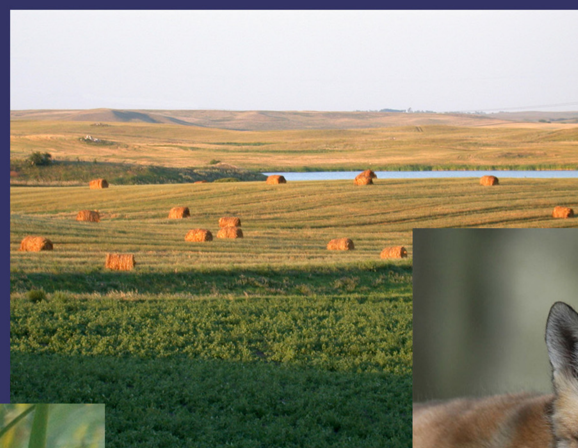
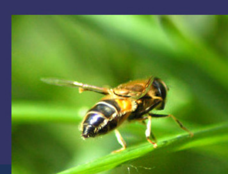
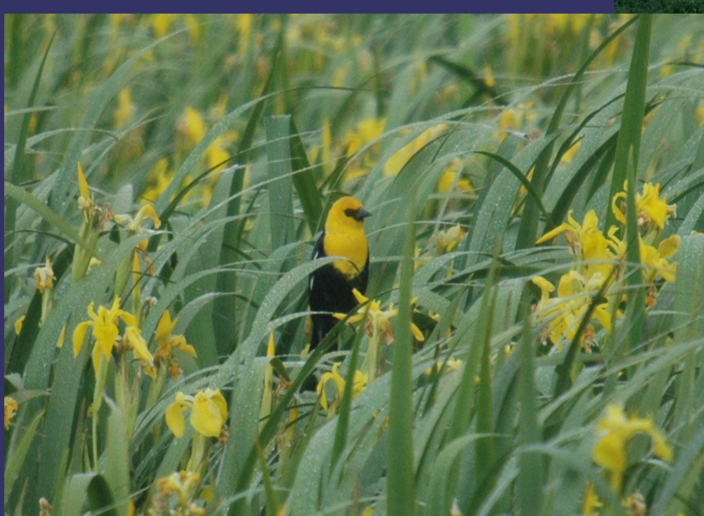
HABITAT IN AGRICULTURAL LANDSCAPES: HOW MUCH IS ENOUGH?

A STATE-OF-THE-SCIENCE LITERATURE REVIEW

BY KRISTEN BLANN







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WEST LINN, OREGON • WASHINGTON D.C.

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EXECUTIVE SUMMARY

Biodiversity — the number and variety of plants, animals, ecological communities, ecosystems and landscapes — is threatened, both globally and in North America. Agriculture (including agricultural practices, land conversion, water diversion, pesticides, and fertilizers) continues to be a leading cause of species endangerment. Land use change resulting from agriculture has altered the abundances and varieties of native species; introduced novel and potentially detrimental species to new areas, disrupted natural water and nutrient cycles, and significantly altered natural disturbance patterns (e.g., fire, flooding).

At the same time, the 500 million acres of U.S. land in farmland landscapes harbor a substantial portion of plant and animal species. Agricultural lands in many cases provide more suitable habitats for native wildlife and birds than do fragmented and extensively modified urban or suburban lands. Such lands often serve as a buffer between natural areas and more highly altered landscapes, providing food, cover, and breeding habitat, enabling movement and exchange of plant and animal populations.

The goal of this paper is to provide a comprehensive synthesis of current understanding regarding conservation of fish and wildlife habitat and biodiversity in agricultural landscapes, and to establish a framework for setting conservation goals, policy, and future research priorities. For the purposes of this effort, our "vision" of the agricultural landscape is one that supports viable populations of native species of plants and animals in functioning ecosystems — reversing the decline of threatened and endangered and at-risk species, and maintaining those species and communities whose numbers are stable or increasing.

Agricultural lands can provide more suitable habitats for native wildlife and birds than do fragmented and extensively modified urban or suburban lands. Such lands often serve as a buffer between natural areas and more highly altered landscapes, providing food, cover, and habitat which allow movement and exchange of plant and animal populations.

There are many factors accounting for biodiversity loss beyond the destruction of habitat. Minimum habitat areas for species may vary depending on region and landscape context. For most species, little is known about the minimum individual and population level habitat needs. The impact of habitat fragmentation on any given species also has substantial regional variation. Adjacent landscape patches can influence biodiversity by harboring habitat for non-native exotics, edge, predator, or colonist species that compete, reduce the habitat quality, or directly reduce the survival of species in remaining habitat patches and natural remnants. Ecological conditions, settings, habitat needs, and agricultural practices and systems all vary substantially geographically and regionally. Planning for ecosystems and landscape-scale habitat for biodiversity will require both watershed and terrestrial landscape approaches. Conservation must be planned and implemented at larger scales, from ecosystems and landscapes to entire regions.

NOT ALL AGRICULTURE IS CREATED EQUAL — BENEFITS OF MORE ECOLOGICAL AGRICULTURE

For thousands of years, agriculture has involved modification of natural habitats and ecosystems to produce food, fiber, and other products for human use. In many regions, native people intentionally managed the prairie and other natural ecosystems by mimicking natural disturbance patterns on the landscape, such as setting fires to maintain prairie, manage berry crops, and modify wildlife habitat.

In North America, the modern era of biodiversity loss — including the disappearance of many species and populations of plants and animals, landscapes, and ecological phenomena — was initially driven by extensive overexploitation of the American continent's timber, wildlife, fish, and the clearing of land for agriculture in the centuries that followed European settlement. Land was cleared to lay claim to new lands and create opportunities for growing populations to own land. The intensification of agriculture in the second half of the 20th century has further exacerbated biodiversity loss, eliminating the patchwork of fencerows, field edges, pastures, small wetlands, and other remnant natural habitats that provided refuge for many species of native plants and animals. The latest wave of agricultural intensification has been driven by technological advancements in inputs and expanding world markets for agricultural exports.

To stem the tide of biodiversity loss, scientists and conservationists in many fields have begun to recognize the role that some types of working lands play in providing habitat that is more compatible with biodiversity. Many terms have been used to describe a range of alternative, "eco-friendly" agriculture and ranching, from "conservation-friendly farming," "conservation-based agriculture," "sustainable agriculture," "organic agriculture" to "permaculture" and "ecological agriculture" or "ecoagriculture." These visions vary in the degree to which they are compatible with local biodiversity, but all are designed to reduce or reverse the contribution of agriculture to local and regional biodiversity loss.

To be truly "ecological," agricultural practices and systems intended to be compatible with conservation of natural communities should as much as possible resemble the structure and function of the natural ecosystem in its ecoregional context. For example, landscape-scale approaches to grazing are more often designed to mimic the seasonal and spatial patterns of grazing by native ungulates and other wildlife in order to restore the ecological structure and function of grassland ecosystems. Management practices can have significant impacts on whether agricultural lands function as sources or sinks for species which find such habitats attractive.

GUIDELINES AND RECOMMENDATIONS FOR LANDSCAPE SCALE ECOLOGICAL AGRICULTURE: SUMMARY OF GENERAL PRINCIPLES AND RECOMMENDATIONS

- Understanding the history of a landscape as well as its past and current spatial context is critical to conservation of local and regional biodiversity. Consider farmland within a watershed and ecosystem context.

- Agricultural systems should be designed as much as possible to mimic the scale and function of key ecological processes as they have evolved historically in a given region. Different types of agriculture are likely to be more compatible with biodiversity conservation in different landscapes, depending on the natural arrangement of physical features, habitats, and species, and land use history.

- Agricultural systems that mimic structure and function of local ecosystems will be more compatible with biodiversity protection. In prairie and prairie-forest border ecoregions, croplands should be in a matrix of natural grasslands, grazed lands, and pastures. More research is needed to understand the potential value of agroforestry lands in forest ecoregions as habitat and stepping stones for interior forest species.

- Remnant natural habitat patches are responsible for maintaining much of the biodiversity currently present in agricultural landscapes. These natural habitat remnants should be aggressively protected. Converting quality remnant systems to agricultural production or development should be avoided.

- Conservation in agricultural landscapes should focus on maintaining adequate patch size and restoring linkages and connectivity for native plants and animals. Large blocks of habitat are needed to strengthen regional networks of conservation reserves.

- Minimize connectivity of artificial habitats that tend to spread exotics and pest species, such as clearcuts, regularly cleared agricultural fields, and roadsides.

- Continue efforts to inventory and assess the biodiversity functions of existing farm and ranchlands. These include important ecosystem services, such as pollination, pest control, beneficial predation, flood and erosion control, nutrient cycling, groundwater recharge, and maintenance of water quality.

- Avoid intensification or conversion of farm and ranchlands where such lands are providing important biodiversity functions. Protection of agricultural systems and practices that are partially compatible with biodiversity protection should be a priority where these lands are threatened by fragmentation by more intensive land uses such as urban and suburban development.

- Focused efforts that concentrate ecological farming initiatives and incentives to areas that have been identified as biologically significant, or in the context of habitat requirements for threatened and endangered species, species of special concern, umbrella or keystone species, are likely to prove of greatest conservation value.

- Conservation of aquatic biodiversity will require substantial reductions in the aquatic ecosystem impacts of nonpoint source pollution from agricultural lands as well as the substantial effects of altered hydrology on aquatic ecosystem structure and function.

- Nutrient, soil, chemical, and sediment losses to surface and groundwater must be dramatically reduced. Intensive row-cropping and tillage should be eliminated, modified, or minimized on all highly erodible land and marginal land, including highly sloping lands, floodplains, erodible soils, riparian lands, and wetlands. Substantial contiguous buffer areas should be maintained around streams, water bodies, groundwater recharge areas, and coastal zones. Floodplain and riparian land should be restored or maintained with perennial, preferably native, species that can trap and filter sediment and nutrients as well as provide migration corridors as well as habitats for wetland, floodplain and riparian dependent species as well as birds.

- Remaining natural wetlands should be conserved to the maximum extent possible, as restored or created wetlands generally do not provide the same degree of ecological function. At least 10 percent of any watershed where wetlands were historically present should be conserved in wetlands and important associated upland zones in order to maintain biodiversity, water quality, and flood storage services. In many landscapes, such as the prairie pothole region, a much higher percentage of wetlands may be required to adequately maintain biodiversity services, particularly in light of climate change (Johnson et al. 2005). The natural diversity of wetland types in any given ecological region should be conserved in order to maintain the full range of different but important hydrological and ecological functions.

Conserving native biodiversity in agricultural landscapes will likely require a combination of:

- Limiting further conversion of native landscapes;
- Restoring some converted lands to native vegetation; and
- Implementing more eco-friendly agricultural practices on a substantial percentage of active farm acreage.

Achieving these changes and monitoring to make sure they are effective will require substantial societal will, public education and involvement. There are many unknowns. The scale, complexity, and unpredictability of ecological systems continue to elude scientific understanding. Although there are many small-scale studies, we lack data and predictive analyses on the landscape-scale biodiversity implications of different patterns of production systems, land use, and management in agricultural landscapes. Effective conservation will require integration of local knowledge, science, and deliberative democratic participation in land and natural resource management decisions and policy-making. Policies, market mechanisms, and other institutional arrangements are needed to create effective incentives for conservation land uses at local, state, and federal scales that adequately reflect public values for biodiversity as well as the value of ecosystem services. In many cases, these proposals will meet with substantial resistance.

It will be difficult to take the necessary actions without a shift in values and a broader public understanding of the benefits of biodiversity or the need for more than just habitat restoration and conservation of a few isolated remnant areas. Some may view such changes as unnecessary "sacrifice" or "tradeoffs." But if we do not do so voluntarily, similar tradeoffs and sacrifices will be imposed upon us and our children in the future, in response to the unsustainability of our current agricultural and food systems.

FOREWORD

Defenders of Wildlife is a national conservation organization focused on the long-term maintenance of biodiversity. In this context, protecting biodiversity means conserving native species of plants and animals large and small, in functioning ecosystems. It also means addressing biodiversity needs across the landscape, on public and private lands.

Many of the private lands that are relatively undeveloped are used for agriculture. While agricultural activities have adversely impacted biodiversity, these lands also offer tremendous opportunities to conserve and restore certain species and habitats.

Conservationists generally recognize that maintaining naturally functioning ecosystems with the full compliment of native species is not possible in many areas of the United States, especially those that are extensively developed and intensively managed for human uses. It is not realistic from a political or ecological perspective to expect large sections of the country to be returned to pre-European condition.

So how much conservation land is enough, and how should it be managed? Federal, state and local governments and the private sector spend billions of dollars each year to conserve land through direct acquisition, easements, landowner incentives, and a wide range of regulatory and collaborative activities. What does success look like? How will we know when our collective conservation goals have been reached?

Foreword by

Sara Vickerman

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The sad truth is that we have no collective conservation goals, or a shared vision of what we want the American landscape to look like. This paper was commissioned to serve as a starting point for determining how we might address biodiversity conservation needs in agricultural landscapes. Defenders asked Kristen Blann, an ecologist from Minnesota, to review the relevant scientific literature and summarize it to serve as background information for a continuing dialogue.

This report provides a wealth of technical information for resource professionals, landowners, conservation groups and policy makers who see the opportunity to conserve biodiversity on agricultural lands, and want to be a part of the effort to do so. It also provides a sense of direction for us to ponder, a new way of looking at our food and fiber producing lands.

INTRODUCTION

For the purposes of this document, we refer to ecological agriculture broadly to encompass the concept of agriculture that is most compatible, at a landscape scale, with conservation of biological diversity that is based on mimicking natural ecosystems function and services.

One third of all plants and animal species in the United States are at risk or subject to concern, and more than 500 species are already missing or presumed extinct. Nearly 60 percent of the North American continental land area has experienced significant loss of natural vegetation since European settlement. Many of the most threatened ecosystem types have declined from their original extent by 98 percent or more. These include the eastern deciduous old growth forests, midwestern oak savannah and prairie pothole wetlands, pine rocklands of south Florida, native grasslands and oak woodlands of California, and Palouse prairie in the Pacific Northwest (Bryer et al. 2000). Aquatic ecosystems, in particular, are among the most imperiled. For instance, 28 percent of amphibians, 34 percent of fishes, 65 percent of crayfishes, and 73 percent of freshwater mussels in North America are ranked extinct, imperiled, or rare (Allan and Flecker 1993).

Agriculture (including agricultural practices, land conversion, water diversion, pesticides, and fertilizers) continues to be a leading cause of species endangerment, affecting 38 percent of listed species (Groves et al. 2003). Agriculture affects everything from the quality and quantity of wildlife habitat to drinking water, local creeks, fish nurseries and shellfish in major estuaries (Jackson 2002). An estimated 60 percent of rivers in the U.S. are substantially impaired due to agricultural runoff (Imhoff 2003). Land use change resulting from agriculture has altered the abundances and varieties of native species; introduced novel and potentially detrimental species to new areas, disrupted natural water and nutrient cycles, and significantly altered natural disturbance regimes to which ecological communities are adapted, such as flooding and wildfire.

HOW MUCH IS ENOUGH?

This is as much a question of values, culture, and economic preferences as it is a question of science. What conservation values will shape policy and guidance? Do we want to restore biodiversity in natural landscapes, or do we only want to retain what we have now?

At the same time, agricultural regions often coincide with the most productive lands and natural areas of high biodiversity and endemism. The 500 million acres of U.S. land in farmland landscapes still harbor a substantial portion of native plant and animal species. Agricultural lands often serve as a buffer around natural areas, providing food, cover, and other critical components of habitat — enabling movement and exchange of plant and animal populations (Freemark et al. 2002). Thus, agro-ecosystems are under increasing pressure to play a much greater role in providing habitats, while continuing to provide food and other goods and services. Agricultural land use is often less damaging than urban, suburban, or other more intensive uses (Kerr and Cihlar 2004). In many parts of the country, conservation of working agricultural and grazing lands is a priority for conservation organizations seeking to avoid additional fragmentation and habitat loss due to urban, suburban, and commercial development. The presence of wild species in agroecosystems is important in many regions where there is limited potential to expand or establish additional biodiversity reserves. Thus, agriculture has a critical and potentially constructive role to play in the maintenance of biodiversity and ecosystem services.

Can our farmlands be managed to conserve and restore biodiversity while meeting growing national and global demand for food and commodities? Is modern agriculture compatible with conservation of natural ecosystems, landscapes, and species? Can sustainable and ecological farm management practices on a portion of the landscape contribute to meeting the challenge of conserving biodiversity? If so, how much is enough? What policy guidance can be derived from the existing research and literature on the amount and quality of habitat necessary to sustain the remaining biodiversity in America's landscapes?

This paper is designed to address the questions above, summarizing:

- What do we already know?

- What guidance can be derived from existing literature at the landscape scale?

- What is uncertain, and what research is needed to provide specific guidance for goal-setting at local, regional, and national scales?

Our vision of the agricultural landscape is one that supports viable populations of native species of plants and animals in functioning ecosystems — reversing the decline of threatened and endangered and at-risk species — and maintaining those species and communities whose numbers are stable or increasing. For the purposes of this paper, the working assumption is that an effective policy will require some combination of protecting intact natural areas from further conversion, restoring some cultivated areas to more natural vegetation and function, and implementing eco-friendly farming techniques. Biodiversity is recognized in law and policy as a public good, and the goal of conserving species and ecosystems has been widely embraced by our democratic society. Stopping the loss of species and taking steps to restore ecosystems and species at risk seems prudent given the fact that our understanding of ecosystem functioning remains very incomplete (Loreau et al. 2001). As Aldo Leopold wrote in *Sand County Almanac* (1949), "the first rule of intelligent tinkering is to keep every cog and wheel."

Recommendations and guidelines in this document are based on a synthesis of scientific literature and theory with respect to the goal of conserving existing biodiversity. Regional variation and the uniqueness of local context preclude one-size fits all recommendations.

The question of **How much is enough?** is as much a question of values, culture, and economic preferences as it is a question of science. What conservation values will shape policy and guidance? Do we want to restore biodiversity in natural landscapes, or do we only want to retain what we have now? Do we want to maintain and conserve wildlife populations at no less than the levels they are today, or do we want to restore declining populations and degraded ecosystems? Are we as a nation content to save mainly those common wildlife species compatible with agricultural land uses, such as deer and quail, or do we hope that future generations may once again experience the vanished wildlife, natural communities, landscapes, and ecological phenomena of the past, such as the thunderous migrations of the bison or skies darkened by flocks of migratory waterfowl? As the state of our scientific knowledge grows, we may learn much more about the value of biodiversity and its interrelated benefits. We may find that an even more dramatic transformation of our economic systems and personal choices is required than we have yet imagined.

