III. Environmental Monitoring and Evaluation

Grasshopper control does not take place in a vacuum but in complex rangeland ecosystems. Researchers funded by the Grasshopper Integrated Pest Management Project carefully studied the effects of various control regimes on aquatic organisms, small mammals, birds, and bees. (Photo by R. Miller, submitted through chapter author James R. Fisher and reproduced by permission.)

III.1 Introduction

L. C. McEwen

Grasshopper integrated pest management (GHIPM) is the preferred alternative for grasshopper control listed in the 1987 Environmental Impact Statement for the 17 Western States with rangeland. In conducting the U.S. Department of Agriculture (USDA) cooperative grasshopper control programs, it is necessary to meet the requirements of environmental protection laws, especially the National Environmental Policy Act, the Endangered Species Act, and laws to protect surface and ground water.

Three of the registered methods for the cooperative programs use liquid insecticide formulations. Although the amount of active ingredient applied has been reduced by using ultralow-volume spray techniques, these pesticides can still affect the ecosystem. Grasshopper sprays blanket the rangeland habitat and expose nontarget animal life to the chemicals. Though the spray programs effectively reduce grasshopper densities in the short term, effects on nontarget species and rangeland ecology need to be evaluated. Some aspects deserve continued monitoring after USDA's GHIPM Project ended in 1994.

Use of dry baits for grasshopper control, with less potential for unintended effects on nontarget life, was investigated in the field. Grasshopper baits carrying chemical or biological control materials have great promise for use in environmentally sensitive areas. Also, new candidate grasshopper control methods and materials, such as diflubenzuron and *Beauveria bassiana,* were examined for effects on American kestrels (sparrowhawks) in field studies of nestlings and fledglings. These materials appear to have little, if any, direct toxicity to birds.

Several field and laboratory studies of GHIPM materials or methods have been conducted since the inception of the GHIPM Project in 1987. Birds have received the most attention because they are usually more susceptible than mammals to direct toxicity and to indirect ecological changes, such as loss of insect food. Studies have varied from determining total avian population response following large-scale grasshopper control programs (on areas greater than 10,000 acres) to physiological and behavioral measurements in individual birds sublethally exposed to GHIPM materials.

Two species of endangered fish have been studied intensively for toxicity of malathion and carbaryl. Effects on nontarget invertebrates (both aquatic and terrestrial) were also investigated. Other GHIPM Project-sponsored environmental impact studies included (1) avian and mammalian brain and blood cholinesterase measurements, (2) use of American kestrels and killdeer as bioindicators of possible effects on closely related endangered species, (3) effectiveness of bird predation for regulating grasshopper population densities, (4) postspray pesticide residue concentrations in environmental samples and biota (fauna and flora), (5) results of aquatic field monitoring of spray treatments, (6) small mammal live-trapping recapture tests, and (7) field experiments to investigate the indirect effects (loss of food base) on productivity of nesting birds associated with application of malathion and Sevin® 4-Oil liquid sprays and carbaryl bait. Preliminary results of golden eagle postfledging survival after aerial spray of Sevin 4-Oil to nest areas are also reported in this Environmental Monitoring and Evaluation section.

The important question of potential effects on endangered plant species and their insect pollinators is addressed in a summary of several studies. Authors also discuss untreated buffer-zone requirements to protect endangered plants, aquatic habitats, nests of endangered birds such as peregrine falcons, and other environmentally sensitive sites.

Knowledge of GHIPM relationships to nontarget life and rangeland ecology is critical for successful grasshopper population management. The days are long past when estimating the grasshopper kill was the only concern while other effects of a spray program were ignored. For many years, aldrin, dieldrin, and other organochlorine compounds were extremely efficient at killing grasshoppers, but USDA stopped using those pesticides in the mid 1960's because of their effects on nontarget life. Organochlorine pesticides harmed wild mammals, migratory birds, endangered raptors, reptiles, aquatic life, and western rangeland ecosystems (McEwen 1982).

Dieldrin, for example, is a stable compound that circulated through food chains and ecosystems for years and was highly toxic to all fish and wildlife. The Environmental Protection Agency criterion for chronic dieldrin contamination in fresh water is only 0.0019 parts per billion (Nimmo and McEwen 1994), but the bioconcentration factor in aquatic life can be 49,000 times the level of contamination in the water (Moriarity 1988). Animals exposed to sublethal organochlorine contamination may be unable to reproduce—particularly many fish species, fish-eating birds, and endangered raptors—and may also be more vulnerable to disease, pathogens, predators, and other stresses.

The insecticides currently registered for GHIPM programs are not only less toxic to terrestrial nontarget wildlife (McEwen 1982, Stromborg et al. 1984, Smith 1987) but also much less persistent in the environment than organochlorine chemicals. Today's grasshopper insecticides soon degrade into biologically inactive compounds that do not circulate through food chains (U.S. Department of Agriculture, Animal and Plant Health Inspection Service, 1987). The primary questions to be answered concerning the current control materials are (1) significance of sublethal toxic effects on birds, mammals, and fish, particularly cholinesterase inhibition; (2) degree of hazard to endangered fish, wildlife, and plants, and other species of concern; (3) indirect effects due to reduction of insect or invertebrate food supply; (4) effects on nontarget insects, including pollinators of endangered plants; and (5) evaluation of wildlife population effects related to wide area GHIPM treatments. The answers to these questions are more difficult to determine than the relatively simple wildlife carcass counts and pesticide residue analyses that were used to investigate the old organochlorine pesticides.

The current, more comprehensive, investigations of sublethal and indirect effects reflect the need to determine the complex ecological impacts of GHIPM on nontarget life. The findings support GHIPM strategy, including recognition that healthy, vigorous, rangeland ecosystems are the most permanent solutions to range grasshopper problems in the long term.

References Cited

McEwen, L. C. 1982. Review of grasshopper pesticides vs. rangeland wildlife and habitat. In: Peek, J. M.; Dalke, P. D., eds. Proceedings of the wildlife–livestock relationships symposium; 20–24 April 1981; Coeur d'Alene, ID. Moscow, ID: University of Idaho: 362–382.

Moriarity, F. 1988. Ecotoxicology (2d ed.). London and New York: Academic Press. 289 p.

Nimmo, D. W.; McEwen, L. C. 1994. Pesticides. In: Calow, P., ed. Handbook of ecotoxicology, vol. 2. Oxford, UK: Blackwell Scientific Publishers: 155–203.

Smith, G. J. 1987. Pesticide use and toxicology in relation to wildlife: organophosphorus and carbamate compounds. Resour. Publ. 170. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service. 171 p.

Stromborg, K. L.; McEwen, L. C.; Lamont, T. 1984. Organophosphate residues in grasshoppers from sprayed rangelands. Chemistry in Ecology 2: 39–45.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 1987. Final environmental impact statement: Rangeland Grasshopper Cooperative Management Program. USDA-APHIS FEIS 87-1. Washington, DC: U.S. Department of Agriculture, Animal and Plant Health Inspection Service: 2–9, 2–10, 2–11.

III.2 Direct and Indirect Effects of Grasshopper Integrated Pest Management Chemicals and Biologicals on Nontarget Animal Life

L. C. McEwen, C. M. Althouse, and B. E. Petersen

NOTE: Acephate is no longer approved by EPA for rangeland grasshopper control.

Initially there were 16 objectives (11 terrestrial and 5 aquatic) for the environmental monitoring studies of the Grasshopper Integrated Pest Management (GHIPM) Project. Most of the terrestrial objectives were concerned with determining effects of the grasshopper control methods and materials on birds. Studies varied from total bird population response after spray operations or bait treatments to toxicology tests with individual birds.

Small-mammal population effects and toxicology were investigated with one chemical (acephate). Some limited small-mammal observations also were obtained in areas sprayed with malathion and Sevin® 4-oil. Aquatic objectives were to investigate toxic effects of malathion and carbaryl on endangered fish in tank tests and to determine effects of grasshopper spray programs on fish and aquatic invertebrates in the field.

Other objectives included (1) evaluation of hazards to endangered species through study of related surrogate species, (2) determination of the significance of bird predation as a biological control of grasshoppers in an IPM program, and (3) wildlife tests with the candidate materials *Beauveria bassiana* (a fungal organism) and diflubenzuron (an insect growth inhibitor). More than 20 papers have been published in peer-reviewed journals on the GHIPM Project's environmental monitoring work, and other papers are in press.

Direct Effects

Direct effects on nontarget fish and wildlife of GHIPM materials may be lethal or sublethal. Unlike the organochlorine pesticides, such as dieldrin, chlordane, heptachlor, and toxaphene, formerly used for range grasshopper control (and still in use in some parts of the world) the current GHIPM chemicals do not kill wildlife by direct toxicity (McEwen 1982). There may be some rare exceptions to this statement, such as individual small nestlings of passerine (bird) species that are unusually sensitive to carbaryl or malathion being directly sprayed on an open nest. On the whole, however, GHIPM Project-funded investigators have seen only a very few such possible cases in a large number of nest observations. And none of these bird deaths could be positively attributed to chemical control materials.

At the malathion ultralow-volume (ULV) application rate of 8 fl oz/acre (0.58 kg/ha) and the Sevin 4-Oil formulation rate of 20 fl oz/acre (1.44 kg/ha) (carbaryl active ingredient [AI] rate of 0.56 kg/ha), there is very little possibility of toxicity-caused mortality of upland birds, mammals, or reptiles, and none has been observed.

However, these pesticides are more toxic to aquatic life: direct overspray of small ponds kills many aquatic invertebrates and may kill sensitive fish species. The risk is lower in flowing streams because the chemical is transported downstream and diluted more rapidly. Consequently, nonspray buffer zones around aquatic habitat must be observed (see chapter III.8). Lower-level exposure from pesticide drift or runoff (in contrast to direct overspray) does not kill fish but can be lethal to certain aquatic invertebrates (Beyers et al. 1995; also see chapter III.6).

One of our main environmental monitoring objectives was to determine effects of grasshopper control treatments on rangeland bird populations. We investigated 13 different grasshopper control treatments with GHIPM materials (malathion, Sevin 4-Oil, carbaryl bait, or *Nosema locustae).* We studied effects on total bird populations by concurrently conducting extensive line transect counts (Emlen 1977) before and after insecticide application in both treatment and control (untreated) plots. Total birds (total individuals of all species) did not change ($P > 0.05$) in the posttreatment periods (George et al. 1995). Populations of one highly insectivorous species, the western meadowlark *(Sturnella neglecta),* did consistently decrease at 10 and 21 days posttreatment. We presumed that was due to reduced food availability because there was no evidence of toxic signs in the remaining meadowlarks, and no dead ones were found. Comparative avian population response to many different pesticides used or tested for grasshopper control can be found in a report by McEwen (1982).

Sublethal Effects

Sublethal exposure to GHIPM pesticides is highly probable for wildlife inhabiting sprayed rangeland. The routes of exposure include dermal from direct hit or by moving through sprayed vegetation, ingestion in food or drinking water, and inhalation. The effects of sublethal exposure can vary from biological insignificance to convulsions and near death followed by recovery. Severe toxic signs have not been observed in terrestrial wildlife following GHIPM treatments. The potential for sublethal toxic effects can be minimized by use of bait formulations. Dry bait formulations use less actual chemical per acre or hectare and limit the route of exposure primarily to ingestion of affected insects. In comparison, liquid sprays result in multiple exposure routes (dermal, inhalation, and ingestion of coated vegetation as well as insects). Consumption of bait (bran particles) by wildlife is negligible because of the small size of bran particles and the low treatment rates used for GHIPM (2 to 5 lb/acre or 2.2 to 5.6 kg/ha of bait containing 2 percent carbaryl).

Use of bait treatments provides an environmentally safe means of obtaining some reduction of grasshopper densities in environmentally sensitive areas (such as habitat for endangered plants or animals). Vesper sparrow survival, growth, and fledging rates were not affected by carbaryl bait treatments around the nest areas (Adams et al. 1994). Total bird numbers were not reduced in a large area treated for grasshopper control with carbaryl bait (George et al. 1992a). Bait treatments at GHIPM rates reduce the potential for aquatic contamination (less drift and less chemical). Baits also appear safe for bees and pollinators of endangered plants (see chapters III.4 and III.5).

Cholinesterase Inhibition

All three of the GHIPM chemicals—carbaryl, malathion, and acephate—are cholinesterase (ChE) inhibitors. In vertebrates, acetylcholinesterase and butyrylcholinesterase are essential for normal function of the nervous system. Severe inhibition (>60 percent) often leads to death of the animal (fig. III.2–1). Moderately severe inhibition (40–60 percent) affects coordination, behavior, and foraging ability and can lead to death from other stresses of survival in the wild, such as weather or predators. Effects of lower levels of brain ChE inhibition (<40 percent) are still an open question regarding biological significance (Grue et al. 1991). In our samples of birds and mammals from areas treated with carbaryl, malathion, or acephate, we have not found any animals with >40 percent brain ChE inhibition, and only a few individuals inhibited >20 percent (Fair et al. 1995, George et al. 1995, and Petersen et al., in prep).

Figure III.2–1—Several highly toxic pesticides were field-tested to determine efficacy for grasshopper control and effects on nontarget life. Those chemicals found to be too toxic and hazardous to wildlife were not registered for use on rangeland. Most of the chemicals not registered were severe cholinesterase inhibitors and caused paralysis and death of beneficial birds, such as these Wilson's phalaropes. (Photo by G. Powell of the U.S. Fish and Wildlife Service; reproduced by permission.)

In a study of fish exposed to light drift of carbaryl (Sevin 4-Oil), Beyers et al. (1995) detected no effects on brain ChE. Blood plasma ChE also can be used as an indicator of pesticide exposure: effects of malathion on kestrels and carbaryl (Sevin 4-Oil) effects on golden eagles were reported by Taira (1994).

These results suggest that ChE inhibition is not a problem for upland wildlife when GHIPM chemicals are applied but do not mean that attention to accuracy and rigor of applications can be relaxed. Beyers et al. (1994) found that in water, concentrations of carbaryl as low as 1.3 mg/L (p/m) and of malathion as low as 9.1 mg/L were lethal to fish. Young kestrels died from malathion exposures of only 30 mg per kg of body weight (McEwen et al. 1993 unpubl.), much lower than lethal dosages for other species of birds (>100 to >400 mg/kg, Smith 1987).

A recent study by Nicolaus and Lee (1999) suggested a formerly unrecognized effect of organophosphate exposure. Birds that fed on affected insects developed a strong aversion to those insect species and would no longer capture them for food, even after the insects were free of contamination. Thus surviving birds were indirectly denied major food sources.

Indirect Effects

The most frequently asked question about effects on wildlife of grasshopper control is, "What about the effects on birds of the loss of the insect food base?" Much of our environmental monitoring effort was directed at this problem.

A 3-year investigation of indirect effects of malathion on nesting birds was conducted in Idaho. After a year of pretreatment study, two areas of rangeland were sprayed with the standard 8 fl oz/acre (0.58 kg/ha) ULV formulation of malathion. Intensive studies were conducted to measure effects on the insect and invertebrate populations and on survival and growth of Brewer's sparrow *(Spizella breweri)* and sage thrasher *(Oreoscoptes montanus)* nestlings (Howe 1993, Howe et al. 1996 and 2000).

Although the total invertebrate availability was significantly reduced by the spray applications, nesting birds switched their diets to the remaining insects and reproduced as successfully as birds on untreated comparison plots (Howe et al. 1996 and 2000). Adults had to forage longer on sprayed plots, and nestlings showed a higher propensity for parasitic blowfly *(Protocalliphora braueri)* infestation (Howe 1991, 1992), both of which might affect survival in some situations. Those effects were not significant in this study. Prespray grasshopper densities were low (1–4 per square yard or square meter) on all plots and were significantly reduced in the postspray period. This probably made the food availability test more rigorous than an operational grasshopper control program, where prespray densities are much higher and even postspray grasshopper densities usually exceed 1 or 2 per square yard or square meter.

Effects of Sevin 4-Oil sprays on killdeer populations were investigated in North Dakota. Two large treated areas were studied. One was sprayed with the standard rate of 20 oz/acre of formulation (16 oz Sevin 4-Oil + 4 oz diesel oil), and the other area received a lower rate of 16 oz/acre (12 oz Sevin 4-Oil + 4 oz diesel oil). These rates translated to 0.56 and 0.45 kg/ha of carbaryl AI respectively. No toxic signs and no mortality were observed in the killdeer.

Effects on foraging and diet of the killdeer were examined by both direct observation and analysis of stomach contents (Fair et al. 1995a). The insect capture rate by foraging killdeer increased during the period when affected insects were easily available 2 days after treatment (Fair et al. 1995b). No other differences in food habits were detected.

A test of carbaryl bait effects on vesper sparrow *(Pooecetes gramineus)* nestling growth and survival was conducted in North Dakota. This study simulated the "hot spot" method of treating small grasshopper infestations with carbaryl bait. There was no difference in any of the productivity parameters between nests on treated and untreated sites (Adams et al. 1994). Adult sparrows on treated sites had to forage farther from the nests to obtain food but did so successfully. Grasshoppers comprised 68 percent of all food deliveries to nestlings even though grasshopper densities were <1 per square meter. The ability of birds to capture a preferred food, even when grasshopper densities are extremely low, supports the value of predation by birds as a preventive force against grasshopper increase in an IPM approach to grasshopper management (see chapter I.10, "Birds and Wildlife as Grasshopper Predators").

Biennial grasshopper infestations in southeastern Alaska provided an opportunity to examine bird population response to the extreme differences in grasshopper abundance and availability that occur naturally. Densities alternate between >25 per square yard in high years and <1 per square yard in low years. This phenomenon apparently occurs because of a synchronized 2-year life cycle of the *Melanoplus sanguinipes* grasshopper species in the population. Birds were counted on permanently marked transects in 2 high and 2 low years, and nesting success of Savannah sparrows *(Passerculus sandwichensis),* the most abundant bird species, was measured. Total bird populations did not differ among years ($P > 0.05$).

Nesting success showed a trend of lower clutch size and nestling growth rates in the low grasshopper years (1991 and 1993) but not significantly ($P > 0.05$) (Miller et al. 1994). Grasshoppers constituted >45 percent of the birds' diet numerically and an even greater proportion of biomass in the high grasshopper years (1990 and 1992)

(McEwen et al. 1993 unpubl., Miller and McEwen 1995). The birds also managed to search out and capture grasshoppers in the low years, indicating their preference for this important food source. However, the breeding birds were able to switch their main food items to other insects (beetles, Hemiptera, larvae of Lepidoptera and others) in the low grasshopper years.

Rangeland wildlife has adapted to variable food availability and environmental conditions over the long term. Evidence of this was observed in North Dakota studies. An extreme drought in 1988 resulted in avian nesting failures and population declines. The effects on bird populations did not carry over to the succeeding years, when precipitation was in the normal range (George et al. 1992b; see also chapter III.7).

Small Mammal Studies

Small mammals generally are not affected as much as birds in the same area where a pesticide application is made, probably because small mammals generally are not exposed to as much toxicant as birds are. Most small mammals are nocturnal and are often in underground burrows during and immediately after a treatment; thus there is more time for the chemical to dissipate before small mammals are exposed (fig. III.2–2). Deer mice *(Peromyscus maniculatus)* collected on a malathion-sprayed area had lower residues than birds from the same sites (McEwen et al. 1989 unpubl.). Many small-mammal species also are inherently more resistant to specific toxicants than birds (Nimmo and McEwen 1994).

Figure III.2–2—Kangaroo rat being released after capture in a live-trap for study on a rangeland-grasshopper control area. Small mammals were generally less vulnerable to pesticide effects than birds inhabiting sprayed areas. (Photo by L. C. McEwen of Colorado State University; reproduced by permission.)

Effects of acephate and methamidophos (an acephate metabolite) on small mammals were studied on short grass range in Colorado. Results have not been completely analyzed, but preliminary data indicate a decrease in populations of certain species due to a combination of greater sensitivity to chemical toxicity and reduced competitive ability with other species. Deer mice were twice as sensitive to methamidophos (the lethal dose to 50 percent, or LD_{50}, was 9 mg/kg) than the other two most common species, grasshopper mice *(Onychomys leucogaster)* and 13-lined ground squirrels *(Spermophilus tridecemlineatus)*. The LD_{50} for both the latter was 21 mg/kg (Stevens 1989). Field live-trapping studies indicated postspray decreases of deer mice but not of the grasshopper mice and ground squirrels. Data analysis and manuscripts are still in progress on these studies (Althouse et al. unpubl., McEwen et al., in prep.).

Limited live trapping studies on malathion-sprayed areas in North Dakota showed no posttreatment decreases in abundant populations, primarily deer mice, and studies of carbaryl-sprayed areas at other locations had a similar outcome (McEwen et al. unpubl. 1988). An investigation of malathion ULV (8 fl oz/acre or 0.58 kg/ha) applied in Nebraska found no effects on small-mammal populations (Erwin and Sharpe 1973).

