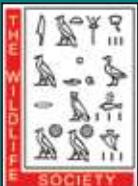




Fish and Wildlife Response to Farm Bill Conservation Practices



Technical Review 07-1
September 2007



A Partnership of the
Conservation Effects
Assessment Project

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This document is the second of two literature reviews focused on fish and wildlife and the Farm Bill. It is a conservation practice-oriented companion to the Farm Bill conservation program-focused literature synthesis released in 2005 (*Fish and Wildlife Benefits of Farm Bill Conservation Programs: 2000-2005 Update*, The Wildlife Society Technical Review 05-2).



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Executive Summary

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Conservation benefits of the Farm Bill are allocated through the various conservation programs including the Conservation Reserve Program (CRP), Environmental Quality Incentives Program (EQIP), Wildlife Habitat Incentives Program (WHIP), and other related programs. Each program has its stated purpose and operational guidelines. However, conservation incentives are actually accomplished through use of specific practices that are identified independently of the programs. Most of these practices can be utilized in more than one conservation program. For example, range planting is a practice that can be used in a project administered through CRP, EQIP, WHIP, or other conservation programs. While it is important to understand benefits to fish and wildlife accrued through use of conservation programs, it is also important to understand the benefits that have been documented for specific practices. This volume addresses conservation practices that can be used to provide fish and wildlife benefits through the Farm Bill. It does not specifically focus on investigations of actual Farm Bill funded projects, but rather summarizes investigations that have addressed various benefits or impacts to fish and wildlife resources associated with the primary practices utilized for fish and wildlife objectives within Farm Bill programs. The chapters in this volume do not attempt to provide a complete review of all literature pertaining to these practices, but rather to provide documentation of fish and wildlife responses reported in the literature. Chapters

are designed to address primary practices and their fish and wildlife benefits associated with croplands, established grasslands, linear conservation practices, native grasslands, wetlands, and aquatic ecosystems. In addition, a final chapter discusses the importance and need for use of adaptive management.

Brady (this volume) discussed the responses of fish and wildlife to the primary conservation practices used in croplands. He noted that agriculture has had the greatest effects on wildlife habitat of any anthropogenic cause. Many cropland conservation practices are targeted at reducing soil erosion. Reducing sediment delivery and run-off of agricultural pollutants will have positive effects on aquatic systems and species. He noted that such practices may also benefit wildlife populations when properly planned, but may have little or no benefits without this planning. He noted the importance of considering the landscape context in agricultural settings and the importance of providing appropriate plant communities and habitat elements within agricultural landscapes if wildlife benefits are to be provided.

Jones-Farrand et al. (this volume) discussed the wildlife benefits associated with the establishment of grasslands, focusing primarily on practices that apply to the Conservation Reserve Program, but that could equally apply to application of such practices in other programs. They reported substantial benefits to wildlife that have been produced through establishment of grasslands, especially in comparisons to wildlife benefits from row crop agriculture. This was espe-

cially true for bird populations that have received the most investigation. They noted a lack of research that has focused on responses to many other taxa. They also noted variability in wildlife responses and the need for additional investigations that included landscape analyses. Because of the complexities caused by differences in sites, size, and shape of established grasslands, surrounding landscape parameters, temporal factors, and other considerations, specific benefits to wildlife of grassland establishment will be species- and site-specific.

Clark and Reeder (this volume) discussed the benefits to wildlife of many linear practices that are used primarily in croplands for water and soil conservation, but that can also provide some benefits to wildlife. Example practices include filter strips, grassed waterways, buffers, contour strips, riparian strips, and windbreaks and shelterbelts. Their review of the literature revealed that the small area and high edge-interior ratios of these practices limited the benefits to wildlife. Most studies, as was found for establishing grasslands, focused on bird populations, and information on most other taxa are inadequate. Landscape influences also need additional attention. Clark and Reeder (this volume) concluded that with careful planning and management, various benefits to wildlife can be produced with linear practices, especially in comparison with the alternative of having areas remain in row crops.

Haufler and Ganguli (this volume) discussed wildlife responses to conservation practices applied on rangelands, with specific focus on the grasslands of the Great Plains. Investigations of wildlife responses to prescribed grazing reported both benefits and impacts to wildlife. Similarly, prescribed burning investigations also found both positive and negative responses by wildlife species, but generally burning produced favorable results for wildlife. Range planting and restoration of declining habitat were generally reported to produce positive benefits to wildlife, but a complicating factor was how to identify comparisons to treated areas. “Native” ecosystems were found to be poorly defined in many investigations. A number of studies revealed the need to enhance grassland heterogeneity, best defined in reference to ecosystems produced under historical disturbance regimes. This information has been lacking, so grassland investigations have used a variety of definitions

of “native” grasslands for comparative purposes. Other grassland practices were reviewed by Haufler and Ganguli (this volume) including fencing, pest management, brush management, and tree planting and shelterbelts. These practices were found to have both positive and negative effects on wildlife. Birds were the taxon most studied, with relatively few investigations of other taxa. More information on all species is needed, especially in terms of factoring in site effects, surrounding landscape conditions, and cumulative assessments.

Rewa (this volume) reviewed literature pertaining to wildlife responses to wetland practices. He reported similar findings to those of other chapters in this volume — that bird responses to practices have received the most attention. A majority of studies found that bird communities in restored wetlands were similar to those of natural wetlands. Wetland restoration was found to produce rapid responses by amphibians and invertebrates. Factors that influenced wildlife responses included size of restored wetlands, proximity to other wetlands, the age and complexity of a restored wetland, and the management of the wetland. As with other chapters in this volume, the chapter by Rewa (this volume) stressed the need for additional information on taxa other than birds and longer term studies on responses by all taxa.

Knight and Boyer (this volume) summarized the responses of aquatic species and their habitats to conservation practices. They reported benefits and impacts to fish and aquatic fauna produced by these practices. They stressed the importance and need for evaluating responses within watersheds, as aquatic resources are influenced by not only the direct practices occurring in aquatic ecosystems, but also those that influence the inputs to aquatic ecosystems. Knight and Boyer (this volume) reviewed a number of practices designed to reduce inputs of sediments, nutrients, or pesticides into aquatic ecosystems. They also reviewed many practices used to improve or maintain riparian or shoreline condition, which in turn helps maintain water quality and aquatic species and habitats. Other practices they reviewed included direct management of aquatic resources such as fish passages, fish pond management, pond establishment, shallow water management, and stream habitat improvement and management. In general, practices they reviewed help reduce impacts of agri-

cultural activities on aquatic ecosystems and produce benefits to aquatic species and their habitats. They noted some exceptions to this, where certain practices can result in impacts to various aquatic resources. They noted the complexity of variables influencing responses and reported on many additional information needs.

Franklin et al. (this volume) provided a description of adaptive management and stressed the importance of incorporating this concept in the monitoring of fish and wildlife responses to conservation practices. The need for additional information stressed in all of the previous chapters points to the need for new approaches to monitoring and documenting responses to Farm Bill practices. A systematic approach to defining expected responses and then monitoring if these responses were produced was described. Four case studies describing applications of adaptive management with implications for its use in monitoring Farm Bill practices were presented.

In total, the chapters in this volume provide a summary documentation of the numerous benefits to fish and wildlife that can be produced through Farm Bill practices. However, most practices can produce both positive and negative responses by different species, requiring that specific objectives be articulated as a basis for evaluating positive responses. The complexities of fish and wildlife responses with factors emerging at various scales make simple conclusions difficult. Much additional research is needed if responses to practices are to be adequately understood for effective planning. Responses by many taxa are virtually unknown. These information gaps emphasize the need for application of adaptive management in a systematic manner as part of an expanded monitoring program. ■



Effects of Cropland Conservation Practices on Fish and Wildlife Habitat

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ABSTRACT A literature review of commonly applied cropland soil and water conservation practices and their impact on fish and wildlife habitat is presented. Agriculture has had the most extensive effect on wildlife habitat of any human-induced factor in the United States. Any practice that improves runoff water quality and/or reduces sediment delivery will have beneficial effects to aquatic ecosystems. Many soil and water conservation practices have additional benefits to wildlife when applied in a habitat-friendly manner, but may have little or no benefit when applied otherwise. Wildlife and agriculture can coexist if land is managed to conserve sufficient biological integrity in the form of plant communities and habitat elements compatible with the surrounding landscape.

A variety of soil and water conservation practices are widely applied to croplands for the primary purposes of controlling soil erosion, managing runoff water, conserving soil moisture, improving soil quality, protecting crops, managing nutrients and pests, or otherwise avoiding soil degradation. While each conservation practice has specific primary purposes for application, many also affect other resources. Primary effects are often well documented in the literature and to some extent secondary effects are also recognized. Unfortunately, however, there is little documentation of broader ecological effects to other resources such as fish and wildlife habitat. Allen and Vandever (2003) studied Conservation Reserve Program (CRP) participants, reporting most farm

operators recognize economic, environmental, and societal benefits stemming from establishment of CRP conservation practices, with greater than 75 percent of farm operators responding to their survey identifying wildlife as an important product of their conservation activities. This paper reviews literature documenting effects of cropland soil and water conservation practices on fish and wildlife habitat. Cropland is defined here to include land used for the production of food, feed, fiber, and oil seed crops. This definition includes land used to grow row crops, close grown crops, orchards, vineyards, and tame hay, but excludes forest, pasture, range, and native hay (i.e., marsh hay or wild hay). The term habitat is used generically in this discussion to refer to resources or conditions present that will produce

occupancy by some wildlife species. Proper use of the word habitat requires a species-specific definition (Hall et al. 1997), which is impractical in this review.

The goal of reducing soil erosion rates down to the tolerable level has been based on soil characteristics for continued production. Each soil map unit is assigned a tolerable soil loss limit or “T-value” to represent the amount of erosion loss it can withstand without sacrificing long-term productivity. Soil characteristics such as depth of the A horizon, depth to bedrock or other restricting layer, texture, and similar attributes help determine the tolerable limit for each soil map unit. T-values typically range from 1 to about 4 or 5 tons/acre/year (2.2 to 9 or 11.2 tons/ha/year). While the T-value is a useful concept for maintaining long-term sustainability of the site, there are conditions on the landscape where those values could result in excessive sediment delivery to receiving waters to the detriment of fish and other aquatic organisms. In addition to the T-value and soil sustainability concerns, site conditions in relation to receiving waters should be considered when evaluating soil conservation treatment alternatives for cropland.

There were 369.7 million acres (149.6 million ha) of cropland in the 48 conterminous states in 2001 (USDA NRCS 2003) representing about 27 percent of nonfederal rural land. Nearly 85 percent of cropland is cultivated annually while the remainder is used to produce perennial or semi-perennial crops. About 56 percent of cropland is classified as prime farmland, while 27 percent is classified as highly erodible land (HEL). Soil erosion rates were at, or below, the tolerable level on about 72 percent of all cropland in 2001. From 1982 to 2001 soil erosion rates on all cropland declined from 3.1 billion tons (2.8 billion metric tons) per year to 1.8 billion tons per year (1.6 billion metric tons) (USDA NRCS 2003), a net reduction of 1.3 billion tons per year (1.2 billion metric tons), or 42 percent. One can only conclude that extensive conservation treatment has been applied to achieve this significant reduction. However, 18 percent of the non-HEL and 55 percent of HEL cropland still exhibit soil erosion rates greater than the tolerable level (USDA NRCS 2003). This represents 103.8 million acres (42 million ha) of cropland, or 28 percent, where additional conservation treatment is needed immediately.

While cropland soil conservation practices can affect the quality of fish and wildlife habitat, it needs

to be recognized that land use is the principal factor determining the base level of abundance of endemic wildlife species in agricultural ecosystems (Edwards et al. 1981). The extent and intensity of land use determines how much of the landscape is available as wildlife habitat since land use determines the kinds, amounts, relative permanence, and distribution of vegetation. The extent to which cropland conservation practices enhance or diminish the landscape’s ability to meet habitat needs of terrestrial wildlife is a function of how significantly conservation complements the mix of perennial or residual cover types. Wildlife habitat management is largely based upon managing plant communities and related resources to furnish fundamental needs such as cover and food for wildlife. In agricultural ecosystems, this often includes using agronomic practices and crops in the management plan. The literature is replete with studies documenting wildlife response to various vegetation and land management practices (e.g., nesting cover, winter cover, food plots, etc.). However, little has been published documenting specific effects of most soil and water conservation practices on terrestrial wildlife habitat. The same is true for wetland and aquatic habitats; however, conservation practices that reduce soil erosion and sediment delivery or that otherwise improve the quality of runoff water (e.g., vegetative filter or buffer strips) play significant roles in improving aquatic habitat quality.

Agricultural Land Use Effects on Habitat

Perhaps no human activity has had a more profound impact on American wildlife than has agriculture (Burger 1978). Farris (1987:2) concluded that “farm legislation has a greater impact on wildlife habitat than any other human-related factor in this country, including all of our combined wildlife management efforts.” Initially, as forest and prairies were converted to agricultural uses, there were positive responses by some species to habitat openings and additional food resources that agriculture provided. However, most wildlife species began to decline when agriculture expanded to the point of replacing extensive tracts of native habitats. Variability among wildlife species exists in their ability to respond to agricultural land use intensification; however, for many

species there are thresholds of disturbance beyond which further agricultural expansion or intensification is not tolerated. Those thresholds vary by species as well as by landscape setting; consequently, definitive thresholds have not been defined. An analysis of breeding birds in Iowa agricultural landscapes (Best et al. 1995) found potential numbers of nesting species increased from 18 to 93 over four landscape management scenarios representing a progression from intensively farmed row crop monoculture to a diverse mosaic of crop and non-crop habitats.

The following discussion furnishes a brief summary of land management and technological changes driving agricultural land use intensification that have affected the quality and distribution of wildlife habitats and populations associated with agricultural ecosystems. More specific details are available in the following references: Baxter and Wolfe (1973), Burger (1978), Taylor et al. (1978), Samson (1980), Edwards et al. (1981), Warner et al. (1984), Warner and Etter (1985), Wooley et al. (1985), Potts (1986), Robbins et al. (1986), Berner (1984, 1988), Brady (1985, 1988), Brady and Hamilton (1988), and Flather and Hoekstra (1989), Warner and Brady (1994), Flather et al. (1999), Heard et al. (2000), and Higgins et al. (2002).

Agricultural land use effects were first manifest by extensive conversion of native habitats to diversified, small-scale agricultural production. Forest and wetland wildlife were dramatically impacted while shifts in presence, abundance, and distribution of grassland wildlife occurred somewhat gradually at first. The mixed agricultural landscape coupled with low intensity farming practices retained connectivity among habitat patches. As native prairie was converted to non-native forage grasses and legumes, many grassland birds were able to persist because this pseudo-prairie was structurally complex and heterogeneous. Between the early 1900s and 1950 in Illinois, for example, there was little change in most grassland bird populations (Forbs and Gross 1922, Graber and Graber 1963), as introduced forage grasses and legumes offered a pseudo-prairie for most grassland birds (Warner 1994). These forage crops were important for livestock production and legumes were important to supply nitrogen in rotation with grains. Soon after World War II, horses were replaced by machinery, greatly reducing the need for forages, and

nitrogen became commercially available, eliminating the need for legumes in rotations. The growing presence of livestock confinement facilities and feedlots further reduced the need for pasture and rangeland as agriculture became even more industrialized and landscapes became less diverse in the crops produced and habitat provided. Improved varieties of alfalfa replaced mixed forage stands (Warner 1994) and the development of improved crop varieties, herbicides, and pesticides further permitted row crop agriculture to expand (Burger 1978). Transportation and marketing developments along with vertical integration of businesses allowed specialized agricultural products to be produced where natural conditions were most optimum, then shipped fresh to markets. Farms and rural grain markets became specialized and many landscapes became dominated by just one or two crops. Grassland birds typically declined in relative abundance by 80 percent to more than 97 percent during this period (Graber and Graber 1963, Robbins et al. 1986, Herkert 1991, Warner 1994). During the 30-year period beginning in 1956, dramatic declines in the hunter harvest of ring-necked pheasants (*Phasianus colchicus*) and northern bobwhite quail (*Colinus virginianus*) in Illinois were highly correlated with increasing amounts of row crops, while declines in the harvest of cottontail rabbits (*Sylvilagus floridanus*) were highly correlated with declines in hay and small grains (Brady 1988). At the same time, survival of ring-necked pheasant chicks to 5 to 6 weeks of age declined from 78 percent to 54 percent (Warner 1979). This decline was the result of fewer acres of forage crops, small grains, and idle areas where chicks forage for insects. Consequently, due to the diminished presence of suitable cover and less available food, the area needed to ensure survival of pheasant broods nearly tripled (Warner 1984, Warner et al. 1984).

