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FIELD STUDIES ON PESTICIDES AND BIRDS: UNEXPECTED AND UNIQUE RELATIONS

LAWRENCE J. BLUS AND CHARLES J. HENNY

U.S. Geological Survey, Biological Resources Division, Forest and Rangeland Ecosystem Science Center, 3080 S.E. Clearwater Drive, Corvallis, Oregon 97333 USA

Abstract. We review the advantages and disadvantages of experimental and field studies for determining effects of pesticides on birds. Important problems or principles initially discovered in the field include effects of DDT (through its metabolite DDE) on eggshell thickness, reproductive success, and population stability; trophic-level bioaccumulation of the lipid-soluble organochlorine pesticides; indirect effects on productivity and survival through reductions in the food supply and cover by herbicides and insecticides; unexpected toxic effects and routes of exposure of organophosphorus compounds such as famphur and dimethoate; effects related to simultaneous application at full strength of several pesticides of different classes; and others. Also, potentially serious bird problems with dicofol, based on laboratory studies, later proved negligible in the field. In refining field tests of pesticides, the selection of a species or group of species to study is important, because exposure routes may vary greatly, and 10-fold interspecific differences in sensitivity to pesticides are relatively common. Although there are limitations with field investigations, particularly uncontrollable variables that must be addressed, the value of a well-designed field study far outweighs its shortcomings.

Key words: birds; direct effects; experimental studies; field studies; indirect effects; pesticides; sensitive species.

INTRODUCTION

Traditionally, most toxicity data related to pesticide registration were obtained from relatively short-term tests under uniform conditions and generally with few species. Obviously, this process has a number of positive aspects, but there are drawbacks, some of which are of prime concern for interpreting toxicity to wildlife populations. Because field studies with pesticides are, or should be, conducted in an ecological context with the toxic insult being one of many variables, it is obviously simpler and less expensive to rely upon experimental studies under controlled conditions in pens or laboratories. The limitations of projecting experimental results to field situations become apparent when one reviews findings of field research studies like ours or any number of other contaminant investigations during the last 50 years. One could say that field research documents problems that slip through the system. Herein, we describe the uniqueness and indispensability of field studies in interpreting toxic effects of pesticides on birds (Table 1). With some limitations, these principles probably apply to all components of the biota.

The objectives of this review are to: (1) identify the benefits and shortcomings of experimental studies for predicting effects of pesticides on wild populations, (2) describe the benefits of field studies as well as their disadvantages, and (3) offer suggestions for refining field-testing protocols that identify and utilize the most sensitive species, conditions, end points, and habitats in order to decrease important problems from pesticides.

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EXTRAPOLATION FROM EXPERIMENTAL STUDIES TO THE FIELD

When one attempts to use LD_{50}, LC_{50}, reproductive effects, and other data from experiments to predict potential effects of a pesticide on wild bird populations, the extreme complexity of this exercise rapidly emerges (Table 1). One of the major difficulties is that limited numbers of avian species, primarily quail and a few other upland game birds, waterfowl (particularly ducks), and some song birds, are used extensively in the experiments conducted under uniform conditions. In the field, wide arrays of bird species are exposed to pesticides under widely varying conditions related to routes of exposure, presence of other pesticides or pollutants, weather, food supply, cover, and a host of other environmental factors. Selection of those avian species most sensitive to a specific pesticide for use in experiments or field studies is complicated by the myriad of forms at risk from exposure, possible shifts in interspecific sensitivity from one compound to another, dif-

TABLE 1. Important effects, principles, or practices initially discovered or expanded through field studies of pesticides and birds.

