BIRDS IN AGRICULTURAL AREAS - REDUCING PESTICIDE RISKS TO BIRDS USING A RISK ASSESSMENT ANALYSIS

by

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A Dissertation
Submitted to the
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of
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Master of Science
George Mason University, 2003

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DEDICATION

This work is dedicated to my family and friends without whom I'd never have been able to finish the task!

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ABSTRACT

BIRDS IN AGRICULTURAL AREAS - REDUCING PESTICIDE RISKS TO BIRDS USING A RISK ASSESSMENT ANALYSIS

Susan A. Heath, PhD

George Mason University, 2008

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Bird presence is well documented in agricultural areas and conversion of native habitats to agricultural land has resulted in shifts in species composition and abundance and alteration of geographic ranges. Pesticides have been widely used on agriculture since World War II, and while massive avian mortality events have been documented, the long-term effects of exposure to pesticides are unknown. Although the bioaccumulating organochlorine pesticides have mostly been banned, their replacements, the organophosphate pesticides are more toxic to wildlife. This study attempted to: 1) establish an extensive database on what bird species are using agriculture, where they are found and how they are using agricultural areas; 2) analyze these data to determine if there are significant relationships between avian population abundance, agricultural intensity, and crop types; and 3) determine if there are negative population trends associated with pesticide use. I built an MS Access database documenting bird use of

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agricultural areas from an extensive literature review including 577 citations, mostly from peer-reviewed journals. To analyze bird use of agriculture, I divided the north central states into 20 areas based on Omernik's Ecoregions and mapped Breeding Bird Survey routes to those areas using ArcView 3.3. Using North American BBS data analyzed with the route-regression method, I calculated species trends and abundances for all grassland and ground nesting bird species with sufficient data to meet statistical requirements. Using National Agricultural Statistics Service and National Center for Food and Agricultural Policy data, I calculated six independent variables including agricultural intensity, percent herbicide use, percent insecticide use, acute toxicity, chronic toxicity, and an herbicide indirect parameter. Linear regressions showed a significant positive relationship between Killdeer (*Charadrius vociferous*) abundance and agricultural intensity and significantly negative relationships between Upland Sandpiper (*Bartramia longicauda*) and Grasshopper Sparrow (*Ammodramus savannarum*) abundance and agricultural intensity ($p < 0.05$). Linear regressions also showed significant positive relationships between Killdeer abundance and amount of corn and amount of soybeans and between Bobolink (*Dolichonyx oryzivorus*) abundance and amount of oats ($p < 0.05$). Additionally, linear regressions showed significantly negative relationships between Dickcissel (*Spiza americana)*, Grasshopper Sparrow, and Western Meadowlark (*Sturnella neglecta*) abundance and amount of oats, between Bobolink abundance and amount of soybeans, between Bobolink and Grasshopper Sparrow abundance and amount of corn and finally between Dickcissel, Grasshopper Sparrow, and Western Meadowlark abundance and amount of alfalfa ($p < 0.05$). Linear regressions between the independent variables and the percentage of negative species trends by area showed the chronic toxicity variable to be the most important in predicting negative species trends ($p < 0.05$). Multiple regressions between a reduced set of independent variables and individual species trends gave inconclusive results because of the difficulty in separating pesticide effects from agricultural intensity effects.

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INTRODUCTION

Over half of the land area of the United States (U.S.) is devoted to agriculture (Vesterby and Krupa 1997). This land is divided among the three primary agricultural uses: cropland, grazing land, and special uses (e.g. farmsteads and farm roads). Cropland and grazing land represent 20 and 25% of land use, respectively (Vesterby and Krupa 1997). Although world demand for food is expected to rise in response to increasing world population, in most developed nations these demands will be met by advances in technology and agricultural intensification rather than by an increase in agricultural land (OECD 2001). In the United States, cropland has decreased by about three percent over the last 25 years (Vesterby and Krupa 1997), and over the next 20 years, the amount of land designated for agricultural use is not expected to change significantly.

Bird presence is well documented in agricultural areas and conversion of native habitats to agricultural land has resulted in shifts in species composition and abundance and alteration of geographic ranges (Rodenhouse et al. 1995). Rodenhouse et al. (1993) reported that 215 species of Neotropical migrants use agricultural areas in North America. Nine of these Neotropical migrant species are currently listed as threatened or endangered or are candidates for listing, and agriculture is implicated in the decline of all of them (Rodenhouse et al. 1993). Only 10 of the 215 are known to cause agricultural

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damage over wide geographic areas. The overwhelming majority of these species are utilizing agriculture in a benign or even beneficial way during the breeding and/or migratory seasons (Rodenhouse et al. 1993). Kirk et al. (1996) reported that the predatory activities of birds can suppress insect populations, at least at medium to low infestation levels, an ecological service that should not be overlooked in integrated pest control plans. Despite this and other assessments of use of agricultural areas by birds, the total number of bird species using agricultural areas in North America is not currently known.