This paper stops well short of a thorough analysis or synthesis of how such landscape goals should be achieved. The question of who pays or what policies should be adopted to achieve these goals is not addressed in this paper (see McNeely and Scheer 2003, Keeney and Kemp 2003, and Roling and Wagemakers 2003 for such a discussion). Inasmuch as the goal of conserving biodiversity benefits not just farmers and rural communities, but society as a whole, the attempt to achieve an ecological agriculture compatible with landscape-scale conservation should be viewed as a public good. Thus, it is not expected that farmers, rural communities, or any other specific group would be expected to bear the full cost or expense of biodiversity protection. The design of agriculture and food systems that are at once economically, socially and ecologically sustainable is a goal that society as a whole should embrace, pursue, and assume responsibility for achieving.

A LANDSCAPE TRANSFORMED

In 1949, Aldo Leopold envisioned a farm landscape 50 years into the future in which the production of crops and livestock mattered, as well as the provisioning of healthy soil, clean water, wildlife, biodiversity, and an aesthetically beautiful countryside. It was a future in which a nation of farmers felt as much pride in the health of the biotic community as they did in their per acre production totals. To Leopold, an agriculture was possible in which people were in harmony with the land, in which the farmer saw the land in its ecological context and managed it not just for economic production, but for the full range of essential services including habitat for wild pollinators, birds, bats and insects that fed on crop pests, wildlife for meat, pelts, and enjoyment; and woodlots for timber and shelter, as well as functions such as nutrient cycling, groundwater capture and recharge, and flood control. Leopold recognized that for both practical and moral reasons, government could provide no substitute for a stewardship ethic on the part of the private landowner.

Yet in many ways Leopold's vision of conservation agriculture has only receded in the intervening years. The early expansion of agriculture to open up new lands for a growing population with growing food needs and the intensification of production systems in the latter half of the 20th century in response to expanded world export markets has had a profound, largely negative, effect on natural and domesticated plant and animal biodiversity in North America as well as globally. Agricultural systems have been subject to a general loss of diversity among and within domesticated economic species. Farmers themselves are an increasingly endangered species, with fewer farmers every year and the average age of farmers increasing. More than 45 percent of the landscape is now managed by less than 2 percent of the farmer population (Natural Resource Conservation Service 1996).

OREGON: The Willamette Valley Alternative Futures Project examines a range of future landuse scenarios for impacts on ecosystem services such as: (a) agriculture; (b) the wood products industry; and (c) taxpayers and developers who bear the burden of costs for the infrastructure and services needed by new development.

http://www.econw.com/wvaf/about.html

Like Leopold, we as individuals and citizens have the choice and the obligation to envision the future that we desire and take steps towards that vision. Across the continent, community based ecosystem protection initiatives are seeking to do just that. In 2005, for example, stakeholders in Oregon envisioned a plan for

50 years into the future in which farms and agricultural lands were integrated into a healthy working landscape, one in which society explicitly valued all the biodiversity and ecological services provided by the patchwork of natural, grazing, agricultural, and forest lands (see link on page 11).

Exactly how this vision is to be achieved is largely an open question that will rely on the efforts and initiatives of thousands of individuals working at local, regional, and landscape scales. Clearly, however, significant changes in our current agricultural system will be required — both to reduce the role that agriculture plays in direct habitat loss and conversion, and to reduce the negative impacts of agriculture on neighboring ecosystems. This will involve a range of strategies including developing better systems and technologies for reducing agricultural inputs, reducing the impacts of inputs at both ends (environmental impacts of manufacture as well as runoff), and modifying agricultural lands so that they function within a landscape-scale matrix that is compatible with biodiversity protection on adjacent lands. Agricultural lands need to be managed as part of the matrix surrounding lands managed for conservation, while protected areas must be viewed within the context of surrounding agricultural lands (Pirot, Meynell, and Elder 2000). As one well-known conservationist put it: “Parks cannot conserve biodiversity in a world of shopping malls and soybean fields. Working landscapes can, and must, be managed for biodiversity as well as for resources” (Redford 1997).

ROLE OF AGRICULTURE IN BIODIVERSITY LOSS

North America has undergone two dramatic transformations leading to biodiversity loss. The first was the wave of extinctions and losses associated with European settlement and the conversion of the landscape to agriculture in the 19th century. The second major transformation was precipitated by demographic, technologic, macroeconomic, and policy changes in agriculture and the larger economic system in the second half of the 20th century.

The initial impact of European settlement and agriculture on biodiversity, in addition to its devastating impacts on Native American populations, cultures, and natural resource management systems, was severe and dramatic. The direct loss of many species and populations of plants and animals, landscapes, and ecological phenomena was driven by extensive overexploitation of the American continent's timber, wildlife, fish, and the clearing of land for agriculture. The bison were eliminated, fish and game populations were decimated, forests were clearcut, and large predators were eliminated. Streams and rivers were degraded by damming, channelization, movement of timber to markets, and massive soil erosion that accompanied land conversion. Settlers also brought with them exotic non-native plants, pasture grasses, and ornamental species that drastically altered North American ecosystems and led to the threatened status today of many native plants and animals (Mack et al. 2000). The overall impact has been to simplify and homogenize natural systems.

By the 1930s, the federal soil conservation service was established to address the problems brought on by massive soil erosion and cultivation of marginal lands. A network of national parks, public and private nature preserves had been established, and Aldo Leopold could envision a future of conservation farmers.

Instead, agriculture has taken a different trajectory. Driven by technological advances under the modernist scientific progress paradigm, the last century witnessed a second dramatic transformation of farming country (Jackson 2002). Energy, machinery, agrochemicals, and irrigation have nearly eclipsed land as the principal elements of agricultural production technology (Allen 1995). These changes have propelled the agricultural landscape of the American Midwest from a matrix of small privately owned farmsteads in a vast region of prairies, savannas, riverside forests and pothole wetlands to one dominated by intensive, high-input, high-yield, and high-impact monocultures (Jackson 2002). The farmland matrix envisioned by Leopold has become instead a wasteland when it comes to wildlife. In many regions, protected remnant natural areas represent a small archipelago of islands in an ocean of inhospitable monocultures. Most of the American Midwest is now managed for agricultural production with ever increasing intensity, dominated by a few genetic varieties of corn and soybeans (Jackson 2002). The grassy fencerows, hay meadows, and pastures have vanished as well. Fields are larger, crop rotations are simpler and less frequent, and agrochemicals play a major role in crop production. Equipment used for tillage, planting, application of agrochemicals, and harvesting, have led to soil compaction, depletion, and erosion, resulting in greater dependence on fertilizers and pesticides (Bender 1984). As farm animals have been replaced by machinery, livestock operations have instead been consolidated into large-scale operations involving concentration of animals and increased use of sub-therapeutic antibiotics and other chemicals. These changes have also resulted in surface and groundwater contamination (Ribaudo 1989). The volume of manure produced by animal confinement facilities is 130 times that produced by humans, and much of this reaches waterways and groundwater untreated (Imhoff 2003). Runoff from agricultural lands still carries thousands of tons of topsoil and sediment into the nation's waterways each year, along with a soup of nutrients, fertilizers, pesticides, herbicides, and other agrochemicals. The impacts on wildlife and biodiversity are profound, ranging from habitat elimination to long-term effects of agrochemicals on water quality and reproductive success of fish, birds, and other wildlife (Capel et al. 1993; Allen 1995).

Modern agriculture reaches deep into aquatic ecosystems in other ways as well. Radical changes in hydrology have resulted from widespread drainage and replacement of sponge-like prairie soils with plowed ground (Jackson 2002). Millions of acres have been tiled and drained, isolated wetlands and basins have been connected to existing stream networks, stream channels have been ditched and straightened, and rivers have been dammed and diverted and disconnected from their floodplains by extensive levees. Sediment, nitrogen, and phosphorus from the corn belt of the Midwest have given rise to a hypoxic Dead Zone in the Gulf of Mexico that covers up to 8500 square miles in the summer months. Hypoxia and coastal eutrophication due to agricultural and other sources of nonpoint pollution are common at the mouths of other rivers around North America. Alteration of nutrient cycles and disturbance regimes has shifted the competitive balance in many ecosystems and endangered a large proportion of native and endemic plants. The single minded pursuit of maximum production has driven the effort to harness rivers, drain wetlands, and farm every inch of available bottomland and floodplain, destroying millions of acres of valuable riverine, wetland, and aquatic habitat, and contributing to the long list of threatened and endangered aquatic species, while contributing to increasing flood damages (Hey and Phillippi 1995; Rasmussen 2003).

Because modern agriculture is primarily oriented towards maximizing a single variable — crop production — it has gradually shed its connection with and resemblance to natural ecosystems. Maximizing "efficiency" of production has come at the expense of other ecological services, whose valueis often poorly captured by markets or subsidies.

Modern food production systems are demonstrably unsustainable from the standpoint of inputs, water use, soil loss, and nonpoint source pollution (Trenbath et al. 1989). Modern agriculture is too dependent on fertilizers, chemicals, and inputs from the subsidized petroleum industry to maintain soil fertility and power extensive machinery, as well as too energy intensive in the transport and processing of agricultural commodities from farm to consumer. Rates of soil erosion, water withdrawals from groundwater and surface waters, as well as nonpoint source pollution from agriculture to surface and groundwater, are also unsustainable in many agricultural regions.

Because modern agriculture is primarily oriented towards maximizing a single variable — crop production — it has gradually shed its connection with and resemblance to natural ecosystems. Maximizing "efficiency" of production has come at the expense of other ecological services, whose value is often poorly captured by markets or subsidies. Increased production has been made possible only with the addition of inputs made economically feasible by subsidies and commodity payments in the form of cheap energy, subsidized transportation, consumptive water withdrawals, soil mining, and by exceeding the waste assimilation capacities of the nation's air, water, and fish and wildlife to recycle nutrients, sediment, and contaminants. The real costs to society of poor water quality, environmental contamination, and the loss of ecological services such as flood protection, fisheries and wildlife, natural amenities, and pollination are substantial (Costanza et. al. 1997).

"A just relation of peoples to the earth rests not on exploitation, but rather on conservation — not on the dissipation of resources, but rather on restoration of the productive powers of the land and on access to food and raw materials. If civilization is to avoid a long decline, like the one that has blighted North Africa and the Near East for 13 centuries, society must be born again out of an economy of exploitation into an economy of conservation."

— W.C. Lowdermilk 1939, 1st chief of the Soil Conservation Service.

Many of these costs impair the performance of agriculture itself. For example, biodiversity losses affect above and below ground biological systems that play a critical role in pollination, control of agricultural pests, breakdown of agricultural residues and wastes, and recycling of nutrients critical to plant growth. Ultimately, these cumulative costs may exceed the benefits derived from increased outputs under intensified production.

NOT ALL AGRICULTURE IS CREATED EQUAL — BENEFITS OF MORE ECOLOGICAL AGRICULTURE

If one teases apart these causes, it is not agriculture itself, but the way agriculture is practiced, in conjunction with other trends in society, that has led to declining biodiversity in North America. An emerging body of scholarly work argues that we should model agroecosystems on native ecosystems not just to conserve biodiversity, but to achieve economically and ecologically sustainable agricultural production (Soulé and Piper 1992).

Prior to European settlement, the Native American farmers had developed agricultural and natural resource management systems that were based on limited modification of natural ecosystems. These farming practices were thus compatible with the biodiversity European settlers initially encountered. In parts of North America, Native Americans had continuously cultivated maize, beans, squash, and other crops for more than five thousand years (Altieri 1989). In other regions, native people intentionally managed the prairie and other natural ecosystems by mimicking natural disturbance patterns on the landscape, such as setting fires to manage berry crops and maintain and modify prairie for wildlife habitat.
For thousands of years, agriculture has involved modification of natural habitats and ecosystems to produce food for human consumption. Farming systems have evolved from close observation of natural cycles since

agriculture began some 12,000 years ago (Thrupp 1997). In places where native peoples may have lived continuously, humans have adapted to environmental change, and ecosystems have evolved in concert with human natural resource management systems. Such systems appear to have been sustainable over long periods, or at least over relevant and discrete spatial and temporal scales (Nabhan 1998).

Yet history is also filled with examples of societies that have collapsed and dispersed, possibly because they failed to prevent land degradation and associated internally induced changes brought on by their own management systems or by external environmental change (Hillel 1991; Redman 1999; McNeely and Scherr 2003).

Agriculture reduces native biodiversity to the extent that:

- Agriculture alters the structure, function, community composition, and habitat value of the ecosystems it replaces; and

- Agriculture degrades the structure, function, and habitat value of adjacent or downstream ecosystems.

However, in landscapes that have been transformed by roads, urban, residential, commercial and industrial development, farmland and other blocks of less disturbed habitat may help to conserve the integrity and ecological function (i.e. health) of natural communities in remnant reserves. These semi-natural areas in agricultural landscapes, particularly the smaller, less intensively managed farms in the patchwork envisioned by Leopold, function as refuges, “stepping stones” and corridors for migratory birds, insect pollinators, and other wildlife.

STATUS AND TRENDS BY HABITAT TYPE

In recent years, there have been numerous large-scale attempts to conduct a comprehensive assessment of status and trends of biodiversity in the United States. These include comprehensive reviews by Flather and Hoekstra (1989), *Our Living Resources* (LaRoe, ed. 1995) for the U.S. Fish and Wildlife Service, *Our Precious Heritage* by The Nature Conservancy (2000), and *The State of the Nation's Ecosystems* by the Heinz Center for Science, Economics, and the Environment (2002). Land use and land cover patterns provide a coarse description of the amounts and quality of habitat needed by fish and wildlife (Flather and Hoekstra 1989).

CROPLANDS

A recent assessment of the State of the Nation's Ecosystems estimated that about 25 percent of the farmland landscape in the Midwest and 40 to 50 percent elsewhere is non-cropland: forest, wetlands, grassland or shrubland (Heinz Center 2002). Most farmland species are associated with these remnant habitat types that farmlands replaced, remaining in the area and taking advantage of these smaller habitat patches, but present at lower population levels than prior to land conversion. Some species actually favor the kinds of conditions found in areas with extensive farmlands. These species have become in many cases much more common than they were before conversion to agriculture. These include species associated with particular crops as well as species inhabiting edges, hedgerows, woodlots, field margins, and pastures. Generalist species such as white-tailed deer, raccoons, and pheasants (the latter introduced from China) are far more abundant than at settlement, and in many cases pose a biodiversity threat to native species resulting from herbivory, predation, or competition. Row crop agriculture, particularly large fields in monocultures of corn, soybeans, and other major commodities, generally provides attractive or suitable habitat for the least number and diversity of species.

The past 50 to 100 years have brought substantial changes in the mix of cropland and pastureland. These changes have been dramatic at the farm scale. The expansion of cropland relative to hay and pasture has been accompanied by more intensive farming practices, including increases in farm size, and a reduction in shelterbelts, field borders, and odd habitat areas that were inconvenient to farm.

Over the past 50 to 100 years, there have been substantial changes in the mix of cropland and pastureland. These changes have been particularly dramatic at the farm scale. The expansion of cropland relative to hay and pasture has been accompanied by more intensive farming practices, including increases in farm size, and a reduction in shelterbelts, field borders, and odd habitat areas that were previously inconvenient to farm. Fencerow to fencerow farming has eliminated much nesting, feeding, and winter cover for wildlife. As both row crop and livestock operations have become increasingly concentrated and specialized, the connection between them has been lost, leading to increased leakage of nutrients and chemicals to natural systems. Livestock waste is no longer directly recycled into functioning soil eco-systems to fertilize crops; meanwhile, off-farm nutrient inputs are required in the form of purchased feed for concentrated livestock or fertilizer for feed crops.

FOREST LANDS

At the time of European settlement, it is estimated that 46 percent of the United States land mass was forested (compared to 33 percent today). The initial wave of settlers moving west was accompanied by extensive timber harvest, and much of the land cleared in this fashion was then settled for agriculture. On steep slopes, poor soils, in northern regions, or other marginal areas for agriculture production, many of these initially cleared forest lands were ultimately abandoned, and reverted back to second growth forest. However, the composition of North American forests was forever and drastically altered. Besides the almost total disappearance of the American chestnut, which was once among the most abundant canopy trees in eastern United States and a key food source for an array of wildlife species (Greller 1988), there have been significant changes in forest composition, including a general shift towards earlier successional forest types and declines in a number of forest types, especially old growth, pine, oak savanna and other fire-dependent forest communities. Bryer et al. (2000) concluded that 58 percent of the conterminous United States no longer supports natural vegetation. Mature and old-growth softwood stands are becoming increasingly rare in the major timber producing regions of the Pacific Northwest and southeastern United States. The oak woodlands of the west coast are also critically threatened, having declined by more than 99 percent. Although the total acreage of cropland is not expanding greatly relative to natural lands, there are some alarming exceptions. In western states (OR, WA, CA) oak woodlands continue to be converted to vineyards, in some cases organic vineyards, because these lands are more likely to be free of chemicals upon conversion (Merenlender 2000, Heaton and Merenlender 2000, Hilty and Merenlender 2003).

GRASSLANDS

The central North American prairie, as with the temperate grassland biomes in general, is one of the most endangered ecosystems on earth. Once the greatest grassland on Earth, covering a quarter of the continental U.S. along with portions of southern Canada and northern Mexico, less than one percent of the native prairie of North America's Great Plains remains undisturbed by the plow (Klopatek et al. 1979; Samson and Knopf 1994).

A key habitat element in the grasslands of the midwest was the prairie potholes (Tiner 2003). These wetlands, formed by the melting and receding of glaciers in the intervals between ice ages, generally ranged in size from 1/5th of an acre to 25 acres. Although the pothole region comprises only 10 percent of North America's waterfowl breeding areas, it produces over half of the continent's waterfowl. Besides serving as breeding and watering sites for wildlife, potholes collect rain and flood waters, recharging groundwater or releasing waters slowly to surface water outlets. Yet it is estimated that more than 90 percent of the potholes have been drained for agriculture. Isolated wetland basins once unconnected to surface water networks have been ditched and channeled into rivers and streams (Tiner 2003), and the proportion of water cycled directly into the atmosphere through evapotranspiration has decreased, increasing peak flows and effective discharge in prairie streams leading to in stream and stream bank erosion. In Iowa, for example, as much as 300 miles of streams and small rivers had been eliminated by straightening by 1975 (Bulkley 1975). The great increase in row crops and the increased use of subsurface drainage accompanying agricultural intensification in the latter half of the 20th century further altered ecosystem dynamics of prairie grasslands. It is estimated that by 1985 at least 30 million hectares of cropland in the U.S. (roughly 15 percent) and 14 million hectares of pasture, range, and forestland, had been extensively artificially drained by subsurface tile drains and/or surface drainage ditches (Skaggs et al. 1994). These alterations of the hydrologic cycle have had significant hydrologic and water quality impacts on stream, river, wetland, and coastal ecological communities.