Soil and Water Conservation Practice Effects on Habitat

Generally, as soil conserving measures increase, upland wildlife habitat quality also improves (Lines and Perry 1978, Miranowski and Bender 1982). Direct changes in land use can have greater effects on habitat quality than changes in management practices can (Miranowski and Bender 1982). This is illustrated by

data from Illinois where between 1967 and 1982, a 46 percent decline in the harvest of farmland game was attributed to a 48 percent increase in area of “cropland adequately treated” for soil erosion control (Brady and Hamilton 1988). However, during the same period the proportion of cropland used for row crops increased from 70 percent to 85 percent. Within the context of the landscape setting and with the assumption that certain minimum habitat elements are available, then cropland conservation practices can have a beneficial effect on fish and wildlife habitat. However, they represent the last increment of habitat elements within the landscape context. Soil and water conservation practices offer benefits to wildlife only when installed to complement existing habitat within the landscape setting. Of course any practice that improves runoff water quality or reduces sediment delivery is beneficial to aquatic systems. In most cases, selection of soil and water conservation practices that also benefit wildlife requires land users to choose features that enhance wildlife habitat from among unequal options. For example, native grasses such as switchgrass (*Panicum virgatum*) may furnish greater long-term and seasonal benefits to wildlife than introduced grasses such as smooth brome (*Bromus inermis*).

In the following section, the effect on fish and wildlife habitat of commonly applied soil and water conservation practices is discussed. Some conserva-

tion practices were combined together for discussion as appropriate. Definitions and purposes of each practice are provided in Appendix A. Published literature is reviewed, but there is a paucity of relevant literature documenting specific effects for many practices on wildlife and their habitats.

Conservation Tillage

(residue management; no-till, strip-till, mulch-till, ridge-till)

Conservation tillage is practiced on more than 111 million acres (45 million hectares) world-wide, primarily to protect soils from erosion and compaction, to conserve moisture, and reduce production costs (Holland 2003). The agronomic values of conservation tillage are generally very good, accounting for its widespread adoption. It is also believed this conservation practice generally improves habitat values of crop fields for some wildlife species. Various forms of intermediate tillage (strip or mulch tillage) may be used to chop or shred crop residue to facilitate planting, or to incorporate soil amendments or pesticides, all of which reduce the value of the cropland to wildlife, due to additional disturbance as well as diminished availability of cover and food resources.

Robertson et al. (1994) studied soil-dwelling invertebrates in a semi-arid agro-ecosystem in northeastern Australia. They reported that the highest population densities of detritivores and predators occurred in zero-tilled fields while conventional cultivation displayed the lowest abundance. Populations of these beneficial invertebrates in reduced tilled fields were intermediate. The numbers of herbivorous soil insects were similar between tillage treatments at each sampling time. The authors concluded zero tillage may further increase the ecological sustainability of agro-ecosystems by maintaining high populations of soil-ameliorating fauna and predators of insect pests. Altieri (1999) explored the role of biodiversity as it pertains to crop protection and soil fertility. He suggests the persistence of biodiversity-mediated renewal processes and ecological services depend on the maintenance of biological integrity and diversity in agro-ecosystems. No-till fields have a greater abundance and diversity of arthropods than conventionally tilled fields. This increased diversity was reported to be the result of greater abundances of beneficial insects (Blumberg and Crossley 1983, Warburton and Klimstra 1984). While many of these arthropods are



No-till production techniques for soil conservation in Alabama. (Photo courtesy of USDA NRCS)

important food resources for birds and mammals, Basore et al. (1987) found no increase in insect numbers in no-till fields vs. conventionally tilled fields during the pheasant brood rearing period in Iowa.

Several studies report on nesting and nest success of birds in minimum tillage crop fields. Best (1986) suggested minimum tilled crops represent ecological traps that attract nesting birds away from safer habitats only to see the nests destroyed by subsequent farming operations. Certainly this could happen, especially in ridge-till systems where cultivation is required. Cropping systems that reduce the number of field operations should be used where possible and maximum amount of crop residues should be retained on the soil surface (Rodenhouse et al. 1993).

Warburton and Klimstra (1984) found a greater abundance of invertebrates, birds, and mammals in no-till than in conventionally tilled cornfields in southern Illinois. Castrale (1985) found deer mice (*Peromyscus spp*) to exhibit a negative relationship with residue amounts, while house mice (*Mus mus*) were more dependent on greater residue in no-tilled row crop fields. Clark and Young (1986) reported no relationship between deer mouse abundance and the varying residue amounts in conventional vs. no-till row crops. The increased residue amounts created by no-till generally result in greater diversity rather than density of small mammals. Concerns over crop damage by small mammals in no-till fields are not warranted (Stallman and Best 1996) in crop fields. However, that may not be true where corn is no-tilled into pasture or hayfields (Best 1985).

Basore et al. (1986) found substantially greater diversity and density of birds nesting in Iowa no-till fields (12 species, 36 nests/247 acres or 100 ha) than in conventionally tilled fields (4 species, 4 nests/247 acres). Nest success was comparable to levels recorded in idle areas, such as fencerows and waterways. Duebbert and Kantrud (1987) found that minimum tillage in fall-seeded crops was more attractive and productive for nesting ducks than was conventional tillage in North Dakota. Nest success was 27 percent for 5 duck species and nest density was 7 nests/247 acres (100 ha). Cowan (1982) found nest density was 1.4-1.5 times greater in no-till fields, and duck nest success in no-till winter wheat was 42 percent vs. 13 percent on conventionally tilled farms. Loekmoen and Beiser (1997) report equivalent, or higher, nest

success in minimum tillage fields than recorded within conventionally tilled fields.

Martin and Forsyth (2003) studied bird use of fields used for spring cereals, winter wheat, and summer fallow farmed using either conventional or minimum tillage (i.e., no-till or strip-till) in southern Alberta, Canada. The authors found savannah sparrows in spring cereal and winter wheat and chestnut-collared longspurs in summer fallow tended to prefer minimum tillage. Minimum till spring cereal and winter wheat were more productive for savannah sparrows (*Passerculus sandwichensis*) than were conventionally tilled habitats. Summer fallow of either tillage regime did not appear to be as productive as were minimum tilled cereal fields for savannah sparrows. Chestnut-collared longspurs (*Calcarius ornatus*) occurred predominantly in minimum till summer fallow and spring cereal habitat. McCown's longspurs (*Calcarius mccownii*) tended to have higher productivity in minimum till plots. The authors concluded that minimum tillage appeared to confer benefits in productivity to bird species that nested in farmland. Shutler et al. (2000) reported higher relative abundance of 37 upland bird species in Saskatchewan on wild than on farmed sites, as well as higher abundance on minimum tillage than on conventionally tilled farms.

Cotton generally provides the least suitable habitat for most early successional songbirds among the major agricultural crops in the southeastern United States due to the high intensity of tillage practices and dependence on pesticides to maintain productivity. Cederbaum et al. (2004) reported both conservation tillage and clover stripcropping systems improved conditions for birds in cotton, with strip-cropped fields providing superior habitat. Although the clover treatment attracted the highest avian and arthropod densities, conservation tilled fields still provided more wildlife and agronomic benefits than did conventional management.

Rodenhouse and Best (1983) reported vesper sparrow (*Poocetes gramineus*) nests produced an average of 2.8 young/pair in conventionally tilled croplands, probably below replacement levels. They suggested breeding success likely would be greater if the number of tillage operations was reduced and crop residue was retained on the fields. These authors (1994) also reported on foraging patterns

of vesper sparrows in Iowa corn and soybean fields, concluding the sparrows preferred to forage in fields with the most crop residue. Therefore, reduced tillage farming methods may enhance foraging opportunities for this species.

Crop residues left undisturbed over winter furnish additional wildlife benefits from conservation tillage. Undisturbed harvested crop fields receive greater use by wintering wildlife than do fall-tilled crop fields in Indiana (Castrale 1985). The waste grain is an important source of energy for many wildlife species. (Baldassorre et al. 1983). However, that benefit is compromised when intermediate tillage methods are employed. Multiple-pass tillage operations commonly used for corn, or single-pass tillage with twisted shank chisel plows, may be as detrimental to the availability of waste grain as the moldboard plow (Warner et al. 1989).

Pesticide effects were neatly summarized in the NRCS Wildlife Habitat Management Institute's literature review (USDA NRCS 1999):

Although the increased attractiveness of no-till crop fields as nesting and brood rearing habitat was shown to have potential pesticide exposure, Little (1987) pointed out that greater usage of herbicides was not necessarily required for no-till or reduced tillage farming. Flickinger and Pendleton (1994) reached the same conclusion in a Texas study that measured the use of herbicides in reduced and conventionally tilled fields. In addition to conservation tillage not having to greatly increase the use of herbicides and insecticides above those used in conventional tillage, some work has shown that less toxic choices are available. Some herbicides, such as glyphosate, are very low in toxicity and have little direct impact on nests (Cowan 1982, Castrale 1985, Nicholson and Richmond 1985). Although insecticides also are of concern, Best (1985) noted that insecticide use had more to do with cropping sequence than tillage practices. Also, recent studies of the impacts of direct spraying and the consumption of poisoned insects on bobwhite quail chicks in North Carolina showed that modern

insecticides are less toxic than those used in the past (Palmer et al. 1998).

In summary, conservation tillage systems, i.e., no-till, have widely been reported to provide improved habitat values over conventional tillage systems. Reports consistently indicate no-till fields have greater densities and more species of birds than found within conventionally tilled fields. In relation to the needs for wildlife habitat, the best systems are those leaving the greatest amounts of crop residue on the surface and those having the fewest number of disturbances from farming operations. Mulch-till systems may meet soil conservation standards, but the intermediate tillage treatments they employ adversely affect wildlife food and cover.

Grassed Waterways

Grassed waterways have been extensively established to safely remove concentrated flows of runoff water from agricultural fields. The size of grassed waterways is highly variable depending upon topography, soil texture, and local rainfall patterns. Typical waterway size in Illinois or Iowa is about 35 to 60 feet (11-18 m) wide with lengths ranging from a few hundred feet to nearly one-half mile (60-800 m). Bryan and Best (1991) reported 48 species using smooth brome grass waterways during the breeding season in Iowa, compared with only 14 species using adjacent corn and soybean fields. Total bird abundance was also



Grassed waterway in an agricultural field in Missouri. (Photo by C. Rahm, USDA NRCS)

higher, averaging 2,198 birds observed/census/247 acres (100 ha) in waterways, compared with 682 in crop fields. The peak of bird species abundance (53 percent) occurred during July 4 to July 22. The temporal patterns in bird abundance were attributed primarily to aspects of the waterways and surrounding cropland that changed over time, such as vegetation height. In a subsequent paper (1994) these authors reported 10 bird species nested in waterways, achieving a nest density of 1,104 nests/247 acres (100 ha). Nest success was low (8.4 percent red-winged blackbirds, 22 percent dickcissels), with 57 percent of all nest losses due to predation, while 16 percent of nests lost were attributed to mowing. The authors believed nest success could be increased by delaying mowing until late August or September. Grassed waterways also are assumed to provide habitat value during other seasons of the year, but those have not been documented.

Bryan and Best (1994) noted, "Annual mowing is not necessary to maintain grass vigor after the waterway is established; however, mowing every three to four years may be required." This statement is correct as it relates to grass vigor, but it is in conflict with NRCS guidance for waterway maintenance. Grassed waterways are designed to have a convex or trapezoidal shape with maximum depths ranging from about 1 to 3 feet (0.3-1 m) deep. They are typically designed with capacity to carry runoff from the 10-year storm event at a non-erosive velocity to a stable outlet. The grass type, slope, and shape help determine the hydrologic retardance factor. Waterways typically are densely seeded to grasses such as smooth brome or tall fescue and designed based upon the assumption of regular mowing. The purpose of regular mowing is to maintain velocity and encourage grass density by production of rhizomes and tillers. As grasses grow taller, hydrologic retardance increases, causing a reduction in the runoff velocity. Sediment is deposited into the dense sod as runoff velocity decreases, causing the waterway ultimately to lose capacity. Sediment then builds up in the waterway to the point that it can no longer receive runoff from the adjacent field. The water then runs down the unprotected (i.e., cropland) sides of the waterway, causing additional gullies. Typical cost (in 2005) to build a grassed waterway ranges from about \$2,000 - \$2,400 per acre (Gene Barickman and Mark Lindflott, personal

communication). Wetter site conditions also may require drainage tile for part or all of the length of the waterway, adding an additional \$1.25 to \$2.00 per linear foot. Waterways with taller grasses (or a higher mowing height) to benefit wildlife can be accommodated during the planning phase by designing for higher water velocities. However, all grassed waterways require good maintenance to ensure proper functioning and protection of investment.

Grade Stabilization Structures

These structures are installed to control gully erosion and to reduce head cutting uphill. Grade stabilization structures are often required at the downstream end of a grassed waterway to provide a stable outlet. Grade stabilization structures may be made of concrete, corrugated metal, or treated lumber and are designed to handle concentrated flows. These structures typically have berms on each side to direct water over the notch or toward the inlet of a pipe in front of an earthen dam. The berm or dam is designed to provide temporary storage of water while it is released at a controlled rate (determined by the weir or pipe size). On-farm applications typically are designed for the 10-year storm event to flow through the pipe or over the weir with temporary water storage up to the 25-year storm event behind the berms or dam. Peak storm flows in excess of the 25-year event would be routed around the berms to an emergency spillway. Grade stabilization structures provide wildlife habitat to the extent that they permit small terrestrial and wetland habitats to develop with associated shallow pools that may be permanently or seasonally flooded.

Little has been published about the wildlife benefits of grade stabilization structures with the exception of pipe drop structures. The latter have been studied in Mississippi. Smiley et al. (1997) recorded 100 species of vertebrate wildlife using the habitats created by pipe drop structures. The highest species richness at pipe drop structures occurred in scrub-shrub and intermittent riverine wetlands. Habitat values are optimized with larger and deeper pool sizes and a buffer of robust grasses to trap sediment before it is delivered to the pool area. Cooper et al. (1997) reported the highest percent capture abundance among all habitat types occurred with amphibians, followed by fish, birds, mammals, and

reptiles. Habitat benefits were minimal for sites smaller than 0.2 ac (0.08 ha), sites lacking woody vegetation, and sites that did not have at least 20 percent of their area below the inlet weir elevation (Shields et al. 2002).

