to assume an approximate correspondence for male individuals.

Reproductive value is a function of the age, life expectancy, and fertility of the (female) individuals of the population, and the reproductive value at age X (V_X) relative to the reproductive value at birth (V_0) is estimated by:

$$\frac{V_X}{V_0} = \left(\frac{e^{IX}}{L_X} \right) \left(\sum_{Y=X}^{\infty} e^{-IY} L_Y M_Y \right) \quad (1.8-33)$$

where I is the instantaneous rate of population increase (per individual), commonly known as the Malthusian parameter. Provided that the net reproductive rate, R_0 (Eq. 1.8-29), is near unity, I can be approximated* by

$$I \approx \frac{\log_e R_0}{T} \quad (1.8-34)$$

where T is the generation time computed by:

$$T = \frac{\sum_{x=0}^{\infty} (X) (L_X) (M_X)}{\sum_{x=0}^{\infty} (L_X) (M_X)} = \frac{\sum_{x=0}^{\infty} (X) (L_X) (M_X)}{R_0} \quad (1.8-35)$$

The summation value in Eq. 1.8-33 represents the number of female offspring that will be produced by a female from age X to the end of her life, and the term Y is used to designate all the ages she will pass through from X to infinity (∞). The reproductive value for age 0 (V_0) is always equal to one, so that Eq. 1.8-33

*See Appendix 1A for the computation of a more precise estimate of I when R_0 deviates from the equilibrium condition.

provides the reproductive value of V_x as a multiple of V_0 . For example, $V_x = 4$ means that a female surviving to age X can be expected to produce four times as many female offspring as another female which has just been born (1.8-36). Wilson and Bossert (1.8-36) also point out that, in words, V_x is defined as

$$V_x = \frac{\text{the number of female offspring produced at this moment by females of age X and older}}{\text{the number of females which are age X at this moment}}$$

It can be seen that for a population at stable equilibrium, i.e., when $R_0 = 1.0$ and $I = 0$, Eq. 1.8-33 reduces to

$$V_x = \sum_{y=x}^{\infty} (L_y/L_x) M_y \quad , \quad (1.8-36)$$

where the term L_y/L_x represents the probability of living from age X to age Y, and M_y is the average reproductive success (fertility) of an individual at age Y (1.8-58). However, for a population which is changing in size, Eq. 1.8-33 must be used to calculate V_x since the definition of reproductive value in a changing population is the present value of future offspring. In an increasing population the reproductive value of very young individuals relative to older individuals is lower than in a declining population because the future breeding population is destined to be larger, and offspring to be produced later will contribute less to the total gene pool than offspring currently being born. Similarly, for a declining population, offspring to be expected at some future date are worth relatively more than current progeny because the total future population is destined to be smaller (1.8-58). This point is emphasized in Figures 1.8-13 and 1.8-14, which are plots of reproductive value vs age for Hudson River striped bass and white perch, using $R_0 = 0.5, 1.0$, and 1.5 for each population.

In a general sense it can be said that the number alive in any age category is inversely proportional to the relative reproductive value at that age, i.e., that reproductive value at age X is inversely proportional to the survivorship (L_x) to age X (1.8-59). This is true at least up to the age of breeding and may hold after breeding age in exploited fish populations where there are no reproductively senile age groups and fertility (as estimated by individual fecundity) tends to

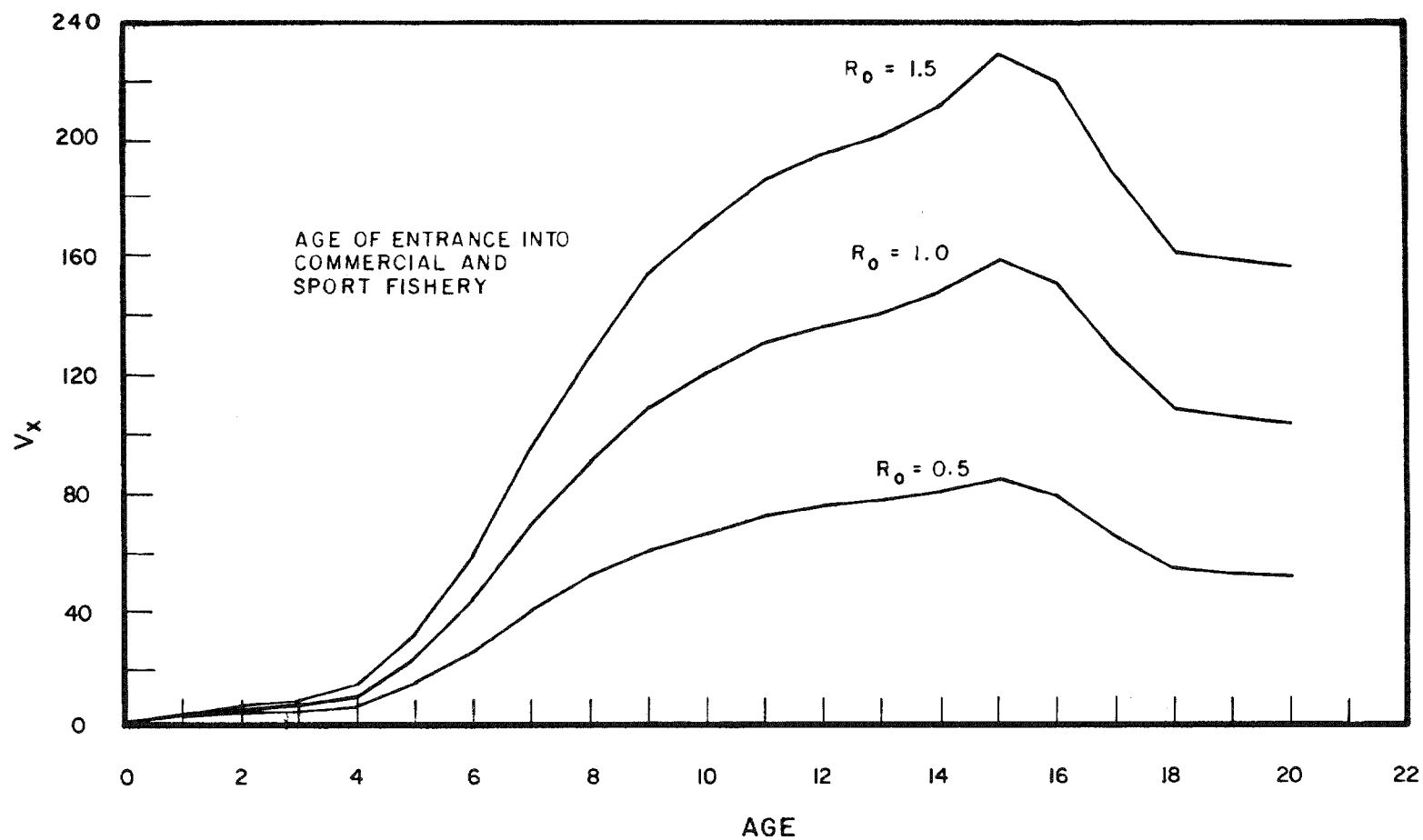


Figure 1.8-13. Reproductive value (V_x) plotted against age (X) for increasing ($R_0 = 1.5$), decreasing ($R_0 = 0.5$), and equilibrium ($R_0 = 1.0$) populations of Hudson River striped bass.

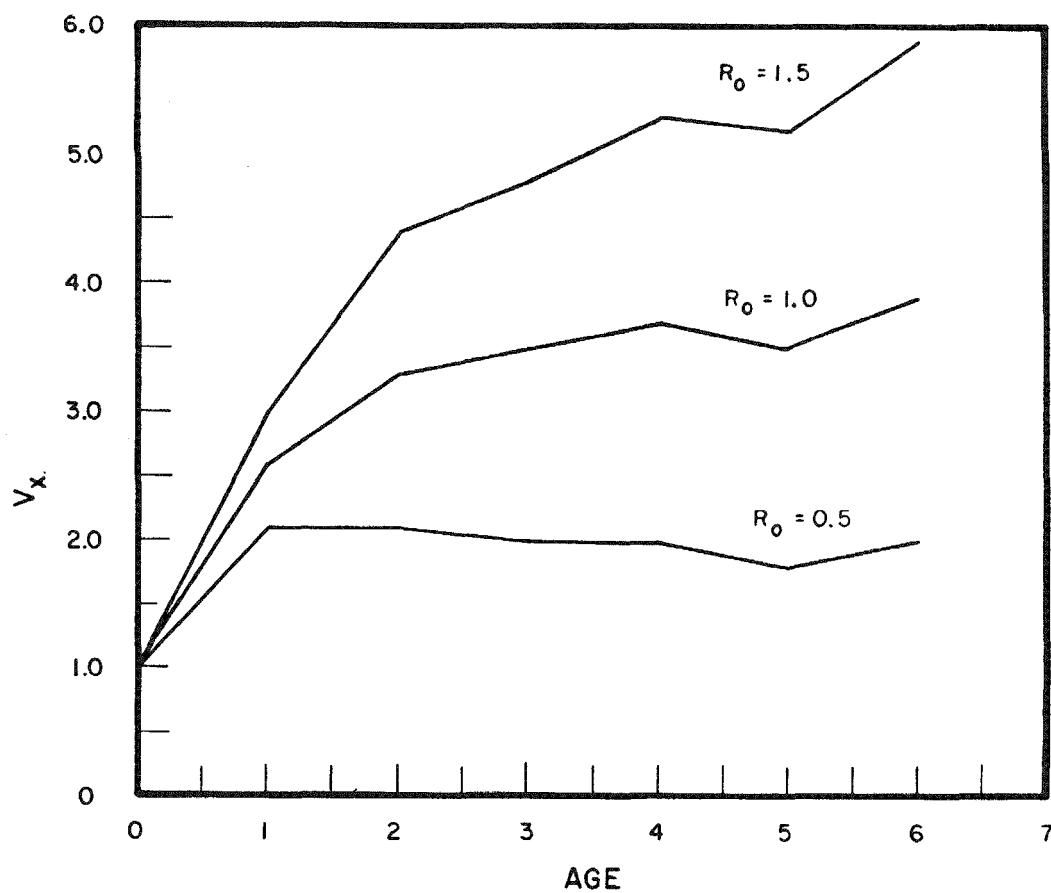


Figure 1.8-14. Reproductive value (V_x) plotted against age (X) for increasing ($R_0 = 1.5$), decreasing ($R_0 = 0.5$), and equilibrium ($R_0 = 1.0$) populations of Hudson River white perch.

increase throughout life. Slobodkin (1.8-59) goes on to point out that it is therefore possible to have a large number of individuals in the population with negligible reproductive value and that "... these evolutionarily and demographic-ally trivial animals can be fished without any real fear." He emphasizes that the stock will, of course, be reduced but that the number of fish in a stock is a very unsophisticated measure of the effective evolutionary and ecological population size. A stock comprised of a very large number of fish with very little reproductive value is in a more tenuous situation than a considerably smaller stock made up of individuals with higher reproductive value.

Admittedly, it is most difficult to obtain precise life table data for open-water fish stocks, but Slobodkin (1.8-59) makes it clear that proper management decisions can be made using the rather coarse survivorship (L_x) and fertility (M_x) data typically available in order to approximate the shape of the reproductive curve (reproductive value plotted against age). Such management decisions would be based on the conclusion reached by several workers in population dynamics, including Slobodkin (1.8-59) and Wilson and Bossert (1.8-36), that the truly skillful predator or the "prudent" predator would want to concentrate on the age groups with the lowest reproductive values, thus obtaining the largest amount of protein with the least diminution of the exploited population.

Life tables based predominantly on empirical mortality and fecundity data were constructed for Hudson River white perch and striped bass in order to approximate the reproduction curve for each species under different regimes of population growth ($R_0 = 0.5, 1.0, 1.5$). These species were chosen because sample data were available, and because together they comprise the bulk of entrainment and impingement at Hudson River power plant intakes. Entrainment and impingement are very size-selective for these species and are directed almost entirely at yearling (one-year-old) and younger fish.

The data in Tables 1.8-20 through 1.8-25 show age-specific annual mortality (Q_x), survivorship (L_x), fertility (M_x), reproductive value (V_x), gross reproductive rate (GRR), net reproductive rate (R_0), the intrinsic rate of natural increase (I), the finite rate of increase (J), generation time (T), and egg to juvenile survival (P). Estimates for these parameters, with the exception of R_0 and Q_x values which are needed as input, were calculated by the computer program LIFETAB, written in the language BASIC and documented in Appendix 1A.

Table 1.8-20

TABLE OF POPULATION STATISTICS FOR A DECLINING POPULATION OF STRIPED BASS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.30	1.0000000000	.0000	.0000	.0000	1.0
1	.20	.2000000477	.0000	.0000	.0000	4.5
2	.20	.1600000560	.0000	.0000	.0000	5.1
3	.40	.1280000508	.4459	.0571	.1712	5.7
4	.60	.0768000325	.7803	.0599	.2397	7.9
5	.53	.0307200104	2.0571	.0632	.3160	16.1
6	.53	.0144384056	5.6359	.0814	.4882	26.9
7	.53	.0067860512	13.6651	.0527	.6491	40.9
8	.53	.0031894441	19.9124	.0635	.5081	52.1
9	.53	.0014990389	26.3473	.0395	.3555	61.8
10	.53	.0007045483	29.7251	.0209	.2094	67.9
11	.53	.0003311377	33.4408	.0111	.1218	73.2
12	.53	.0001556347	35.2986	.0055	.0659	76.3
13	.53	.0000731463	35.9741	.0026	.0342	78.5
14	.53	.0000343797	36.9875	.0013	.0178	81.6
15	.53	.0000161585	43.9121	.0007	.0106	85.5
16	.53	.0000075945	45.6010	.0003	.0055	79.2
17	.70	.0000035694	47.2900	.0002	.0029	65.6
18	.90	.0000010708	48.9789	.0001	.0009	54.9
19	.95	.0000001071	50.6678	.0000	.0001	53.6
20	1.00	.0000000054	52.3567	.0000	.0000	52.4

GROSS REPRODUCTIVE RATE (GPR) = 529.077

NET REPRODUCTIVE RATE (R0) = .5

INTRINSIC RATE OF NATURAL INCREASE (I) = -.1040891

FINITE RATE OF INCREASE (J) = .9011

GENERATION TIME (T) = 6.39416

EGG TO JUVENILE SURVIVAL (F) = .0000337785

Table 1.8-21

TABLE OF POPULATION STATISTICS FOR AN EQUILIBRIUM POPULATION OF STRIPED BASS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.80	1.0000000000	.0000	.0000	.0000	1.0
1	.20	.2000000477	.0000	.0000	.0000	5.0
2	.20	.1600000560	.0000	.0000	.0000	6.2
3	.40	.1280000508	.8918	.1141	.3424	7.8
4	.60	.0768000335	1.5606	.1199	.4794	11.5
5	.53	.0307200104	4.1142	.1264	.6317	24.9
6	.53	.0144384056	11.2719	.1627	.9765	44.3
7	.53	.0067860512	27.3302	.1855	1.2982	70.3
8	.53	.0031894441	39.8249	.1273	1.0162	91.4
9	.53	.0014990389	52.6945	.0790	.7109	109.7
10	.53	.0007045483	59.4502	.0419	.4189	121.2
11	.53	.0003311377	66.8915	.0221	.2436	131.4
12	.53	.0001556347	70.5972	.0110	.1318	137.2
13	.53	.0000731483	71.9483	.0053	.0684	141.8
14	.53	.0000343797	73.9750	.0025	.0356	148.6
15	.53	.0000161585	87.8242	.0014	.0217	158.4
16	.53	.0000075945	91.2021	.0007	.0111	151.0
17	.70	.0000033694	94.5799	.0003	.0057	127.2
18	.90	.0000010708	97.9578	.0001	.0019	108.6
19	.95	.0000001071	101.3356	.0000	.0002	106.6
20	1.00	.0000000054	104.7135	.0000	.0000	104.7

GROSS REPRODUCTIVE RATE (GRR) = 1058.15

NET REPRODUCTIVE RATE (R0) = 1

INTRINSIC RATE OF NATURAL INCREASE (r) = .0000000

FINITE RATE OF INCREASE (J) = 1.0000

GENERATION TIME (T) = 6.39415

EGG TO JUVENILE SURVIVAL (F) = .0000675571

Table 1.8-22

TABLE OF POPULATION STATISTICS FOR AN INCREASING POPULATION OF STRIPED BASS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.20	1.0000000000	.0000	.0000	.0000	1.0
1	.20	.2000000477	.0000	.0000	.0000	5.3
2	.20	.1600000560	.0000	.0000	.0000	7.1
3	.40	.1280000568	1.3376	.1712	.5137	9.5
4	.60	.0768000335	2.3409	.1798	.7191	14.5
5	.53	.0397200104	6.1713	.1896	.9479	32.5
6	.53	.0144384056	16.9979	.2441	1.4647	53.7
7	.53	.0067860512	40.9953	.2782	1.9474	97.3
8	.53	.0031894441	59.7374	.1905	1.5242	127.7
9	.53	.0014990389	79.0418	.1185	1.0664	154.4
10	.53	.0007045483	89.1754	.0628	.6283	171.1
11	.53	.0003311377	100.3223	.0332	.3654	186.0
12	.53	.0001556347	105.8957	.0165	.1973	194.5
13	.53	.0000731483	107.9224	.0079	.1026	201.1
14	.53	.0000343757	110.9625	.0038	.0534	211.5
15	.53	.0000161585	131.7353	.0021	.0319	228.3
16	.53	.0000075945	136.8031	.0010	.0166	219.3
17	.70	.0000035654	141.8699	.0005	.0086	197.4
18	.90	.0000010708	146.9366	.0002	.0023	161.9
19	.95	.0000001011	152.0034	.0000	.0003	159.4
20	1.00	.0000000054	157.0702	.0000	.0000	157.1

GROSS REPRODUCTIVE RATE (GRR) = 1587.23

NET REPRODUCTIVE RATE (R0) = 1.5

INTRINSIC RATE OF NATURAL INCREASE (I) = .0649833

FINITE RATE OF INCREASE (J) = 1.0671

GENERATION TIME (T) = 6.39416

EGG TO JUVENILE SURVIVAL (F) = .0001013356

Table 1.8-23

TABLE OF POPULATION STATISTICS FOR A DECLINING POPULATION OF WHITE PERCH

AGE X	MORTALITY q(x)	SURVIVORSHIP L(x)	FERTILITY M(x)	L(x)M(x)	XL(x)M(x)	REPRODUCTIVE VALUE V(x)
0	.52	1.0000000000	.0000	.0000	.0000	1.0
1	.21	.3799999999	.0000	.0000	.0000	2.1
2	.16	.3031999855	.5567	.1671	.3342	2.1
3	.58	.1921200026	.9650	.1854	.5552	2.0
4	.72	.0806937516	1.3361	.1078	.4313	2.0
5	.90	.0225942507	1.5588	.0352	.1761	1.8
6	1.00	.0022594258	1.9670	.0044	.0267	2.0

GROSS REPRODUCTIVE RATE (GRR) = 6.38361

NET REPRODUCTIVE RATE (R0) = .5

INTRINSIC RATE OF NATURAL INCREASE (I) = -.2202992

FINITE RATE OF INCREASE (J) = .8023

GENERATION TIME (T) = 3.04893

EGG TO JUVENILE SURVIVAL (F) = .0000371140

Table 1.8-24

TABLE OF POPULATION STATISTICS FOR AN EQUILIBRIUM POPULATION OF WHITE PERCH

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.62	1.0000000000	.0000	.0000	.0000	1.0
1	.21	.3799999999	.0000	.0000	.0000	2.6
2	.36	.3001999855	1.1134	.3342	.6685	3.3
3	.58	.1921280026	1.9299	.3708	1.1124	3.5
4	.72	.0816937516	2.6722	.2156	.8625	3.7
5	.90	.0225942507	3.1176	.0704	.3522	3.5
6	1.00	.0022594258	3.9341	.0089	.0533	3.9

GROSS REPRODUCTIVE RATE (GRR) = 12.7672

NET REPRODUCTIVE RATE (R0) = 1

INTRINSIC RATE OF NATURAL INCREASE (I) = .0000000

FINITE RATE OF INCREASE (J) = 1.0000

GENERATION TIME (T) = 3.04693

EGG TO JUVENILE SURVIVAL (P) = .0000742280

Table 1.8-25

TABLE OF POPULATION STATISTICS FOR AN INCREASING POPULATION OF WHITE PERCH

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.62	1.0000000000	.0000	.0000	.0000	1.0
1	.21	.3799999999	.0000	.0000	.0000	3.0
2	.36	.3001999855	1.6701	.5014	1.0027	4.4
3	.58	.1921280026	2.8949	.5562	1.6686	4.8
4	.72	.0806937516	4.0083	.3234	1.2938	5.3
5	.93	.0225942507	4.6764	.1057	.5283	5.2
6	1.00	.0022594258	5.9011	.0133	.0800	5.9

GROSS REPRODUCTIVE RATE (GRR) = 19.1508

NET REPRODUCTIVE RATE (R₀) = 1.5

INTRINSIC RATE OF NATURAL INCREASE (I) = .1354646

FINITE RATE OF INCREASE (J) = 1.1451

GENERATION TIME (T) = 3.04893

EGG TO JUVENILE SURVIVAL (P) = .0001113419

Mortality (Q_x) estimates for striped bass up through four years old are hypothetical estimates based on data for other species, knowledge of vulnerability of striped bass to predators and impingement, and knowledge of their recruitment into the sport and commercial fisheries. Annual mortality for striped bass older than five years old is based on growth data obtained from the commercial fishery as input to the Beverton-Holt formulation described in Section 1.8.3 (Eq. 1.8-5). Mortality estimates for white perch are derived from catch curve analysis (Section 1.8.3.1). Once a mortality schedule has been decided upon, survivorship, i.e., a schedule of the probability of surviving to a given age, may be calculated by:

$$L_{x+1} = L_x - Q_x L_x$$

where $L_0 = 1.000$, since all of the young are alive at time $X = 0$ set by the investigator to demarcate the beginning of life.

Age-specific fecundity estimates are based on actual egg counts for Hudson River fish, but fertility (M_x) values require additional calculations using the equation:

$$M_x = PA_x (F_x/2)$$

where:

A_x = proportion of females sexually mature at age X

$F_x/2$ = age-specific fecundity adjusted to reflect female offspring only (assumes 1:1 sex ratio for eggs)

$$P = \frac{\sum_{x=0}^{\infty} L_x M_x}{\sum_{x=0}^{\infty} L_x A_x (F_x/2)} = \frac{R_0}{\sum_{x=0}^{\infty} L_x A_x (F_x/2)}$$

The quantity P is a survival term reflecting survival from a spawned egg to an age 0 female juvenile. Age 0 fish are defined as young-of-the-year fish when they are fully recruited to the 1/4 inch square mesh seines and trawls used in sampling (three to four months after spawning). Note that for a given survival (L_x) schedule, maturation (A_x) schedule, and egg complement (F_x), the egg to juvenile survival term (P) is dictated by the value chosen for the net reproductive rate (R_0).

Using appropriate data for the number of age groups in the population, the mortality schedule, and the maturation and fecundity schedules, program LIFETAB calculates age-specific reproductive values for given values of R_0 . The reproductive values contained in Tables 1.8-20 through 1.8-25 for Hudson River striped bass and white perch are plotted in Figures 1.8-13 and 1.8-14. It is generally evident that the youngest fish have the lowest reproductive value and that losses of fish from this end of the age spectrum would mean much less to the success of future generations than numerically equivalent losses from the older age groups. For example, the removal of 1,000 female yearling striped bass from an equilibrium population ($R_0 = 1.0$) means that there will be approximately 5,000 fewer age 0 female offspring produced in the next generation. On the other hand, the removal of 1,000 seven-year-old females, for example, means the potential loss of approximately 70,300 age 0 females which would have been born into the next generation. Another way to view the situation is to realize, for an equilibrium population, that the loss of 500,000 female yearlings through impingement (equivalent to impingement of 1,000,000 male and female yearlings), i.e., the reproductive loss of 2,500,000 offspring to the next generation, would be equivalent to the loss from commercial and sport fishing of approximately 47,508 male and female striped bass evenly distributed between all age groups from ages 4 through age 20.*

For a given population, the exact relationship between the reproductive value for younger vs older fish depends on whether the population is increasing, stable, or decreasing, and it can be seen from Figures 1.8-13 and 1.8-14 that younger fish increase in reproductive importance (relative to older fish) in a declining population. Therefore, it should be expected that the impact from a given level of entrainment and impingement (assuming selectivity for young fish) would be greater for a population with $R_0 < 1.0$ than for the same population in an equilibrium or ascendant phase. Although the numbers given above are only approximate and only as good as the L_x and M_x values used in the calculations, this approach, as Slobodkin (1.8-59) implicitly points out, is certainly useful in lending perspective to the relative impact from entrainment/impingement cropping vs sport/commercial fishing.

$$\frac{*500,000(V_1)}{\sum_{x=4}^{20} V_x} = \frac{2,500,000}{1789.2} = 1397.3 \text{ females per age group 4 through 20}$$

1.9 STOCK-RECRUITMENT RELATIONSHIPS

1.9.1 Introduction

Stock-recruitment relationships are a class of population models which represent the number of progeny that attain spawning age (recruits), as a function of the size of the spawning stock which produced them (spawners). Such models have proven extremely useful in the field of fisheries management, and have similar potential for application to impact assessment. To see how these models may be helpful in analyzing the potential impacts of power plant cropping, it is useful to discuss the biological factors considered in their development.

