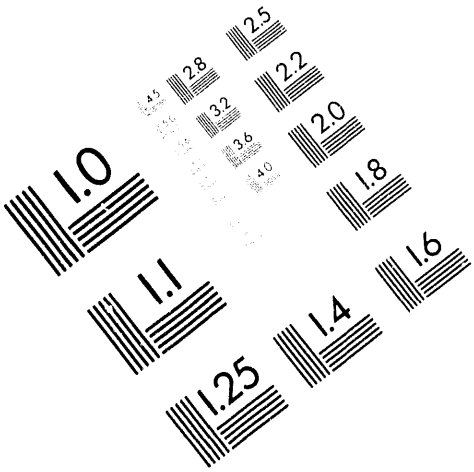

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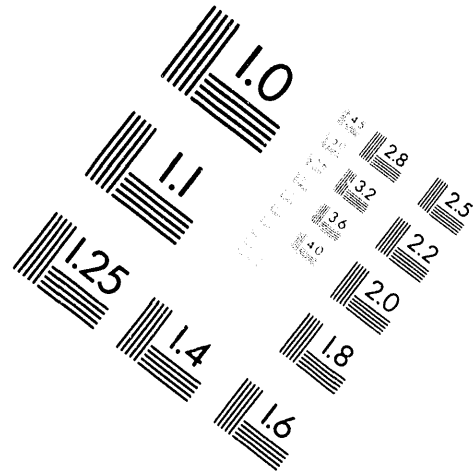

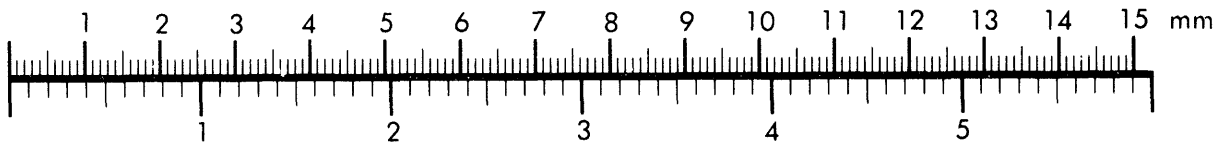

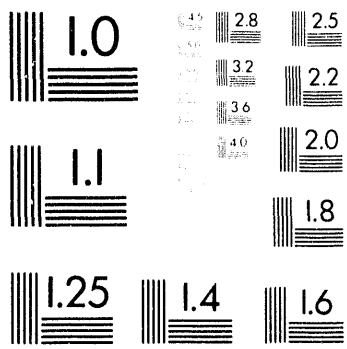

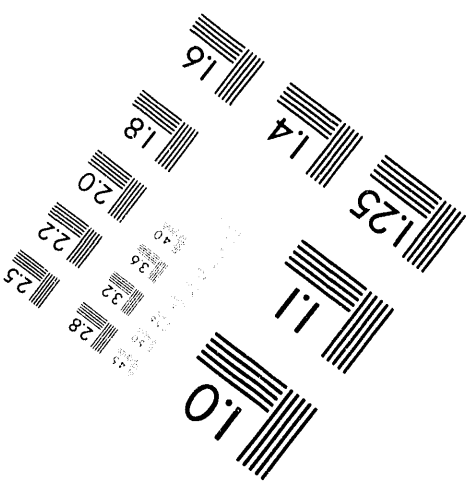

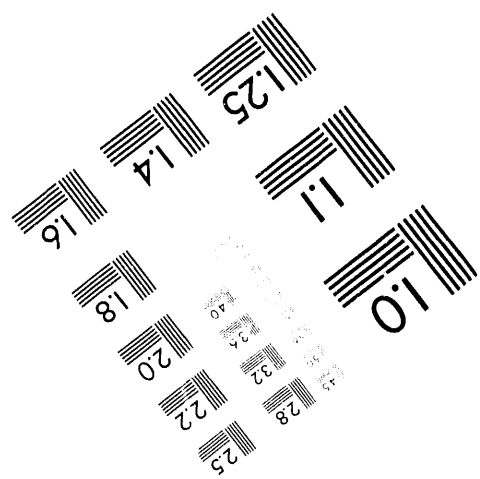

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PESTICIDES AND THEIR EFFECTS ON WILDLIFE

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Pesticides and Their Effects on Wildlife

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About 560 active ingredients are currently used as pesticides. Applications of these pesticides are made to agricultural lands and other areas inhabited by wildlife. Unfortunately, many agricultural-use pesticides also entail some measure of risk to organisms other than the pest species. Because testing of pesticides prior to registration cannot evaluate all the potential environmental-pesticide-wildlife/fish interactions, current methods of risk assessment do not always provide sufficient safety to nontarget organisms. This is evidenced by die-offs of fish and wildlife from applications of pesticides at environmentally "safe" rates (Grue et al. 1983; National Research Council of Canada 1975), the linking of population declines of some species with agrochemical use (Fox et al. 1989; Grue et al. 1986, 1989), and observations of survival-threatening behavioral changes in laboratory and field animals exposed to typical field levels of pesticides (Costa and Murphy 1982; White et al. 1979; Galindo et al. 1985). It is important to note, however, that the majority of pesticides, when properly applied, have not caused significant injury to wildlife (Cooper 1991).

A brief summary of pesticide effects on wildlife and fish are presented for the common classes of pesticides in use today.