Golden Eagle Study

Golden eagles *(Aquila chrysaetos)* are a protected species and also are designated as a “species of concern” by wildlife conservation and land management agencies. This species also has special significance for Native Americans. Golden eagles nest in remote rangeland areas and often are found on areas slated for grasshopper control. Because of these concerns and problems, a study was initiated on the Western North Dakota IPM Demonstration Area where nesting territories and spray blocks often overlap.

Active nests of golden eagles were located and randomly selected for Sevin 4-Oil treatments or left unsprayed in 1993 and 1994. Overall, 12 nest areas were sprayed with Sevin 4-Oil at 20 fl oz/acre (1.4 kg/ha) or 8 oz/acre AI

(0.56 kg/ha AI) carbaryl. Approximately 10 ha were treated around each nest. For comparison, the investigators left eight nest areas untreated. At these control nests, the spray plane flew the same pattern and length of time but did not release any spray. Some nests contained two nestlings and some, a single nestling. The total number of treated nestlings was 17, and untreated totaled 11. Treatments were made when the eaglets were 4–7 weeks of age.

When the nestlings neared fledging age (10–11 weeks) they were captured to (1) take biological measurements, (2) take a 4- to 5-mL blood sample, and (3) attach a radio transmitter for postfledging location and observations (telemetry) (O'Toole et al. 1999). Field work and data analysis are incomplete, but preliminary results can be reported.

In 1993, two untreated and three treated fledglings died from various causes unrelated to the treatments. In 1994, a better prey year, all 6 untreated and 10 treated fledglings survived. Postfledging telemetry studies indicated two behavior differences in the eagles from sprayed nest areas: "sprayed" eagles tended to perch longer and to preen more in afternoon observation periods. These results will be reported by O'Toole et al. (in prep.). All fledglings dispersed from their hatch areas by November each year (except for one, which left by December 3, 1994), and radio signals could no longer be detected in ground searches. Aerial telemetry searches were conducted in 1995 to obtain more information on movements and long-term survival rates.

Blood plasma ChE and other blood components were measured. Golden eagles were found to have a higher proportion of butyrylcholinesterase (75 percent) than acetylcholinesterase (25 percent) in plasma (Taira 1994). Blood samples from the treated nestlings had higher total ChE activity than untreated, but not significantly ($P = 0.11$). This was somewhat predictable in that blood samples were not taken until 3 to 5 weeks after exposure, and an overcompensation or "rebound effect" has been found in other species after light exposure to carbamates.

In summary, it appears that Sevin 4-Oil sprayed at the GHIPM rate offers little risk to nesting golden eagles. With global positioning system technology, spray planes could shut off and leave a small unsprayed area of a few acres or hectares around active nests, to leave the eagles completely unaffected. Similar studies of effects of malathion sprays (8 fl oz/acre or 0.58 kg/ha) for rangeland grasshopper control need to be conducted with young golden eagles.

References Cited

Adams, J. S.; Knight, R. L.; McEwen, L. C.; George, T. L. 1994. Survival and growth of nestling vesper sparrows exposed to experimental food reductions. The Condor 96: 739–748.

Beyers, D. W.; Sikoski, P. J. 1994. Acetylcholinesterase inhibition in federally endangered Colorado squawfish exposed to carbaryl and malathion. Environmental Toxicology and Chemistry 13: 935–939.

Beyers, D. W.; Keefe, T. J.; Carlson, C. A. 1994. Toxicity of carbaryl and malathion to two federally endangered fishes, as estimated by regression and ANOVA. Environmental Toxicology and Chemistry 13: 101–107.

Beyers, D. W.; Farmer, M. S.; Sikoski, P. J. 1995. Effects of rangeland aerial application of Sevin-4-oil on fish and aquatic invertebrate drift in the Little Missouri River, North Dakota. Archives of Environmental Contamination and Toxicology 28: 27–34.

Emlen, J. L. 1977. Estimating breeding season bird densities from transect counts. Auk. 94: 455–468.

Erwin, W.,J.; Sharpe, R. S. 1978. Effect of wide area ultra-low volume application of malathion on small mammal populations. Transactions of the Nebraska Academy of Science 5: 25–28.

Fair, J. M.; Kennedy, P. L.; McEwen, L. C. 1995a. Effects of carbaryl grasshopper control on nesting killdeer in North Dakota. Environmental Toxicology and Chemistry 14: 881–890.

Fair, J. M.; Kennedy, P. L.; McEwen, L. C. 1995b. Diet of nesting killdeer in North Dakota. Wilson Bulletin 107: 174–178.

George, T. L.; McEwen, L. C.; Fowler, A. C. 1992a. Effects of a carbaryl bait treatment on nontarget wildlife. Environmental Entomology 21: 1239–1247.

George, T. L.; Fowler, A. C.; Knight, R. L.; McEwen, L. C. 1992b. Impacts of a severe drought on grassland birds in western North Dakota. Ecological Applications 2(3): 275–284.

George, T. L.; McEwen, L. C.; Petersen B. E. 1995. Effects of grasshopper control programs on bird populations in western rangelands. Journal of Range Management 48: 336–342.

Grue, C. E.; Hart, A.D.M.; Mineau, P. 1991. Biological consequences of depressed brain cholinesterase activity in wildlife. In: Mineau, P., ed. Cholinesterase-inhibiting insecticides: their impact on wildlife and the environment. (Vol. 2 of the Chemicals in Agriculture series.) Amsterdam, London, New York: Elsevier: 152–209.

Howe, F. P. 1991. Two new host species for the parasitic blow fly *Protocalliphora braueri.* Wilson Bulletin 103: 520–521.

Howe, F. P. 1992. Effects of *Protocalliphora braueri* (Diptera: Calliphoridae) parasitism and inclement weather on nestling sage thrashers. Journal of Wildlife Diseases 28(1): 141–143.

Howe, F. P. 1993. Effects of a grasshopper insecticide application on diet, food delivery rates, growth, and survival of shrubsteppe passerines. Ph.D. dissertation. Fort Collins, CO: Colorado State University. 108 p.

Howe, F. P.; Knight, R. L.; McEwen, L. C.; George, T. L. 1996. Direct and indirect effects of insecticide applications on growth and survival of nestling passerines. Ecological Applications 6: 1314–1324.

Howe, F. P.; Knight, R. L.; McEwen, L. C.; George, T. L. 2000. Diet switching and food delivery by shrubsteppe passerines in response to an experimental reduction in food. Western North American Naturalist 60: 139–154.

McEwen, L. C. 1982. Review of grasshopper pesticides vs. rangeland wildlife and habitat. In: Peek, J. M.; Dalke, P. D., eds. Proceedings of the wildlife–livestock relationships symposium; 20–24 April 1981; Coeur d'Alene, ID. Moscow, ID: University of Idaho: 362–382.

Miller, C. K.; McEwen, L. C. 1995. Diet of nesting savannah sparrows in interior Alaska. Journal of Field Ornithology 66: 152–158.

Miller, C. K.; Knight, R. L.; McEwen, L. C. 1994. Responses of nesting savannah sparrows to fluctuations in grasshopper densities in interior Alaska. The Auk 111: 960–967.

Nicolaus, L. K.; Lee, H. 1999. Low acute exposure to organophosphate produces long-term changes in bird feeding behavior. Ecological Applications 9: 1039–1049.

Nimmo, D. W.; McEwen, L. C. 1994. Handbook of ecotoxicology-pesticides. In: Calow, P., ed. Handbook of ecotoxicology, vol. 2. Oxford, UK: Blackwell Scientific Publications. Chapter 8.

O'Toole, L. T.; Kennedy, P. L.; Knight, R. L.; McEwen, L. C. 1999. Postfledging behavior of golden eagles. Wilson Bulletin 111: 472–477.

Smith, G. J. 1987. Pesticide use and toxicology in relation to wildlife: organophosphorus and carbamate compounds. Resour. Publ. 170. U.S. Department of the Interior, Fish and Wildlife Service. 171 p.

Stevens, P. D. 1989. Acute toxicity and inhibition of cholinesterase activity in small mammals following exposure to methamidophos. M.S. thesis. Ft. Collins, CO: Colorado State University. 51 p.

Taira, T. 1994. Blood analysis of American kestrel and golden eagle nestlings exposed to malathion or carbaryl. M.S. thesis. Fort Collins, CO: Colorado State University. 80 p.

References Cited–Unpublished

Althouse, C. M.; McEwen, L. C.; Hoffman, L. A. Influence of prior northern grasshopper mouse *(Onychomys leucogaster)* and 13-lined ground squirrel *(Spermophilus tridecemlineatus)* live trap captures on succeeding intra and interspecific captures. Submitted to Journal of Mammalogy.

McEwen, L. C.; George, T. L.; Petersen, B. E.; Beyers, D. W.; Fowler, A. C.; Howe, F. P. 1989. Environmental monitoring. In: Cooperative Grasshopper Integrated Pest Management Project, 1988 annual report. Boise, ID: U.S. Department of Agriculture, Animal and Plant Health Inspection Service: 82–102.

McEwen, L. C.; Petersen, B. E.; Beyers, D. W.; Althouse, C. M.; Fair, J. M.; Schleve, G. R.; Bednarski, L. T.; Taira, T. 1993. Environmental monitoring. In: Cooperative Grasshopper Integrated Pest Management Project, 1993 annual report. Boise, ID: U.S. Department of Agriculture, Animal and Plant Health Inspection Service: 127–142.

McEwen, L. C.; Althouse, C. M.; Logan, J. A.; Stevens, P. D. (In preparation). Small mammal long-term population response to 2 rates of methamidophos spray on mixed shortgrass/*Atriplex* rangeland. Journal of Range Management or Environmental Toxicology and Chemistry.

O'Toole, L. T.; McEwen, L. C.; Knight, R. L. [In preparation.] Behavioral responses of post-fledging golden eagles *(Aquila chrysaetos)* to Sevin 4-Oil applications in the Little Missouri National Grassland, North Dakota. Environmental Toxicology and Chemistry.

Petersen, B. E.; DeWeese, L. R.; McEwen, L. C.; Palmer, D.; Ryder, R. A. Nesting success of lark buntings on rangeland treated with acephate. Submitted to Journal of Range Management.

III.3 Impact of Control Programs on Nontarget Arthropods

Mark A. Quinn

Introduction

Grasshopper control programs potentially can have a large impact on the rangeland ecosystem. Of particular concern are the effects of large-scale control programs on natural enemies of grasshoppers, pollinators of seed crops and endangered plant species, endangered species of vertebrates, and general biodiversity of grasslands. Here, I will be addressing two main questions: (1) What are the immediate and more long-term effects of grasshopper control treatments on nontarget species? and (2) Does the disruption in communities of nontarget arthropods affect the population dynamics of grasshoppers and the potential for outbreaks?

Effect of Grasshopper Control Treatments on Nontarget Arthropods

There is very little information on the effects of grasshopper control treatments on beneficial and other nontarget arthropods (animals with exoskeletons, such as insects, spiders, and crayfish). Insecticidal sprays can cause high mortality of grasshoppers, so it should be assumed that sprays can cause large reductions in other arthropod populations as well. The potential for a significant impact on nontarget arthropods is large because they are often very active when grasshopper control treatments are typically applied. For example, Quinn et al. (1993) showed a relationship between the presence of nymphal grasshoppers, the stage usually treated in control programs, and the activities of some groups of nontarget arthropods, such as ants, ground beetles, wolf spiders, sphecid wasps, and robber flies.

As part of the Grasshopper Integrated Pest Management (GHIPM) Project work in South Dakota, Quinn et al. (1990, 1991, 1993) studied the effects of large-scale aerial applications of bran bait containing 5 percent active ingredient (AI) carbaryl at 1.5 lb/acre (1.68 kg/ha), and ultralow-volume (ULV) malathion (91 percent AI) at 0.58 lb AI/acre (0.65 kg/ha) on nontarget arthropods of mixed-grass rangeland. Table III.3–1 lists the groups of nontargets that my colleagues and I collected with malaise (aerial) and pitfall (ground) traps before treatments were applied. Of all the groups of nontargets collected in malaise traps, only two are considered predators of grasshoppers—sphecid wasps (15 percent) and robber flies (3 percent). Both of these groups feed on a variety of insects and not just grasshoppers. The most abundant groups collected in malaise traps were ichneumonid wasps (32 percent) and moths (27 percent). Most of the ichneumonid wasps collected were Lepidoptera parasites.

Most of the groups of nontarget arthropods collected in the pitfall traps were grasshopper predators. The two most abundant groups were blister beetles (36 percent) and ants (31 percent). Blister beetle larvae may be significant predators of grasshopper egg pods (Parker and Wakeland 1957, Rees 1973). Ants feed on molting grasshoppers. Other abundant groups of nontarget arthropods were darkling beetles (11 percent), wolf spiders (8 percent), and ground beetles (7 percent).

Some groups of nontarget arthropods were affected by both the insecticidal bait and spray treatments (table III.3–2). Activities of darkling beetles, ground beetles, and field crickets were reduced by 49 percent to 89 percent after 1 week in plots treated with either the insecticidal bait or spray. The dominant species of darkling beetles and ground beetles were similarly reduced by the two treatments (Quinn et al. 1990, 1991). Populations of these groups did not change in the control plots over the same time period. These groups were most likely affected by the insecticidal bait because they either consumed the bait directly or because they fed on infected grasshoppers. Other groups were affected by the insecticidal spray, but not the bait. For example, activities of blister beetles and ichneumonid wasps were reduced by 59 percent and 56 percent, respectively, in the malathion spray plots but did not change in the bran bait or untreated (control) plots. Activities of two species of ground beetles, *Cratacanthus dubius* and *Discoderus parallelus*, were reduced by 81 percent and 66 percent, respectively, in the insecticidal bait plots but did not seem to be affected by the insecticidal spray.

Pfadt et al. (1985) conducted a study to determine the effects of ULV malathion at 8 fluid oz/acre (0.58 lb AI/acre) on nontarget organisms of shortgrass rangeland in Wyoming. Pfadt's team concluded that (1) aerial applications of insecticidal sprays are not likely to have a large impact on nontargets because most species are protected (in nests, soil, and plants), and (2) the only arthropods likely to be affected are those that inhabit

Table III.3–1—Relative abundance (percent) of nontarget arthropods collected with malaise and pitfall traps, July 2–8, 1986, at mixed-grass rangeland plots, Butte County, SD (adapted from Quinn et al. 1993)

Nontarget group		Feeding habits	Relative abundance
			Percent
Malaise traps			
Ichneumonidae	Ichneumonid wasps	Mostly moth parasites	31.6
Lepidoptera	Moths	Plant feeders (as larvae)	26.6
Sphecidae	Sphecid wasps	General predators*	14.7
Odonata	Damsel flies	General predators	9.4
Mutillidae/ Tiphiidae	Velvet ants/ tephiid wasps	Wasp, bee, and beetle parasites	9.3
Pompilidae	Spider wasps	Spider predators	5.8
Asilidae	Robber flies	General predators*	3.0
Chrysididae	Cuckoo wasps	Wasp and bee parasites	1.8
Halictidae	Halictid bees	Pollen feeders/bee parasites	1.4
Others			1.8
Pitfall traps			
Meloidae	Blister beetles	Pollen feeders/grasshopper egg predators*	35.9
Formicidae	Ants	Seed and plant feeders/general predators*	31.0
Tenebrionidae	Darkling beetles	General scavengers/detritus feeders	10.9
Lycosidae	Wolf spiders	General predators*	7.8
Carabidae	Ground beetles	General predators/plant feeders*	6.9
Gryllidae	Field crickets	General predators/plant feeders*	2.6
Buprestidae	Metallic wood-boring beetles	Plant feeders	1.6
Other spiders		General predators*	1.1
Others			2.2

*Feed on grasshoppers

Table III.3–2—Effect of carbaryl bran bait and malathion ULV spray on change in activities of nontarget arthropods between the pretreatment and 1 week posttreatment sampling intervals, Butte County, SD

Nontarget group	Trap	Treatment	% change ($\bar{x} \pm$ SEM[1])	n
Blister beetles	Pitfall	Bran bait	–10.1 ± 13.6	10
		Malathion	–58.5 ± 6.4	10
		Control	–35.1 ± 15.9	9
Ants	Pitfall	Bran bait	32.6 ± 43.6	7
		Malathion	–39.6 ± 3.0	9
		Control	509.3 ± 447.6	5
Darkling beetles	Pitfall	Bran bait	–89.3 ± 4.2	10
		Malathion	–80.9 ± 9.5	10
		Control	210.2 ± 132.4	8
Wolf spiders	Pitfall	Bran bait	–80.5 ± 4.9	10
		Malathion	–76.1 ± 4.1	10
		Control	–61.6 ± 13.2	9
Ground beetles[2]	Pitfall	Bran bait	–88.0 ± 4.6	10
		Malathion	–53.0 ± 8.4	9
		Control	41.8 ± 37.8	9
Field crickets	Pitfall	Bran bait	–82.5 ± 0.1	9
		Malathion	–49.3 ± 14.6	9
		Control	24.4 ± 64.2	6
Ichneumonid wasps	Malaise	Bran bait	143.9 ± 68.7	10
		Malathion	–56.1 ± 6.9	10
		Control	71.1 ± 35.6	8
Sphecid wasps	Malaise	Bran bait	0.1 ± 18.1	10
		Malathion	–17.5 ± 13.7	10
		Control	32.8 ± 61.9	8
Spider wasps	Malaise	Bran bait	–1.8 ± 24.4	10
		Malathion	–9.9 ± 39.7	10
		Control	50.0 ± 57.5	8
Robber flies	Malaise	Bran bait	39.8 ± 27.7	10
		Malathion	–29.5 ± 30.2	9
		Control	–44.9 ± 13.3	7

[1]Standard error of the mean.

[2]Does not include *Amara impuncticollis*, which was not present in traps before treatments but was present after treatments.

foliage during the day. For example, this study showed that the ant *Formica obtusopilosa*, which is commonly found foraging on flowers, was affected by the insecticides. However, colonies of all ant species were not affected. Pfadt's results also indicated that immature Ephemeroptera (mayflies) and Odonata (dragonflies and damselflies) in ponds may have been affected by the malathion.

Swain (1986 unpubl.) conducted a study on desert grassland in New Mexico to determine the effects of malathion ULV (8 oz/acre–0.58 lb AI/acre), carbaryl (0.54 lb AI/acre), and 2 percent (AI) carbaryl bran bait (1.5 lb/acre) on nontarget arthropods. Her study showed that mean abundance of most groups of nontargets declined immediately after treatments. In particular, all treatments seemed to affect populations of ants and only the insecticidal sprays affected populations of spiders.

Swain (1986) and Quinn et al. (1990, 1991, 1993) found that large-scale application of insecticidal sprays and baits had little long-term impact on the groups of nontargets examined. For example, my team found that activities of four dominant species of ground beetles and three dominant species of darkling beetles rebounded to the pretreatment levels 1 year after treatment. Only one species of darkling beetle, *Eleodes tricostatus*, may have been affected 1 year after treatment. Quinn et al. (1993) also found that field crickets, ichneumonid wasps, and blister beetles, as groups, rebounded to or above the pretreatment levels 1 year after treatment.

Pollinators, such as honey bees and solitary bees, are important components of rangeland and adjacent cropping systems. Although the effects of large-scale control treatments on bees have not been examined thoroughly, insecticidal sprays should be presumed to exert a serious impact on bee populations because they are particularly susceptible to commonly used insecticides (carbaryl, malathion). The effects of insecticides on native bees and rare rangeland plants are reviewed in chapters III.4 and III.5 in this section of the User Handbook.

In summary, large-scale applications of nonselective insecticidal sprays can cause large reductions in populations of nontarget species of arthropods immediately after treatment. Species that are active during treatments or that feed on infected grasshoppers are particularly susceptible. These include ground beetles, darkling beetles, blister beetles, spiders (especially wolf spiders), field crickets, foraging bees, and ants. In contrast, insecticidal baits affect only species that consume the baits directly or prey that have consumed the baits. These species include darkling beetles, ground beetles, field crickets, and ants.

Although reductions in nontarget arthropods can last throughout the year of application, there is little evidence that grasshopper control treatments cause any long-term effects on nontargets. Besides the resiliency of populations, there may be numerous other explanations for this lack of evidence of long-term treatment effects. Inadequate sample sizes and large population variability inevitably lead to a conclusion that treatments have no effect, when in fact, one may exist. No studies of nontarget arthropods have examined the possibility of making such an error (by conducting a statistical power analysis). An additional problem with existing studies is that they frequently assess effects on whole families and not species. When lumping of species is done, species emerging after treatments can dilute the effects of treatments and cause one to find no treatment effect when one actually exists (Quinn et al. 1993). Thus, these studies must be viewed with caution.

Effect of Control Treatments on Grasshopper Outbreaks

In general, nonselective insecticides can cause pest resurgence when they disrupt populations of natural enemies. Similarly, large-scale grasshopper control programs can potentially *enhance* grasshopper outbreaks by killing off grasshopper predators and parasites or by affecting their behavior. Although it seems clear that insecticide applications can affect natural enemies of grasshoppers, at least in the short term, it is less clear that reductions in natural enemies automatically affect grasshopper population dynamics.