Fire, once an important element in forming and maintaining the prairie, has also been eliminated across much of the landscape, resulting in major shifts in vegetation composition. In many parts of the Great Plains, fire suppression in combination with water management practices and the planting of farm and ranch shelterbelts has resulted in encroachment of trees into grasslands habitat (Willson 1995). Far more woody plants and forest patches are present than before European settlement. Expansion of agriculture and resulting ecological changes have benefited some species at the expense of others. Endemic grassland birds have declined, replaced by eastern forest species moving into newly wooded habitats. Isolated trees provide perches for raptors, increasing predation on grassland species, while expanding the amount of edge habitats benefiting generalist species.

Fragmentation of native grassland has eliminated keystone species such as bison, white-tailed prairie dog, and wolf. The original plowing of the prairie from the 1850s to the 1890s eradicated prairie plants and animals with large area requirements (Smith 1992). These changes shifted the balance of ecological relationships, drastically altering plant and animal communities, toward species that benefit from agricultural disturbance and edge habitat. As top predators such as wolves, mountain lions, and grizzlies have been eliminated, meso-scale predators such as skunks, coyotes, raccoons, and opossums, etc have become both more abundant and more geographically widespread. Predation from these now more common meso-scale predators — in combination with the decline of prairie pothole wetland and grassland habitats — is perhaps the primary factor behind the decline of many grassland species. Extensive areas of cropland have likely resulted in the loss of habitat for the small mammals that form the base of the food web in grassland systems (Hunt 1993; Marzluff et al. 1997). Currentagricultural practices are contributing to population declines of some farmland grassland birds (Koford and Best 1996; Murphy 2003), although not necessarily to the exclusion of factors acting during migration and wintering (Rodenhouse et at. 1993, 1995; Herkert 1994, 1995; Warner 1994).

At first glance, grazing would appear more compatible with the mimicking of ecosystem structure and function in grasslands than in other biomes. Most prairie mammals and birds are adapted to moderate levels of grazing, and some species of ground squirrels may persist even in heavily-grazed grasslands (Bradley and Wallis 1996; Michener 1996). However, the grasslands of North America where massive herds of bison, elk, and antelope once roamed were the first to yield to the plow, and were converted overwhelmingly to cropland. Much of the land we consider rangeland now consists of lands in the arid west and southwest

TABLE 1. Eight examples of 27 ecosystem types that have declined by 98% or more in the U.S. since European settlement (Noss et al. 1995)

1.	Old-growth forest in eastern deciduous forest biome
2.	Spruce-fir forest in the southern Appalachians
3.	Longleaf pine forests and savannas of the southeastern coastal plain
4.	Oak savanna in the Midwest
5.	Streams in the Mississippi alluvial plain
6.	Native grasslands (all types) in California
7.	Prairies (all types) and oak savannas in the Willamette Valley and in the foothills of the Coast Range, Oregon
8.	Tallgrass prairie east of the Missouri River and on mesic sites across range

Source: Bryer et al. 2000

that were too dry or marginal for cropland agriculture. Livestock densities on many of these arid lands have been far higher than the density of native ungulate grazers. Overgrazing combined with fire suppression — particularly in arid lands — reduces the plant cover and the food resources necessary for whole suites of plant and animal species (Platt 1974), and can result in irreversible ecological changes by driving the conversion of grasslands into woody shrublands. Cattle also have more severe impacts on riparian systems in arid regions, trampling riparian vegetation and exacerbating stream bank erosion. As a result, the majority of non-federal rangelands are in fair to poor condition (Flather and Hoekstra 1989). Although range condition appears to be improving with better management, ongoing concerns include the loss and fragmentation of grassland habitats in the east, degradation of riparian habitats in the arid west, and shifts in rangeland condition to mesquite shrublands due to overgrazing in the southwest (National Research Council 1994).

In spite, or perhaps because, grassland systems have been so dramatically altered, the potential for ecological agriculture to function in a restorative capacity is perhaps greater than in many other systems.

Converting cropland back into pasture-based rotational grazing systems is proving to be beneficial not just for grassland species and soil health, but a profitable alternative strategy for farmers feeling pinched by the ever increasing input costs of conventional commodity production. Activities such as ranching may help protect the prairie against fragmentation because ranchers prefer large blocks of pasture land to graze cattle (Paton 2002). There is an evolving literature on landscape-scale approaches to grazing designed to restore grassland ecosystems (Jackson 2002; Soule 2002; Fuhlendorf and Engle 2001).

WETLANDS

Dahl (1990) estimated that more than half of the wetlands present at settlement in the conterminous U.S. have been drained or converted to agriculture since 1790. Although wetlands account for just 5 percent of the total land area in the U.S., distribution of hydric soils on county soils maps suggests that in some regions as much as 80 percent of today's cropped land was originally wetland (Jackson 2002). Wetland area has continued to decline significantly over the past several decades despite aggressive national and state policy attempts to prevent additional losses. From 1954 to 1974, forested wetlands declined by nearly 11 percent, emergent wetlands declined by 14 percent, and estuarine wetlands declined by 6.5 percent. Since 1980, net wetlands losses have declined by 80 percent, but these are disproportionately distributed. Wetlands in agricultural landscapes and on the urban-suburban fringe continue to be eliminated by drainage and development (Dahl 2000).

Fire, once an important element in forming and maintaining the prairie, has also been eliminated across much of the landscape, resulting in major shifts in vegetation composition. In many parts of the Great Plains, fire suppression in combination with water management practices and the planting of farm and ranch shelterbelts has resulted in encroachment of trees into grasslands habitat.

AQUATIC ECOSYSTEMS

Regional studies indicate that the capacity of the nation's waters to support warm and coldwater fisheries has declined, primarily due to degradation of aquatic habitat and introductions of competing fish species. About 80 percent of the nation's flowing waters have problems with quantity, quality, fish habitat, or fish community composition (Imhoff 2003). A 1997 assessment of aquatic biodiversity concluded that the three leading threats to aquatic species nationwide were agricultural nonpoint source pollution, alien species, and altered hydrologic regimes due to dams, impoundments, and land use change (Richter et al. 1997). There were important geographic differences to the nature of threats, with nonpoint pollution dominating with alien species, habitat degradation, and altered hydrology together constituting the main threat in the West. Soil erosion from agricultural lands leads to mortality from turbidity, while sedimentation of aquatic substrates degrades macroinvertebrate communities and interferes with fish spawning and rearing. Nutrient inputs have drastically altered in stream and in-lake trophic pathways, altering aquatic plant and animal communities and resulting in seasonally anoxic or hypoxic conditions. Stream thermal regimes are also affected by irrigation withdrawals, changes in basin water yields, and altered riparian vegetation. Dams and water level management for agriculture and barge transportation result in barriers to fish movements and threats to emergent and coastal zone vegetation communities. For mussel species in particular, for which North America is a global hotspot for endemism and diversity, a major threat is river level management and maintenance of barge navigation channels on the major river systems.

A 2000 national assessment of the state of the nation's ecosystems found substantial water quality problems nationwide (Heinz Center 2002). Nitrate concentrations exceeded federal drinking water standards (10 parts per million) for 20 percent of groundwater wells and 10 percent of stream sites tested nationwide, with the highest rates in farmland regions. Three-fourths of farmland stream sites had phosphorus concentrations exceeding the level recommended by Environmental Protection Agency to protect against excess algae growth. Groundwater samples from areas dominated by agricultural use had higher concentrations of nitrate than urban or forested areas, with forested lands having the lowest of the three. Only 3 percent of urban groundwater sites had nitrate concentrations above the 10 parts per million federal drinking water standard. Sedimentation of aquatic habitats and pervasively detected trace levels of pesticides were also of serious concern.

STATUS AND TRENDS OF NORTH AMERICAN BIODIVERSITY BY ECOLOGICAL GUILD

An estimated 20 percent of all losses of honeybee colonies involve some degree of pesticide exposure, despite the fact that honeybees, unlike their wild counterparts, are often removed by beekeepers from fields during periods of pesticide application. Pollinators remaining in small fragments of natural habitat in arid landscapes are particularly susceptible to insecticides.

OVERALL TRENDS:

- In the United States, 330 vertebrate species are listed as federally threatened or endangered, an increase of 130 species since the 1980 national assessment of wildlife and fish. In addition, there are approximately 1,000 candidate plant and animal species for which the U.S. Fish and Wildlife Service has sufficient information to initiate formal listing procedures. Additional species are listed as endangered or threatened in Mexico and Canada (Kerr and Cihlar 2004; COSEWIC 2002).

- The majority of breeding bird populations has remained stable since the mid-1960s. However, a significant subset (13 percent) of the breeding bird fauna has declined. The number of breeding bird species that have shown recent population declines are higher in the east than the west. Grassland birds have been in decline since the 1960s (Jackson et al. 1996).

- Migratory bird populations, except geese, have also generally declined. Breeding duck populations declined from 44 million in the early 1970s to about 30 million birds in the mid-1980s. Estimates for some species of breeding ducks were at or near record low levels in the early 1990s (Austin 1998). Populations of most species common in the prairie pothole region (PPR) have since recovered to levels near or above the population goals of the North American Waterfowl Management Plan (United States Fish and Wildlife Service 1998).

- There are no data comparable to the Breeding Bird surveys to track small mammals, amphibians, reptiles, and invertebrates (Jackson 2002). Trends in mammal populations vary. Population trends are divergent for agriculture and forest species. Those small game species associated with agricultural lands have shown significant declines over the last 20 years, while most woodland populations have remained stable. Big game species across all regions have increased, except for deer species of the Pacific coast. Populations of the two most commonly hunted big game species, white-tailed deer and wild turkey, have more than doubled. White-tailed deer in many regions, in fact, represent a significant threat to forest regeneration as well as protection and restoration of native plant communities.

NATIVE PLANT COMMUNITIES

Although existing reserves and protected areas may be adequate to protect most plant species based on area requirements alone, fragmentation poses significant threats especially in the context of climate change. Plants usually have low dispersal mobility (Lennartsson 2002). Fragmentation changes interactions between plants and more mobile organisms with which they have co-evolved (Lennartsson 2002), especially pollinators (Kearns et al. 1998), seed vectors (Santos et al. 1999), and herbivores and seed predators (Jennersten and Nilsson 1993; Zabel and Tscharntke 1998). Several studies have demonstrated effects of fragmentation on patch occupancy of insect populations (Hanski et al. 1995), in particular on specialist species (Zabel and Tscharntke 1998; Kruess and Tscharntke 2000) and on species with restricted mobility (Thomas 2000). An examination of rates of tree migrations in response to historical climate change suggested that a large proportion of plant diversity is at risk of extinction as a consequence of global warming (Schwartz 2003).

INSECTS AND POLLINATORS

More than 90,000 described insect species inhabit North America (Mason, Jr. 1995; Hodges and Powell 1995). This includes 4000 species of native bees (Vaughn et al. 2004), and 11,500 species of butterflies and moths (Powell 1995). Largely due to their great abundance and diversity, comprehensive data on the status of invertebrates, including insects and other arthropods, is relatively spotty. There are few comprehensive assessments in the academic literature (Mason, Jr. 1995; Master et al. 2000). Despite the lack of comprehensive data, insects are acknowledged to play a critical role in pollination of plants in agricultural and natural ecosystems (Buchmann and Nabhan 1996). More than 100 crops in North America require insect pollinators in order to be productive. Even crops that self pollinate, such as tomatoes, peppers and eggplants, often produce more, larger, or higher-quality fruit when cross-pollinated by insects. Although the introduced European honeybee gets most of the credit for this service, native bees are also important pollinators.

Unfortunately, both wild and managed pollinators are disappearing at alarming rates owing to habitat loss, pesticide poisoning, diseases and pests. North American feral and domestic honeybees have undergone drastic population declines over the past decade in the wake of the introduction of the Varoa mite, an introduced parasite (Watanabe 1994; Loper 1995). In 1994, local bee shortages for the first time forced many California almond growers to import the bulk of the honey bees they needed to pollinate their $800 million per year almond crop (Ingram 1996).

Many crops that would benefit in quality and quantity from more thorough pollination are not sufficiently pollinated because of heavy pesticide applications. A large number of insecticides used in agriculture are toxic to pollinating insects, and the U.S. now applies twice the amount it used when Rachel Carson published Silent Spring in 1962 (Curtas and Profeta 1993). Agrochemicals also eliminate nectar sources for pollinators, destroy host plants for larval moths and butterflies, and deplete nesting materials for bees (Buchmann and Nabhan 1996). An estimated 20 percent of all losses of honeybee colonies (at a cost of at least $13 million per year) involve some degree of pesticide exposure, despite the fact that honeybees, unlike their wild counterparts, are often removed by beekeepers from fields during periods of pesticide application (Pimentel et al. 1992). Pollinators remaining in small fragments of natural habitat in arid landscapes are particularly susceptible to insecticide spraying on adjacent croplands (Suzan et al. 1994). Aerial spraying of coniferous forest pests in Canada in the early 1970s reduced native bee populations to the point that blueberry yields fell measurably (Kevan 1975).

Natural areas that once served as a source of new pollinators after insecticide applications killed resident insects and bees have been eliminated. Many agricultural landscapes lack sufficient natural habitat to support native pollinators (Ingram et al. 1996).

WILDLIFE

Status and trends in populations, harvests, and uses of wildlife and fish resources are closely linked to habitat trends. Although trends vary by guilds, those species associated with agricultural, mature and old-growth forest, native grassland, and wetland habitats have had declining or unstable populations. The diversity of wildlife habitats on farms has continued to decline, first large mammals and birds, then those species with smaller and smaller area requirements (Dinsmore 1994).

BIRDS

The biodiversity status of birds is known in somewhat greater detail due to their widespread conservation appeal, as evidenced by participation in monitoring and assessment efforts including the North American Breeding Bird Survey and Audubon Christmas Bird Counts. Nevertheless, estimates of changes in bird abundance are uncertain because there were no widespread, systematic surveys of bird populations before the mid-1960s.

NEOTROPICAL MIGRATORY BIRDS. In forested landscapes that had been compromised by extensive agricultural corridors, Ford et al. (2001) documented increased nest predation and brood parasitism by native brown-headed cowbirds at the expense of neotropical migrant birds. Murphy (2003) concluded that avian population trends are linked strongly to changes in agricultural land use such as relative area of Conservation Reserve Program, cropland, and shrubland — regardless of migratory behavior or nesting habits. Thirty-eight neotropical migratory birds, which form the dominant part of midwestern farmland bird communities, are common in at least one farmland habitat in the midwest, and two-thirds of the common native species are neotropical migratory birds (Johnson 1996). The conversion of midwestern forest, savanna, and prairie to agriculture, mostly prior to 1920, altered the abundance and distribution of neotropical migratory birds (Johnson 1996). This included extinction and local extirpation of some species (Johnson 1996).

GRASSLAND BIRDS. Grassland birds form a large fraction of the common farmland neotropical migratory birds, and probably did not decline much between the 1920's and 1950's when diversified farming was common (Koford and Best 1996). However, more recently they have shown steeper, more consistent, and more widespread declines than any other avian group including neotropical migrants (Knopf 1994). Declines are the result of cumulative factors. Population declines of grassland birds coincide with the more recent era of intensive farming, and appear to have continued in recent years (Koford and Best 1996). Declines could be due mainly to loss of habitat (Rodenhouse et al. 1995) or to a combination of loss of habitat and degradation of habitat (e.g., low nest success or survival rates in attractive habitats and landscapes). Agricultural habitats that appear to have declined the most are pastureland and strip cover. Agricultural management practices likely have significant impacts on whether agricultural lands function as sources or sinks for species which find such habitats attractive.

WATERFOWL AND WETLAND DEPENDENT BIRDS. Since the 1970s, numbers of some waterfowl species such as mallard (*Anas platyrhynchos*), blue-winged teal (*A. discors*), and northern pintail (*A. acuta*) have reached or nearly reached the lowest ever recorded (Schaffer and Newton 1995). Population trends among breeding ducks tend to reflect availability of ponded wetlands in the Prairie Potholes region in the spring (Austin 1998, Sovada et al 2002). Low nest success (the proportion of nests in which one or more eggs hatch) in key breeding areas, including the U.S. Prairie Pothole region, is partly responsible for declines in duck numbers (Klett et al. 1988; Johnson et al. 1992). High levels of predation may limit recruitment and inhibit the recovery of duck populations when habitat conditions are favorable (Greenwood et al. 1995, Sovada et al 2002, Johnson et al. 2005).

Newton and Schaffer (1995) reported that nest success for five species of ducks studied was and probably still is too low to maintain stable numbers of breeding ducks in most areas of the Prairie Pothole region. Except for pintails, whose nest success generally increased, they found no consistent trends in nest success across periods. In Canada, where more than two-thirds of the Prairie Pothole region is located, Greenwood et al. (1987) found mallard nest success averaged 12 percent and rarely exceeded 15 percent from 1982 to 1987. They concluded nest success in much of prairie Canada during the study period was too low to maintain stable numbers of breeding mallards. Predators caused most nest failures in both studies. Common predator species such as red fox (*Vulpes vulpes*), striped skunk (*Mephitis mephitis*), and raccoon (Procyon lotor) are important predators of waterfowl nests. Red foxes also take many female ducks during the breeding season (Sargeant et al. 1984; Sargeant et al. 1993). Newton and Schaffer (1995) reported that nest success in central South Dakota was much higher than in other regions for a period for which data were available.

Grassland birds have shown steeper, more consistent, and widespread declines than any other avian group including neotropical migrants. Declines could be due mainly to loss of habitat or to a combination of loss of habitat and degradation of habitat (e.g., low nest success or survival rates in attractive habitats and landscapes).

This region probably contributed a “surplus” of ducks that helped compensate for overall declines. Research is needed on landscape configuration and ecological characteristics of areas where waterfowl numbers are stable and increasing.

SHORELAND / COASTAL BIRDS. The major threat to shoreland and coastal bird species is conversion and degradation of habitat. Many staging and wintering areas along the U.S. coasts have been degraded or eliminated by development since 1800. In Mexico, recent development has created a new threat to important shorebird sites on the west coast of that country. The tendency of many species to flock during migration and wintering makes them vulnerable to stochastic events, including both human induced and natural catastrophes.