Organophosphate and Carbamate Insecticides

Organophosphate pesticides (OP) and carbamates (CB) are among the most heavily used agricultural pesticides in the United States. Both OP and CB insecticides act principally by inhibiting the enzyme acetylcholinesterase (AChE) (Casarett and Doulls 1986). AChE normally cleaves the neurotransmitter, acetylcholine, from activated neural and myoneural junctions, thus restoring the nerve or muscle to the resting state. Inhibition of AChE by OPs or CBs results in excessive buildup of acetylcholine at the nerve junctions. This buildup disrupts nerve impulse transmission and can cause death by blocking neurotransmission in the respiratory center of the brain or at myoneural junctions of the respiratory muscles (O'Brien 1967; Casarett and Doulls 1986). Sublethal signs of intoxication in wild animals include ataxia, tremors, salivation, lacrimation, loss of righting reflex, tetany, diarrhea, piloerection (or fluffed feathers), and polydipsia.

Many OPs bind permanently to AChE. Several weeks may be required to synthesize new AChE and re-establish normal levels of the enzyme in the animal. (Robinson and Beiergrohslein 1980; Fleming 1981). Therefore, additional exposures to AChE inhibitors (OPs or CBs) during this recovery period can diminish the AChE stores to hazardous levels. In contrast, little cumulative toxic action can occur with CBs because the chemical reaction of the CB with AChE is completely and rapidly reversible (Casarett and Doulls 1986) and the parent compounds rapidly degraded (Tucker and Crabtree 1970; Busby et al. 1987). Multiple applications of dimethoate, chlorpyrifos, methomyl, carbaryl and methyl parathion have been shown to reduce the survival of birds inhabiting edge habitats adjacent to croplands and orchards (White et al. 1990). Recent consideration of the effects of multiple exposures on reproduction in birds suggests that increasing exposure to pesticides impairs songbird (Passeriformes) productivity (Patnode and White in press; Mineau and Peakall 1987). OPs and CBs also inhibit a number of other enzymes. Inhibition of some of these enzymes appears to be the mechanism by which exposure to one OP/CB potentiates or decreases the toxic effect of another (DuBois et al. 1968; Murphy and Cheever 1968).

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[a] Pacific Northwest Laboratory is operated for the U.S. Department of Energy by Battelle Memorial Institute under Contract DE-AC06-76RLO 1830.

Direct Effects on Wildlife

Lethality

There are many confirmed incidents of wildlife mortality resulting from proper use (i.e., application according to label requirements) of OPs and CBs (Grue et al. 1983). In North America, over 70% of these unintentional poisonings involved either diazinon, parathion, fenthion, phosphamidon or carbofuran (Eisler 1985). It should be noted that the reported incidents probably reflect highly visible situations, and there may actually be many undetected incidents for each one reported (Hoffman et al. 1990).

Birds are the wildlife group most often reported in OP and CB mortality incidents. The avian species most frequently poisoned are waterfowl (Anatidae), kinglets and thrushes (Muscicapidae), sparrows (Emberizidae), and warblers (Parulidae). Birds that forage or nest in the sprayed vegetation appear to be those most exposed to and affected by OP/CB applications (Pearce et al. 1979; DeWeese et al 1979; Blus et al. 1989). Few reports involve mammals, probably because they tend to be less conspicuous and less sensitive to OPs and CBs than birds.

The sensitivity of wildlife to acetylcholinesterase (AChE)-inhibiting pesticides varies with species and chemical. In general, birds are more sensitive to OPs and CBs than mammals (Table 1) and mammals are more sensitive than amphibians (Gaines 1969; Tucker and Crabtree 1970; Schafer 1972; Schafer and Cunningham 1972; Hudson et al. 1979; Schafer and Brunton 1979; Kenaga 1979; Cholakis et al. 1981). The sensitivity of reptiles to OP and CB exposure is similar to that of birds or mammals (Hall and Clark 1982). Fish and aquatic invertebrates can vary greatly in their sensitivity to OP and CB exposures (Table 1).

Over 50 incidents of bird poisonings have been attributed to use of granular carbofuran (Fox et al. 1989). Most of the die-offs involved ducks or geese in lands associated with agricultural fields (Stickel 1975; Flickinger et al. 1980). However, there have been five reports of bird kills associated with the use of 10G granules in pine plantations (Overgaard et al. 1983; U.S. EPA 1979). Granules were incorporated by hand into the hole made for the seedling or by means of a deep-injection device, which reportedly achieves better than 99% incorporation. Song bird and mouse die-offs have also been reported on vegetable crop fields treated with various granule formulations of carbofuran (Balcom et al. 1984). Granular application of carbofuran at 0.5 lb/acre resulted in deaths of red-winged blackbirds, three species of sandpipers (*Erolia* spp.), frogs, earthworms, crayfish, leeches, and four species of fish within 48 h (Flinkinger et al. 1980). Twelve incidents of non-intentional secondary poisoning of birds from consumption of prey contaminated with granular carbofuan have been reported (Balcom 1983; Fox et al. 1989; Overgaard et al. 1983; Stone and Bradoni 1985a) and include kills of bald eagles, red-shouldered hawks, red-tailed hawks, loggerhead shrikes, northern harriers, owls, and ravens. This insecticide has been implicated in the population decline of the burrowing owl in Canada (Fox et al. 1989).

Methyl parathion is highly toxic to wildlife and is one of the most heavily used pesticides, yet the only reported bird kills associated with the chemical followed applications of methyl parathion mixed with parathion, toxaphene, or methomyl (Grue et al. 1983; Smithson and Sanders 1978). Wildlife die-offs have occurred following azinophos-methyl when it was mixed with parathion (Smith 1987). Dimethoate-induced bird kills apparently occur when applications are made during high temperatures (Blus et al. 1989).