Spawners exert a number of influences, both direct and indirect, on their young. The size, age distribution, and weight composition of the spawning stock determine the number of eggs spawned in a given year. In some stocks, egg production does not increase directly with an increase in these factors, because under crowded conditions females may retain and reabsorb a greater proportion of eggs than when the stock is less dense. A shortage of suitable spawning grounds may also decrease spawning rate and/or viability of the spawn in the less preferred habitats. Egg and larval survival is then dependent on the levels of competition, predation, and cannibalism present in the system, as well as the influence of prevailing environmental conditions. Spawners may be thought of as "setting" the level of competition through their role in determining initial abundance. In addition, the rate of cannibalism may be dependent on the abundance of both spawners and their young. Throughout life the progeny will be subject to a host of additional factors which will influence their abundance. However, it is generally thought that as they mature, environmental (i.e., density-independent) factors will tend to gain importance, as the influence of density-dependent factors diminish.

It is apparent that the relationship between spawning stock size and subsequent recruitment is due to the interplay of a variety of factors. The general form of the relationship will depend on the relative strengths of various density-dependent influences, while the accuracy of predictions obtained using such a relationship will in part depend on the relative importance of density-dependent and density-independent factors. Thus, when recruitment is compared with spawning stock abundance, one sees the "result" of the interplay of many factors without being able to decipher which specific mechanisms are responsible for the pattern observed. Although this may appear to be a disadvantage, it is in fact, the real

value of stock-recruitment curves. That is, they are tools through which population behavior may be examined, despite the fact that the population regulating mechanisms responsible for them are often exceedingly difficult to uncover. Moreover, the general form of the relationship may indicate the predominant type of density-dependent processes operating. For example, a curve which predicts asymptotically increasing recruitment concurrent with increasing stock abundance may indicate that mortality rate continuously decreases as progeny abundance decreases through time. Alternatively, a curve which depicts declining recruitment at high levels of stock abundance is consistent with the notion that mortality rate during the early life stages is fixed by the size of the spawning stock at reproduction. Although these inferences are of a general nature, they may aid in directing research toward study of specific mechanisms which would result in the implied form of density-dependent mortality.

As mentioned above, stock-recruitment relationships are valuable in quantifying the level of density-dependence operative in a population, without requiring that each mechanism contributing to the relationship be known or quantified. McFadden et al. (1991) provide an exhaustive and up-to-date review of the literature surrounding the question of the role of dependent processes in population regulation. Their review finds a great deal of evidence supporting the notion that such processes are important in the regulation of any persistent population and of fish populations in particular. It can readily be seen (as the authors also point out) that a model which does not incorporate some notion of density-dependence cannot adequately represent population dynamics. Such models will invariably predict that a stock initially at equilibrium will inexorably decline to extinction if any additional source of mortality, no matter how small, is placed upon it. Clearly this does not conform with observations of the maintenance of numerous stocks under moderate to high levels of exploitation.

Although density-dependent processes are known to be important in population regulation, it is an extreme rarity to find quantitative information on the specific mechanisms involved. This is, in part, due to the fact that many factors contribute to the overall response observed, and any individual factors may be difficult to detect statistically unless a long-term series of data is available. An additional complication is the fact that the mechanisms involved may vary in importance from year to year in response to changes in environmental or other factors, and some mechanisms may only play a role during a very limited portion of the life cycle or over a limited range of stock abundances. Thus the lack of an adequate

data base places a large constraint on the modeler who wishes to incorporate density-dependent processes in a complicated population model. In contrast, the stock-recruitment approach may be workable with only a record of population abundance over a period of years. The availability of this type of data is greater than that of the kind mentioned previously.

Although stock-recruitment models have many advantages, there are several limitations that must be recognized before applying them to a particular population. Probably the major disadvantage is that a long time series of data is necessary for their construction. Furthermore, in multiple age spawning stocks, it is highly desirable to have a record of age composition. Even where a relatively long time series of data is available it may still be extremely difficult to construct a stock-recruitment curve. One reason is that the basic models in use differ most from one another at the extreme high and low levels of spawner abundance. Often the available data will cover only a limited range of stock abundance, which makes it extremely difficult to determine which model is most appropriate.

Environmental factors also play a role in obscuring an underlying stock-recruitment relationship. The relationship between spawners and their young is known to be moderated by environmental cues. Thus, a particular stock-recruitment relationship may be appropriate under a limited range of environmental conditions. If a persistent and directional change in environmental influences has occurred during the time the population has been observed it is necessary to have appropriate environmental information to develop a relationship appropriate for current conditions. Even if environmental forces have not acted in a unidirectional mode, they may increase the variability of stock-recruitment data to such a degree that the collection of data points may appear to be a random scatter exhibiting no discernible pattern.

An additional note of caution is that stocks for which appropriate data are available have often been subject to commercial or sports fishery exploitation. The rate of exploitation also exerts an influence on the spawner-recruit relationship and thus, some measure of the levels of exploitation may be desirable. If, however, the exploitation rate has been, and will continue to be, relatively constant through time it is generally not necessary to estimate its magnitude.

Despite the many limitations associated with stock-recruitment relationships, they can still be useful if care is taken in their development and interpretation. Many techniques have been developed which attempt to account for some of the difficulties described above, and further research in this area appears promising. The next section provides guidance for choosing an appropriate spawner-recruit model, and describes some of the basic models in use.

1.9.2 Choosing An Appropriate Stock-Recruitment Relationship

Due to factors discussed above, when plots of stock vs recruitment are made a large degree of scatter is often observed. A spectrum of curve fitting possibilities may be applied, and it is the investigator's challenge to choose an appropriate curve from the practically infinite variety available to him. When Ricker (1.9-2) first proposed the theory of stock and recruitment, he described some of the desirable characteristics a stock-recruitment curve should possess. An obvious, yet important requirement is that the curve must pass through the origin, since recruits cannot be generated in the absence of spawners. In addition, the origin should be the only point at which the curve meets the abscissa. That is, there is no level of parent stock for which successful reproduction is completely eliminated. For a stock to be persistent, recruitment must equal or exceed parental stock over a significant portion of the range of stock abundances.

A linear relationship can immediately be rejected based on the argument that a density-independent model cannot accurately reflect population dynamics. Beverton and Holt (1.9-3) and Lawler and Englert (1.9-4) provide additional support for rejecting the spectrum of linear relationships. Unfortunately, elimination of linear relationships does not narrow the field of possibilities by much.

One approach that has often been taken is to adopt one of the classical stock-recruitment relationships as a working approximation of fish population behavior. The two basic models are the Ricker stock-recruitment relationship, and the Beverton-Holt stock-recruitment relationship. Both were developed in the 1950's, the first by Ricker in Canada (1.9-2); the other by Beverton and Holt in England (1.9-3).

These relationships are illustrated in Figure 1.9-1. In each case it is presumed that spawners and recruits are measured on the same units. Thus the line

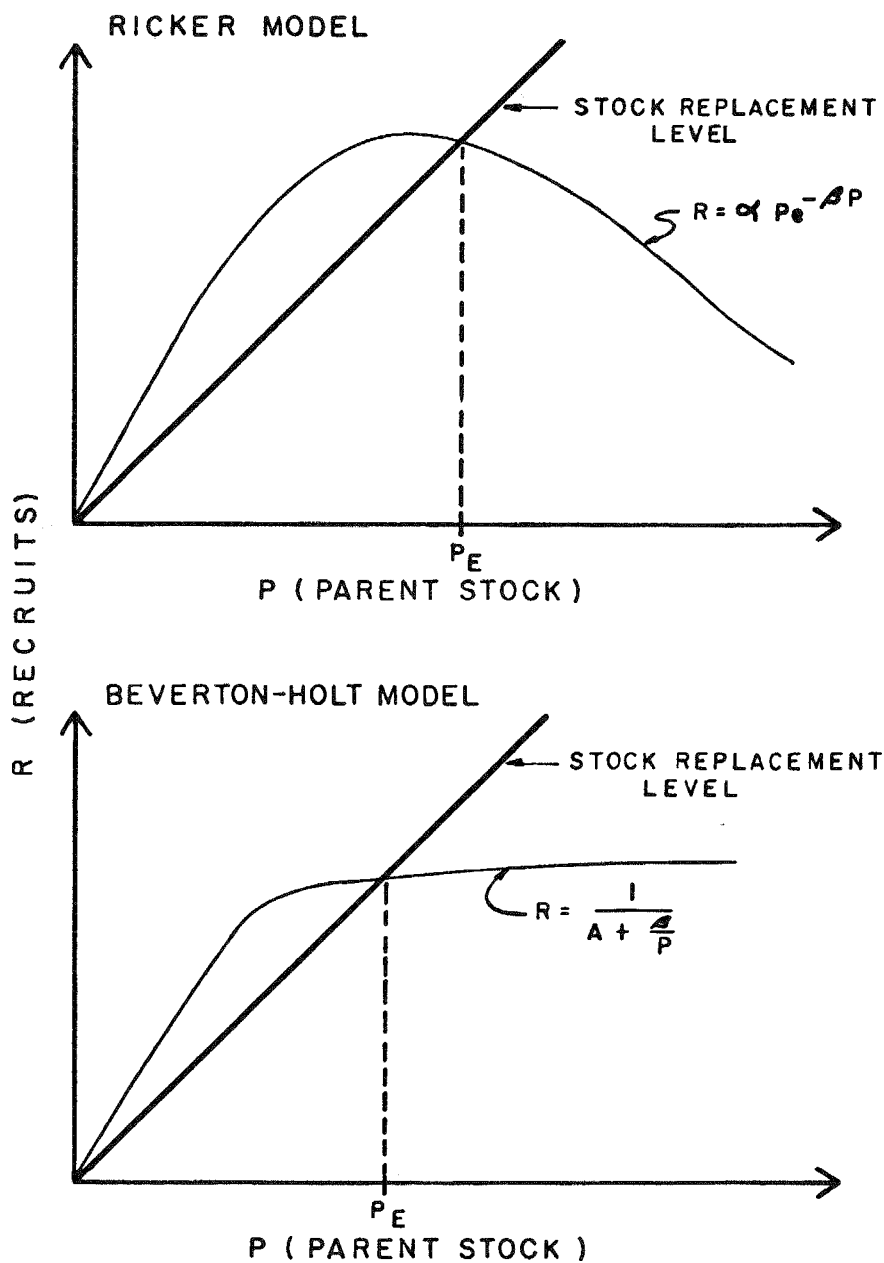


Figure 1.9-1. Beverton-Holt and Ricker Stock-Recruitment Relationships

which intercepts the origin at a 45 degree angle represents the set of points for which recruitment would exactly balance spawning stock (either in abundance, egg production, or biomass), and thus is termed the replacement line. The point at which the curves intersect the replacement line is termed the equilibrium, or replacement stock. The major difference between the two models is that the Ricker curve predicts declining recruitment at high levels of spawner abundance, whereas the Beverton-Holt curve depicts recruitment as increasing asymptotically toward some maximum level. This difference is generated by a difference in the defining differential equations appropriate to each. The Ricker model assumes recruit survival to be set by the initial level of parent stock, as for example, when cannibalism is an important factor governing young-of-the-year mortality. In contrast, the Beverton-Holt model presumes that recruit survival is continuously density-dependent up to some critical age.

Each of these models has been applied to the study of numerous fish stocks. Their widespread use appears to be due to the fact that they are consistent, in a general way, with observations of numerous stocks, and that they are relatively easy to apply. Another major reason that models such as those developed by Ricker and by Beverton and Holt have been selected, is that they have some degree of theoretical foundation, as opposed to a simple curve fit that not only does not meet many population characteristics, but also has no underlying explanations.

Despite the fact that such models are known to be good general descriptors of population dynamics, it is also recognized that particular populations may deviate (perhaps dramatically) from one of these two simple relationships. Not all density-dependent processes behave in a compensatory manner. Stocks for which density-dependent mortality influences are important would be expected to have reproduction curves dissimilar to the classic forms. Peterman (1.9-5) proposed that the recruitment curve depicted in Figure 1.9-2 be used to describe the dynamics of the pink salmon (Onchorhynchus gorbuscha) in the Atnarko River in British Columbia. Such a model had previously been discussed on theoretical grounds by Ricker (1.9-2), Paulik (1.9-6), and Holling (1.9-7). The model depicts a population which has both an upper and lower domain of stability. That is, if the population were to be depressed to a low level of abundance it would tend to persist at the lower level and similarly would tend to maintain a high level abundance when initially present at that level. Peterman proposes that the recruitment curve for Pacific salmon may be generated by the presence of an important predator which exhibits a Holling Type III, or S-shaped functional response (1.9-8). This type of feeding behavior is

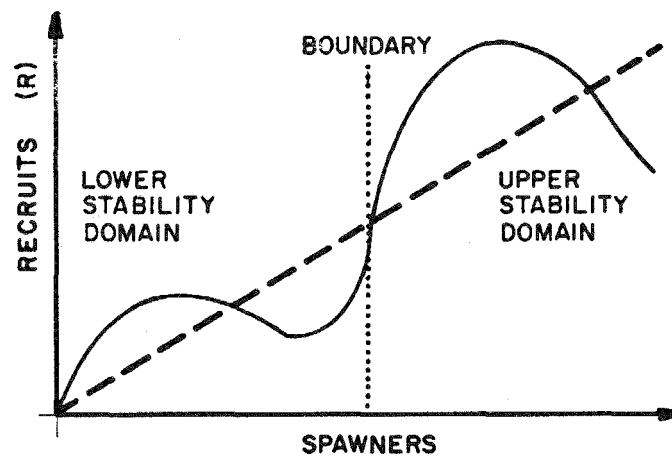


Figure 1.9-2. A recruitment curve showing the effect of depensatory mortality.

characteristic of predators that "learn" and is illustrated in Figure 1.9-3. When prey abundance is extremely low, the predators may find alternate prey more attractive, and exert little influence on the prey population in question. However, as prey numbers increase, predators may "switch on" to what has now become a readily available food source, and the number of prey taken per predator increases quickly. Finally, predators are subject to handling time and other constraints so that individual feeding rates tend towards an asymptotic level as prey abundance increases further. Murdock and Oaten (1.9-9) show how an S-shaped predation function gives rise to the type of "depensatory recruitment curve" illustrated in Figure 1.9-3.

Other types of recruitment curves may similarly be generated by alternative types of density-dependent phenomena. In multiple-aged spawning stocks it may be difficult to relate a single index of spawning stock to the survival success of their offspring. Instead, it may be necessary to consider the influences of spawner biomass, age composition, abundance, and egg production as separate factors. Furthermore, the success of a group of recruits may also depend on the age composition and abundance of the entire stock (i.e., not limited to the influence of spawning adults). For example, Ricker and Smith (1.9-10) found evidence for the existence of interaction between neighboring year classes of Skeena River sockeye salmon (Oncorhynchus nerka), having an impact on recruit survival.

Thus to construct a stock-recruitment curve for a particular population, it is necessary to exercise extreme caution and to utilize all pertinent information known about the stock. The primary source of information will often be the stock-recruitment data base itself. The data must be examined to determine whether they deviate greatly from the classical models. Life history information on the stock should also be compiled and analyzed to uncover the type of density-dependent factors that may be important in population control. Comparative studies of related species will often provide much insight into potential population control mechanisms and should not be overlooked.

When all pertinent material has been thoroughly researched a determination of whether such information is consistent with one or the other of the basic models is made. If a reasonable degree of consistency is observed it would then seem appropriate to adopt the model as a working approximation and to estimate its parameters using the available data. If, on the other hand, existing information conflicts with the basic models, they clearly should not be used. If enough

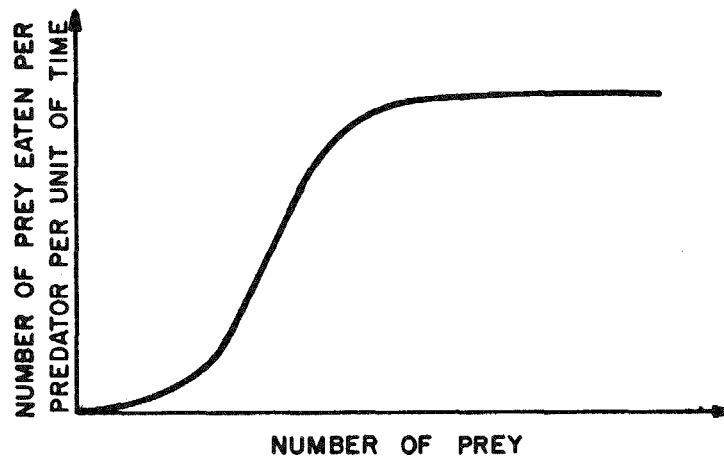


Figure 1.9-3. A Type III Predator Functional Response Curve

is known about the important factors governing the stock, it may be possible to construct an alternate model on theoretical grounds. Where no explanation can be found for the pattern observed, it is best to use an empirically based curve, or perhaps to try an entirely different approach if such a curve appears to have little predictive capacity.

In many circumstances one of the basic models has been, and will be found to be, consistent with population information. For this reason, the use of these models and their modifications is discussed in considerable detail below.

1.9.3 The Ricker Model

The Ricker stock-recruitment model was initially developed for the study of single-aged spawning stocks. Extensions to a multiple-aged spawning stock have since been developed, and these are discussed in a section below. The original model proposed that the overall mortality rate of recruits is composed of both density-independent, and density-dependent mortality which is reflected to some degree by the level of stock abundance.

The defining differential equation for the Ricker model is:

$$\frac{dR(t)}{dt} = - [k_1(t) + k_o(t) P_o R(t)] \quad (1.9-1)$$

where

$R(t)$ = the number of (potential) recruits at time t

P_o = initial spawner abundance

$k_1(t)$ = the instantaneous density-independent mortality rate at time t

and

$k_o(t)$ = the stock-dependent instantaneous mortality rate at time t

This is integrated to yield:

$$\int_{P_o}^{R(T)} \frac{dR(t)}{R(t)} = \int_0^T (k_1(t) + k_o(t) P_o) dt \quad (1.9-2)$$

$$R(T) = P_o E e^{-\bar{k}_1 T} \cdot e^{-\bar{k}_o t_c} P_o \quad (1.9-3)$$

$$R = R(t) = \alpha P e^{-\beta P} ; \alpha = E e^{-\bar{k}_1 T} \quad \text{and} \quad \beta = k_o t_c \quad (1.9-4)$$

Where T represents the age at which spawning is attained, t_c is the age beyond which density-dependent factors no longer exert any influence and E represents the average egg production per spawner. The defining differential equation above presents both stock-dependent and density-independent mortality rates as functions of time. This is done in recognition of the fact that such rates will vary among life history stages and from year to year. This also represents a modification of the original Ricker equation in which these rates are presumed to be constants. The time-dependent nature of both density-dependent and density-independent variables explains the scatter of the data about the curve generated from the standard, two parameter Ricker curve. However, the heuristic difference in the defining equation will usually not be reflected in the stock-recruitment curve obtained. This is because the predominant method of obtaining a Ricker curve is to solve for the parameters and through application of regression techniques. Thus, regardless of the time-dependent nature assumed in the defining differential equation, the values of and obtained will be the same, and will reflect time averaged mortality rates.

The easiest way to solve for the parameters and is to rearrange Eq. 1.9-4 to the following form:

$$\log_e (R/P) = \log_e \alpha - \beta P \quad (1.9-5)$$

and perform linear regression of $\log_e (R/P)$ vs P . This method was originally proposed by Rounsefell (1.9-11). As Ricker (1.9-12) states "the expected" or "most frequent" value of $[R/P]$, as calculated from the logarithmic relationship for a particular P , is the geometric, rather than arithmetic mean of actual observed R/P values at that level of P . The geometric mean (GM) is always less than the corresponding arithmetic mean (AM). The relationship between the two can be represented by an expression modified from formula (8) of Jones (1.9-13). The formula is:

$$\log_{10} (AM/GM) = 0.2172 S^2 (N-1)/N \quad (1.9-6)$$

where S is the standard deviation from the regression line of the normally-distributed natural logarithm of the variates, and N is the number of items in the series.

The slope of the regression line (Eq. 1.9-5) yields an estimate of β , and $\log_e \alpha$ is estimated by the ordinate intercept. This geometric mean line can then be converted to an AM line using Eq. 1.9-6. The widespread availability of computers makes use of non-linear, iterative techniques feasible for estimation of model parameters, and these techniques are often preferred over the standard least squares method for the linearized version of the curve. Another variation in the curve fitting process is to use a weighting technique which weights data points which lie at the extremes of spawner abundance more heavily than intermediate points. The rationale for such a process is that often most data points will be clustered around replacement level and thus may be thought of as possessing less information as to the actual shape of the curve than the relatively rare data points which lie at the extremes. The appropriateness of various regression techniques depends in part on error structure, and professional statistical advice should be sought in choosing the technique to be employed.

Before considering how the Ricker equation may be used to evaluate power plant impacts, it is important to discuss the meanings of the α and β values obtained. Alpha (α) is the slope of the reproduction curve at zero stock abundance. It is composed of the product of the average parental egg production and survival rate when only density-independent processes are considered. That is, represents potential rate of increase (intrinsic rate less density-independent mortality) with no stock biomass inhibition.

Beta (β) is the unit rate inhibition of stock on recruitment or the density-dependent population mortality rate. Beta comprises all mechanisms of density-dependent mortality, direct and indirect, and intrinsic and extrinsic to the population (1.9-2).

By definition, recruitment exactly balances spawning stock abundance at equilibrium. Replacing R and P by P_E (equilibrium stock size) in Eq. 1.9-4 yields:

$$P_E = \frac{\log_e \alpha}{\beta} \quad (1.9-7)$$

Power plant cropping is generally believed to be a density-independent source of mortality. As such, it will lead to a fractional reduction in the level of α . The relationship is:

$$\alpha' = \alpha(1-m) \quad (1.9-8)$$

where α' replaces α as the appropriate parameter when the plant is in operation and m is the conditional natural mortality rate discussed in another section.

It is more difficult to determine the effect of power station mortality on the level of β . Beta is composed of the product of the "average" density-dependent mortality rate per spawner and the time interval during which density-dependent processes are operative. One might presume that if the recruit population was thinned without a concurrent reduction in spawner abundance (as is the case with power plant cropping) that competition among recruits would be reduced, promoting faster growth. Since mortality due to predation and cannibalism is generally believed to be a decreasing function of both organism size and abundance, β would probably be reduced under these circumstances. A reduction in β leads to an increase in recruitment for a given level of parental stock and fixed value of β . Thus, the assumption that β is unchanged under the influence of power plant cropping appears to be conservative.

One way to evaluate power plant impact is to consider the fractional reduction in equilibrium population abundance due to plant operation. Under the assumption of a constant pre- and post-operative β , this equilibrium reduction is given by:

$$\frac{P_E - P'_E}{P_E} = \frac{\frac{\log_e \alpha}{\beta} - \frac{\log_e \alpha(1-m)}{\beta}}{\frac{\log_e \alpha}{\beta}} = 1 - \frac{\log_e \alpha(1-m)}{\log_e \alpha} \quad (1.9-9)$$

It can easily be seen that the level of reduction expected for a given level of m values inversely with α . Thus, stocks with a greater value of α will be able to withstand greater levels of exploitation than those for which α is small. It is for this reason that α has sometimes been termed a measure of "compensatory reserve."

Although the stock-recruitment relationship has been described as one between spawner and recruit abundance, it may in some instances be more appropriate

to use some other stock index, such as biomass or egg production. For example, if competition among recruits is the primary density-dependent mechanism, spawner egg production would probably be a better index to use.

Some alternate procedures may be useful in estimating the parameters of the reproduction curve. For example, if recruits pass through a series of discrete life history stages with which particular instantaneous density-independent mortality rates are associated, α will take the following form:

$$\alpha = E \prod_{i=1}^n e^{-k_{1i}(t_{i+1} - t_i)} \quad (1.9-10)$$

where

k_{1i} = the instantaneous density-independent mortality rate during the i th life history stage

and

n = the number of life history stages prior to attaining spawning

If information is available on the magnitude of the various k_{1i} 's it is possible to estimate α directly from Eq. 1.9-10. The main disadvantages to this technique are that such information is difficult to obtain and the confidence intervals about such estimates are often large. Multiplication of a number of such estimates will compound each of the individual errors. If α is obtained in this manner, then given an estimate of P_E , β can be estimated from Eq. 1.9-7, and the stock-recruitment curve is completely determined.

McFadden et al. (1.9-1) have proposed a method of deriving β prior to estimating α . Their argument is the following. Assume that R_1 and R_2 can be measured and are the recruits from P_1 and P_2 respectively. In the Ricker model, their ratios are:

$$\frac{R_1}{R_2} = \frac{\alpha P_1 e^{-\beta P_1}}{\alpha P_2 e^{-\beta P_2}} \quad (1.9-11)$$

Equation 1.9-11 is equivalent to the following expression:

$$\beta = \frac{\log_e R_2 + \log_e P_1 - \log_e R_1 - \log_e P_2}{P_1 - P_2} \quad (1.9-12)$$

If three R and P pairs are available, three estimates of β are possible and their average, $\bar{\beta}$, is an estimate of the unit rate inhibition of stock and recruitment. Eq. 1.9-4 can be rearranged as:

$$\alpha = \frac{R}{P e^{-\beta P}} \quad (1.9-13)$$

The estimate of β obtained can now be substituted into Eq. 1.9-13 along with each individual R and P pair, to obtain an α for that specific generation. The authors propose that this approach has an advantage over the traditional regression techniques for stocks near equilibrium and for which little information is available at low levels of stock abundance (i.e. the level at which compensatory phenomena may be operating most strongly).