Pesticide Use in Agriculture

Prior to World War II, agricultural pesticides were relatively simple derivatives of naturally occurring plant products and minerals (Hoffman 2003). Synthetic organic pesticides were initially developed for commercial agricultural use in the late 1940s and 1950s. Despite Rachael Carson's warnings in *Silent Spring* (Carson 1962), pesticides were widely adopted by the mid-1970s, and their use has increased ever since. The main pesticide groups are insecticides, rodenticides, avicides, fungicides and herbicides. Of these, only herbicides have a low acute toxicity to birds and insecticides have had the most widespread and publicized negative effect on birds (Brown 1978).

The organochlorine insecticides including dichlorodiphenyltrichloroethane (DDT) and its analogs dieldrin, endrin, heptachlor, and chlordane have high solubility in fats and a long

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environmental persistence, which resulted in significant bioaccumulation in top carnivores (Blus 2003). For example, egg shell thinning resulted in significant population declines of top carnivores such as the bald eagle (*Haliaetus leucocephalus*), the osprey (*Pandion haliaetus*), the peregrine falcon (*Falco peregrinus*), and the brown pelican (*Pelecanus occidentalis*) (Blus 2003). Research that demonstrated these effects played a major role in the registration cancellation of many organochlorine pesticides (Hoffman 2003).

Organochlorine insecticides were replaced by organophosphorus (OP) and carbamate (CB) compounds, because they are short-lived in the environment and are readily metabolized by wildlife (Hill 2003). Thus, they are not prone to bioaccumulation, but their direct toxicity to wildlife is much higher than the organochlorines. In addition, their mode of action as cholinesterase inhibitors results in a wide array of behavioral and physiological effects that are hard to identify and quantify (Hill 2003). Several of these compounds have resulted in massive kills of birds and other wildlife. In Argentina, an estimated 20,000 Swainson's hawks were poisoned when they fed on grasshoppers sprayed with monocrotophos (Hooper et al. 1999). In North Dakota, 37 bird species, with estimates from 5,000 to 25,000 total individuals, were poisoned from a single aerial application of fenthion (Seabloom et al. 1973). Mineau (2003b) estimated, based on several industry-led field studies that at its peak, the insecticide carbofuran was killing 17 to 91 million songbirds annually in the U.S. corn belt. Despite extensive evidence of the

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dangers to wildlife, compounds similar to these remain the mainstay of insect, mite, and nematode control today.

Herbicides and fungicides tend to have lower acute toxicities to wildlife than insecticides and thus are not associated with well publicized mortality events. Although few herbicides have a high enough acute toxicity to cause significant avian mortality, European studies have shown that they can reduce avian populations through indirect effects, including limiting the availability of nesting habitat and the insect fauna supported by that habitat which is critical for foraging (Wilson et al. 1999). This relationship is not as well documented in North America, however. In addition, several herbicides have been found to be embryotoxic (Brown 1978). Mercury containing fungicidal seed treatments caused extensive mortality of seed-eating birds and their raptorial predators in Sweden, Canada, and France (Mineau 2003a, Fimreite et al. 1970). In addition, laboratory studies show that the use of some fungicides can have reproductive effects including cessation of egg laying, early embryonic death, and teratogenesis (Heinz 1976).

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Pimentel et al. (1992) reported that one billion pounds of 600 different types of pesticides are used annually in the U.S. The National Center for Food and Agricultural Policy's National Pesticide Use Database reports a similar number used on 87 crops in the 48 contiguous states in 1997 (NCFAP 1997). These pesticides are grouped by their 220 active ingredients including 37 fungicides, 66 insecticides, 96 herbicides, and 21 classified as “other.” “Other” includes fumigants, growth regulators, and defoliants (Gianessi and Marcelli 2000). These statistics include applications to cropland only and do not include applications to range lands and application of rodenticides.

Pesticide Impacts on Birds

Birds are especially sensitive to the more toxic OP and CB insecticides, and avian reproduction has been shown to be vulnerable to a wide range of pesticides (Hill 2003). The Avian Incident Monitoring System (AIMS), a joint project of the U.S. Environmental Protection Agency (EPA) and the American Bird Conservancy (ABC), currently lists 113 pesticides which have caused direct bird mortality (ABC 2005a). More than half of these are insecticides. Pimentel et al. (1992) estimated that 672 million birds are directly exposed to pesticides in agriculture each year. They took a conservative estimate of 10% mortality and determined that 67 million birds are killed by pesticide use annually. Note that this is the result of acute toxicity only and does not include secondary losses resulting from reductions in invertebrate prey or behavioral impacts to intoxicated individuals resulting in reduced survivability or fecundity.