Effects of Organophosphates and Carbamates on Reproduction and Survival

OPs have been shown to cause a number of responses in animals that may affect reproduction or survival of wildlife. Severe body weight loss (up to 40%) is often associated with OP exposure and is apparently caused by OP-induced anorexia (Costa and Murphy 1982; Grue 1982, Stromborg 1981). Application of an OP (fenitrothion) for spruce budworm control resulted in significantly reduced growth of nestling white-throated sparrows (*Zonotrichia albicollis*) and reduced fledgling weights (Pearce and Busby 1980). Reduced litter size in mice (Spyker and Avery 1977) and

Table 1. Relative Toxicity of Some Pesticides to Fish and Wildlife

	Toxicity Class[a]					
	Aquatic Invertebrates	Fish	Birds		Mammals	Die-offs
Pesticide	(LC_{50})	(LC_{50})	Acute Oral (LD_{50})	Dietary (LD_{50})	Acute Oral (LD_{50})	Reference[b]
Insecticides						
Organophosphates						
Acephate	L	L	M	L	M	
Azinophos-methyl	M	VH	H	M	VH	3
Chlorpyrifos	VH	VH	H	M	H	1
Diazinon	H	M	VH	M	M to H	1
Dimethoate	-[c]	M	VH	M	M	2
Malathion	VH	H	L	L	M	
Methyl Parathion	H	M	VH	M	VH	1
Naled	VH	L	H	L	M	
Parathion	H	H	VH	H	VH	1,3
Phosmet	-	H	M to H	L to M	H	
Carbamates						
Carbaryl	M	M	L	L	M	
Carbofuran	-	H	VH	M	VH	
Synthetic Pyrethroids						
Bifenthrin	H	VH	L	L	M	
Fenvalerate	-	VH	L	L	M	
Fluvalerate	-	VH	L	L	M	
Permethrin	H	VH	L	L	L	
Pyrethrins	VH	VH	-	-	M	
Organochlorines						
Dicofol	M	M	M	-	M	
Dienochlor	-	H		-	L	
Endosulfan	M to H	H	M	-	H	
Lindane	M	H	L	L	H	
Sulfite Esters						
Propargite	-	H	L	-	M	

Table 1. (contd)

	Toxicity Class[a]					
	Aquatic Invertebrates	Fish	Birds		Mammals	Die-offs
Pesticide	(LC_{50})	(LC_{50})	Acute Oral (LD_{50})	Dietary (LD_{50})	Acute Oral (LD_{50})	Reference[b]
Insect Growth Regulators						
Diflubenzuron	H	L	L	-	L	
Soil Fumigants						
Methyl Bromide	-[c]		-	-	VH	
Dazomet	-	H	-	-	M	
Herbicides						
Amitrole	L	L	L	L	L	
Pendimethalin	-	H	L	-	L	
Oxyfluorfen	-	H	L, H[d]	-	L	
Fungicides						
Benomyl	-	H	-	M	L	
Rodenticides						
Diphacinone	-	M	L	-	VH[e]	
Strychnine	-	-	VH	-	VH	

(a) Toxicity classes modified after Grue et al. (1986): invertebrates 96 h LC_{50} (synthetic pyrethroids 24 h LC50), Low (L) = >1.0, Moderate (M) = 0.01-1.0, High (H) = 0.001-0.01, and Very High (VH) = <0.001 mg/L; fish 96 h LC_{50} (mg/L), L = > 100, M = 1-100, H = 0.1-1, VH = < 0.1; birds and mammals (mg/kg), L = >1000, M = 201-1000, H = 41-200, VH = < 40. Values from Verschueren (1983), Smith (1987), Hudson et al. (1984), Weed Society of America (1983), and Royal Society of Chemistry (1990).

(b) Reports of wildlife die-offs from unintentional poisoning: 1 = Grue et al. 1983, 2 = Blus et al. 1989, 3 = Smith 1987.

(c) Information not available

(d) Toxic to gamebirds but not to waterfowl.

(e) Requires repeated exposures to be lethal.

inhibited egg production in ring-necked pheasants (*Phasianus colchicus*) and bobwhite quail (*Colinus virginianus*) (Stromborg 1981) have resulted from sublethal diazinon exposure. Loss of body weight from OP exposure apparently results in increased susceptibility to environmental stressors (e.g., low temperature) in small animals with high metabolic rates (Pope and Ward 1972; Grue et al. 1983). For example, cold weather contributed to the death of about 1500 geese (*Anser anser* and *A. brachyrhynchus*) that fed on winter wheat treated with carbophenthion (Stanley and Bunyan 1979). Elevated temperatures may have exacerbated dimethoate toxicity to sage grouse (*Centrocedrus urophasianus*) in alfalfa fields (Blus et al. 1989). In laboratory tests, a 60% mortality of American kestrels (*Falco sparverius*) was observed when the birds were exposed to methyl parathion at -5 °C (23°F), whereas exposure to the same concentrations of methyl parathion at 22°C (72°F) caused only sublethal effects (Rattner and Franson 1984). Chronic exposure to heat or cold enhanced parathion toxicity twofold in quail chronically exposed to heat or cold (Rattner et al. 1982; Rattner et al. 1987).

OP-induced changes in parental care may result in reduced reproduction. Female starlings dosed with OPs made fewer trips to feed their young and remained away from the nest for longer time periods compared to controls (Grue et al. 1982). Reduced nest attentiveness (White et al. 1979) and time spent incubating eggs (White et al. 1983) were observed in laughing gulls (*Larus atricilla*) exposed to parathion. However, nest-defense behavior and hatching success were not affected in gulls recieving a single sublethal dose of parathion (King et al. 1984). It should be noted that reproductive effects in wild populations have been inconsistent and, for the most part, have been monitored in populations exposed to a single application of pesticide (Robinson et al. 1988; Bednareck and Davidson 1967; DeWeese et al. 1979; Powell 1984; Meyers et al. 1990).