1.9.4 Extension of the Ricker Model to a Multiple-Aged Spawning Stock

An extension of the basic Ricker model to a multiple-aged spawning stock has been developed. The multiple-aged spawning stock model assumes that no qualitatively different population phenomena occur. That is, no inter-year class interactions are presumed to occur other than the primary relationship between the spawning adult stock and their progeny. The multiple-age spawner model generally compares the egg production of a group of spawning adults with the subsequent egg production of the recruits throughout. It is given by the equation:

$$REP = \alpha PEP e^{-\beta P} \quad (1.9-14)$$

where

REP denotes lifetime egg production of recruits and

PEP denotes the egg production of the associated group of spawners

The derivation of the multiple-aged spawner model is based on the combination of two equations. The first is:

$$R(t_i) = PEP \prod_{j=1}^i e^{-k_{1j}(t_{j+1} - t_j)} e^{-\beta P_o t_c} \quad (1.9-15)$$

which is equivalent to the stock-recruitment relationship for a single-aged spawning stock. The second equation relates age composition, reproductive capacity, and abundance to the lifetime egg production of the recruits. That is:

$$REP = \sum_{i=Y_1}^{Y_n} R(t_i) E_{fi} f_{mi} f_{fi} \quad (1.9-16)$$

where

f_{fi} = the fraction of age i individuals that are female
 f_{mi} = the fraction of i year old females that are mature
 E_{fi} = the eggs produced/mature female at age i

and

$Y_1 \rightarrow Y_n$ are the n ages which make some contribution to the spawn.

Eq. 1.9-14 is obtained by substituting the right hand side of Eq. 1.9-15 for $R(t_i)$ in Eq. 1.9-16, and rearranging terms. The stock-recruitment curve that is obtained is nearly identical in form to the single-age spawner-recruitment model and the same techniques discussed above for fitting data to the curve may also be used in the present case. The major difference in fitting the models is the nature of the data required. For the multiple-aged spawner model it is highly desirable to have direct information on the age composition of the stock for all years observed. If such data are not available it may still be possible to apply the approach, but the results obtained will probably only roughly approximate the real life situation. One case in which the approach may retain some degree of realism is when either the commercial catch or spawning stock tends to be dominated by a single age group, and thus catch statistics would convey some information about age structure through comparisons of several years' worth of data.

1.9.5 The Beverton-Holt Model

The Beverton-Holt model, as discussed previously, has also been widely used in fisheries management. Since many of the comments that were made regarding the

Ricker curve can be applied equally to the Beverton-Holt curve, only the defining differential equation and resulting relationship will be discussed.

The rate of change of recruit abundance through time is given by:

$$\frac{dR}{dt} = - [k_1 + k_o R]R; R|_{t=0} = P_o \bar{E} \quad (1.9-17)$$

All terms in the equation have the same meaning as those encountered in the defining differential for the Ricker curve, with the exception that k_o in this instance is the density-dependent mortality rate per recruit (rather than per spawner). Solution of Eq. 1.9-17 yields:

$$R(t) = \left[\frac{k_1/k_o}{\left\{ \left[1 + \frac{k_1/k_o}{P_o \bar{E}} \right] e^{k_1 t} \right\} - 1} \right] \quad (1.9-18)$$

The more familiar form of Eq. 1.9-18 is:

$$R = \frac{1}{\alpha + \frac{\beta}{P}} \quad (1.9-19)$$

The curve may be written in a linear form as either

$$\frac{1}{R} = \alpha + \frac{\beta}{P} \quad (1.9-20)$$

or

$$\frac{P}{R} = \beta + \alpha P \quad (1.9-21)$$

In either case regression analysis would estimate $1/R$ rather than R , so that the R line is obtained by inverting computed values. Values of $1/R$ represent the harmonic mean of expected recruitment at each P . Ricker (1.9-12) points out that this line tends to be considerably below an arithmetic mean line fitted to observed R , and recommends the use of nonlinear iterative techniques.

1.9.6 Other Modifications of the Basic Stock-Recruitment Relationships

Possible modifications of the stock-recruitment approach are practically infinite and can only be discussed in a general manner. The prevalent types of modifications include changing or adding detail to the defining differential equations, attempting to account for the effects of environmental factors explicitly, and stochastic treatment of model parameters.

Along the lines of modifying the basic equations, Paulik (1.9-5) has proposed that a model of the form:

$$R = \alpha P e^{(-\beta P^\delta)}$$

provides a considerable amount of flexibility without unduly overextending the available data base. Some have suggested combining aspects of both Ricker and Beverton-Holt curves as an alternate modification. McFadden et al. (1.9-1) have proposed the inclusion of an additional density-dependent mortality term to represent cannibalism.

Attempts to account for environmental factors generally begin with a presumed stock-recruitment relationship and attempt to account for deviations from the curve by regressing important environmental cues against the residuals, or a similar procedure. Using such a combined recruit-environment model, Nelson et al. (1.9-14) were able to account for over 84% of the variation on their survival index of Atlantic menhaden (*Brevoortia tyrannus*). Lawler and Englert (1.9-4) consider the effect of a temperature increase due to thermal discharge on the long-term population reduction due to plant cropping. McFadden et al. (1.9-1) incorporate a flow index as a modification of a Ricker curve.

Utilization of stochastically varying parameters makes it possible to form a probability distribution of impact, rather than merely obtaining a single impact estimate based on a deterministic approach. Lawler and Englert (1.9-4) describe a procedure in which a probability distribution of impact is generated. Under the assumption that the distribution of the linear fit (to the Ricker stock-recruitment equation), various statistics can be extracted, including the correlation between α and β . A random number generator is used in connection with

these statistics to generate random pairs of α and β . As they state: "The proposition is that these random pairs of α and β , when arranged on probabilistic fashion, give the probability distribution of the response of the system in the future." This assumes that the future environment will be generally definable by the overall historical statistics. Then a probability distribution of the impact parameter, that is, the reduction in the equilibrium population, can be developed by computing the model output for each set of α and β .

Figure 1.9-4 indicates results that can be obtained using this statistical approach. In this illustration, given a conditional mortality rate of 5% there is a 10% probability of the impact being no greater than 3%, a 50% probability of impact being no greater than 4% and a 90% chance of its being no greater than about 5%. This means that 90% of all possible time series of population data that retain the same overall statistics of past behavior will result in lowering the equilibrium population by no more than 5.3%. Information presented in this fashion can improve the decision-makers' sense of the significance of various levels of estimated impact. These and related approaches have the potential to make a substantial impact on the decision-making process.

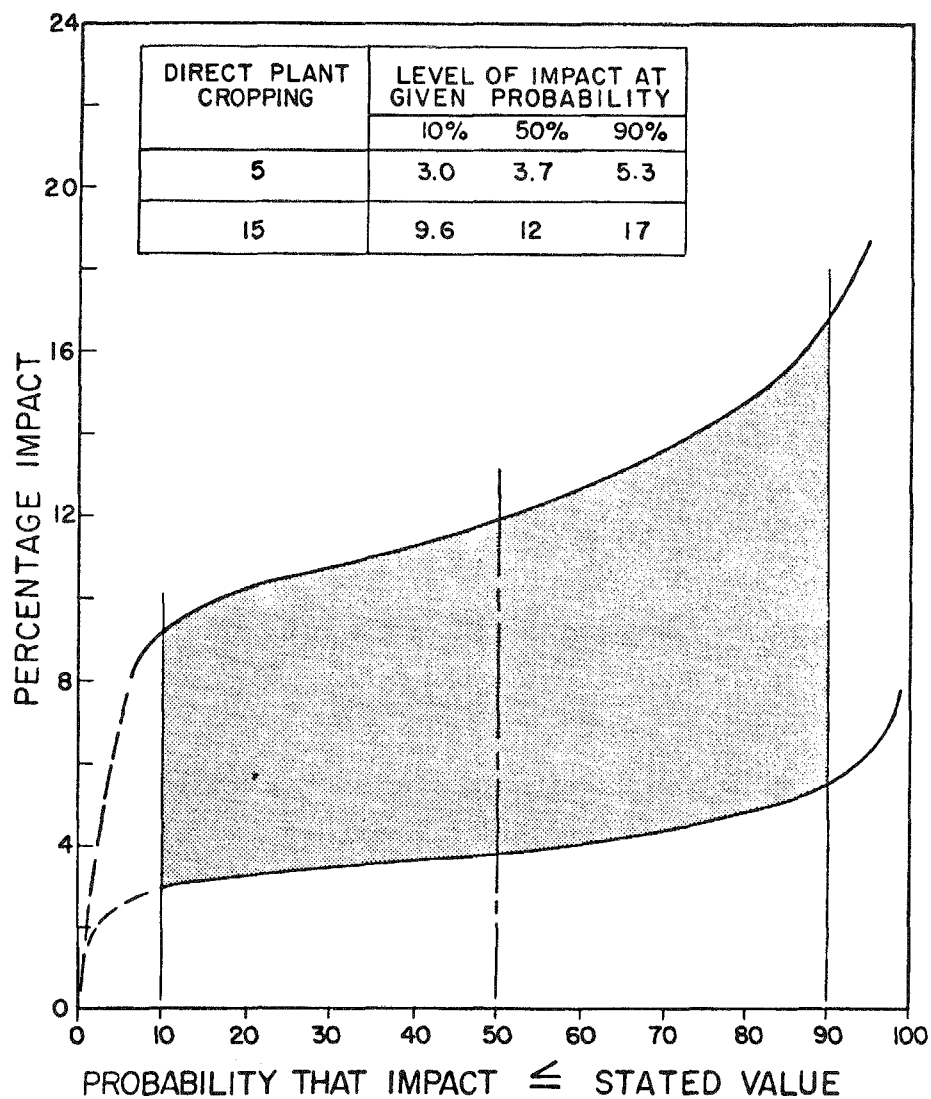


Figure 1.9-4. Stochastic Treatment of Impact

Source: Lawler and Englert 1977 (1.9-4)

1.10 LIFE CYCLE MODELS

1.10.1 Introduction

Life cycle models (LCM's) consist of a description of the population behavior of fish or major invertebrates over their entire life history. This description encompasses the various life stages through which the individuals progress and the primary mechanisms which collectively describe the survival strategy of the given species.

The time scales used in these models can vary from a single year of life, in which a sequence of life stages is examined, to many generations of individuals, i.e., 50 to 100 years. These time scales correspond to short- and long-term impacts, respectively.

The objective of the LCM's described here is to predict impacts of power plant operation on population levels. The emphasis, clearly, is to focus on those components of a life cycle which are susceptible to power plant effects and to incorporate these effects at the appropriate stages in the life history of the species, and finally to project these effects of the species under consideration on the long-term population levels.

1.10.2 Basis for Life Cycle Models

At first glance, the variety of species and their mechanisms and the diversity and changeability of nature, preclude a quantitative view of species behavior. Species are continually evolving, the genetic basis for their adaptability is only beginning to be understood.

When a more detailed view of species behavior is taken, the initial reluctance to model readily disappears. What we are really modeling is a single stage within an ecological succession of stages with regard to a given species. We make no attempt at this time to describe where a species is going in an evolutionary sense but simply capsulize its current status. This boils down to capturing its present physiological and behavioral adaptations which are operative over the time scale of interest. We simulate a given species, i.e., describe how it reacts to the biome (environment) in which it carries out its life cycle rather than describe why it reacts the way it does. The description of the factors affecting a species' behavior forms the basis for the life cycle models of impact assessment.

1.10.3 Elements of Life Cycle Models

The elements comprising LCM's are simply the relevant environmental (exogenous) and behavioral (endogenous) components which regulate the population, i.e., dictate its numbers during the various stages of the life cycle. The interaction of the power plant with these stages is then superimposed on the life cycle.

Although each species has its own unique set of behavioral characteristics, the major components of the life cycles of fish and major invertebrates are quite similar and appear to be variations of a common theme. This common theme can be best demonstrated by a specific example, the life history of the Hudson River striped bass (Morone saxatilis). Figure 1.10-1 schematically depicts the basic life cycle of striped bass. Also included are the locations within the life cycle of direct plant effects, i.e., entrainment and impingement.

The description of the life cycle will begin with the egg stage, progress through subsequent stages and finally return to the egg stage, thus completing and tracing a single generation of the life cycle. During the spring of each year, primarily in April and May, large numbers of adult striped bass migrate upriver to spawn in relative fresh water. Upon finding suitable spawning sites, the females begin dropping unfertilized eggs near the water surface over a short period of time. A typical adult female can carry and drop about 500,000 eggs. The majority of the spawning activity occurs during a four- to six-week period. As the eggs are dropped, a group of adult males broadcast sperm in the water and vigorous activity of adult bass ensues to enable some mixing and contact of the sperm and unfertilized eggs.

This spawning activity is depicted by arrows directed from adult age groups 4-13 to the egg symbol in Figure 1.10-1. The broken line arrows indicate that not all of the individuals of a given age group are sexually mature and therefore do not partake in the spawning activity. Here the term adult is used to indicate an individual that has survived the first year of life as opposed to one that is sexually mature. The size of the circles in the figure represents the relative numbers of a given life stage which generally decreases with the varying degrees of mortality of each stage.

Of the enormous amount of eggs spawned, many die off from such natural causes as lack of fertilization, predation, settlement to the river bottom. During

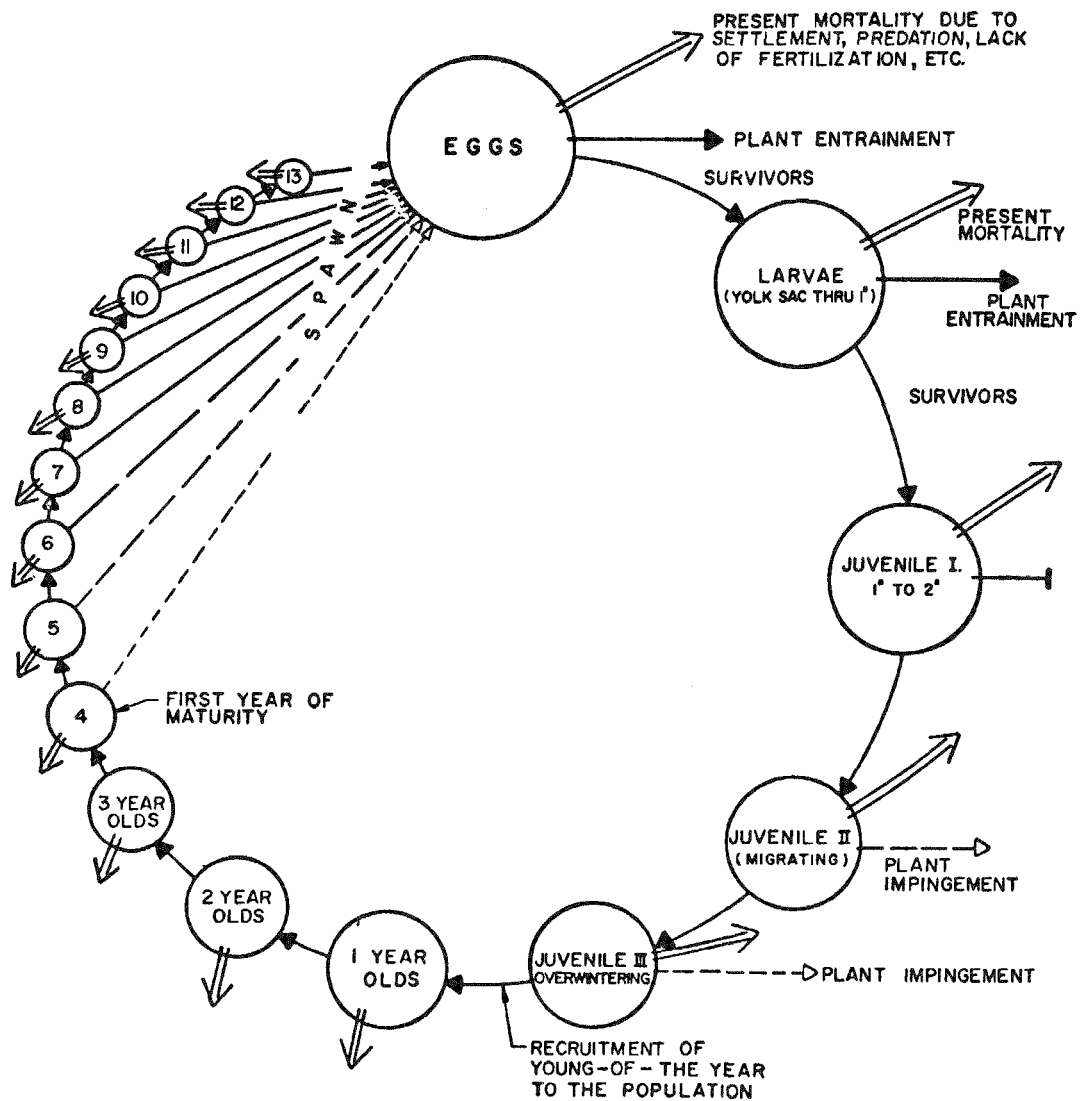


Figure 1.10-1. Schematic Life Cycle for Striped Bass in the Hudson River

the egg stage, which lasts about one to two days, depending primarily on river temperatures, the plant entrains those eggs in the condenser water flow with subsequent mortality because of elevated temperatures and pressure and mechanical stresses as the eggs pass through the cooling water system. The natural mortality associated with the eggs is shown by a double-lined solid arrow directed outward in Figure 1.10-1 leaving the egg stage and the plant-induced mortality (entrainment) by a single solid arrow directed outward.

As the surviving eggs hatch, they emerge as yolk-sac larvae. The yolk sac attached to the larvae serves as a food supply which is gradually absorbed over an interval of approximately one to two weeks, giving the larvae time for additional physiological development, after which they pursue natural food sources. This post-yolk-sac stage lasts about three to four weeks, at which time the larvae are about half-inch long. The larval stage is characterized by rapid development and a change from mostly passive behavior toward the environment to active behavior, i.e. active movements within the water body to avoid predators, search out food, etc. During the larval stage, active vertical movements (diel migration) occur within the water column, and this motion appears to be the dominant mechanism used to avoid rapid downstream movement and allow the larvae to maintain their positions within various estuarine reaches of the Hudson River.

The Lower Hudson River is estuarine for approximately 150 mi. to the Atlantic Ocean and is characterized by a two-layer flow regime in this region because of the combined effect of tidal action and density-(salinity) induced circulation. The two-layer flow regime is such that the net movement over any one tidal cycle (~12.5 hours) is upstream in the lower layer of the river and downstream in the upper layer. The net movement, both layers considered, is downstream and equal to the freshwater input. The larval striped bass have apparently adapted their behavior to this phenomenon and use it as an integral component of their survival strategy.

With respect to losses during this stage, both natural and plant induced mortality occur in a manner similar to that of the egg stage. The notable difference is that larval susceptibility to entrainment can be markedly different, depending on the location of the larvae within the water column and across the water cross section and the location of the plant intake withdrawal zone.

The larval stage is followed by the Juvenile-I stage which lasts about three to four weeks arbitrarily terminated at the time juveniles reach a two-inch length, the length at which they are too large to pass through the intake screens of power plants. This stage is characterized by further physiological development and active behavior, i.e., the fish are no longer partially controlled by river currents and can actively select preferred habitats in the river. Both natural mortality and entrainment mortality occur during this stage, but at reduced levels. Natural mortality is reduced by the increased ability of juveniles to locate food sources and avoid predators as well as by the decreasing number and type of predators which utilize the juvenile bass as prey because of its increased size. Entrainment mortality is reduced because of their decreased susceptibility to being passively withdrawn by the intake flow.

The Juvenile-II stage, commencing when the young bass reach two inches in size, is characterized by active movements within the river to nursery areas in search of both food and environmental conditions favorable to completing their remaining development and growth. The favorable nursery areas are believed to be shallows along the river with depths less than about 18 feet. Natural mortality is further decreased during this stage for the same reasons as during the Juvenile-I stage. Although entrainment, by definition, is absent here, organisms exposed to intake flows are susceptible to impingement on the intake screens thus causing plant induced mortality. Although the susceptibility to impingement decreases with size, as the juveniles grow, their mobility increases, and the impingement mortality rates appear to be controlled by their behavioral reactions to their environment rather than by their increased mobility. Actual impingement rates appear to depend more on water temperatures, salinity and the like than on Juvenile-II sizes.

The next stage, Juvenile III, begins around early December when river temperatures are sufficiently low to significantly diminish base food chain productivity and metabolic processes of the juvenile striped bass. The stage is characterized by low activity of the bass as they overwinter to adapt to their limited environment. This stage lasts until the ensuing spring when the surviving juveniles, having completed their major development phases, enter their first adult year. During their overwintering, the juveniles are vulnerable to natural mortalities due primarily to the severe environment which exists. Reduced temperatures and diminished food supplies contribute to their losses. The largest impingement rates also occur during this period, particularly January and February, when river temperatures are at their lowest, increasing the juveniles vulnerability to intake flows.

Figure 1.10-2 summarizes the early life-stage time periods during the first year of life for a typical spawning period. The remainder of the striped bass life cycle consists of approximately 13 adult years, sexual maturity reached by a fraction of 4-year-old adults, a larger number of 5-year-old adults, a greater number of 6-year-old adults, and all older adults.

The first two adult stages (1- and 2-year olds) are non-migratory and reside within the Hudson River or in its vicinity.

Subsequent adult stages migrate along the Atlantic coast, join other striped bass stocks, and return to the Hudson River annually for their spawning run in early April.

The resident adults are subject to some impingement losses, although this occurrence is usually sporadic and relatively small. The non-resident adults are not affected by the plant; their mortality is due to natural causes (predation, etc.) and both sport and commercial fishing.

To place the entire life cycle in perspective, fewer than two of the 500,000 eggs dropped annually by a ripe female need survive to become a mature adult to maintain a persistent striped bass population. Obviously, the reproductive strategy of the striped bass is to provide an over abundance of eggs to offset massive mortalities which occur during the early life stages.

The example presented illustrates several points common to all life cycles of interest to impact assessment. First, each life cycle can be divided into a consecutive set of discrete stages. Each stage in turn is characterized by certain behavioral characteristics, which coupled with the environmental conditions during that stage, dictate its susceptibility and vulnerability to plant impacts. Applying the plant effects to the early life stages will yield the short-term impact on the given species; tracking the short term impact through successive stages and succeeding generations yields the long-term impact.

Although the example used illustrated the striped bass life cycle, other life cycles pertinent to impact assessment will vary in detail but retain the same basic components. For example, different species will have different reproductive strategies and differing environmental conditions will have a major influence on

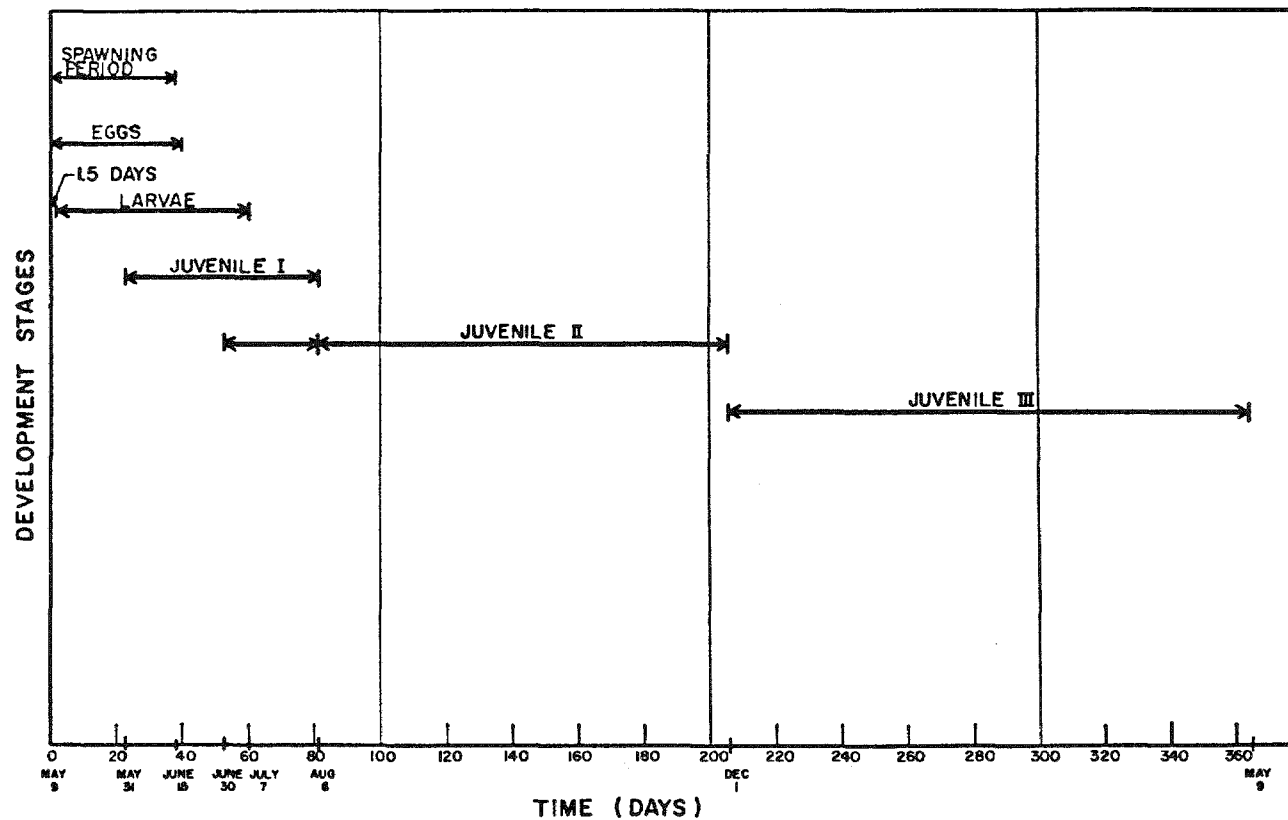


Figure 1.10-2. Striped Bass Development from Fertilized Egg to One Year of Age

the distribution of early life stages. They may spawn for a single year only, rather than multiple years, or alter their form drastically, such as the metamorphosis of blue crabs between their larval and juvenile stages. Coupling the various stages and the behavior associated with each stage and the pertinent environmental variables affecting the behavior with the operation of the power plant(s) constitutes the elements of life cycle modeling.