LARGE PREDATORS

The native large carnivores of North America, namely grizzly bears (*Ursus arctos*) and wolves (*Canis lupus*), were extirpated across most of their original range by the early 1900s. The loss of wolves that likely contributed to the abundance of meso-scale predators (skunks, raccoons, coyotes) is hypothesized to be a factor in the decline of many ground nesting birds and small mammals. Berger et al. (2001) described severe impacts of moose herbivory (browsing) on the structure and density of riparian vegetation, and corresponding declines in songbird richness and density, in areas missing their native large carnivores, grizzly bears and wolves, as contrasted with matched sites where such carnivores were present or hunting was permitted.

Many landscapes that currently provide sufficient habitat for grizzly bears and wolves are subject to thresholds beyond which they would no longer be large enough to provide acceptable habitat for long-term population persistence (COSEWIC 1996). Active cooperation between government, industry and the public will be required to support ongoing research and to understand the implications of monitoring results for coordinated land use.

The restoration of predators to farmland landscapes has been highly controversial. In areas where wolves are returning, some success is being achieved with non-lethal approaches to predator management, including relocation, compensation for livestock damage, modification of farm systems, and extensive education and outreach.

Resistance to predator restoration may be reduced. Wolves persist in areas with high human density in some regions of Europe and Asia where human attitudes and cultures differ significantly from the United States (Harrison and Chapin 1998).

Status and Trends resources:

Our Living Resources, *LaRoe, ed.*

http://biology.usgs.gov/s+t/

The State of the Nation's Ecosystems

http://www.heinzctr.org/ecosystems/

GUIDELINES AND RECOMMENDATIONS FOR LANDSCAPE SCALE ECOLOGICAL AGRICULTURE: HOW MUCH IS ENOUGH?

What proportion of the landscape should be devoted to natural areas? What portion in some form of conservation-based, ecological, or environmentally friendly agriculture? The answer depends very much on the type of conservation-based agriculture (how compatible it is with the native species present), the condition and use of the remaining lands in the region, the status and viability of remaining biodiversity regionally, landscape and land use history, and ultimate conservation goals.

Over the past 40 years, the scientific community has devoted much research to understanding the impacts of fragmentation and habitat loss on biodiversity — including the influence of habitat patch size, shape, configuration, quality, connectivity, and landscape configuration. For the vast majority of species, including most of those that are listed as threatened or endangered, we lack the detailed knowledge to quantify how population viability will respond to changes in habitat quantity and quality. However, all else being equal, the chances of maintaining a species increases as the size and number of habitat patches increases (Groves et al. 2003). Considerable benefits for biodiversity protection have been achieved through habitat protection and management on public lands and in nature preserves (especially in the short-term), in conjunction with restrictions on harvest and consumptive uses (e.g., hunting and fishing laws).

Many scholars in the fields of island biogeography, conservation biology, landscape ecology, as well as a range of social sciences have concluded that for many reasons, simply creating "reserves" where most human uses and activities are forbidden is an untenable strategy for long-term biodiversity preservation. The largest, most well-managed preserves may be adequate to conserve plants and small to medium sized terrestrial animals, but these are the exception (Jackson 2002). Island biogeography theory suggests that even if biodiversity represented in existing preserves were securely conserved, 30 to 50 percent of existing species would be lost (MacArthur and Wilson 1967). In North America, the majority of natural lands that have already come under protection are disproportionately located in northern regions, mountainous regions, and other lands marginal for human economic

Less than 10 percent of land area is under legal protection and many of these preserves are inadequate because they are too small or too isolated to effectively protect the full complement of species represented, or they exist on paper only. In North America, national parks and other conservation reserves cover less than 8 percent of the land area, and are adequate to sustain only a fraction of total biodiversity.

uses, not necessarily in those areas harboring the greatest biodiversity. Land ownership patterns show that areas of lower elevation and more productive soils are most often privately owned and already extensively converted to urban and agricultural uses (Scott et al. 2001). Globally, less than 10 percent of land area is under legal protection, and many of these preserves are inadequate because they are either too small or too isolated to effectively protect the full complement of species represented, or they exist on paper only. In North America, national parks and other conservation reserves cover less than 8 percent of the North American land area, and are adequate to sustain only a fraction of total biodiversity (Reid and Miller 1989). For example, 40 percent of known plant communities have less than 10 percent of their area protected by nature reserves (Groves et al. 2000).

For an index of The Nature Conservancy's ecoregional plans, search the online directory at http://www.ConserveOnline.org.

Most nature reserves are too small or isolated to sustain whole suites of organisms, from top predators to migratory species to species that rely on patchy resources (Diamond 1976; Shafer 1990). The vast human-altered landscapes separating reserves serve as barriers to dispersal and create isolated and fragmented populations of native plants and animals. Small isolated nature preserves are "living dead" (Jackson 2002). Small, poorly dispersing populations are vulnerable to genetic isolation and catastrophic disturbances. Rare plants may fail to reproduce due to lack of pollinators, and wildlife may fail to find suitable mates. For example, the federally threatened eastern prairie fringed orchid relies entirely on sphinx moths for pollination. Yet in order to cross-pollinate small patches of orchids on isolated remnant prairies, these moths must often cross many miles of insecticide treated fields (Jackson 2002).

Furthermore, reserves are often too small or too constrained by neighboring land uses to allow crucial natural renewal processes and disturbances such as migration, fire, and flooding to be restored at a functional scale. Very few are designed explicitly for the protection of aquatic species and ecosystems.

Conservation increasingly must be planned and implemented at larger scales — from ecosystems and landscapes to entire regions. Chaplin et al. (2003), using a computer model algorithm, estimated that the absolute minimum area required to encompass all conservation targets (focusing on 2800 imperiled species) could be theoretically achieved with a network of conservation lands representing 6 percent of the U.S. land area, while a portfolio of small watersheds representing 15 percent of small watershed areas in the lower 48 could represent much of the nation's aquatic diversity. Using a different methodology, Dobson et al. (1997) estimated that 14 percent of U.S. land area would be required to ensure representation of all federally listed species. None of these methods, however, fully took into account long-term population viability, dispersal and metapopulation dynamics, interdependence of ecological communities, maintenance of historical disturbance regimes, the need to conserve multiple metapopulations, or other elements of conservation design. Large-scale habitat and ecoregional conservation planning efforts in the U.S. typically call for 15-30% of a landscape to be protected (Stein et al. 2000, Shaffer 2002). Internationally, conservation organizations typically strive for a rough minimum guideline of 10-12% of each ecoregion or ecosystem type (James et al. 2001). However, many conservation scientists readily admit that this goal has as much basis in political realism as in rigorous scientific analysis, and that protection of only 10% of Earth's ecosystems could make at least half of all terrestrial species vulnerable to anthropogenic extinction in the near future (Soule and Sanjayan 1998). Extrapolating based on a handful of other assessments using similarly coarse scale filters (e.g., Cox 1994, The Nature Conservancy 1998), Shaffer et al. (2002) estimated that as much as 25% of the land in the coterminous U.S., of which roughly half is in private ownership, would be needed for an adequate system of habitat conservation areas.

It should be noted, however, that such analyses presume a worst case scenario, in which lands outside of the network are virtually worthless from the standpoint of biodiversity. Despite near-term constraints on how much land can be devoted to nature conservation exclusively, semi-natural lands in the buffer provide some level of protection.

There are substantial sociopolitical implications of having large percentages of land owned and managed by nonresident public or private conservation owners. Many of these conflicts and processes are only beginning to play out across the country. Local units of government are often under pressure to limit land acquisition and ownership by conservation buyers. Numerous counties and municipalities have in fact voted to restrict public or private conservation land ownership or agricultural land retirement. Although opposition is often framed in terms of the property tax implications, it often masks more fundamental differences in cultural, social, and political values.

CONSIDER FARMLAND WITHIN A WATERSHED AND ECOSYSTEM CONTEXT

As unprotected lands are further fragmented by incompatible development and intensification, the value of surrounding landscapes as buffers with partial ecosystem function becomes more obvious. Linked to larger areas within the matrix and/or embedded in semi-natural buffer zones, existing and expanded reserves are more likely to succeed in ensuring the survival of native species (Anderson and Bernstein 2003).

To arrest and reverse biodiversity declines, some combination of natural area management buffered by and in conjunction with conservation-based working lands will be needed. Many existing agricultural lands serve to provide ecological services that have largely not been inventoried or adequately recognized for the value they provide to local economies and landscapes. The composition and spatial structure of agricultural landscapes influences their habitat value. Attempts should be made to inventory and appropriately value the functions of these lands so as to avoid continued biodiversity loss due to intensified management of farmlands.

Some species' habitat needs may be incompatible with all but the minimal forms of human use and disturbance. These include large carnivores and other animals with very large home ranges, and species that generate human-wildlife conflict such as top predators. Such species, like wolves, do best in the least disturbed, most intact natural areas. For conservation of such species, recommendations for the amount of natural vegetation and ecological agriculture in the landscape are available in various alternative ecoregional planning efforts such as those pursued by The Nature Conservancy and state and federal conservation authorities. Such planning efforts continue to develop target-specific guidance for adequate size, shape, and landscape configuration of conservation areas, as well as to identify regionally appropriate forms of agriculture and management practices that are compatible with biodiversity conservation.

DESIGN AGRICULTURAL SYSTEMS TO MIMIC SCALE AND FUNCTION OF KEY ECOLOGICAL PROCESSES.

Agricultural practices designed to be compatible with conservation of natural communities should as much as possible mimic the structure and function of the natural ecosystem in its ecoregional context (Lefroy et al. 1999). For example, a given configuration of large cleared fields and pastures is likely to have a greater impact on native ecological communities in historically forested ecosystems than a similar patchwork of farmlands in historically grassland ecosystems. Practices designed to accomplish some conservation objectives in the absence of attention to ecological context may be detrimental to the achievement of others. For example, windbreaks implemented to reduce wind erosion may provide perches for forest raptors that prey on grassland birds or prairie species, or serve as a source of woody invasive shrubs into prairie areas. Clearly, one size does not fit all. Best practices must be customized based on observation and local context, and adaptive management is needed to ensure that practices designed to maintain biodiversity are achieving the desired or expected results.

CONNECTIVITY

Because connectivity of habitats is essential for daily and seasonal movements of animals throughout their life history, for example between feeding and nesting habitats, as well as for genetic exchange among metapopulations, much conservation literature calls for the establishment and protection of critical habitat corridors and/or "stepping stones" for birds and other animals that can move short distances across unsuitable habitats.

PROTECT REMAINING NATURAL HABITATS IN AGRCULTURAL LANDSCAPES. Nearly all science-based conservation plans and peer-reviewed studies highlight the importance of conserving remaining natural habitats — prairie, savannah, old-growth forest, functioning wetlands in most cases as many places and in patches as large as possible (Kodolf and Best 1996; also see any of The Nature Conservancy ecoregional plans.) These semi-natural habitats and remnants are extremely significant for biodiversity in agricultural landscapes. Duelli and Obrist (2003) found that 63 percent of animal species (from invertebrates to mammals) living in an agriculturally managed landscape were dependent on these remnant patches of natural habitats.

Increasingly, local and international conservation organizations are seeking to work with individuals, groups, government entities, and landowners to permanently preserve remaining tracts of natural habitat on private lands, using fee title acquisition or easements requiring management practices consistent with protection of native species and communities.

CONSIDER LANDSCAPE HISTORY AND CONTEXT. Part of the difficulty in providing quantitative guidance about how much habitat is needed is because it very much depends on what the rest of the landscape looks like. Landscape context greatly influences habitat suitability and ecological interactions. What is the history of the landscape and its ecological communities? Are conservation targets grassland-dependent species as in the Midwest or forest interior birds as in the eastern forests? What does the rest of the landscape look like? Are there large blocks of natural areas buffering farmland, or is the landscape outside of sustainable managed farms dominated by intensive row crop monocultures, rural or urban development?

The abundance of many species' populations appears to be governed at regional scales. Brown trout population abundance in southeastern Minnesota streams responds to regional trends in climate and agricultural practices, rather than local habitat features (Thorn et al. 1997).

MAINTAIN ADEQUATE PATCH SIZE AND CONNECTIVITY. All else being equal, larger areas of habitat sustain greater population sizes and a greater number of interior, specialist, and native species due to increased habitat diversity and more core area (Harris 1984; Forman 1995; Kennedy et al. 2003,). Some species declines, such as those of grassland birds, are unlikely to be reversed without active restoration and land use change on a substantial portion of the landscape. In historically forested watersheds, Black et al. (2004) found the total number of aquatic macroinvertebrate taxa declined rapidly when forest land cover within the local watershed decreased below 80 to 90 percent. They determined a land cover optimum of 70 to 80 percent forest land cover at the catchment scale and 80 to 90 percent cover at the local watershed scale.

MATRIX INFLUENCE AND PATCH CONDITION. Matrix influence is a measure of the positive or negative influence of the surrounding landscape on a habitat patch (Environment Canada 2004). The intensity of adjacent land use has a profound effect on biodiversity of a given patch (Lindenmayer and Franklin 2002). Effects of small patch size are likely to be more pronounced in landscapes where similar habitat is scarce than in landscapes where such habitat is common (Johnson 2001). For example, Andrén (1994, 359) suggested that "the decline in population size of a species living in the original habitat seems to be linearly related to the proportion of original habitat lost, at the initial stages of habitat fragmentation. At some threshold, area and isolation of patches of original habitat will also begin to influence the population size in the original habitat patches." The probability that certain forest birds will occur in small patches has been found to depend on the percentage of forest in the surrounding landscape (Askins et al. 1987; Dorp and Opdam 1987; Robbins et al. 1989). However, the effects of fragmentation may not occur until the original habitat is reduced by 70 to 90 percent (Andrén 1994).

Both the quality of habitat patches and the condition of the matrix between patches matters. Large patches with relatively degraded habitat — such as those dominated by non-native species, diminished diversity, modified hydrology, etc. — may have less value than small patches with high ecological integrity (structure, processes, and function). The condition of the matrix influences the effective size of remaining fragments and degree to which patches are isolated (Andrén 1994). This affects whether or not species can successfully disperse among habitat patches and whether important ecosystem processes, such as fire and hydrologic cycling, will occur on the landscape (Fahrig and Merriam 1994). If the matrix can support populations or allows for adequate species dispersal or migration between fragments, then communities of native plants and animals in remnant patches may retain diverse and viable populations. Wetland bird communities depend not only on their local habitat, but on the amount of wetlands within a surrounding three kilometer buffer (Fairbairn and Dinsmore 2001). Herkert et al. (1999) speculated that northern harriers (*Circus cyaneus*) and probably also short-eared owls (*Asio flammeus*) respond more strongly to the total amount of grasslands within the landscape than to the sizes of individual grassland tracts, responding to isolation as well as patch size.

Different types of agriculture are likely to be more compatible with biodiversity conservation in different landscapes, depending on the natural arrangement of physical features, habitats, and species, and the implications of land use history (Forman 1995). A study of breeding bird communities in central Pennsylvania, for example, found that forests within agricultural landscapes had fewer forest-associated species, long-distance migrants, forest canopy and forest understory nesting species, and a greater number of edge species than forest landscapes primarily disturbed by silviculture, irrespective of the effect of disturbance (Rodewald and Yahner 2001). In Colorado, ranchlands and protected reserves were found to be more compatible with species of conservation concern (including songbirds, carnivores, and plant communities) than exurban developments, which tended to support only human-adapted species (Maestas et al. 2001).

Research is needed to determine critical thresholds for proportions of suitable habitat relevant to geographic settings in which conservation-based farming is practiced. Some of The Nature Conservancy's ecoregional plans have developed baseline percentages for ecological community conservation targets, such as 10 percent of existing cover (Anderson and Bernstein 2003). Such targets were avoided in The Nature Conservancy ecoregional planning guidance, however, because existing cover is rarely representative in quantity or quality of historic cover, and because percentage figures derived from species-area relationships and island biogeography theory are inappropriate for ecoregions and communities where habitats are contiguous and boundaries are permeable. Estimating the proportion of suitable habitat in a landscape is a coarse-scale method of determining how much suitable habitat should be conserved to ensure the persistence of species in a region. In the case of some species declines, active consideration of landscape scale restoration will be needed to conserve interior forest or grassland-dependent species.

SINKS AND ECOLOGICAL TRAPS. The attractiveness of a habitat to species may not necessarily correlate with the quality of that habitat. Creating attractive habitat may not necessarily promote conservation if the new habitat functions as a "sink," i.e. an area that absorbs more individuals of a given population than it reproduces (Wiens and Rotenberry 1981; Pulliam 1988). Ecological traps, i.e. poor-quality habitats that nonetheless serve to attract individuals, have been observed in both natural and human-altered settings (Kristan 1993). Preferential use of poor habitat may in fact contribute to population declines and ultimately elevate extinction risk (Kristan 1993). Ecological traps may go undetected when population sizes are large, but when populations drop below threshold levels, they may thwart replacement in spite of the availability of high-quality habitat. Similarly, Kareiva and Marvier (2003) warn that a focus on “hot spots” may in fact be counter to conservation by focusing on edges and ecotones rather than source habitats.

Identification of sources and sinks is thus an important step in implementing a conservation strategy for wildlife in farmlands (Rodenhouse et al. 1993; Donovan et al. 1995; Koford and Best 1996). Because ecological traps are most likely to be associated with the rapidly changing and novel habitat characteristics primarily produced by human activities, they should be considered an important and potentially widespread conservation concern. For example, grassland birds that attempt to nest on farmlands are usually unsuccessful. A review of nest success in farmland indicates that few species are known to be reproducing at levels sufficient to balance estimated mortality. Estimates of nest success even on conservation reserve program fields and moderately grazed pastures were “discouraging” (Rodenhouse et al. 1993).

However, even sinks can play an important role in maintaining gene exchange among healthy populations and metapopulations. The more robust species (those with large dispersal, high fecundity, and high survivorship; usually the more widespread species) can persist in even the most extensively fragmented systems with only 25 to 50 percent of suitable habitat. For such species, source habitats can support as little as 10 percent of the metapopulations, while maintaining up to 90 percent of the total population (Pulliam 1998). However, rare species and habitat specialists like the Northern spotted owl may require up to 80 percent of suitable habitat in order to persist in a region (Lande 1987, 1988; Lamberson et al 1992).