Other Sublethal Effects

Other sublethal effects of OP exposure include induction of hypothermia in birds (Rattner and Franson 1984) and mammals (Chattopadhyay et al. 1982), changes in immune response (Street and Sharma 1975), alterations in hormone levels (Rattner et al. 1982a,b), and behavioral changes in vigilance (Levin and Rodnitzky 1976), food-seeking (Adams 1977), visual acuity (Plestina and Piukovic-Plestina 1978), and song structure and production by breeding birds (Forsyth 1980; Grue and Shipley 1981). OP-induced behavioral changes may also increase the susceptibility of exposed animals to predation. Predation of captive northern bobwhite following exposure to methyl parathion was increased over that of controls (Galindo et al. 1985). Following a sublethal dose of malathion, loss of radio-tagged sharp-tailed grouse (*Tympanuchus phasianellus*) to predators was increased (McEwen and Brown 1966). Zinkl et al. (1981) and others have noted that predatory birds appear to be attracted to areas treated with OPs. Grass shrimp (*Palaemonetes pugio*) exposed to sublethal levels of ethyl or methyl parathion displayed increased spontaneous activity and were, as a consequence, more easily detected by the gulf killifish (*Fundulus grandis*). Endurance of the shrimp was also reduced resulting in increased capture by pursuing killifish (Farr 1977, 1978). Predatory behavior (e.g., attack efficiency) in juvenile Atlantic Salmon (*Salmo salar*) was also impaired by exposure to very low levels of OPs (Morgan and Kiceniuk 1990).

Indirect Effects on Wildlife

OPs and CBs may indirectly impact the survival and reproduction of wildlife species by reducing invertebrate food resources at critical times. Synchronization of reproduction with peaks in insect populations provides many wildlife species with needed high-protein diets during important developmental stages. Decreased insect abundance as a result of pesticide applications may result in emigration of birds and small mammals from pesticide-treated areas (Keith and Flickinger 1965; McEwen et al. 1965, 1972; Barrett and Darnell 1967; Doane and Schaefer 1971; Moulding 1976; and Bart 1979) with subsequently less successful second nest attempts (Grue et al. 1982). Studies also show that OP-reduction in insects and aquatic invertebrates may result in reduced growth and survival of young birds (Grue et al. 1986; Potts 1977; Powell 1984). In agricultural areas, survival of grey partridge (*Perdix perdix*) chicks was directly correlated to OP-induced reduction in insect abundance (Potts 1977). Waterfowl production in the pothole areas of the United States and Canada may also be decreasing because of food reduction during the egg-laying period of the females and/or during early growth stages of the young. Both bird mortality and invertebrate reduction were greater than expected when aerial applications of ethyl and methyl parathion were made according to label directions to the pothole region (Grue et al. 1986; Grue et al. 1989). Reproductive success of red-winged blackbirds (*Agelaius phoeniceus*) was not affected by a reduction in the principal food source of over 50% after application of fenthion (Powell 1984). However, Powell noted that insect abundance may have been above average during the study and the impact of food-source reduction to the nesting birds less pronounced than when food resources are naturally low.

Population Effects

No permanent declines in overall populations of wildlife species have been attributed to OP/CB exposure. The CB carbofuran, however, has recently been implicated in the population decline of the burrowing owl (*Athene cunicularia*) in Canada (Fox et al. 1989) and several OPs and CBs have been associated with decreases in waterfowl production in the Prairie Pothole Region of the United States and Canada (Grue et al. 1986, 1989). Also, several studies have documented declines in local wildlife

populations following OP and CB applications. Malathion application (2 lb/acre) to a small forested watershed caused a population decline in chipmunks (*Tamias striatus*) of 55% and a decline in white-footed mice (*Peromyscus leucopus*) of 20 to 45% (Giles 1970). Buckner and McLeod (1975) monitored the impact of fenitrothion on small forest mammals and found that juvenile numbers were reduced at an application rate of 0.4 lb/acre and the number of adults was reduced at 0.9 lb/acre. The seasonal reproductive cycles of the mammals were interrupted at 1.3 lb/acre, but were restored to normal within the year. Brood abandonment and, possibly, a reduction in the invertebrate prey base reduced the survival of mallard (*Anas platyrhynchos*) and blue-winged teal (*Anas discors*) ducklings from 58% on control fields to 16% on fields receiving aerial application of 1.6 lb/acre of methyl parathion (Brewer et al. 1988). Possible declines in forest bird populations have been observed following applications of carbaryl (DeWeese et al. 1979; Zinkl et al. 1979). A gradual reduction in bird species diversity has also been reported (Moulding 1976). However, other studies at comparable application rates have not resulted in changes in bird numbers or diversity following carbaryl sprays Bart 1979; DeWeese et al. 1979). Although malathion is considered to be one of the safest of the anticholinesterase pesticides in use today, it is one of the few pesticides for which measurable reductions in wild mammal populations are documented. Application of 0.81 kg/ha malathion was shown to reduce mice and chipmunk populations by 30% (Giles 1970). Effects on avian populations, on the other hand, are minimal and transient (McEwen 1982; Giles 1970). The toxicity of malathion is potentiated by many other pesticides (Casarett and Doulls 1986), making application history and spray drift important considerations prior to spray.