1.10.4 Population Persistence

Fundamental to the application of modeling of life cycles for impact assessment is the notion of persistence. By persistence we mean the regulation of a population within some finite range of abundance over a long period of time under a variety of environmental variation, both endogenous and exogenous to the population.

Over any appreciable period of time, evolutionary and genetic changes play a major role in the perpetuation of a species; over shorter periods of time, i.e., the time scales used in long-term impact assessment, minor evolutionary changes, if any, occur, and the current adaptation of the species is responsible for "absorbing" a wide range of environmental conditions and regulating its population size. The adaptation of the species consists of the organism's physiology and its behavioral mechanisms and their integration (the strategy) to regulate numbers. The species is continually faced with change, be it temporal or spatial, and is reacting to a variety of environmental variations.

Intimately linked with the notion of persistence is the necessity of density-dependent responses by a population (1.10-1). As density, i.e., abundance within some area, increases, the population will respond in some manner to curtail the increase to maintain abundance within the habitat's carrying capacity; conversely, as density decreases, the population will respond to curtail the decline and increase abundance. Although a population could increase its areal extent in reacting to population increases and thereby maintain its density, it is a limited and short-term reaction which must cease as its potential habitat is exhausted. Thus, to maintain abundance, some adjustment to density must occur.

The fundamental question in impact assessment is whether or not the species reacts to the changes in density or some exogenous environmental variation is responsible for (correlated with) the regulation of the species numbers.

If persistence is attributable to density-dependent responses by the population, then the species can potentially mitigate some or all of the plant impact; if persistence is due to the environment, which the plant does not regulate, then the population cannot compensate for losses. Environmental factors and density-dependent responses are actually linked in complex and unpredictable ways due to the stochastic nature of the environment.

The current approach to the problem of persistence is best exemplified by the work of Royama (1.10-1). The tactic taken was to simulate both answers for typical populations and their life cycles. Based on his results, he concluded that internal regulation of a population by its active reactions rather than external regulation by the environment more likely explains the known persistence of populations. Other evidence, such as 25 to 75% sustained commercial fishing exploitation rates, also supports this viewpoint.

1.10.5 Integration of Plant Effects

To adequately assess the impact of plants on the life cycle of various species, two types of effects need be considered: direct and indirect. The direct effects, entrainment and impingement, can be incorporated in a straightforward manner, as described previously, given a knowledge of the vulnerability of the appropriate stages in the life cycle. The indirect effects such as temperature, habitat alteration, selective impingement of weaker individuals, recycling of entrained organisms to the water body and the like, are highly variable with different species and between life stages of a given species.

The susceptibility of various life stages to direct and indirect impacts depends strongly on the life strategy of the species and the various behavioral characteristics of the life stages. For example, those species which nest and incubate their eggs avoid egg susceptibility to entrainment to a great extent. Those species which bear their young live also avoid egg susceptibility as well as larval susceptibility for the period of time prior to birth.

Various strategies for perpetuation of the species throughout the life cycle have evolved. Generally those species with low birth rates have developed protective mechanisms to ensure a high survival rate during the developmental stages, whereas those with high birth rates (e.g., 500,000 eggs/fecund female) can sustain

a low survival rate through many of the early life stages and thus exhibit less complex protective mechanisms.

The susceptibility to entrainment during the larval and early juvenile stages, up to the period at which their size precludes passage through the screen system, varies greatly, depending primarily on the physiological (swimming speed, growth rate, etc.) and behavioral (pelagic, benthic, etc.) characteristics of the species. Both the physiological development and behavioral characteristics change rapidly during larval life, and the susceptibility therefore can change markedly as the organisms develop. These changes can occur abruptly (e.g., absorption of the yolk-sac by striped bass larvae) or gradually (e.g., the change from pelagic to benthic feeding in young spot).

As the juvenile stage progresses the fish are susceptible to impingement, again affected by their behavioral and physiological characteristics. Although the direct effects always reduce a population size immediately (the short-term impact), the long-term impact of direct effects can be significantly reduced by compensatory (density-dependent) responses by the population. These density-dependent mechanisms are more likely to be present during the first year of life, i.e., the egg, larval and juvenile stages, when densities are largest and changing the fastest. During this period, rapid development of the species and alterations in its behavior are also most prominent.

The potential for compensation and its occurrence within the life cycle correspond to the period when both entrainment and impingement occur, thus offering greater potential for immediate, as opposed to delayed, responses by the population to reductions in its numbers.

On the other hand, indirect effects can tend to either reduce or expand a population size. Moderate elevations of water body temperature can accelerate the development of early life stages and allow these organisms to better survive the remaining hazards of their first year; high elevations of water body temperature can adversely affect both their environment, by depressing dissolved oxygen levels, and their own physiology, by increasing metabolic activity thus requiring more food for sustenance and more rapid growth.

Many other indirect effects have similar two-sided effects. Each, in turn, must be considered for the species and the various life stages in question.

1.10.6 Current Life Cycle Models Used in Impact Assessment

Life cycle models can be classified a number of different ways dependent on the objective of the classification. For example, classification can be based on the spatial complexity (one-, two- or three-dimensional) or the spawning behavior (catadromous, anadromous, etc.). Regardless of the form of the model, the discussion presented here will center upon the structural aspects and the manner and degree in which life history and power plant effects have been incorporated into the models. To this end, the existing models are divided into two categories: simple LCM's and complex LCM's. Both types have their particular use in impact assessment.

The simple LCM's take a gross view of the life cycle itself, the environment in which the life cycle exists, and the details of plant effects. These models are used to estimate probable ranges of impact with a minimum of effort. In many cases they are used to focus rapidly on a variety of species and delineate those species where further assessment is required.

The complex LCM's take a more detailed view of one or more of the items grossly considered by the simple LCM's. The resolution given to the water body is increased, and the additional details of the life cycle and the plant's effects on it are incorporated. The temporal resolution is also increased to include more detailed environmental variation and the corresponding behavioral reactions to the environment. As the resolution is increased in the complex LCM's, the models have become site- and species-specific to a large degree. The compromise here sacrifices generality for accuracy.

Examples of the simple LCMS are the completely mixed striped bass model (CMSBM) for the Hudson River (1.10-2) and the generalized fish life cycle model developed by ORNL (1.10-3).

In the case of the CMSBM, the resolution given to the river system was to assume that all of the young striped bass (eggs through Juvenile III) were equally susceptible to the plant, i.e. the river was treated as a completely mixed water body. Spawning was assumed to recur at a constant rate over about a five-week period, and except for intake water flows, other plant site-specific characteristics were not directly included in the model. Other relevant parameters of the life cycle (stage survivals, fraction of males and females in adult spawning years, etc.)

were chosen to represent an equilibrium status without the plant operating, i.e., the population was persistent at the given level for the parameters chosen, in the absence of any stresses on the system. Initial versions of this model did not contain any population regulation mechanisms such as density-dependent survival. However, it soon became apparent that the addition of any continuous stress to the model, e.g., removing a single adult per year, would cause the population to collapse.

Later versions of the model incorporated a simple density-dependent regulation mechanism (variation of Juvenile II survival with Juvenile II density) to control population extinctions. The overall logic used in the CMSBM is shown in Figure 1.10-3. Typical values of life cycle parameters used in the CMSBM are shown in Tables 1.10-1 through 1.10-5. The output of the CMSBM consists of the population of early life stages as a function of time within a year and adult population levels at the beginning of each spawning year. Typical outputs of the model are shown in Figures 1.10-4 to 1.10-10.

Figure 1.10-4 is a graph of total adult striped bass vs time for the life cycle parameters indicated for an equilibrium selection of parameters. The plot demonstrates the ability of the CMSBM to maintain a persistent (and constant) population over a 180-year period in the absence of external stresses. Figure 1.10-5 illustrates the ability of the CMSBM to reach a new lower equilibrium population after a temporary stress of plant operation for a one-year period. The lesser population results because of the absence of a population-regulation mechanism in the version of the CMSBM used to generate these results. Figure 1.10-6 demonstrates the effect of randomly varying one of the life cycle parameters (Juvenile II survival, in this case) in the absence of external stresses and population regulatory mechanisms. For the condition shown, the random variation creates an unbounded population growth.

Figure 1.10-7 demonstrates the regulating effect of density-dependence. Here random variation was used for Juvenile I stage survival with the Juvenile II stage survival depending on its density. Although the results show high variability in the population levels, no upward or downward trend is evident during the 180 years.

Figures 1.10.8 and 1.10-9 illustrate the sensitivity of the CMSBM to the adult female fecundity values and changes in egg spawning rates over a fixed spawning

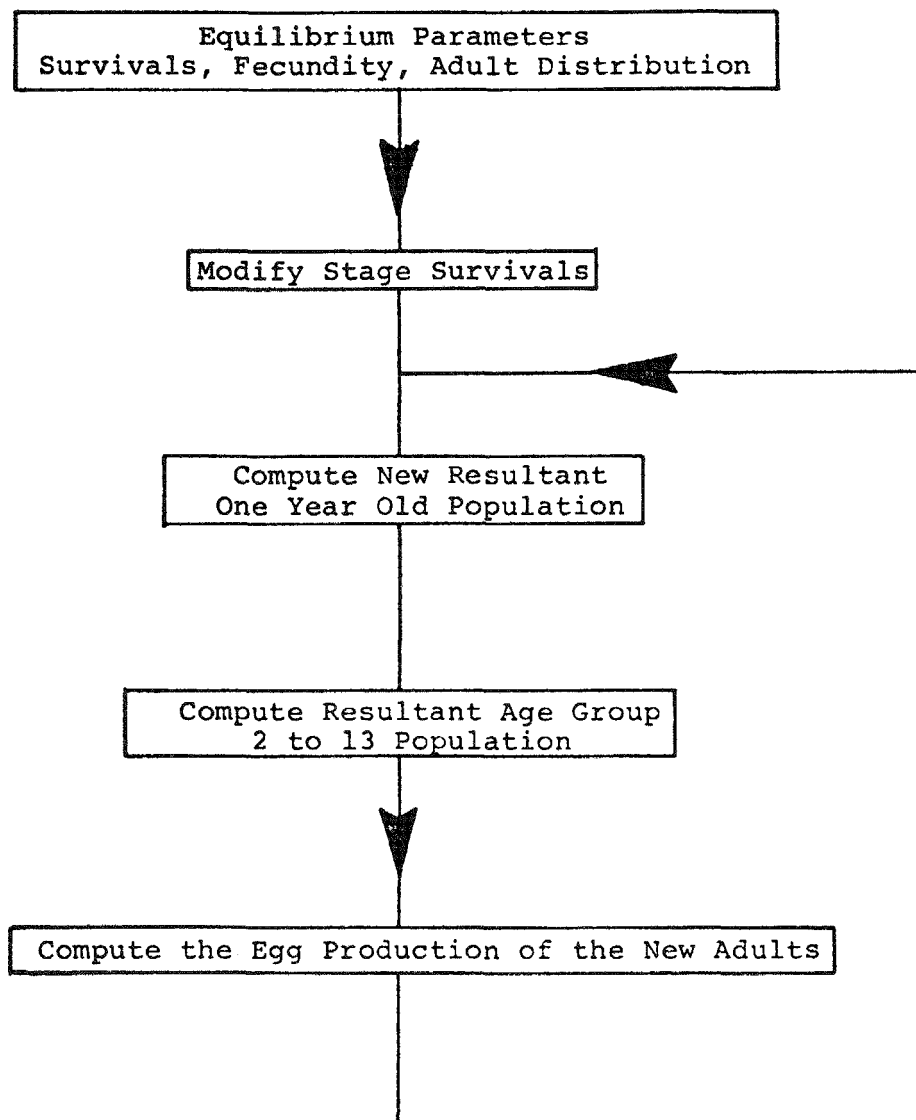


Figure 1.10-3. Computation of adult striped bass population when equilibrium is disturbed.

Table 1.10-1

STAGES AND ASSOCIATED PARAMETERS
OF THE FIRST YEAR OF LIFE FOR STRIPED BASS

Stage	Description	Period of Life	Time of First Appearance		Passage to Next Stage		Size at time of Passage to next Stage (inches)	References and Remarks
			Earliest Date	Latest Date	Earliest Date	Latest Date		
Egg	Spawn to hatch	1.5	May 9	June 15.5	May 10.5	June 17	0.134	Mansueti (3) Fer- tilized & water hardened
Larval	Hatch to free swimmers	21	May 10.5	June 17	May 31.5	July 8	3/4 to 1	Pearson (8), Hump- ries & Cummings(16)
J _I	Non-screenable	30	May 31.5	July 8	June 30.5	Aug. 7	2	Mansueti (3), Rathjen & Miller(6)
J _{II}	Screenable, migrating	116-153.5	June 30.5	Aug. 7	Dec. 1	Dec. 1	3.3	Carlson - McCann (7)
J _{III}	Screenable, over wintering	159	Dec. 1	Dec. 1	May 9 of following year		3-4	Little growth during overwintering stage

NOTES:

1. Period of Spawn = 37.5 days
 2. This date coincides with the earliest date of 1st appearance of the next stage
 3. This date coincides with the latest date of 1st appearance of the next stage
- Source: Lawler (1.10-4).

Table 1.10-2

GROWTH DATA BASED ON AVERAGE SIZES FOR VARIOUS AGES
OF LARVAL AND YOUNG STRIPED BASS

Mansueti, 1958		Pearson, 1958		Patuxent River Seine Collections		Hudson River Data Cornwall Report 1965, 1968		Hudson River Data Q&M Collections, 1971		Hudson River Data Rathjen & Miller 1955		Humphries & Cumming	
Observed Age	Ave. Total Length (mm)	Observed Age	Ave. Total Length (mm)	Estimated Age	Ave. Total Length (mm)	Estimated Age*	Ave. Total Length (mm)	Estimated Age*	Ave. Total Length (mm)	Estimated Age*	Ave. Total Length (mm)	Estimated Age	Ave. Total Length (mm)
Hatching	2.9	Hatching	3.2										
1 day	3.6	---											
1.5 days	---	1.5 days	4.4										
2 days	5.1	2 days	---										
3 days	5.6	3 days	5.2										
		4 days	5.8										
		6 days	6.0									3 wks	17 (.7")
		8 days	9.0										
		16 days	13.0 (.5")							(1955) 5 wks	26 (1")		
		3-4 wks	36.0 (1.4")	4 wks	42.0 (1.7")	(1968) 7 wks	23 (.9")	7 wks	38 (1.5")			7 wks	49 (1.8")
				8 wks	68.0 (2.7")	(1968) 8 wks	33 (1.3")			(1955) 8-9 wks	51 (2")		
						10-11 wks	51 (2")	11 wks	61 (2.4")			11-12 wks	51 (2")
						(1965) 19 wks	79 (3.1")						

* Age calculated from May 28 which is the mid-point in the spawning season.
Source: Lawler (1.10-4).

Table 1.10-3

SURVIVAL RATE SELECTION

<u>Stages</u>	<u>Percentage Survival Ranges</u>	<u>Percentage Survivals Selected for Use in Model</u>		
		<u>Run #1</u>	<u>Run #2</u>	<u>Run #3</u>
<u>Eggs</u>	1 to 10	1	10	10
<u>Larvae</u>	0.05 to 1	0.5	1	1
<u>Juveniles</u>				
I - non-screenable	20	20	20	20
II - screenable, migrating	40 to 60	40	50	60
III - screenable, overwintering	20 to 40	20	18.4*	40
<u>Developing Adults</u>				
One & Two Year Olds	30 to 99*	99*	51.8**	66.8*
Three Year Olds	30 to 90	80.1*	51.8**	30
<u>Mature Adults</u>				
Four to 13 Year Olds	30 to 90	80	51.8	30

* Chosen to permit the total set of survival and fertility factors to satisfy the condition of equilibrium.

**Chosen originally as 50%. Associated mortality rate, K, was then computed and rounded. This resulted in a slightly higher survival of 51.8%.

Source: Lawler (1.10-4).

Table 1.10-4

SELECTED FERTILITY FACTORS

<u>Age Group</u>	<u>Female Fraction*</u>	<u>Female Maturity**</u>	<u>Fecundity (Eggs/Fertile Female)***</u>
1	.5	0	0
2	.52	0	0
3	.54	0	0
4	.56	.25	345,000
5	.58	.75	438,000
6	.6	.95	615,000
7	.62	1	752,000
8	.64	1	820,000
9	.66	1	909,000
10	.68	1	910,000
11	.7	1	964,000
12	.7	1	1,136,000
13	.7	1	908,000

* (Ref.19) Values between age group 1 and 11 were linearly interpolated from estimates of age group 1 and 11 female fractions.

** (Ref.20,21) Age groups 4 - 6 are computed estimates.

*** (Ref.18) Computed by weighted averages of individual measurements.

Source: Lawler (1.10-4).

Table 1.10-5
AGE GROUP DISTRIBUTION OF ADULTS
AT BEGINNING OF SPAWNING (MAY 9)

(Equilibrium Model)

<u>Year Class</u>	<u>Numbers</u>
1	1,455
2	1,440
3	1,426
4	1,141
5	913
6	731
7	585
8	468
9	374
10	299
11	240
12	192
13	153
TOTAL	9,417 say 10,000

<u>Stage</u>	<u>% Survival Used In Model</u>
Eggs	1
Larvae	.5
Juveniles	
I	20
II	40
III	20
Developing Adults	
One Year Olds	99*
Two Year Olds	99*
Three Year Olds	80.1
Mature Adults	80

* to satisfy equilibrium equation
Source: Lawler (1.10-4).

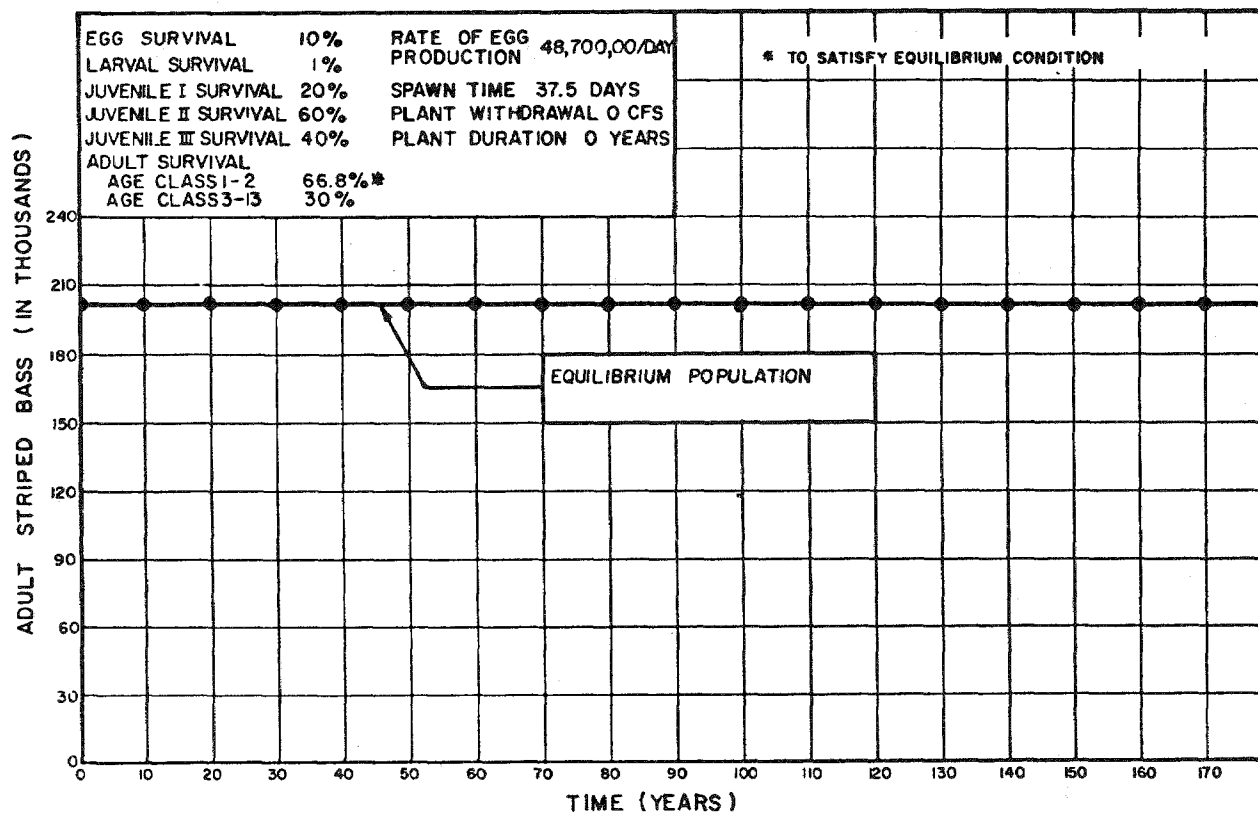


Figure 1.10-4. Adult Striped Bass Equilibrium Population in the Hudson River

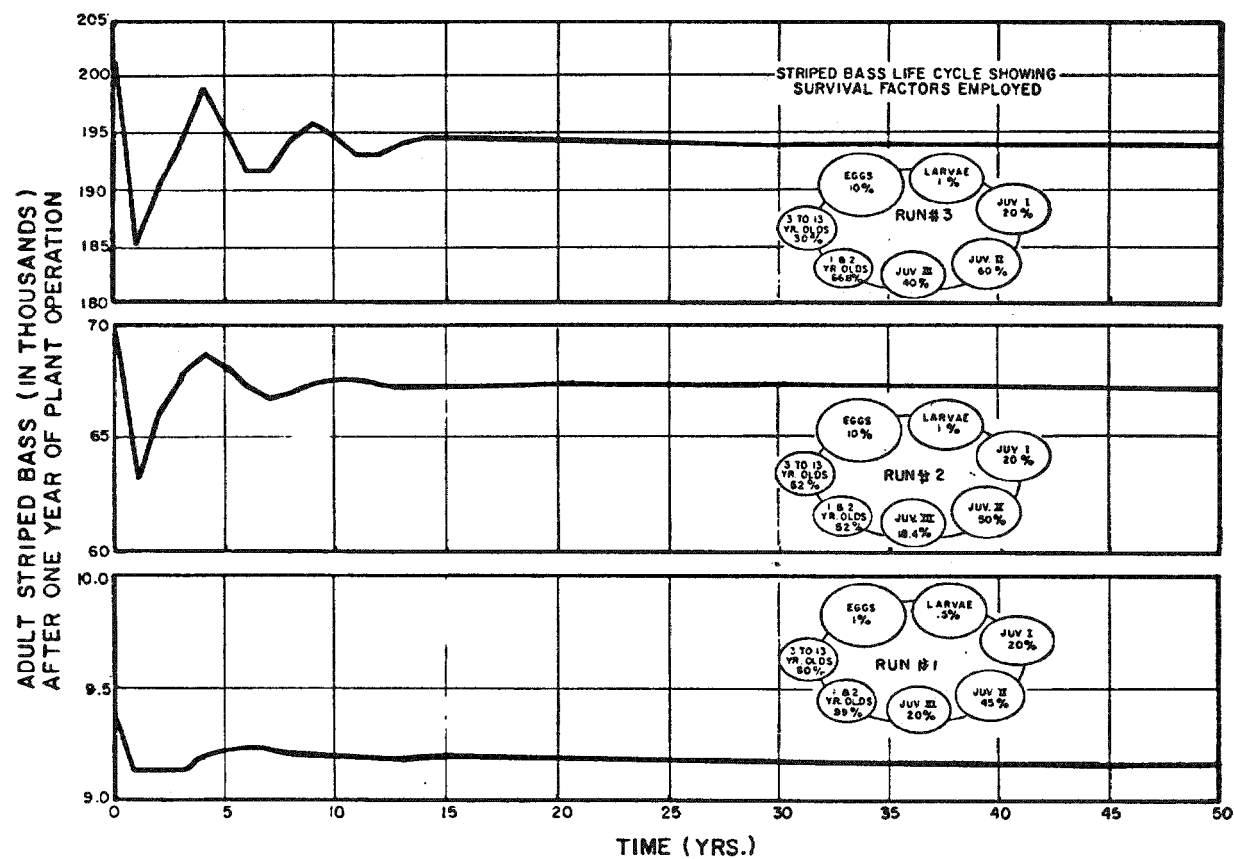


Figure 1.10-5. Three estimated equilibrium adult striped bass populations after year's operation of Indian Point Units 1 and 2 at design flow.