Grass Backed and Grass Ridged Terraces

Terraces have been extensively used to manage runoff water and reduce sheet erosion. Terraces are best suited to deep soils on long gentle slopes but are poorly suited to soils that are shallow (to bedrock) or occur on short, choppy slopes where contour farming is difficult. Terraces may be broad-based and farmed or may be narrow-based with grassed ridges or grassed back slopes. Grassed back slope terraces are usually built on steeper sites, while the grass ridged terraces are narrow-based (about 10 to 14 feet wide, or 3 to 4.3 meters) and more appropriate for slopes. Grassed terraces are less expensive to build than are broad-based terraces, but the grassed portion is lost from crop production. Broad-based terraces have no direct benefit to wildlife, but the grassed terraces increase the diversity and interspersion of vegetative types in cropland settings. Terrace construction could lead to the loss of habitat if waterways are replaced with underground tile outlets or if new field alignments remove old, grown-up fencerows and odd areas of habitat.

Hultquist and Best (2001) observed 26 bird species using grassed terraces in Iowa. Red-winged

blackbirds and dickcissels accounted for 58 percent of the total bird abundance. Bird abundance in terraces was less than in other strip-cover habitats such as grassed waterways and roadsides, but greater than in rowcrops. However, all terraces evaluated were dominated by smooth brome grass averaging over 70 percent cover. Therefore, results may be different on terrace systems with greater plant diversity or those dominated by native warm season grasses and/or forbs, which generally are believed to provide greater quality habitat for wildlife.

Beck (1982) reported 35 species of vertebrates using grassed back slope terraces in Iowa. Additionally, he reported pheasant nest success was 22.5 percent, or one successful nest per 12.5 acres (5 ha) of grass in these terraces. While this density is low, it is an improvement over no grassy cover or no nests at all from broad-based terraces.

Filter Strips and Field Border Strips

These two practices have been combined for discussion because their ecological effects are similar. Filter strips are established between agricultural fields and “environmentally sensitive” areas such as streams and aquatic systems. Field border strips are established around the perimeter of crop fields. Filter strips reduce erosion, trap sediments, filter pollutants, and provide wildlife food and cover. Few studies have been reported on these two practices until recently. Both practices have become increasingly popular as a result of the USDA National Conservation Buffer Initiative and the Conservation Reserve Program practice “CP33” (Bobwhite Buffers). The latter provides land rental payments to land users who participate.

Puckett et al. (2000) examined how the addition of filter strips around crop fields and along crop field drainage ditches impacted northern bobwhite quail in North Carolina. The authors reported that the presence of filter strips shifted habitat use patterns, especially during spring and early summer, and improved crop fields as habitat for breeding bobwhite quail. Bobwhites occurring on filter strip sections of their study area had significantly smaller breeding season ranges than those captured where filter strips were not present. Filter strips have the potential to increase quail recruitment by providing what is often the only



Example of strip-till production, an intermediate tillage technique. (Photo courtesy of USDA NRCS)

available nesting and brood-rearing cover during spring and early summer (Puckett et al. 2000).

Smith et al. (2005a) reported field border effects over winter differed by bird species and adjacent plant community types in Mississippi, but greater densities of several sparrow species were observed along most bordered transects. Smith et al. (2005b) also studied bird response to field borders during the breeding season and concluded from their Mississippi study that “within intensive agricultural landscapes where large-scale grassland restoration is impractical, USDA conservation buffer practices such as field borders may be useful for enhancing local breeding bird richness and abundance.” Smith (2004) suggested the percentage of the land base established in field borders may play a greater role in eliciting population responses of northern bobwhite than field border width. Smith (2004:87) summarized his results with this statement: “Therefore, given my results in the context of those reported in Puckett et al. (1995, 2000) and Palmer et al. (Tall Timbers Research Station, unpublished data), I suggest that at least 5 percent to 10 percent of a site be placed in field border habitats to elicit measurable responses from northern bobwhite populations. USDA conservation practices, such as the recently announced CP-33 practice, may provide opportunities to enhance northern bobwhite habitat with minimal changes in primary land use.”

Conover (2005) conducted a three-year study to evaluate the response of breeding and wintering avian communities to field borders in an agricultural landscape in Mississippi. Results from his study revealed substantial avian benefits provided by field borders. Field border habitat generally provided greater avian richness, abundance, and conservation value over traditional “ditch-to-ditch” row-crop practices. Field borders were particularly valuable if established at widths greater than 33 feet (10 m) and when vegetative composition was dominated by forbs. During the breeding season nearly all species that commonly inhabit field edges had significantly greater abundances on bordered margins. Avian richness, abundance, and conservation value were higher in bordered field margins and adjacent agricultural fields regardless of width. Avian response to field borders was variable by species. Dickcissels (*Spiza americana*) appeared to benefit mostly from wide

borders and were not abundant on narrow-bordered margins. Nesting birds displayed extreme preference for wide border nest-sites. Dickcissel and red-winged blackbird (*Agelaius phoeniceus*) nest success estimates were comparable to other studies, suggesting field border habitat does not likely represent an ecological trap. Nest-site selection favored borders with increased forb composition over grass and greater vertical cover.

Kammin (2003) studied 92 filter strips in central Illinois and reported 89 species of birds using them. Seventeen species nested in filter strips, but 76 percent of 411 active nests were destroyed by predation. The author concluded filter strips provide adequate cover and food resources to support several bird species, but are only marginally suitable as breeding habitat due to elevated rates of predation.

Bromley et al. (2002) studied bird response to field borders in North Carolina and found that farms with field borders had higher nest density, particularly for field sparrows (*Spizella pusilla*) and common yellowthroats (*Geothlypis trichas*) and had greater nesting bird diversity than did farms without field borders. However, songbird nest success was low because of heavy depredation, which was not reduced by removing mesomammal predators such as raccoons (*Procyon lotor*), opossums (*Didelphis virginianum*), and foxes (*Vulpes vulpes*). Northern bobwhite abundance during summer was greater on farms containing field borders. Consistently more bobwhite coveys were heard on farms with field borders than heard on farms without field borders. However, the authors reported no differences in the number of coveys heard between predator reduction and non-reduction farms. Farms with both field border and predator reduction had more coveys heard compared with other farm blocks, but predator reduction would usually not be economically feasible.

Henningson and Best (2005) studied grassland bird use of riparian filter strips in Iowa and found 46 bird species using filter strips, with 41 species in sites dominated by cool season grasses and 31 species in sites dominated by warm season grasses. Mean species richness did not differ among sites. Seven bird species were significantly more abundant in filter strips lacking nearby woody vegetation compared with those adjacent to a wooded edge, and mean spe-

cies richness was significantly greater in non-wooded sites. There were no significant differences in relative nest abundance between cool and warm season grass-dominated sites. Nine avian species nested in cool season grass sites; seven species nested in warm-season grass sites. Twenty-seven percent of all nests were successful, while 62 percent were depredated.

Hedgerows

Hedgerows consist of rows of shrubs or small trees planted along the side of a field. There is an extensive literature base documenting the value of hedgerows for insects in Europe where some hedgerows may be centuries old. In the United States, Best (1983) reported on bird use of woody fencerows and Best et al. (1990) reported on the importance of edge habitats for birds in Iowa. Best (1983) reported as many as 30 species of birds using fencerows in Iowa farmlands during the breeding season. Fencerows with greater coverage of trees and shrubs supported a more diverse and abundant avifauna. A monotypic row of a single shrub species was not found to support the diverse bird communities that could occur from multiple woody species providing diverse structure. Hedgerows and other linear covers are generally perceived to be beneficial to most wildlife species inhabiting agriculturally dominated landscapes (Cable 1991). However, when established in landscapes dominated by grasslands, they may serve to fragment grassland habitats with negative consequences for grassland wildlife (O'Leary and Nyberg 2000).

Contour Strip Cropping

No literature citations were found documenting the wildlife effects of this practice, but inferences can be drawn from other work. Contour strip cropping is a technique used to control erosion by interspersing strips about 90 to 120 feet (27 to 36 m) wide of close-grown crops (e.g., hay and small grains such as oats) on the contour between strips of row crops. Alternating strips of corn, oats, and hay can provide the juxtaposition and configuration of cover types necessary to provide for the needs of wildlife during periods of limited mobility, such as when pheasants are tending young broods (Warner et al. 1984,

Warner 1988). As previously noted, ring-necked pheasant brood survival to 5 to 6 weeks of age had significantly declined from 78 percent to 54 percent in Illinois during a 30-year period concomitant to a threefold increase in the foraging area observed for pheasant broods (Warner 1979, 1984, Warner et al. 1984). This decline was the result of fewer acres of forage crops, small grains, and idle areas that chicks use to forage for insects. Contour strip cropping can make a substantial contribution to minimizing this problem by increasing the diversity of vegetation covers in a relatively small area.

System Effects

In those parts of the country where agricultural land uses are part of a matrix consisting of forest, range, and other land uses, wildlife abundance is usually not a problem unless it becomes one of crop depredation. However, wildlife habitat can be a daunting challenge where intensive land uses prevail. The fundamental principle guiding preservation and enhancement of wildlife habitats in such situations is to conserve as much of the biological integrity of the landscape as possible in the form of natural, or nearly natural, plant communities—"to keep every cog and wheel is the first precaution of intelligent tinkering" (Leopold 1966). Relatively natural habitats in agriculturally dominated landscapes often occur as riparian corridors, wetlands, woodlots, "odd" areas that aren't farmed for some reason, and brushy or weedy fencerows and roadsides. The greater the extent of those residual patches of biotic integrity, the greater the probability wildlife species will respond to the habitat elements provided, often secondarily, from the soil and water conservation practices described above. Any one of those practices alone may not have a great effect, but when implemented as part of a holistic resource management system, the cumulative effect can be substantial. The combination of grass-ridged terraces, grassed waterways, conservation tillage, and field border strips will provide habitat, food resources, and travel lanes, greatly enriching the biological characteristics of the landscape. Many other combinations of conservation practices can also be combined to enhance biological resources to fit various other landscape settings.

Wildlife response to land management activities is scale-dependent and the geographic scale of concern is dependent upon the wildlife species of interest. Grizzly bears demand huge landscapes, while meadow voles require very little. Most of the individual cropland soil and water conservation practices described here fall below the habitat thresholds for many species. Wildlife may utilize those habitat elements for part of their life cycle, but not all of it. Consequently, it does not make sense to try to elucidate direct cause and effect relationships at too fine a scale, as other habitat elements on the landscape confound the interpretation. Rather, the research needed should be at the resource management system level, where wildlife response to large scale agricultural land management systems is conducted while land use is controlled. Individual wildlife benefits from any traditional conservation practice may not be immediately obvious. However, when used in combination and in relation to landscapes that provide covers other than those annually disturbed, the conservation practices described above can only serve to elevate the quality of the landscape for terrestrial species. The water quality benefits described for many of these conservation practices undoubtedly reach far beyond the borders of fields containing the conservation activities. ■

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Appendix A

Definitions and purposes of cropland conservation practices (Conservation Practice Physical Effects, USDA NRCS).

Residue Management, No Till/Strip Till:

Managing the amount, orientation, and distribution of crop and other plant residues on the soil surface year-round, while growing crops in narrow slots, or tilled or residue-free strips in soil previously untilled by full-width inversion implements.

This practice may be applied as part of a conservation management system to support one or more of the following: reduce sheet and rill erosion, reduce wind erosion, maintain or improve soil organic matter content, conserve soil moisture, manage snow to increase plant-available moisture or reduce plant damage from freezing or desiccation, and to provide food and escape cover for wildlife.

Residue Management, Mulch Till: Managing the amount, orientation, and distribution of crop and other plant residue on the soil surface year-round, while growing crops where the entire field surface is tilled prior to planting.

This practice may be applied as part of a conservation system to support one or more of the following: reduce sheet and rill erosion, reduce wind erosion, maintain or improve soil organic matter content and tillage, conserve soil moisture, manage snow to increase plant-available moisture, and provide food and escape cover for wildlife.

Residue Management, Ridge Till: Managing the amount, orientation, and distribution of crop and other plant residues on the soil surface year-round, while growing crops on pre-formed ridges alternated with furrows protected by crop residue.

This practice may be applied to support one or more of the following purposes: reduce sheet and rill erosion, reduce wind erosion, maintain or improve soil organic matter content, manage snow to increase plant-available moisture, modify cool wet site conditions, and provide food and escape cover for wildlife.

Residue Management, Seasonal: Managing the amount, orientation, and distribution of crop

and other plant residues on the soil surface during a specified period of the year, while planting annual crops on a clean-tilled seedbed, or when growing biennial or perennial seed crops.

This practice may be applied to support one or more of the following purposes: reduce sheet and rill erosion, reduce soil erosion from wind, reduce off-site transport of sediment, nutrients or pesticides, manage snow to increase plant-available moisture, and provide food and escape cover for wildlife.

Contour Buffer Strips: Narrow strips of permanent, herbaceous vegetative cover established across the slope and alternated down the slope with parallel, wider cropped strips.

This practice may be applied to support one or more of the following purposes: reduce sheet and rill erosion; reduce transport of sediment and other water-borne contaminants down slope, on-site or off-site; or enhance wildlife habitat.

Contour Farming: Tillage, planting, and other farming operations performed on or near the contour of the field slope.

This practice may be applied to support one or more of the following purposes: reduce sheet and rill erosion or reduce transport of sediment and other water-borne contaminants.

Herbaceous Wind Barriers: Herbaceous vegetation established in rows or narrow strips in the field across the prevailing wind direction.

This practice may be applied to support one or more of the following purposes: reduce soil erosion and/or particulate generation from wind, protect growing crops from damage by wind-borne soil particles, manage snow to increase plant-available moisture, and provide food and cover for wildlife.

Strip Cropping: Growing row crops, forages, small grains, or fallow in a systematic arrangement of equal-width strips across a field.

This practice may be applied to support one or more of the following purposes: reduce soil erosion from water and transport of sediment and other water-borne contaminants, reduce soil erosion from wind, and protect growing crops from damage by wind-borne soil particles.

Filter Strip: A strip or area of herbaceous vegetation situated between cropland, grazing land, or disturbed land (including forestland) and environmentally sensitive areas.

This practice may be applied to support one or more of the following purposes: reduce sediment, particulate organics, and sediment-absorbed contaminant loadings in runoff, reduce dissolved contaminant loadings in runoff, serve as Zone 3 of a Riparian Forest Buffer, Practice Standard 391, reduce sediment, particulate organics, and sediment-absorbed contaminant loadings in surface irrigation tailwater, restore, create or enhance herbaceous habitat for wildlife and beneficial insects, and maintain or enhance watershed functions and values.

Grade Stabilization Structure: A structure used to control the grade and head cutting in natural or artificial channels.

Grassed Waterway: A natural or constructed channel that is shaped or graded to required dimensions and established with suitable vegetation.

This practice may be applied as part of a conservation management system to support one or more of the following purposes: to convey runoff from terraces, diversions, or other water concentrations without causing erosion or flooding; to reduce gully erosion; and to protect/improve water quality.

Sediment Basin: A basin constructed to collect and store debris or sediment.

This practice may be applied to support one or more of the following purposes: preserve the capacity of reservoirs, wetlands, ditches, canals, diversion, waterways, and streams; prevent undesirable deposition on bottom lands and developed areas; trap sediment originating from construction sites or other disturbed areas; and reduce or abate pollution by providing basins for deposition and storage of silt, sand, gravel, stone, agricultural waste solids, and other detritus.

Terrace: An earth embankment, or a combination ridge and channel, constructed across the field slope.

This practice may be applied as part of a resource management system to reduce soil erosion and retain runoff for moisture conservation.

Water and Sediment Control Basin: An earth embankment or a combination ridge and channel generally constructed across the slope and minor watercourses to form a sediment trap and water detention basin.

This practice may be applied to support one or more of the following purposes: improve farmability of sloping land, reduce watercourse and gully erosion, trap sediment, reduce and manage onsite and downstream runoff, and improve downstream water quality.