Organochlorine Insecticides

OC insecticide use was sharply curtailed in the 1970s when it was established that DDT and related compounds caused eggshell-thinning in several species of piscivorous and raptoral birds (Cooke 1973). Banning or curtailing the use of OCs mitigated the reproductive and population declines observed in sensitive avian species (Peakall et al. 1975; Spitzer et al. 1978; Blus 1982; Grier 1982; Wiemeyer et al. 1984). However, continuing environmental contamination by DDE raised concerns that contamination of some of the safer OCs still registered for use in the United States may be contributing to DDE burdens in fish and wild birds. In particular, the U. S. (EPA) was concerned about the continued use of dicofol products (Table 1), which contain DDT and related compounds (Moore 1986; Hunt et al. 1986). As a result of the registration review of dicofol, EPA required that the combined content of DDT, DDE, and DDD in dicofol formulations be $\leq 0.1\%$ by the end of 1988. However, recent studies investigating the reproductive impact of dicofol itself showed that although DDE/DDT concentrations were minimal in screech owls (*Otus asio*) fed diets containing the new dicofol formulations, shell weight and thickness were reduced by more than 15% (Wiemeyer et al. 1989). Dicofol is the only environmental contaminant other than DDT and DDE known to cause this magnitude of eggshell thinning (Cooke 1973; Wiemeyer et al. 1984). It has been shown that bird populations experience reproductive problems when eggshell thickness is reduced by 15 to 20% for several years (Anderson and Hickey 1972). Indeed, no raptor population in North America has maintained itself while experiencing a reduction in shell thickness of $\geq 18\%$ (Lincer 1975). Dicofol fed to doves not only caused eggshell thinning but also reduced egg production (Schwarzbach et al. 1988). Although there is a potential for dicofol-induced reproductive impairment in birds, concentrations shown to cause eggshell thinning and reduction in clutch size (i.e., 3 ppm) have not been reported in wildlife to date (Clark et al. 1990). It should be noted, however, that dicofol burdens in wildlife food chains have not been systematically monitored. In regions of intense dicofol use, dicofol has been found in 76% of the eggs collected from 6 species of birds (Clark et al. 1990). The concentrations of dicofol (and its metabolites) in eastern screech owl eggs were the highest and ranged from 2.1 ppm to 4.6 ppm (Clark et al. 1990). In general, these values fall below DDE levels in eggs associated with nest failure in most bird species. Only in those species most sensitive to eggshell thinning, such as the brown pelican (*Pelecanus occidentalis*), have comparable levels of DDE (i.e., 3 ppm DDE) in eggs caused reproductive decline (Blus 1984). The presence of species of comparable sensitivity to eggshell-thinning as the brown pelican and the potential for chronic food-chain exposure should be considered in areas of intended high dicofol use.

Fish are the organisms that are at greatest risk from exposure to most of the OCs listed in Table 1. For example, the 24-h LC_{50} for endosulfan in rainbow trout (*Salmo gairdneri*) is 2.1 µg/L and as low as 0.02 µg/L in harlequin fish (*Rasbora heteromorpha*) (Schoettger 1970). To avoid impacting aquatic systems, the product label prohibits application or drift of the OC pesticides to lakes, streams, or ponds. Often a buffer zone of up to 200 m from the target site is established to prevent deposit of the highly toxic insecticide in aquatic systems (Payne et al. 1988). Yet, large fish kills have been associated with the operational use of endosulfan (National Research Council of Canada 1975). Recent studies on the off-target deposition of endosulfan from conventional agricultural sprays applied by aircraft showed that the commonly used buffer zones may not be sufficiently large to protect aquatic systems (Ernst et al. 1991). Endosulfan deposits at 200 m from the target area resulted in 90% mortality in a moderately sensitive fish species, the threespine stickleback (*Gasterosteus aculeatus*) (LC50 of 7.8 µg/L). More than 50% mortality was observed in some aquatic invertebrates at 50 m.

Synthetic Pyrethroid Insecticides

Synthetic pyrethroids (SPs) pose very little direct threat to wildlife because of their very low mammalian and avian toxicity. However, these chemicals are extremely toxic to fish. About 40% of the LC_{50}s values for fish are below 1 µg/L (Jolly et al. 1978; Stephenson 1982). However, field applications of synthetic pyrethrins have not resulted in fish kills. Larvidical application rates have caused sublethal effects in fish. These effects include gill necrosis, erratic swimming, altered schoolng activity and other behavioral changes and may lead to reduced filter-feeding, reproduction, and survival under field conditions (Coats et al. 1989; Day and Kaushik 1987; Day 1989).

Aquatic invertebrates are the most susceptible organisms to pyrethroids. Exposure of aquatic habitats to the SPs from spray drift have caused changes in invertebrate abundance (Smith and Stratton 1986). Reduction in invertebrate abundance from SPs has caused changes in feeding habits and indicates potential indirect impacts from reduced quantity and quality of food (Kingsbury and Kreutzweiser 1987). Lethal concentrations of SPs for most invertebrates are also below 1 µg/L (Anderson 1982; Friesen et al. 1983; Day and Kaushik 1987; Mokry and Hoagland 1990). Concentrations in streams contaminated by spray drift were 0.4 to 1.7 µg/L (Crossland et al. 1982). Forest environments treated with aerial applications of permethrin at 8.8 to 70 g/ha resulted in stream concentrations of 0.23 to 1.80 µg/L (Kingsbury and Kreutzweiser 1987). Loranzo et al. (in press) evaluated the effects of the synthetic pyrethroid, esfenvalerate, on the aquatic organisms in 12 littoral enclosures in a 2-ha pond. Drastic reductions in or elimination of most crustaceans, chironomids, juvenile bluegills (*Lepomis macrochirus*), and larval cyprinids occurred within the first 4 days of application at exposures of 1 and 5 µg/L esfenvalerate. Rapid knockdown of aquatic insects has been observed when applications have occurred near streams and ponds (Smies 1980). Caddis fly naiads and rotifer populations were greatly reduced by exposure to trace amounts of SPs, but populations recovered quickly. Mayfly populations did not recover for the entire season once exposed. Permethrin levels of 0.5 to 5 µg/L reduced zooplankton diversity and the density of larger cladocdrans and copepods in aquatic enclosures in a small euthrophic pond and a 10-ha mesotrophic lake (Yasuno et al 1988; Kaushik et al 1985). In contrast to the water column, sediment and plant concentrations of esfenvalerate are persistent and cumulative (Loranzo et al. in press; Heinis and Knuth in press). Accumulation of esfenvalerate in these sorptive pools resulted in delayed and long-term reduction in littoral invertebrates (Loranzo et al. in press). At concentrations of 0.08 to 0.2 µg/L the abundance of several copepod and insect genera were reduced. These invertebrate communities were impacted for 25 to 53 days following application.