PATCH LOCATION AND CONFIGURATION. Because connectivity of habitats is essential for daily and seasonal movements of animals throughout their life history, for example between feeding and nesting habitats, as well as for genetic exchange among metapopulations, much conservation literature calls for the establishment and protection of critical habitat corridors and/or “stepping stones” for birds and other animals that can move short distances across unsuitable habitats. Concerns have been raised about the possibility that such corridors might also function as sinks or ecological traps by concentrating predation (Boswell et al 1998). Little research exists on the extent to which corridors may act as sinks or traps for some species, thus undermining conservation efforts.

EDGES. Edge habitats are important in nature, and tend to be local hot spots of biodiversity. Edges tend to attract and benefit generalist species at the expense of interior habitat specialists; thus edges may often function as ecological sinks in nature. Agriculture has traditionally tended to increase edge habitats. Because ecological agriculture encourages structural diversity on smaller patches of landscape, small diverse farms increase edge habitat relative to conventional agriculture. Landscape scale shifts from conventional towards an ecological agriculture, if composed of smaller farms with greater diversity of habitats, are likely to have a regional effect on the abundance of edge species and generalists. In some cases this may occur to the detriment of interior habitat or specialist species, if it elevates regional abundance of competitive edge species and exotics, or contributes to increased mortality due to predation, parasitism, or nest failure. Chalfoun et al. (2002) found that predator species richness was significantly higher in forest edge. Forest vegetation structure was very similar between edge and interior. Differences in predator abundance, distribution, and species richness were driven by factors at larger spatial scales such as landscape context.

BOUNDARY ZONES. The amount of contrast between a patch edge and surrounding landscape matrix defines the severity of edge effects and dispersal abilities of wildlife populations (Kennedy et al. 2003). Edge effects are higher when there is greater contrast between patches and the surrounding matrix (Franklin 1993). In forested ecosystems, agroforestry and permaculture techniques, in which there is a gradual thinning of vegetation from row crops to shrubs and fruit trees and taller trees, provides a means to minimize edge effects and facilitate the movement of species between a patch and the surrounding matrix (Forman and Godron 1981).

GUIDELINES BY HABITAT TYPE

CROPLANDS

Both because of the continuing effects of agricultural land conversion and practices on terrestrial species, and because of the significant negative impacts of agriculture and croplands on aquatic ecosystems, all croplands should be managed to minimize soil erosion and to dramatically reduce the use and escape of fertilizers, pesticides, and breakdown products onto adjacent lands. The following practices, adapted from Granatstein (1997) and Bird et al. (1995), should be adopted on all croplands:

- Manage croplands using evolving best management practices to minimize soil erosion, sedimentation, nutrient and agrochemical losses to surface and groundwaters. These include increasing the protective cover on the soil surface, with practices such as no-till, cover crops, and windbreaks, and applying conservation measures such as contour strip cropping and grass waterways where appropriate.

- Eliminate or minimize intensive row-cropping and tillage on all highly erodible land and sensitive lands such as floodplains, riparian areas, wetlands, and extremely steep slopes.

- Use a greater variety of crops grown in more complex rotations to break weed and disease cycles, protect and build soil, and spread labor requirements over a longer period with less peak needs. Reduce total acreage and dominance of major crops, such as corn and soybeans in the midwest.

- Provide a variety of higher quality habitats to encourage and enhance wildlife diversity. Use cover crops and soil-building crops like legumes, clover, and grass. Integrate crops and livestock production with intensively managed grazing and recycling of manure to build soils.

- Implement less disruptive pest control tactics, e.g. using integrated pest management, in which pest levels are monitored, biological controls are used

Some analysts have expressed skepticism that voluntary or market-based approaches to nutrient reductions will be adequate, and have advocated more radical regulatory, tax, and economic reforms. Reducing nonpoint source pollution and restoring natural hydrologic regimes to levels sufficiently protective of aquatic biodiversity is likely to require significant changes in existing agricultural systems.

wherever available, and chemicals used only when an economic threshold is reached.

- Improve nutrient management to maximize efficiency and minimize nutrient movement to surface and groundwater. Use soil and plant testing to determine nutrient need. Add nutrients during peak crop use.

- Properly store and apply animal manures, and consider composting manures and other wastes.

- In arid regions and other areas relying heavily on irrigation, develop and implement management systems that use water as efficiently as possible. Water-intensive crops that compete with instream uses often impose a high cost on local ecosystems. Cropping systems should therefore be better matched to local and regional climatic and environmental conditions.

REDUCING NUTRIENT, SOIL, CHEMICAL, AND SEDIMENT LOSSES TO SURFACE WATERS AND AQUATIC ECOSYSTEMS. Although conservation of aquatic biodiversity will require drastic reductions in nonpoint source pollution of surface waterways, specific minimum targets will vary by ecoregion. To effectively address the problem of Gulf of Mexico hypoxia, for example, studies have estimated that nitrogen losses from croplands in the Mississippi Basin must be reduced by 30 to 70 percent. Concurrent and complementary sets of strategies proposed in the final report Integrated Assessment of Hypoxia in the Gulf of Mexico (a peer-reviewed, interdisciplinary collaborative scientific assessment convened by EPA and NOAA and developed by hundreds of scientists from a wide array of different fields during the 1990s) (Mitsch et al. 2001) include:

- Offering incentives to reduce the use of agricultural fertilizer;

- Replacing aging tile drainage systems with up to date systems that allow water drained off fields to be retained and reused;

- Employing comprehensive nutrient waste management for livestock operations such as dairies; and

- Applying precise and minimum nutrient application on fields, pinpointing only those areas in need of fertilizer (Mitsch et al. 2005).

Some analysts have expressed skepticism that voluntary or approaches to nutrient reductions will be adequate, and have advocated more radical regulatory, tax, and economic reforms. Reducing agricultural nonpoint source pollution and restoring natural hydrologic regimes to levels sufficiently protective of aquatic biodiversity is likely to require significant changes in existing agricultural systems (Lichatowich 2001; Keeney and Kemp 2003). For example, in the Upper Mississippi River catchment basins, meeting required targets for nitrogen losses can perhaps be achieved most easily by a large-scale transition from row crops on significant acreage. Nitrate nitrogen losses are 30 to 50 times higher from lands planted to row crops than land in perennial hay crops or grass systems (Randall et al 1997; Randall 2000). Randall (2001) concluded after years of studying corn and soybean rotations, that the two-crop rotation itself is unsustainable from the point of view of nitrogen and soil manangement. Many scientists from across the spectrum have also concluded that the current commodity corn-soybean rotation is unsustainable (Meadows et al. 2001, Keeney and DeLuca 2003). Significant acreage of lands currently used for row crops should be converted to hay, small grains, forage and pasture, woodlands, wetlands, and conservation uses. John Day of Louisiana State University and William Mitsch of Ohio State University, who played lead roles in the scientific assessment of Gulf hypoxia, are calling for the restoration of five million acres of wetlands and 19 million acres of riverside forest or grasslands in the Midwest — or 3 percent of the current farmland in the basin. Restoration of these natural filtering systems could reduce the Mississippi's nitrate load by 0.6 million metric tons per year.
The Committee on Environment and Natural Resources Report also recommended using crop rotation practices that employ perennials on 10

percent of farmland, effectively reducing the nitrate load by 0.5 million metric tons (30 percent) annually (Hey 2002).

Many analysts have proposed that we should consciously mimic aspects of nature to develop farming systems that "work with nature the way nature works" (Lefroy et al. 1999, Soule and Piper 1992). For example, agricultural systems should be designed to take advantage of a range of environmental conditions and ecological niches, e.g., matching crops to slope, soil type, and natural hydrology. The idea is coalescing under the banner called natural systems agriculture. Replacing annual grain crops such as corn and wheat with perennial plant mixtures, including nitrogen fixing legumes, would retain and rebuild soil structure, reduce insect and weed pressure, and reduce dependence on fuel-intensive tillage. As much floodplain land as possible should be restored and revegetated with native or perennial species that can trap and filter sediment and nutrients as well as provide adequate migration corridors, feeding and nesting habitat for birds, wetland and riparian dependent species. Agricultural systems in floodplains and seasonally flooded wetlands should be adapted to seasonal variations in flow and the natural movement of river corridors. For example, croplands in the areas behind the levees on the Mississippi could remain in production but shift to more flood tolerant or agroforestry crops (Core 2004). Hey (2002) has advocated removing excess nitrogen from streams and rivers by paying farmers to restore floodplain and riparian wetlands and/or to allow agricultural fields to temporarily flood as a means of removing some harmful nitrates and restoring other ecological functions.

In many wetland systems, wetland crops such as rice may provide an economic use compatible with maintenance of seasonal wetland habitat (Shennan and Bode 2003). Of all the agricultural practices in California's Central Valley, for example, rice production appears to be the most favorable to waterfowl by providing artificial wetlands that compensate, to some degree, for the extensive historic loss of native wetland habitat in this region (CH2M Hill 1992). Recent radical changes in rice field management in the Central Valley have improved their habitat value. Winter flooding of fields, which restores the historic, wet winter condition of much of the land where rice is farmed, greatly increases the habitat value of rice fields because waterfowl depend heavily on Central Valley habitat during winter. Rice farms are also the preferred neighboring land use in some coastal estuarine systems such as the Mad Island Marsh Preserve along the Texas Gulf Coast (Soulé 2002). However, care must be taken in irrigated systems to avoid overuse of in stream flows as well as the potential build-up of toxic salts in irrigation drainage water.

Analysts have proposed that we should consciously mimic aspects of nature to develop farming systems that "work with nature the way nature works." For example, agricultural systems should be designed to take advantage of a range of environmental conditions and ecological niches.

RESTORING ECOSYSTEM PROCESSES TO FARMLANDS. Jackson (2003) provides an overview of practices that would restore prairie processes to farmlands in the Midwest. Traditional rotation-based cropping systems in the American Midwest from the 1900s to the 1950s maintained important elements of the prairie ecosystem: diverse perennial hayfields, biological nitrogen fixation by legumes, well-timed delivery of nitrogen to plants via the breakdown of organic matter; and an intact food chain, including ruminant herbivores.

In prairie and prairie-forest border ecoregions, croplands should be in a matrix of natural grasslands, grazed lands, and pastures to provide habitat for grassland species such as ground squirrels and other prey species (Hunt 1993; Marzluff et al. 1997; Paton 2002). In forested ecosystems, a key biodiversity strategy is to maintain large patches of forest with adequate interior forest habitat. A shift from conventional farming to permaculture, agroforestry (tree crops), and silviculture that mimics forest ecosystems may lessen the impacts of agriculture in such regions. Studies in tropical forest regions have shown mixed, mostly positive impacts of agroforestry for birds and other species. More research is needed in North American systems to understand the potential value of agroforestry lands in riparian or forested ecosystems as habitat and stepping stones for birds (Core 2004).

For more guidance on required forest patch size to maintain different fractions of interior species, see

How Much Habitat Is Enough?

Forest habitat guidelines, pp 30-44 and appendices.

http://www.on.ec.gc.ca/wildlife/factsheets/fs_habitat-e.html

ORGANIC FARMING. In rural areas surrounding major urban centers, the economics of organic production and local food systems offer growing potential for economically viable organic agriculture. Many studies have demonstrated higher soil, insect, and biodiversity values on organic farms relative to conventional farms (Freemark and Kirk 2001; Hole et al. 2005). However, conversion of natural habitats to organic agriculture to speed the organic certification process (rather than converting conventional farmland to organic) is a net loss for biodiversity. It should be noted that organic production, while a positive development, is not "sufficient" to conserve biodiversity, and should not be viewed as a panacea for biodiversity loss. Organic production, particularly of annual vegetable crops, is intensive, and involves disturbance of the soil and substantial modification of natural habitats. It can be equally stressful to aquatic ecosystems in terms of irrigated water use and in stream withdrawals, and can be done in large monocultures. Relatively little research and analysis has been done explicitly comparing the relative local and landscape-level value and function of different environmentally friendly or organic production systems — such as organic vegetable production versus grass-based farming — for overall impacts on biodiversity.

GRASSLANDS

Throughout North America, rotational and management-intensive grazing systems are increasingly being developed to mimic the role of bison and other native grazers on the landscape as well as to improve soil health and restore hydrologic function, especially relative to plowed or irrigated production systems. However, many grassland communities co-evolved with a suite of grazers including elk, bison, wild sheep, deer, and pronghorns. Much research confirms that appropriately managed grazing is considerably more compatible with native biodiversity than extensive areas of cropland, particularly irrigated cropland.

Restoration strategies on many lands under conservation ownership or management promote substituting cattle grazing land for irrigated corn. In the Platte River system of Nebraska, a habitat used by tens of thousands of migrating sandhill and whooping cranes and other migrating grassland birds, was plowed under for corn during the 1970s, an acreage loss that rivaled the total loss during settlement (Soulé 2003). There The Nature Conservancy is actively restoring formerly irrigated corn fields for grazing (Soulé 2003). At Red Canyon Ranch in Wyoming, rotational grazing mimics the herd effects and migration patterns of native grazers as they moved around the landscape. Studies at Red Canyon have documented increases in neotropical migrant birds and fish (Soulé 2003). A potentially beneficial relationship with grazing is also suggested in Michigan's degraded fens. Where tile and drainage ditches from surrounding fields empty into a fen, the sheet flow from groundwater seepage that makes a fen is disrupted. The altered nitrogen balance often leads to invasion by cattails, phragmites, and purple loosestrife, and the altered water and disturbance regimes lead to replacement of prairie vegetation with shrubs. Limited experiments with grazing have shown some promise in restoring prairie vegetation. However, the impact of grazing on fens remains an open question.

Grazing can have positive or neutral effects on ecosystems and diversity. Large-scale influences such as serious drought or soil variation have much greater effect than grazing (Wallis DeVries 1998; Soulé 2003). Changes in microclimate cycles over the short history of European settlement on the North American landscape make it difficult to separate natural variation from anthropogenic disturbances. The importance of grazing versus fire as the appropriate disturbance regime is still debated. In the U.S., conservationists have tended to favor fire management over cattle grazing, blaming the latter for reduced diversity, overgrazing, and spread of invasive exotics (Wallis DeVries 1998). For disturbance adapted ecosystems, a varied set of disturbance agents such as fire and grazing should be incorporated at historic frequencies and intervals (Fuhlendorf and Engle 2001).

Cooperative grazing partnerships have the potential to restore grazing patterns on much larger landscape units. The Malpai Borderlands Group has perhaps received the most attention in the literature and popular press (Wolf 2001). The Malpai Group is a collaborative effort, organized and led by ranchers along the Mexican border in southern Arizona and New Mexico, built around the goals of protecting their community, way of life, and wild landscape from development and subdivision from nearby towns. The group has activities in several program areas directed at protecting and restoring the ecological diversity and productivity of native grassland and savanna habitat, including restoring fire as a natural landscape process. The group has established conservation easements over 42,000 acres of land, and has pioneered "grassbanking" whereby ranchers can rest their ranches from grazing by moving their herds under reciprocal conservation agreements. The Malpai group also has an extensive monitoring and science program designed to assess the effectiveness of their restoration activities on the landscape.

The Malpai Borderlands Group (see box below), The Nature Conservancy, and other partners in these landscape-scale restoration initiatives recognize the need to monitor and use adaptive management to generate feedback on the success of management theories and practices and guide ongoing and future management. Working landscape restoration will increasingly require collaborative efforts among farmers, ranchers, researchers, agencies, and conservation organizations to monitor outcomes, recognize problems, and identify emerging issues.

MALPAI BORDERLANDS GROUP:

Formed in the early 1990's, the group consists of approximately two dozen landowners whose ranches span nearly a million acres in New Mexico, Arizona, and northern Mexico. The group was formed by ranchers concerned about the long-term effects of state and federal fire suppression and overgrazing that lead to critical invasions of brush and woody species into what had formerly been luxuriant grasslands.

Visit their website:

http://www.malpaiborderlandsgroup.org/

STREAMS, RIVERS, AND RIPARIAN SYSTEMS

Riparian areas possess a relatively high diversity of species and environmental processes (Naiman and Decamps 1997). Considerable research has demonstrated the effectiveness and functionality of riparian buffer zones along rivers, streams, and wetlands in attenuating sediments and nutrients, providing corridors and refugia for wildlife, and quality habitat for fish and aquatic species. Ideally, buffers should extend along all perennial, intermittent, and ephemeral streams, lakes, shorelines, and adjacent wetlands (Weller et al. 1998, Wenger 1999).

In addition to protecting water quality, small isolated riparian habitat patches have been found to function as important wildlife corridors in otherwise fragmented landscapes. For example, riparian corridors provide vital stop-over sites for en route migratory birds in the southeastern United States (Skagen et al. 1998) and breeding habitat for many different grassland, shrubland, and forest-nesting birds (Peak et al. 2004). However, for some of the most sensitive species, even wide riparian forest buffers are inadequate (Peak et al. 2004).

The effectiveness of buffers in providing habitat and mitigating watershed impacts depends on a range of variables, including the composition of vegetation in the buffer, the degree of disturbance in the surrounding landscape, the surrounding topography, and the contiguity of the buffer. For example, a larger width may be required for buffers surrounding more pristine or highly valued wetlands or streams; in close proximity to high impact land use activities; or with steep bank slopes, highly erodible soils, or sparse vegetation (Castelle et al. 1994; Fischer and Fischenich 2000). Lowrance et al. (1997) found that the value of a buffer in trapping nutrients diminished over time if it was in a landscape where it was expected to absorb large amounts of nutrients from agricultural runoff.

BUFFER WIDTH. As with other conservation thresholds, the scientific literature does not support uniform buffer width applicable in all circumstances (Kennedy et al. 2003). At a minimum, a riparian buffer should encompass "the stream channel and the portion of the terrestrial landscape from the high water mark towards the uplands where vegetation may be influenced by elevated water tables or flooding, and by the ability of soils to hold water" (Naiman et al. 1993).

The majority of reviews and studies suggest that buffers must be a minimum of 25 meters in width to provide nutrient and pollutant removal; a minimum of 30 meters to provide temperature and microclimate regulation and sediment removal; a minimum of 50 meters to provide detrital input and bank stabilization; and over 100 meters to provide for wildlife habitat functions (Kennedy et al. 2003; Castelle et al. 2004).

EXTENT. The effectiveness of a buffer in providing ecological services depends partly on the continuity, or linear extent, of the buffer along the entire aquatic system (Wenger 1999). Protection efforts should prioritize the establishment of continuous buffer strips along the maximum reach of stream, rather than focusing on widening existing buffer fragments (Weller et al. 1998). Protection of the headwater streams as well as the broad floodplains downstream, which generally encompass less than 10 percent of total land area, is also recommended (Naiman et al. 1993).