Discovery	Discovery site: Laboratory	Discovery site: Field	Pesticides involved	Rationale for being overlooked, not measured, or inadequately followed up in laboratory experiments
Eggshell thinning		X	DDT†	DDE not used; some species insensitive
Decreased productivity	X		OCs‡ antiChEs herbicides	No studies on pesticides such as heptachlor; study design allows limited interpretation of field data
Population decrease/extirpation		X	OCs‡ antiChEs§ herbicides	Endpoint not measurable
Bioaccumulation		X	OCs	Inadequate numbers and types of organisms
Decreased food supply/cover		X	OCs antiChEs herbicides	Endpoint essentially not measurable
Endocrine disruption	X		OCs	Little follow-up after initial discovery; eggshell thinning, female–female pairing, and skewed sex ratios discovered in field
Unique route of exposure		X	famphur	Method of application considered low risk
Species sensitivity	X		many	Limited probability of finding sensitivity among the relatively few species used
Simultaneous application		X	many	Practice not widely known or considered unimportant
Illegal application/misuse		X	many	Birds in most experiments exposed to high levels of pesticides, but it is difficult to simulate field conditions

† The parent material is listed, even though its metabolite, DDE, is responsible for eggshell thinning.
‡ OCs (organochlorines) account for most of these problems.
§ AntiChEs are compounds that affect brain cholinesterase activity.

ficulty in keeping these sensitive species in captivity, their scarcity in the field, and other factors. Also, most experiments delineating reproductive effects are designed so that they are of limited use in interpreting residues in eggs and tissues in relation to the field situation (Blus 1995). Experimental studies that have established lethal levels of the lipid-soluble organochlorine compounds in critical tissues such as the brain are very helpful in interpreting pesticide-induced mortality in the field (Bernard 1963, Stickel et al. 1966).

Few biomarkers developed during experimental studies have proved useful for interpreting effects of pesticides on wild birds. An exception is the assay of cholinesterase activity, particularly in the brain, that is used widely to identify exposure or probable death of birds from anticholinesterase (antiChE) compounds (Hill and Fleming 1982). Another important biomarker, eggshell thinning, was discovered in the field (Ratcliffe 1967) and was then verified experimentally (Heath et al. 1969).

FIELD STUDIES OF DIRECT EFFECTS OF SPECIFIC PESTICIDES

DDT and metabolites

Although DDT (an organochlorine [OC] pesticide) was known to be moderately toxic to experimental birds and to cause die-offs of large numbers of birds under circumstances of unusually heavy applications (Hickey and Hunt 1960, Wurster et al. 1965), field studies revealed that its most devastating effect was sublethal to adult birds. The discovery of eggshell thinning in eggs of Peregrine Falcons (*Falco peregrinus*) and Eurasian Sparrowhawks (*Accipiter nisus*) in Great Britain (Ratcliffe 1967) immediately led to experimental studies, verifying that DDE, a metabolite of DDT, was responsible for eggshell thinning and lowered reproductive success in birds (Heath et al. 1969). Subsequently, a virtual flood of papers verified the link between DDE and eggshell thinning and reproductive success in both experimental and wild birds (Hickey and Anderson 1968, Wiemeyer and Porter 1970, Cooke 1973). In wild populations, average eggshell thinning $\geq$18% over several years was associated with population declines (Hickey and Anderson 1968). Although other contaminants may cause some eggshell thinning, the major role of DDE in eggshell thinning has been established in many studies.

The use of eggshell thinning as a biomarker was not verified by >20 years of extensive experimentation with DDT and birds, and there was no evidence whatsoever regarding the role of DDE. One problem was that species tolerant to DDT were used in many of these studies. As a result, there was no way to establish the link between DDT and lowered reproductive success and population declines in wild birds without first learning about its seemingly innocuous metabolite, DDE. DDE is not used as a pesticide, but it is the metabolite found most commonly and at highest concentrations in avian tissues and eggs. With the 1972 ban on most uses of DDT in the United States and subsequent ban on this material in much of the world, many populations of Brown Pelicans (*Pelecanus occidentalis*), Peregrine Falcons, Osprey (*Pandion haliaetus*), Bald Eagles (*Haliaeetus leucocephalus*), Double-crested Cormorants (*Phalacrocorax auritus*), and other species have experienced total or partial recovery (see review by Blus 1995).

DDD is a metabolite of DDT that usually ranks second in occurrence to DDE in tissues and eggs of wild birds; it has received some use as an insecticide. Its most noted use occurred in California, where it was applied three times from 1949 to 1957 to Clear Lake to control gnats. The number of breeding pairs of Western Grebes (*Aeschmorphorus occidentalis*) decreased from 1000 to zero in the year following the first spray; the population remained depressed for many years (Hunt and Bischoff 1960, Herman et al. 1969). Although the local extirpation of grebes was a significant event, the Clear Lake study was probably the first to document trophic-level bioaccumulation that was essential in explaining the persistence of residues and residual effects of the OCs (Keith 1991).