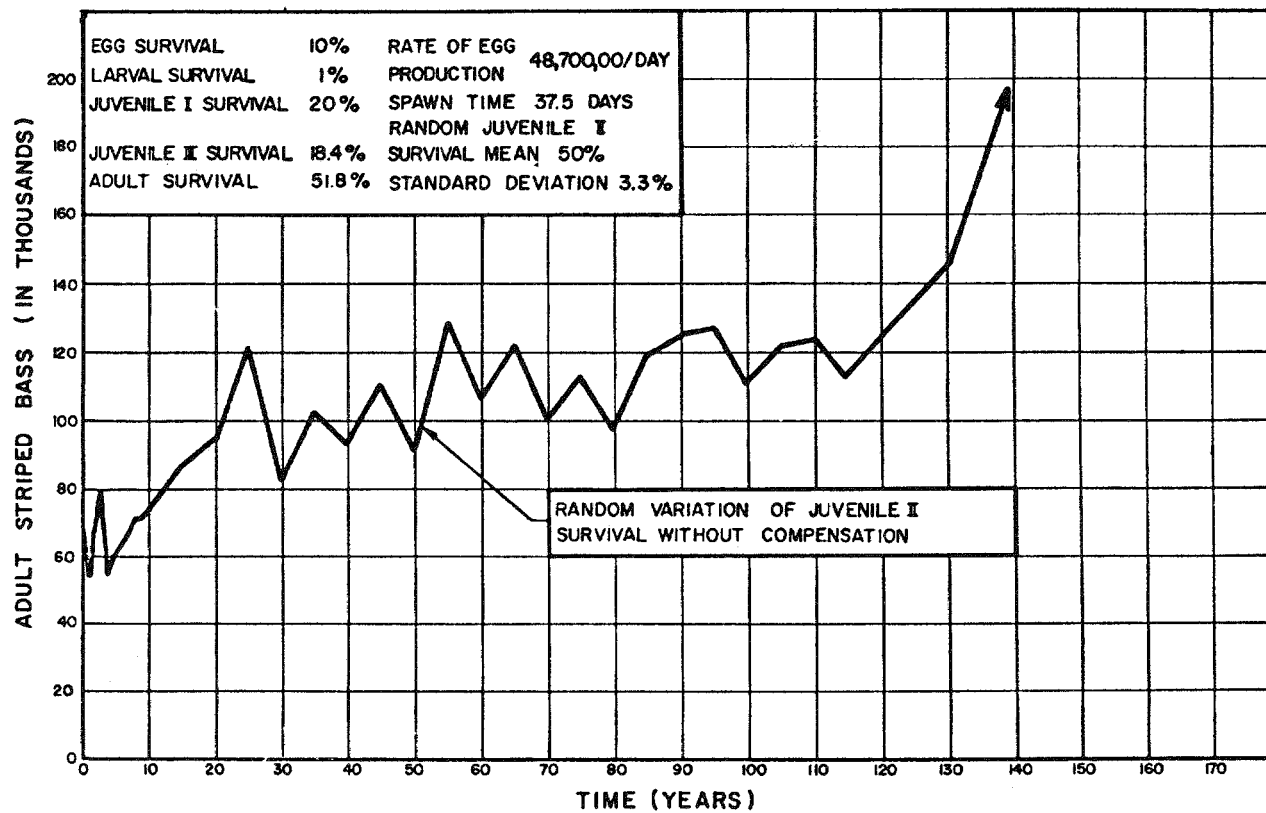


Figure 1.10-6. The Effect on Random Variation of Adult Striped Bass Survival Without Compensation

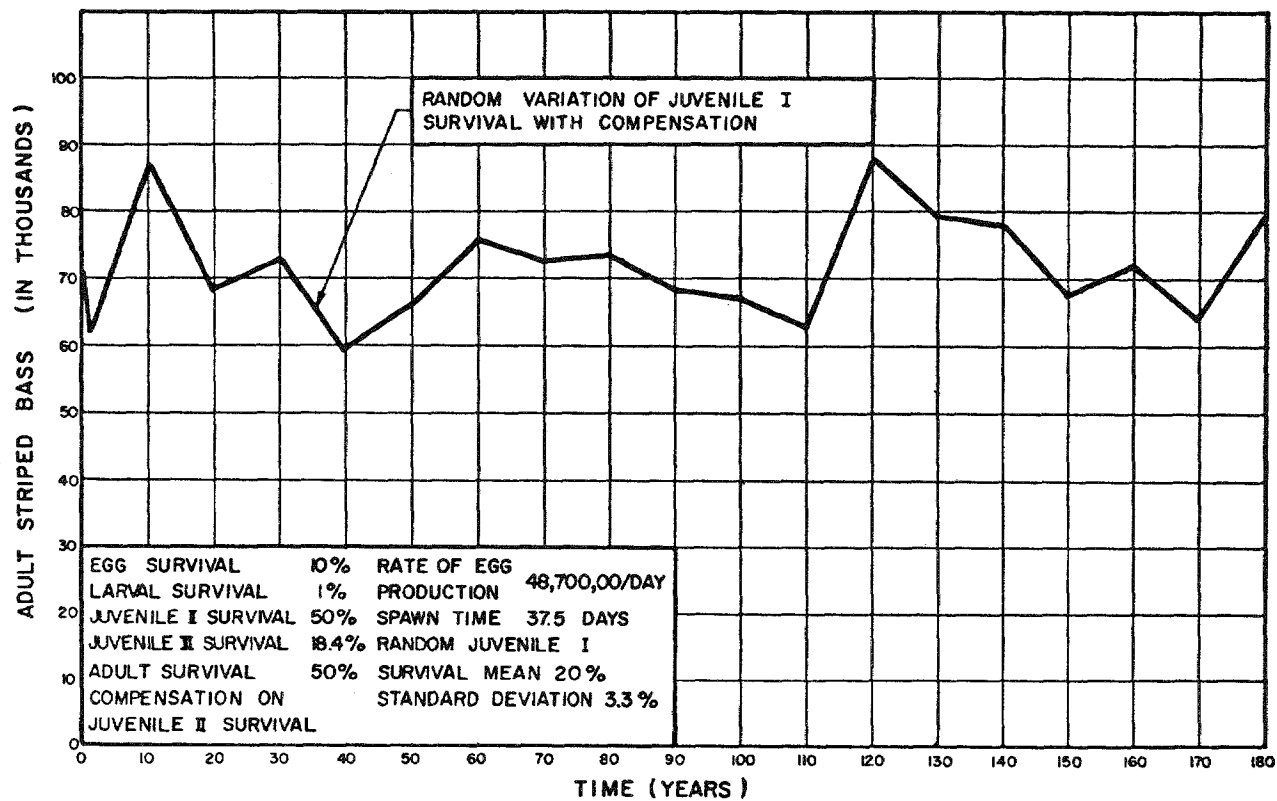


Figure 1.10-7. The Effect of Compensation on Random Variation of Adult Striped Bass Survival

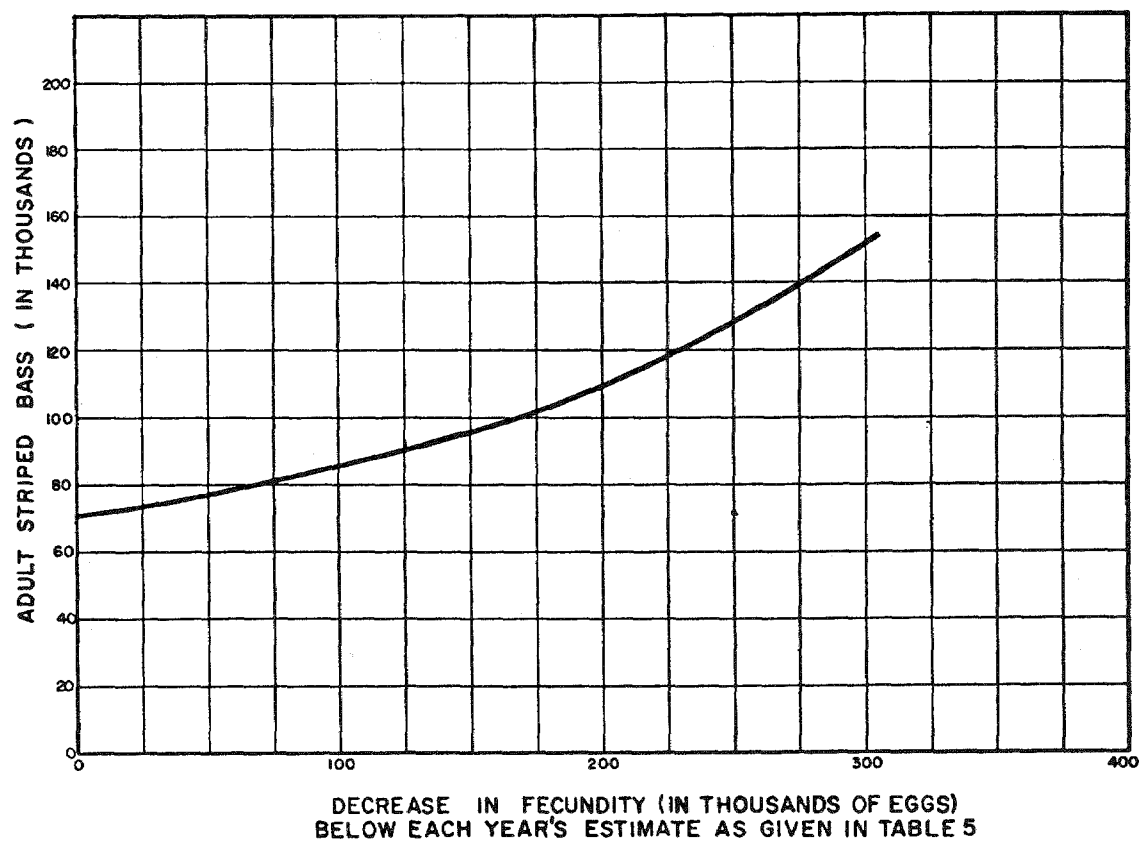


Figure 1.10-8. Effect of reduction in fecundity of mature females on the Hudson River adult striped bass population.

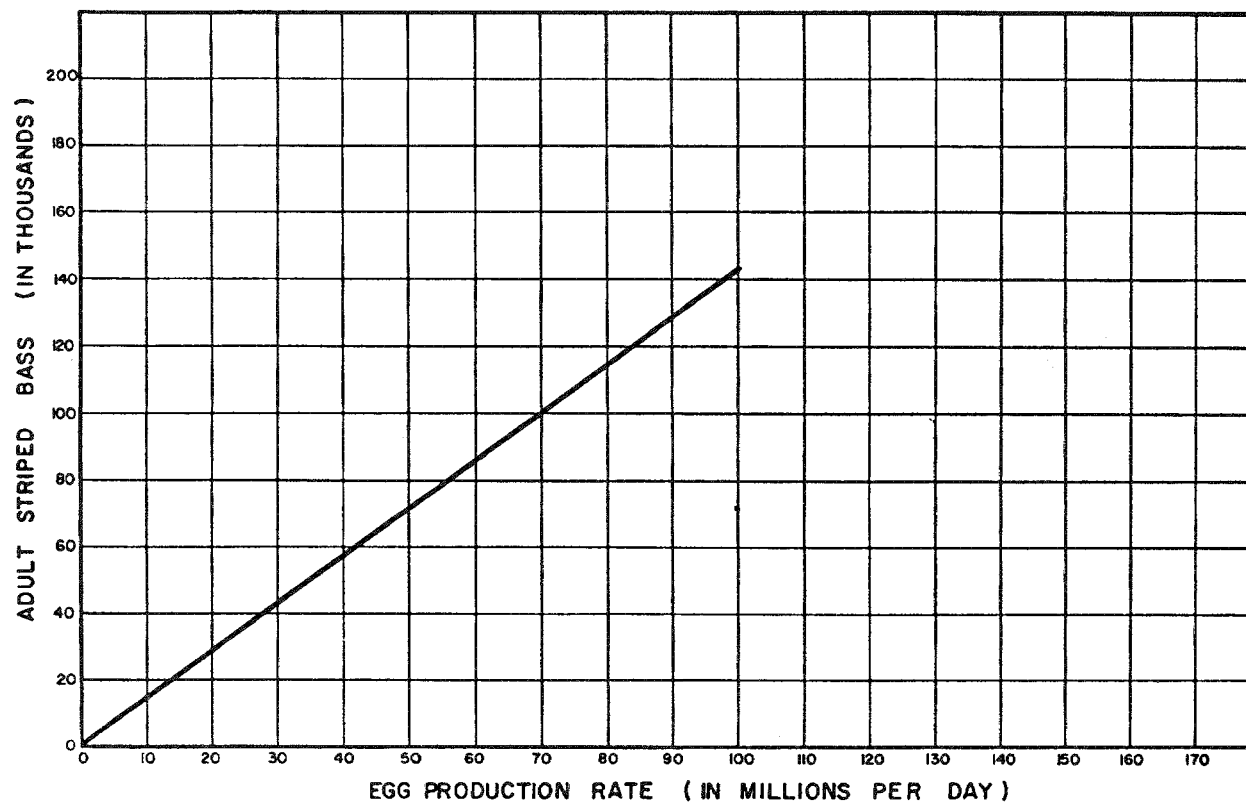
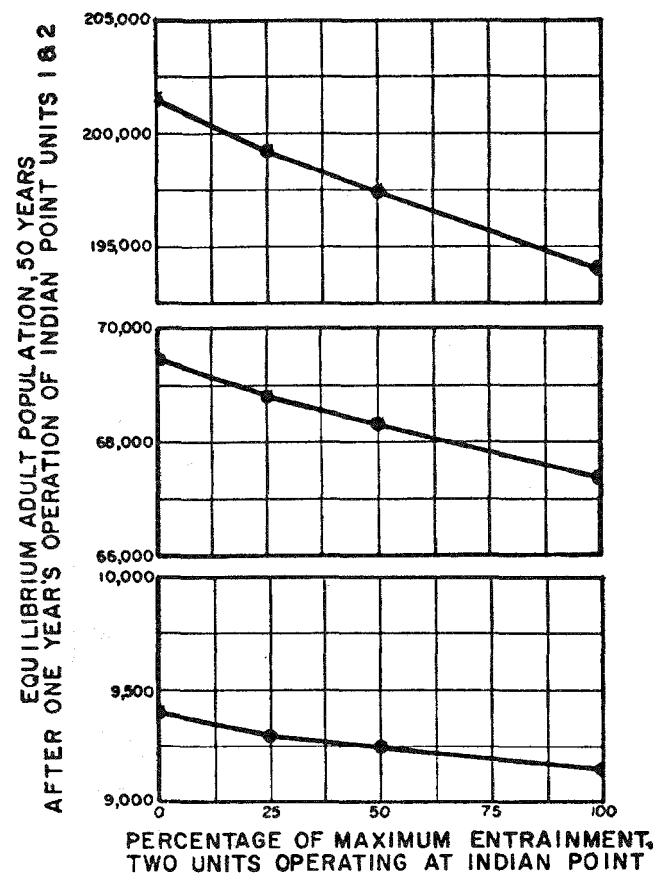
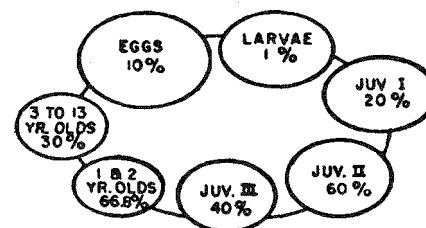


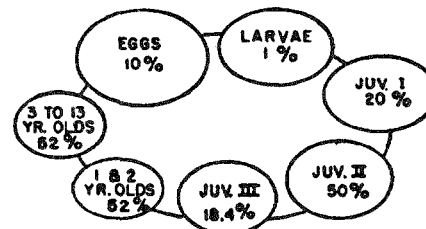
Figure 1.10-9. Adult Striped Bass vs Egg Production Rate



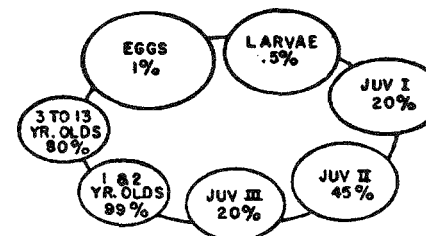
STRIPED BASS LIFE CYCLE SHOWING SURVIVAL FACTORS EMPLOYED



RUN # 3



RUN # 2



RUN # 1

Figure 1.10-10. Effect of one year's operation of Indian Point Units 1 and 2 on Hudson River striped bass population.

period, respectively. Finally, Figure 1.10-10 illustrates the sensitivity to the choice of susceptibility/vulnerability values expressed as the percentage of maximum entrainment (100% is completely mixed, i.e., all susceptible; and 0% survival through plant cooling system, i.e., all vulnerable) for three choices of life cycle parameters.

The CMSBM illustrates the type of simple LCM's which have been developed. They can usually be applied to other life cycles with minor modifications since the resolution is usually gross. This allows the use of estimated parameter values to be input for the missing details as opposed to the development of empirical or scientific formulae to describe the details.

The complex LCM's include a greater amount of detail regarding the species, the water body, the environmental variation, and plant operational characteristics than the simple LCM's. They require extensive literature searches and data collection programs to support this level of detail. Here clearly the compromise made is to sacrifice generality for accuracy.

Typical complex models are the Hudson River Striped Bass Transport Model (1.10-2, 1.10-5) HRSBTM, the Hudson River Striped Bass Real-Time Model (1.10-6) HRSBRTM, the Oak Ridge National Laboratory Models (1.10-7) ORNLM's and the Cape Fear Estuary Model, FPM.

All of these models have included time variable environmental parameters, detailed time variable plant characteristics, several behavioral mechanisms, and at least one density-dependent population regulation mechanism (except the FPM model, see below). The first three models, HRSBTM, HRSBRM and ORNLM, have been used to predict the impact of single and multiple power plants on the Hudson River striped bass population. The latter two have been continually modified as additional data collection efforts and analyses are completed. These, as well as other complex LCM's, are best described as evolutionary models which attempt to keep pace with the refinement for the understanding of life cycle dynamics.

The ORNLM's are really two programs which taken together represent the overall life cycle. The first program includes young-of-the-year details and impacts, and the second program, given a summary of results from the first program, simulates the long-term life cycle behavior of striped bass.

The Cape Fear Estuary Model simulates only the early life cycle for both spot and croaker. Marine spawners, spot and croaker, first enter the estuary in the larval stage. Little is known about their adult stages. Therefore, a great deal of life cycle detail on the larval through juvenile stages was included in the FPM Model but no population regulation mechanisms were incorporated. The FPM Model is typical of those models which address a portion of the life cycle completely but cannot address the long-term impact directly because of an incomplete understanding of basic components of the life cycle of the species studied.

1.10.7 Advantages and Limitations of Current LCM's

LCM's developed to date have provided a quantitative framework to assess both the short-term (one year) and long-term (up to 100 years) impact of power plant operation on several species of fish and major invertebrates. In so doing, the stresses placed on given populations by both direct and indirect plant effects have been placed in perspective, including the analogous behavior of direct plant effects and both sport and commercial exploitation of population stocks.

In general, it has been shown that no prejudice should be placed on power plant usage of a resource (in this case the resource includes population stocks, the intake water, etc.) when the plant is viewed as a single user of a multiple user resource.

In addition to the direct use of LCM's to assess impact, many by-products of the LCM approach exist. LCM's serve as both integrators of our available knowledge of life histories as well as indicators of the gaps that exist in our knowledge. Sensitivity analysis of the LCM can be used to indicate which gaps are significant and where "knowledge-collection-efforts" (field data, literature searches, laboratory experiments and the like) are needed.

The major limitation of the LCM's developed to date is their narrow scope. This limitation has arisen for a variety of reasons. Modeling of the life cycle for impact assessment has a relatively short history and the models have been developed in an atmosphere where specific concerns were emphasized and time constraints were excessive. This has limited the adaptability of the current models to an undesirable extent, particularly for detailed assessments of new species or the same species at a different site. These limitations apply to the manner in which

the components of the models have been aggregated and integrated, not to the technique itself.

REFERENCES

- 1.2- 1. S. L. Weiss. A Sampling Frame for Entrainment Surveys. Unpublished Manuscript, 1977.
- 1.2- 2. W. S. Abbott. "A Method of Computing the Effectiveness of an Insecticide." J. Economic Entomol., Vol. 18, 1925, pp. 265-267.
- 1.2- 3. B. C. Marcy Jr. "Vulnerability and Survival of Young Connecticut River Fish Entrained at a Nuclear Power Plant." J. Fish. Res. Bd. Can., Vol. 30, No. 8, 1973, pp. 1195-1203.
- 1.2- 4. S. L. Weiss and D. T. Logan. "Statistical Problems Associated with the Analysis of Viability Studies Performed Under Uncontrolled Conditions." Presented at the Fourth Biennial International Estuarine Research Conference, 1977.
- 1.2- 5. Ecological Analysts, Inc. Bowline Generating Station Entrainment Survival and Abundance Studies: 1975 Annual Interpretive Report. Report submitted to Orange and Rockland Utilities, Inc., 1976.

REFERENCES

- 1.3- 1. K. Sharma and R. F. Freeman III. Survey of Fish Impingement at Power Plants in the United States. I. The Great Lakes. Argonne National Laboratory, 1977. ANL/ES-56.
- 1.3- 2. R. F. Freeman and R. K. Sharma. Survey of Fish Impingement at Power Plants in the United States. II. Inland Waters. Argonne National Laboratory, 1977. ANL/ES-56.
- 1.3- 3. R. C. Stupka and R. K. Sharma. Survey of Fish Impingement at Power Plants in the United States. III. Estuaries and Coastal Waters. Argonne National Laboratory, 1977. ANL/ES-56, 310 p.
- 1.3- 4. L. R. King, J. B. Hutchinson Jr., and T. G. Huggins. "Impingement Survival Studies on White Perch, Striped Bass and Atlantic Tomcod at Three Hudson River Power Plants." In Fourth National Workshop on Entrainment and Impingement, pp. 217-233. Melville, N.Y.: Ecological Analysts, Inc., 1978.
- 1.3- 5. T. R. Tatham, D. L. Thomas, and G. J. Miller. "Survival of Fishes and Macroinvertebrates Impinged at Oyster Creek Generating Station." In Fourth National Workshop on Entrainment and Impingement, pp. 235-243. Melville, N.Y.: Ecological Analysts, Inc., 1977.
- 1.3- 6. J. C. White Jr. and M. L. Brehmer. "Eighteen-Month Evaluation of the Ristroph Traveling Fish Screens." In Third National Workshop on Entrainment and Impingement, pp. 367-380. Melville, N.Y.: Ecological Analysts, Inc., 1976.
- 1.3- 7. C. Coutant. "Temperature, Reproduction and Behavior." Chesapeake Sci., Vol. 10(3 and 4), 1969, pp. 261-274.
- 1.3- 8. A. C. Gross. "Comparison and Prediction of Fish Impingement Rates at Power Plant Cooling Water Intake Sites." In Third National Workshop on Entrainment and Impingement, pp. 255-267. Melville, N.Y.: Ecological Analysts, Inc., 1976.
- 1.3- 9. T. L. Page, D. A. Neitzel, and R. H. Gray. "Comparative Fish Impingement at Two Adjacent Water Intakes on the Mid-Columbia River." In Fourth National Workshop on Entrainment and Impingement, pp. 257-266. Melville, N.Y.: Ecological Analysts, Inc., 1978.
- 1.3-10. R. A. Alevras. "The Estimation of Fish Impingement at Indian Point Units 1 and 2." Testimony before U.S. Atomic Energy Commission. U.S.A.E.C. Docket No. 50-247, 1973.
- 1.3-11. Consolidated Edison Company of New York, Inc. and Power Authority of the State of New York. Near-Field Effects of Once-Through Cooling System Operation on Hudson River Biota. New York, 1977.
- 1.3-12. I. P. Murarka. A Model for Predicting Fish Impingement at Cooling Water Intakes. Report prepared for U.S. Energy Research and Development Administration by Argonne National Laboratory, 1977. ANL/AA-8.
- 1.3-13. Tetra Tech. Inc., personal communication with Marc Lorenzen.

REFERENCES

- 1.3-14. J. M. Loar, J. S. Griffith, and K. D. Kumar. "An Analysis of Factors Influencing the Impingement of Threadfin Shad at Power Plants in the Southeastern United States." In Fourth National Workshop on Entrainment and Impingement, pp. 245-255. Melville, N.Y.: Ecological Analysts, Inc., 1978.
- 1.3-15. J. E. Kerr. "Studies on Fish Preservation at the Contra Costa Steam Plant of the Pacific Gas and Electric Company." State of California Department of Fish and Game, Fish Bulletin, No. 92, 1953.
- 1.3-16. Orange and Rockland Utilities, Inc. Bowline Point Generating Station: Near-Field Effects of Once-Through Cooling System Operation on Hudson River Biota. Spring Valley, N. Y.: Orange and Rockland Utilities, Inc., 1977.
- 1.3-17. P. J. Eisele and J. F. Malaric. "A Conceptual Model of Causal Factors Regarding Gizzard Shad Runs at Steam Electric Power Plants." In Fourth National Workshop on Entrainment and Impingement, pp. 291-298. Melville, N.Y.: Ecological Analysts, Inc., 1978.
- 1.3-18. R. L. Wyman, J. Payne, and D. Andrews. "Hydroacoustic Assessments of Fish Populations." Twentieth Conference of Great Lakes Research: Abstracts. International Association for Great Lakes Research, 1977.
- 1.3-19. I. P. Murarka and D. J. Bodeau. Sampling Designs and Methods for Estimating Fish-Impingement Losses at Cooling Water Intakes. Argonne National Laboratory, 1977. ANL/ES-60.
- 1.3-20. I. P. Murarka, A. Spigarelli, and D. J. Bodeau. "Statistical Comparison and Choices of Sampling Designs for Estimating Fish Impingement at Cooling Water Intakes." In Fourth National Workshop on Entrainment and Impingement, pp. 267-280. Melville, N.Y.: Ecological Analysts, Inc., 1978.
- 1.3-21. W. G. Cochran. Sampling Techniques. 2nd ed. New York: John Wiley & Sons, 1963.
- 1.3-22. K. D. Kumar and J. S. Griffith. "Temporally Stratified Sampling Programs for Estimation of Fish Impingement." In Fourth National Workshop on Entrainment and Impingement, pp. 281-289. Melville, N.Y.: Ecological Analysts, Inc., 1978.
- 1.3-23. P. V. Sukhatme and B. V. Sukhatme. Sampling Theory of Surveys With Applications. 2nd ed. Ames, Iowa: Iowa State University Press, 1970, 452 p.
- 1.3-24. I. P. Mararka and D. J. Bodeau. Software Documentation and User's Manual or Fish-Impingement Sampling Design and Estimation Method Computer Programs. Argonne National Laboratory, 1977. ANL/ES-61.