BUFFER PROTECTION. To ensure that buffers function adequately, all major sources of disturbance and contamination should be excluded from the buffer zone, including dams, stream channelization, water diversions and extraction, heavy construction, impervious surfaces, logging roads, forest clear cutting, mining, septic tank drain fields, agriculture and livestock, waste disposal sites, and application of pesticides and fertilizers (Wenger 1999; Pringle 2001).

COMPOSITION OF VEGETATION. To maximize ecological function, buffers should have diverse vegetation that is native and well-adapted to the ecosystem and disturbance regime (Kennedy et al. 2003). Diversity of species and vegetation structure (e.g., grass and herbaceous understory, shrubs, and mature trees) is recommended to allow for greater

THE ROLE OF CONSERVATION PROGRAMS IN THE U.S. FARM BILL

Conservation programs in the U.S. Farm Bill have for many years provided many soil conservation, water quality, wildlife habitat, and other natural resource benefits. Of these, the Conservation Reserve Program is the largest program both in terms of acres enrolled and federal outlays (Ribaudo et al 2001). With about 36 million acres under contract (as of 2004), up to 10% of US cropland is idled under the program. For comparison, the National Wildlife Refuge System holds about 15 million acres in the continental United States (USFWS, 1999). The Conservation Reserve Program is a voluntary long-term cropland retirement program that provides participants an annual per acre rent and cost-share for establishing permanent land cover in exchange for retiring highly erodible and/or environmentally sensitive cropland from production for 10-15 years. Annual rental payments paid to enrollees in fiscal year 2001 by the U.S. government totaled roughly $1.5 billion. The Wetlands Reserve Program also provides cost-share and easements for to protect wetlands on agricultural lands. The Wetlands Reserve Program cap was raised under the 2002 Farm Bill from 1 to nearly 2.5 million acres, with annual outlays of around $150 million. The Wildlife Habitat Incentives Program provides cost sharing to landowners to develop and improve wildlife habitat. Annual outlays range from $15-$85 million, and over 1.4 million acres have been enrolled since 1998. Finally, the Farmland Protection Program provides funds to help keep productive farmland in agricultural use and avoid subdivision and conversion for development.

New programs authorized under the 2002 Farm Bill included the Conservation Security Program and the Grassland Reserve Program. In recent years, the Conservation Security Program has received much attention as a program to promote adoption of a wide range of structural and/or land management practices to address a variety of local and/or national natural resource concerns. The 2002 Farm Bill estimated spending on Conservation Security Program provisions of $369 million for 2003-2007. The Grassland Reserve Program has been authorized to protect up to 2 million acres of grasslands (for comparison, the U.S. is estimated to have 525 million acres of privately-owned grasslands).

The environmental performance of conservation provisions of U.S. agricultural policy are extensively reviewed elsewhere (Haufler 2005). Conservation objectives in the Farm Bill have grown increasingly complex and multidimensional over the years to meet growing list of concerns (Anderson 1995). Farm programs are under political pressure to be all things to all constituents, and this can create conflicts between competing objectives. Environmental programs under the Farm Bill have at times created perverse incentives, as when farmers preemptively drained wetlands to avoid Swampbuster provisions, or plowed prairie or other lands marginal for agriculture in order to qualify for enrollment in land set-aside programs.

Most significantly, conservation outlays under the Farm Bill pale in comparison to its provisions for commodity price supports. Total annual outlays for conservation programs average $5-7 billion in recent years, compared to total expenditures of $38-40 billion. Commodity programs are the elephant in the room, an economic driver that artificially inflates acreage and production totals, and can drive farm-level decisions that undermine the gains made through conservation spending.

Continued on next page

THE ROLE OF CONSERVATION PROGRAMS IN THE U.S. FARM BILL, *continued from previous page*

Nevertheless, many conservation organizations have in fact viewed the Farm Bill, through the Conservation Security Program and other conservation provisions, as the appropriate tool to make significant progress in implementing a comprehensive national system of conservation lands. For example, the $2.7 billion estimated annual cost of acquiring, restoring, and maintaining 24 million acres of wetlands in the Mississippi River Basin to reduce nitrogen loads driving Gulf of Mexico hypoxia (as recommended in the Final Integrated Assessment Report, Mitsch et al. 1999) represents a fraction of farm bill spending, and is comparable to existing Conservation Reserve Program outlays. If Conservation Reserve Program funds could be used to establish nitrogen farms, then commodity price control and nitrate reduction would be advanced simultaneously. Furthermore, costs are often well under the costs of mitigation after-the-fact for natural disasters made more expensive in the wake of poor land use. For example, the flood of 1993, which caused, by some estimates, $16 billion of damage, could have been contained within 13 million acres of restored wetlands in the upper Mississippi basin (Hey and Philippi 1995).

resilience to possible fluctuations in environmental conditions (e.g., water levels, temperature, herbivory), and to provide for more ecological function (e.g., wildlife habitat) (see Fischer and Fischenich 2000 for further guidance on vegetation type, diversity, and function).

ECONOMIC USE OF BUFFERS. Many agricultural consultants and academics have promoted adoption of riparian buffers by advocating their potential economic uses for fuel wood, biomass, or other products. Economic use of buffers should be approached from an ecological perspective. For example, hybrid poplar buffers are being widely promoted as a "sustainable" practice by some technical experts and consultants (Perry et al 2000). However, the use of herbicides may be needed for successful establishment of native riparian buffers in some areas. Likewise, the frequency and method of harvest will influence whether the buffer serves as an effective sediment and nutrient trap, or a periodic source of pollutants to the water body. Furthermore, hybrid poplar plantations provide only a fraction of the full range of ecological functions provided by native floodplain and riparian forests.

WETLANDS

Many species of fish and wildlife rely on multiple habitat types throughout their life history, and many rely on wetlands or riparian areas for critical parts of their life cycle. The ecological and biodiversity services and benefits provided by wetlands are better understood and acknowledged, from local soil and water conservation plans to a "No Net Loss" goal articulated (if not achieved) in national wetlands policies.

WETLAND SIZE AND EXTENT. Small wetlands of less than 2 hectares can support a surprisingly high species richness of amphibians (Richter and Azous 1995; Kennedy et al 2003). Proximity to core habitat and local habitat heterogeneity rather than area alone may better correlate with reptile and amphibian richness (Burbink et al. 1998).

Critical ratios of wetland area to watershed area vary according to channel slope, as well as historical and current land use/land cover (Environment Canada 2004). Wetland habitat guidelines should be based on historical extent and patterns. Wetlands should also be well distributed across each sub-watershed.

As a minimum guideline, at least ten percent of a watershed, and six percent of any sub-watershed, should be comprised of wetlands (Environment Canada 2004). Most studies show that the effect of wetlands in reducing watershed yield, reducing flooding, sustaining base flows, and reducing high flows flattens above 10 percent of wetland cover. When wetlands comprise about 10 percent of a watershed, flooding is greatly reduced and base flows are better maintained (Johnson et al. 1996). Hey and Philippi (1995) calculated that strategic restoration of 5.3 million ha of wetlands in the Upper Mississippi River Basin (or 10% of the Upper Mississippi River Basin) could have provided storage to sufficient to accommodate the 1993 Mississippi floods, vastly reduced the estimated $16 billion in flood damages that were sustained. Mitsch et al. (2001) recommended restoring 2.1-5.3 million ha of wetland and 7.8-20.0 ha (3-7%) of bottomland hardwood forest within the 300 million ha Mississippi River Basin, primarily as nitrogen sinks that could effectively reduce N loads in the Mississippi River and control Gulf of Mexico hypoxia.

THE CRITICAL FUNCTION ZONE OF A WETLAND SHOULD BE NATURALLY VEGETATED. Uplands adjacent to wetlands provide a variety of critical functions, such as nesting habitat for wetland waterfowl or other wetland-associated fauna, or a groundwater recharge area important for wetland function. This upland area within which functions and attributes related to the wetland occur has been termed the "Critical Function Zone" (*Environment Canada* 2004). Along with the wetland itself, the Critical Function Zone should be protected from adverse effects arising outside of the wetland, such as contaminants and intrusions, by a buffer or filter strip, varying in size with required function. The combined Critical Function Zone and required buffer area comprises the naturally vegetated adjacent lands needed around a wetland. The appropriate size of the Critical Function Zone, wetland, and buffer must be determined based on site-specific knowledge of important ecological attributes and their sensitivities, and on management objectives. Minimum guidelines for natural vegetation in wetland buffers (*Environment Canada* 2004) were as follows:

OTHER RESOURCES:

For a more extensive summary of quantitative findings from the scientific literature regarding minimum patch size, minimum proportions of suitable habitat, minimum riparian buffer width, which are found to maintain long-term persistence of viable populations or communities pertaining to a range of species and community types, see Appendices A-D, page 59.

Excerpts from Kennedy, C., J. Wilkinson, and J. Balch. 2003. Conservation Thresholds for Land-Use Planners. Washington, D.C.: Environmental Law Institute.

- The entire catchment of a bog;
- 100 meters adjacent to a fen, or as determined by a hydrological study, whichever is greater; and
- 100 meters adjacent to swamps and marshes.

Wetlands of various sizes, types, and hydroperiods will be used by different wildlife. Larger swamps and marshes tend to have greater habitat heterogeneity — i.e., greater variety of habitats — which in turn tends to support more wildlife species (e.g., emergent versus submerged vegetation). Smaller marshes will be less likely to have multiple marsh communities of sufficient size for use by wildlife. However, small and ephemeral wetlands are often used preferentially by breeding amphibians. A variety of natural wetland types should be conserved across a landscape.

As a minimum guideline, at least ten percent of a watershed, and six percent of a sub-watershed, should be comprised of wetlands.

FLOODPLAINS

Floodplains play an important role in the ecological function and dynamics of riverine ecosystems, including fish spawning, sediment dynamics, flood storage and attenuation, and nutrient cycling. Human impacts on river hydrology, geomorphology, biodiversity, and ecological processes in river systems — notably disconnecting rivers from their floodplains with levees and water level management that has eliminated important seasonal flood pulse dynamics — is responsible for a large proportion of federal and state listings of aquatic species as rare, threatened, or endangered. Floodplains should be restored and protected from further development and levees. Policy should seek to gradually discourage cropping of floodplains at the expense of natural variations in flow. If floodplains are to be used for agriculture, agroforestry and other types of agriculture compatible with natural cycles are preferable to annual monocultures requiring annual plowing, fertilizing, etc.

ADDITIONAL CONSIDERATIONS FOR AQUATIC ECOSYSTEMS

Even if substantial portions of the terrestrial landscape are managed in natural cover or ecological agriculture, the goal of conserving all remaining biodiversity cannot be met without substantial attention to restoration of aquatic ecosystems. A major challenge will be to assess the magnitude of impacts from hydrologic alterations and to reestablish hydrologic connectivity and natural patterns (Pringle 2001). As a result of dam building, water diversions, groundwater extraction, and other broad scale impacts, aquatic ecosystems have become degraded, dewatered, and disconnected. These alterations are often beyond the control of reserve managers and are often ignored by conservationists. The location of a reserve within a watershed often determines how it will respond to human-altered hydrological processes (Pringle 2001). For example, reserves located in middle or lower portions of watersheds often suffer the most direct impacts, whereas reserves in upper watersheds become isolated from one another and lose anadromous or migratory species due to downstream activities or barriers.

Table 2.
Example guidance on landscape scale habitat minima from the literature *(also see Appendices A-D)*

Parameter	Guideline	Source
Natural habitat	20-60% of natural habitat in a landscape should be conserved.	Kennedy et al. 2003
Croplands restored to wetlands, riparian buffers, and floodplains	70% reduction in the nitrogen load is needed to reduce the extent of the Gulf of Mexico hypoxia problem. Restoration of five million acres of wetlands and 19 million acres of riverside forest or grasslands in the Midwest—or 3 percent of the current farmland in the basin.	Hey 2002; Mitsch and Day 2003
Wetland habitat guidelines		
Percent wetlands	Greater than 10 percent of each major watershed in wetland habitat; greater than six percent of each subwatershed in wetland habitat; or restore to original percentage of wetlands in the watershed.	Detenbeck et al. 1999
Amount of natural vegetation adjacent to the wetland	For key wetland functions and attributes, the identification and maintenance of the Critical Function Zone and its protection, along with an appropriate Protection Zone, is the primary concern. Where this is not derived from site-specific characteristics, the following are minimum guidelines: - Bog: the total catchment area - Fen: 100 m or as determined by hydrogeological study, whichever is greater - Marsh: 100 m - Swamp: 100 m	Environment Canada 2004
Wetland type	The only two wetland types suitable for widespread rehabilitation are marshes and swamps.	Environment Canada 2004
Wetland location	Wetlands can provide benefits anywhere in a watershed, but particular wetland functions can be achieved by rehabilitating wetlands in key locations, such as headwater areas for groundwater discharge and recharge, flood plains for flood attenuation, and coastal wetlands for fish production. Special attention should be paid to historic wetland locations or the site and soil conditions.	Environment Canada 2004
Wetland size	Wetlands of a variety of sizes, types, and hydroperiods should be maintained across a landscape. Swamps and marshes of sufficient size to support habitat heterogeneity are particularly important.	Environment Canada 2004
Wetland shape	As with upland forests, in order to maximize habitat opportunities for edge-intolerant species, and where the surrounding matrix is not natural habitat, swamps should be regularly shaped with minimum edge and maximum interior habitat.	Environment Canada 2004

Table 2, *continued*
Example guidance on landscape scale habitat minima from the literature

Riparian Habitat Guidelines		
Buffer zones		
Width of buffer	3-200 m buffers along streams effective for different functions. Very non-linear results. Larger adjacent lands areas are needed where adjacent land use is intense. Based on the majority of scientific findings, buffer strips must be a minimum of 25 meters in width to provide nutrient and pollutant removal; 30 meters to provide temperature and microclimate regulation and sediment removal; 50 meters to provide detrital input and bank stabilization; and over 100 meters to provide for wildlife habitat functions.	*Environment Canada* 2002
Extent of buffer	30 m buffer along 75% of stream length defined a threshold for fish community degradation in Toronto area streams.	Strus et al. 1995, *Environment Canada* 2002
Composition	Buffer zones should be in native or perennial vegetation.	

CONSIDERATIONS IN DETERMINING "HOW MUCH IS ENOUGH"

GUIDELINES ARE NOT TARGETS

A growing body of experience in conservation suggests that success requires establishing benchmarks and quantitative goals that can effectively focus efforts and resources, and against which performance can be evaluated and assessed. At the same time, ecological conditions, settings, habitat needs, agricultural practices and systems, etc. all vary substantially geographically and regionally. Quantitative recommendations and research summarized herein should be interpreted as guidelines rather than targets. The vast majority of scientific papers reviewed emphasize that biodiversity needs vary with local and landscape context; cultural and natural conditions influence the effectiveness and applicability of recommendations; and one-size fits all recommendations do not apply. Furthermore, little is known about the minimum individual and population level habitat needs for most species. Overall, reviews of the ecological literature make it clear that quantitative habitat guidelines, such as 30 percent natural vegetative cover on farms in forested landscapes, or 10 percent thresholds for impervious surface or wetlands in a watershed, represent desirable minima (Environment Canada 2004, Tear et al. 2005). Biodiversity protection will be enhanced whenever and wherever these standards can be exceeded. The principal factor governing the compatibility of management practices with biodiversity protection is the degree to which regional agricultural systems mimic natural, historical, and local ecoregional structure and function on the landscape.

Biodiversity needs vary. The vast majority of scientific papers reviewed emphasize that biodiversity needs vary with local and landscape context; cultural and natural conditions influence the effectiveness and applicability of recommendations; and one-size fits all recommendations do not apply.

LANDSCAPE MATRIX. The question of how much habitat is enough must be considered in the ecological, land use, and historical context of a watershed/ecosystem. What is happening in the rest of the landscape? A given percentage of conservation-friendly farms may support higher biodiversity in a landscape containing numerous patches of natural vegetation and public or protected lands than in one dominated by intensive agricultural production. Adjacent landscape patches can influence biodiversity by harboring habitat for non-native exotics, edge, predator, or colonizing species that compete, reduce the habitat quality, or directly reduce the survival of species in remaining habitat patches and natural remnants. They can also serve as a vector for wildlife diseases (Mack et al. 2000).

LANDSCAPE-SCALE APPROACH. Planning for ecosystems and landscape-scale habitat for biodiversity requires both watershed and terrestrial landscape approaches. Management of habitats for fish and wildlife can neither be restricted to watershed boundaries nor to terrestrial political or administrative boundaries. Ecoregional planning in which conservation organizations work with federal, state, and local agencies and producers represents a positive step towards integrating landscape-scale terrestrial and aquatic ecosystem planning and protection. Each of the 50 states also completed a wildlife action plan in late 2005 that lays out species and habitats of greatest conservation need, threats to those species and habitats, and conservation actions needed to address those threats. Many of these regional and statewide plans recognize the importance of privately owned and working lands in the matrix and explicitly call for policies that create incentives and promote more environmentally compatible agriculture practices. Some also recommend management agreements, covenants, and/or deed restrictions on critical habitat portions of the landscape.

Wildlife Action Plans: For more information and state by state updates, go to

http://www.teaming.com/state_wildlife_strategies.htm

To view the eight required elements:

http://www.biodiversitypartners.org/bioplanning/elements.shtml

To view guiding principles, go to:

http://www.biodiversitypartners.org/bioplanning/IAFWA_CWCSGuidingPrinciples.pdf

But how should scarce conservation dollars be allocated and distributed? Should conservation resources be allocated for conservation farming over large acreages in which monies might be diverted to achieve non-conservation objectives, or should conservation dollars be targeted more intensively to land acquisition and enforceable protection of smaller areas where the designated land use is biodiversity protection? Such questions continue to pose challenges to conservationists, but there are increasing tools available, such as the state wildlife action plans, to continue addressing these questions.

SCALE AND GEOGRAPHIC VARIATION. Results of ecological studies have been shown to be dependent on the scale at which an organism is investigated (Noss 1991; Trine 1998; Black et al. 2004). Depending on their mobility, different organisms experience their environment as fragmented at different spatial scales of habitat heterogeneity (Thomas 2000). Minimum habitat areas may vary for species depending on region and landscape context. Distribution is influenced by factors over a wide range of spatial scales. For example, fragmentation for migratory songbirds that travel thousands of miles and require several acres of intact forest habitat for successful nesting and rearing will be experienced very differently than fragmentation by a population of butterflies. The impacts of habitat fragmentation on any given species also have substantial regional variation. For example, Trine (1998) found that minimum habitat area required to maintain viable wood thrush populations were much larger in the midwest than in the eastern United States.