Hedgerow Planting: Establishment of dense vegetation in a linear design.

This practice may be applied to provide one or more of the following functions: food, cover, and corridors for terrestrial wildlife; food and cover for aquatic organisms that live in watercourses with bank-full width less than 5 feet; to intercept airborne particulate matter; to reduce chemical drift and odor movement; to increase carbon storage in biomass and soils, living fences, boundary delineation, contour guidelines, screens and barriers to noise and dust; and improvement of landscape appearance.

Field Border: A strip of permanent vegetation established at the edge or around the perimeter of a field.

This practice may be applied to support one or more of the following purposes: reduce erosion from wind and water, soil and water quality protection, management of harmful insect populations, provide wildlife food and cover, increase carbon storage in biomass and soils, and improve air quality.



Grassland Establishment for Wildlife Conservation

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ABSTRACT Establishing grasslands has important implications for wildlife, especially in areas historically rich in grasslands that have since been converted to row crop agriculture. Most grasslands established under farm conservation programs have replaced annual crops with perennial cover that provides year-round resources for wildlife. This change in land use has had a huge influence on grassland bird populations; little is known about its impacts on other terrestrial wildlife species. Wildlife response to grassland establishment is a multi-scale phenomenon dependent upon vegetation structure and composition within the planting, practice-level factors such as size and shape of the field, and its landscape context, as well as temporal factors such as season and succession. Grassland succession makes management a critical issue. Decisions on how frequently to manage a field depend on many factors, including the location (especially latitude) of the site, the phenology at the site in the particular year, the breeding-bird community associated with the site, and weather and soil conditions. The benefits for a particular species of any management scenario will depend, in part, on the management of surrounding sites, and may benefit additional species but exclude others. Thus, the benefits of grassland establishment and management are location- and species-specific.

Prior to European settlement, prairies and other grasslands covered an estimated 300 million ha (740 million acres) of the United States (Risser 1996) and were the largest vegetation type in North America (Samson and Knopf 1994). Major grassland ecosystems can be classified into six distinct types based on geography and vegetation structure: the tallgrass, mixed-grass, and shortgrass prairies of the central plains, the desert grasslands of the Southwest, the California grasslands, and the Palouse prairie of the Northwest (Risser 1996). Additionally, subtropical grasslands occurred in Florida and the eastern gulf plain of Texas, and smaller grasslands occurred in the eastern United States and intermountain west (Rich et al. 2004).

Grasslands have been termed the nation's most threatened ecosystem (Noss et al. 1995, Samson and Knopf 1994). Although they were unable to attain data for several states, Sampson and Knopf (1994) reported reductions in the U.S. central plains of 82.6 percent to 99.9 percent for tallgrass prairies, 30 percent to 77.1 percent for mixed-grass prairies, and 20 percent to 85.8 percent for shortgrass prairies. Reductions for grassland types in other portions of the country are similar to those of tallgrass prairie, including California grasslands (99 percent) and the Palouse prairie (99.9 percent) (reviewed by Noss et al. 1995). Losses of native grasslands have been (and continue to be) primarily due to conversion to agricultural or suburban land uses, though woody invasion after fire suppression (Rich et al. 2004) and the planting of trees and other non-native plants in the post-dust bowl era also contributed (Samson and Knopf 1994). In addition to quantitative losses, grasslands have been impacted qualitatively by alterations of natural disturbance regimes (fire, grazing pressure, and hydrology) and changes in species composition caused by invasive and non-native species (Rich et al. 2004, Noss et al. 1995, Samson and Knopf 1994).

Concomitant with losses and degradation of grasslands have been declines of wildlife populations. Disappearance of the massive bison (*Bison bison*) herds from the Great Plains is well known, but many other grassland species are endangered, threatened or candidates for listing (e.g. black-footed ferret (*Mustela nigripes*), prairie dog (*Cynomys sp.*), and mountain plover (*Charadrius montanus*)). There are many more species for which we lack good informa-

tion. Our best national data on wildlife populations exists for birds. Most grassland-nesting birds have been experiencing significant population declines for the 37 years of Breeding Bird Survey monitoring (Sauer et al. 2004), despite the fact that most grassland losses occurred before the survey began (Noss et al. 1995). Research has documented breeding in the Great Plains by 330 of the 435 bird species that breed in the United States (Samson and Knopf 1994), including almost 40 percent of the species on Partners In Flight's continental Watch List (Rich et al. 2004). Additionally, U.S. grasslands are important wintering habitat for birds of the Northern Forest Avifaunal Biome, which stretches from the northeastern United States northwest across Canada, as well as grassland breeding birds (Rich et al. 2004).

The Conservation Reserve Program (CRP) has played an important role in stemming the losses of U.S. grasslands. Beginning as part of the Food Security Act of 1985 (a.k.a. the 1985 Farm Bill), the CRP retired highly erodible cropland for a period of 10 years. Producers received rental and incentive payments to plant perennial vegetation. Most (>75 percent) of the 14 million ha (34.8 million acres) enrolled in CRP has been planted to grass or a mixture of grasses and forbs or legumes (Table 1). New grass plantings in the continental United States have been established in areas that were historically grassland (Figures 1-4). Although many conservation practices (CP) may incorporate grass (e.g., permanent wildlife habitat, CP4), seven exclusively establish grass or grass-based herbaceous mixtures: new introduced grasses and legumes (CP1), new native grasses (CP2), grass waterways (CP8), existing grasses and legumes (CP10), filter strips (CP13 and CP21), contour grass strips (CP15), and cross wind trap strips (CP24).

This manuscript discusses the impact of grass field establishment and management on wildlife species. We focus on CRP, specifically CP1 and CP2, because this program is the primary vehicle for establishment of grass fields and has been the focus of most of the research into the wildlife impacts of farm conservation practices. Our discussions are valid for CP10 as these acres are primarily re-enrollments of CP1 and CP2 fields. Most research has been conducted on avian communities in the Great Plains, Midwest, and Southeast. Thus, our discussion of benefits to wildlife necessarily concentrates on birds; we discuss other

information where available. Discussion of the benefits of other grass-based establishment practices can be found in the chapter on linear strips and conservation buffers. Although the management and spatial context issues discussed here are equally pertinent to conservation of rangelands, please see the rangeland chapter for a detailed treatment.

Desired Fish and Wildlife Benefits

Wildlife conservation was a secondary consideration of the 1985 Farm Bill but was elevated to co-equal status with erosion and water quality concerns with the 1996 re-authorization. Still, it was widely assumed that the establishment of CRP plantings would positively affect grassland wildlife populations (e.g., Berner 1988), by providing perennial food and cover resources. In their review of the literature, Ryan et al. (1998) listed 92 species of birds observed using CRP grass plantings in the central United States during spring and summer (i.e., the breeding season), including at least 42 species nesting in CRP. Recent research has added only one species to that list; Evard (2000) noted three rough-legged hawks (*Buteo lagopus*) hunting CRP fields in Wisconsin. Best et al. (1998) recorded 40 species using CRP fields in the Midwest during winter, five of which do not use the fields during the breeding season. Mammals, reptiles, and invertebrates also have been shown to use CRP grass plantings (reviewed by Farrand and Ryan 2005). The benefits provided by planting grass fields can be measured, in part, by the response of wildlife species to the grass relative to the crop land they replaced. Such benefits are related, in part, to the vegetation composition and structure of the plantings and how these factors change naturally over time (i.e., succession).

Retiring Cropland

Replacing annual crops with perennial grasses has the potential to provide stable cover and food resources for wildlife. Indeed, avian studies have shown higher abundances or densities of birds in CRP grass fields than in the crop lands they replaced. King and Savidge (1995) reported avian abundance to be four times greater in CRP fields than crop fields in

Nebraska. Analogously, in southeastern Wyoming, Wachob (1997) found higher densities of grassland birds in CRP fields (as well as in native rangeland) than in croplands. In the Midwest, Best et al. (1997) detected from 1.4 to 10.5 times more birds in CRP grass fields than rowcrop fields during the breeding season. Interestingly, the total number of bird species observed in CRP plantings by Best et al. (1997, 1998) did not differ markedly from the number of species they observed in nearby rowcrop fields. However, 16 species of birds were unique or substantially more abundant in CRP fields than in nearby rowcrop fields. Three of the four bird species they frequently observed in CRP (dickcissel [*Spiza americana*], grasshopper sparrows [*Ammodramus savannarum*], and bobolinks [*Dolichonyx oryzivorus*]) have been undergoing significant population declines. Additionally, Henslow's sparrow (*Ammodramus henslowii*) and sedge wren (*Cistothorus platensis*), species of high conservation concern in the Midwest (Herkert et al. 1996), occurred only in CRP fields. The Henslow's sparrow also is listed as a continental Watch List species (Rich et al. 2004). Of the five species unique or substantially more abundant in rowcrops than in CRP fields (Best et al. 1997), only one, the lark sparrow (*Chondestes grammacus*), is of moderate conservation concern in the Midwest (Herkert et al. 1996). Summer observations of ring-necked pheasants (*Phasianus colchicus*) in western Kansas, analyzed by Rodgers (1999), showed they used CRP fields more than their availability in northwestern Kansas but not in southwestern Kansas, where shorter grass plantings may not provide better habitat than cropland. Pheasant indices in Wisconsin CRP fields were 10-fold higher than in surrounding private farmland (Evard 2000). Johnson and Igl (1995) projected declines in the populations of 15 grassland bird species breeding in North Dakota CRP if those grass fields were reverted back to cropland.

Greater benefits are accrued to those species that breed successfully in planted grass fields than to those that simply use the fields for food or cover (Ryan 2000), because the breeding season is the part of the annual cycle that most strongly influences the population size of birds. Assessing the reproductive rate is much more challenging than determining population size; grassland birds are notoriously secretive in their breeding habits. Such behavior is

Table 1. Summary of grass area and total area in the Conservation Reserve Program by state and the proportion of area in Conservation Practices that establish whole-field grass-based plantings. Numbers presented here reflect conditions as of March 2005.

State ^a	Grass (ha)	Total (ha)	%Grass	%CP1 ^b	%CP2 ^b	%CP10 ^b
Alabama	50,949	196,783	25.9	4.0	2.9	92.2
Alaska	11,858	12,066	98.3	19.6	0.0	80.4
Arkansas	15,707	81,813	19.2	7.8	8.0	70.3
California	54,322	58,940	92.2	3.9	1.2	94.9
Colorado	818,246	926,006	88.4	2.4	29.7	67.8
Connecticut	103	129	80.2	27.5	13.3	51.4
Delaware	610	3,134	19.5	3.5	1.5	2.0
Florida	1,019	35,213	2.9	11.8	6.0	82.0
Georgia	3,911	123,457	3.2	5.9	4.0	75.9
Idaho	259,855	319,949	81.2	14.0	3.1	82.7
Illinois	262,128	413,485	63.4	27.7	6.1	38.8
Indiana	91,508	116,681	78.4	16.9	12.6	38.7
Iowa	537,793	773,352	69.5	22.3	11.0	44.2
Kansas	1,046,509	1,161,142	90.1	0.7	31.0	66.8
Kentucky	122,732	136,421	90.0	29.1	12.6	46.1
Louisiana	8,629	98,505	8.8	0.7	11.4	84.8
Maine	8,588	9,436	91.0	6.1	0.5	92.7
Maryland	24,348	34,178	71.2	19.6	5.9	6.9
Massachusetts	47	49	95.9	0.0	0.0	45.7
Michigan	79,886	105,749	75.5	17.3	9.5	51.4
Minnesota	338,672	713,815	47.4	29.3	16.1	35.4
Mississippi	58,624	380,740	15.4	4.0	0.3	90.1
Missouri	574,829	627,322	91.6	25.9	12.9	58.0
Montana	1,234,173	1,376,732	89.6	23.1	27.2	49.7
Nebraska	408,382	483,350	84.5	4.6	35.5	57.6
New Hampshire	70	80	87.8	5.8	0.0	0.0
New Jersey	859	926	92.8	53.5	17.2	21.8
New Mexico	238,503	241,337	98.8	0.2	30.9	68.8
New York	18,589	24,613	75.5	12.8	1.7	84.0
North Carolina	11,735	50,064	23.4	7.8	5.7	62.2
North Dakota	753,405	1,351,363	55.8	21.9	3.5	74.1
Ohio	83,891	112,834	74.3	12.3	13.9	46.5
Oklahoma	407,143	417,669	97.5	1.9	39.0	58.9
Oregon	187,974	204,956	91.7	23.8	11.4	64.2
Pennsylvania	68,800	76,587	89.8	48.3	16.3	33.9
Puerto Rico	186	448	41.5	23.5	0.0	76.5
South Carolina	7,421	85,600	8.7	3.7	0.6	60.9
South Dakota	367,173	593,500	61.9	18.3	25.2	55.6
Tennessee	89,485	110,653	80.9	14.3	18.7	62.6
Texas	1,565,462	1,602,024	97.7	2.8	42.2	54.8
Utah	81,314	81,732	99.5	28.7	7.4	63.8
Vermont	105	626	16.8	0.0	0.0	44.6
Virginia	9,919	25,338	39.1	17.1	11.1	54.8
Washington	478,310	563,134	84.9	10.6	49.2	33.1
West Virginia	299	1,062	28.1	1.4	3.0	89.0
Wisconsin	188,804	251,179	75.2	10.2	12.0	71.8
Wyoming	100,690	113,755	88.5	22.9	3.0	74.1
Undesignated	13	91	14.7	0.0	100.0	0.0
Total (ha)	10,673,588	14,098,018	75.7	13.1	24.8	57.6
Total (ac)	26,363,762	34,822,105				

^aStates and territories with CRP enrollments. Arizona, Hawaii, Nevada, and Rhode Island did not have enrollments.

^bConservation Practices that establish whole-field grass-based plantings are: CP1 – new introduced grasses and legumes; CP2 – new native grasses; and CP10 – existing grasses and legumes.

Figure 1. Land in active CRP contracts in the U.S. and Puerto Rico as of 30 April 2005 for new introduced grasses and legumes (CP1). Disclosure indicates data unavailable due to privacy restrictions required by the Farm Security and Rural Investment Act of 2002.

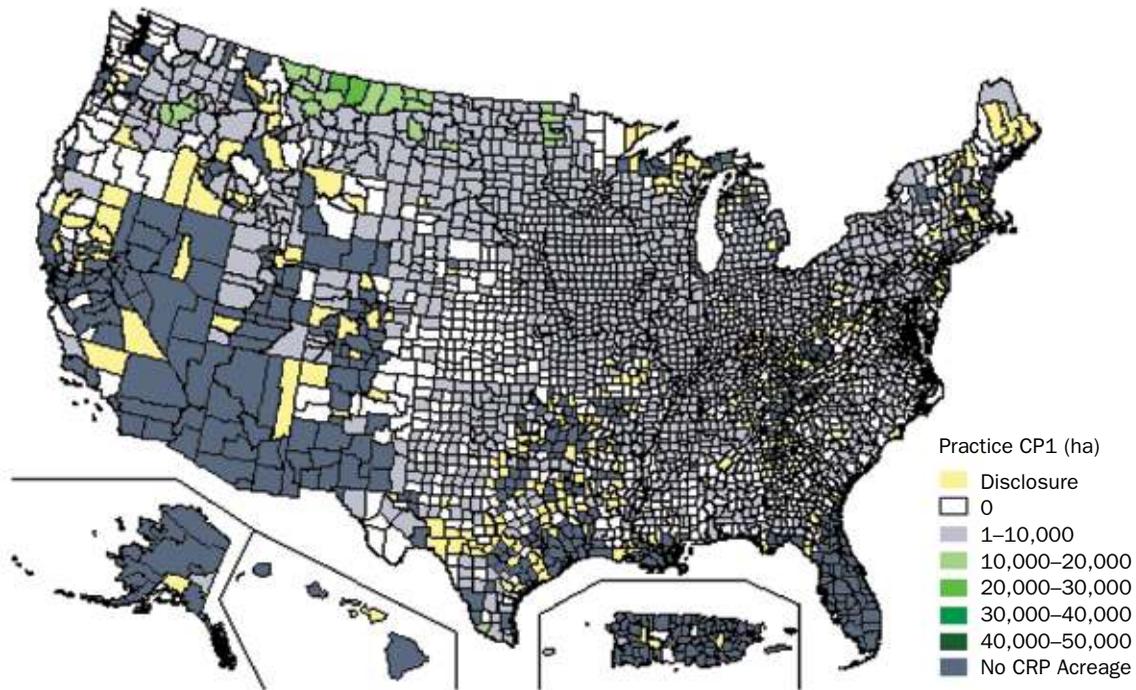


Figure 2. Land in active CRP contracts in the U.S. and Puerto Rico as of 30 April 2005 for new native grasses (CP2). Disclosure indicates data unavailable due to privacy restrictions required by the Farm Security and Rural Investment Act of 2002.