Field experiments with radiolabeled-DDT in aquatic and terrestrial environments (Meeks 1968, Forsyth et al. 1983) verified trophic-level accumulation in a number of food chains, with those organisms atop each food chain generally having the highest residues.

Heptachlor

The extreme toxicity of heptachlor (an OC insecticide) to birds was known from both experimental and field studies (Blus 1995). Despite this, there were no studies of the effects of heptachlor on reproductive success of birds until field studies of the Canada Goose and American Kestrel (*Falco sparverius*) were conducted in the late 1970s (Henny et al. 1983, Blus et al. 1984). The heptachlor originated from treated wheat seed ingested directly by the geese, and secondarily or tertiarily by kestrels feeding on prey that had ingested the wheat; the heptachlor was readily metabolized to heptachlor epoxide after ingestion. Kestrels were much more sensitive to heptachlor than were geese. The lower effect level of heptachlor epoxide in eggs that was associated with adverse effects on nest success was <3 $\mu g/g$ for the kestrels and ~ 10 $\mu g/g$ for the geese. As predicted by toxicity tests, die-offs of adult Canada Geese and other seed eaters occurred, and there was also some mortality of adult kestrels and other raptors (Henny et al. 1983, Blus et al. 1984). A local goose population declined in the area where the treated wheat seed was used; the population recovered after lindane, a much shorter lived OC, was substituted as a seed treatment.

Famphur

Famphur (Warbex) is an organophosphorus (OP) insecticide used to control warbles (fly larvae) in cattle. The liquid is poured on the back of the animal at the midline; significant quantities may remain on the hair and skin for months. The material acts as a systemic, and larvae are killed in the bloodstream. Under this treatment regime, problems with wildlife seemed remote. Death of a captive Great-horned Owl (*Bubo virginianus*), after it was fed Black-billed Magpies (*Pica pica*) found dead near a cattle treatment area, stimulated a study in the state of Washington to determine effects of famphur on birds (Henny et al. 1985). A field experiment with magpies involved two similar study areas. One area had no cattle and served as a reference. The second area consisted of small ranches where famphur use was rare; we selected this area to treat cattle with famphur at the recommended rate. Following application of famphur, dead magpies were found throughout the treated area and a local population decline ensued there, but not in the reference area. At the time, Black-billed Magpies were undergoing a regional population decline. A Red-tailed Hawk (*Buteo jamaicensis*) died from secondary poisoning in the Washington study area after eating a magpie contaminated with famphur. Subsequently, numerous cases of secondary poisoning and one case of tertiary poisoning of hawks, eagles, and owls were recorded from throughout the United States (Henny et al. 1987). Some deaths were related to lacing of livestock carcasses with famphur to intentionally kill birds (White et al. 1989). Death from famphur was verified by assaying brain cholinesterase (ChE) activity and by detecting famphur residues in the upper gastrointestinal (GI) tract.

There was no indication from laboratory experiments that famphur as a pour-on would pose a risk to birds, even though its extreme toxicity was established. Apparently, none of the initial toxicity testing involved the pour-on formulation. The persistence of high residues on hair for months after application and the habit of magpies and several other species to associate with cattle and to acquire toxic levels, especially from ingestion of treated hair, was a unique scenario that had no parallel in experimental studies.