REFERENCES

- 1.4- 1. C. G. J. Peterson. "The Yearly Immigration of Young Plaice into the Limfjord from the German Sea." Rep. Dan. Biol. Sta., Vol. 6, 1896, pp. 1-48.
- 1.4- 2. D. B. DeLury. "On the Planning of Experiments for the Estimations of Fish Populations." J. Fish. Res. Bd. Can., Vol. 8, 1951, pp. 281-307.
- 1.4- 3. D. G. Chapman. "Inverse, Multiple and Sequential Sample Censuses." Biometrics, Vol. 8, 1952, pp. 286-306.
- 1.4- 4. H. A. Regier and D. S. Robson. "Estimating Population Number and Mortality Rates." In The Biological Basis of Freshwater Fish Production, edited by S.D. Gerking, p. 31. Oxford: Blackwell; 1967.
- 1.4- 5. G. A. F. Seber. The Estimation of Animal Abundance and Related Parameters. New York: Hafner Press, 1973.
- 1.4- 6. W. E. Ricker. Computation and Interpretation of Biological Statistics of Fish Populations. Ottawa: Dept. of the Environment Fisheries and Marine Service, 1975.
- 1.4- 7. K. E. F. Watt. Ecology and Resource Management. New York: McGraw-Hill Book Company, 1968.
- 1.4- 8. R. A. Parker. "A Method for Removing the Effect of Recruitment on Peterson-Type Population Estimates." J. Fish. Res. Bd. Can., Vol. 12, 1955, pp. 447-450.
- 1.4- 9. W. S. Overton. "Estimating the Numbers of Animals in Wildlife Population." In Wildlife Management Techniques, edited by R. H. Giles Jr., p. 633. Washington, D.C.: Edwards Brothers, Inc., 1971.
- 1.4-10. D. G. Chapman. "Some Properties of the Hypergeometric Distribution With Applications to Zoological Sample Censuses." Univ. Calif. Publ. Stat., Vol. 1, 1951, pp. 131-160.
- 1.4-11. G. A. F. Seber. "The Effects of Trap Response on Tag-Recapture Estimates." Biometrics, Vol. 26, 1970, pp. 13-22.
- 1.4-12. D. S. Robson and H. A. Regier. "Sample Size in Peterson Mark-Recapture Experiments." Trans. Am. Fish. Soc., Vol. 93, 1964, pp. 215-226.
- 1.4-13. R. Jones. "3. Tagging: Theoretical Methods and Practical Difficulties." In Fish Population Dynamics, edited by J. A. Gulland, pp. 46-66. New York: John Wiley & Sons, 1977.
- 1.4-14. B. F. J. Manly and M. J. Parr. "A New Method of Estimating Population Size Survivorship and Birth Rate from Capture-Recapture Data." Trans. Soc. Brit. Ent., Vol. 18, 1968, pp. 81-89.
- 1.4-15. W. C. Latta. "Significance of Trap-Net Selectivity in Estimating Fish Population Statistics." Pap. Mich. Acad. Sci. Arts. Lett., Vol. 44, 1959, pp. 123-138.

REFERENCES

- 1.4-16. J. M. Hamley and H. A. Regier. "Direct Estimates of Gillnet Selectivity to Walleye (Stizostedion vitreum)." J. Fish. Res. Bd. Can., Vol. 30, 1973, pp. 817-830.
- 1.4-17. J. M. Hamely. "Review of Gillnet Selectivity." J. Fish. Res. Bd. Can., Vol. 32, 1975, pp. 1943-1969.
- 1.4-18. Z. E. Schnabel. "The Estimation of the Total Fish Population of a Lake." Am. Math. Mon., Vol. 45, 1938., pp. 348-352.
- 1.4-19. D. G. Chapman. "The Estimation of Biological Populations." Ann. Math. Stat., Vol. 25, 1954, pp. 1-15.
- 1.4-20. D. G. Chapman. "A Mathematical Study of Confidence Limits of Salmon Populations Calculated by Sample Tag Ratios." Int. Pac. Salmon Fish Comm. Bull., Vol. 2, 1948, pp. 67-85.
- 1.4-21. S. D. Gerking. "Vital Statistics of the Fish Population of Gordy Lake, Indiana." Trans. Am. Fish. Soc., Vol. 82, 1953, pp. 31-78.
- 1.4-22. G. M. Jolly. "Explicit Estimates from Capture-Recapture Data with Both Death and Immigration-Stochastic Model." Biometrika, Vol. 52, 1965, pp. 225-247.
- 1.4-23. G. A. F. Seber. "A Note on the Multiple-Recapture Census." Biometrika, Vol. 52, 1965, pp. 249-259.
- 1.4-24. R. M. Cormack. "The Statistics of Capture-Recapture Methods." Oceanogr. Mar. Bio. Ann. Rev., Vol. 6, 1968, pp. 455-506.
- 1.4-25. B. J. Rothschild. "5. Fishing Efforts." In Fish Population Dynamics, edited by J. A. Gullard, pp. 96-115. New York: John Wiley & Sons, 1977.
- 1.4-26. J. A. Gullard. "Manual of Methods for Fish Stock Assessment. Part 1. Fish Population Analysis." FAO (Food Agric. Organ., U.N.) Man. Fish. Sci., Vol. 4, 1969, pp. 1-154.
- 1.4-27. M. C. Pennington and M.D. Grosslein. "Accuracy of Abundance Indices Based on Stratified-Random Trawl Surveys." ICNAF Res. Doc., 78/VI/77, Serial No. 5264, Unpublished.
- 1.4-28. S. H. Clark. "Application of Bottom-Trawl Survey Data to Fish Stock Assessments." In Proceedings of the Northeast Fish and Wildlife Conference, 1978.
- 1.4-29. W. G. Cochran. Sampling Techniques. 2nd ed. New York: John Wiley & Sons, 1963.
- 1.4-30. W. E. Ricker. "Relation of 'Catch Per Unit Effort' to Abundance and Rate of Exploitation." J. Fish. Res. Bd. Can. Vol. 5, 1940, pp. 43-70.
- 1.4-31. J. A. Gullard. "Catch Per Unit Effort as a Measure of Abundance." Rapp. P.-V. Reun. Cons. Perm. Int. Explor. Mer., Vol. 155, 1964, pp. 8-14.

REFERENCES

- 1.4-32. D. L. Alverson and W. T. Pereyra. "Demersal Fish Explorations in the Northeastern Pacific Ocean - An Evaluation of Exploratory Fishing Methods and Analytical Approaches to Stock Size and Yield Forecasts." J. Fish. Res. Bd. Can., Vol. 26, 1969, pp. 1985-2001.
- 1.4-33. Orange and Rockland Utilities, Inc. Bowline Point Generating Station: Near-Field Effects of Once-Through Cooling System Operation on Hudson River Biota. Spring Valley, New York: Orange and Rockland Utilities, Inc., 1977.
- 1.4-34. J. Loesch, J. Bishop, A. Crowe, R. Kuckyr and P. Wagner. "Technique for Estimating Trawl Efficiency in Catching Brown Shrimp (Penaeus aztecus), Atlantic Croaker (Micropogon undulatus), and Spot (Leiostomus xanthurus)."
Gulf Research Reports, Vol. 5, 1976, pp. 29-33.
- 1.4-35. M. A. Kjelson and D. R. Colby. Estuarine Processes. New York: Academic Press Inc., 1977.
- 1.4-36. P. H. Leslie and D. H. S. Davis. "An Attempt to Determine the Absolute Number of Rats on a Given Area." J. Anim. Ecol., Vol. 8, 1939, pp. 94-113.
- 1.4-37. K. S. Ketchen. "The Use of Catch-Effort and Tagging Data in Estimating a Flatfish Population." J. Fish. Res. Bd. Can., Vol. 10, 1953, pp. 459-485.
- 1.4-38. I. N. Voevodin. "On the Problem of Estimating the Supply on Zanders in the North Caspian." (In Russian) Tr. Pervoi Vsesoyuznoi Nauchnoi Rybnokhozyaistvennoi Konferentsii, Vol. 2, 1938, pp. 3-12.
- 1.4-39. F. E. J. Fry. "Statistics of a Lake Trout Fishery." Biometrics, Vol. 5, 1949, pp. 27-67.
- 1.4-40. W. E. Ricker. "Derzhavin's Biostatistical Method of Population Analysis." J. Fish. Res. Bd. Can., Vol. 28, 1971, pp. 1666-1672.
- 1.4-41. D. G. Chapman. "Population Estimation Based on Change of Composition Caused by Selective Removal." Biometrika, Vol. 42, 1955, pp. 279-290.
- 1.4-42. R. H. Lander. "A Method of Estimating Mortality Rates from Change in Composition." J. Fish. Res. Bd., Vol. 19, 1962, pp. 159-168.
- 1.4-43. G. J. Paulik and D.S. Robson. "Statistical Calculations for Change-In-Ratio Estimators of Population Parameters." J. Wild. Man., Vol. 33, 1969, pp. 1-27.
- 1.4-44. R. S. Rupp. "Generalized Equation for the Ratio Method of Estimating Population Abundance." J. Wild. Man., Vol. 30, 1966, pp. 523-526.
- 1.4-45. G. I. Murphy. "Population Dynamics and Population Estimation." In Inland Fisheries Management, edited by A. Calhoun, pp. 1-18. California: The Resources Agency, 1966.

REFERENCES

- 1.4-46. D. H. Cushing. The Uses of Echo Sounding for Fishermen. London: Her Majesty's Stationery Office, Ministry of Agriculture, Fisheries, and Food, 1963.
- 1.4-47. R. G. Urick. Principles of Underwater Sound. New York: McGraw-Hill Book Co., 1975.
- 1.4-48. R. G. Urick. Principles of Underwater Sound for Engineers. New York: McGraw-Hill Book Co., 1967.
- 1.4-49. J. Lozow and J. Suomala Jr. The Application of Hydroacoustic Methods for Aquatic Biomass Measurements. A Note on Echo Envelope Sampling and Integration. Cambridge, Mass: Charles Stark Draper Lab., 1971.
- 1.4-50. J. B. Suomala Jr. A Short Note on the Applicability of Hydroacoustic Methods for Demersal Fish Counting and Abundance Estimations. International Commission for the Northwest Atlantic Fisheries, 1975. ICNAF Res. Doc. 75/95.
- 1.4-51. K. I. Yudanov, J. B. Suomala Jr., V. M. Vorbyov, and K. A. Smith. The Application of Hydroacoustic Methods for Aquatic Biomass Measurements. Preliminary Report of the First Joint USA - USSR Hydroacoustic Experiment in the ICNAF Convention Area, 11 March -15 April 1974. Cambridge, Mass: Charles Stark Draper Lab., 1974. C-4117.
- 1.4-52. J. Lozow and J. Suomala Jr. The Application of Hydroacoustic Methods for Aquatic Biomass Measurements. An Informal Report to the Northeast Fisheries Center on the Present State of Hydroacoustic Methods and Recommendations Leading to Their Practical Application. Cambridge, Mass: Charles Stark Draper Lab., 1973. C-3948.
- 1.4-53. D. Hamilton, J. Lozow, J. Suomala Jr., and R. Werner. The Application of Hydroacoustic Methods for Aquatic Biomass Measurement. A Hydroacoustical Measurement Experiment to Examine Target Quantification Methods. Cambridge, Mass: Charles Stark Draper Lab., 1973. R-759.
- 1.4-54. J. Lozow and J. Suomala Jr. A Short Note on Hydroacoustical Echo Signal Components and Their Effect on Fish Target Density Estimations. Cambridge, Mass: Charles Stark Draper Lab., 1976.
- 1.4-55. A. Orlowski. Echo Sounder and Camera Control Unit. Gdynia, Poland: Polish/UNSF Highseas Fisheries Research Project, MIR, 1972.
- 1.4-56. P. W. Osborne, E. Hoffman, J. V. Merriner, G. E. Richards, and R. W. Lovelady. "Automatic Electronic Fish Tracking System." Oceans, Vol. 76, 1976, pp. 15A-1 - 15A-6.
- 1.4-57. T. B. Thorson, G. F. Esterberg, and J. H. Johnson. "Ultrasonic Shark Tag Monitoring System." Univ. Nebr. Dept. Zool. Tech. Rept., 1969.

REFERENCES

- 1.4-58. D. G. Pincock, D. McG. Luke, D. W. Church, and A. B. Stasko. "An Automatic Monitor for Detecting and Recording Passage of Transmitter-Fitted Fish." Fish. Mar. Serv. Res. Dev. Tech. Rept., Vol. 499, 1974, pp. 1-59.
- 1.4-59. A. D. Hawkins, D. N. MacLennan, G. G. Urguhart, and C. Robb. "Tracking Cod Gadus morhua L. in a Scottish Sea Loch." J. Fish. Biol., Vol. 6, 1974, pp. 225-236.
- 1.4-60. B. Holland, I. Mohus, and R. Bernsten. "Devices and Results 1974. Sintef and Reguleringssteknikk. 7034 NTN- Trondheim, Norway." Fish Telen., Rept. 6, STF 48, A 740 49, 1974, pp. 1-89.
- 1.4-61. C. Groot. "Migrations of Yearling Sockeye Salmon (Oncorhynchus nerka) as Determined by Time-Lapse Photography of Sonar Observations." J. Fish. Res. Bd. Can., Vol. 29, 1972, pp. 1431-1444.
- 1.4-62. K. Braithwaite. "A Sonar Fish Counter." J. Fish. Biol., Vol. 3, 1971, pp. 73-82.
- 1.4-63. R. E. Throne and J. J. Dawson. "An Acoustic Estimate of the Escape-ment of Sockeye Salmon (Oncorhynchus nerka) into Lake Washington in 1971." J. Fish. Res. Bd. Can., Vol. 31, 1974, pp. 222-225.
- 1.5-64. G. L. Hergenrader and A. D. Hasler. "Influence of Changing Seasons on Schooling Behavior of Yellow Perch." J. Fish. Res. Bd. Can., Vol. 25(4), 1968, pp. 711-716.
- 1.4-65. J. W. Mullar and R. L. Applegate. "Use of an Echosounder in Measuring Distribution of Reservoir Fishes." Paper Bur. Sport Fish. Wildl., No. 19, 1969, pp. 1-16.
- 1.4-66. J. R. M. Kelso, E. E. Pickett, and R. G. Dowd. "A Digital Echo-Counting System Used in Determining Abundance of Freshwater Pelagic Fish in Relation to Depth." J. Fish. Res. Bd. Can., Vol. 31, 1974, pp. 1101-1104.
- 1.4-67. D. W. Chapman. "Acoustic Estimates of Pelagic Ichthyomass in Lake Tanganyika With an Inexpensive Echo Sounder." Trans. Am. Fish. Soc., Vol. 1976(5), 1976, pp. 581-587.
- 1.4-68. B. F. White. "A Transportable Acoustic Fish Census System for Lake and River Studies." Oceans, Vol. 76, 1976, 13E1 - 13E6.
- 1.4-69. R. L. Wyman, J. Payne, and D. Andrews. "Hydroacoustic Assessment of Fish Populations." Twentieth Conference of Great Lakes Research: Abstracts. International Association for Great Lakes Research, 1977.
- 1.4-70. G. L. Hergenrader and A. D. Hasler. "Diel Activity and Vertical Distribution of Yellow Perch (Perca flavescens) Under the Ice." J. Fish. Res. Bd. Can., Vol. 23, 1966, pp. 499-509.
- 1.4-71. D. H. Cushing. "Direct Estimation of a Fish Population Acoustically." J. Fish. Res. Bd. Can., Vol. 25(11), 1968, pp. 2349-2364.

REFERENCES

- 1.4-72. R. E. Thorne. "Digital Hydroacoustic Data-Processing System and Its Application to Pacific Lake Stock Assessment in Port Susan, Washington." Fish. Bull., Vol. 71(3), 1973, pp. 837-843.
- 1.4-73. O. A. Mathisen and E. P. Nunnallee. "Acoustic Stock Estimation as a Total in Management of Sockeye Salmon (*Oncorhynchus nerka* Walbaum)." In EIFAC/FAO Symposium on Methodology for the Survey, Monitoring and Appraisal of Fishery Resources in Lakes and Large Rivers. Avamore, Scotland, 1974.
- 1.4-74. T. Ishi. "Studies on Counting the Echo Pattern of Individual Fish by Pattern Analysis Method - I. Total System and Hard Wear [sic]." Bull. Jap. Soc. Scient. Fish., Vol. 42(3), 1976, pp. 251-264.
- 1.4-75. R. P. Hewitt, P. E. Smith, and J. C. Brown. "Developments and Use of Sonar Mapping for Pelagic Stock Assessment in California Current Area." Fish. Bull., Vol. 74(2), 1976, pp. 281-300.
- 1.4-76. J. S. M. Rusby. "Long Range Survey of a Herring Fishery by Side-Scan Sonar." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 7-14. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.
- 1.4-77. W. I. Dunn and S. T. Forbes. "The Use of the Aberdeen Fish Counter as a 400 Channel Echo Integrator." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 159-161. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.
- 1.4-78. H. Yamanaka, M. Yukinawa, J. Morita, and M. Nishimura. "Acoustic Fish Counting System For Tuna and Related Species." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 174-184. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.
- 1.4-79. W. C. Acker. "Acoustic Assessment of North Pacific Salmon Stocks." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 189-195. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.
- 1.4-80. K. A. Johannesson and A. N. Robles. "Echo Surveys of Peruvian Anchoveta." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 237-244. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 10, 1977.
- 1.4-81. K. I. Yudanov, J. B. Suomala Jr., and K. A. Smith. "The Application of Hydroacoustic Methods for Aquatic Biomass Measurements. Preliminary Report of the Last Joint USA-USSR Hydroacoustic Experiment in the East Central Atlantic - 14 May to 4 June 1976. Cambridge, Mass.: Charles Stark Draper Lab., 1976. C-4744.
- 1.4-82. O. Nakken and K. Olsen. "Target Strength Measurement of Fish." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 52-69. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.

REFERENCES

- 1.4-83. J. H. Gee and P. A. Gee. "Alteration of Buoyancy by Some Central American Stream Fishes, and a Comparison with North American Species." Can. J. Zool., Vol. 54(2), 1976, pp. 386-391.
- 1.8-84. H. V. Reyer. "The Role of the Swim-Bladder in Vertical Movement of Fishes." Biol. Behav., Vol. 2, 1977, pp. 109-128.
- 1.4-85. K. Olsen. "Some Experiments on the Effect on Target Strength of Fish Undertaking Vertical Migrations." ICES Gear and Behavior Committee, Vol. 42, 1976. CM 1976/B.
- 1.4-86. R. W. G. Haslett. "Automatic Plotting of Polar Diagrams of Target Strength of Fish in Roll, Pitch, and Yaw." In Hydroacoustics in Fisheries Research, edited by A. R. Margetts, pp. 74-81. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.
- 1.4-87. S. Olsen, O. Chruichshank, and K. Hansen. "Target Strength of Porcupine Fish - An Outstanding Deviation from the Established Target Strength-Length Regression." In Hydroacoustics in Fisheries Research, edited by A. R. Margett, pp. 82-83. Denmark: Rapp. P-V. Reun. Cons. Int. Explor. Mer, Vol. 170, 1977.
- 1.4-88. R. H. Love. "Maximum Side-Aspect Target Strength of an Individual Fish." J. Acoust. Soc. Amer., Vol. 46, No. 3, Pt. 2, 1969, pp. 746-752.
- 1.4-89. R. H. Love. "Dorsal-Aspect Target Strength of an Individual Fish." J. Acoust. Soc. Am., Vol. 49, No. 3, Pt. 2, 1971, pp. 816-823.
- 1.4-90. R. H. Love. "Measurements of Fish Target Strength: A Review." Fish. Bull., Vol. 69(4), 1971, pp. 703-715.
- 1.4-91. R. H. Love. "Target Strength of an Individual Fish at Any Speed." J. Acoust. Soc. Am., Vol. 62(6), December, 1977, pp. 1397-1403.
- 1.4-92. J. R. M. Kelso and C. K. Minns. "Summer Distribution of the Near-shore Fish Community Near a Thermal Generating Station as Determined by Acoustic Census." J. Fish. Res. Bd. Can., Vol. 32, 1976, pp. 1409-1418.
- 1.4-93. K. L. Zankel, L. H. Bongers, T. T. Polgar, W. A. Richkus, and R. E. Thorne. "Size and Distribution of the 1974 Striped Bass Spawning Stock in the Potomac River." Potomac River Fish. Prog. Rept. Ser., 1975. PRFP-75-1.

REFERENCES

- 1.5- 1. J. P. Lawler. "Physical Measurements -- Their Significance in the Prediction of Entrainment Effects." Presented at The Third National Workshop on Entrainment and Impingement, New York, February 2-4, 1976.
- 1.5- 2. J. P. Lawler. "Differential Equations: Stream and Estuarine Analysis Manual." Tenth Summer Institute in Water Pollution Control. New York: Manhattan College, 1964.
- 1.5- 3. J. P. Lawler. "Mathematical Analysis of Estuarine Pollution: Engineering Aspects of Marine Waste Disposal." Environmental Health Science Engineering Center. Cincinnati, Ohio, 1965.
- 1.5- 4. Quirk, Lawler & Matusky Engineers. Hudson River Waste Quality and Waste Assimilation Capacity Study. Report to the New York State Department of Health, 1970.
- 1.5- 5. J. P. Lawler. "Receiving Water Temperature Distributions from Power Plant Thermal Discharges: An Estuarine Model." In Proceedings 17th Southern Water Resources & Pollution Control Conference. Chapel Hill, N.C., 1968.
- 1.5- 6. J. P. Lawler and K. A. Abood. "Thermal State of the Hudson River and Potential Changes." In Proceedings Second Symposium on Hudson River Ecology, Sterling Forest Conference Center, Tuxedo, New York, 1969.
- 1.5- 7. Quirk, Lawler & Matusky Engineers. Effect of Indian Point Cooling Water Discharge on Hudson River Temperature Distribution. Report to Consolidated Edison Company of New York, Inc., 1968.

REFERENCES

- 1.6- 1. M. P.. Weinstein, S.L. Weiss, R.G. Hodson, and L.R. Gerry. "Retention of Three Species of Postlarval Fishes in an Intensively Flushed Tidal Estuary, Cape Fear River, North Carolina." Estuar. Coast. Mar. Sci., forthcoming.
- 1.6- 2. P. A. Hackney and J.C. Webb. "A Method for Determining Growth and Mortality Rates of Ichthyoplankton." In Proceedings Fourth National Workshop on Entrainment and Impingement, pp. 115-124 Melville, N.Y.: Ecological Analysts, Inc., 1977.
- 1.6- 3. P. A. Hackney. "Calculating Survival Rate, Biomass Production, Growth Rate, and Assessing Entrainment of Lacustrine Ichthyoplankton." Unpublished Manuscript, 1977.
- 1.6- 4. W. G. Pearcy. "Ecology of an Estuarine Population of Winter Flounder, Pseudopleuronectes americanus (Waldbaum)." Bull. Bingham Oceanogr. Collect., Vol. 18, 1962, pp. 1-78.
- 1.6- 5. W. E. Ricker. Computation and Interpretation of Biological Statistics of Fish Populations. Ottawa: Dept. of the Environment Fisheries and Marine Service, 1975.

REFERENCES

- 1.7- 1. P. H. Leslie. "On the Use of Matrices in Certain Population Mathematics." Biometrika, Vol. 33, 1945, pp. 183-212.
- 1.7- 2. P. H. Leslie. Some Further Notes on the Use of Matrices in Population Mathematics." Biometrika, Vol 35, 1948, pp. 212-245.
- 1.7- 3. S. R. Searle. Matrix Algebra for the Biological Sciences: Including Applications in Statistics. New York: John Wiley & Sons, Inc., 1966.
- 1.7- 4. J. M. Emlen. Ecology: An Evolutionary Approach. Reading, Mass.: Addison-Wesley, 1973.
- 1.7- 5. D.S. Vaughan and S.B. Saila. "A Method for Determining Mortality Rates Using the Leslie Matrix." Trans. Am. Fish. Soc., Vol. 105, 1976, pp. 380-383.
- 1.7- 6. D. S. Vaughan. "Confidence Intervals on Mortality Rates Based on the Leslie Matrix." In Proceedings of the Conference on Assessing the Effects of Power-Plant-Induced Mortality on Fish Populations, pp. 128-150. New York: Pergamon Press, 1977.
- 1.7- 7. T. J. Horst. "Effects of Power Station Mortality in Fish Population Stability in Relationship to Life History Strategy." In Proceedings of the Conference on Assessing the Effects of Power-Plant-Induced Mortality on Fish Populations, pp. 297-310. New York: Pergamon Press, 1977.
- 1.7- 8. R. N. MacArthur and E. O. Wilson. The Theory of Island Biogeography. Princeton, N.J.: Princeton University Press, 1967.
- 1.7- 9. S. B. Saila and E. Lorda. "Sensitivity Analysis Applied to a Matrix Model of the Hudson River Striped Bass Population." In Proceedings of a Conference on Assessing the Effects of Power-Plant-Induced Mortality on Fish Populations, pp. 311-322. New York: Pergamon Press, 1977.