Several chapters in this User Handbook address the effects of natural enemies on grasshoppers. Results from studies summarized in these chapters indicate that grasshoppers are attacked by a wide variety of predators and parasites and that grasshopper mortality can be quite high, at least on a local level. For example, birds can reduce grasshopper densities by 30 to 50 percent (see chapter I.10 on “Birds and Wildlife as Grasshopper Predators”). Parker and Wakeland (1957) estimated that an average of 19 percent of grasshopper egg pods were destroyed by predators but that at the local level, mortality may be as high as 100 percent. Parasitism rates of grasshoppers can also be quite high at the local level (exceeding 50 percent), although they do not usually exceed 10 percent (Lavigne and Pfadt 1966, Rees 1973). As discussed by Capinera (1987), the collective effects of all the different mortality factors may add up to an overall large effect on grasshoppers. It seems clear that we should not underestimate the effects of grasshopper natural enemies and that we should work to preserve these organisms.

There is some evidence that grasshopper populations are regulated by natural enemies (particularly birds) under certain conditions (see chapter VII.14 on “Grasshopper Population Regulation”). In effect, natural enemies may be responsible for keeping grasshopper populations at low levels. Once the natural enemies are removed (for example, by nonselective insecticides), then grasshopper populations can no longer be regulated and outbreaks can occur. Once grasshoppers reach high densities, natural enemies are no longer able to suppress their populations. Unfortunately, few studies have examined the role of natural-enemy reductions, caused by nonselective insecticides, on subsequent grasshopper outbreaks.

In a review of grasshopper population dynamics over several years, Lockwood et al. (1988) found that the duration and stability of grasshopper outbreaks were greater in northern Wyoming, compared with southern Montana, and suggested that the more intensive grasshopper control programs in Wyoming may have contributed to this. In a study of the effects of an insecticidal spray (malathion) and bait (carbaryl on bran) on grasshopper and nontarget arthropod populations, Quinn et al. (1989, 1991, 1993) found that populations of most dominant grasshopper species, four species of ground beetles, and numbers of other nontargets rebounded to or above pretreatment levels a year after treatment. An exception was *Ageneotettix deorum*. Densities of this species remained low a year after treatment. These results indicate that some nontarget arthropods and grasshopper species are very resilient. Clearly, until more is known about the effects of natural enemies on grasshopper population dynamics and the effects of grasshopper control programs on resiliency of natural enemies, scientists and land managers should act to preserve these communities.

References Cited

Capinera, J. L. 1987. Population ecology of rangeland grasshoppers. In: Capinera, J. L., ed. Integrated pest management on rangeland: a shortgrass prairie perspective. Boulder, CO: Westview Press: 162–182.

Lavigne, R. J.; Pfadt, R. E. 1966. Parasites and predators of Wyoming rangeland grasshoppers. Sci. Monogr. 30. Laramie, WY: University of Wyoming and Wyoming Agricultural Experiment Station. 31 p.

Lockwood, J. A.; Kemp, W. P.; Onsager, J. A. 1988. Long-term, large-scale effects of insecticidal control on rangeland grasshopper populations (Orthoptera: Acrididae). Journal of Economic Entomology 81: 1258–1264.

Parker, J. R.; Wakeland, C. 1957. Grasshopper egg pods destroyed by larvae of bee flies, blister beetles, and ground beetles. Tech. Bull. 1165. Washington, DC: U.S. Department of Agriculture: 1–29.

Pfadt, R. E.; Marston, N. L.; Lawson, F. A. 1985. Effect of ULV malathion grasshopper spray on non-target arthropods. Sci. Monogr. 48. Laramie, WY: University of Wyoming and Wyoming Agricultural Experiment Station. 31 p.

Quinn, M. A.; Kepner, R. L.; Walgenbach, D. D.; Bohls, R. A.; Pooler, P. D.; Foster, R. N.; Reuter, K. C.; Swain, J. L. 1989. Immediate and 2nd-year effects of insecticidal spray and bait treatments on populations of rangeland grasshoppers. Canadian Entomologist 121: 589–602.

Quinn, M. A.; Kepner, R. L.; Walgenbach, D. D.; Foster, R. N.; Bohls, R. A.; Pooler, P. D.; Reuter, K. C.; Swain, J. L. 1990. Effect of habitat and perturbation on populations and community structure of darkling beetles (Coleoptera: Tenebrionidae) on mixed-grass rangeland. Environmental Entomology 19: 1746–1755.

Quinn, M. A.; Kepner, R. L.; Walgenbach, D. D.; Foster, R. N.; Bohls, R. A.; Pooler, P. D.; Reuter, K. C.; Swain, J. L. 1991. Effect of habitat characteristics and perturbation from insecticides on the community dynamics of ground beetles (Coleoptera: Carabidae) on mixed-grass rangeland. Environmental Entomology 20: 1285–1294.

Quinn, M. A.; Kepner, R. L.; Walgenbach, D. D.; Foster, R. N.; Bohls, R. A.; Pooler, P. D.; Reuter, K. C.; Swain, J. L. 1993. Grasshopper stages of development as indicators of nontarget arthropod activity: implications for grasshopper management programs on mixed-grass rangeland. Environmental Entomology 22: 532–540.

Rees, N. E. 1973. Arthropod and nematode parasites, parasitoids, and predators of Acrididae in America north of Mexico. Tech. Bull. 1460. Washington, DC: U.S. Department of Agriculture. 288 p.

Swain, J. L. 1986. Effect of chemical grasshopper controls on non-target arthropods of rangeland in Chaves County, New Mexico. M.S. thesis. Las Cruces, NM: New Mexico State University. 102 p.

III.4 Direct and Indirect Effects of Insecticides on Native Bees

D. G. Alston and V. J. Tepedino

NOTE: Acephate is no longer approved by EPA for rangeland grasshopper control.

The successful reproduction of plants in both natural and agricultural ecosystems is highly dependent upon adequate populations of pollinators. The role of bees as pollinators in natural ecosystems, such as rangelands, is less obvious to the casual observer. The fact is that the majority of rangeland plants require insect-mediated pollination. Native, solitary bee species are the most important pollinators on western rangelands (Tepedino 1979).

Indiscriminate use of broad-spectrum insecticides is likely to cause changes throughout the rangeland community. In addition to controlling the targeted pest (grasshoppers), rangeland insecticides can have direct and indirect effects on nontarget insects and related animals (see also III.3). Potential negative effects of insecticides on pollinators are of special concern because a decrease in their numbers has been associated with decline in fruit and seed production of plants. And this decline may have dramatic repercussions throughout the rangeland food chain. Some of the possible negative effects to the ecosystem include changes in future vegetation patterns via plant competition, reduction in seed banks, and influences on the animals dependent upon plants for food.

Direct effects are those that are lethal in nature and cause direct mortality that can be attributed to use of insecticides. Indirect or sublethal effects are much more difficult to document. They generally act over a longer period of time and can result in negative effects on reproductive potential, lifespan, activity levels, body size, and behavior of current and future generations.

Important Characteristics of Native Bees

When choosing the timing of insecticide applications to rangelands, one should consider some important characteristics of native bees, of the insecticide applied, and of the growth cycle of native plants. The typical solitary bee overwinters in its nest and emerges as an adult the following spring to early summer (fig. III.4–1). Adult females are exclusively responsible for feeding the young and thus play the major role in plant pollination while foraging for nectar and pollen.

There is tremendous variation among bee species in the length of time that adults are active and foraging (fig. III.4–1). The seasonal activity period of solitary bees may extend from spring through early fall due to multiple generations per year and continual availability of blooming plants. Therefore, land managers cannot assume that simply avoiding the application of insecticides on rangeland during the major time of plant bloom will avoid endangering the native bee population.

Exposure of bees to insecticides is also influenced by foraging behavior and flight distance. For most native bees, our knowledge of foraging behavior is limited to information on flower associations, such as a particular species that has been seen collecting the pollen and/or nectar of certain plants. The leaf-cutting habit of the alfalfa leafcutter bee makes it particularly susceptible to residues of contact insecticides on plant foliage. Contaminated leaves, mud, water, or resins used for nest construction may result in detrimental effects to the young. Bees' flight range can greatly affect their exposure to insecticides. Extensive flight distances between nests and flowering plants increase their foraging time and make them more vulnerable to insecticides (see III.8).

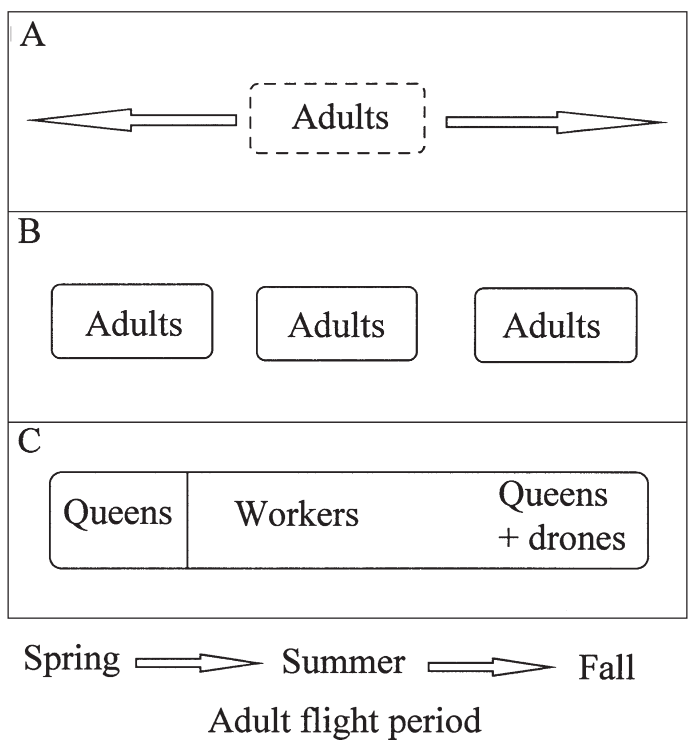

Figure III.4–1—Adult flight periods for three general life cycles of native bees: (A) Single generation per year, e.g., *Nomia* or *Osmia;* dotted lines indicate that flight period can shift in time depending on species. (B) Two or more generations per year, e.g., *Megachile* or *Ashmeadiella.* (C) Social, e.g., *Bombus.*

Body size of native bees also may affect susceptibility to insecticides in field situations. The greater surface-to-volume ratio of small bees increases their relative exposure to contact insecticides (Johansen 1972). Studies in a Montana forest (Flavell et al. 1975) found that, although the total bee population was not reduced following an application of the insecticide trichlorfon, the percentage of smaller bees (predominantly solitary species) present in the forest was significantly reduced. If this same effect is found in other ecosystems, then the greater susceptibility of smaller bees to insecticides is of particular concern for western rangelands.

Important Characteristics of Insecticides

Pesticide formulation strongly influences toxicity. Dusts and wettable powders tend to be more hazardous to bees than solutions or emulsifiable concentrates, while granular and bait formulations are generally low in hazard. Application technique is also important in determining toxicity; aerial spraying offers less opportunity for avoidance behavior and greatly increases drift (National Research Council of Canada 1981).

Currently, only broad-spectrum insecticides (acephate, carbaryl, and malathion) are registered for use on rangelands for grasshopper control. All three have received a high toxicity rating for their negative effects on bees (National Research Council of Canada 1981, Johansen and Mayer 1990, Johansen et al. 1983), and, therefore, are not registered for use on blooming crops or weeds if commercial bees are visiting the treatment area. Yet these insecticides are being sprayed on rangelands when native plants are in bloom and being visited by pollinators. Contact sprays can be very toxic to small, native bees because of direct contact with the insecticide or insecticide residue. Therefore, insecticides that are more selective in activity are highly desirable to reduce negative effects on bees.

One insecticide with promise for selectivity is carbaryl incorporated into bran flakes. Because such flakes act only upon ingestion, they are much more selective than contact formulations (Peach et al. 1994). Bees likely would encounter bran bait only when gathering pollen and nectar from open upright flowers into which particles of bait have fallen. Ingestion of the insecticide would have to occur in order for the bee to receive a toxic dose.

Lethal Effects

The direct, or lethal, effects of insecticides on bees have been the focus of much research. The majority of toxicological information has been obtained for three distantly related species: *Apis mellifera,* the honey bee; *Nomia melanderi,* the alkali bee; and *Megachile rotundata,* the alfalfa leafcutting bee. Toxicological data for the latter two species are of greater relevance to natural situations because of these bees' solitary nesting lifestyle and the primary role of adult females in foraging activities and provisioning the young. The greatest body of toxicity literature exists for the honey bee, but unfortunately these data have proved of limited use in prediction of toxicity to many species of native bees because of the major differences in lifestyle, behavior, physiology, and size.

On western rangelands where native plants are rare or their populations threatened, bait formulations of carbaryl have been suggested as a possible alternative to contact sprays. Liquid formulations of carbaryl can be quite toxic to all three bee species previously mentioned when bees directly contact insecticides or insecticide residues (Johansen and Mayer 1990). In contrast, under laboratory conditions, only extremely high doses of ingested carbaryl resulted in toxic effects to alfalfa leafcutting bee larvae when incorporated into the pollen provision either as liquid (Guirguis and Brindley 1974) or as bran bait (Peach et al. 1994). Such high rates of carbaryl are much greater than a bee would encounter in the field.

There were also no lethal effects of carbaryl bran bait on adult alfalfa leafcutting bees, even when they were fed a sustained diet of honey solution contaminated with carbaryl bait for up to 40 days (Peach et al. 1994). Other studies have found that young adult bees of this species (up to 4 days old) readily detoxify topically applied carbaryl, but this ability rapidly declines after day 4 (Lee and Brindley 1974).

Sublethal Effects

Other effects of insecticides to bees may not be as obvious. The long-term sublethal effects of insecticides to bees that would be most likely to lower visitation rates to flowers, and thereby reduce plant reproductive success, include negative changes in longevity of bees, adult activity levels, and number, size, and sex ratio of offspring produced. Such chronic effects could occur from the slow poisoning of the young through ingestion of contaminated pollen and exposure of foraging bees to insecticides through translocation in nectar. Although sublethal effects of insecticides can be subtle, in the long run they may have as great a weakening effect on bee populations as the mortality caused by direct toxicants.

Although few studies have addressed the subtle effects of insecticides on bees, some detrimental effects have been found. Female alfalfa leafcutting bees treated with contact applications of organophosphate insecticides showed reduced longevity and lower nesting rates and egg production than bees not treated (Torchio 1983, Tasei and Carre 1985, Tasei et al. 1988).

Approximately 40 percent of larvae of this bee fed provisions contaminated with deltamethrin could not successfully complete development (Tasei et al. 1988). However, studies with carbaryl bran bait found no sublethal effects on adults or larvae (Peach et al. 1994). There seems to be little reason for concern that any carbaryl eaten by foraging adult females from the nectar of open flowers will affect any aspect of reproduction. Again, it appears that the use of carbaryl bran bait on rangelands is a relatively safe option for pollinators (fig. III.4–2).

Figure III.4–2—Domestic bees often need protection during grasshopper conrol treatments using chemical sprays. Beekeepers can move the bees out of the application area, or control-program managers can leave a sufficient buffer zone to protect the bees. Applications of bran bait normally will be of little concern for beekeepers. (APHIS file photo.)

Implications for Management of Grasshoppers on Western Rangelands

Because of the multiple-use concept employed by managers of public lands, there is certain to be continual conflict among different users of the lands. The U.S. Department of Agriculture, Forest Service and the U.S. Department of the Interior, Bureau of Land Management have the unenviable task of making land-management decisions based on wide-ranging demands and input from recreational use and preservation of biodiversity to logging, mining, and grazing. Because of the current status of pest management technology, it is likely that use of insecticides for control of grasshoppers on western rangelands will continue for some time. Despite this current situation of conflict, there does appear to be some alternative in choice of insecticides that are more selective in their effects to nontarget plants and animals.

One such selective insecticide that appears well suited for use on rangelands is carbaryl bran bait. Demanding laboratory and greenhouse tests performed with the alfalfa leafcutting bee, a solitary nester, found no lethal or sublethal effects on adults and only minimal effects on larvae when doses much higher than would be encountered in the field were incorporated into their pollen provisions. However, there are more limitations to choosing carbaryl bran bait as a rangeland pest control tool. Because not all grasshopper species feed equally well on the bait (see II.12), proper identification of grasshopper species is especially important.

Although carbaryl bran bait may be a relatively safe option for a representative solitary bee, no one should feel comfortable with this assessment until there is further research on other pollinator species' susceptibility to various insecticides. Such research is critical for the preservation of insect biodiversity, as well as the biodiversity of the plants whose flowers cannot reproduce sexually without insect visits.

References Cited

Flavell, T. H.; Tunnock, S.; Barry, J. W.; Ekblad, R. B.; Ciesla, W. M. 1975. Western spruce budworm: a pilot control project with carbaryl and trichlorfon—environmental monitoring of aquatic organisms, birds, and insect pollinators. March 1978. Rep. 787-5. Missoula, MT: U.S. Department of Agriculture, Forest Service.

Guirguis, G. N.; Brindley, W. A. 1974. Insecticide susceptibility and response to selected pollens of larval alfalfa leafcutting bees, *Megachile pacifica* (Panzer)(Hymenoptera: Megachilidae). Environmental Entomology 3: 691–694.

Johansen, C. A. 1972. Toxicity of field-weathered insecticide residues to four kinds of bees. Environmental Entomology 1: 393–394.

Johansen, C. A.; Mayer, D. F.; Eves, J. D.; Kious, C. W. 1983. Pesticides and bees. Environmental Entomology 12: 1513–1518.

Johansen, C. A.; Mayer, D. F. 1990. Pollinator protection: bee and pesticide handbook. Cheshire, CT: Wicwas Press. 212 p.

Lee, R. M.; Brindley, W. A. 1974. Synergist ratios, EPN detoxification, lipid, and drug-induced changes in carbaryl toxicity in *Megachile pacifica.* Environmental Entomology 3: 899–907.

National Research Council of Canada. 1981. Pesticide–pollinator interactions. Ottawa, ON: National Research Council of Canada. 190 p.

Peach, M. P.; Alston, D. G.; Tepedino, V. J. 1994. Bees and bran bait: is carbaryl bran bait lethal to alfalfa leafcutting bee (Hymenoptera: Megachilidae) adults or larvae? Journal of Economic Entomology 87: 311–317.

Tasei, J. N.; Carre, S. 1985. Effets du traitement de luzerne en fleurs *(Medicago sativa* L.) avec de la deltamethrine et de la phosalone sur l’abeille solitaire: *Megachile rotundata* F. (Hym., Megachilidae). Acta Oecologica 6: 165–173.

Tasei, J. N.; Carre, S.; Moscatelli, B.; Grondeau, C. 1988. Recherche de la D.L. 50 de la deltamethrine (Decis) chez *Megachile rotundata* F. abeille pollinisatrice de la luzerne (*Medicago sativa* L.) et des effets de doses infralethales sur les adultes et les larves. Apidologie 19: 291–306.

Tepedino, V. J. 1979. The importance of bees and other insect pollinators in maintaining floral species composition. Great Basin Naturalist Memoirs 3: 139–150.

Torchio, P. F. 1983. The effects of field applications of naled and trichlorfon on the alfalfa leafcutting bee, *Megachile rotundata* (Fabricius). Journal of the Kansas Entomological Society 56: 62–68.

III.5 The Reproductive Biology of Rare Rangeland Plants and Their Vulnerability to Insecticides

Vincent J. Tepedino

NOTE: Acephate is no longer approved by EPA for rangeland grasshopper control.

The Western United States is an area of high plant and animal diversity. Many of the plants on this vast expanse of mountain, plain, and desert occur nowhere else in the world (Cronquist et al. 1972, Barbour and Billings 1988). Currently about 150 of these plant species are so rare that they have been listed under the Endangered Species Act as either threatened or endangered. Four are shown in figure III.5–1 (a–d). Most of these rare plants have been found on public rangelands (fig. III.5–2).

Figure III.5–1—Rare rangeland plants. A = Blowout penstemon (Nebraska), B = Dwarf bear-poppy (Utah), C = Dudley Bluffs twinpod (Colorado), D = San Rafael cactus (Utah).

Preserving rare plant species means removing or reducing threats to existing individuals and ensuring that those individuals can reproduce. Plants reproduce both asexually and sexually. For example, the rare plants *Cycladenia humilis* var. *jonesii* in Utah and *Mirabilis macfarlaneii* in Idaho and Oregon both reproduce sexually by seeds and asexually by the production of rhizomes. However, in seed plants, sexual reproduction is the predominant method. All rare plants that my associates and I studied and described in this chapter reproduce sexually. Sexual reproduction is particularly important because it enables plants to generate and maintain in their offspring the genetic variability necessary to cope with unusual circumstances. In contrast, asexual reproduction produces only copies of the parent plant, not variations on the theme.