Herbicides and Fungicides

Those pesticides demonstrating very low toxicity to both fish and wildlife are, predominantly, chemical herbicides and biorational pesticides (i.e., chemical analogues of naturally occurring biochemicals such as growth regulators and various pest control agents such as bacteria). Based on current data, use of these pesticides in agricultural environments cause very little to no adverse impact to fish or wildlife populations when used in accordance with label restrictions.[a] Some low-risk pesticides are listed in Table 2. However, some herbicides and fungicides exhibit high toxicity to aquatic fauna. Although most of the toxic herbicides and fungicides are strongly adsorbed to the soil (Fadayomi and Warren 1977; Weed Science Society of America 1983) and are unlikely to leach into aquatic systems, concern over the potential impacts to fish arises from the spray drift contamination recently observed for OC compounds. Because OC compounds have toxicities, application rates, and current application restrictions (i.e., buffer zone sizes) similar to toxic herbicides and fungicides, off-target deposition of OCs may also be harmful to fish.

Direct Effects on Wildlife

The herbicides listed in Table 1 pose little direct risk to avian or mammalian species. A possible exception is the triazine herbicide simazine, which appears to be moderately toxic to ruminant animals (Palmer and Radeleff 1964). To protect wild ruminants (e.g., Cervidae), applications that result in consumption of ≥ 50 mg/kg daily for more than 14 days should be avoided. Typically, the equivalent of 2 to 4 lb/acre of simazine are applied for selective weed control under most conditions. (Higher concentrations are used for non-selective weed control.) Assuming a forage consumption factor of 3% of the body weight (Buck et al. 1976), about 1,800 lb/acre of edible biomass (grass + forbes) in clearcuts (McCorquodale in press), and no degradation of the herbicide with time, a deer would consume a maximum of about 13 mg/kg of simazine per day following a 4 lb/acre application. Ingestion of simazine may also be reduced by the declining palatibility of the poisoned plants. Only if much higher rates are applied in multiple applications is there likely to be any adverse impact to ruminant wildlife from this herbicide.

Few field data are available on the persistence, fate, and wildlife impacts of the fungicides listed in Table 1. However, the toxicities of the fungicides are relatively low and the potential for acute lethal exposure is minimal. There have been no reports of these fungicides causing adverse effects in fish or wildlife. Although data on invertebrate toxicity is lacking, it is unlikely these compounds will have deleterious effects on invertebrate populations at environmental levels. Ziram has been shown to both inhibit and stimulate algal growth (Vyas 1988). However, disruption of food web structure by this chemical has not been demonstrated.

Bacillus thuringiensis poses no direct risk to wildlife or aquatic fauna. Broad-scale use, however, may impact wildlife species that feed predominantly on the target pest.

Indirect Impacts on Wildlife

Some of the low-risk herbicides are toxic to invertebrates and could indirectly impact fish and wildlife populations by reducing their prey base. Among the most toxic herbicides to invertebrates are 2,4-D esters, picloram, atrazine, and simazine. However, the median lethal concentrations (LC_{50}s) for these herbicides are one to two orders of magnitude less toxic to aquatic organisms than most of the insecticides (Sheenan et al. 1986) and will probably have little impact on food resources of fish and wildlife.

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[a] Impacts to fish and wildlife from intentional habitat modification as a result of herbicide use are not considered in this evaluation.

Table 2. Low-Risk Pesticides to Wildlife

	Toxicity Class[a]				
	Aquatic Invertebrates	Fish	Birds		Mammals
Pesticide	(LC_{50})	(LC_{50})	Acute Oral (LD_{50})	Dietary (LD_{50})	Acute Oral (LD_{50})
Herbicides					
Phenoxy					
MCPA	-[b]	L	-	L	M
Fluazifop-butyl	L	L	L	L	L
Dichloroprop	L	M	L	L	L
Dalapon	-	L	L	L	L
2,4-D	L	L, H[c]	L	L	M
Pyridines					
Trichlopyr	M	L	L	L	M
Picloram	M	M	L	L	L
Clopyralid	-	L	L	L	L
Triazines					
Atrazine	L	M	L	L	L
Hexazinone	-	L	L	L	L
Simazine	L	M	L	L	L
Other Herbicides					
Dicamba	L	M	L	L	L
Dichlobenil	L	M		L	L
Glyphosate	L	L	L	L	L
Imazapyr	L	L	L	L	L
MSMA	-	L			M
Oryzalin	-	M	L	M	L
Pronamide	-	M	L	-	L
Sufometuron-methyl	-	-	-	-	L
Fungicides					
Metalaxyl	-	L	L	L	M
Ziram	-	M	M	L	L
Chlorothalonil	-	M	L	L	L
Insecticides/Nematicides					
Bacillus thuringiensis	L	L	L	L	L
Dichloropropene	L	M	L	-	M

(a) Toxicity classes modified after Grue et al. (1986): invertebrates 96 h LC_{50} (synthetic pyrethroids 24 h LC50), Low (L) = >1.0, Moderate (M) = 0.01-1.0, High (H) = 0.001-0.01, and Very High (VH) = <0.001 mg/L; fish 96 h LC_{50} (mg/L), L = > 100, M = 1-100, H = 0.1-1, VH = < 0.1; birds and mammals (mg/kg), L = >1000, M = 201-1000, H = 41-200, VH = < 40. Values from Verschueren (1983), Smith (1987), Hudson et al. (1984), Weed Society of America (1983), and Royal Society of Chemistry (1990).