REFERENCES

- 1.8- 1. R. J. H. Beverton and S. J. Holt. "On the Dynamics of Exploited Fish Populations." Fishery Invest. Lond., Ser. 11, Vol. 20, 1957.
- 1.8- 2. A. H. Weatherley. Growth and Ecology of Fish Populations. New York: Academic Press, 1972.
- 1.8- 3. R. E. Foerster. "The Relation of Lake Population Density to Size of Young Sockeye Salmon (Oncorhynchus nerka)."
J. Fish. Res. Bd. Can., Vol. 6, 1944, pp. 267-280.
- 1.8- 4. A. W. Eipper. "Growth, Mortality Rates and Standing Crops of Trout in New York Farm Ponds." Cornell Univ. Agr. Exp. Sta., Mem. 388, 1964, pp. 67.
- 1.8- 5. O. A. Mathisen. "Growth of Sockeye Salmon in Relation to Abundance in the Kvichak District, Bristol Bay, Alaska." Fish. Der. Skr. Ser. Hav. Unders., Vol. 15, No. 3, 1969, pp. 172-185.
- 1.8- 6. T. A. Lindstrom, A. Fagerstrom, and K. J. Gustafson. "Fishing Pressure, Growth, and Recruitment in a Small High Mountain Lake." Inst. Fresh-water Res. Drottningholm Rept., Vol. 50, 1970, pp. 100-115.
- 1.8- 7. M. C. Healey. "Dynamics of Exploited Whitefish Populations and Their Management with Special Reference to the Northwest Territories." J. Fish. Res. Bd. Can., Vol. 32, 1975, pp. 427-448.
- 1.8- 8. J. Van Oosten. "Life History of the Lake Herring (Leucichthys artedii LeSueur) of Lake Huron, as Revealed by Its Scales, With a Critique of the Scale Method." Bull. U.S. Bur. Fish., Vol. 44, 1929, pp. 265-448.
- 1.8- 9. H. A. Schuck. "Relationship of Catch to Changes in Population Size of New England Haddock." Biometrics, Vol. 5, No. 3, 1949, pp. 213-231.
- 1.8-10. R. Hile. "Body-Scale Relation and Calculation of Growth in Fishes." Trans. Am. Fish. Soc., Vol. 99, 1970, pp. 468-474.
- 1.8-11. G. G. Simpson, A. Roe, and R. C. Lewontin. Quantitative Zoology. New York: Harcourt Brace Jovanovich, 1960.
- 1.8-12. F. J. Rohlf and R. R. Sokal. Statistical Tables. San Francisco: W. H. Freeman and Co., 1969.
- 1.8-13. W. E. Ricker. "Effects of Size-Selective Mortality and Sampling Bias on Estimates of Growth, Mortality, Production, and Yield." J. Fish. Res. Bd. Can., Vol. 26, 1969, pp. 479-541.
- 1.8-14. W. E. Ricker. Computation and Interpretation of Biological Statistics of Fish Populations. Ottawa: Dept. of the Environment Fisheries and Marine Service, 1975.

REFERENCES

- 1.8-15. R. J. H. Beverton and S. J. Holt. "A Review of Methods for Estimating Mortality Rates in Exploited Fish Populations With Special Reference to Sources of Bias in Catch Sampling." Rapp. P-V. Reun. Cons. Perm. Int. Explor. Mer, Vol. 140, Pt. 1, 1956, pp. 67-83.
- 1.8-16. C. J. Krebs. Ecology: The Experimental Analysis of Distribution and Abundance. New York: Harper & Row, 1972.
- 1.8-17. W. E. Ricker. "Mortality Rates in Some Little-Exploited Populations of Freshwater Fishes." Trans. Am. Fish. Soc., Vol. 77, 1949, pp. 114-128.
- 1.8-18. D. H. Cushing. Marine Ecology and Fisheries. New York: Cambridge University Press, 1975.
- 1.8-19. W. G. Pearcy. "The Ecology of an Estuarine Population of Winter Flounder, Pseudopleuronectes americanus (Waldbaum)." Bull. Bingham Oceanogr. Coll., Vol. 18, Art. 1, 1962, pp. 5-77.
- 1.8-20. C. B. Dew and J. H. Hecht. "Observations on the Population Dynamics of Atlantic Tomcod (Microgadus tomcod) in the Hudson River Estuary." In Hudson River Ecology Fourth Symposium on Hudson River Ecology, Paper 25. Hudson River Environmental Soc., 1976.
- 1.8-21. L. A. Walford. "A New Graphic Method of Describing the Growth of Animals." Biol. Bull., Vol. 90, No. 2, 1946, pp. 141-147.
- 1.8-22. E. Ford. "An Account of the Herring Investigations Conducted at Plymouth During the Years from 1924-1933." J. Mar. Biol. Assoc., N.S., Vol. 19, 1933, pp. 305-384.
- 1.8-23. W. E. Ricker. "Handbook of Computations for Biological Statistics of Fish Populations." Bull. Fish. Res. Bd. Can., No. 119, 1958.
- 1.8-24. W. F. Thompson and F. N. Bell. "Biological Statistics of the Pacific Halibut Fishery. 2. Effect of Changes in Intensity Upon Total Yield and Yield per Unit of Gear." Rep. Int. Fish. Comm., Vol. 8, 1934, pp. 49.
- 1.8-25. W. E. Ricker. "Big Effects from Small Causes: Two Examples from Fish Population Dynamics." J. Fish. Res. Bd. Can., Vol. 20, 1963, pp. 257-264.
- 1.8-26. R. P. Silliman. "Studies on the Pacific Pilchard or Sardine (Sardinops caerulea). 5. A Method of Computing Mortalities and Replacements." U.S. Fish. Wild. Serv. Spec. Sci. Rep., No. 24, 1943.
- 1.8-27. R. J. H. Beverton. "Notes of the Use of Theoretical Models in the Study of the Dynamics of Exploited Fish Populations." U.S. Fish. Lab. Beaufort, N.C., Misc. Contr., No. 2, 1954.

REFERENCES

- 1.8-28. J. E. Paloheimo. "Studies on Estimation of Mortalities. 1. Comparison of a Method Described by Beverton and Holt and a New Linear Formula." J. Fish. Res. Bd. Can., Vol. 18, 1961, pp. 645-662.
- 1.8-29. R. H. Lander. "A Method of Estimating Mortality Rates from Change in Composition." J. Fish. Res. Bd. Can., Vol. 19, No. 1, 1962, pp. 159-168.
- 1.8-30. D. G. Chapman and G. I. Murphy. "Estimates of Mortality and Population from Survey-Removal Records." Biometrics, Vol. 21, 1965, pp. 921-935.
- 1.8-31. G. A. F. Seber. The Estimation of Animal Abundance and Related Parameters. New York: Hafner Press, 1973.
- 1.8-32. G. I. Murphy. "A Solution of the Catch Equation." J. Fish. Res. Bd. Can., Vol. 22, 1965, pp. 191-202.
- 1.8-33. K. E. F. Watt. Ecology and Resource Management. New York: McGraw-Hill Publications, 1968.
- 1.8-34. J. T. Tanner. "Effects of Population Density on Growth Rates of Animal Populations." Ecology, Vol. 47, No. 5, 1966, pp. 733-743.
- 1.8-35. L. B. Slobodkin. Growth and Regulation of Animal Populations. New York: Holt, Rinehart and Winston, 1961.
- 1.8-36. E. O. Wilson and W. H. Bossert. A Primer of Population Biology. Sunderland, Mass.: Sinauer Assoc., Inc., 1971.
- 1.8-37. A. C. Simpson. "The Fecundity of the Plaice." Fish. Invest., London, Vol. 2(17), 1951.
- 1.8-38. R. J. Mansueti. "Movements, Reproduction, and Mortality of the White Perch, Roccus americanus, in the Patuxent Estuary, Maryland." Chesapeake Sci., Vol. 2 (3-4), 1961, pp. 142-205.
- 1.8-39. T. B. Bagenal. "The Ecological and Geographical Aspects of the Fecundity of Plaice." J. Mar. Biol. Assoc. U. K., Vol. 46, 1966, pp. 161-186.
- 1.8-40. J. Loesch. "A Study of the Blueback Herring, Alosa aestivalis (Mitchill), in Connecticut Waters." Ph.D. Thesis, Univ. of Connecticut, 1969.
- 1.8-41. G. W. Kissil. "Contributions to the Life History of the Alewife, Alosa pseudoharengus (Wilson) in Connecticut." Ph. D. Thesis, Univ. of Connecticut, 1969.
- 1.8-42. J. G. Holsapple and L. E. Foster. "Reproduction of White Perch in the Lower Hudson River." N. Y. Fish Game J., Vol. 22(2), 1975, pp. 122-127.

REFERENCES

- 1.8-43. B. A. Lehman. "Fecundity of Hudson River Shad." U. S. Fish Wildl. Serv., Res. Rept., No. 33, 1953, pp. 18-28.
- 1.8-44. W. S. Davis. "Ova Production of American Shad in Atlantic Coast Rivers." U.S. Fish. Wildl. Serv., Res. Rept., No. 49, 1957, pp. 1-5.
- 1.8-45. R. M. Lewis and R. R. Bonner Jr. "Fecundity of the Striped Bass, Roccus saxatilis (Walbaum)." Trans. Am. Fish. Soc., Vol. 95(3), 1966, pp. 328-331.
- 1.8-46. A. N. Sheri and G. Power. "Reproduction of White Perch, Roccus americanus, in the Bay of Quinte, Lake Ontario." J. Fish. Res. Bd. Can., Vol. 25(10), 1968, pp. 2225-2231.
- 1.8-47. S. H. Taub. "Fecundity of the White Perch." Prog. Fish-Cult., 1969, pp. 166-168.
- 1.8-48. J. T. McFadden. "Numerical Changes and Population Regulation in Brook Trout, Salvelinus fontinalis." J. Fish. Res. Bd. Can., Vol. 24(7), 1967, pp. 1425-1459.
- 1.8-49. C. M. Breder Jr. and D. E. Rosen. Modes of Reproduction in Fishes. New York: Natur. Hist. Press, 1966.
- 1.8-50. H. C. Boyar and R. A. Clifford. "An Automatic Device for Counting Dry Fish Eggs." Trans. Am. Fish. Soc., Vol. 96(3), 1967, pp. 361-363.
- 1.8-51. F. E. Perkins and V. C. Anthony. "A Note on the Fecundity of Herring (Clupea harengus harengus) from Georges Bank, the Gulf of Maine, and Nova Scotia." ICNAF Res. Doc. 69/60, 1969.
- 1.8-52. T. B. Bagenal. "Eggs and Early Life History, Part I. Fecundity." In Methods for Assessment of Fish Production in Fresh Waters. IBP Handbook No. 3, edited by W. E. Ricker. Oxford, Engl.: Blackwell Sci. Publ., 1971.
- 1.8-53. T. B. Bagenal. "A Short Review of Fish Fecundity." In The Biological Basis of Freshwater Fish Production, edited by S. D. Gerking. New York: John Wiley and Sons, 1967.
- 1.8-54. G. L. Kesteven (ed.). "Manual of Field Methods in Fisheries Biology." FAO Man. Fish. Sci., No. 1, 1960.
- 1.8-55. G. V. Nikolsky. The Ecology of Fishes. New York: Academic Press, 1963.
- 1.8-56. R. A. Fisher. The Genetical Theory of Natural Selection. Oxford: Clarendon Press, 1930.

REFERENCES

- 1.8-57. R. A. Fisher. The Genetical Theory of Natural Selection. 2nd. ed. New York: Dover, 1958.
- 1.8-58. E. R. Pianka. Evolutionary Ecology. New York: Harper and Row, 1978.
- 1.8-59. L. B. Slobodkin. "Summary and Discussion of the Symposium." In Fish Stocks and Recruitment, edited by B. B. Parrish, pp. 7-14. Denmark: Rapp. P-V. Reun. Cons. Perm. Int. Explor. Mer, Vol. 164, 1970.

REFERENCES

- 1.9- 1. J. T. McFadden, Texas Instruments, Inc., and Lawler, Matusky & Skelly Engineers. "Influence of the Proposed Cornwall Pumped Storage Project and Steam Electric Generating Plants on the Hudson River Estuary with Emphasis on Striped Bass and Other Fish Populations, Revised." Report to Consolidated Edison Company of New York, Inc., 1978.
- 1.9- 2. W. E. Ricker. "Stock and Recruitment." J. Fish. Res. Bd. Can., Vol. 11, 1954, pp. 559-623.
- 1.9- 3. R. J. H. Beverton and S. H. Holt. On the Dynamics of Exploited Fish Populations. London: H. M. Stationery Off., Fish Invest. Ser. 2, 1957.
- 1.9- 4. J. P. Lawler and T. L. Englert. "Models Useful for the Estimation of Equilibrium Population Reduction due to Power Plant Cropping." In Fourth National Workshop on Entrainment and Impingement, edited by L. D. Jensen, pp. 103-113. Melville, New York: Ecological Analysts, Inc., 1978.
- 1.9- 5. R. M. Peterman. "A Simple Mechanism that Causes Collapsing Stability Regions in Exploited Salmonid Populations." J. Fish. Res. Bd. Can., Vol. 34, 1977, pp. 1130-1142.
- 1.9- 6. G. J. Paulik. "Studies of the Possible Form of the Stock and Recruitment Curve." In Fish Stocks and Recruitment, edited by B. B. Parrish, pp. 302-315. Denmark: Rapp. P.-V. Reun. Cons. Perm. Int. Explor. Mer, 1973.
- 1.9- 7. C. S. Holling. "Resilience and Stability of Ecological Systems." Ann. Rev. Ecol. System., Vol. 4, 1973, pp. 1-23.
- 1.9- 8. C. S. Holling. "The Functional Response of Predators to Prey Density and its Role in Mimicry and Population Regulation." Mem. Entomol. Soc. Can., Vol. 45, 1965, pp. 1-60.
- 1.9- 9. W. W. Murdoch and A. Oaten. "Predation and Population Stability." Adv. Ecol. Res., Vol. 9, 1975, pp. 1-32.
- 1.9-10. W. E. Ricker and H. D. Smith. "A Revised Interpretation of the History of the Skeena River Sockeye Salmon, (Onchorhynchus nerka). " J. Fish. Res. Bd. Can., Vol. 32, 1975, pp. 1369-1381.
- 1.9-11. G. A. Rounsefell. "Factors Causing Decline in Sockeye Salmon of Karluk River, Alaska." U. S. Fish. Wildl. Serv., Fish. Bull., Vol. 58, 1958, pp. 83-169.
- 1.9-12. W. E. Ricker. "Computation and Interpretation of Biological Statistics of Fish Populations." Fish. Res. Bd. Can. Bull., Vol. 191, 1975, 382p.
- 1.9-13. R. Jones. "The Analysis of Trawl Haul Statistics with Particular Reference to the Estimation of Survival Rates." Rapp. P.-V. Reun. Cons. Perm. Int. Explor. Mer, Vol. 140, 1956, pp. 30-39.
- 1.9-14. W. R. Neslon, M. C. Ingram, and W. E. Schaff. "Larval Transport and Year-Class Strength of Atlantic Menhaden, Brevoortia tyrannus." Fish. Bull., Vol. 75, 1977, pp. 23-41.

REFERENCES

- 1.10.1. Royama, T. "Population Persistence and Density Dependence." Ecol. Monogr. Vol. 47, No. 1, 1977.
- 1.10-2 J. P. Lawler. "Testimony of John P. Lawler Ph.D., Quirk, Lawler & Matusky Engineers, on The Effect of Entrainment at Indian Point on the Population of the Hudson River Striped Bass, April 5, 1972." In the matter of Consolidated Edison Company of New York, Inc. (Indian Point Station, Unit No. 2). AEC Docket 50-247.
- 1.10-3. S. W. Christensen, D. L. DeAngelis, and A. G. Clark. "Development of a Stock-Progeny Model for Assessing Power Plant Effects on Fish Populations." In Proceedings of the Conference on Assessing the Effects of Power-Plant-Induced Mortality on Fish Populations, edited by W. Van Winkle, pp. 196-226. New York: Pergamon Press, 1977.
- 1.10-4. J. P. Lawler. "Lecture on Techniques Employed in Modelling Power Plant Impact on Aquatic Life." Westinghouse International School for Environmental Management, Colorado State University. New York: Quirk Lawler & Matusky Engineers [Lawler, Matusky & Skelly Engineers], 1973.
- 1.10-5. Lawler, J. P. "Effect of Entrainment and Impingement at Cornwall on the Hudson River Striped Bass Population." (Testimony) U.S. Federal Power Commission (Now the Federal Energy Regulatory Commission) Project No. 2338, October 1974.
- 1.10-6. Lawler, Matusky & Skelly Engineers. "Report on Development of a Real-Time, Two-Dimensional Model of the Hudson River Striped Bass Population." Prepared for Consolidated Edison Company of New York, Inc., 1975.
- 1.10-7. A. H. Eraslan, et al. A Computer Simulation Model for the Striped Bass Young-of-the-Year Population in the Hudson River. Oak Ridge, Tennessee: Oak Ridge Nat. Lab., Dec. 1976. ORNL/NURGE-8, ESD-766.

BIBLIOGRAPHY

Carlson, F., and J. McCann. "Hudson River Fisheries Investigations, 1965-1968: Evaluation of a Proposed Pumped Storage Project at Cornwall, New York in Relation to the Fish in the Hudson River." Hudson River Policy Committee, New York State Conservation Dept. and Consolidated Edison Company of New York, Inc., 1969.

Humphries, E. T., and K. B. Cumming. "An Evaluation of Striped Bass Fingerling Culture." Trans. Amer. Fish. Soc., Vol. 102(1), 1973, pp. 13-20.

Lewis, M., and R. R. Bonner Jr. "Fecundity of the Striped Bass, Roccus saxatilis (Walbaum)." Trans. Amer. Fish. Soc., Vol. 95(3), 1966, pp. 328-331.

Mansueti, J. "Eggs, Larvae and Young of the Striped Bass, Roccus saxatilis." Md. Dept. Res. Ed. Contrib., Vol. 112, 1958, 33p.

Merriman, D. "Studies on the Striped Bass, Roccus saxatilis of the Atlantic Coast." Fish. Bull., U.S. Fish. Wildl. Serv., No. 50, 1941, pp. 1-77.

Pearson, J. S. "The Life History of the Striped Bass or Rockfish, Roccus saxatilis (Walbaum)." Bull. U.S. Bur. Fish., Vol. 49(28), 1938, pp. 825-851.

Rathjen, F., and L. C. Miller. "Aspects of the Early Life History of the Striped Bass (Roccus saxatilis) in the Hudson River." N.Y. Fish. Game J., Vol. 4 (1), 1957, pp. 43-68.

Trent, L. and W. W. Hassler. "Gill Net Selection, Migration, Size and Age Composition, Sex Ratio, Harvest Efficiency, and Management of Striped Bass in the Roanoke River, North Carolina." Chesapeake Sci., Vol. 9(4), 1968, pp. 217-232.

Vladykov, V. D., and D. H. Wallace. "Studies of the Striped Bass, Roccus saxatilis (Walbaum), with Special Reference to the Chesapeake Bay Region During 1936-1938." Bull. Bingham. Oceanogr. Coll., Vol. 14, No. 1, 1952, pp. 132-177.

Appendix 1A

COMPUTER PROGRAM LIFETAB COMPUTATION OF POPULATION SURVIVAL AND REPRODUCTIVE CHARACTERISTICS

PROGRAM LANGUAGE

HP-3000 BASIC

PURPOSE AND METHOD OF OPERATION

This program computes the proportion (L_x) of an initial cohort surviving to each age x , given annual mortality rates (Q_x), using the equation:

$$L_x = L_{x-1} - (Q_{x-1} L_{x-1}), \quad (1A-1)$$

where L_0 is defined as 1.0. It also calculates age-specific fertility (M_x) using the equation:

$$M_x = PA_x (F_x/2) \quad (1A-2)$$

where

A_x = proportion of females sexually mature at age x

$F_x/2$ = age-specific fecundity adjusted to reflect female offspring only
(assumes 1:1 sex ratio for eggs)

$$\text{and } P = \frac{\sum_{x=0}^{\infty} L_x M_x}{\sum_{x=0}^{\infty} L_x A_x (F_x/2)} = \frac{R_0}{\sum_{x=0}^{\infty} L_x A_x (F_x/2)} \quad (1A-3)$$

which is a survival term reflecting survival from spawned egg to a three to four month old (age 0) juvenile, fully recruited to the 1/4 inch square mesh seines and trawls used in sampling.

*Documentation of a program developed for analyses of reproductive value, Section 1.8.5.3.

Note that for a given survival (L_x) schedule, maturation (A_x) schedule, and egg complement (F_x), the egg to juvenile survival term (P) is dictated by the value chosen for the net reproduction rate (R_0), where:

$$R_0 = \sum_{x=0}^{\infty} L_x M_x \quad (1A-4)$$

If the population is assumed to be in stable equilibrium, a value of $R_0 = 1.0$ is appropriate; if the population is increasing or decreasing, a value of $R > 1.0$ or $R < 1.0$, respectively, should be chosen as input.

Program LIFETAB also calculates the gross reproductive rate as the sum of the M_x values, which represents the total number of female offspring that would be produced by an average female in the absence of mortality. Generation time (T), representing the average time period between birth of parents and birth of offspring, is calculated as:

$$T = \frac{\sum_{x=0}^{\infty} x L_x M_x}{R_0} \quad (1A-5)$$

An initial estimate of the intrinsic rate of natural increase (I) is provided in LIFETAB by the approximate formula,

$$I \approx \frac{\log_e R_0}{T} \quad (1A-6)$$

which represents an exact estimate of I only when $R_0 = 1.0$. In order to obtain a more precise estimate of I when R_0 deviates from the equilibrium condition, program LIFETAB uses Euler's implicit equation derived by Lotka (1A-1, 1A-2):

$$\sum_{y=x}^{\infty} e^{-Iy} L_y M_y = 1 \quad (1A-7)$$

The initial estimate of I from Eq. 1A-6 is substituted into Eq. 1A-7 and if the deviation of the sum from unity is more than 0.001, then a new trial value of

I, adjusted by 20% of the difference from one, is again substituted into Eq. 1A-7. This procedure iterates until the sum is within 0.001 of unity, and this value of I is then multiplied by the final sum to obtain the final estimate of I. This procedure is similar to that recommended by Krebs (1A-3, p. 171).

The parameter I, also known as the Malthusian parameter (Fisher, 1A-4), is a measure of the instantaneous rate of change of population size (per individual). In a closed population I is defined as the instantaneous birth rate (per individual), b, minus the instantaneous death rate (per individual), d. In an open population $I = (\text{births} + \text{immigration}) - (\text{deaths} + \text{emigration})$. For a population at stable equilibrium, $R_0 = 1.0$ and $I = 0$; for a declining population where death and/or emigration exceed birth and/or immigration, I will take on a negative value. The finite rate of increase (J) is calculated by LIFETAB as

$$J = e^I, \quad (1A-8)$$

which means that for every individual present this year (or season), there will be J present next year (or season).

Most of the above parameters are quite straightforward and would not generally require a computer program for their calculation. However, the main purpose of this program is to calculate the reproductive value of each age group in a population. As originally defined by R.A. Fisher, the reproductive value at age X (V_x) relative to the reproductive value at birth (V_0) is estimated by

$$V_x/V_0 = \left(\frac{e^{IX}}{L_x} \right) \left(\sum_{Y=X}^{\infty} e^{-IY} L_Y M_Y \right) \quad (1A-9)$$

The summation value pertains to the number of female offspring that will be produced by a female from age X until the end of her life. The term Y is used to label all the ages she will pass through from X to ∞ . In practice V_0 is made equal to one, so that the above equation provides the reproductive value of V_x as a multiple of V_0 . For example, $V_x = 4$ means that a female who has reached the age of X can be expected to produce four times as many female offspring as another female who has just been born (1A-5). It can be seen that for a population at stable equilibrium ($R_0 = 1.0$, $I = 0$) the above equation reduces to

$$V_x = \sum_{y=x}^{\infty} (L_y M_y) / L_x \quad (1A-10)$$

However, for a changing population, Eq. 1A-9 must be used to calculate V_x since the definition of reproductive value in a changing population is the present value of future offspring. In an increasing population the reproductive value of very young individuals relative to older individuals is lower than in a declining population because the future breeding population will be larger and offspring to be produced later will contribute less to the total gene pool than offspring currently being born. Similarly for a declining population, offspring expected at some future date are worth relatively more than current progeny because the total future population will be smaller (1A-6). This fact can be observed from Figures 1.8-13 and 1.8-14 which are plots of the reproductive value output from LIFETAB for Hudson River striped bass and white perch, using $R_0 = 0.5, 1.0, \text{ and } 1.5$ for each population.