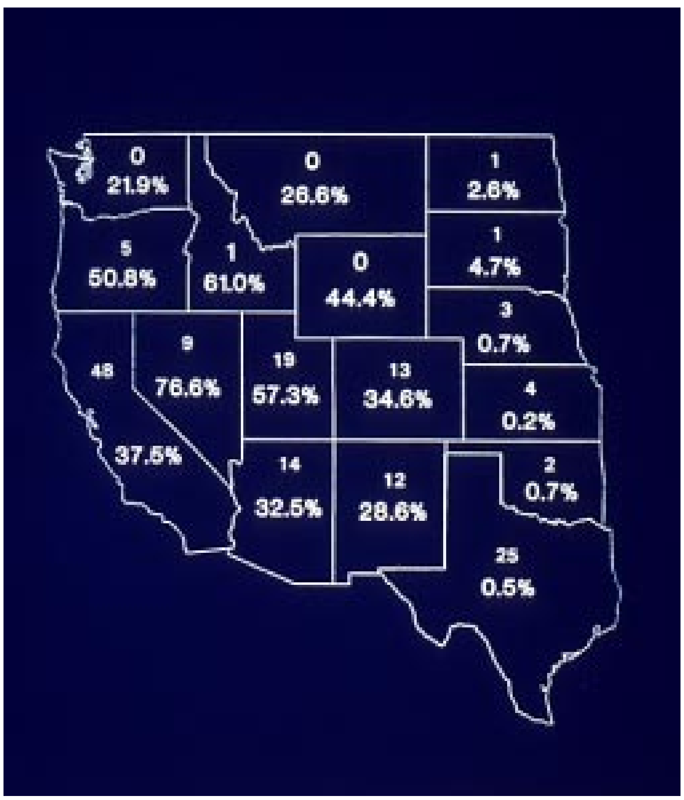

Figure III.5–2—Number of threatened and endangered plant species listed under the Endangered Species Act as of August 1993 (U.S. Fish and Wildlife Service 1993, upper figure) and percent total area administered by the Bureau of Land Management and Forest Service (lower figure), by State, in the West.

In seed plants, sexual reproduction depends on the movement of mature pollen from the anthers to a receptive stigma (pollination). To complete the process, pollen grains must germinate and send pollen tubes down the style to fertilize one or more ovules in the ovary (fertilization). Sexual reproduction may take place between individuals, or individuals may fertilize themselves if they are self-compatible, meaning their stigmas are receptive to their own pollen.

Because plants are immobile, they require “go-betweens” to move pollen from anthers to stigma. Such assistance comes mostly from insects–although wind, water, gravity, and other animals may occasionally be agents of pollination for some species. Although butterflies, moths, flies, ants, and beetles may pollinate flowers as they visit them to eat pollen and/or nectar, the truly essential pollinators for North American flowering plants are bees.

The bees to which we refer are not honeybees, which are of Eurasian origin, but native bees, which have evolved in North America. The North American bee fauna is quite diverse. In the State of Wyoming alone, there are more than 600 species (Lavigne and Tepedino 1976). In the Western United States, there are well over 2,500 species. Many of these bees are quite specialized in the plants that they visit and pollinate. For example, *Perdita meconis*, an uncommon bee that pollinates the endangered dwarf bearclaw poppy, *Arctomecon humillis*, visits only plants in the genera *Arctomecon* and *Argemone* for pollen.

Most bees that visit rare plants are solitary rather than social (the familiar honeybee). Like social bees, solitary bee females care for their offspring. Individual females carefully construct nests without the aid of workers, usually in the ground (fig. III.5–3) or in dead wood (fig. III.5–4). These nests will hold and protect the young bees and the food provided for them. The nesting material varies from species to species and may be quite specific. For example, for certain species, the ground must have a certain slope or soil moisture content or texture (Cane 1991).

Figure III.5–3—Entrance/exit holes at a nest-site of a ground-nesting bee.

Bees provision these nests with pollen and nectar molded into a loaf (fig. III.5–4) for the young to eat. Adults also eat nectar and pollen while foraging. In addition, bees may forage for water or other extraneous materials needed to construct the nest, such as leaf pieces (fig. III.5–5), resin, mud, etc., (Stephen et al. 1969). Adult females must launch many foraging expeditions from their nest-sites to obtain these resources. Frequently the best nesting substrate is not in the same area as food or other necessities, and bees must travel some distance to obtain nest materials.

Unfortunately, bees are generally vulnerable to most commonly used insecticides, including those that are approved for use to control grasshoppers on Federal rangelands: acephate, carbaryl, and malathion (Johansen et al. 1983). Bees that are forced to travel widely to gather their resources are most vulnerable because they must forage over larger areas and are therefore more likely to encounter a spray area. If bees are vulnerable, so may be the plants that depend on them for pollination services. Because of the potential vulnerability of both bees and plants, the U.S. Department of the Interior's U.S. Fish and Wildlife Service (FWS) and the U.S. Department of Agriculture's (USDA) Animal and Plant Health Inspection Service must hold joint consultations before aerially treating rangelands with insecticides. Usually, insecticide-free safety zones called buffers must be left around rare plant populations to reduce effects on both plant and pollinators.

Figure III.5–4—The nest of a twig-nesting bee, split open to expose feeding larvae, their food provisions, and the partitions between cells.

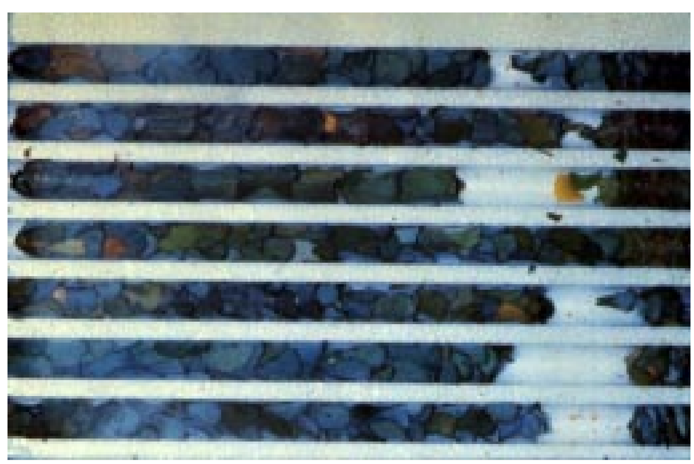

Figure III.5–5—Several leafcutter bee nests in an artificial domicile, exposed to show the numerous cells enfolded in leaves.

Questions about optimal buffer zone size and vulnerability of rare plant reproduction to insecticides are important. If flowers normally self-fertilize automatically, then grasshopper spraying programs are unlikely to be of consequence because pollinators will not be necessary for reproduction. Thus, scientists first must determine whether the flowers of the plant species in question are capable of self-fertilization, and, second, if self-fertilization is automatic. We also must determine whether fruit and seed set are improved by cross-pollination and identify the agents of pollination. When this is accomplished, we will have described the breeding system of the plant and will have some idea about the life history of its pollinators.


Figure III.5–6—Cattle grazing at a Brady pincushion cactus site (Arizona).

The size of the buffer zone that should be left around rare plant populations that rely exclusively on insect pollination depends on how far bees fly to obtain their resources. Presently, a buffer zone of 3 miles is being left around rare plant populations, but this is provisional in that it is based on best guesses rather than accurate estimates. By experimentation, we can help resolve questions about the value of buffer zones and whether they should be expanded or contracted in size.

Conducting a Study

To uncover general patterns in the reproductive biology of rare plants on western rangelands, I elected to study the breeding systems and pollinators of a large number of species rather than to conduct very detailed studies on a few species.

I gave study priority to rare plant species on actively grazed public rangelands (fig. III.5–6) in counties with high probabilities of having large numbers of grasshoppers, and thus of being sprayed. The approximate locations of the species studied are shown in figure III.5–7. With two exceptions *(Penstemon harringtonii* in Colorado and *Castilleja aquariensis* in Utah), all are listed as threatened or endangered under the Federal Endangered Species Act.

To describe the plant breeding system, we conducted a series of experiments using mesh bags or cages to prevent insects from visiting the flowers. Individual flowers, entire inflorescences (flower clusters), or entire plants (where necessary) were bagged or caged just prior to the onset of flowering (fig. III.5–8). Each of the following treatments was applied to a different flower: for self-pollination, flowers were hand-pollinated with the pollen of another flower on the same plant; for cross-pollination, flowers were hand-pollinated with pollen from a flower on a distant plant; to test for automatic self-pollination, flowers were left untreated; and, as a control, some flowers were left unbagged (open-pollinated). My associates and I carried out a complete series of treatments, one of each, on each of 15 to 25 experimental plants. Treatments were randomized on each plant to remove any effects of order or position on fruit or seed set.

We observed and collected naturally occurring pollinators as they visited the flowers during several time periods each week. Insects were pinned and identified later using the insect collections at the USDA, Agricultural Research Service, Bee Biology and Sytematics Laboratory in Utah, and the collection at Utah State University.

Estimating the distances a bee typically flies on its foraging trips proved very difficult because of its size, the speed at which it moves, and the size of the area to be monitored. Because native bees are too small to track with radio collars or electronic chips, as many mammals and birds can be, other methods were necessary. We used both direct (A below) and indirect (B, C, D) methods:

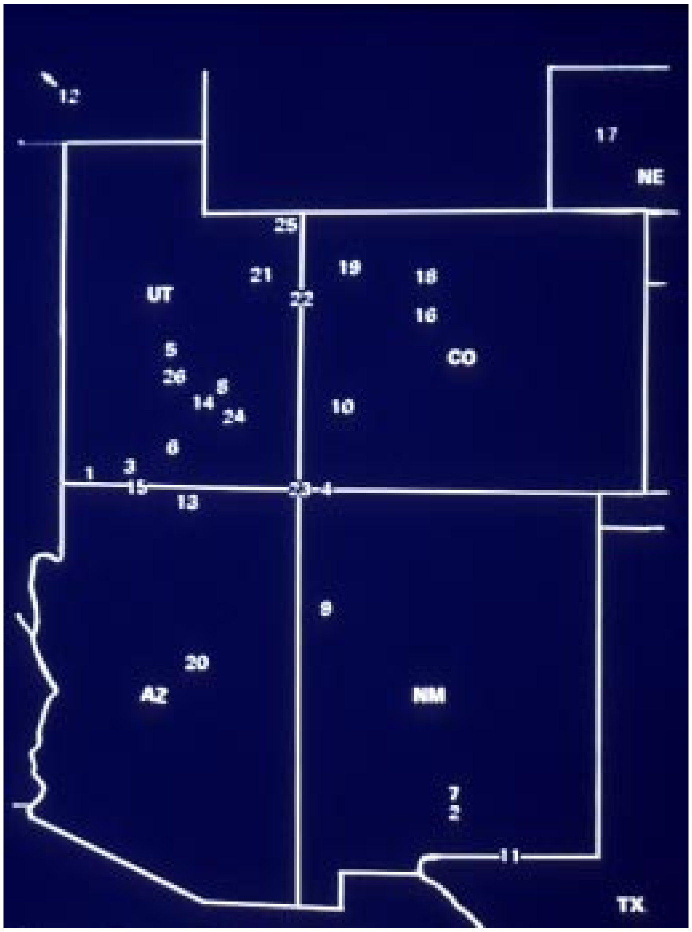

Figure III.5–7—Locations of specific threatened and endangered plants studied from 1988 to 1993. 1 = dwarf bear-poppy, 2 = Sacramento prickly-poppy, 3 = Welsh's milkweed, 4 = Mancos milkvetch, 5 = Heliotrope milk-vetch, 6 = Aquarius paintbrush, 7 = Sacramento Mountains thistle, 8 = Jones' cycladenia, 9 = Zuni fleabane, 10 = clay-loving wild-buckwheat, 11 = McKittrick pennyroyal, 12 = McFarlane's four-o'clock, 13 = Brady pincushion cactus, 14 = San Rafael cactus, 15 = Siler pincushion cactus, 16 = Harrington beard-tongue, 17 = blowout penstemon, 18 = Penland beard-tongue, 19 = Dudley Bluffs twinpod, 20 = Arizona cliffrose, 21 = shrubby reed-mustard, 22 = Uinta Basin hookless cactus, 23 = Mesa Verde cactus, 24 = Wright fishook cactus, 25 = Ute ladies'-tresses, 26 = last chance townsendia.

(A) Foraging bees were captured, marked on the thorax with a dot of water-resistant paint that was nontoxic to plants and insects, released, and then searched for on subsequent days at other plant populations at set distances from the marking site (fig. III.5–9 and 10).

(B) Nontoxic fluorescent powders (pollen analogs or imitators) were placed in "donor" flowers, where they would be picked up and spread by foraging bees, and were searched for in the evening with a black light in other flowers at different distances from the donors.

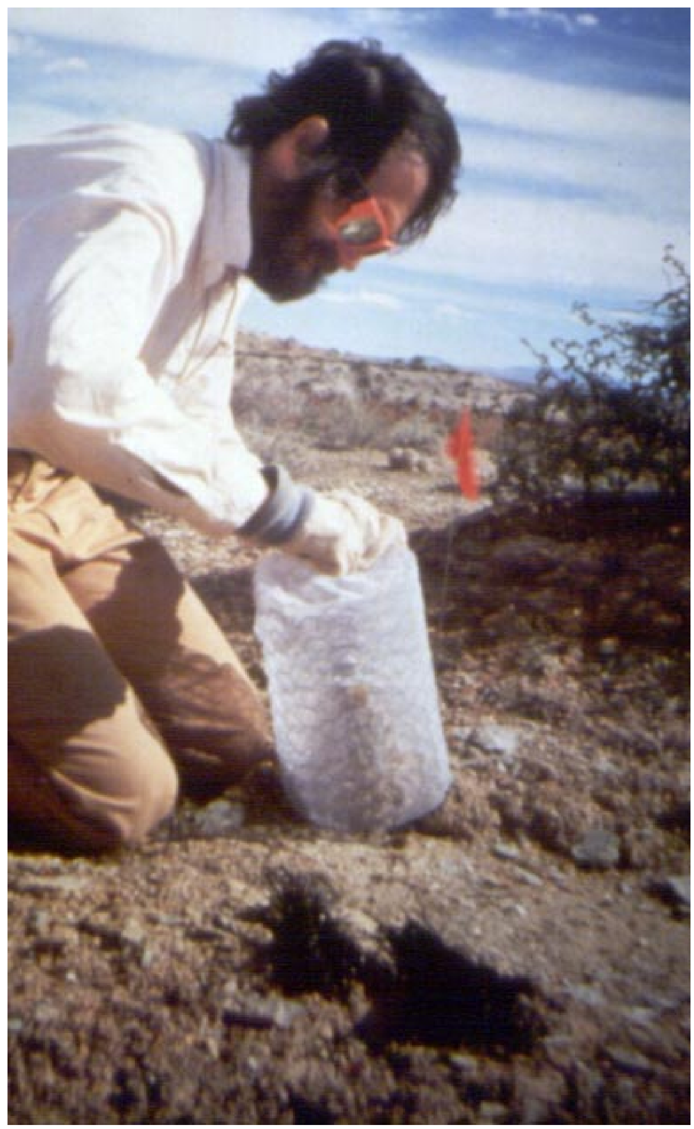

Figure III.5–8—Fitting a cage over a cactus plant to exclude insects.

Figure III.5–9—Coaxing a bee into a marking tube.

Figure III.5–10—The coaxed bee marked on the thorax.

Figure III.5–11—An artificial bee “condominium” offers bees cheap housing.

(C) Trap-nests (artificial nests that bees will use, figure III.5–11) were placed at different distances from donor flowers, and the provisions of the cells made therein were examined for fluorescent powder.

(D) A “mobile garden,” a pickup truck with a bed full of blooming potted plants, was used to attract marked bees that had earlier foraged on flowers dusted with fluorescent powders (see above) (fig. III.5–12). The “mobile garden” was parked at different distances from areas where bees had been marked and flowers had been dusted. My associates and I then recorded marked bees visiting plants in the garden or any flowers with fluorescent powder deposited on them.

Figure III.5–12—The oldest floating “mobile garden” in Arizona.

Study Results

Three clear patterns were evident from the data. First, rare plants do not tend to be automatic self-fertilizers. Indeed, just the opposite is the case. With the exception of two species *(Astragalus montii* in central Utah and *Schoencrambe suffrutescens* in eastern Utah), all species are primarily outcrossing (table III.5–1). Many are also self-compatible, meaning pollen moved from one flower to another on the same plant will sometimes cause fertilization, but in most cases the fruits and seeds produced are inferior either in number or size to those produced as a result of cross-pollination. In any case, pollinators also are needed to cause this type of self-pollination, which is not automatic.

The second pattern is that the most abundant visitors to the flowers of these plants are almost always native bees (table III.5–1). In some cases, bee pollination is supplemented by other animals. For example, in New Mexico the Sacramento Mountains thistle *(Cirsium vinaceum)* also is pollinated by several species of hummingbirds, flies, and butterflies.

Table III.5–1—Summary of the reproductive characteristics of 26 species of rare plants

Common name	Species name	Status	State	BrSys	I	Pollinators	L
Dwarf bear-poppy	*Arctomecon humilis*	E	UT	CR SI	Y	Bees, many	N
Sacramento prickly-poppy	*Argemone pleiacantha pinnatisecta*	E	NM	CR PS	Y	*Dialictus*	?
Welsh's milkweed	*Asclepias welshii*	T	UT	?	Y	Bees, wasps	?
Mancos milk-vetch	*Astragalus humillimus*	E	CO NM	CR SC	Y	Bees, many	N
Heliotrope milk-vetch	*Astragalus montii**	T	UT	AS SC	?	*Osmia*	N
Aquarius paintbrush	*Castilleja aquariensis**		UT	CR SI	Y	*Bombus*	?
Sacramento Mountains thistle	*Cirsium vinaceum*	T	NM	CR PS	Y	Various	?
Jones cycladenia	*Cycladenia humilis* var. *jonesii**	T	UT	CR SI	Y	Bees, many	?
Zuni fleabane	*Erigeron rhizomatus*	T	NM	CR PS	Y	Various	N
Clay-loving wild-buckwheat	*Eriogonum pelinophilum*	E	CO	CR SC	Y	Various	?
McKittrick pennyroyal	*Hedeoma apiculatum*	T	NM TX	CR SC	Y	Halictidae	N
MacFarlane's four-o'clock	*Mirabilis macfarlanei**	E	ID OR	CR PS	Y	Bees, many	?
Brady pincushion cactus	*Pediocactus bradyi*	E	AZ	CR SI	Y	*Dialictus*	N
San Rafael cactus	*Pediocactus despainii*	E	UT	CR SI	Y	Bees, many	N
Siler pincushion cactus	*Pediocactus sileri*	E	AZ UT	CR SI	Y	Bees, many	N
Harrington beardtongue	*Penstemon harringtonii*		CO	CR PS	Y	Bbees, many	?
Blowout penstemon	*Penstemon haydenii*	E	NE	CR PS	Y	Bees, many	N
Penland beardtongue	*Penstemon penlandii*	E	CO	CR SC	Y	Bees, many	N
Dudley Bluffs twinpod	*Physaria obcordata*	T	CO	CR SI	Y	Bees, many	N
Arizona cliffrose	*Purshia subintegra*	E	AZ	CR PS	Y	Bees, many	Y
Shrubby reed-mustard	*Schoencrambe suffrutescens**	E	UT	AS SC	?	Halictidae	N
Uinta Basin hookless cactus	*Sclerocactus glaucus**	T	CO UT	CR SI	Y	Bees, many	Y
Mesa Verde cactus	*Sclerocactus mesae-verdae**	T	CO NM	CR PS	Y	Halictidae	N
Wright fishhook cactus	*Sclerocactus wrightiae*	E	UT	CR SI	Y	Halictidae	N
Ute ladies'-tresses	*Spiranthes diluvialis**	T	CO UT	CR SC	Y	*Bombus*	N
Last chance townsendia	*Townsendia aprica*	T	UT	CR PS	Y	*Osmia*	N

T = threatened, E = endangered. BrSys describes the plant's breeding system: CR = cross-pollinated, AS = automatic self-pollination, SI = self-incompatible, SC = self-compatible; PS = partially self-compatible. I = insect pollinated, Y = yes. Pollinators: genus or family of bee given when possible, many = several bee taxa, various = several animal taxa. L = evidence that fruit or seed set is being limited by inadequate pollination, N = no, Y = yes; * = uncommonly visited species.

The third pattern is that the flowers of about one-third of the plant species studied received few visits (table III.5–1). For several species, insect visitation was so low that we were forced to abandon the original pollinator observation and collection schedules. In these cases insects were simply captured whenever possible. Such low numbers of flower visitors are of concern, especially for rare plants that can produce seeds only when visited by pollinators.