STRESSORS BEYOND HABITAT. There are many factors accounting for biodiversity loss beyond the loss of habitat. Many species are declining, and the precise reasons for the decline are unknown even where habitat appears to be increasing. Agricultural practices that result in loss of nutrients, pesticides, or sediment; hydrologic alterations; commercial, municipal and industrial discharges of nutrients and toxics; atmospheric deposition of mercury and other persistent pollutants such as hormonally active substances; the spread of invasive exotic species; and global and local climate change all represent significant potential threats to biodiversity. Many of these stressors are beyond the control of individual land managers, and are often inadequately addressed by conservation strategies. Such concerns are only peripherally addressed in this document.

Furthermore, although agriculture is the focus of this paper, it is not the only land use that has implications for biodiversity. Like farmlands, forestlands and rangelands also provide significant biodiversity services, especially if managed on ecological principles or in part for conservation objectives. Thus many of the principles discussed here also apply to management of working forests and rangelands.

NOT ALL SUSTAINABLE AGRICULTURE IS CREATED EQUAL. Many terms are used to describe agriculture, from "conservation-friendly farming," "conservation-based agriculture," "sustainable agriculture," "organic agriculture," to "permaculture" and "ecological agriculture". These terms have been developed and applied to different discourses at different times to refer to different environmental quality goals. Take, for example, "sustainable agriculture." What does sustainable agriculture sustain? A narrow interpretation of the term which receives wide usage in the literature refers to practices that are sustainable in terms of greatly reduced soil erosion and nutrient loss. This is the sense of the term used, for example, in much of the academic literature on conservation tillage (Coughenour and Chamala 2003). Yet conservation tillage alone, despite its promise in reducing erosion and the attendant effects on aquatic ecosystems, would not in and of itself be "sustainable" from the point of view of biodiversity, particularly because of its questionably heavy reliance on herbicides for weed control. This adds to the difficulty of determining "How much is enough?" It could be argued that ultimately, for society to succeed at the twin goals of adequate food production and maintaining biodiversity and ecosystem services, 100 percent of agriculture must be "sustainable." We can't continue to wash soil downriver, deplete groundwater, pollute coastal and freshwater aquatic ecosystems, and substitute inputs based on cheap fossil fuel energy for healthy soil and efficient management of ecological cycles, or we are liable to go the route of the lost civilizations of Easter Island, the American Southwest, and the ancient Middle East (Lowdermilk 1953). Such a transition may ultimately involve some mixture of row crop agriculture under intensive production, with extremely judicious and minimal use of agricultural chemicals and inputs, combined with increased areas of integrated agriculture, perennial production, permaculture, diverse farms, modified grazing systems, and restoration of substantial acreages of marginal, riparian, wetland, and floodplain lands.

INTENSIVE, SEGREGATED USE VERSUS "WORKING LANDSCAPES." Biodiversity loss is most observable locally within high input agroecosystems, but low-input and extensive systems can also bring about significant biodiversity loss through increased conversion of natural habitats (Wood et al. 2000). Some have advanced the argument that intensive management of land, such as in the American Midwest, maximizes economic production and "efficiency," allowing marginal or higher value conservation lands to be set aside for biodiversity protection elsewhere. The argument advocates a kind of "triage" in which the "best" lands are sacrificed for intensive production. A familiar argument in the discourse is that intensive farming and forestry practices in northern temperate zones can spare tropical rainforest, with higher per acre biodiversity, and/or habitats and regions perceived to have higher conservation, amenity, or wilderness values (e.g., Horsch and Fraley 1998; Avery 1995.) In intensively managed croplands, there is indeed a sharp trade-off between agricultural production and biodiversity. Increasing agricultural productivity and sustainability on existing agricultural lands may be an important part of the strategy for slowing or reversing the conversion of wildlands to agriculture (McNeely and Scherr 2003). Yet notwithstanding demand theory, there is little evidence to suggest that intensive land use in temperate sacrifice zones translates directly into reduced pressure on natural lands in more biodiverse regions.

With ecosystem services required across the landscape in all regions, biodiversity in one region or state or country cannot be sacrificed to preserve biodiversity in another. Furthermore, the notion that some lands can be sacrificed is at best tenuous in a world in which isolated preserves are inadequate to protect whole interconnected ecosystems, in which society is belatedly learning the full value of ecological services that working agricultural, forest, and rangelands provide, and in which human influence in the form of climate change, atmospheric deposition of mercury and other persistent contaminants, penetrates into even the most remote ecosystems. Both the sustainability and efficiency of intensive agricultural production systems are called in to question when all these external costs are factored in, from energy and transportation subsidies to the environmental and infrastructure costs of agricultural water supply, use and water quality protection; crop insurance and natural disaster spending.

Finally, people value nature everywhere, not just in isolated preserves. In his forward to the 1996 Natural Resource Conservation Service publication *A Geography of Hope*, Chief Paul Johnson wrote “today we understand that narrowly circumscribed areas of natural beauty and protected land alone cannot provide the quality of environment that people need and want... A land comprised of wilderness islands at one extreme and urban islands at the other, with vast food and fiber factories in between, does not constitute the ‘geography of hope’. But private land need not be devoted to a single purpose enterprise. With a broader understanding of land and our place within the landscape, our Nation's farms, ranches, and private forest land can and do serve the multiple functions that we and all other life depend upon.”

McNeely and Scherr (2003) describe the gradual evolution of thinking about an alternative “ecoagriculture” as a continuous evolution from an agriculture focused on maximizing production to an agriculture that views agriculture as part of landscapes managed for production of food, ecosystem services, and wildlife habitat. They describe how even modern commercial agriculture — building on theories and advances in conservation biology (Meffe and Carroll 1997), landscape ecology (Forman 1995), and systems thinking (Röling and Jiggins 2003) — has formally begun to adopt elements of ecological principles into production agriculture.

EXPERTS, UNCERTAINTY, AND ADAPTIVE MANAGEMENT

The history of conservation is one of passionate advocates working to communicate the importance of protecting and managing the planet's outstanding natural heritage legacy. It is also one of politically charged struggles between private landowners, environmental advocates, and government institutions over rights and responsibilities regarding natural resources.

Science has played a critical role both in shaping our understanding of the importance of biodiversity, and in calling our attention to the threats that face it. Unfortunately, science has not always been put to the best uses. In some cases, perceptions about bureaucratic ineptitude or skepticism about the abuses and excesses of government may be rooted in elements of past experience. At least some of the responsibility for our current legacy of biodiversity loss stems from our society's own “expert” management. For example, the Soil Conservation Service (now the Natural Resources Conservation Service) long provided technical and monetary assistance to farmers to drain wetlands, channelize creeks, build dams, and introduce exotic species such as multiflora rose, and kudzu (Jackson 2002). Forestry, wildlife, and fishery agencies have frequently promoted silvicultural practices, fish and game management and stocking policies that have resulted in historical and ongoing threats to biodiversity. Federal and state natural resource management agencies are beginning to be more sensitive about restoration, management and conservation. However, in many places, agencies continue to work at cross purposes, e.g., working to eradicate noxious weeds while simultaneously promoting agricultural use of non-native Siberian elm or hybrid poplar, or promoting “environmental” practices whose costs may ultimately prove to be far in excess of the espoused environmental benefits, such as corn production for renewable fuel (Oliveira et al. 2005). Such a mixed record would perhaps argue for some humility on the part of management and policy.

Increasingly, effective conservation solutions call for collaborations among diverse entities, disciplines, and institutions. “Experts” cannot do conservation alone. Many biological interactions and large-scale ecological processes are essentially unmanageable. The scale, complexity, and unpredictability of ecological systems continues to elude scientific understanding. The uncertainty and complexity of natural and social systems is evidenced by the long standing debate over stability versus diversity, i.e. whether ecosystems with more diversity are more likely to contain complex food webs and feedback loops that confer resilience to disturbance and collapse (McCann 2000).

Adaptive management is an approach to management that explicitly acknowledges uncertainty, and therefore integrates design, management, and monitoring to systematically test assumptions in order to adapt and learn (Salafsky et al. 2002). It has been applied to ecosystem and natural resource management problems at least since 1978 as an approach to “learning-by-doing,” moving forward in the face of inevitable complexity and uncertainties, particularly where management decisions were controversial (Holling 1978). As such, adaptive management emphasizes the need to treat policies and decisions explicitly as hypotheses and opportunities for learning, requiring continuous monitoring, feedback, reflection, and revision. Over the past 30 years, many excellent publications have emerged on the practice, theory, history, case studies, and performance assessments of adaptive management (Lee 1995, Gunderson et al. 1995, Lee 1999, Nyberg 1999, Salafsky et al. 2002). Roling and Jiggins (2003) edited an excellent volume on using adaptive management to facilitate development and implementation of more ecologically sound agricultural systems and practices, through participatory approaches and appropriate institutional support and policy structures.

CONCLUSIONS: CRITICAL UNCERTAINTIES AND FUTURE RESEARCH

Although the literature contains many peer-reviewed studies and recommendations on local impacts of practices to maintain biodiversity, there are few quantitative data and predictive analyses on the landscape-scale biodiversity implications of different patterns of production systems, land use and management in agricultural landscapes. This is partly a function of the difficulty and complexity of doing ecological science at large scales and partly a function of failure of scientific paradigms and allocation of efforts (Levins and Lewontin 1985).

The presence of significant uncertainty should not serve as a justification for delaying implementation until "more research" is conducted. Rather we should strive to implement recommended strategies for biodiversity conservation as soon as possible and then monitor their performance in achieving the predicted or desired effects, using adaptive management to monitor and adjust as additional feedback is obtained.

Whether agriculture designed to be compatible with biodiversity succeeds in providing high quality habitat and buffering natural areas against more intensive uses depends largely on political will and quality of decision making at multiple scales. Whether agricultural lands serve as a source or sink for wildlife populations can hinge on management practices as simple as the timing of mowing (Frawley and Best 1991; Koford and Best 1996); thus many sustainable management practices must be tailored based on extensive local knowledge and understanding of natural ecosystem functions and the ways in which modification alters key processes and habitats. Yet we have an incomplete understanding of many factors affecting population dynamics and ecosystem functions in agricultural ecosystems.

Most studies of biodiversity on conservation or ecological farms are based on habitat use, but provide little data on the long-term quality of such habitat or the ability of such lands to sustain populations over time or landscape context. Additional assessment of fitness implications is needed, including identifying habitat features and management practices that influence whether such lands serve as habitat sources or sinks (Koford and Best 1996). Simulation modeling can be useful in evaluating potential actions (e.g., Cowardin et al. 1983). Status and trends of target species or guilds should be monitored to evaluate the results. On-farm research should be coordinated in the context of a comprehensive framework for conservation and management planning (Freemark et al. 1993, 1995). Many of the state wildlife action plans completed in late 2005 represent good reference points for habitat quality and conservation action information.

Additional guidance for policy-makers and land use planners could benefit from a meta-analysis exploring the implications for agricultural landscapes of adopting the recommended guidelines and quantitative restoration targets suggested by the literature. For example, DeVelice and Martin (2000) calculated that protection of roadless areas on national forests would increase the size of contiguous reserves and the number of ecoregions and vegetation types that have 12 percent of their area protected.Landscape scale modeling could examine on a regional and national basis the land use and cropland changes implied as well as the amount of habitat that would be created if recommendations such as 100m riparian corridors, 300 m shoreland and wetland buffers, and changes in floodplain use were adopted.

LAGGED RESPONSE TIME TO LANDSCAPE DISTURBANCE

Many species and populations exhibit lagged responses to habitat loss, especially in landscapes undergoing rapid change. Documented cases of extinction have been shown where the landscape changed more rapidly than the demographic response time of the population. For example, songbirds in landscapes undergoing rapid change might not be assessed as "at risk" until the population's demographic potential has been seriously eroded. Assessment of a species' extinction risk may vary widely among landscapes of similar structure, depending upon how quickly the landscape achieved its current state (Schrott et al. 2005). Less vagile species that require larger territories are likely to exhibit a response earlier (McLellan et al. 1986). Thus, information on the current landscape state (e.g., amount of habitat or degree of fragmentation) may not be sufficient for assessing long-term population viability and extinction risk in the absence of information on the history of landscape disturbance.

CAUSES OF DECLINES INCOMPLETELY UNDERSTOOD

Important ecological processes driving specific biodiversity declines are often incompletely understood, due to the cumulative nature and complexity of ecological threats and changes. For example, population trends of North American grassland birds remain poorly understood (Knopf 1995). As a group, grassland birds have declined more than birds of other North American vegetative associations. Unlike other neotropical migrants, which have experienced declines primarily in the northeastern deciduous forests (Robbins et al. 1989), declines in grassland species are occurring at a continental scale. The decline in numbers of the mountain plover, Cassin's sparrow, and lark bunting are occurring across their ranges. The lack of understanding of the wintering ecology of grassland birds precludes optimistic projections, especially for these species experiencing widespread, geographic declines.

CLIMATE CHANGE

The potential ecological impacts of climate change are significant, and will include substantial changes in duration, distribution, and character of key ecological processes and functions (NAST 2000, Covich et al., Johnson et al. 2005). Yet the uncertainties surrounding specific impacts of climate change on biodiversity are at this time virtually infinite.

VALUE OF BIODIVERSITY SERVICES

The science of measuring and communicating the economic value of biodiversity services is still in its infancy. A better understanding of the indirect and external costs and benefits of existing agricultural systems will be necessary in many cases to develop support for local and more conservation-based farming systems and methods (Daily 1997, Bjorkland et al. 1999, Alexander et al. 2001, Pretty et al. 2001).

ADAPTIVE MANAGEMENT: IMPLEMENT, MONITOR, AND ADJUST SIMULTANEOUSLY

The presence of significant uncertainty should not serve as a justification for delaying implementation until "more research" is conducted. Rather we should strive to implement recommended strategies for biodiversity conservation as soon as possible and then monitor their performance in achieving the predicted or desired effects as well as generating unexpected outcomes, using adaptive management to monitor and adjust as additional feedback is obtained. Indeed, many have suggested that adaptive management offers the only real potential for effective learning about large-scale ecological phenomena and management (Walters 1997).

CONCLUSION

Conservation of existing biodiversity will require substantial societal will, public education and involvement. It will be difficult to take the necessary actions without a shift in values and a broader understanding in society regarding the benefits of biodiversity or the need for more than just habitat restoration and conservation of a few isolated remnant natural areas. There is a critical need for private landowners to be involved in stewardship efforts and developing an ecological understanding of their own lands. Effective conservation will increasingly require integration of local knowledge, science, and deliberative democratic participation in land and natural resource management decision and policy-making (Schwarz and Thompson 1990; Gunderson et al. 1995; McNeely and Scherr 2003; Röling and Jiggins 2003).

Attempts to enact policy changes, acquire and manage land to meet conservation objectives or restore important ecological functions, such as reintroducing fire, reconnecting rivers with their floodplains, removing ineffective flood control structures, or restoring large predators often meet with considerable resistance among segments of the public, interest groups, as well as local and national leaders in government. However, such resistance can be transformed over time. In Florida, managers overcame initial resistance to undertake large-scale restoration of the Kissimmee River and reconnect it with its floodplain. The recognition that the initial channelization project had been an ecological mistake led to an investment in restoration many times greater than the initial project cost.

Some inevitably view such changes as unnecessary "sacrifice" or "tradeoffs." But if we do not make them voluntarily, similar tradeoffs and sacrifices will ultimately and undoubtedly be imposed upon us and our children in the future, in response to the unsustainability of our current agricultural and food systems.

The cost of creating networks of conservation lands and of supporting agricultural systems that are compatible with conservation objectives is undoubtedly large. However, the benefits are numerous, redundant, mutually reinforcing, and in the long run by far the most cost-effective. For example, wetland habitat protection and restoration provides not only biodiversity services but may substantially reduce required expenditures for flood damages, water quality protection, and endangered species management programs. Proactive conservation costs are often considerably lower than the costs of mitigation after-the-fact for natural disasters made more expensive by poor land use. For example, the flood of 1993, which caused, by some estimates, $16 billion of damage, could have been contained within 13 million acres of restored wetlands in the upper Mississippi basin (Hey and Philippi 1995).

Furthermore, when viewed in light of total spending and outlays under the federal Farm Bill, acreage restoration goals and targets articulated in various ecosystem-scale assessments seem reasonable. The estimated $2.7 billion annual cost of restoring 24 million riparian and wetland acres in the Mississippi Basin sufficient to effectively reverse Gulf of Mexico hypoxia is in the neighborhood of existing annual spending on the Conservation Reserve Program. If the funds could be used for reducing nitrogen, then commodity price control and nitrate reduction would be advanced simultaneously. This cost represents only a fraction of annual farm bill spending. Similarly, the initial “ballpark” cost estimate derived by Shaffer et al. (2003) for securing a national system of habitat conservation areas of between $5 billion and $8 billion annually for 30 years, represents roughly one-fourth to one-third the cost of maintaining the national highway system over the same period.