Dimethoate

Die-offs of Sage Grouse (*Centrocercus urophasianus*) in Idaho were attributed to OP insecticides. This led to a radiotelemetry study to determine and quantify effects of pesticides on Sage Grouse (Blus et al. 1989). In late spring, some Sage Grouse move from sagebrush to cropland, especially alfalfa fields, to feed on the succulent foliage until late summer. At the sagebrush–cropland interface, we captured and radio-collared 82 grouse. Nearly 20% of the radio-marked Sage Grouse were known to occupy alfalfa fields when they were sprayed with the OP dimethoate. Of these, 16% of the juveniles and none of the adults died from dimethoate poisoning, as revealed by brain ChE activity and presence of residues in the upper GI tract (Blus et al. 1989). Intoxicated or dead grouse were found around five alfalfa fields sprayed with dimethoate in Idaho. One 240-ha alfalfa field sprayed with dimethoate contained 200 Sage Grouse, including six radio-marked birds, five of which subsequently died. Dead and sick grouse were recorded from the day after spray until 11 days later. In all, we found 63 dead grouse (including five radio-marked birds); most of these were verified as dying from dimethoate. We attached radio collars to 32 grouse found intoxicated in or near sprayed fields; ≥20 (63%) of these died from dimethoate.

The Sage Grouse study was somewhat unique, in that no other direct mortality of wild birds was attributed to dimethoate (Smith 1987), and a marked population of birds was used to estimate mortality from pesticides. Sage Grouse feed extensively on alfalfa foliage, even immediately after spraying, and this may have made them particularly vulnerable to dimethoate. Dermal absorption and respiration are other routes of exposure. In Argentina, a die-off of Swainson's Hawks (*Buteo swainsoni*) occurred after dimethoate was reportedly sprayed on alfalfa (Goldstein et al. 1996); however, no samples were collected for ChE assays or residue analysis (M. Goldstein, *personal communication*). Other studies of birds, particularly waterfowl occupying fields treated with other antiChE compounds, describe relatively frequent die-offs that seem to be related most commonly to ingestion of treated vegetation (Smith 1987). Also, Sage Grouse may be unusually sensitive to dimethoate because of physiological attributes associated with metabolism or excretion, because no other bird mortality from dimethoate has been reported. The major die-off of Sage Grouse occurred during extremely hot weather in August; stress, including extreme temperature, can exacerbate the toxicity of antiChE compounds (Rattner 1982).

Effects of dimethoate on upland game bird populations in Great Britain will be discussed later under *Indirect Effects.*

Dicofol

In most of the examples cited previously, field studies discovered problems or defined biomarkers that were not identified in laboratory studies. The reverse held true for dicofol or Kelthane (an OC insecticide), in that it was considered a potentially serious risk to bird populations. Experimental birds given diets containing this organochlorine pesticide laid eggs with thin shells, experienced reduced reproductive success, and accumulated residues of dicofol and metabolites in their eggs and tissues (Wiemeyer et al. 1989, Clark et al. 1990). Nevertheless, no adverse effects were documented in field studies, and rare detections of dicofol in the biota were generally low (Clark 1990, Clark et al. 1995). Thus, the ecotoxicological ramifications were unclear until field studies were completed. The apparent contradiction of the degree of risk derived from laboratory vs. field studies was primarily due to the relatively short half-life of dicofol after it is applied to the environment (Clark et al. 1995). Because dicofol is produced, from DDT as the starting material, DDT and a number of its metabolites are contaminants of dicofol. Potential problems from dicofol were reduced when regulations stipulated a reduction in DDT and its metabolites to ≤0.1%.

Simultaneous application of pesticides

In the United States and possibly other countries, it is legal to mix and simultaneously apply pesticides at full strength unless the label specifically prohibits a particular combination. For example, Blus et al. (1991) reported mortality of 160 Canada Goose (*Branta canadensis*) goslings when a seed alfalfa (*Medicago sativa*) field was sprayed with the OP insecticides disulfoton and dimethoate, and the carbamate insecticide carbofuran. Disulfoton and carbofuran were applied at full strength, whereas a small amount of dimethoate was added to the mixture. Carbofuran appeared to cause deaths of the goslings, because carbofuran residues in alfalfa ingesta constituted an apparently lethal dose. It must be noted that pesticides on the ingesta were not absorbed, but they presumably represent a lethal dose absorbed previously. In another study, Wood Ducks (*Aix sponsa*) died after consuming seed treated with carbofuran and the OP diazinon (Stone and Gradoni 1985).