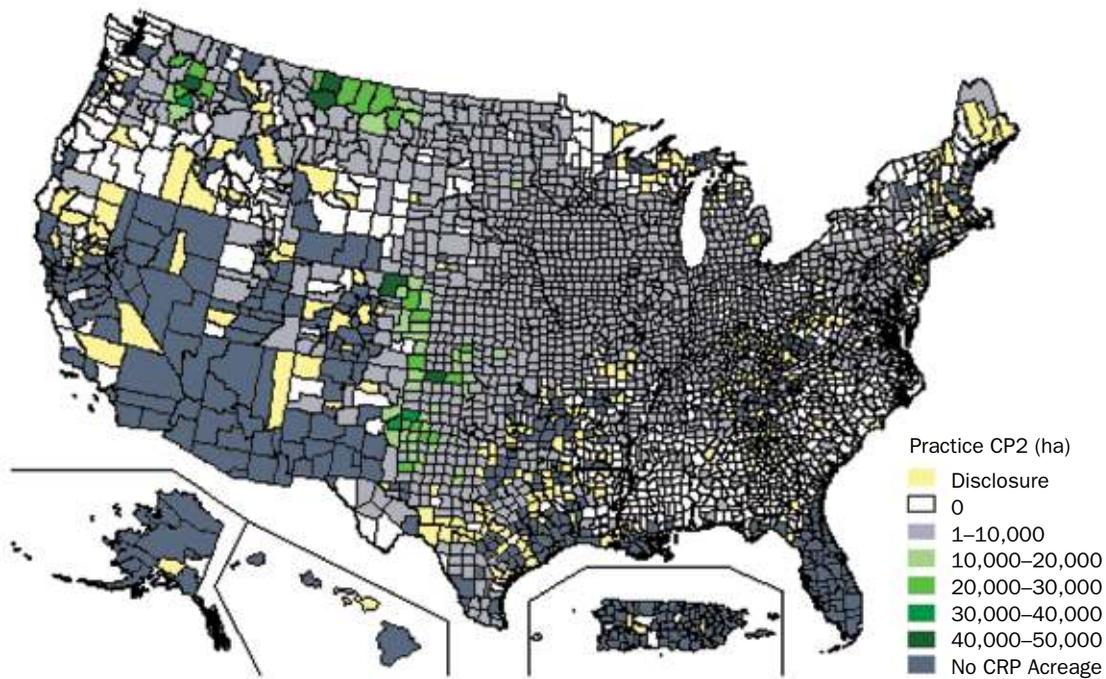


Figure 3. Proportion of active CRP contracts in new introduced grasses and legumes (CP1) for the U.S. and Puerto Rico as of 30 April 2005. Disclosure indicates data unavailable due to privacy restrictions required by the Farm Security and Rural Investment Act of 2002.

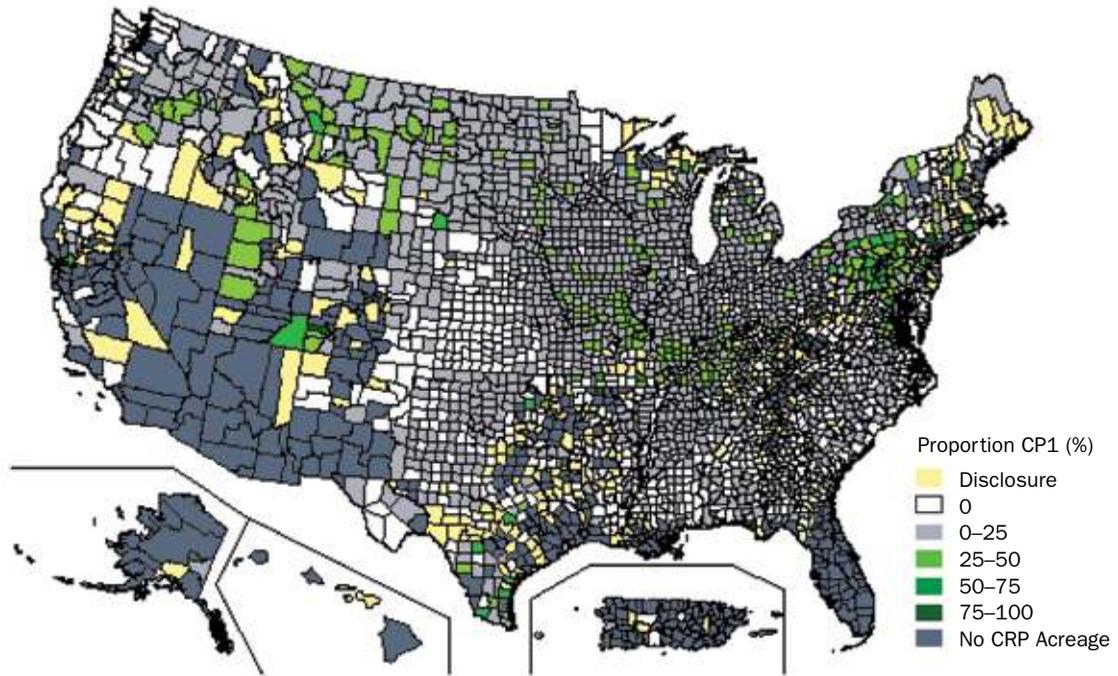
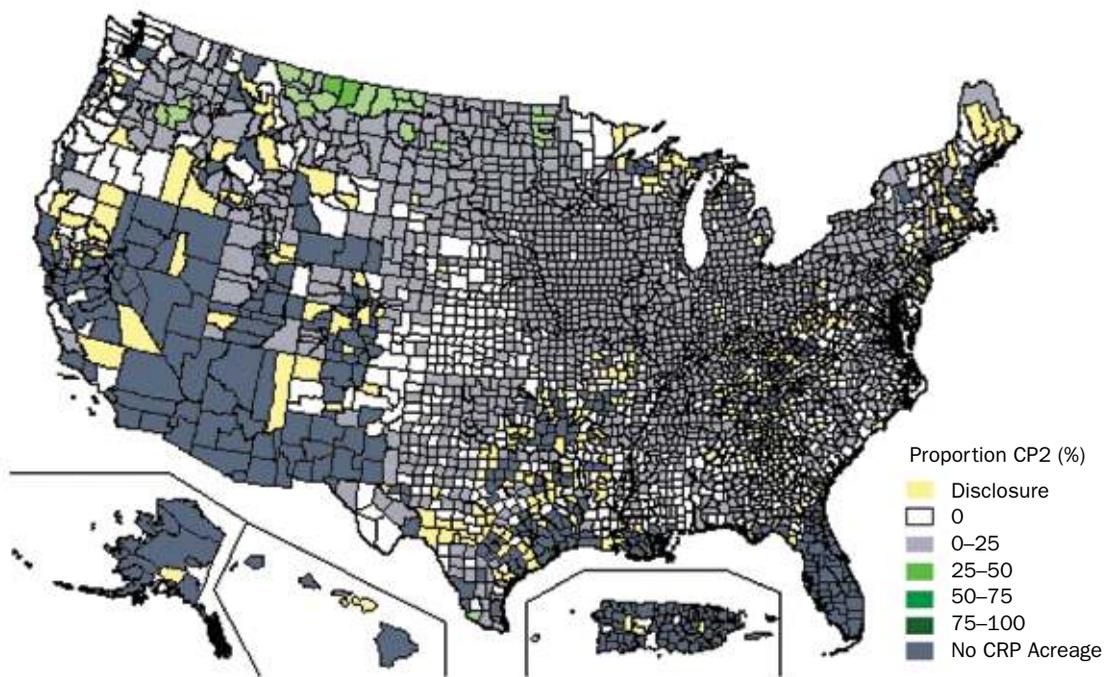


Figure 4. Proportion of active CRP contracts in new native grasses (CP2) for the U.S. and Puerto Rico as of 30 April 2005. Disclosure indicates data unavailable due to privacy restrictions required by the Farm Security and Rural Investment Act of 2002.



necessary to avoid drawing the attention of a wide range of species that depredate nests in grasslands. Avian reproductive success has not been well studied in CRP fields in the Great Plains, but the studies that have been conducted indicate that birds, including several grassland species of conservation concern, are at least as successful in CRP fields as in other land cover types. In northwest Texas, Berthelsen et al. (1990) found approximately six pheasant nests per 10 acres of CRP grassland, but no nests in cornfields. Berthelsen and Smith (1995) found a number of nongame bird nests incidental to their upland game-bird study in Texas. Most common species recorded were red-winged blackbirds (*Agelaius phoeniceus*), grasshopper sparrows, Cassin's sparrows (*Aimophila cassinii*), and western meadowlarks (*Sturnella neglecta*). Nest success values were higher than those typically reported in other studies in the agricultural Midwest. Koford (1999) found nests of red-winged blackbirds, grasshopper sparrows, and savannah sparrows to be most common in CRP fields in his North Dakota study sites, while in Minnesota sites the most numerous species were red-winged blackbirds, bobolinks, grasshopper sparrows, and savannah sparrows (*Passerculus sandwichensis*). He found fledging success of ground-nesting birds in CRP fields was lower than on Waterfowl Production Area plantings, but not significantly so.

In the Midwest, CRP plantings have been extensively used for nesting by grassland birds. Murray and Best (2003) found 20 species nesting in switchgrass (*Panicum virgatum*) CRP fields in 1999 and 2000 in Iowa; red-winged blackbirds comprised 56 percent of the sample. Best et al. (1997) located 1,638 nests of 33 bird species in CRP fields versus only 114 nests of 10 species in a similar area of rowcrops. In row-crop, they most frequently discovered red-winged blackbird, vesper sparrow (*Pooecetes gramineus*), and horned lark (*Eremophila alpestris*) nests. Nests of red-winged blackbirds, dickcissels, and grasshopper sparrows were the most frequently located in CRP fields by Best et al. (1997). Similar lists of species nesting in CRP have been produced by recent studies (Davison and Bollinger 2000, McCoy et al. 2001a). House sparrow (*Passer domesticus*) was the most common avian species nesting in CRP fields in north-east Kansas (Hughes et al. 2000). CRP also appears to be important nesting habitat for mourning doves

(*Zenaida macroura*) in Kansas (Hughes et al. 2000). In Wisconsin, ring-necked pheasant, gray partridge (*Perdix perdix*), northern harrier (*Circus cyaneus*), short-eared owl (*Asio flammeus*), and duck nests have been reported (Evard 2000). In Missouri, 55 percent of northern bobwhite (*Colinus virginianus*) nests and 46 percent of brood-foraging locations occurred in CRP fields that comprised only 15 percent of the largely agricultural landscape (Burger et al. 1994).

Grass fields also provide important resources for birds in winter. Although Morris (2000) reported higher species richness in crop fields in southern Wisconsin, she reported lower abundances in crop fields than CP2 fields. Avian abundance in crop fields was higher during periods of incomplete snow cover than during periods with 100 percent snow cover, while the reverse was true for CP2 sites. Morris (2000) did not observe if grassland birds were using CP1. However, total bird use in winter did not differ between introduced grasses with legumes (CP1) and switchgrass monocultures (CP2) in Missouri (McCoy et al. 2001a). During the winter months, ring-necked pheasants, northern bobwhites, American tree sparrows (*Spizella arborea*), dark-eyed juncoes (*Junco hyemalis*), and American goldfinches (*Carduelis tristis*) were the most abundant or widely distributed species observed in CRP fields (Best et al. 1998). All but the goldfinch have been undergoing long-term population declines (Sauer et al. 1996). King and Savidge (1995) reported use in Nebraska by American tree sparrows, ring-necked pheasants, red-winged blackbirds, western meadowlarks, horned larks, and northern bobwhites. Delisle and Savidge (1997) noted only American tree sparrows, ring-necked pheasants, and meadowlarks (*Sturnella sp.*) (eastern and western meadowlarks were not distinguishable) wintering on their Nebraska study areas. Burger et al. (1994) provided evidence that CRP plantings in Missouri provided important winter cover for northern bobwhites. They documented that 69 percent of nighttime roosts occurred in CRP fields in an area where CRP made up only 15 percent of the landscape. Rodgers (1999) used counts of droppings to compare winter pheasant use of weedy wheat stubble and CRP in north central Kansas. Despite offering comparable concealment, dropping density was 2.75 times greater in wheat stubble than CRP. Dropping data suggested that pheasants were using CRP for night-time roost-

ing. CRP may be less valuable to pheasants in winter due to fewer food sources, excessive litter, and the less rigid stems of the planted grass.

Information comparing mammalian use of planted grass fields with crop fields is scarce, and information on reproductive activity is virtually non-existent. Olsen and Brewer (2003) reported that a three-year, winter wheat (*Triticum aestivum*) rotation in southeastern Wyoming had higher rodent abundance and diversity than CRP at both sites in both years studied. A study of white-tailed deer (*Odocoileus virginianus*) habitat use in South Dakota revealed that CRP fields were used proportionately greater than habitat availability during periods of deer activity during spring, and during evening and midnight periods during



Dickcissel. (Photo by S. Maslowski, USFWS)



Red-winged Blackbird. (Photo by D. Dewhurst, USFWS)

summer (Gould and Jenkins 1993). Increased use of CRP between spring and summer corresponded with rapid vegetation growth and fawning. Similarly, white-tailed deer in southeastern Montana used CRP in greater proportion than its availability in all seasons except fall (Selting and Irby 1997). Indirect evidence of mammalian use of CRP comes from the nest predation literature. Hughes et al. (2000) listed potential nest predators at their sites in Kansas, including coyotes (*Canis latrans*), raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), opossums (*Didelphis virginianum*), feral cats (*Felis domesticus*), and badgers (*Taxidea taxus*). Evard (2000) attributed duck nest predation to mammalian predators, including red fox (*Vulpes vulpes*), striped skunk, and raccoon, though hard evidence was lacking.

Other mammalian species incidentally noted in CRP included white-tailed jackrabbits (*Lepus townsendii*), white-tailed deer fawns, and a coyote den with three pups (Evard 2000).

As with mammals, information on benefits accrued to other groups of wildlife is rare. Burger et al. (1993) reported mean invertebrate abundance and biomass in CRP fields were four times higher than in soybean fields. Phillips et al. (1991) detected a low in-

cidence of cotton pests and found beneficial predator species in Texas CRP. Davison and Bollinger (2000) identified four species of snakes common on their study sites in east-central Illinois, including prairie kingsnake (*Lampropeltis calligaster*), common garter snake (*Thamnophis sirtalis*), black rat snake (*Elaphe obsoleta obsoleta*), and blue racer (*Coluber constrictor*). Hughes et al. (2000) listed bullsnakes (*Pituophis melanoleucus*) as a potential nest predator in Kansas CRP.