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The effect of pesticide exposure on avian reproduction is perhaps the most difficult to quantify. In addition to egg shell thinning from persistent organochlorine bioaccumulation, several studies have documented changes in levels of reproductive hormones leading to decreased song production and displays (Grue et al. 1997, Hill

2003). Also, decreases in food consumption have lead to a decrease in the number of eggs laid and reduced time spent incubating in birds exposed to OP and CB insecticides (Grue et al. 1997, Hill 2003). Mineau et al. (1994) points out that a large proportion of currently registered pesticides have the potential to affect reproductive processes at levels that are not toxic to the parents. Risk assessment models and bird poisoning incidents show the hazard of pesticides to birds, but the aggregate impact of all pesticides to bird populations is unknown largely because the population effects of OP and CB insecticides have not been extensively documented (Sullivan 2003).

EPA Regulation Process

The EPA is tasked with assessing the impacts of pesticides on wildlife during the registration process. Testing protocols are developed under the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) and the Toxic Substances Control Act (TSCA). Currently, pesticide registration in the U.S. requires an acute oral toxicity test, a subacute dietary test, subchronic dietary toxicity tests, and chronic toxicity tests with reproductive effects as the primary endpoint (Hoffman 2003). In the U.S. all toxicity testing is conducted on two species, the Northern Bobwhite (*Colinus virginianus*) and the Mallard (*Anas platyrhynchos*). Several other countries add the Japanese Quail (*Coturnix japonica*) to the suite of test species (Hoffman 2003).

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The acute oral toxicity test is administered as a single dose at each of five or six predetermined dosage levels to overnight-fasted, young adult birds (Hoffman 2003). Birds must be at least 14 weeks of age and not yet mated (EPA 1996a). The doses are administered by gavage or capsule and the birds are fed immediately after dosing. All birds are observed for signs of intoxication for a minimum of 14 days post-treatment. Necropsies are performed on all birds that die and on a sub-sample of the survivors (Hoffman 2003). This test determines the median lethal dose (LD_{50}) that will cause 50% mortality of the test population (Mineau et al. 2001) and provides a preliminary indication of the lethal hazard of the chemical (Hoffman 2003).

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The subacute dietary test consists of a five-day feeding trial in which birds are monitored for mortality and signs of intoxication (Hoffman 2003). Food consumption is measured at 24-hour intervals on juvenile birds that are less than 14 days old (EPA 1996b). Observations are continued for a minimum of three days after administration of dosed food is discontinued (Hoffman 2003). This test determines the median lethal concentration (LC_{50}) in the diet that will lead to 50% mortality of the test population (Mineau et al. 2001) and serves as a composite indicator of vulnerability to a contaminated diet, allowing for metabolic changes that occur over time (Hoffman 2003).

The LD_{50} and LC_{50} values can both be used in risk assessments, but Mineau et al. (2001) argue against using the LC_{50} because the test provides unreliable results due to the difficulty of determining exposure during the test. The results are greatly influenced by the exact age and condition of the test population and the correlation of LC_{50} values among test species is weak, making extrapolation from test species to wild species difficult. In addition, field studies indicate that laboratory derived LC_{50} values are poor predictors of risk (Mineau et al. 2001). Because of this, current avian risk assessment depends almost entirely on the results of the LD_{50} test.

Regardless of which test is used, it is administered to at most only three species of birds and extrapolation of the values obtained for these species to the more than 800 avian species occurring in the U.S. has proved troublesome (Mineau et al. 2001). To resolve this problem, Mineau et al. (2001) present acute toxicity values that can be used for assessing the relative acute risk of different pesticides to any species of bird. Their reference values are based on a distribution-based method which incorporates a scaling factor for body weight to improve cross-species comparisons of toxicological susceptibility (Mineau et al. 1996). The reference values are expressed in terms of the HD_5, which is the 5% hazardous dose. It is based on the median estimates of the LD_{50} at the 5% lower tail of the avian species' sensitivity distribution for each pesticide. HD_5 values are corrected to be representative of species ranging from 20 to 1,000 g in weight, which accounts for the majority of avian casualties seen in documented bird kills.

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In the U.S., a pesticide manufacturer must also conduct an avian reproduction study for compounds which meet any of the following requirements:

1) the prescribed application procedures for this pesticide will result in birds being subjected to repeated or continuous exposure to the pesticide or any of its metabolites and degradation products, especially preceding or during the breeding season,
2) the pesticide is stable in the environment to the extent that potentially toxic amounts may persist in avian feed,
3) the pesticide is accumulated in animal or plant tissues, or
4) other information, such as that obtained from mammalian reproduction studies, indicates that reproduction in terrestrial vertebrates may be adversely affected (McLane 1986).

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In the avian reproduction test, the chemical to be tested is mixed into the bird's diet for a period of ten weeks before laying begins, which is controlled through photoperiod manipulation (Mineau et al. 1994). This test is administered to birds that are at least seven months old (EPA 1996c). During the egg-laying period, which lasts eight to ten weeks, eggs are removed from the adults the day they are laid and are incubated to hatching. The variables recorded during the test include adult body weight and food consumption, number of eggs laid, proportion of eggs placed in the incubator that are fertile, proportion of fertile eggs containing viable embryos at three weeks of