These experiments also can be used to indicate species that may be producing fewer than the highest number of seeds, perhaps because of insufficient pollinator visits. Species whose seed production is low are of special concern because they may not be producing enough new individuals to replace those that are dying. Fortunately, only *Purshia subintegra* in central Arizona and *Sclerocactus glaucus* in eastern Utah gave any indication of underpollination. Because these two species set significantly fewer seeds in open-pollinated treatments than in cross-pollinated treatments, these plants should be studied further to determine if underpollination is common.

My results in estimating distances traveled by foraging bees were surprising. While it was easy to recapture bees in the general vicinity in which they were marked, or to detect fluorescent powders in flowers in the general area of the donor flowers, it was very difficult to find either marked bees or fluorescent particles at distances beyond a few dozen yards from the marking point. The record for distance moved was about a quarter mile (400 m) from a donor flower in a study of *Pediocactus sileri* in northern Arizona (Peach et al. 1993).

Implications for Chemical Sprays

To say that most plants reproduce sexually and that most depend on insects to pollinate them does not necessarily mean that rare plants do so. Indeed, prior to this study, there were reasons to suspect that rare plants were more likely than common plants to automatically self-pollinate and less likely to require insect visitors to achieve sexual reproduction (Tepedino 1979, Karron 1991). If this were true, then insecticide spraying for grasshoppers would have little effect on reproduction by rare plants, and land managers would not need to be concerned about the potential effects on the plants' pollinators.

The results obtained in this study show that rare plants on rangelands do not commonly self-pollinate. Almost all species studied set seed only when native bees visit their flowers. Because these bees are likely susceptible to liquid insecticide sprays, land managers should consider the implications of some reduction in pollinators as a result of spraying. Significant reduction of pollinators is likely to reduce the seed production of rare plants.

In addition, land managers should consider that many of the insect pollinators may be vulnerable to insecticides at any time of the year. Unless there is a perfectly synchronized, one-generation-per-year specialist pollinator for a plant, and my associates and I found none of those, the conservative approach—until more is known—is to avoid spraying within the buffer zone around each rare plant population at any time. However, if the plan is to use carbaryl bran bait (2 percent active ingredient), a nonliquid treatment, no buffer zones are needed (see III.4).

Overall, the pollinator situation on Federal rangelands may not be as perilous as some scientists had feared. Despite past spraying history, there is little indication that rare plants on rangelands are currently producing fewer seeds than they are capable of producing. While this is a conclusion that cries out for additional corroboration, it is also encouraging to find that seed production of open-pollinated flowers of rare plants do not seem to be pollinator limited. In most cases, visitation rates of bees to flowers, and by implication, bee numbers, appear to be sufficient to support maximum seed production. It is probable that bee numbers and seed production of native forbs have not been impacted because large-scale insecticide spray programs to control or suppress populations of grasshoppers on rangeland are not usually applied in the same areas in successive years. This policy must continue if rangeland pollinators are to have ample time to recover from spray episodes. Other researchers working in Canadian forests have shown that bee numbers will usually return to prespray levels in 1 to 3 years, depending upon the species of bee and the insecticide used (Plowright and Thaler 1979, Kevan and LaBerge 1979, Wood 1979, Miliczky and Osgood 1979). Recovery times and patterns for rangeland pollinators also should be studied.

Scientists regard the absence of evidence for long-distance movement of pollen grain analogues (fluorescent powders) less as evidence that native bees do not move long distances than as an indication of a logistical problem in testing. It is simply impossible for one or two people effectively to cover the area that must be censused. A complicating factor is that every study to look at pollen dispersal has reported drastic reductions in pollen deposition with distance (Handel 1983). By the time one samples flowers more than 33 ft (10 m) from the source, the number of pollen grains deposited is minimal. Again, this does not mean that pollen flows only over very short distances but that investigators are faced with detecting a very small needle in a very large haystack.

Other studies of bee movement and gene flow are of little help because they are invariably conducted over relatively short distances (Handel 1983). Pollen can, however, move long distances. Kernick (cited in Levin 1984) noted that several species of crop plants must be isolated by as much as 1.24 miles (2 km) to maintain varietal purity. Several other studies have examined the homing ability of solitary species of bees. They have shown that bees are capable of returning to their nests from distances of up to 5 miles (Fabre 1925, Rau 1929 and 1931; reviews by Packer 1970 unpubl. and Roubik 1989). While such experiments in no way tell us the distance that a bee normally flies on a typical foraging trip, they help to put an upper bound on bees' movements.

Conclusions

Although much valuable information has been obtained on both plants and their pollinators, much remains to be done. There are four areas in which additional research should be encouraged. First, the pollination biology of other plant species listed under the Endangered Species Act must be studied. The Grasshopper Integrated Pest Management Project has supported studies of 26 species in 13 families (see table III.5–1) or roughly 17 percent of the plant taxa in the Intermountain West which are listed under the Endangered Species Act. Thus, we feel confident in concluding that, in general, the flowers of rare plants must be pollinated by native bees to produce seeds. However, unless administrators and land managers are willing to assume that all rare plants must be managed as if they required bee pollinators, the reproductive biology of the remaining species must be studied.

Second, to make informed recommendations about the size of buffer zones to be left around rare plant populations, better information is needed on the distances pollinators and/or pollen travel. Laboratory methods that demonstrate genetic differences between the enzymes produced by different plants can be used, together with theoretical population genetic models, to provide information on gene flow between plant populations separated by a range of distances and on the genetic isolation of selected plant populations (Slatkin 1985 and 1993, Slatkin and Barton 1989). Long-distance pollinator movement can be documented by showing that certain forms of particular enzymes, which are primarily or exclusively restricted to one population, have moved to other populations. Indeed, these techniques can be used to give a rough approximation of the average number of individual plants per generation that are the result of pollen migration between populations.

Third, information is needed on the toxic effects to native bees of the liquid insecticides commonly used to treat rangeland grasshoppers. Current knowledge has been obtained from studies of the honey bee and the alfalfa leafcutter bee (both introduced species) and the alkali bee because they are cultured for crop pollination and are easily obtainable. Little is known about how susceptible the 2,500-plus species of rangeland bees are to insecticides because their populations are too small, or too difficult to obtain, to yield adequate sample sizes for experimentation of this kind. Prior to studying the toxicology to native species, it will be necessary to build up their populations to a sufficient size for experimentation by raising them in large field cages or greenhouses.

Fourth, decisionmakers must be advised when it is safe to spray. As noted earlier in this chapter, such decisions cannot be made by simply using flowering phenology records for the rare plant species because its pollinators may be active at other times of the year. Information must be available on the flight times of adult pollinators and on their activity patterns for the potential season of spraying. Thus far, activity patterns for pollinators of only one rare plant species have been studied (Peach et al. 1993).

References Cited

Barbour, M. G.; Billings, W. D., eds. 1988. North American terrestrial vegetation. New York: Cambridge University Press. 434 p.

Cane, J. H. 1991. Soils of ground-nesting bees (Hymenoptera: Apoidea): texture, moisture, cell depth and climate. Journal of the Kansas Entomological Society 64: 406-413.

Cronquist, A.; Holmgren, A. H.; Holmgren, N. H.; Reveal, J. L. 1972. Intermountain flora, vol. 1. New York: Hafner Publishing Co. 270 p.

Fabre, J. H. 1925. The mason bees. Garden City, NY: Garden City Publishing Co. 315 p.

Handel, S. N. 1983. Pollination ecology, plant population structure, and gene flow. In: Real, L., ed. Pollination biology. New York: Academic Press: 163–211.

Johansen, C. A.; Mayer, D. F.; Eves, J. D.; Kious, C. W. 1983. Pesticides and bees. Environmental Entomology 12: 1513–1518.

Karron, J. D. 1991. Patterns of genetic variation and breeding systems in rare plant species. In: Falk, D. A.; Holsinger, K. E., eds. Genetics and conservation of rare plants. New York: Oxford University Press: 87–98.

Kevan, P. G.; LaBerge, W. E. 1979. Demise and recovery of native pollinator populations through pesticide use and some economic implications. In: Caron, D. M., ed. Proceedings of the fourth international symposium on pollination. Misc. Publ. 1. College Park, MD: Maryland Agricultural Experiment Station: 489–508.

Lavigne, R.; Tepedino, V. J. 1976. Checklist of the insects in Wyoming. I. Hymenoptera. Agric. Exp. Stn. Resour. Jour. 106. Laramie, WY: University of Wyoming. 61 p.

Levin, D. A. 1984. Immigration in plants: an exercise in the subjunctive. In: Dirzo, R.; Sarukahn, J., eds. Perspectives on plant population ecology. Sunderland, MA: Sinauer Associates: 242–260.

Miliczky, E. R.; Osgood, E. A. 1979. The effects of spraying with Sevin-4-oil® on insect pollinators and pollination in a spruce–fir forest. Tech. Bull. 90. Orono, ME: University of Maine and Maine Agricultural Experiment Station: 1–21.

Peach, M. L.; Tepedino, V. J.; Alston, D. G.; Griswold, T. L. 1993. Insecticide treatments for rangeland grasshoppers: potential effects on the reproduction of *Pediocactus sileri*(Englem.) Benson (Cactaceae). In: R. Sivinski; Lightfoot, K., eds. Proceedings of the southwestern rare and endangered plant conference. Misc. Pub. 2. Santa Fe, NM: New Mexico Forestry and Resources Conservation Division: 309–319.

Plowright, R. C.; Thaler, G. R. 1979. The effect of biocides on forest pollination in New Brunswick. In: Caron, D. M., ed. Proceedings of the fourth international symposium on pollination. Misc. Publ. 1. College Park, MD: Maryland Agricultural Experiment Station: 483–487.

Rau, P. 1929. Experimental studies in the homing of carpenter and mining bees. Journal of Comparative Psychology 9: 35–70.

Rau, P. 1931. Additional experiments on the homing of carpenter-and mining-bees. Journal of Comparative Psychology 12: 257–261.

Roubik, D. W. 1989. Ecology and natural history of tropical bees. New York: Cambridge University Press. 514 p.

Slatkin, M. 1985. Rare alleles as indicators of gene flow. Evolution 39: 53–65.

Slatkin, M. 1993. Isolation by distance in equilibrium and non-equilibrium populations. Evolution 47: 264–279.

Slatkin, M; Barton, N. 1989. A comparison of three indirect methods for estimating average levels of gene flow. Evolution 43: 1349–1368.

Stephen, W. P.; Bohart, G. E.; Torchio, P. F. 1969. The biology and external morphology of bees. Corvallis, OR: Oregon State University and Oregon Agricultural Experiment Station. 140 p.

Tepedino, V. J. 1979. The importance of bees and other insect pollinators in maintaining floral species composition. Great Basin Naturalist Memoirs 3: 139–150.

U.S. Fish and Wildlife Service. 1993. Endangered and threatened wildlife and plants. U.S. Fish and Wildlife Service. 50 CFR 17.11 and 17.12. Washington, DC: U.S. Government Printing Office: 1993-342-447/94033. 40 p.

Wood, G. W. 1979. Recuperation of native bee populations in blueberry fields exposed to drift of fenitrothion from forest spray operations in New Brunswick. Journal of Economic Entomology 72: 36–39.

Reference Cited—Unpublished

Packer, J. S. 1970. The flight and foraging behavior of the alkali bee *Nomia melanderi* (Ckll.) and the alfalfa leafcutter bee *Megachile rotundata* (F.). Ph.D. dissertation. Logan, UT: Utah State University. 119 p.

III.6 Grasshopper Treatment Effects on Aquatic Communities

D. W. Beyers and L. C. McEwen

Concern about potential for adverse effects on endangered species from inadvertent exposure to insecticides was partially responsible for initiation of the Grasshopper Integrated Pest Management (GHIPM) Project. Investigation of effects of grasshopper control operations on aquatic communities was one aspect of the Project and had two major emphases.

The first emphasis was evaluation of the toxicity of carbaryl and malathion to two federally endangered fishes that inhabit rivers of the Colorado River Basin (the Colorado River and tributaries in Wyoming, Colorado, Utah, New Mexico, and Arizona). The second area of research involved environmental monitoring of the effects of operational grasshopper insecticide applications on aquatic invertebrates and fish in ponds and streams. Results of these studies provide information on potential effects of pesticide application practices and allow evaluation of adequacy of no-spray buffer zones around aquatic habitats.

Toxicity Testing With Endangered Fishes

The Colorado pikeminnow *(Ptychocheilus lucius)* and bonytail *(Gila elegans)* are large minnows historically found throughout the Colorado River Basin. Populations of both species have declined as a result of interactions with introduced fishes, construction of dams, and habitat modification. Young Colorado pikeminnow and bonytail occupy shallow, low-velocity, near-shore nursery habitats. These habitats have low rates of water exchange, and pesticides deposited in them may persist in sufficient concentration and duration for toxic effects to occur.

The timing of grasshopper control programs coincides with the presence of potentially sensitive early life stages of Colorado pikeminnow and bonytail in nursery habitats. But the infrequency and low application rate of pesticide use in Federal grasshopper control programs present a minor risk to these endangered fishes in comparison to other hazards, such as cropland chemicals, instream flow changes, and introduced (exotic) species. Nevertheless, data are needed on the IPM chemical effects.

Because of uncertainty in predicting the sensitivity of Colorado pikeminnow and bonytail to carbaryl and malathion, Beyers et al. (1994) estimated toxicity of these chemicals using methods recommended by U.S. Environmental Protection Agency and the American Society for Testing and Materials. The toxicity of technical carbaryl, Sevin® 4-Oil, and technical malathion was estimated by determining (1) 96-hour median lethal concentrations, and (2) concentrations that affected survival and growth in 32-day early life-stage tests (Beyers 1993, Beyers et al. 1991 and 1994).

One concern responsible for initiation of toxicological studies was that Colorado pikeminnow or bonytail might be supersensitive to carbaryl or malathion. To evaluate this possibility, we compared the sensitivity of Colorado pikeminnow and bonytail to other commonly studied fishes. We concluded that Colorado pikeminnow and bonytail were 2 to 10 times more sensitive to carbaryl than fathead minnow *(Pimephales promelas)* but were about as sensitive to malathion as fathead minnow (Beyers et al. 1994, Mayer and Ellersieck 1986). Some pesticide formulations are more toxic than their technical compounds; however, toxicity of Sevin 4-Oil (49 percent carbaryl) is approximately one-half that of technical carbaryl. No synergistic or antagonistic toxic effects due to formulation of carbaryl as Sevin 4-Oil were observed.

Results of standardized toxicity tests provided quantitative description of toxicant effects, but the tests did not simulate chemical exposure conditions likely to occur in the field. Therefore, we conducted studies of brain acetylcholinesterase (AChE) inhibition in order to estimate toxicant effects at a scale consistent with the duration of exposure and concentration range typically observed in the field. AChE activity was measured in Colorado pikeminnow after 24-hour *in vivo* exposure to technical carbaryl or malathion (Beyers and Sikoski 1994).

A comparison of the potency of the 2 toxicants showed that technical carbaryl was about 13 times more toxic than malathion to Colorado pikeminnow. Toxicant concentrations that significantly affected AChE activity were 15 times lower for carbaryl and 4 times lower for malathion than concentrations that affected growth or survival in 32-day early life-stage tests. These differences were attributed to development of physiological tolerance over the 32-day period used for early life-stage tests, and greater sensitivity of biochemical processes (AChE inhibition) compared to whole-organism responses (growth or survival).

Environmental Monitoring

Insecticides used to control grasshopper infestations pose a potential hazard to fish and invertebrates because, although no-spray buffer zones are observed around aquatic habitats, pesticide may be deposited by drift or mobilized from upland areas by runoff. We investigated effects of several aerial grasshopper control pesticide applications within the Little Missouri National Grasslands in western North Dakota (Beyers et al. 1995, Beyers and Myers 1996).

Environmental monitoring in aquatic habitats involved collection of water samples for pesticide analysis and study of sublethal and lethal effects on invertebrates and fish. In pond studies, we used enclosures called mesocosms to divide a portion of a pond into independent experimental units. Each mesocosm contained sediment, plants, and invertebrates that occurred naturally in the pond. We monitored survival of invertebrates within mesocosms for up to 4 days after pesticide application. *In situ* toxicity tests using naturally occurring invertebrates were also conducted with mesocosms.

The effects of pesticide application on river-dwelling organisms in the Little Missouri River were investigated on two separate occasions. Potential effects on aquatic invertebrates were investigated by quantifying daytime invertebrate drift. Normally, aquatic invertebrate drift in rivers is low. However, when pesticides are introduced, catastrophic drift may occur as invertebrates attempt to avoid toxicant exposure or suffer toxic effects (Wiederholm 1984). Sublethal effects on fish in the Little Missouri River were evaluated by studying fish-brain AChE inhibition. AChE activity of flathead chub *(Platygobio gracilis)* collected from control and treatment sites before and after pesticide application was measured.

Results of monitoring showed that when the standard 500-ft (152-m) no-spray buffer was employed, trace amounts of pesticide were always detected in aquatic habitats. The amount of deposition was dependent on the size of the aquatic habitat; smaller ponds had higher pesticide concentrations. Detection of trace amounts of pesticides does not necessarily result in biological effects on aquatic organisms.

We intensively studied six ponds but found evidence of direct mortality of pond-dwelling organisms in only one. On this occasion, a 0.6-acre (0.23-ha) pond containing abundant amphipods was monitored during an application of Sevin 4-Oil. All amphipods in treatment enclosures died within 24 hours of pesticide application. Subsequent collections confirmed that the amphipod population in the pond had declined. Amphipods are known to be extremely sensitive to carbaryl and malathion (Mayer and Ellersieck 1986). Other taxa in the pond appeared to be unaffected by the application.

Studies in the Little Missouri River during a drought year (1991), when discharge and the dilution potential of the river was low, detected an increase in invertebrate drift during the first 3 hours after pesticide application (Beyers et al. 1995). This increase was primarily composed of Ephemeroptera, especially Heptageniidae. There was no change in drift at the reference site. Subsequent sampling during the day of pesticide application showed that the increase in invertebrate drift was transient and undetectable after 3 hours.

The biological significance of increased invertebrate drift due to pesticide application is uncertain but probably of minimal consequence. The increase in invertebrate drift was mostly due to Ephemeroptera; other taxa were unaffected. Because a relatively small portion of the Little Missouri River was within the spray block (3.2 river-miles or 5.2 river-km), mortality was probably compensated by recolonization from unaffected organisms living in the substrate or upstream. Thus only a portion of the invertebrate community may have been affected, and the likelihood of rapid recovery of affected populations was high. Analyses of brain AChE activity in flathead chub showed that fish were not affected by the pesticide application. Similar monitoring studies conducted during a year when precipitation was above average (1993) did not detect any increase in aquatic invertebrate drift or effects on fish (Beyers et al. 1995). The overall conclusion was that these grasshopper control operations had no biologically significant affect on aquatic resources.

A factor that may reduce the potential for toxic effects to aquatic organisms is the natural degradation of carbaryl and malathion. Both pesticides hydrolyze (decompose chemically) rapidly in waters with pH >7 (Beyers and

Myers 1996). All aquatic habitats monitored in North Dakota had pH's greater than 7. Although the amount of pesticide deposited in aquatic habitats may be potentially toxic to some aquatic life, the short duration of the exposure can reduce or eliminate toxic effects.

Our investigations were designed to detect AChE inhibition or invertebrate mortality within 96 hours of pesticide application. If toxic effects were manifested over a longer time scale it is unlikely that effects would have been detected by our investigations. Toxicity endpoints other than death of aquatic organisms (such as swimming ability, avoidance of predators, feeding behavior, and reproductive effects) also are receiving attention by others in the field of aquatic ecotoxicology (Nimmo and McEwen 1994).

A Note on Quality Assurance for Pesticide Monitoring

One of the reasons why carbaryl and malathion are used to control grasshopper infestations is that they degrade relatively rapidly in the environment. Short persistence assures less potential for nontarget effects; however, these qualities complicate sampling for pesticide analysis because, if precautions are not taken, degradation may continue to occur after a sample has been collected and pesticide concentration estimates will be in error.

An important aspect of quality assurance (QA) that can be used to guard against this eventuality is fortification (spiking with measured pesticide amounts) of similar environmental samples. Prior to pesticide application, samples for fortification should be collected at the same localities where pesticide monitoring samples will be collected. A known amount (for example, 1 mL) of a fortification standard should be added to each QA sample. To prevent investigator bias, QA samples should not be identified any differently than posttreatment monitoring samples. QA samples should be handled and submitted for chemical analysis along with other monitoring samples. In general, QA samples should be fortified to approximately 10 times the detection limit reported by the analytical laboratory and the number of QA samples should be about 10 percent of total number of samples submitted for analysis. If only a few monitoring samples are being collected (fewer than 10), then at least 2 QA samples should be submitted. Fortification standards should be obtained from the laboratory that will be conducting the analytical work (see Chapter III.9). When reporting results of pesticide monitoring, percent recovery from fortified samples also should be reported. The importance of including QA samples cannot be overstated: they provide the only method for judging accuracy of reported results.