APPENDIX A

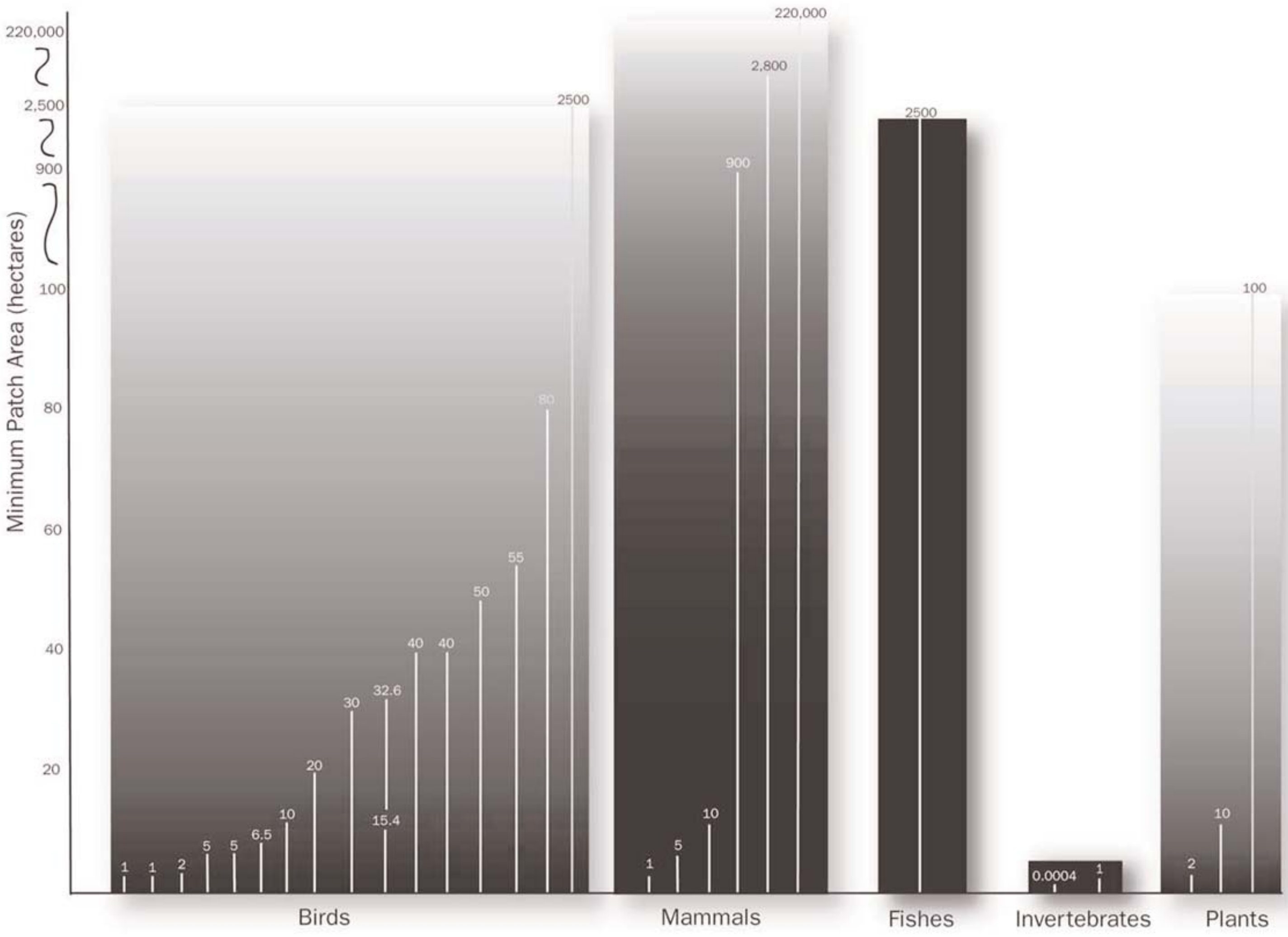

Figure 1. Minimum patch area requirements (in hectares) needed to maintain populations or communities of birds, mammals, fishes, invertebrates, or plants species in the United States, as cited in the scientific literature. Numbers represent the recommended minimum patch area sizes; two numbers along one line indicate a recommended range (*see* Appendix A for specific findings). Lines extend from zero to the recommended minimum patch area sizes to indicate the span of habitat needed for protection.

Kennedy, C., J. Wilkinson, and J. Balch. 2003. *Conservation Thresholds for Land-Use Planners.* Washington, D.C.: Environmental Law Institute. Page 11.

APPENDIX B

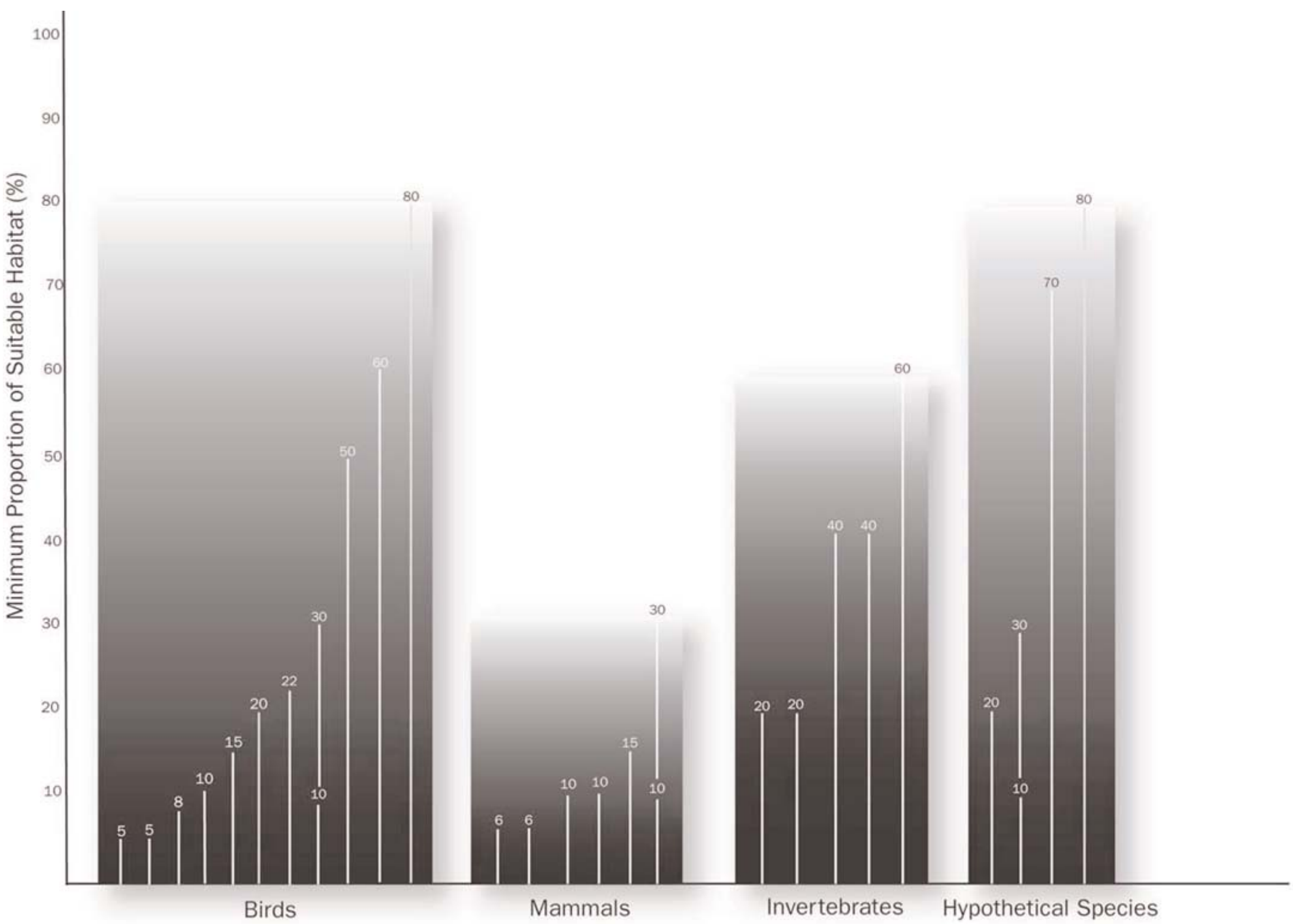

Figure 2. Recommended minimum proportions of suitable habitat (in percentages) needed to maintain populations or communities of birds, mammals, invertebrates, or hypothetical species (as determined by models) in the United States, as cited in the scientific literature. Numbers represent the recommended minimum proportions of habitat; two numbers along one line indicate a recommended range (*see* Appendix B for specific findings). Lines extend from zero to the recommended proportion to indicate the span of habitat needed for protection.

Kennedy, C., J. Wilkinson, and J. Balch. 2003. *Conservation Thresholds for Land-Use Planners.* Washington, D.C.: Environmental Law Institute. Page 15.

APPENDIX C

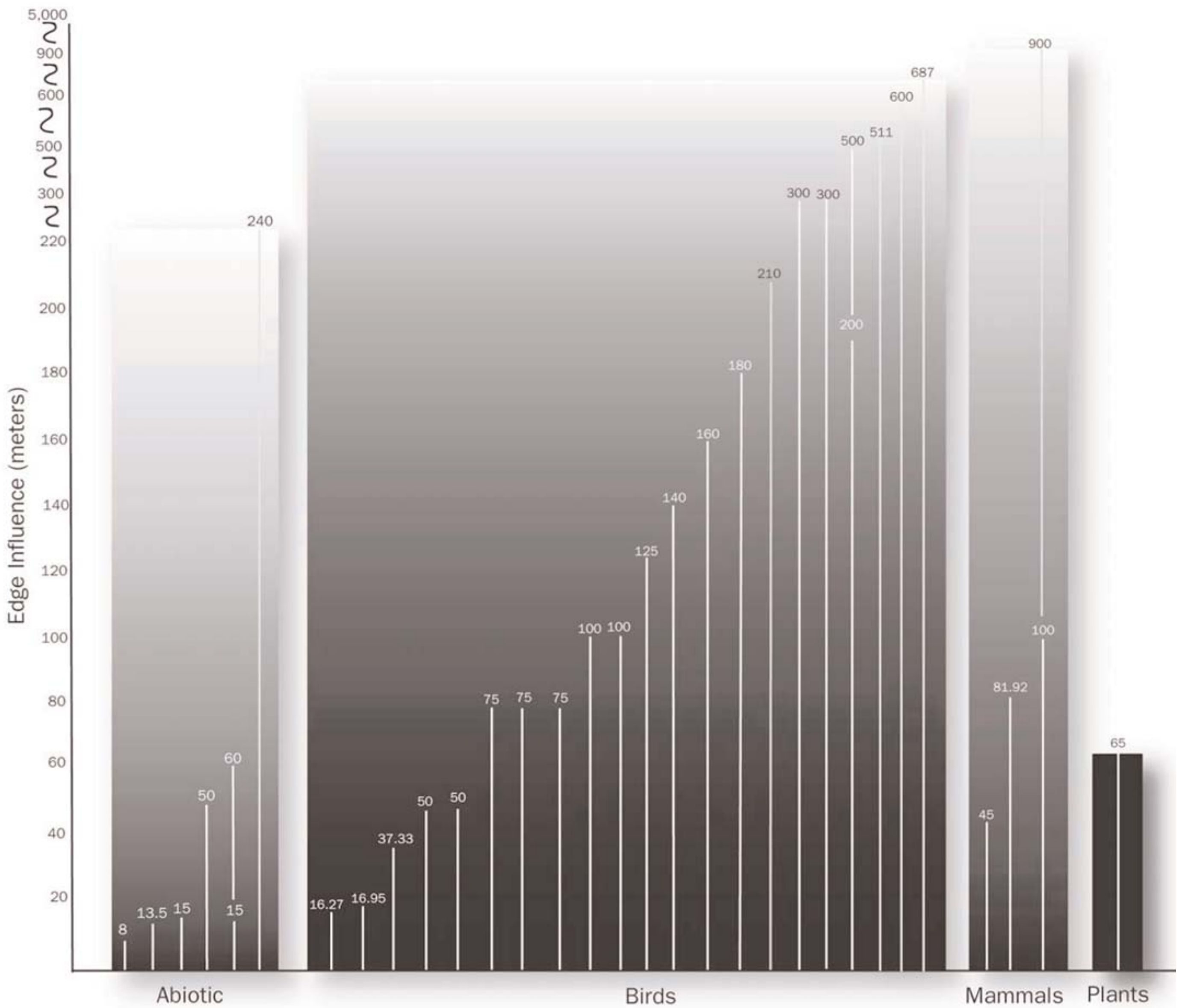

Figure 3. Distances (in meters) that edge effects penetrate into habitats in the United States, as cited in the scientific literature. E width is measured by abiotic, bird, mammal, or plant responses; abiotic responses include microclimate changes, such as changes temperature, humidity and light. Numbers represent edge width distance findings; two numbers along one line indicate a range of ed width distance (*see* Appendix C for specific findings). Lines extend from zero to the determined edge widths to indicate the span of tat that is affected by edge effects.

Kennedy, C., J. Wilkinson, and J. Balch. 2003. *Conservation Thresholds for Land-Use Planners.* Washington, D.C.: Environmental Law Institute. Page 18.

APPENDIX D

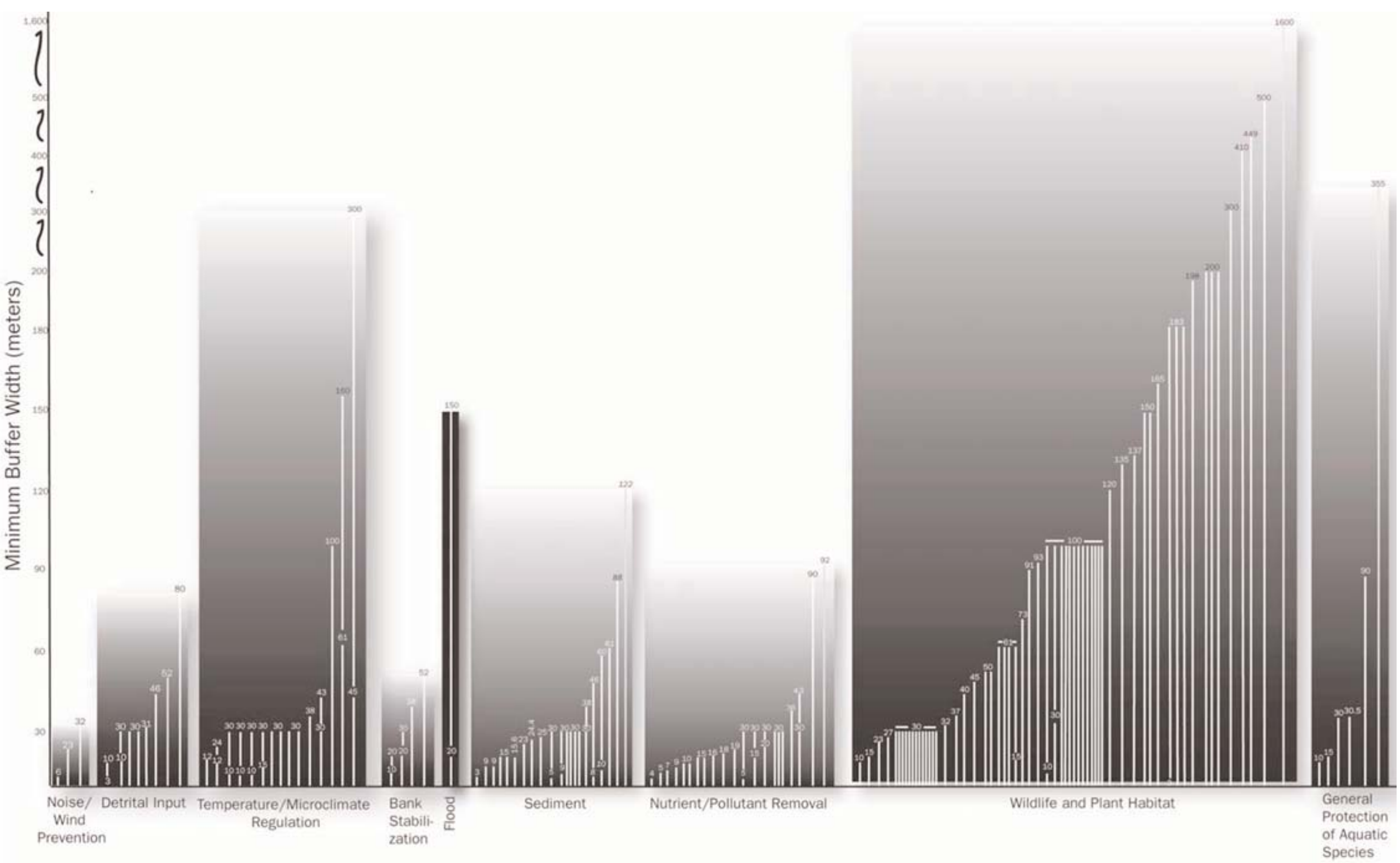

Figure 4. Recommended minimum riparian buffers (in meters) from each side of a water body (e.g. stream bank) needed to prevent noise/wind damage; provide detrital input; moderate temperature/microclimate; stabilize banks; provide flood attenuation; control sediment; reduce nutrients/pollutants; and provide wildlife habitat functions and general protection of aquatic systems in the United States, as cited in the scientific literature. Numbers represent the recommended minimum buffer widths; two numbers along one line indicate a recommended range (*see* Appendix D for specific findings). Lines extend from zero to the recommended buffer widths to indicate the span of habitat needed for protection.

Kennedy, C., J. Wilkinson, and J. Balch. 2003. *Conservation Thresholds for Land-Use Planners.* Washington, D.C.: Environmental Law Institute. Page 22.

Table 3.
Wildlife Use of Various Sized Habitats

Area	Forest/treed swamp	Marsh
1 hectare	• Edge tolerant mammals (Gray Squirrel) • Common edge-tolerant birds (Blue Jay, American Crow) • A few birds may be associated with mature trees (Black-capped • Chickadee, Eastern Wood-Pewee)	• Small populations of Muskrat • Edge-tolerant birds (Red-winged • Blackbird, Canada Goose, Mallard) • Persistent and common herpetofauna (such as Green Frog and Midland Painted Turtle)
4 hectares	• A very few common edge-tolerant birds (Downy Woodpecker, • Great Crested Flycatcher) • Eastern Chipmunk may be present	• Similar species as above, but may also support Bullfrog
10 hectares	• Still dominated by edge-tolerant species may have very small • areas of interior habitat supporting low numbers of modestly area sensitive species (Hairy Woodpecker, White-breasted Nuthatch)	• May support Marsh Wren, other waterfowl species
30 hectares	• May be large enough to support some species of salamander • Small populations of edge-intolerant species (Winter Wren, • Brown Creeper, Black-and-White Warbler)	• Similar marsh bird species as above, plus possibly Black Tern
50 to 75 hectares	• A variety of area-sensitive species may be present; some will • be absent if there is no nearby suitable habitat • Still predominantly edge influenced, but will support small • populations of most forest bird species • Some will be absent if there is no nearby suitable habitat	• Least Bittern may be present in • marshes of this size
100 to 400 hectares	• All forest-dependent bird species • Many will still be in low numbers and may be absent if there • is no nearby suitable habitat • Woodland Jumping Mouse may be present	• Small numbers of diving ducks possible (e.g., Redhead, Canvasback, Ruddy Duck)
1,000 hectares	• Suitable for almost all forest birds • Some forest-dependent mammals present, but most still absent	• All marsh species, although some may still have small populations
10,000 hectares	• Almost fully functional ecosystem, but may be inadequate for a few mammals such as gray wolf, bobcat, grizzly bear (100 000 ha has been suggested as a minimum)	• Largely-functional ecosystem

Source: Environment Canada 2002.

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