No data have been located regarding simultaneous exposure of experimental birds to several pesticides at maximum application rates. Exposure of experimental animals to an OP and later to a carbamate resulted in a 5- to 15-fold decrease in toxicity of the carbamate (Miyaoka et al. 1984), whereas treatment with a carbamate and then an OP resulted in a 3- to 8-fold increase in toxicity of the OP (Gordon et al. 1978). Although birds are exposed to a mixture of pesticides and other pollutants, we have inadequate information regarding interactions of these compounds, especially when several are applied simultaneously at full strength.

Field Studies of Indirect Effects

In Great Britain, effects of OPs such as dimethoate and various herbicides were apparently related to population declines of the Gray Partridge (*Perdix perdix*), Ring-necked Pheasant (*Phasianus colchicus*), and possibly other upland game birds. Even though adults experienced no significant mortality or any significant physiological effects (Potts 1973, 1986, 1990), the problems seemed to be related to a reduction in arthropod numbers in cropland, causing young birds to spend more time covering more area in search of food. Partridge and pheasants, like most upland game birds, require a high-protein food source during the first few weeks after hatching. The diet of adults consists primarily of vegetation. Mortality of partridge and pheasant chicks was much greater in treated vs. untreated areas. The few chicks found dead seemed to have died from cold and starvation (Potts 1986). The action of the herbicides differed from that of the insecticides such as dimethoate, in that the former reduced the very component of the vegetation (forbs) that contained the greatest mass of arthropod prey, whereas the latter killed the prey directly (Southwood and Cross 1969, Potts 1973, 1986, 1990). Potts (1990) concluded that frequent use of dimethoate could "wipe out" the resident wild game birds in the treated areas. It should be noted that young Sage Grouse killed by dimethoate in Idaho were hatched and reared in sagebrush until they were old enough to move to cropland. By that time, most of their food consisted of vegetation; thus, probable reductions in arthropod prey apparently had little or no effect on Sage Grouse young (Blus et al. 1989).

Another variation in food depletion by pesticides occurs when adult and young birds both feed extensively on the same foods, as in the case of Blue Tits (*Parus caeruleus*). Application of the OP malathion to study areas in Spain almost eradicated the larvae of an important lepidopteran prey species, but another similar prey species was not affected. As a result, no food shortage occurred on the treated areas, and reproductive success was similar on both treated and reference areas (Pascual 1994).

Although several experimental studies have attempted to simulate effects on birds of food deprivation coupled with pesticide exposure (Keith and Mitchell 1993), the effects are mediated through adult birds. Secondary effects on birds that result from reduction of their food supply by pesticides, are exceedingly difficult to determine and quantify through field studies. This is true whether they are coupled with direct (primary) toxic effects to some or all components of the population, or whether direct toxic effects are unimportant, but the food supply is critically diminished for either the young or all components of the population. The excellent design of the British studies, including use of radiotelemetry and measurement of arthropod populations, enabled researchers to document effects of pesticides on young game birds through reductions in available arthropod food. Limited circumstantial evidence for the loss of young pheasants due to reduced prey availability was also reported in the United States (Messick et al. 1974). Upland game birds are decreasing over a widespread area, primarily from loss of suitable habitat; however, pesticides may have a secondary, but important, role in some of these declines (Potts 1990).

Aside from herbicides on cropland, many compounds are used to control various types of plants in forests (primarily clearcuts), brushlands, riparian areas, various aquatic habitats, and grasslands. Santillo et al. (1989) summarized probable effects of herbicides on birds as related to changes in vegetation, including alteration of structure, foliage diversity, and species composition. Bird populations were reduced overall in clearcuts treated with herbicides in Maine (Santillo et al. 1989) and Norway (Slagsvold 1977), but not in Oregon (Morrison and Meslow 1984). Also, avian populations were reduced in clearcuts in Nova Scotia, Canada, only when applications were twice the recommended rate (Freedman et al. 1988).

Two studies on the same area in California indicate the complexity of the herbicide issue. Beaver (1976) found that herbicide treatments had little effect on avian species composition, population size, or relative abundance of birds two years after application. Savidge (1978) found reductions in individuals and species of birds six years after application. The difference was attributed largely to the collapse of dead shrubs that had provided nesting cover during the earlier study. In sagebrush treated with herbicide, Brewer's Sparrows (*Spizella breweri*) showed reductions of 67% and 99% in their populations one and two years, respectively, following herbicide application (Schroeder and Sturges 1975).