References Cited

Beyers, D. W. 1993. Toxicity of carbaryl and malathion to Colorado squawfish and bonytail. Ph.D. dissertation. Fort Collins, CO: Colorado State University. 92 p.

Beyers, D. W.; Myers, O. B. 1996. Use of meta-analysis to predict degradation of carbaryl and malathion in freshwater for exposure assessment. Human and Ecological Risk Assessment 2: 366–380.

Beyers, D. W.; Sikoski, P. J. 1994. Acetylcholinesterase inhibition in federally endangered Colorado squawfish exposed to carbaryl and malathion. Environmental Toxicology and Chemistry 13: 935–939.

Beyers, D. W.; Carlson, C. A.; Tessari, J. D. 1991. Solid-phase extraction of carbaryl and malathion from pond and well water. Environmental Toxicology and Chemistry 10: 1425–1429.

Beyers, D. W.; Keefe, T. J.; Carlson, C. A. 1994. Toxicity of carbaryl and malathion to two federally endangered fishes, as estimated by regression and ANOVA. Environmental Toxicology and Chemistry 13: 101–107.

Beyers, D. W.; Farmer, M. S.; Sikoski, P. J. 1995. Effects of rangeland aerial applications of Sevin 4-Oil on fish and aquatic invertebrate drift in the Little Missouri River, North Dakota. Archives of Environmental Contamination and Toxicology 28: 27–34.

Mayer, F. L., Jr.; Ellersieck, M. R. 1986. Manual of acute toxicity: interpretation and data base for 410 chemicals and 66 species of freshwater animals. Resour. Publ. 160. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service. 506 p.

Nimmo, D. W.; McEwen, L. C. 1994. Pesticides. In: Calow, P., ed. Handbook of ecotoxicology, vol. 2. London, Edinburgh, Boston: Blackwell Scientific Publications: 155–203.

Wiederholm, T. 1984. Responses of aquatic insects to environmental pollution. In: Resh, V. H.; Rosenberg, D. M., eds. The ecology of aquatic insects. New York: Praeger Publishers: 508–557.

III.7 Bioindicator Species for Evaluating Potential Effects of Pesticides on Threatened and Endangered Wildlife

L. C. McEwen, B. E. Petersen, and C. M. Althouse

Monitoring pesticide applications for possible effects on wildlife is an integral part of pesticide registration and regulation and of a successful grasshopper integrated pest management (GHIPM) system. During grasshopper outbreaks, U.S. Department of Agriculture cooperative grasshopper control programs have treated as much as 13.1 million acres (5.3 million ha) of rangeland in a single season (U.S. Department of Agriculture, Animal and Plant Health Inspection Service 1987).

Large numbers of insectivorous birds may inhabit, or congregate in, areas where these insecticide applications are made. One grasshopper egg bed found in Otero County, CO, encompassing 2 acres (0.8 ha), was populated by "about 200 western horned larks and lark buntings," which were seen feeding heavily on the grasshopper nymphs (Wakeland 1958). An effective GHIPM program should retain the natural controls on grasshoppers and not disrupt the rangeland ecosystem, including threatened and endangered species.

Wiens and Dyer (1975) reported breeding-season bird densities averaging approximately 0.8 to 1.3 birds/acre (1.9 to 3.3 birds/ha) on rangeland. Johnson et al. (1980) summarized avian densities for grassland–sagebrush habitats as averaging 1.2 to 5.0 breeding birds/ha. Therefore, large numbers of birds and other wild vertebrates can be exposed to a chemical during a single pesticide application (McEwen 1987). In areas not monitored during an application, mortality, and particularly sublethal effects, caused by pesticides can be overlooked because mortality "usually affects only part of the fauna, is scattered in space and time, and generally occurs where there is no biologist to record it" (Stickel 1975).

Toxicity evaluation has employed the use of white rat species in a laboratory setting utilizing test animals that are common species, easily bred, maintained, and handled. Controlled tests are pertinent for determining baseline data and comparing relative toxicity of chemicals. However, to understand pesticide effects in the natural environment, all the intricate interactions of cover, weather, food, exposure routes, and animal behavior, must be considered. Toxicity tests in the laboratory can only predict ecotoxicity in the field setting within broad limits.

An intermediate step between laboratory and field investigations is the use of caged or penned vertebrates located within an application block as used by Kreitzer and Spann (1968). However, it was found that the cage-in-field method resulted in less exposure to the pesticide than free-ranging wildlife received and actually protected the experimental animals from possible predation related to sublethal effects (Heinz et al. 1979).

Sublethal effects can be observed in the controlled environment of laboratory investigations, and researchers often surmise that "a sublethal effect seen in the laboratory would also occur in the field and that this effect would result in mortality or reproductive problems" (Heinz 1989). These effects can also be misleading or overlooked. For example, Grue et al. (1982) found that free-living starlings differed from captive birds by losing weight after dosing with dicrotophos, an organophosphate (OP) insecticide. Field investigations are a necessary step in evaluating the overall effects of large-scale pesticide applications.

It has been recognized that data on effects of OP's and other classes of pesticides are incomplete (Grue et al. 1983, Kirk et al. 1996). The Avian Effects Dialogue Group (1994) set forth some recommendations for more effective techniques in gathering data. Several issues of concern were studies on focal avian species, study sites, carcass searching, population changes, modeling, use of radio telemetry, and dissemination of information.

Species of critical concern are usually unavailable for any hands-on laboratory or field toxicity studies, thus making the need for surrogate species a necessity. Lower and Kendall (1990) suggested some criteria for selecting a sentinel species (one in which effects may be interpreted as indicators of similar disturbances in other species) when evaluating synthetic compounds, such as pesticides in the field. This approach has several limitations.

For example, can the toxicity of a chemical to a chicken, duck, or quail predict toxic effects on a falcon or eagle? How do the differences in a species' physiology, food, habitats, and ecology affect the animal's exposure and reaction to the chemical? When threatened or endangered (T and E) species may be at risk, they of course, cannot be collected for chemical analysis, pathology examination, or food-habits study. Thus, the next best approach is to estimate potential effects on T and E species by study of closely related sentinel species.

The American kestrel *(Falco sparverius)* has been shown to be more sensitive to anticholinesterase insecticides than other avian species (such as quail and ducks) used to establish toxicity (Rattner and Franson 1984, Wiemeyer and Sparling 1991). Consequently, the kestrel is a conservative bioindicator of possible effects on the related peregrine falcon *(Falco peregrinus).*

Our environmental monitoring team's studies have utilized the American kestrel and killdeer *(Charadrius vociferus),* as surrogates for other Falconiformes and Charadriidae, such as the peregrine falcon and mountain plover *(Charadrius montanus),* respectively. Kestrels and killdeer are representative of their genera, are widely distributed, and are found in much greater numbers than their endangered relatives.

The American and European kestrels have been utilized in toxicology studies for many years (Wiemeyer and Lincer 1987). Studies of the American kestrel, the smallest and most abundant falcon throughout North America, have progressed from laboratory toxicity tests to field ecotoxicology investigations over the past 20 years. Since kestrels are commonly present on rangelands where grasshopper outbreaks occur, they are excellent subjects for examining direct and indirect effects of control programs. Kestrel use of nest boxes (fig. III.7–1) and tolerance of disturbance and observers makes it possible to investigate all stages of their life cycle. Henny et al. (1983) examined productivity of free-ranging kestrels using nest boxes beginning in 1978 for investigating the adverse effects of the pesticide heptachlor in Oregon's Columbia River Basin.

Figure III.7–1—Kestrel nest box used on rangeland. Access to the eggs and nestlings is through a hinged side of the box. Field crews can check nests periodically to determine egg hatchability, growth measurements, and survival of young, and to affix leg bands and attach transmitters. (Photo by L. C. McEwen of Colorado State University; reproduced by permission.)

On rangelands, population densities of American kestrels may be restricted by the lack of natural tree cavities for nesting sites. Investigation of pesticide effects could be difficult to document because of small sample sizes of kestrels, but nesting populations can be increased by adding artificial nest box structures. Frocke (1983) summarized the use of nest boxes in avian management and research; cavity-nesting species have exhibited a readiness to use, and possibly a preference for, nest boxes over natural cavities. Kestrels are very adaptable and will easily accept the use of human-made nest boxes.

Kestrels favor open-space sites for hunting, so establishing new nest sites in these open areas for experimental purposes can be effective. Although Loftin (1992) found in Florida that nest boxes placed in pastures or areas away from known kestrel use were ineffective in increasing American kestrel populations, we did not find this to be true. We had >50 percent use of all nest boxes in six different geographic locations from Colorado to Alaska. However, in some areas, it took 2–3 years to reach maximum use of boxes. (Plans and directions for construction and placement of nest boxes are given in chapter I.11 of this Handbook.)

Seven years of production data have been compiled on nesting American kestrels during the Grasshopper Integrated Pest Management (GHIPM) Project. Approximately 560 nest boxes were in place by the sixth year among 6 locations: the 2 GHIPM demonstration areas in Idaho and North Dakota, Alaska, Wyoming, and 2 parts of Colorado—the northwestern section and in the Front Range (fig. III.7–2). Data on clutch size, hatchability, and numbers of nestlings fledged were collected annually (table III.7–1).

Productivity is presented as baseline data for each location and compared between years. Mean clutch sizes did not vary among locations, but yearly differences were observed (*P* < 0.05). Alaskan kestrels surpassed birds

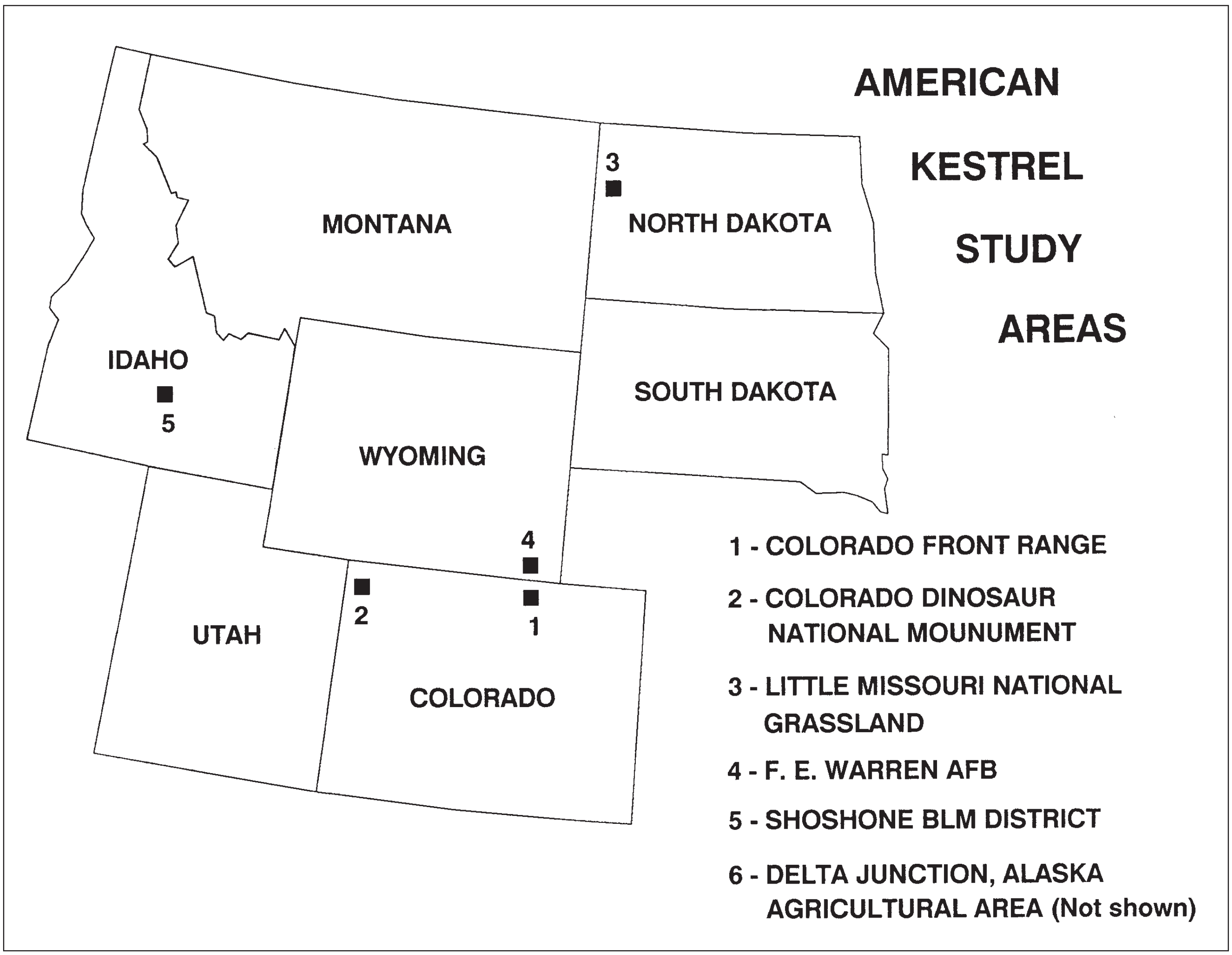

Figure III.7–2—Locations of kestrel study areas where >500 nest boxes have been placed (total of all areas). Key: 1 = Colorado, Front Range; 2 = Colorado, Dinosaur National Monument; 3 = Little Missouri National Grasslands; 4 = F. E. Warren Air Force Base; 5 = Bureau of Land Management's Shoshone District. (A sixth location, an agricultural area in Delta Junction, AK, is not shown.)

Table III.7–1—Variation in nesting productivity of American kestrels in the GHIPM demonstration areas and other treatment and reference areas during 1988–94

Location and years	Mean no. of nests/yr	% of nests hatched[1]	% of nests fledged[2]	Mean no. fledged per nest attempt
Alaska				
1990–93	33	85–97	82–97	3.5–4.3
Colorado, Front Range				
1988–94	26	61–88	55–81	2.0–2.9
Colorado, northwestern				
1988–94	24	81–89	79–84	2.9–3.1
Idaho				
1988–93	62	60–90	48–81	1.8–3.5
North Dakota				
1988–94	83	58–88	50–70	1.5–3.0
Wyoming				
1989–94	12	31–100	19–100	0.6–3.8

[1] Hatched nest: ≥ 1 egg hatched.
[2] Fledged nest: ≥ 1 young fledged.

from all other areas sampled in mean number of eggs hatched and young fledged in 1990 through 1993, but the differences were not statistically significant ($P > 0.05$).

Lower kestrel productivity in Idaho and North Dakota coincided with drought years and with the one extreme high-precipitation year in the Dakotas but otherwise was similar for most years (table III.7–1). The results illustrate the variability in kestrel nesting success due to natural factors and emphasize the importance of having concurrent untreated nest boxes for observation when investigating possible pesticide effects on nests in sprayed areas. Comparison of comparable untreated nests with sprayed nests over the same time period, is necessary to differentiate effects of weather, predation on nestlings by great horned owls *(Bubo virginianus),* and other natural factors from pesticide treatment effects.

In 1990–94, a limited number of nest boxes in several locations, excepting Idaho, were used to study sublethal effects on kestrel nestlings and fledglings of (1) *Beauveria bassiana,* a fungus bioinsecticide; (2) carbaryl, a carbamate (sprays and bran-bait treatments); (3) malathion, an organophosphate; and (4) diflubenzuron (Dimilin®), an insect growth regulator. These results are presented in separate sections.

Field Applications

A carbaryl bran-bait treatment was examined at the Delta Agricultural Project in Alaska where five kestrel nest sites with heavy grasshopper infestation were selected for study of the effects of carbaryl bait. At the time of application, nestlings were approximately 18–22 days of age. Three of these nests had 2 percent carbaryl bran-bait applied at approximately 2.2 lb/acre on 40 acres (16.2 ha) adjacent to the nest box entrances, and 2 nests were left untreated. No adverse effect was noted on the treated nests, and all kestrel nestlings fledged normally. It was also found that numbers of breeding birds in North Dakota on line transects before and after application did not differ when controlling grasshoppers with carbaryl bait (George et al. 1992).

Possible effects on killdeer from spray applications of two formulations of Sevin® 4-Oil (20 or 16 fl oz/acre, with each containing 4 fl oz of diesel oil; active ingredient [AI] of carbaryl was 8 and 6.4 fl oz/acre or 0.56 and 0.45 kg/ha, respectively) were investigated in North Dakota during 1992. Brain AChE activities were monitored at 2, 8, and 21 days after applications and found not to differ from normal (Fair et al. 1995). Whole body carbaryl residues were low (averaging <0.1 to 1.4 p/m [parts per million]) but significantly ($P < 0.05$) greater for birds collected from the sprayed areas compared to birds from unsprayed surrounding locations. No toxic signs were observed in any killdeer. On the treated areas, birds captured invertebrate prey at rates significantly higher than on reference areas at 2 and 8 days after spraying (Fair 1993) presumably due to the availability of dying insects.

Acute Oral Dosing Treatments and Procedures

Growth, nestling and fledgling survivability, and postfledging movements of young wild kestrels were measured in the field after exposure to an acute sublethal oral dose of one of the following standard or experimental IPM materials: *Beauveria bassiana*, diflubenzuron, carbaryl, malathion, or their formulation carriers (diesel or corn oil). A minimum of four young per brood were used in these studies. The remaining nestling(s), if any, in each box served to maintain a normal brood size and provided an untreated comparison to the dosed birds. Their ages varied from 8 to 16 days when nestlings were randomly selected and given a single dose of one of the following: corn oil, pesticide formulation, the petroleum-based oil used in the formulation (carrier oil or #2 diesel fuel), or the technical material. Behavior and growth data were collected every 4 days following dosing.

Surviving test nestlings were fitted with transmitters at 26–31 days of age (fig. III.7–3). After fledging, all birds were located daily or every other day until transmitters failed or young moved too far from the nest box area to be located.

Beauveria bassiana Sublethal Test

This investigation was conducted in the short-grass prairies of north-central Colorado during 1992. Thirteen nest boxes containing 55 young were tested (table III.7–2). Two of the nests were given challenge dosages of 5 µL

Figure III.7–3—Young kestrel with small transmitter attached for the study of postfledging behavior, movements, and survival. (Photo by B. E. Petersen of Colorado State University; reproduced by permission.)

Table III.7–2—Survival of American kestrel nestlings dosed with *Beauveria bassiana* formulation, carrier oil, corn oil, or untreated in north-central Colorado, May–August 1992

	Beauveria formulation[1]	Carrier oil[2]	Corn oil[2]	Untreated control
No. nestlings dosed	14	13	13	15
No. nestlings survived	11	12	13	15
No. fledglings with radios	11	12	13	2
No. fledglings survived	10	10	12	2

[1] Contains formulation oil and *Beauveria bassiana* spores. Dosage was based on 500,000 spores/µL and 1 µL/g of body weight.

[2] Dosages based on 1 µL/g of body weight.

(microliters)/gram of body weight for the formulation and carrier oil; for the main test, broods were dosed at 1 µL/gram of body weight. No statistical significance was detected in either growth rates or behavior data among treated and untreated groups ($P > 0.05$). Transmitters were attached to 38 kestrels. Data were collected on survival and movements of 28 of those birds (10 radio attachments failed). No detectable differences in survival or movements were found among treated and untreated kestrels.

Seven treated fledglings, ages 31–42 days, were collected for examination. Two additional fledglings were found dead and also the remains of one eaten by predators. Necropsies were performed on all collected birds at the Colorado Veterinary Teaching Hospital; no visible gross pathology was detected.

Diflubenzuron Sublethal Test

This investigation was conducted in north-central Colorado during 1993–94. Forty nest boxes containing 170 young were used (table III.7-3). Two of the nests were given preliminary challenge dosages of 64 mg/kg of body weight of technical diflubenzuron (Dimilin) to estimate toxicity, if any. (In English measure, this is the equivalent of 0.0009 oz diflubenzuron per pound of body weight). All following dosages will be given in metric units as used in toxicology. Kestrel broods in the main study were dosed at 10.2 mg/kg.

No statistical differences were detected in nestling growth rates, behavior data, or survival among treated and untreated birds ($P > 0.05$). Although no differences were found in nestlings, possible effects on fledgling survival were seen the first year. Transmitters were attached to 42 fledgling kestrels. During 1993 approximately half the fledgling kestrels dosed with diflubenzuron formulation died or were lost, warranting a second year of research. In 1994, however, more than 70 percent of the 43 kestrels fitted with transmitters survived, and no differences were observed between treated and control fledglings.

Several treated fledglings, ages 27 to 45 days, were found dead due to predation or other causes. Necropsies were performed on all the dead birds, and no gross pathology was detected.

Carbaryl Sublethal Test

American kestrel nestlings in nest boxes on the North Dakota GHIPM demonstration area were administered sublethal acute oral doses of Sevin 4-Oil formulation in 1992 to determine effects on growth and postfledging survival. Two 10-day-old nestlings were given 200 mg/kg body weight of Sevin 4-Oil (40.5 percent carbaryl or 81 mg/kg AI) to establish a lethal dosage. Brain acetylcholinesterase (AChE) activity was depressed 80 percent at death in 27–35 minutes. Four additional nestlings all survived Sevin 4-Oil dosages of 30–100 mg/kg.