Planting Perennial Vegetation

Wildlife response to changes in land use is species-specific, depending on life-history requirements. Thus, issues regarding the composition of the planting (e.g., introduced or native species, monoculture of grass or a mixture of grasses and forbs/legumes, seeding rate, etc.) and its resultant structure (e.g., height, plant density) will play an important role in determining what species can benefit from the practice.

The primary farm conservation practices that establish new grass fields are CP1 (introduced grasses and legumes) and CP2 (native grasses). As the names suggest, the primary difference between the two is the origin of grass and legume seed. Either practice can be planted as a grass monoculture or as a mixture of grasses with or without forbs and/or legumes; eligible plant lists are developed by individual states. Each planting must conform to NRCS Practice Standard 327 – Conservation Cover (NRCS 2002). The standard sets forth base criterion for each establishment including: minimum seeding rates; guidelines for the seeding rate, seedbed preparation, and companion crops; and management considerations. The standard also includes “Additional Criteria for Enhancement of Wildlife Habitat,” which gives guidelines related to plant selection, native forb establishment, an adjustment factor (0.75) to reduce seeding rates if erosion control guidelines can still be met, and maintenance recommendations. The combination of the practice standard with the individual land owner’s conservation plan yields flexibility to meet the land owner’s needs and variability in the practice’s wildlife habitat value.

Few studies have directly compared avian response to CP1 and CP2 plantings. McCoy et al. (2001a) found that species richness, abundance and nesting success of grassland birds during the breed-

ing season did not differ between CP1 (introduced grasses and legumes) and CP2 (switchgrass monocultures) in Missouri. However, species-specific Mayfield nest success often differed between CP1 and CP2 within years, and the better type switched between years in several cases. However, means differed only for red-winged blackbird. Parasitism rates did not differ between the practices for any species, but varied with host species (mean=18%, range 0-40%). Fecundity of dickcissel, a continental Watch List species (Rich et al. 2004), and nesting success and fecundity of red-winged blackbirds were higher on CP2 than on CP1 habitat, but both practices were likely sinks ($\lambda < 1$) for these species. For grasshopper sparrows, a species of national concern (Rich et al. 2004), nest success was 49 percent in CP2 compared with 42 percent in CP1. Both practices were likely source ($\lambda > 1$) habitat for grasshopper sparrows, whereas only CP1 fields were likely a source for eastern meadowlarks (*Sturnella magna*) and American goldfinches (McCoy et al. 2001a).

Morris (2000) compared winter use by grassland birds of CRP, crop fields, pastures, and restored and native prairies in southern Wisconsin. In this study, species diversity was highest in crop fields, followed by restored prairie, CP2 fields (a mixture of native warm-season grasses and two forbs), native prairie remnants, and pastures, while avian abundance was highest in pastures, followed by restored prairie, CP2, crop fields, and native prairie. No species were observed using CP1 fields (a mixture of introduced grasses and legumes) in this study. In contrast, McCoy et al. (2001a) found that total bird use in the winter did not differ between CP1 and CP2 in Missouri.

Although we know of no studies directly examining mammalian response to CP1 versus CP2, two studies have compared CP1 fields to native prairies. Hall and Willig (1994) found that CP1 fields simulated shortgrass prairies of northwest Texas in small mammal diversity but not in species composition, suggesting that CRP was not mimicking natural conditions. Of the 11 species captured in the study, only the southern plains woodrat (*Neotoma micropus*) was not captured on CRP. Also in northwest Texas, Kamler et al. (2003) reported that both adult and juvenile swift fox (*Vulpes velox*) strongly avoided CP1 fields. Whereas CRP comprised 13 percent of the available habitat for adults and 15 percent of the available habitat for juve-

niles, only 1 of 1,204 locations was recorded in a CRP field. The authors believed this was due to the taller, denser vegetation of CP1 (introduced warm-season grass plantings) compared with the native short grass prairie preferred by swift foxes.

Several studies have focused on invertebrate response to CP1 and CP2 plantings. Burger et al. (1993) reported that CP1 fields planted to timothy (*Phleum pretense*) and red clover (*Trifolium pretense*) had significantly higher invertebrate abundance and biomass than CP1 or CP2 grass monocultures or CP1 fields planted orchard grass (*Dactylis glomerata*) and Korean lespedeza (*Kummerowia stipulacea*). Carroll et al. (1993) determined CRP grasses (native and exotic) to be marginal over-wintering habitat for boll weevils (*Coleoptera: curculionidae*) in Texas. Also in Texas, McIntyre and Thompson (2003) reported that CP1 and CP2 fields had less vegetative diversity and lower arthropod diversity than native shortgrass prairie, but did support avian prey groups. The CRP types were similar in terms of invertebrate abundances (i.e., no support that different types of grasses possess different prey availabilities for grassland birds). In a concurrent study, McIntyre (2003) surveyed CP1, CP2 and native shortgrass prairie in the Texas panhandle for endangered Texas horned lizards (*Phrynosoma cornutum*) and their food supply, harvester ants (*Pogonomyrmex*). Ant nest densities varied within the classes but not between, suggesting that planting type (exotic vs. native) did not affect habitat value. Lizards also were seen on both types of CRP, but only at sites with ant nests.

Several studies investigated the effect of forb abundance on wildlife response. Hull et al. (1996) examined the relationship between avian abundance and forb abundance in native-grass CRP fields in northeast Kansas. The expected significant relationship was not found, but no field had > 24 percent forbs, which the authors surmised was too low to produce a response. Their data also did not support the hypothesis that invertebrate biomass was correlated positively with forb abundance. However, Burger et al. (1993) concluded that planting legumes may improve CRP plantings for northern bobwhite brood-rearing habitat due to increased invertebrate biomass. Swanson et al. (1999) reported that savannah sparrows used fields with less forb canopy cover.

Vegetation Succession

Although the initial planting mixture and density is important, changes in structure will occur over time. McCoy et al. (2001b) studied vegetation changes on 154 CRP grasslands in northern Missouri and reported that during the first two years following establishment, fields are characterized by annual weed communities with abundant bare ground and little litter accumulation. Within three to four years, CRP fields became dominated by perennial grasses with substantial litter accumulation and little bare ground. They suggested that vegetation conditions three to four years after establishment might limit the value of enrolled lands for many wildlife species and some form of disturbance, such as prescribed fire or disking, might be required to maintain the wildlife habitat value of CRP grasslands.

Few studies have examined avian response to field age. In an analysis of Breeding Bird Survey data combined with CRP contract data, Riffell and Burger (2006) showed the abundances of northern bobwhite and common yellowthroat were positively correlated with the density of CRP fields <4 years old. Eggebo et al. (2003) observed more crowing pheasants in old, cool-season, CRP fields than any other age or cover type in South Dakota. Delisle and Savidge (1997) noted that grasshopper sparrow densities declined in the CRP fields in Nebraska each year of their study from 1991 to 1994. They attributed that change to a build-up of litter and dead vegetation. Swanson et al. (1999) evaluated avian use of two- to seven-year-old CRP (CP1, CP2 and CP10) fields in Ohio and reported that neither species richness nor total abundance was related to field age. However, these coarse summary metrics may mask shifts in community composition (Nuttall et al. 2003).

As with birds, little information exists on mammalian response to aging fields. Furrow (1994) captured eight small mammal species on CRP fields planted to exotic grasses (CP1) in Michigan. Deer and white-footed mice (*Peromyscus spp.*) dominated younger fields and meadow voles (*Microtus pennsylvanicus*) dominated older (>2 years) fields. *Peromyscus* numbers were positively correlated with bare ground and forb canopy cover, and voles were positively correlated with litter depth. Fields <2-years-old had a greater diversity of small mammalian species than

older fields, while relative abundance increased with age. Millenbah (1993) reported greater insect abundance on one- to two-year-old fields, which may have contributed to greater small mammal diversity on these age classes. Conversely, Hall and Willig (1994) detected no significant differences in mammalian diversity due to age of CP1 plantings. However, their sites were only one to three years post-planting compared with Furrow's one- to six-year-old sites. Furrow (1994) also surveyed mid-sized mammals using scent stations and noted a decreasing trend in detections with increasing age of the CRP field. The decreasing trend was attributed to decreases in ease of movement and prey diversity.

Principles for Application

Wildlife habitat selection and use is a multi-scale phenomenon (e.g., Gehring and Swihart 2004, Best et al. 2001, Johnson 1980). In addition to the within-field factors (vegetation composition, structure, and succession) described above, response to implementation of a particular planting is dependent upon practice-level factors (e.g., size, shape), the landscape context in which those plantings are placed (e.g., to what extent are alternative grasslands available), and how the fields are managed over time.

Field Size, Shape and Landscape Context

The size of a grassland patch and its surrounding landscape can markedly influence the use of that site by grassland birds. Some patches may be too small to be colonized by certain species, or birds using smaller patches may suffer more from competition or predation than do birds in larger patches. Also, smaller patches have a relatively greater proportion of their area near an edge, so edge effects can be more pronounced in smaller patches. Edge effects are phenomena such as avoidance, predation, competition, or brood parasitism that operate at different levels near a habitat edge than in the interior of a habitat patch (e.g., Faaborg et al. 1993, Winter and Faaborg 1999). Brown-headed cowbirds (*Molothrus ater*) are brood parasites; they lay their eggs in nests of other birds and leave them for the host birds to raise, usually to the detriment of the host's own young. Cow-

birds use elevated perch sites to find nests to parasitize; such perches are more frequent along edges of grasslands because of the presence of trees, fence posts, and the like. Isolation from other grassland patches is a landscape feature that can affect either the use by birds or the fate of their nests in a patch.

Each of these factors—patch size, amount of edge, and isolation—can affect 1) the occurrence or density of birds using a habitat patch; 2) reproductive success, through either predation rates or brood parasitism rates; or 3) competition with other species (Johnson and Winter 1999, Johnson 2001). These features have been shown to operate among several species of grassland birds (e.g., Herkert et al. 2003; Winter et al. In press; reviewed by Johnson 2001). In CRP habitat specifically, Johnson and Igl (2001) related the occurrence of species and their densities to patch size in CRP fields. They conducted 699 fixed-radius point counts of 15 bird species in 303 CRP fields in nine counties in four states in the northern Great Plains. Northern harriers, sedge wrens, clay-colored sparrows (*Spizella pallida*), grasshopper sparrows, Baird's sparrows (*Ammodramus bairdii*), Le Conte's sparrows (*Ammodramus caudacutus*), and bobolinks were shown to favor larger grassland patches in one or more counties. In contrast, two edge species, mourning doves and brown-headed cowbirds, tended to favor smaller grassland patches. Horn (2000) sampled 46 CRP fields in North Dakota during 1996 and 1997. He reported bobolinks, grasshopper sparrows, and red-winged blackbirds were more common in large grassland patches than in smaller ones. In contrast, brown-headed cowbirds preferred smaller fields. Field size also was an important factor influencing the occurrence and/or abundance of grassland songbirds in switchgrass plantings in Iowa (Horn et al. 2002). In southeastern Wyoming, Wachob (1997) noted that sharp-tailed grouse favored larger CRP patches for nesting but not for brood-rearing. Conversely, Rodgers (1999) postulated that pheasants in western Kansas had not benefited from CRP as much as expected due to the large size of the plantings.

Use of CRP (CP1, CP2 and CP10) fields by several grassland-dependent species in Ohio was related to field size (eastern meadowlarks and bobolinks) or field size plus adjacent grasslands (grasshopper sparrows) (Swanson et al. 1999). All species recorded in this study were more abundant in CRP fields contiguous with other grassland.

McCoy (2000) compared measures of grassland bird use and habitat quality between CRP fields located in landscapes with high (20-35 percent) or low (5-12 percent) amounts of CRP and high (55-75 percent) or low (20-35 percent) amounts of grassland. Dickcissels and sedge wrens were more likely to be present in CRP fields in landscapes with higher levels than lower levels of CRP. Total species richness was highest in high CRP, high grassland landscapes, and total bird abundance was higher in high grassland than low grassland landscapes, but there were no similar effects for grassland birds as a group. Nesting success was higher for wild turkey (*Meleagris gallopavo*) in high grassland than low grassland landscapes, and was higher for red-winged blackbirds in high CRP than low CRP landscapes.

Best et al. (2001) investigated the effect of landscape context, including proportion in CRP, on avian use of rowcrop fields in Iowa. Some species showed a strong response to landscape composition (including dickcissel and indigo bunting [*Passerina cyanea*]), while others did not (e.g., American robin [*Turdus migratorius*], American goldfinch, and killdeer [*Charadrius vociferus*]). Seven species differed significantly between landscapes; for these the lowest numbers in crop fields occurred in areas of intensive agriculture. Species with different habitat affinities (grass or wood) showed similar aversion to rowcrops. Grassland birds occurred more often in landscapes with more grass (block or strip). Generalists, crop specialists, and aerial foragers were not affected by landscape composition.

Merrill et al. (1999) compared landscapes (1.6-km radius) surrounding greater prairie-chicken leks with random non-lek points and found greater amounts of CRP in the landscape for leks. Toepfer (1988) documented nesting in Minnesota CRP, but success was lower in CRP than in native grasslands (J. Toepfer, unpublished data, in Merrill et al. 1999). The shape of grassland and woodland patches was significant but had low predictive power for comparisons between temporary and traditional leks. Merrill et al. (1999) believed CRP might be important, especially near temporary lek sites. Svendsky et al. (2000) recommended that 30 percent of the grassland surrounding greater prairie-chicken leks be managed to provide spring nesting cover and be in close proximity to brood cover to maintain populations. Wachob (1997) noted that sharp-tailed grouse leks were

more common closer to CRP fields and in areas with extensive CRP within 0.6 mile (1 km).

Recent studies have examined the landscape scale effects of CRP across large regions. Riffell and Burger (2006) examined the abundances of 15 bird species associated with grasslands in the eastern United States and found positive correlations between bird abundance and amount of CRP in the landscape. Bird responses varied by species and by ecological region, but tended to be stronger in regions where grasslands were relatively scarce. Similarly, Veech (2006a) investigated the relationship between northern bobwhite population trends and land use across its range. He found that landscapes with increasing populations had significantly more useable land (e.g., cropland and grassland). In a separate analysis, Veech (2006b) examined the population trends of 36 grassland nesting birds in the Midwest and Great Plains relative to land use. Restored grasslands (e.g., CRP) were typically rare, but were more common in landscapes with increasing than decreasing populations.

In contrast to these studies, Hughes et al. (2000) found that mourning dove Daily Survival Rate (DSR) was influenced by vegetation structure within the field, but not field edge or landscape (800 m) factors. Landscape effects were thought to be lacking due to the generalist nature of doves. For ring-necked pheasants in northwestern Kansas, the amount of CRP in areas where home ranges were located had no detectable effect on size of home ranges (Applegate et al. 2002). Females tended to have smaller home ranges (average of 127 ha) in high-density (25 percent) CRP sites than low-density (8 to 11 percent) CRP sites (average 155 ha), but males showed the reverse trend. Horn et al. (2002) also found no effect of landscape on the relations between avian occurrence, abundance, and field size. They noted that the literature is contradictory concerning landscape effects on area sensitivity and postulated that the amount of woodland cover, ranges in field sizes among landscapes, and amounts of shrub and forb cover within CRP fields may have confounded any relationship with landscape composition.

Management Practices

As previously mentioned, plant communities on CRP grasslands are not static, but rather change in species composition and structure over the 10-year

lifespan of the contract. Successional changes can be mitigated through management practices such as mowing, disking, burning, or herbicide applications. Until the 2002 reauthorization, grazing and haying were not permitted practices under the CRP, except during weather-related emergencies (e.g., drought). All management practices affect wildlife populations indirectly through changes in vegetation structure, but also directly as a potential cause of mortality.