(b) Information not available

(c) Some formulations are toxic to fish, e.g., acid esters; others are not toxic.

Because phytoplankton form the food base for many aquatic organisms, the indirect impacts of herbicides to aquatic and terrestrial fauna through food web disruption must be considered. The triazine herbicides are soluble in water and are among the pesticides showing the greatest losses to surface waters (Wauchope 1978). Phytotoxic levels of triazine residues are commonly found in lakes, streams, and groundwater as a result of run-off from treated agricultural fields (Bodo 1991; Eisler 1989). Peak triazine levels in watersheds occur after storm events within the summer spray season and persist at diminished but detectable levels into the following year (Roberts et al. 1979; Frank et al. 1982; Frank and Logan 1988; Triplett et al. 1978). Concentrations causing nonreversible adverse effects in sensitive aquatic plants and algae have not been documented over extended periods in the environment (Eisler 1986). Persistent concentrations (i.e., the concentrations expected in the water column during the winter months) of these photosynthesis inhibitors have been shown to cause transient changes in phytoplankton composition and abundance (DeNoyelles et al. 1982; Jenkins and Buikema 1990) during laboratory tests. However, the relative lack of dependence of winter zooplankton communities on autotrophic phytoplankton (Jenkins and Buikema 1990) indicates that triazine applications are unlikely to have even indirect deleterious impacts on fish or wildlife food sources under field conditions.

Field and laboratory studies with glyphosate and the pyridine herbicides triclopyr and picloram show that natural dissipation mechanisms reduce exposure to these herbicides to safe levels in aquatic systems even for direct overhead sprays (Thompson et al. 1991; Fontaine 1990; Morgan et al. 1991; Norris et al. 1987; Wan 1986; Newton et al. 1984; Mayes et al. 1987). Maximum levels of herbicides in streams following direct overhead aerial application have been reported to be 0.1- to 0.35 mg/L for triclopyr (Norris et al. 1987; Thompson et al. 1991) and 0.02- to 0.27 mg/L for formulations of glyphosate (Wan 1986; Newton et al. 1984). These levels are well below toxic concentrations for fish and invertebrates. However, exposure to sublethal concentrations may reduce survivability and reproduction in sensitive fish species. Changes in behavior that may increase exposure to predation, and affect feeding, migration, and reproduction patterns have been observed in fish exposed to herbicides (Sandheinrich and Atchison 1990; Little and Finger 1990; Morgan et al. 1991). In particular, salmonids have been shown to abandon localized habitats contaminated with triclopyr and glyphosate (Hildebrand et al. 1982; Morgan et al. 1991). Concern that sublethal concentrations of these chemicals may cause acute behavioral and physiological changes in juvenile salmonids has led to the determination of threshold response levels for triclopyr, picloram, and glyphosate (Mayes et al. 1987; Morgan et al. 1991). Threshold concentrations causing behavioral changes, including avoidance, in salmonid fry are two to three orders of magnitude higher than the maximum levels reported for streams receiving overhead applications. Thus, little impact on fish or wildlife populations is anticipated from the use of these herbicides. Establishment of protective buffer zones (as required by label) further increases the margin of safety for conventional applications, particularly for those herbicides with moderate toxicity to fish (Table 1).

Soil Fumigants

Dazomet decomposes in water and moist soil to form the fumigant methyl isothiocyanate. It is highly toxic to fish and label restrictions prohibit its application near aquatic habitats. Methyl bromide is a highly toxic gas. The major potential harm from these fumigants is exposure of ground or near-ground nesting animals and burrowing mammals during breeding season, exposure of congregating animals (e.g., migration), or endangered or threatened species. Ground disturbance/preparation prior to application of the fumigants may discourage nesting, but may encourage foraging on treated ground. Reduction of rodent prey base may affect nesting raptors if use of these compounds is extensive. However, extensive applications are unlikely because of application restrictions. No wildlife kills have been associated with the use of these soil fumigants and, because these compounds evaporate rapidly, any effects will be transient.

Rodenticides

Broad-scale or improper use of the rodenticides listed in Table 1 have resulted in many deaths of nontarget organisms, including threatened or endangered species (Hudson et al. 1984). Diphacinone is an anticoagulant rodenticide that is very toxic to mammals, but not to birds (Royal Society of Chemistry 1990). It is generally used in bait form and requires multiple feedings to produce lethal effects. Strychnine does not require multiple exposures to be lethal and is highly toxic to both birds and mammals. Safe use of these compounds is dependent on site- and target species-specific handling and baiting to minimize contact with this rodenticide by nontarget species. Neither of these compounds should be used in areas where protected species may come in contact with the bait, especially forest edge areas, are particularly hazardous localities for baiting.

Factors Influencing Vulnerability of Fish and Wildlife to Pesticides

From the above discussion the following factors have been found to influence the vulnerability of fish and wildlife to pesticides. Particular emphasis is given to the pesticides with high potential for harming nontarget vertebrate organisms. The factors described below should be taken into consideration when application of a pesticide in forested lands is being evaluated. Possible mitigation measures are identified for each factor.

Previous Exposure to OPs or CBs

Because of the potential for accumulative toxic action from repeated exposure to OPs and possible potentiation of toxicity from prior OP exposures, consideration of recent spray history in any proposed application area should be made prior to OP application. Multiple applications of OPs should be minimized or spaced several weeks apart to allow the nervous system to recover to at least 80% of normal. The potential for spray drift from or to neighboring areas should considered in the spray history. Spray drift can cause a significant fraction of the total application to be deposited in nontarget areas (Crabbe et al. 1980; Crabbe and McCooyee 1985).