Table III.7–3—American kestrel nestling and fledgling survival after dosing with technical or formulation diflubenzuron, diesel oil #2, corn oil, or untreated in north-central Colorado during 1993–94

	Diflubenzuron		Diesel	Corn	No
	Technical	Formulation	oil #2	oil	treatment
No. nestlings dosed	[1]40	40	40	39	11
No. nestlings survived	32	33	34	32	10
No. fledglings with radios	25	27	27	6	—
No. fledglings survived	22	19	21	3	—

[1] One bird dosed with technical diflubenzuron was collected prior to radio transmitter fitting.

Sublethal dosages then were given to 32 nestlings (8 to 14 days old). Sixteen were dosed at 15 mg/kg and 16 at 30 mg/kg with Sevin 4-Oil. Sixteen additional nestlings were given corn oil at 2 µL/g of body weight as untreated controls subjected to the same handling procedures. Blood samples were collected from the nestlings for analysis of plasma cholinesterase activity at 1 hour, 24 hours, and 7 to 14 days after dosing. Radios were placed on 30 of the nestlings for study of postfledging movements and survival. Twenty-one of the nestlings and fledglings were collected at 10 to 38 days after treatment for brain AChE activity measurements, carcass residue analysis, and necropsy. Carbaryl residues were no longer detectable in the carcasses, but three had 0.08–0.15 p/m in their gastrointestinal tracts (analyzed separately). No gross pathology was found.

None of the 21 birds had significant inhibition of brain AChE activity or any signs of gross pathology. The lack of brain AChE inhibition was not unexpected because of the sublethal dosage levels and the rapid reversibility of carbaryl inhibition. Blood plasma samples showed mild AChE inhibition at 1 hour after treatment (averages = 4 percent at 15 mg/kg and 12 percent at 30 mg/kg). Recovery from the low degree of plasma AChE inhibition was evident in all carbaryl-dosed nestlings by 24 hours after treatment.

Malathion Sublethal Test

American kestrel nestlings in North Dakota were administered sublethal acute oral malathion dosages in 1993 and 1994. To establish the sublethal treatment dosages, it was first necessary to determine the acute oral lethal levels by conducting preliminary range-finding toxicity tests. Based on reported malathion toxicity to other avian species, dosages ranging from 49 to 500 mg/kg were administered to seven nestlings, and all dosages were found to be lethal. In further tests, it was determined that lethal malathion dosages began at 20 to 40 mg/kg (Taira 1994). These results indicated that young kestrels are much more sensitive to malathion toxicity than many other bird species for which LD_{50}'s (lethal dose to 50 percent of the birds) range from >100 to >400 mg/kg (Smith 1987). Part of this sensitivity may be age related, but scientists do not know the acute oral LD_{50} of malathion for adult American kestrels.

Young birds in 17 nest boxes were given malathion at 1 of 2 dosage levels: 5 or 20 mg/kg. An equal number were given corn oil or left untreated. Posttreatment blood samples were taken for plasma AChE and butyrylcholinesterase (BChE) assay from each bird at 1 hour, 24 hours, and between 7 and 14 days after treatment. At the 20 mg/kg dosage, both AChE and BChE were severely inhibited (77.1 and 71.6 percent respectively) at 1 hour posttreatment (table III.7-4). AChE activity was still inhibited 60.3 percent at 24 hours. BChE recovered more quickly, showing 21.9 percent inhibition at 24 hours. Nestlings dosed with 5 mg/kg were not as strongly affected but had plasma AChE inhibition of 45.4 percent and BChE inhibition of 60.8 percent at 1 hour. These results support the conclusion from the range-finding tests that young kestrels are more sensitive to malathion than many other avian species (Taira 1994).

Nestlings that were casualties in the malathion range-finding tests were analyzed for carcass residue concentrations. Whole-carcass residues ranged from 0.38 p/m in the lowest-dosed bird (49 mg/kg) to 46.5 p/m in the highest-dosed nestling (500 mg/kg). Gastrointestinal tracts (including contents) were analyzed separately, and residues varied from 12.1 p/m to 4,860 p/m corresponding to dosage levels. Only 6 of the sublethally dosed nestlings/fledglings were recovered for analysis. Residues were not detectable except in one carcass, which contained 0.21 p/m of malathion.

Summary and Conclusions

Field studies of bioindicator species are a useful approach for estimating potential ecotoxicological effects of pest control operations on threatened or endangered (T and E) species or other wildlife species of special concern. Species selected as bioindicators should be widely distributed and relatively abundant in the habitat types subjected to pest controls. Species closely related to T and E species also may be considered “surrogates” for those species and for others of concern.

In our environmental monitoring studies, we have investigated effects on American kestrels as bioindicators for peregrine falcons (and other small raptors) and effects on killdeer as bioindicators for mountain plovers. Our data on total bird populations in treated and untreated rangeland sites also could be examined in retrospect if questions arise concerning other species such as long-billed curlews, burrowing owls, ferruginous hawks, loggerhead shrikes, or rare species of sparrows.

From our GHIPM work, these two conclusions can be drawn:

(1) Young kestrels are more vulnerable to toxicity of malathion and anticholinesterase pesticides than many other avian species. Therefore, nonspray buffer zones

Table III.7–4—Mean percentage of plasma cholinesterase (ChE) activity in malathion-dosed kestrel nestlings compared to control ChE activity

	Dosages					
	5 mg/kg			20 mg/kg		
Posttreatment collection time	ChE	Total AChE[1]	BChE[2]	ChE	Total AChE	BChE
1 hour	51.1	54.6	39.2	24.2	22.9	28.4
24 hours	74.8	73.8	80.5	46.4	39.7	78.1
7 days	94.0	94.5	91.6	89.0	86.9	101.8
14 days	98.3	100.8	88.2	94.6	97.0	84.7

[1] Acetylcholinesterase.

[2] Butyrylcholinesterase.

around active nests of the closely related peregrine falcon should always be observed when liquid pesticide formulations are applied. However, bait formulations of IPM chemicals and biologicals are safe and pose no significant hazard even if used in the immediate vicinity of the nests. Acute dosages of diflubenzuron or *Beauveria bassiana* formulations indicate very low direct toxicity to young kestrels. These materials would have no direct effects on nontarget terrestrial wildlife but might reduce the insect food base in some cases. These findings should also apply to other nesting raptors on rangeland.

(2) Studies of Sevin 4-Oil grasshopper sprays (16 or 20 fl oz/acre) indicated little or no effect on killdeer (Fair et al. 1995). Cholinesterase activity was not significantly inhibited, whole-body carbaryl residues were low (<0.1 to 1.4 p/m), and food-habits studies showed that the birds maintained adequate diets. No gross pathology was found on necropsy of the killdeer. Whole body lipids were measured as an indicator of body condition and did not differ between killdeer from sprayed and unsprayed sites.

These results indicate that Sevin 4-Oil applications at 20 fl oz/acre (0.56 kg/ha carbaryl AI) or lower pose little hazard to the closely related mountain plover, a Category 1 species that may be listed in the future as endangered. However, areas known to be in the immediate vicinity of mountain plover nests should be excluded from spray applications because of the variation in individual bird response to synthetic chemical compounds. Bait formulations would be the least hazardous method of grasshopper control in mountain plover habitat.

References Cited

Avian Effects Dialogue Group. 1994. Final report of the AEDG, 1988-1993. Washington, DC: RESOLVE, Center for Environmental Dispute Resolution. 156 p.

Fair, J. M. 1993. The effects of carbaryl grasshopper control on nesting killdeer *(Charadrius vociferus).* M.S. thesis. Fort Collins, CO: Colorado State University. 41 p. + appendices.

Fair, J. M.; Kennedy, P. L.; McEwen, L. C. 1995. Effects of carbaryl grasshopper control on nesting killdeer in North Dakota. Environmental Toxicology and Chemistry 14: 881–890.

Frocke, J. B. 1983. The role of nestboxes in bird research and management. In: Snag habitat management: proceedings of the symposium. Gen. Tech. Rep. RM-99. Ft. Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station: 10–13.

George, T. L.; McEwen, L. C.; Fowler, A. 1992. Effects of a carbaryl bait treatment on nontarget wildlife. Environmental Entomology 21: 1239–1247.

Grue, C. E.; Powell, G.V.N.; McChesney, M. J. 1982. Care of nestlings by wild starlings exposed to an organophosphate pesticide. Journal of Applied Ecology 19: 327–335.

Grue, C. E.; Fleming, W. J.; Busby, D. G.; Hill, E. F. 1983. Assessing hazards of organophosphate pesticides to wildlife. Transactions of North American Wildlife and Natural Resources Conference 48: 200–220.

Heinz, G. H. 1989. How lethal are sublethal effects? Environmental Toxicology and Chemistry 8: 463–464.

Heinz, G. H.; Hill, E. F.; Stickel, W. H.; Stickel, L. F. 1979. Environmental contaminant studies by the Patuxent wildlife research center. In: Kenega, E. E., ed. Avian and mammalian wildlife toxicology. Spec. Tech. Publ. 693. Philadelphia: American Society for Testing and Materials: 9–35.

Henny, C. J.; Blus, L. J.; Stafford, C. J. 1983. Effects of heptachlor on American kestrels in the Columbia Basin, Oregon. Journal of Wildlife Management 47: 1080–1087.

Johnson, R. R.; Haight, L. T.; Riffey, M. M.; Simpson, J. M. 1980. Brushland/steppe bird populations. In: Workshop proceedings: management of western forests and grasslands for nongame birds. Gen. Tech. Rep. INT-86. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station: 98–113.

Kirk, D. A.; Evenden, M. D.; Mineau, P. 1996. Past and current attempts to evaluate the role of birds as predators of insect pests in temperate agriculture. Current Ornithology 13: 175–269.

Kreitzer, J. F.; Spann, J. W. 1968. Mortality among bobwhites confined to a heptachlor contaminated environment. Journal of Wildlife Management 32: 874–878.

Loftin, R. W. 1992. Use of nest boxes by the Florida kestrel. Florida Field Naturalist 20: 57–88.

Lower, W. R.; Kendall, R. J. 1990. Sentinel species and sentinel bioassay. In: McCarthy, J. F.; Shugart, L. R., eds. Biomarkers of environmental contamination. Boca Raton, FL: CRC Press: 309–332.

McEwen, L. C. 1987. Function of insectivorous birds in a shortgrass IPM system. In: Capinera, J. L., ed. Integrated pest management on rangeland: a shortgrass prairie perspective. Boulder, CO: Westview Press: 324–333.

Rattner, B. A.; Franson, J. C. 1984. Methyl parathion and fenvalevate toxicity in American kestrels: acute physiological responses and effects of cold. Canadian Journal of Physiology and Pharmacology 62: 787–792.

Smith, G. J. 1987. Pesticide use and toxicology in relation to wildlife: organophosphorous and carbamate compounds. Resour. Publ. 170. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service. 171 p.

Stickel, W. H. 1975. Some effects of pollutants in terrestrial ecosystems. In: McIntyre, A. D.; Mills, C. F., eds. Ecological toxicology research. New York: Plenum Publishing Corp.: 25–74.

Taira, T. 1994. Blood analysis of American kestrel and golden eagle nestlings exposed to malathion or carbaryl. M.S. thesis. Ft. Collins, CO: Colorado State University. 80 p.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 1987. Rangeland Grasshopper Cooperative Management Program: final environmental impact statement. Washington, DC: U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 221 p.

Wakeland, C. 1958. The high plains grasshopper. Tech. Bull. 1167. Washington, DC: U.S. Department of Agriculture, Agricultural Research Service. 168 p.

Wiemeyer, S. N.; Lincer, J. L. 1987. The use of kestrels in toxicology. In: Bird, D. M.; Bowman, R., eds. The ancestral kestrel. Raptor Res. Rep. 6. Ste. Anne de Bellevue, PQ: Raptor Research Foundation, Inc., and Macdonald Raptor Research Centre of McGill University: 165–178.

Wiemeyer, S. N.; Sparling, D. W. 1991. Acute toxicity of four anticholinesterase insecticides to American kestrels, eastern screech-owls and northern bobwhites. Environmental Toxicology and Chemistry 10: 1139–1148.

Wiens, J. A.; Dyer, M. I. 1975. Rangeland avifaunas: their composition, energetics, and role in the ecosystem. In: Smith, D. R., coord. Proceedings of the symposium on management of forest and range habitats for nongame birds; 6–9 May 1975, Tucson, AZ. Gen. Tech. Rep. WO-1. Washington, D.C.: U.S. Department of Agriculture, Forest Service: 146–182.

III.8 Buffer Zones: Their Purpose and Significance in Grasshopper Control Programs

L. K. Winks, L. C. McEwen, R. N. Foster, Mike W. Sampson, Michael Green, and V. J. Tepedino

NOTE: Acephate is no longer approved by EPA for rangeland grasshopper control.

A buffer zone is a distance or space around an environmentally sensitive area that acts as a deterrent to harm and/or disturbance of that area and its plant and animal life. For Federal cooperative grasshopper control or suppression operations, buffer zones are strips or areas of land left untreated and free of grasshopper suppression chemicals or materials.

Such zones, also called buffers, are pesticide-free areas established to protect (1) species listed or proposed as threatened or endangered (T and E) under the Federal Endangered Species Act, (2) designated or proposed critical habitats of T and E species, (3) aquatic sites (water or wetlands) of all types, and (4) other areas such as residences, parks, campgrounds, schools, cropland, apiaries and insectaries, and habitat for other sensitive species. Before any lands are treated in large-scale U.S. Department of Agriculture (USDA)-sponsored cooperative grasshopper management programs, land management agencies meet with USDA's Animal and Plant Health Inspection Service (APHIS) to consider all aspects of an operational plan to protect the T and E species and sensitive sites in the proposed treatment area.

Land-management agencies typically include the U.S. Department of the Interior's Bureau of Land Management and U.S. Fish and Wildlife Service (FWS) and USDA's Forest Service. An APHIS-prepared biological assessment opens the required consultations, and agencies discuss and negotiate buffer-zone requirements until agreement is reached among APHIS and the affected land-management agencies. At times, discussions and negotiations also involve State agencies.

The agencies determine buffer-zone specifics using existing Federal guidelines, the most recent information, and the best judgment of their personnel. The written agreement reached is expressed in detail in the FWS biological opinion for the site-specific environmental assessment. In practice, optimal treatment of a control block also depends on the experience of the project manager and the skill and experience of the spray pilots or ground applicators and on their observance of buffer boundaries and wind and weather conditions.

Generalized Buffer Zone Requirements

There are two general types of insecticide used for grasshopper control: liquid ultralow-volume (ULV) chemical sprays and insecticide-impregnated wheat-bran flakes. Requirements for use are more stringent for liquid ULV sprays than for bait application because ULV sprays are less selective in action, are more prone to drift, and contain more active ingredient (AI).

For treating grasshoppers in large-scale rangeland programs, APHIS not only follows chemical labeling recommendations but at times adds more restrictions based on environmental concerns. APHIS and other agencies base their current recommendations and mitigation (softening of effects) on guidelines contained in the Rangeland Grasshopper Cooperative Management Program and the Final Environmental Impact Statement (EIS) (U.S. Department of Agriculture, Animal and Plant Health Inspection Service 1987). APHIS also relies on changes agreed to by the FWS and content of the biological opinion. In addition, APHIS considers information that has come from its Grasshopper Integrated Pest Management (GHIPM) Project, which began in 1987.

Protecting areas of water on rangeland is important in grasshopper control programs. Present EIS guidelines state that liquid ULV sprays should not be applied within 500 feet (152 m) of aquatic habitat (reservoirs, lakes, ponds, seasonal pools, springs, streams, rivers, swamps, bogs, marshes, and potholes) or where leaching or surface runoff is likely, or when precipitation seems imminent. In recent years, there has been unresolved discussion about the definition of wetlands, and whether or not dry intermittent creek beds, wet meadows, and seasonally dry potholes qualify under the definition.

Aquatic habitat buffers also apply to areas treated with some baits. When chemical baits are used, the width of the no-treatment zones around aquatic habitats is 200 feet (61 m). When baits are used, buffer zones are smaller, and more of the area harboring grasshoppers can be treated. Bran baits containing the biological control agent *Nosema locustae* can be used without buffer zones. Some pest managers believe that being able to treat a larger proportion of the area lengthens the time period before the site is reinfested.

Baits do have limitations: damp or wet weather hampers use, not all grasshopper species will eat dry baits, baits are more expensive to apply than liquid ULV sprays, and baits provide a lower level of control of susceptible species compared to liquid sprays (see chapter II.12). However, baits do make it possible to reduce the size of buffer zones, obtain some suppression of grasshoppers that otherwise would be untreated using ULV sprays, and minimize insecticide effects on nontarget species.

After no-treatment and no-spray zones for sensitive areas are identified and mapped, the APHIS State plant health director or the authorized APHIS representative should verify the treatment locations in a pretreatment reconnaissance flight with the spray pilot(s). Boundaries should be clearly and adequately marked, preferably with large peices of fluorescent orange material. There should be confirmation of the no-treatment sites. Records and maps also should be signed by APHIS representatives and pilots and dated after the pretreatment flights. The pilots(s) must clearly understand locations and boundaries of buffer zones.

When called for during chemical spray operations, spray-deposit dye cards should be placed within the buffer zones to detect drift or inadvertent treatment of no-spray sites. Lack of spray deposit will verify that buffer zones did prevent exposure to sensitive areas being protected. With bran baits, cards containing adhesive or small pans placed in the buffer zones will detect inadvertent treatment.

Aircraft utilizing an electronic guidance system (Loran C or Global Positioning System) will aid greatly in identifying buffer zones and increasing the accuracy of applying sprays or baits (fig. III.8–1). When acceptable electronic guidance is available and used, ground flagging to mark the areas can be reduced or eliminated. Some guidance systems also are combined with a printed record of the flight showing exact locations of areas treated. A printed record adds to accountability and quality assurance. In the future, Federal agencies may require detailed printed records of insecticide applications in treatment areas.

APHIS has found that only rarely is part of a treatment block treated a second year in a row. Typically, APHIS may treat a block of land only once every several years.

Figure III.8–1—In the era before global positioning systems, agricultural pilots had to turn the nozzles of their spray equipment on and off manually. Pilots did this when they spotted “flagmen” who stood on the ground at the edge of spray plots or buffer areas. It was virtually impossible to adjust the on/off decision in light of near-ground wind, so insecticide drift was common. Naturally, flagmen were exposed to toxicants just like the target pests! Now, however, computerized equipment on the spray planes can automatically starts and stops the flow of pesticides using sophisticated mapping and geostationary satellite coordinates.

Buffer Zones for Endangered Plants

Buffer zones for T and E plants are important, not because of a direct effect of insecticides on plants but to protect any insect pollinators that might be necessary for reproduction of the plants. The only insecticides (malathion, acephate, and carbaryl) registered and approved by APHIS for use in grasshopper control on Federal lands are not known to be toxic to plants at the rates used. The insecticides are toxic to some flower-visiting insects, however.

Is it common for T and E plants to need insect pollinators? The T and E plant species studied during the GHIPM Project demonstrated a dependency on insects, particularly native bee species, to move pollen from one flower to another (chapter III.5). Reproductive success of 24 of 26 plant species studied during the project is greatly increased by the presence of native bees. Grasshopper control efforts must be designed to prevent or minimize insecticide exposure to active pollinators of T and E plants.

The question of adequate buffer-zone size is extremely complex. How can pest managers define “adequate size” in a T and E context? The answer to this question depends on several factors including:

- The distance bee pollinators move between their nesting sites and flower populations,
- The distances over which bees forage for food from flowers, and
- The distances bees must move to gather other needs such as mud, leaf pieces, resin, etc., that are important for nest construction.

The brief answer to questions of adequate size is that scientists and pest managers really do not know what is adequate. One way to determine the size of buffer zones is to base the size on the protection needed; however, determining the protection needed often can be difficult. Some studies to determine at least partial answers to the question of size have not been successful (chapter III.5).

For the most part, bees appear to act in ways that increase their foraging efficiency. When possible, bees nest close to the flowers they visit for pollen and nectar. Sometimes bees cannot do so because the proper nest sites are absent. Sometimes bees also forage farther than usual because flower density is low or because other resources are not available at nesting sites.

Studies noted in chapter III.5 did show that many species of bees are capable of flying several miles to return to their nests. Whether bees do this routinely is not known. Without a complete knowledge of insect pollinator behavior, the common (and some scientists believe the safest) approach is a conservative one. A buffer zone of 3 miles’ (4.8 km) radius usually is employed around T and E plant populations when using liquid insecticides.