Even when overall population declines occurred, the general trend was for populations of some species to decrease while others increased. Herbicides vary widely in their effectiveness in controlling vegetation, depending on a number of factors including the compound itself. Frequently, vegetation is not affected uniformly by the herbicide. Within the treated area, unaffected habitat, sometimes of sizeable blocks, tends to ameliorate effects of the herbicide (Morrison and Meslow 1984). Decreases of some species appear to be related to reductions in invertebrate populations; thus, birds are affected by reductions in cover as well as food supply.

Field Studies of Endocrine Disruption

Although pesticides were found early on to result in certain disruptions in the endocrine system that caused such bizarre effects as feminization of male Ring-necked Pheasants (DeWitt 1956), many years of field work have uncovered variations on this theme, including eggshell thinning and depressed reproductive success.

Since 1950, sex ratios in certain populations of several species of North American gulls and the Caspian Tern (*Sterna caspia*) have changed from essentially balanced to a skewed ratio favoring females (Conover 1983, Conover and Hunt 1984). Homosexual pairings of female Western Gulls (*Larus occidentalis*) accounted for ≥10% of the breeding pairs on Santa Barbara Island, California, from 1972 to 1978 (Hunt et al. 1980); these pairs usually were associated with abnormally large (supernormal) clutch size. Nearly 5% of the clutches of Caspian Terns in Washington and Oregon from 1977 to 1981 contained supernormal clutches compared to 0.7% in a sample collected from 1860–1940 (Conover 1983). It should be pointed out that some homosexual pairs were associated with clutches of normal size, so the actual incidence was probably underestimated in most instances.

Fry and Toone (1981) postulated that the unbalanced sex ratios were related to feminization of male embryos through exposure to DDT and its metabolites; they duplicated feminization experimentally by injecting DDT and its metabolites into eggs of Western and California Gulls (*Larus californicus*), collected from clean areas. Although females in homosexual pairs laid eggs, very few were fertile from promiscuous matings with males. The role of DDT in the unbalanced sex ratios was not accepted universally; for example, Conover and Hunt (1984) postulated that the skewed sex ratio stemmed from high differential male mortality. There was even a suggestion that female–female pairing was a reproductive strategy (Shugart et al. 1988). In the northeastern United States, Herring Gulls (*Larus argentatus*) had a sex ratio slightly skewed toward females, with no firm evidence of female–female pairs and a very low incidence of supernormal clutches (Burger and Gochfeld 1981, Nisbet and Drury 1984). In the Great Lakes, supernormal clutches (4–7 eggs) accounted for 0.7% of Herring Gull nests (Shugart 1980). Female–female pairs of gulls and terns were identified through laparotomies or external measurements (Conover 1983, Shugart et al. 1988).

Evidence for the extent of involvement of DDT in female–female pairs is not clear-cut, because few studies have analyzed gull or tern eggs or tissues for residues of DDT and other endocrine-disrupting compounds. Also, other endocrine-disrupting compounds, such as methoxychlor (Fry and Toone 1981) and polychlorinated biphenyls (Michael Fry, *personal communication*), are capable of inducing these problems. The correlative evidence that sex ratios were skewed toward females, shortly after DDT was widely used, and that the most disproportionate sex ratios and the highest incidence of homosexual pairing in Western Gulls occurred near a "hotspot" of DDT contamination near Los Angeles suggest that DDT did have a major role in this scenario. It remains undetermined whether the high differential male mortality was related to feminization by DDT. Recently, Western Gull populations in southern California have increased, the incidence of female–female pairing is rare, and DDE residues have decreased (Michael Fry, *personal communication*).

For undefined reasons, several species of gulls and the Caspian Tern are essentially the only birds in which female–female pairing has occurred commonly in the field. Homosexual pairing of a wide variety of captive birds occurs when sex ratios are skewed (Huber and Martys 1993). There are concerns that endocrine-disrupting chemicals, including pesticides, are continuing to exert negative effects on many wildlife populations and on human health (Colburn et al. 1993, Hilman 1993). In the future, field studies will play a major role in defining and quantifying these effects.