Mowing or clipping is the most common management practice implemented on CRP grasslands. McCoy et al. (2001b) reported that mowing had short-term effects on vegetation structure (reduced height within the year and increased litter accumulation) and resulted in accelerated grass succession and litter accumulation. Dykes (2005) characterized vegetation structure on 45 CP2 fields in Tennessee and reported that litter cover and depth were greater on fields that had been mowed than those that had been burned. Litter cover and depth were intermediate on unmanaged fields. Conversely, forb coverage was greatest on burned fields, followed by unmanaged and mowed fields (Dykes 2005).

Effects of mowing and haying on wildlife have been fairly well studied. These effects can be divided according to temporal category: immediate, short-term, and long-term. Immediate effects usually include the destruction of nests that are active in the field at the time, fatalities of nesting adults or dependent young, and abandonment of nests or breeding territories that had been established in the field (Rodenhouse and Best 1983, Warner and Etter 1989, Bollinger et al. 1990, Frawley and Best 1991, Dale et al. 1997, McMaster et al. 2005). For example, Labisky (1957) observed that 78 percent of mallard (*Anas platyrhynchos*) and blue-winged teal (*Anas discors*) nests in alfalfa fields were destroyed by haying. In their study of bobolinks (*Dolichonyx oryzivorus*), Bollinger et al. (1990) found that mowing accounted for 51 percent direct mortality in active nests. Subsequent causes of mortality in eggs and of nestlings included abandonment after mowing (24 percent), raking and baling (10 percent), and predation (9 percent); only 6 percent of the clutches fledged successfully. In addition, removal of the vegetation by haying exposes surviving birds, especially young ones, to greater predation pressure (e.g., George 1952, Bollinger et al. 1990).

To mitigate these immediate effects, USDA prohibits regular management activities in CRP grasslands during a set “Nesting Season”; emergency management is also affected. The start date, end date, and length of this restricted period vary from state to state (even by county within some states) based on consultations between USDA and USFWS. A table containing these dates, as well as permissible periods for management under the new Managed Haying and Grazing provision of the 2002 Farm Bill, can be found on the Internet (www.fsa.usda.gov/dafp/cepd/crp/nesting.htm). Restricting management activities to outside the peak nesting period likely has a positive impact on nesting success of grassland birds. However, the benefit of this restriction to populations has not been evaluated and may be limited by annual fluctuations in the timing of peak nesting with annual weather patterns, inability to protect late-season nesting/re-nesting attempts, and a general lack of attention among researchers and managers to the habitat needs of post-fledgling birds.

We consider short-term effects to be those that manifest within about a year after the management action. Johnson et al. (1998) assessed densities of breeding birds in hayed versus idled grassland that had been restored under the Conservation Reserve Program the year after haying occurred. Because the authors used the same fields in all years, they had essentially a before-and-after, treatment-and-control design. They had data from nearly 300 fields that had been hayed and more than 2,600 fields that had been left idle in the previous year; study fields were in eastern Montana, North Dakota, South Dakota, and western Minnesota. Three species typically had heightened densities the year following haying; these were horned lark, chestnut-collared longspur, and lark bunting, all of which favor short and sparse vegetation. The densities of many more species, in contrast, were reduced the year following haying, including vesper sparrow, sedge wren, common yellowthroat, bobolink, clay-colored sparrow, dickcissel, red-winged blackbird, and Le Conte’s sparrow. Some species had responses that varied by study site (and associated climatic regime). Savannah, grasshopper, and Baird’s sparrows tended to respond negatively to mowing in the more arid western study sites but positively in study sites with greater precipitation.

Horn and Koford (2000) reported fewer sedge wrens and, possibly, clay-colored sparrows, Le Conte’s sparrows, and red-winged blackbirds in mowed than in uncut portions of 12 CRP fields (in North Dakota) in the year after mowing. Savannah sparrows and possibly grasshopper sparrows showed the opposite tendency, being more common in mowed CRP.

McCoy et al. (2001a) examined the influence of mowing on birds wintering in CRP fields in Missouri. They noted that mowing of cool-season CRP plantings in late summer and early fall permitted sufficient regrowth to provide habitat for wintering birds. In contrast, the value of mowed warm-season planting was reduced for at least two years.

As might be expected, birds that prefer heavy cover for nesting typically prefer uncut vegetation. For example, Oetting and Cassel (1971) reported that significantly more ducks nested in unmowed stretches of roadside right-of-way than in adjacent mowed stretches. Also, Renner et al. (1995) found that the density of nests of five species of ducks was lower in portions of CRP fields that had been hayed the previous year than in the uncut portions. Overall, densities were twice as high in the uncut vegetation. The earliest nesting species, mallard and northern pintail, especially avoided the hayed portions until sufficient regrowth had occurred. Analogously, Luttschwager et al. (1994) observed a shift in the species composition from mostly mallards in uncut CRP field to primarily blue-winged teal in hayed CRP fields.

It is worth mentioning here that grazing may increasingly be used as a management technique under the new Managed Haying and Grazing provision of the 2002 Farm Bill. Because grazing of CRP historically has been restricted to emergency situations (e.g., drought conditions), little direct information is available. Whereas there has been much research on grazing and birds in rangeland systems, the results are often contradictory (see Ryan et al. 2002 and references therein). In general, grazing, like mowing and haying, can negatively impact wildlife directly or indirectly. Direct effects may include trampling and exposure due to reduced vegetation structure. Indirect effects may include increased exposure (thermal) and predation due to vegetation removal and composition shifts. However, grazing does not impact all birds negatively. Reduced structure may prompt some birds to avoid grazed pastures, but at-

tract other species. Grazing impacts are complex and depend upon the species under consideration, grazing regime (i.e., grazing intensity, timing, frequency, and the livestock species), and other biotic and abiotic factors (Ryan et al. 2002). As noted above, USDA attempts to mitigate direct effects of grazing through timing restrictions, but the benefit of such restrictions is difficult to gauge.

Although our focus has been on breeding birds, there is some relevant information on other taxa, specifically some mammals. For example, Westemeier and Buhnerkempe (1983) noted that nests of small mammals (*Microtus ochrogaster* and *Synaptomys cooperi*) and cottontail rabbits (*Sylvilagus floridanus*) were most abundant in prairie grasses left undisturbed, indicating that they would respond negatively to haying. Leman and Clausen (1984) also commented that meadow voles (*Microtus pennsylvanicus*) and prairie voles (*M. ochrogaster*) were significantly less common on plots with lower residual vegetation; those plots were the ones mowed most recently. In contrast, deer mice (*Peromyscus maniculatus*) were more common on the most recently mowed plot.

By long-term effects, we refer to those occurring more than a year afterward. In addition to the above finding by McCoy et al. (2001) about effects persisting at least two years, Johnson et al. (1998) discovered delayed responses to haying of CRP fields. Some species, such as lark bunting, Le Conte's sparrow, and clay-colored sparrow, showed a response in the second year after haying that was similar to, albeit weaker than, the response in the first year. Although the response by horned larks to haying was positive rather uniformly in the first year, responses in the second year varied geographically, being negative in the drier, western study sites but positive in the more mesic eastern sites. Sedge wrens, reduced the first year after haying, tended to increase the second year. Several species, including common yellowthroat, red-winged blackbird, and bobolink, showed no consistent pattern two years after haying, despite broadly negative responses the first year after haying.

Our knowledge on the effects of other management practices is limited. Madison et al. (1995) examined the effects of fall, spring, and summer disking and burning, and spring herbicide (Round-

up®) treatments on bobwhite brood habitat quality in fescue-dominated, idle grass fields in Kentucky. They reported that during the first growing season following treatment, fall disking significantly enhanced brood habitat quality by increasing insect abundance, plant species richness, forb coverage, and bare ground relative to control plots. However, the benefits of disking were relatively short-lived, with diminished response during the second growing season. During the second growing season following treatment, herbicide treatments provided the best brood habitat quality. Greenfield et al. (2002), examining the effects of disking, burning, and herbicide on bobwhite brood habitat in fescue-dominated CRP fields in Mississippi, likewise reported that disking and burning improved vegetation structure for bobwhite broods during the first growing season after treatment. However, the benefits were short-lived (one growing season). Herbicide treatment in combination with prescribed fire enhanced quality of bobwhite brood habitat for the longest duration (Greenfield et al. 2002).

Concerns or Opportunities

The CRP was amended in the 2002 reauthorization to require mid-contract cover management (i.e., incorporating native seeds, light disking, and burning) on all new covers under new contract (USDA 2003). Additionally, the original provision prohibiting commercial uses of CRP lands was amended to allow managed haying and grazing, as well as biomass harvests and the installation of wind turbines. Whereas managed haying and grazing was specifically restricted to one in three years, no federal guidelines were issued for biomass harvests and cover management practices.

Grasslands are disturbance-dependent ecosystems, so it is natural to consider the role of disturbance in established grasslands compared with natural prairies. Grasslands evolved with, and indeed were maintained by, fire and grazing. Fire was especially important in eastern prairies and the tallgrass prairie, where frequent—often annual—fires restricted the encroachment of woody vegetation. In western prairies especially, bison (*Bison bison*) and other native grazers maintained viable grasslands. Mowing, haying, and disking

are disturbances that are now common in agricultural settings but did not occur naturally. It is reasonable to contemplate if and how those activities should be used in establishing and maintaining grasslands. In our view, human disturbance of established grasslands that mimics the natural disturbance regimes will better provide for species that evolved with grasslands.

Mandated disturbance will address some shortcomings of CRP grasslands as wildlife habitat but also raise some concerns. Management practices such as burning and grazing may mimic natural disturbances, especially if used in combination. By removing vegetation, these practices are likely to benefit grassland bird species associated with shorter, sparser grasslands. If these practices occur in a patchy distribution within a field, across the landscape, and through time, a mosaic of grassland successional stages may form that can sustain a wider array of species. However, if a uniform management is applied to most fields in a landscape (i.e., the same practice applied to whole fields at the same time of year and in the same years), conservation goals for a wide range of species will not be accomplished.

CRP management can only be applied according to a detailed conservation plan (USDA 2003). We recommend such plans carefully consider the timing of management actions. From a purely agricultural perspective, grasses and associated forbs should be harvested at or near the peak of their nutritional quality. That strategy conflicts with providing habitat for nesting birds. The immediate effects of haying are extremely detrimental, of course, but they can be largely avoided by delaying haying until after the bulk of nesting activities has ceased. Establishing a reasonable date to begin haying depends on many factors, including the location (especially latitude) of the site, the phenology at the site in the particular year, the breeding-bird community associated with the site, and weather conditions. Similarly, these factors need to be considered when planning the timing and length of grazing. Other management practices, such as burning, disking, and harvesting biomass for energy (e.g., co-firing switchgrass with coal) can generally be done outside the nesting season and therefore pose less of a dilemma.

Another consideration is the frequency of management. Irregular management will result in a greater variety of grassland successional stages and provide

for a wider array of species. Decisions on how frequently to manage a field depend on many of the same factors as for the establishment of haying dates discussed above. For example, as a result of longer growing seasons and greater rainfall, the rate of natural succession on CRP grasslands throughout the Southeast likely exceeds that observed in the Midwest or Great Plains, making planned disturbance even more important for maintaining habitat quality for early successional species.

Although USDA (2003) contends that wind turbines “generally have a limited impact on wildlife,” their impact may be dependent on placement (e.g., near migratory routes) and species-specific susceptibilities. Avian mortality at wind farms appears to be low relative to the number of birds passing over them, or to communication towers and other tall structures (see Johnson et al. 2002 and references therein). However, turbines may add to the cumulative declines of some species. Wind farms appear to have very little effect on resident bats in Minnesota (Johnson et al. 2004) and Iowa (A. A. Jain, unpublished data). However, substantial numbers of migrating bats suffered collision deaths in both studies. More study is needed to fully understand the impacts of wind turbines on wildlife.

Links with Other Systems

Grasslands established under CRP, or any other program, are linked to varying degrees with other systems in the landscapes in which they are embedded. Perhaps the closest and most important linkage is with riparian and aquatic systems. As mentioned in the introduction, CRP was originally targeted at highly erodible soils to improve and protect water quality. CRP continues to provide those benefits through regular sign-ups and extensions of the program targeted at high value conservation (i.e., Conservation Reserve Enhancement Program). CRP grasslands tend to be established in landscapes already containing more grassland and woodland areas (Weber et al. 2002), likely because these areas tend to have higher slopes and are more difficult to farm than relatively flat areas. These areas also present higher risk to aquatic systems from agricultural runoff of sediment, nutrients, and chemicals. The

Farm Service Agency is currently funding projects to estimate the water quality benefits provided by CRP practices in various regions of the country (S. Hyberg, personal communication).

Conclusions

Establishing grasslands has important implications for wildlife, especially in areas historically rich in grasslands that have since been converted to row crop agriculture. Most grassland established under farm conservation programs has replaced annual crops with perennial cover that provides year-round resources for wildlife. Which wildlife species benefit from grassland establishment depends on many factors at multiple spatial and temporal scales. These factors include within-field factors (vegetation composition, structure, and succession), practice-level factors (e.g., size, shape), the landscape context in which those plantings are placed (e.g., to what extent additional grasslands are available), the season or life-cycle stage the species uses the grassland for, and how the fields are managed over the life of the contract.

Periodic management, especially practices that profit land owners, is a relatively new mandate for established grasslands. It can be argued that as disturbance-dependent systems, grasslands should be manipulated periodically. Such disturbances, however, should occur no more often than is necessary; the frequency depends on factors such as precipitation and species composition of the plants. It should be remembered that the response by breeding birds to such disturbances will depend on the location of the site relative to the breeding ranges of various bird species, the habitat preferences of species whose ranges encompass the site, the environmental conditions—especially soil moisture—prevailing, and the timing of the disturbance. For example, Baird's sparrows prefer grassland habitat with moderately deep litter, vegetation height between 20 and 100 cm, moderately high but patchy forb coverage, and patchy grass and litter cover with little woody vegetation (Dechant et al. 2003). Creating such habitat in Wisconsin, for example, which is well outside the breeding range of the species, is unlikely to provide any benefits to the species. Also, mowing grassland in September will have far different consequences than

mowing it in May. Vegetation will recover from mowing much more quickly when soil moisture is high than when it is not. Further, management scenarios that benefit one species will benefit some others but also exclude some. These considerations lead to the conclusion that a "one size fits all" approach to managing grasslands will not work. ■

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Agricultural Buffers and Wildlife Conservation: A Summary About Linear Practices

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ABSTRACT Conservation practices such as filter strips, grassed waterways, buffers, contour strips, riparian buffers, windbreaks and shelterbelts are eligible under a variety of USDA programs. Most were originally designed to provide benefits regarding reduced soil erosion and improved water quality. Most often grasses, or mixtures of grasses and forbs, are used in these practices, although establishment of trees and shrubs is encouraged in some practices. The small area and high edge-area ratios limit the usefulness of these practices for wildlife. Scientific evidence suggests that enrolling land in linear practices has accumulated in recent years, although most studies still focus heavily on benefits to birds and do not address the larger questions of the animal communities. With careful planning and management, applying linear practices widely within an agricultural landscape could be expected to have positive wildlife benefits compared with continued intensive row cropping.

In Phase I of the Conservation Effects Assessment Project, Clark and Reeder (2005) provided a review of the effects of the Continuous Conservation Reserve Program (CCRP) on the conservation of wildlife in agricultural landscapes. Whereas the first review took a programmatic viewpoint, this chapter summarizes the available research on individual conservation practices that would generally be called “linear or narrow” practices. Grass filter strips or riparian buffers are the most widely used of the practices that we will review. While some of these practices are available in a number of USDA conservation programs, the majority of these practices are available to producers through the CCRP.