The 3-mile buffer zone can be reduced or eliminated if information shows that the species in question is a self-pollinator or reproduces asexually or if the spray is not a potential problem to the pollinator species. Obviously, if no pollinators are needed, there is no effect on the T and E plants from the use of insecticides.

When using the common formulation of 2 percent carbaryl bran bait or other dry baits to treat grasshoppers, it is unlikely that the control program would need any buffer zone (chapter III.4) even with bees present. Because they do not eat bran baits, bees are not directly exposed to the insecticide.

Change in Peregrine Falcon Buffer Zones

The former standard buffer for peregrine falcon *(Falco peregrinus)* aeries (nests), hack sites (release of young peregrines after acclimation and supplemental feeding), and other release or habitat sites was a 10-mile no-treatment or drift radius (for aerial applications). It is now possible to establish buffer zones that are less arbitrary and correspond to the foraging area of the birds–often a long, narrow strip such as a valley or canyon. The foraging areas must be determined by a review team including one representative each from APHIS, FWS, the State conservation agency, and the land manager (or landowner if private land).

Aerial insecticide treatments then can be applied to within 1 mile (1.6 km) of the nest or release site. The boundaries of known foraging areas have a 500-ft (152-m) no-treatment zone. Bait applications with ground equipment can be made to within 0.5 mile (0.8 km) of a nest or release site and within 200 feet (61 m) of foraging areas. Reduced peregrine falcon buffer zones have not been widely used yet in grasshopper control programs, so the zones' use and effect should be part of the project monitoring plan.

Examples of Effective Uses of Buffer Zones

Piping plovers *(Charadrius melodus)*, an endangered species, nest on the sandy shoreline of Lake Sakakawea adjacent to grasshopper control areas in North Dakota. In 1989, a "hot spot" carbaryl bait treatment (2 lb/acre of 2 percent carbaryl bran bait–0.04 lb/acre AI) was applied to land immediately adjacent to a breeding pair of piping plovers with two small chicks and their no-treatment buffer zone (200 ft) near the nest site. Periodic posttreatment observations verified normal development and behavior of the chicks and adults (McEwen and Fowler unpubl.).

In 1991, a 19,200-acre (7,770-ha) area was sprayed with Sevin® 4-Oil at the standard IPM rate. APHIS sprayed liquid Sevin in the block–excluding a 0.5-mile (0.8-km) strip along the lake shore that was treated with carbaryl bait (2 lb/acre–2 percent actual ingredient). APHIS applied the bait and left a 200-ft (61-m) untreated strip at the water line. Observations on the nesting plovers indicated no effect, and breeding piping plovers were found at the same site in the following year (McEwen unpubl.).

This piping plover site is an especially difficult treatment situation because it is near reseeded crested wheatgrass *(Agropyron cristatum)*. Large areas of nearby native range have been reseeded to crested wheatgrass. The plant's clumpy growth form, with bare ground between plants, tends to promote high pest grasshopper densities. Many grasshopper species prefer bare ground for laying eggs. Also, large expanses of crested wheatgrass lose nearly all the bird species associated with native grasses (Reynolds and Trost 1980) that would be preying on the grasshoppers. Part of the loss of breeding birds is based on poor nesting habitat associated with crested wheatgrass.

The authors also have used and evaluated buffer zones around other aquatic sites in western North Dakota. These zones were in relation to large-scale Sevin 4-Oil treatments in 1991 and 1993 adjacent to the Little Missouri River. The standard aquatic buffer zones of 500 ft (152 m) were in place. In both years, carbaryl was detected in the river.

In 1991, a drought year, the maximum concentration of carbaryl detected was 0.085 parts per million (p/m); in 1993, a wetter year, it was 0.013 p/m. These low concentrations were found 1–2 hours after treatment and then rapidly declined (Beyers et al. 1995). Samples at 48 hours contained less than 0.0005 p/m, well below the concentrations generally known to begin affecting other invertebrates (0.002–1.90 p/m) and fish (1.95–39 p/m) (Johnson and Finley 1980). The only biological effect was an increase in the number of Ephemeroptera (mayflies) in the immediate (1–3 hr) postspray drift samples in 1991.

Natural events had greater impact on the aquatic invertebrates in the river in 1991 than did the insecticide. Monitoring of brain acetylcholinesterase (AChE) activity in flathead chubs *(Platygobio gracilis)* collected from the treatment area showed no inhibition, indicating no adverse carbaryl effects. Measurement of AChE activity is a method of detecting toxic effects of pesticides. It was concluded that the light drift of Sevin 4-Oil into the Little Missouri River was biologically insignificant (Beyers et al. 1995).

A study of golden eagle *(Aquila chrysaetos)* response to Sevin 4- Oil treatments around active nests was initiated in 1993 and is still underway (1995) in North Dakota. Nest areas were treated in June 1993 and 1994 when the young eagles were 4–7 weeks of age. Each young eagle was captured at fledging (10–11 weeks of age) so field crews could take biological measurements and blood samples and attach radio transmitters for postfledging observations. Telemetry is used to determine movements, behavior, survival, and dispersal from the natal (hatching) areas. Preliminary results indicate no differences in survival, movements, and dispersal between young golden eagles from sprayed and unsprayed territories.

Eagles from treated nests tended to be less active in afternoon and evening time periods and preened more (Bednarski and McEwen 1994, Bednarski unpubl.). Fledglings from treated areas had slightly higher *(P* = 0.11) blood plasma cholinesterase activity, a normal "rebound" or overcompensation effect commonly seen in birds after a light exposure to an inhibiting pesticide (Taira 1994), Taira and McEwen unpubl.). Territory maintenance, nesting activity, and productivity of the mature pairs of golden eagles in the sprayed and untreated areas are being followed 1 and 2 years after treatment.

Preliminary findings suggest that buffer zones of 500 ft (152 m) or possibly 200 ft (61 m) around the actual nest site will be adequate for protection when treating with Sevin 4-Oil. Further studies may show that buffer zones could be even smaller or possibly eliminated. The large foraging area (± 50 mi^2 or 129 km^2) characterizing an average territory of a breeding pair of golden eagles need not be of concern. A small area (± 5 acres or 2 ha) around each nest easily could be left untreated, without the human disturbance caused when placing flags, by using an electronic guidance system. The human disturbance of people on foot in the immediate vicinity of the nest should be avoided and could cause more problems than the treatment itself. Again, restrictions of the biological assessment and biological opinion will control program design and operation.

Although the effects of carbaryl on nesting golden eagles have been examined during the GHIPM Project, there has been no study of the effects of malathion on golden eagles. A study utilizing malathion also should be done because it was found that another raptor species, the American kestrel *(Falco sparverius)*, is very sensitive to malathion toxicity in the nestling stage (Schleve et al. 1993 unpubl., McEwen et al. 1994 unpubl.).

Potential Consequences of Buffer Zones

Treatment-free buffer zones may appear to be an obvious way to protect sensitive areas. Although liberal use and size of zones may seem safest, unneeded or exaggerated protection may reduce the effectiveness (efficacy) of grasshopper control programs. Buffers have varying impacts on treatment program efficacy, depending on the specific goals of the program (minimum economic level of control or maximum control) and where in the cycle the current grasshopper population exists. While designed to protect nontargets, buffer zones also can provide protection for pests the program seeks to control.

One concern with buffers occurs when the grasshopper population is expected to be about the same or greater in the following year. When the control effort is crisis in nature, maximum control of damaging grasshoppers is the goal. Untreated zones in a treated block may contribute to extending or expanding the problem by harboring grasshoppers, especially when grasshopper populations are cycling upward. In some cases, a large number or size of buffer zones can result in an immediate loss in the integrity of the spray block (less efficacy of treatment). These zones may result in the need for additional treatments and may expose larger tracts of land to pesticide treatments later. Fewer long-term control problems should result from untreated buffer zones when the grasshopper population is expected to decline.

Regardless of the grasshopper population cycle, blocks with large numbers of irregular buffer zones may result in increased treatment difficulties during the actual spray operation. The increased difficulty may be reflected in an increased cost of the application contract. Increased cost may result from marking each zone on the ground to ensure its identify from the aircraft applying the treatment. Marking is required if accurate electronic guidance is not available to the applicator. Additionally, costs associated with environmental monitoring (if required) of the buffer zones also may substantial. Together, these additional costs may be very significant. Coupled with leaving enough of the problem grasshopper population in the buffer zones possibly to reinfest treated areas, these additional costs could reduce the length of the economic benefit of the treatment. There even may be cases where the total buffer-zone acreage or the associated additional costs are so high as to negate the value of a particular treatment.

Buffers around water are the most frequently encountered treatment-free areas within a spray block. However, it is not unusual for grasshoppers to exist at high densities near rivers, streams, lakes, or ponds. In some cases, these areas around water harbor the highest densities of grasshoppers in the entire proposed treatment area. The entire grasshopper population, including that in buffer zones, must be considered for the most economically, biologically sound program to result.

One area of concern for use of buffers is in small, isolated infestations identified as historic hot-spots. In such areas, buffers that prevent effective treatment could be a threat to the concept of treating localized areas before grasshoppers can spread to larger acreages. Large numbers of uncontrolled grasshoppers in buffers–within areas where preventative hot-spot treatment is the foundation of an areawide program–could prevent full implementation of the concept and seriously jeopardize the overall program.

In many cases, a specifically customized treatment may provide the protection needed for a sensitive area while addressing most of the pest population. An example of a customized treatment would be the use of ground-applied bait adjacent to waterways, with an application direction away from the water. If performed properly, such a treatment could be conducted within a few feet of the water. Conscientious consideration–on a case-by-case basis by all participants–should provide an economically, biologically, and environmentally acceptable treatment solution in almost all situations.

Additional research and more knowledge may, in the future, justify modifications to buffer zones and the agreements between Federal agencies and land managers. Until the knowledge is available to call for modifications, the guidelines set forth in the 1987 EIS and guidelines specified for T and E species will dictate how buffer zones are established for grasshopper control programs.

Conclusions

Buffer zones play a vital role in protecting the environment during grasshopper control programs on public lands. APHIS and land-management agencies regularly share information about T and E species, aquatic areas, and sensitive areas necessary to provide effective buffer zones. Currently, APHIS uses the guidelines contained in the 1987 EIS when conducting treatment programs for rangeland grasshopper control and suppression. As noted in the EIS, buffer zones may be subject to revision as new information comes to light.

APHIS bases its treatment programs on sound biological knowledge. At no time does APHIS intentionally jeopardize nontarget species in a treatment block. Buffer zones reflect the desire to provide protection as needed. Customized treatment programs could help resolve difficult situations, especially when grasshopper populations are building and presence of buffers within treatment areas could lead to reinfestation.

References Cited

Bednarski, L. T.; McEwen, L. C. 1994. Response of post-fledging golden eagles *(Aquila chrysaetos)* to Sevin-4-oil application. In: Abstract book, 15th annual meeting, Society of Environmental Toxicology and Chemistry; [date of meeting unknown]; Denver, CO. [Place of publication unknown]: Society of Environmental Toxicology and Chemistry: 249.

Beyers, D. W.; Farmer, M. S.; Sikoski, P. J. 1995. Effects of rangeland aerial applications of Sevin®-4-Oil on fish and aquatic invertebrate drift in the Little Missouri River, North Dakota. Archives of Environmental Contamination and Toxicology 28: 27–34.

Johnson, W. W.; Finley, M. T. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Resour. Publ. 137. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service. 98 p.

Reynolds, T. D.; Trost, C. H. 1980. The response of native vertebrate populations to crested wheatgrass planting and grazing by sheep. Journal of Range Management 31: 122–125.

Schleve, G.; McEwen, L. C.; Clements, W. E. 1993. Sublethal effects of carbaryl and malathion on nestling American kestrels. In: Abstract book, 14th annual meeting, Society of Environmental Toxicology and Chemistry; [date and place of meeting unknown]. [Place of publication unknown]: Society of Environmental Toxicology and Chemistry: 87.

Taira, T. 1994. Blood analysis of American kestrel and golden eagle nestlings exposed to malathion or carbaryl. M.S. thesis. Ft. Collins, CO: Colorado State University. 80 p.

U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 1987. Rangeland Grasshopper Cooperative Management Program: final environmental impact statement. Washington, DC: U.S. Department of Agriculture, Animal and Plant Health Inspection Service. 221 p.

References Cited–Unpublished

Bednarski, L. T. Response of post-fledging golden eagles *(Aquila chrysaetos)* to Sevin® 4-Oil application. M.S. Thesis. Ft. Collins, CO: Colorado State University. [In preparation.]

McEwen, L. C. and A.C. Fowler. 1989. Observations on breeding piping plover *(Charadrius melodus)* and chicks adjacent to a site treated with carbaryl bait for grasshopper control. Unpublished data.

McEwen, L. C. 1991. Observations on piping plover *(Charadrius melodus)* nesting success near a site treated with carbaryl bait and Sevin 4-Oil for grasshopper control. Unpublished data.

McEwen, L. C.; Petersen, B. E.; Beyers, D. W.; Althouse, C. M.; Bednarski, L. T.; Taira, T.; Schleve, G. R. 1994. Grasshopper Integrated Pest management Program Environmental Monitoring and Evaluation. In: Cooperative Grasshopper Integrated Pest Management Project, 1994 annual report. Boise, ID: U.S. Department of Agriculture, Animal and Plant Health Inspection Service: 131–150.

Taira, T.; McEwen, L. C. Blood analysis of American kestrel and golden eagle nestlings exposed to malathion or carbaryl. [Manuscript has been submitted to Environmental Toxicology and Chemistry.]

III.9 Environmental Monitoring of Grasshopper Control Programs

Michael T. Green

Environmental monitoring is the measurement of the effect on the environment of pesticides used for pest control. Monitoring is required by law, is the policy of the Animal and Plant Health Inspection Service (APHIS), and provides useful information for pest-control programs. Monitoring has been, and will continue to be, an important part of grasshopper control operations.

Why Monitor?

Monitoring is required by the National Environmental Policy Act (NEPA) to document the implementation of mitigative (moderating) measures, such as buffers around sensitive sites. In APHIS, we monitor to compare residue levels and nontarget effects resulting from treatments with predictions made in the risk analyses in environmental impact statements written for programs such as grasshopper control.

Sometimes monitoring is conducted under the Endangered Species Act (ESA) to demonstrate protection of threatened and endangered (T and E) species or habitats that are critical for those species. Whether or not to monitor is specified in protection measures agreed to during consultations between APHIS and the U.S. Department of the Interior's U.S. Fish and Wildlife Service (FWS).

Not only is environmental monitoring APHIS policy, it also provides valuable information for APHIS. Information gained from monitoring leads to a greater understanding of the effects of the program on the environment, information that has proven itself useful numerous times. Information gained also is valuable as a tool for assessing the effects of future programs, for educating the public regarding the effects of programs on public health and the environment, and for defense of the program in case of claims or of litigation over purported adverse effects.

In grasshopper programs, monitoring is done mostly out of concern for effects on nontarget plants and animals. Monitoring often is required around sensitive sites (habitats of T and E species, wildlife refuges, aquatic habitats, areas of human occupancy, and other sites of concern to the public) and to demonstrate that standard operating procedures or protective or mitigation measures are adhered to. In addition, monitoring is used to fill gaps in knowledge regarding the fate and transport of program chemicals or biological control treatments.

The Monitoring Plan

Environmental monitoring should be thought of as integral to every grasshopper treatment. APHIS' Environmental Monitoring Team (EMT), within Plant Protection and Quarantine (PPQ), designs the monitoring plans for APHIS programs. EMT should be contacted in the early planning stages for each new control program, such as during the preparation of the site-specific environmental assessment (EA). EMT also should be contacted if treatments are planned for new areas already covered by a previously existing EA and no new EA is being prepared.

The APHIS State Plant Health Director (SPHD) or officer organizing the program should also involve the PPQ environmental monitoring coordinator when contacting EMT. If a site-specific EA is prepared, it should state whether or not monitoring will be conducted and then describe the type of sensitive sites to be monitored. EMT—in coordination with the SPHD, the environmental monitoring coordinator, and the FWS if T and E species are involved—will determine whether any sites should or should not be monitored. If monitoring is required, then EMT personnel will write the monitoring plan.

The monitoring plan will describe where and when sampling will take place, what will be sampled, and how many samples should be collected. The types of samples collected might include flowing or stationary water, soil, sediment, fish, insects, vegetation, and dye cards that measure airborne drift. Trained personnel (environmental monitors) will carry out the monitoring plan and send samples for residue analysis to APHIS' National Monitoring and Residue Analysis Laboratory (NMRAL) in Gulfport, MS. The results from the laboratory are analyzed by EMT and interpreted with the aid of field notes and data collected at the time of treatment and sample collection. These data are reported in monitoring reports by EMT at the end of the treatment season. Addresses and phone numbers are listed on the next page.

Addresses and Phone Numbers

USDA–APHIS–PPQ
National Monitoring and Residue Analysis
Laboratory (NMRAL)
3505 25th Avenue, Building 4
Gulfport, MS 39501
(228) 863–8124
(228) 867–6130 FAX

USDA–APHIS–PPQ
Environmental Monitoring Team
4700 River Road, Unit 150
Riverdale, MD 20737–1237
(301) 734–7175
(301) 734–5992 FAX

Monitoring Tools

There are many tools environmental monitors use to collect samples from the environment. It is important to make a list of the equipment necessary before starting environmental monitoring. NMRAL will send supplies overnight if necessary. The basic tools are dye cards, which are used to measure airborne drift of chemicals and pans or gypsy moth sticky traps to collect drifting bait.

Water is collected by dipping a container into the water body or continuously sampled with a peristaltic pump, depending on the sampling question of interest, the type of water body being monitored, and the chemical being sampled. Soil corers sometimes are used to collect soil; vegetation is collected by (gloved) hand. Water samples must be stabilized by lowering the pH with a special kit, and all samples must be frozen as soon as possible after collecting. This process requires having a large freezer nearby, even at relatively remote sites, and preferably dry ice or an ice bath in which to place bagged, labeled samples in the field. EMT and NMRAL are available to help with questions about collecting sites and methods.

Monitoring plans and techniques require considerable forethought and planning. It is critical, therefore, to get EMT involved early on in any operation, so that an environmental monitoring plan can be written, distributed, and worked into the overall cooperative control operation.

Chemicals in the Water?

The chemical labels for ultralow-volume (ULV) malathion, carbaryl, and carbaryl bait plainly state the risks to aquatic animals. The 2000 Cheminova label for Fyfanon® ULV malathion states, "This product is toxic to fish, aquatic invertebrates, and aquatic life stages of amphibians. For terrestrial uses, do not apply directly to water, or to areas where surface water is present. . . . Drift and runoff may be hazardous to aquatic organisms near the application site." The labels for carbaryl spray and carbaryl bait are similar. For this reason, a 500-ft no-treatment buffer for aerially applied ULV pesticides and a 200-ft buffer for bait applications have been adopted as operational procedures in grasshopper programs.

The technology for detecting chemical residues is such that malathion residues can now be detected in water down to about 1/100th (0.01) of a microgram per liter (µg/L). In a pond 1 acre in size and 1 foot deep, the amount of malathion necessary to create residues near 0.05 µg/L is only about 0.03 fluid oz, or 0.38 percent of the original application (8 fluid oz/acre). Thus, if 99.5 percent of the spray lands on its target or in the buffer, and just 0.5 percent of it reaches a 1-ft-deep 1-acre pond, then the resulting residues would be detectable. The calculations for carbaryl are similar. At 1.0 µg/L, small aquatic crustaceans and aquatic stages of insects become susceptible. These organisms are more tolerant of carbaryl residues, showing sensitivity near 1 to 5 µg/L. Fish are from 10 to 1,000 times more tolerant of malathion and carbaryl than are aquatic invertebrates.

The chemical label states the risks of the pesticides to aquatic organisms and that drift and runoff could be harmful to them. The self-imposed buffers in the grasshopper program are probably sufficient in most cases to prevent harmful residues. Regardless, monitoring is recommended to be sure aquatic ecosystems are unaffected by program activities. Dye cards at the water's edge and water samples will help program managers detect and quantify any residues reaching the water and suggest when buffers might need to be enlarged to minimize residues further.

Although carbaryl and malathion are the most commonly used pesticides in the grasshopper program, other pesticides (such as Dimilin®) might be adopted in the future. Most pesticides that would be effective at grasshopper control probably also will require a no-treatment buffer and residue monitoring around water bodies.

Conclusions

Environmental monitoring is a method of assessing effects of the grasshopper control program on nontarget animals and plants. Monitoring sometimes is required to bring the program in compliance with Federal statutes such as the ESA and the NEPA. APHIS also has the policy of monitoring the environment around pest eradication and control programs such as the cooperative rangeland grasshopper control program.

Whether or not monitoring is required depends on the site, the presence of T and E species, protected areas, wetlands, and other factors. EMT will help determine if monitoring is advisable for specific grasshopper control operations and should be contacted as early as possible during the planning of such operations.

Information gained through monitoring has been of considerable value to the program in the past, and monitoring will continue to be an important part of grasshopper programs in the future.