Discussion and Conclusions

The type and thoroughness of field studies of individual pesticides depend on a number of factors, including chemical structure as well as results of experimental studies of bioaccumulativeness, acute and chronic toxicity, and effects on reproduction and key physiological measurements. Extensive field testing should be reserved for compounds with the potential for greatest harm, with limited testing for those with less apparent risk.

The greatest asset of field studies of pesticide effects on birds is that, if properly conducted, they represent the real world; that is, the findings reflect the influence of a vast array of variables that exist in the environment. On the other hand, the real-world attributes that define the benefits also are drawbacks in designing and conducting field studies, as well as interpreting results. Too many field studies are poorly designed or fail to address appropriate biological endpoints for the specific types and magnitudes of effects that particular compounds exert on wildlife. Therefore, some individuals conclude that it is more productive and less expensive to rely on standard experimental toxicity tests to predict effects on wild populations of birds. However, we provide examples demonstrating that the benefits of well-designed field studies far outweigh their shortcomings. For nearly a century, birds have been used most successfully to detect and monitor effects of pesticides and other contaminants on mortality and population declines (Morrison 1986).

After determining what type of study area is appropriate, one of the prerequisites for selecting an area is to determine the composition of its avifauna. Selecting the species for study depends on a number of factors, the most important being some evidence of adverse effects on that species by the target pesticide(s). Since previous documentation with sensitive avian species is unlikely, especially when registering new pesticides, one must rely on the best ecotoxicological data available regarding species that are likely to be sensitive, based on their response, or that of related species, to the target pesticide or to similar compounds. Actually, the sensitivity concept is not only related to interspecific differences, but also may be related to intraspecific variables such as age, sex, or body condition. Uniform sensitivity sometimes may exist among birds of higher taxa such as genus, family, or even order, but there are insufficient data for a thorough analysis.

In addition to species sensitivity, as determined from experimental or field studies, other important factors include food habits (usually the principle source of exposure), home range, time and type of application, and other factors. Because there are a number of uncontrollable variables, a sound statistical design (Eberhardt and Thomas 1991) should be used in selecting the number and size of treatment and reference study areas. Coordinating avian studies with efficacy tests on insects, other pests, or other nontarget organisms enhances the validity of the studies. Other alternatives include localized field testing of the pesticide to determine effects on the avifauna in general (the most common strategy is to determine what does or does not die), or to forego extensive field work such that adverse effects are determined after the pesticide is in widespread use.

Circumstances leading to unexpected and unique relations of pesticides to birds, initially revealed through field studies, are: (1) no consideration, or inadequate evaluation in experimental laboratory studies, of the critical factors affected by pesticides, such as reproductive success, eggshell thinning, food shortage, or enhanced toxicity through interaction with stressors (e.g., weather); (2) insufficient knowledge of unique habits of certain birds that result in maximum exposure to pesticides applied to local areas or at low concentrations; (3) differences in persistence, either longer or shorter, in experimental vs. field situations as a result of different routes of exposure or other factors; (4) differences in sensitively among species or other groups to various pesticides (10-fold differences in toxicity are common); (5) inadequate knowledge of effects of simultaneous application of several pesticides of different classes; or (6) absence of methodology for measuring certain endpoints, such as population trends, for experimental birds; this obviously requires studies of wild populations over several years.

Field work also reveals the unanticipated real-world findings such as impacts of simultaneous application of several pesticides at full strength. Except for the lacing of carcasses with famphur and the killing of Canada Goose goslings occupying a seed alfalfa field, all examples cited in this paper were related to legal applications. Little evidence exists regarding the extent of illegal applications or misuse of pesticides, and this aspect generally is not considered in the registration process.

Although it is true that one cannot expect experiments or field studies of birds to cover all contingencies, it seems obvious that more attention should be paid to ecological considerations when conducting toxicological research in the field or under controlled conditions. Selection of the best available sensitive species and of comparable treatment and reference study areas are important in ascertaining whether avian populations are at risk from pesticides.

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