DATA INPUT AND RESTRICTIONS

Data is contained in separate data programs for each population and the appropriate program must be appended to LIFETAB prior to running. The required data are:

1. N : the number of age groups in the population (ages 0 to $N-1$). Age 0 is denoted by $X=1$ because HP BASIC arrays are indexed from 1, not 0. N is restricted to 25 (ages 0 through 24) age groups.
2. Q_x : age-specific annual mortality rates. The last value of Q is always 1.0.
3. A_x : proportion of females sexually mature at age X . A value of 0 must be input for each sexually immature age group.
4. F_x : Fecundity or total ovarian egg count of age X . A value of 0 must be input for each sexually immature age group.

Terminal input required:

1. An answer of Y or N for the option of iteration printout showing the calculation of I
2. N5: the maximum number of iterations desired. If the number of iterations chosen is too low to allow convergence to 1.001 or less, then the current value of I (I_0) will be printed out along with the current sum of the Euler equation (Eq. 1A-7), and the program will stop.
3. R_0 : desired value for the net reproductive rate. A value of $R_0 = 1.0$ is appropriate for a population at stable equilibrium, $R_0 < 1.0$ if the population is declining, and $R_0 > 1.0$ if the population is increasing.

OUTPUT

1. The options to be used and the values chosen are printed out.
2. If the iteration printout is chosen, the iteration number, current value of I, and current summation value are printed out.
3. The table of population statistics is composed of age-specific mortality (Q_x), survivorship (L_x), fertility (M_x), the cross-products $L_x M_x$ and $x L_x M_x$, and reproductive values (V_x). In addition, values for the following population parameters are printed out
 - a. gross reproductive rate (GRR)
 - b. net reproductive rate (R_0)
 - c. intrinsic rate of natural increase (I)
 - d. finite rate of increase (J)
 - e. generation time (T)
 - f. egg to juvenile survival (P)

A program listing, sample data, and output for equilibrium populations of striped bass and white perch are attached.

LIFETAB

```

1  MARGIN 80
2  INPUT "DO YOU WANT ITERATION PRINTOUT (Y/N)?",I$
3  INPUT "WHAT IS MAX NUMBER OF ITERATIONS?..",N5
10 DIM Q(25),LC(25),MC(25),Y1(25),Z1(25),AC(25),FC(25),VC(25)
20 PRINT
30 REM Q(X)=ANNUAL MORTALITY RATE
40 PRINT
50 REM L(X)=PROPORTION OF INITIAL COHORT ALIVE AT
60 REM THE BEGINNING OF EACH YEAR.
70 PRINT
80 REM M(X)=AVERAGE NO. FEMALE OFFSPRING PRODUCED/FEMALE AT AGE X
90 PRINT
100 REM Y1(X)=L(X)*M(X) VALUES
105 PRINT
110 REM Z1(X)=XL(X)*M(X) VALUES
120 PRINT
130 REM A(X)=PROPORTION OF FEMALES SEXUALLY MATURE AT AGE X
140 PRINT
150 REM F(X)=FECUNDITY OF TOTAL EGG COUNT AT AGE X.
160 PRINT
170 REM F(X)/2=Eggs DESTINED TO BECOME FEMALES ASSUMING 1:1 SEX RATIO.
180 PRINT
190 REM V(X)=REPRODUCTIVE VALUE AT AGE X.
200 REM DENOTES REDUCTION IN NEXT-GENERATION POPULATION
210 REM CAUSED BY REMOVAL OF SINGLE FEMALE OF AGE X.
220 PRINT
230 READ N
240 PRINT
250 REM N IS THE NUMBER OF AGE GROUPS (AGES 0 TO N-1).
255 REM AGE 0 DENOTED BY X=1 BECAUSE HP BASIC ARRAYS ARE
256 REM INDEXED FROM 1 (NOT 0).
260 PRINT
270 READ (FOR X=1 TO N,Q(X))
280 READ (FOR X=1 TO N,LC(X))
290 READ (FOR X=1 TO N,FC(X))
295 REM DATA CONTAINED IN SEPARATE DATA PROGRAMS WHICH
296 REM MUST BE APPENDED TO LIFETAB.
300 LET L(1)=1
310 FOR X=2 TO N
315 REM MEANS FROM AGE 1 TO N.
320 LET L(X)=L(X-1)-(Q(X-1)*L(X-1))
330 NEXT X
350 PRINT
360 REM RC=NET REPRODUCTIVE RATE=SUM L(X)*M(X) VALUES,DEFINED AS THE
370 REM AVERAGE NO. OF AGE 0 FEMALE OFFSPRING PRODUCED BY AN AVERAGE
380 REM NEWBORN FEMALE DURING ITS LIFETIME.
390 PRINT
420 PRINT "NOTE:RC<1,POPULATION DECLINING;RC=1,POPULATION AT STABLE"
430 PRINT "EQUILIBRIUM;RC>1,POPULATION INCREASING"
431 PRINT
432 INPUT "WHAT IS YOUR VALUE FOR RC?",RC

```

```

435 LET S1=0
440 FOR X=1 TO N
450 LET S1=S1+(L[X]*A[X])*F[X]/2)
460 NEXT X
480 PRINT
490 REM P=PROPORTION OF EGGS SURVIVING TO RECRUITED JUVENILE THREE TO
500 REM FOUR MONTHS AFTER SPAWNING
510 PRINT
515 LET P=R0/S1
-----
525 FOR X=1 TO N
530 LET M[X]=P*A[X]*(F[X]/2)
540 NEXT X
550 FOR X=1 TO N
560 LET Y1[X]=L[X]*M[X]
570 NEXT X
580 FOR X=1 TO N
590 LET Z1[X]=(X-1)*L[X]*M[X]
600 NEXT X
620 PRINT
630 REM T=GENERATION TIME=SUM XL(X)M(X)/R0,DEFINED AS THE MEAN PERIOD
640 REM BETWEEN BIRTH OF PARENTS AND BIRTH OF OFFSPRING.
650 PRINT
655 LET T1=0
660 FOR X=1 TO N
670 LET T1=T1+Z1[X]
680 NEXT X
690 LET T=T1/R0
700 LET I=LOG(R0)/T
701 PRINT
702 REM I=INTRINSIC RATE OF NATURAL INCREASE.IT IS THE INSTANTANEOUS
703 REM RATE REPRESENTING THE DIFFERENCE BETWEEN RATES AT WHICH NEW
704 REM INDIVIDUALS ARE BEING BORN,PER INDIVIDUAL,AND AT WHICH
705 REM PREVIOUSLY EXISTING INDIVIDUALS ARE DYING,PER INDIVIDUAL.
706 PRINT
710 LET I0=I
712 IF I3="Y" THEN PRINT "ITER. NO.",I0,"SUM"
720 LET B2=0
730 LET B1=0
740 FOR X=1 TO N
750 LET B1=B1+(EXP(-I0*(X-1))*Y1[X])
760 NEXT X
780 LET B2=B2+1
790 IF I3="Y" THEN PRINT B2,I0,B1
792 IF ABS(B1-1)<=.001 THEN GO
800 I=I0*B1
810 GOTO 550
820 OOEWD
830 ELSE DO

```

```

840 LET I0=I0*(1+SGN(I0)*.2*(B1-1))
850 IF B2>=N5 THEN DO
860 PRINT "I0=",I0
870 PRINT "SUM=",B1
875 STOP
880 DOEND
885 ELSE GOTO 730
890 DOEND
950 FOR X=1 TO N
960 LET V1=0
970 FOR Y=X TO N
980 LET V1=V1*EXP(-I*(Y-1))*LEYJ*MEYJ
990 NEXT Y
1000 LET V2=(EXP(I*(X-1)))/LEXJ
1010 LET VEXJ=V1*V2
1020 NEXT X
1030 PRINT
1040 REM G=GRR=GROSS REPRODUCTIVE RATE=SUM OF M(X) VALUES.
1050 REM REPRESENTS THE LIFETIME TOTAL NO. FEMALE OFFSPRING THAT WOULD
1060 REM BE PRODUCED BY AN AVERAGE FEMALE IN THE ABSENCE OF MORTALITY.
1070 PRINT
1080 LET G=0
1090 FOR X=1 TO N
1100 LET G=G+MEXJ
1110 NEXT X
1160 PRINT
1170 LET J=EXP(I)
1180 REM J=THE FINITE RATE OF INCREASE.FOR EVERY INDIVIDUAL PRESENT
1190 REM THIS YEAR.THERE WILL BE J PRESENT NEXT YEAR.
1200 PRINT
1210 PRINT
1220 PRINT TAB(21);"TABLE OF POPULATION STATISTICS"
1230 PRINT
1240 PRINT
1250 PRINT TAB(63);"REPRODUCTIVE"
1260 PRINT "AGE";TAB(5);"MORTALITY";TAB(16);"SURVIVORSHIP";TAB(31);
"FERILITY";TAB(66);"VALUE"
1270 PRINT "X";TAB(8);"Q(X)";TAB(20);"L(X)";TAB(33);"M(X)";TAB(41);
"L(X)*M(X)";TAB(52);"XL(X)*M(X)";TAB(67);"V(X)"
1280 PRINT
1290 FOR X=1 TO N
1300 PRINT USING 1310;X-1,QEXJ,LEXJ,MEXJ,V1EXJ,Z1EXJ,VEXJ
1310 IMAGE 004X,0.006X,0.00000000003Y,000.00002X,0.00005X,0.00007X,0&
00.04X
1320 NEXT X
1330 PRINT
1340 PRINT
1350 PRINT "GROSS REPRODUCTIVE RATE (GRR) =" ;G
1360 PRINT
1370 PRINT "NET REPRODUCTIVE RATE (R0) =" ;R0
1380 PRINT
1390 PRINT USING 1400;"INTRINSIC RATE OF NATURAL INCREASE (I) =" ;I
1400 IMAGE 40A,X,M.0000000
1410 PRINT
1420 PRINT USING 1430;"FINITE RATE OF INCREASE (J) =" ;J
1430 IMAGE 29A,X,D.0000
1440 PRINT
1450 PRINT "GENERATION TIME (T) =" ;T
1460 PRINT
1470 PRINT USING 1480;"EGG TO JUVENILE SURVIVAL (P) =" ;P
1480 IMAGE 30A,X.0000000000
1490 END

```

S00DATA

9000 DATA 21

9010 DATA .8,-2,-2,-4,-6,.53,.53,.53,.53,.53,.53,.53,.53,.53,.53,.53,2
.53,.7,.9,.95,1

9020 DATA 0,0,0,-04,.07,.21,.47,.87,.9,1,1,1,1,1,1,1,1,1

9030 DATA 0,0,0,660000,660000,580000,710000,930000,1.31E6,1.56E6,1.76E6

9040 DATA 1.98E6,2.09E6,2.13E6,2.19E6,2.6E6,2.7E6,2.8E6,2.9E6

9050 DATA 3E6,3.1E6

DO YOU WANT ITERATION PRINTOUT (Y/N)?Y
 WHAT IS MAX NUMBER OF ITERATIONS?..100

NOTE:RO<1,POPULATION DECLINING;RO=1,POPULATION AT STABLE
 EQUILIBRIUM;RO>1,POPULATION INCREASING

WHAT IS YOUR VALUE FOR RO?1.5

ITER. NO.	IO	SUM
1	6.34118E-02	1.01017
2	6.35408E-02	1.00938
3	6.36501E-02	1.00865
4	6.37702E-02	1.00797
5	6.38719E-02	1.00735
6	6.39650E-02	1.00679
7	6.40525E-02	1.00625
8	6.41325E-02	1.00576
9	6.42064E-02	1.00531
10	6.42745E-02	1.00489
11	6.43374E-02	1.00451
12	6.43954E-02	1.00415
13	6.44489E-02	1.00383
14	6.44982E-02	1.00353
15	6.45437E-02	1.00325
16	6.45856E-02	1.00299
17	6.46243E-02	1.00276
18	.06466	1.00254
19	6.46928E-02	1.00234
20	6.47231E-02	1.00216
21	.064751	1.00199
22	6.47767E-02	1.00183
23	6.48004E-02	1.00169
24	6.48223E-02	1.00155
25	6.48424E-02	1.00143
26	.064861	1.00132
27	6.48781E-02	1.00121
28	6.48938E-02	1.00112
29	6.49083E-02	1.00103
30	6.49217E-02	1.00095

TABLE OF POPULATION STATISTICS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	KL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.80	1.0000000000	.0000	.0000	.0000	1.0
1	.20	.2000000477	.0000	.0000	.0000	5.3
2	.20	.1600000560	.0000	.0000	.0000	7.1
3	.40	.1280000508	1.3376	.1712	.5137	9.5
4	.60	.0768000335	2.3409	.1798	.7191	14.5
5	.53	.0397200104	6.1713	.1896	.9479	32.5
6	.53	.0144384056	16.9079	.2441	1.4647	59.7
7	.53	.0067860512	40.9953	.2782	1.9474	97.3
8	.53	.0031894441	59.7374	.1905	1.5242	127.7
9	.53	.0014990389	79.0418	.1185	1.0664	154.4
10	.53	.0007045483	89.1754	.0628	.6223	171.1
11	.53	.0003311377	100.3223	.0332	.3654	186.0
12	.53	.0001556347	105.8957	.0165	.1978	194.5
13	.53	.0000731483	107.9224	.0079	.1026	201.1
14	.53	.0000343797	110.9525	.0038	.0534	211.5
15	.53	.0000161585	131.7363	.0021	.0319	228.3
16	.53	.0000075945	136.8031	.0010	.0166	219.3
17	.70	.0000035654	141.8699	.0005	.0086	187.4
18	.90	.0000010708	146.9366	.0002	.0028	161.9
19	.95	.0000001071	152.0034	.0000	.0003	159.4
20	1.00	.0000000054	157.0702	.0000	.0000	157.1

GROSS REPRODUCTIVE RATE (GRR) = 1587.23

NET REPRODUCTIVE RATE (R0) = 1.5

INTRINSIC RATE OF NATURAL INCREASE (I) = .0649833

FINITE RATE OF INCREASE (J) = 1.0671

GENERATION TIME (T) = 6.39416

EGG TO JUVENILE SURVIVAL (F) = .0001013356

DO YOU WANT ITERATION PRINTOUT (Y/N)?Y
WHAT IS MAX NUMBER OF ITERATIONS?..100

NOTE:RO<1,POPULATION DECLINING;RO=1,POPULATION AT STABLE
EQUILIBRIUM;RO>1,POPULATION INCREASING

WHAT IS YOUR VALUE FOR RO?1.0

ITER. NO.	IO	SUM
1	0	1.

TABLE OF POPULATION STATISTICS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.20	1.0000000000	.0000	.0000	.0000	1.0
1	.20	.2000000477	.0000	.0000	.0000	5.0
2	.20	.1600000560	.0000	.0000	.0000	6.2
3	.40	.1280000508	.8918	.1141	.3424	7.8
4	.60	.0768000335	1.5606	.1199	.4794	11.5
5	.53	.0307200164	4.1142	.1264	.6317	24.9
6	.53	.0144384086	11.2719	.1627	.9765	44.3
7	.53	.0067860512	27.3302	.1855	1.2982	70.3
8	.53	.0031894441	39.8249	.1278	1.0162	91.4
9	.53	.0014990389	52.6945	.0790	.7109	109.7
10	.53	.0007045483	59.4502	.0419	.4187	121.7
11	.53	.0003311377	66.8915	.0221	.2436	131.4
12	.53	.0001556347	70.5972	.0110	.1318	137.2
13	.53	.0000731483	71.9483	.0053	.0684	141.8
14	.53	.0000343797	73.9750	.0025	.0356	148.6
15	.53	.0000161585	87.8242	.0014	.0217	158.2
16	.53	.0000075945	91.2021	.0007	.0111	151.0
17	.70	.0000035694	94.5799	.0003	.0057	127.2
18	.90	.0000010708	97.9578	.0001	.0019	108.6
19	.95	.0000001071	101.3356	.0000	.0002	106.6
20	1.00	.0000000054	104.7135	.0000	.0000	104.7

GROSS REPRODUCTIVE RATE (GRR) = 1058.15

NET REPRODUCTIVE RATE (R0) = 1

INTRINSIC RATE OF NATURAL INCREASE (r) = .0000000

FINITE RATE OF INCREASE (J) = 1.0000

GENERATION TIME (T) = 6.39415

EGG TO JUVENILE SURVIVAL (F) = .0000675571

DO YOU WANT ITERATION PRINTOUT (Y/N)?Y
 WHAT IS MAX NUMBER OF ITERATIONS?..100

NOTE:R0<1,POPULATION DECLINING;R0=1,POPULATION AT STABLE
 EQUILIBRIUM;R0>1,POPULATION INCREASING

WHAT IS YOUR VALUE FOR R0?.5

ITER. NO.	I0	SUM
1	-.108403	1.03227
2	-.107704	1.02723
3	-.107117	1.02303
4	-.106624	1.01951
5	-.106208	1.01653
6	-.105856	1.01406
7	-.105559	1.01196
8	-.105306	1.01018
9	-.105092	1.00867
10	-.10491	1.00739
11	-.104755	1.0063
12	-.104623	1.00537
13	-.10451	1.00459
14	-.104414	1.00391
15	-.104333	1.00334
16	-.104263	1.00285
17	-.104203	1.00244
18	-.104153	1.00208
19	-.104109	1.00178
20	-.104072	1.00152
21	-.10404	1.0013
22	-.104013	1.00111
23	-.10399	1.00095

TABLE OF POPULATION STATISTICS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.80	1.0000000000	.0000	.0000	.0000	1.0
1	.20	.2000000477	.0000	.0000	.0000	4.5
2	.20	.1600000560	.0000	.0000	.0000	5.1
3	.40	.1280000508	.4459	.0571	.1712	5.7
4	.60	.0768000335	.7803	.0599	.2397	7.9
5	.53	.0307200104	2.0571	.0632	.3160	16.1
6	.53	.0144384056	5.6359	.0814	.4882	26.9
7	.53	.0067860512	13.6651	.0927	.6491	40.9
8	.53	.0031894441	19.9124	.0635	.5081	32.1
9	.53	.0014990389	26.3473	.0395	.3555	61.8
10	.53	.0007645483	29.7251	.0209	.2094	67.9
11	.53	.0003311377	33.4408	.0111	.1218	73.2
12	.53	.0001556347	35.2986	.0055	.0659	76.3
13	.53	.0000731483	35.9741	.0026	.0342	74.5
14	.53	.0000343797	36.9875	.0013	.0178	81.6
15	.53	.0000161585	43.9121	.0007	.0106	85.5
16	.53	.0000075945	45.6010	.0003	.0055	79.8
17	.70	.0000035694	47.2900	.0002	.0029	65.6
18	.90	.0000010708	48.9789	.0001	.0009	54.9
19	.95	.0000001071	50.6678	.0000	.0001	53.6
20	1.00	.0000000054	52.3567	.0000	.0000	52.4

GROSS REPRODUCTIVE RATE (GFR) = 529.077

NET REPRODUCTIVE RATE (R0) = .5

INTRINSIC RATE OF NATURAL INCREASE (I) = -.1040891

FINITE RATE OF INCREASE (J) = .9011

GENERATION TIME (T) = 6.39416

EGG TO JUVENILE SURVIVAL (F) = .0000337785

WPDATA

9000 DATA 7

9010 DATA .62,.21,.36,.58,.72,.9,1

9020 DATA 0,0,.75,1,1,1,1

9030 DATA 0,0,40000,52000,72000,84000,106000

DO YOU WANT ITERATION PRINTOUT (Y/N)?Y
 WHAT IS MAX NUMBER OF ITERATIONS?..100

NOTE:RO<1,POPULATION DECLINING;RO=1,POPULATION AT STABLE
 EQUILIBRIUM;RO>1,POPULATION INCREASING

WHAT IS YOUR VALUE FOR RO?1.5

ITER. NO.	IO	SUM
1	.132986	1.00785
2	.133195	1.00724
3	.133388	1.00667
4	.133566	1.00614
5	.13373	1.00566
6	.133881	1.00521
7	.134021	1.0048
8	.134149	1.00442
9	.134268	1.00407
10	.134377	1.00375
11	.134478	1.00345
12	.134571	1.00318
13	.134656	1.00293
14	.134735	1.0027
15	.134808	1.00248
16	.134875	1.00229
17	.134937	1.0021
18	.134993	1.00194
19	.135046	1.00178
20	.135094	1.00164
21	.135138	1.00151
22	.135179	1.00139
23	.135217	1.00128
24	.135251	1.00118
25	.135283	1.00109
26	.135313	1.001
27	.13534	1.00092

TABLE OF POPULATION STATISTICS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.62	1.0000000000	.0000	.0000	.0000	1.0
1	.21	.3799999952	.0000	.0000	.0000	3.0
2	.36	.3001999855	1.6761	.5014	1.0027	4.4
3	.58	.1921280026	2.8949	.5562	1.6686	4.8
4	.72	.0806937516	4.0083	.3234	1.2938	5.3
5	.90	.0225942507	4.6764	.1057	.5283	5.2
6	1.00	.0022594258	5.9011	.0133	.0800	5.9

GROSS REPRODUCTIVE RATE (GRR) = 19.1508

NET REPRODUCTIVE RATE (R0) = 1.5

INTRINSIC RATE OF NATURAL INCREASE (I) = .1354646

FINITE RATE OF INCREASE (J) = 1.1451

GENERATION TIME (T) = 3.04893

EGG TO JUVENILE SURVIVAL (P) = .0001113419

DO YOU WANT ITERATION PRINTOUT (Y/N)?Y
 WHAT IS MAX NUMBER OF ITERATIONS?..100

NOTE:RO<1,POPULATION DECLINING;RO=1,POPULATION AT STABLE
 EQUILIBRIUM;RO>1,POPULATION INCREASING

WHAT IS YOUR VALUE FOR RO?1.0

ITER. NO.	IO	SUM
1	0	1

TABLE OF POPULATION STATISTICS

AGE X	MORTALITY G(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.62	1.0000000000	.0000	.0000	.0000	1.0
1	.21	.3799999999	.0000	.0000	.0000	2.6
2	.36	.3001999855	1.1134	.3342	.6685	3.3
3	.58	.1921280026	1.9299	.3708	1.1124	3.5
4	.72	.0816937516	2.6722	.2156	.8625	3.7
5	.90	.0225942507	3.1176	.0704	.3522	3.5
6	1.00	.0022594258	3.9341	.0089	.0533	3.9

GROSS REPRODUCTIVE RATE (GRR) = 12.7672

NET REPRODUCTIVE RATE (RO) = 1

INTRINSIC RATE OF NATURAL INCREASE (I) = .0000000

FINITE RATE OF INCREASE (J) = 1.0000

GENERATION TIME (T) = 3.04893

EGG TO JUVENILE SURVIVAL (F) = .0000742280

DO YOU WANT ITERATION PRINTOUT (Y/N)?Y
 WHAT IS MAX NUMBER OF ITERATIONS?..100

NOTE:RO<1,POPULATION DECLINING;RO=1,POPULATION AT STABLE
 EQUILIBRIUM;RO>1,POPULATION INCREASING

WHAT IS YOUR VALUE FOR RO?..5

ITER. NO.	IO	SUM
1	-.227341	1.02485
2	-.226211	1.02107
3	-.225258	1.0179
4	-.224452	1.01522
5	-.223769	1.01296
6	-.223189	1.01104
7	-.222696	1.00941
8	-.222277	1.00803
9	-.22192	1.00686
10	-.221615	1.00586
11	-.221355	1.00501
12	-.221134	1.00428
13	-.220944	1.00366
14	-.220783	1.00313
15	-.220644	1.00268
16	-.220526	1.00229
17	-.220425	1.00196
18	-.220339	1.00168
19	-.220265	1.00144
20	-.220201	1.00123
21	-.220147	1.00105
22	-.220101	1.00089

TABLE OF POPULATION STATISTICS

AGE X	MORTALITY Q(X)	SURVIVORSHIP L(X)	FERTILITY M(X)	L(X)M(X)	XL(X)M(X)	REPRODUCTIVE VALUE V(X)
0	.62	1.0000000000	.0000	.0000	.0000	1.0
1	.21	.3799999999	.0000	.0000	.0000	2.1
2	.36	.3001999855	.5567	.1671	.3342	2.1
3	.58	.1921280026	.9650	.1854	.5562	2.0
4	.72	.0806937516	1.3361	.1078	.4313	2.0
5	.90	.0225942507	1.5588	.0352	.1761	1.8
6	1.00	.0022594258	1.9670	.0044	.0267	2.0

GROSS REPRODUCTIVE RATE (GRR) = 6.38361

NET REPRODUCTIVE RATE (R0) = .5

INTRINSIC RATE OF NATURAL INCREASE (I) = -.2202992

FINITE RATE OF INCREASE (J) = .8723

GENERATION TIME (T) = 3.04893

EGG TO JUVENILE SURVIVAL (F) = .0000371140

REFERENCES

- 1A-1. A. J. Lotka. "Studies on the Mode of Growth of Material Aggregates." Amer. J. Sci., Vol. 24, 1907, pp. 199-216.
- 1A-2. A. J. Lotka. "A Natural Population Norm." J. Wash. Acad. Sci., Vol. 3, 1913, pp. 241, 289-293.
- 1A-3. C. J. Krebs. Ecology, The Experimental Analysis of Distribution and Abundance. New York: Harper and Row, 1972.
- 1A-4. R. A. Fisher. The Genetical Theory of Natural Selection. 2nd ed. New York: Dover Publ., 1958.
- 1A-5. E. O. Wilson and W. H. Bossert. A Primer of Population Biology. Sunderland, Mass.: Sinauer Assoc., 1971.
- 1A-6. E. R. Pianka. Evolutionary Ecology. 2nd ed. New York: Harper and Row, 1978.