Temperature Extremes

Small birds and mammals with high metabolic rates are most susceptible to OP poisoning. To reduce the impact of environmental stressors on the survival of exposed animals, applications of OPs should not be made during temperature extremes.

Presence of Protected Species

The population stability of species with critically low populations (endangered species) can be affected by a small increase in mortality; therefore, use of high-risk pesticides should be avoided when such species are present. Dicofol causes eggshell thinning and should be limited in use to areas supporting sensitive bird species.

Timing of Application

Pesticides are frequently applied in the morning when weather conditions are optimum for application; however, many wildlife species are actively foraging at this time. To minimize exposure of animals that forage or nest in the treated habitat (e.g., forest canopy, or clear-cuts) to the OPs

and/or CBs and to predation, applications should be made during late evening when most forest birds and mammals are relatively inactive. Up to 80-90% of the deposited pesticide applied over foliated areas is intercepted by the crown canopy of trees. Birds that forage in or nest in the canopy are those most affected by OP/CB applications in forested areas (Pearce et al. 1979; DeWeese et al. 1979). Pearce (1971) reported that the hazards of fenitrothion to forest birds were reduced when spraying took place during the late evening. (Note: application must always be performed during those meteorological conditions that will minimize drift and may not always be possible during evening hours.)

Interaction of Lifecycle and Timing of Application

Poor physiological condition (diminished fat reserves) can cause birds and mammals to ingest greater quantities of contaminated food or be less tolerant to OP-induced anorexia (Hill 1972; Pope and Ward 1972; Grue 1982). To avoid die-offs, OPs should not be applied when migrants are present or when the reproductive period has reduced fat stores (e.g., post-nuptial molt), particularly in small mammals and birds with high metabolic rates.

Concentration of Population in Small Area

High-risk pesticides should not be applied when migrants are present. Granular formulations or baits should be avoided at times and in locations where seed-eating birds congregate (Eisler 1986; Stone and Gradoni 1985b). Discoloration of rodenticides with dyes can afford a measure of protection to birds.

Dependence of Invertebrate Food Supply

Young birds, particularly altricial birds (Grue et al. 1983), and young mammals (Brodeur and DuBois 1967; Benke and Murphy 1975) appear to be more sensitive to pesticides than adults. Timing of sprays to avoid the period when young are dependent on invertebrate food sources will reduce the potential impact of food reduction on the growth and survival of young birds and mammals. Alternatively, spraying areas smaller than the home or foraging range of sensitive species may be considered. Specificity and efficacy of the pesticide should be considered and, in general, broad-scale applications of diazinon, parathion, chlorpyrifos, carbofuran or the SPs should be avoided in areas or at times when young animals dependent on invertebrates as their main food source are present.

Avian Embryo Sensitivity

Oil carriers should be avoided during egg-laying and incubation periods. External exposure of mallard eggs to naled and parathion resulted in embryo mortality when the OPs were applied in an oil vehicle (Hoffman and Eastin 1981). Recent information on salmon returns indicates that oil contamination during egg hatching resulted in reduced ability to identify home streams (Alaska State Department of Environmental Conservation, personal communication). Application of oil carriers to watersheds that drain to spawning areas or support incubating birds should be avoided.

Buffer Zones

Pesticides such as the SPs and several of the OCs are found in harmful concentrations in aquatic systems indicating that common use buffer zone widths may not be sufficiently large enough to

protect sensitive aquatic organisms. Further investigation of appropriate buffer zone distance should be conducted. Meanwhile, applications of these pesticides at distances of less than 200 m should be avoided.

Conclusions

The safety of applying any of the higher risk pesticides is greatly influenced by factors such as temperature, previous pesticide exposure, timing of spray, presence of protected or sensitive species, and off-target drift. Therefore, ca reful consideration of the potential exposure and vulnerability of fish and wildlife in or near any area of intended pesticide application should be made prior to use.

In general, insecticides are more toxic to vertebrate animals than are herbicides (Hudson et al. 1984; Hill and Camardese 1986); (Table 2). Most herbicides and biorational pesticides cause little or no adverse impacts to fish or wildlife populations when used in accordance with label restrictions. Of the insecticides registered for agricultural use, the organophosphate (OP) and carbamate insecticides (CB) pose the greatest direct and indirect risk to both fish and wildlife. A few of these compounds appear to have a significant potential for causing harm to wildlife at label-recommended use rates. Other members of these classes pose lesser hazards to fish and wildlife. Many of the anticholinesterase insecticides are not harmful except when applied under conditions that increase the vulnerability of nontarget organisms (e.g., previous exposure to insecticides) to toxic stress. The organochlorine insecticides are less overtly toxic than the OPs and CBs and their use poses little direct risk to wildlife under current label restrictions. However, these compounds have a high potential for impacting fish populations even when protective buffer zones are employed. The synthetic pyrethroid insecticides (SPs) have little potential for causing direct harm to wild animals at recommended application rates, but can be transported to aquatic systems and adversely affect invertebrate communities that support fish and waterfowl populations. Sublethal effects of the pyrethroids that may lead to reduced reproduction and survival of fish have been observed under field conditions. Some herbicides and fungicides are highly toxic to fish and have the potential of contaminating aquatic systems at harmful levels via spray drift. However, they pose only a moderate or low risk of harm to fish populations because of application practices or environmental degradation processes that limit exposure. In contrast, the difficulty in restricting exposure causes the highly toxic rodenticides to be considered high-risk pesticides in agricultural communities. Soil fumigants are highly toxic to fish but exposure is usually limited to ground dwelling or burrowing animals.

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