As Clark and Reeder (2005) emphasized, the linear shape, small area, and high edge-area ratios have limited the potential direct benefits of linear practices for wildlife. Yet, the replacement of annual crops with perennial habitat, even in small patches, has some conservation benefit. Evidence that wildlife use these linear habitat patches in agricultural landscapes, whether part of a specific conservation program or not, is mounting, although the research is most heavily focused on avian populations and communities. The greatest wildlife benefits of conservation practices and programs accrue when relatively large areas are converted from annual cropland to perennial habitat. This point is easily

illustrated by the well-known benefits of enrollment of large areas into the Conservation Reserve Program (CRP) (Reynolds 2000, Ryan 2000), especially when the habitat is configured in blocks that were the rule under the general signup (Clark et al. 1999, Horn et al. *in press*). Clark and Reeder (2005) also emphasized that the landscape context (i.e., the habitat in the landscape surrounding the project) influences the benefits of linear practices. So a challenge for land managers and producers interested in wildlife benefits is to consider whether practices can be “strategically” located in the landscape to target

wildlife benefits. In fact, this landscape perspective is almost in direct conflict with the application of specific linear conservation practices on individual farm units. There is very little research in the wildlife literature that quantifies the tradeoffs between applications of piecemeal conservation practices versus landscape management of collections of practices. Careful planning and sustained management are keys to gaining the desired wildlife benefits from these plantings. Sustaining wildlife populations and community diversity depends on the functional relationships of species to habitat and

Table 1. Linear and Potentially Linear Conservation Practices on CRP Acres as of April 2005. Adapted from FSA (2005).

Practice		General Signup		Continuous Signup		Total	
		Acres	%	Acres	%	Acres	%
CP1	New Intro. Grasses And Legumes	3,268,929	10	182,813	6	3,451,743	10
CP2	New Native Grasses	6,450,216	20	82,281	3	6,532,497	19
CP3	New Softwood Trees (Not Longleaf)	427,519	1	694	0	428,213	1
CP3A	New Longleaf Pines	184,995	1	0	0	184,995	1
CP3A	New Hardwood Trees	526,105	2	9202	0	535,307	2
CP4	Permanent Wildlife Habitat	2,315,297	7	41,600	1	2,356,897	7
CP5	Field Windbreaks	831	0	74,581	3	75,412	0
CP8	Grass Waterways	1011	0	108,830	4	109,841	0
CP9	Shallow Water Areas For Wildlife	1943	0	48,536	2	50,479	0
CP10	Existing Grasses And Legumes	15,145,051	48	49,564	2	15,194,614	44
CP11	Existing Trees	1,093,037	3	357	0	1,093,394	3
CP12	Wildlife Food Plots	75,473	0	1743	0	77,216	0
CP13	Vegetative Filter Strips	29,458	0	0	0	29,458	0
CP15	Contour Grass Strips	36	0	78,403	3	78,439	0
CP16	Shelterbelts	364	0	29,466	1	29,830	0
CP17	Living Snow Fences	2	0	4252	0	4254	0
CP18	Salinity Reducing Vegetation	0	0	295,130	10	295,130	1
CP 19	Alley Cropping	52	0	0	0	52	0
CP21	Filter Strips (Grass)	0	0	972,156	33	972,156	3
CP22	Riparian Buffers	0	0	712,093	24	712,093	2
CP23	Wetland Restoration	1,569,334	5	91,859	3	1,661,193	5
CP23	Wetland Restoration (Floodplain)	0	0	68,047	2	68,047	0
CP23A	Wetland Restoration (Non-floodplain)	0	0	4832	0	4832	0
CP24	Cross Wind Trap Strips	0	0	687	0	687	0
CP25	Rare And Declining Habitat	656,128	2	38,292	1	694,420	2
CP26	Sediment Retention	0	0	6	0	6	0
CP29	Wildlife Habitat Buffers (Marg Past)	0	0	16,789	1	16,789	0
CP30	Wetland Buffer (Marg Past)	0	0	11,544	0	11,544	0
CP31	Bottomland Hardwood	0	0	10,030	0	10,030	0
CP33	Upland Bird Habitat Buffers	0	0	33,477	1	33,477	0
	Unknown	-21	0	1018	0	997	0
TOTAL		31,745,760	100	2,968,282	100	34,714,042	100

landscape features. Individual conservation practices may not provide the life requisites to sustain satisfactory reproductive success and survival, although data on the functional value of practices on taxa using these plantings is generally limited (Clark and Reeder 2005).

The linear practices that we review (Table 1) are nested in a larger framework of agricultural conservation practices. In Table 1 we present designations used by FSA and have provided an appendix for cross reference with designations used by NRCS. Many of the practices available to producers under a variety of USDA programs could be configured in a linear fashion, depending on the characteristics of the site (Table 2). In this chapter we focus more narrowly on practices that are linear by design, although the principles highlighted in the research reviewed here are applicable to most linear perennial habitat practices.

The standards outlined for the general practice “Conservation Cover” apply to any practice that retires land from agricultural production and establishes permanent vegetative cover (NRCS 2000a). In practice, standards that were established for practices like CP1 (new introduced grasses and legumes), CP2 (new native grasses) and other conservation cover practices recommend following general principles (e.g., avoid the use of invasive species) that should be applied to all practices in addition to the specific guidelines laid out in a specific practice standard. There should be flexibility in conservation practices such that the plantings are suited to the site and the goals of the landowners. For example, the practice standard for Cross Wind Trap Strip specifies that plant materials used for the practice should be selected based on their level of suitability to the site and compatibility with secondary goals such as provision of wildlife food and cover (NRCS 2005). Thus, the rules allow for such strips to be planted with warm or cool season grasses, with or without legumes or other forbs.

Landowners and managers always balance myriad goals and requirements in the planning and implementation of conservation practices. This chapter is designed to assist resource managers weighing the merits of linear conservation practices in relation to wildlife habitat benefits. This summary of practices focuses on the wildlife benefits, although there is

very surely a link between the terrestrial and aquatic communities. We have grouped practices into general categories based on physical structure (herbaceous and tree/shrub) and location (riparian and in-field), so the chapter is organized into four sections based on combinations of those categories. In the sections that follow, we list a number of specific practices that fit under the broader categories.

Herbaceous Practices

In the Midwest, where the intensity of row crop agriculture is the highest, herbaceous practices dominate (Table 2). This fact stems from several causes: a) the pre-agricultural native vegetation was primarily prairie, so natural resource agencies have encouraged the re-establishment of grasses and forbs rather than trees, b) landowners are sometimes averse to the idea of planting trees in an area that has been cleared of trees for agriculture, c) and planting trees is more work and capital-intensive than planting herbaceous vegetation, and trees are also more costly to remove once a program ends.

Research on herbaceous buffers has shown that these practices host greater abundances of wildlife than surrounding row crop fields. Studies of avian use of agricultural areas has demonstrated that, even though some bird species such as vesper sparrows (*Pooecetes gramineus*), dickcissels (*Spiza americana*), and red-winged blackbirds (*Agelaius phoeniceus*) are known to nest in row crop fields, abundances in herbaceous buffers are an order of magnitude greater than in row crops (Best 2000). Grassland specialist bird species use buffer strips in comparatively small numbers (Kammin 2003, Knoop 2004, Henningsen and Best 2005).

Landscape context is particularly important for some species using herbaceous buffers. These species can exhibit behavioral or demographic responses to the proximity of other landscape features, especially trees and shrubs and edges (Ries and Debinski 2001, Fletcher and Koford 2003, Henningsen 2003), as well as to the landscape composition (e.g., Clark et al. 1999, Horn et al. *in press*, Knoop 2004). The width, vegetative composition and structure, and landscape context of these practices all affect wildlife communities using them (Clark and Reeder 2005).

Table 2. Acres of linear conservation practices installed on CRP and CCRP acreage as of April 2005.
(Adapted from FSA 2005.)

States	Field Wind-breaks	Grass Water-ways	Contour Grass Strips	Shelter-belts	Living Snow Fence	Filter Strips	Riparian Forest Buffer	Cross Wind Trap Strips	Field Borders
Alabama	0	47	188	0	0	968	27,940	0	36
Alaska	0	1	0	0	0	0	185	0	0
Arkansas	0	23	0	0	0	5,362	39,785	0	17
California	0	0	0	0	0	0	5,248	0	0
Colorado	1,313	985	444	4,002	37	406	801	28	0
Connecticut	0	0	0	0	0	20	63	0	0
Delaware	0	4	0	0	0	1,403	158	0	0
Florida	0	0	0	0	0	5	68	0	0
Georgia	0	85	41	0	0	1,235	1,320	0	75
Idaho	512	13	64	220	73	1,212	6,928	0	0
Illinois	2,501	28,522	2,011	138	36	147,441	103,759	0	16,259
Indiana	2,172	15,216	208	25	0	56,740	4,841	0	2,383
Iowa	5,999	29,909	30,373	1,949	229	239,909	61,995	41	1,120
Kansas	1,567	7,759	5,482	595	70	26,802	4,765	184	5,294
Kentucky	8	3,532	61	0	0	33,414	13,936	0	806
Louisiana	0	41	0	0	0	636	4,339	0	0
Maine	0	26	0	0	0	126	197	0	0
Maryland	0	228	0	0	0	40,447	16,793	0	7
Massachusetts	0	1	0	0	0	62	5	0	0
Michigan	1,788	803	16	82	3	42,295	3,115	0	0
Minnesota	8,741	4,408	1,273	3,513	2,961	155,354	43,861	7	0
Mississippi	0	61	38	0	0	7,994	132,542	0	196
Missouri	114	1,832	2,232	36	0	42,338	25,307	0	1,674
Montana	409	97	0	260	18	142	2,441	27	0
Nebraska	26,256	1,825	583	2,144	145	20,916	3,136	46	955
New Hampshire	0	0	0	0	0	163	23	0	0
New Jersey	8	21	4	0	0	133	21	0	0
New Mexico	0	0	0	0	0	0	7,330	0	0
New York	11	72	4	0	0	589	10,197	0	0
North Carolina	22	149	0	13	0	6,918	28,220	0	500
North Dakota	4,237	128	0	3,881	306	8,595	575	10	0
Ohio	2,182	7,255	18	88	3	49,656	4,439	4	781
Oklahoma	44	316	2	37	4	1,033	1,483	0	99
Oregon	4	73	19	2	0	2,256	20,438	0	0
Pennsylvania	4	513	133	0	0	1,831	12,349	0	0
Puerto Rico	0	0	0	0	0	0	94	0	0
South Carolina	79	74	0	0	0	6,313	27,422	0	965
South Dakota	16,940	1,168	131	12,743	325	7,262	3,398	15	74
Tennessee	0	171	78	0	0	9,617	5,582	0	1,618
Texas	43	2,230	251	34	0	1,958	25,165	257	571
Utah	5	6	0	0	0	12	154	0	0
Vermont	5	1	0	0	0	147	1,327	0	0
Virginia	3	43	0	0	3	4,347	17,710	38	47
Washington	16	489	33,599	9	0	50,184	19,930	14	0
West Virginia	0	0	0	0	0	49	1,749	0	0
Wisconsin	242	1,700	1,186	26	39	25,312	16,264	0	0
Wyoming	187	13	1	33	4	9	4,654	17	0
Total	75,412	109,841	78,439	29,830	4,254	1,001,614	712,094	687	33,477

Riparian

CP21—Filter Strip

Filter strips are areas of herbaceous vegetation planted between row crop fields and bodies of water. Filter strips are designed to reduce the sediment and contaminant load in surface runoff, to provide habitat for wildlife and beneficial insects, and to enhance watershed functions.

The filter strip is one of the most studied practices with regard to wildlife benefits. The available research suggests that filter strips are valuable to wildlife because they create areas of perennial vegetation that are less disturbed relative to surrounding annual row crops fields. Despite benefits associated with perennial cover, generally, wildlife community composition is not as rich, nor reproductive success as sustaining in filter strips as they are in natural grassland habitats.

Filter strips host a variety of wildlife, including small mammals, arthropods, and birds. However, dominant species within these groups are primarily habitat generalists, like deer mice and red-winged blackbirds. Diverse plantings favor a richer fauna, especially of arthropods, as structural heterogeneity provides a variety of microhabitats. Wider plantings may also support a greater variety of species, because adding interior is more favorable for species that exhibit edge-averse behavior.

Research on the effect of filter strip width has been evaluated for birds and butterflies. In Illinois filter strips, Kammin (2003) found no relationship between strip width and abundance or richness of birds. In Iowa filter strips, the abundance of the eastern meadowlark (*Sturnella magna*) was associated with width (Henningsen 2003). Henningsen (2003) found nest success of only one species, the red-winged blackbird, was positively associated with width of the filter strip. The maximum width of the filter strips in these studies was 40 m (131 ft). Perhaps, for vagile organisms such as birds, the effects of width are not manifested in this range. A study of filter strips 18 to 167 m (59 to 548 ft) wide in Minnesota showed that the diversity of butterflies, as well as the abundance of certain large-bodied butterfly species, was positively associated with strip width (Reeder et al. 2005). The effects of width may be dependent upon the relative vagility of the species of interest and be limited by the range of widths evaluated.

Habitat structure also plays an important role in determining wildlife community structure in filter strips. Vegetative diversity is positively correlated with arthropod diversity and abundance (Benson 2003, McIntyre and Thompson 2003). Arthropods are a primary food source of birds, including pheasant chicks and grassland passerines.

The influence of landscape context on wildlife communities in filter strips has yet to be directly addressed in the literature. It is clear, however, that the configuration of herbaceous cover on the landscape affects the reproduction and distribution of pheasant populations (Clark et al. 1999).

In-field

CP8—Grassed Waterway

Grassed waterways are an in-field conservation practice, engineered to direct runoff within a field and prevent erosion and gully formation. They are typically quite narrow (up to 100 ft wide) and are often mowed to keep the grasses short to allow optimal water flow. The combination of being embedded in a



Grassed waterway in a Georgia agricultural field. (Photo courtesy of USDA NRCS)

row crop matrix rather than being along a field edge, being narrow, and being composed of a relatively homogenous grass mixture leads grassed waterways to offer less habitat potential for wildlife than filter strips or riparian forest buffers. In fact, providing wildlife habitat is not among the stated purposes of this practice (NRCS 2000b). However, grassed waterways host a range of wildlife, from small mammals and snakes to nesting birds, so wildlife considerations can be important in planning and implementing grassed waterways.

A heavy proportion of the species found in grassed waterways are generalists. For example, red-winged blackbirds accounted for 50 percent of the total bird abundance in Iowa grassed waterways, while grassland specialist species such as grasshopper sparrows (*Ammodramus savannarum*), savannah sparrows (*Passerculus sandwichensis*), and vesper sparrows (*Poocetes gramineus*) were found in fewer than five of 33 grassed waterways surveyed (Knot 2004).

Knot (2004) reported that presence of plains garter, eastern garter, and brown (*Storeria dekayi*) snakes was positively correlated with the width of grassed waterways in Iowa. However, in her analyses on the avian community, she found a predictive relationship of grassed waterway width for only 2 of 7 species of songbirds, and the direction of the relationship contrasted. These results suggest that perhaps, for a practice as narrow as a grassed waterway, it is difficult to detect an effect of width on vagile species such as birds. For such species, this practice may represent 100 percent edge habitat.

The effect of habitat structure on wildlife in grassed waterways varies by species. In grassed waterways in Iowa, vegetation vertical density was positively associated with the presence of dickcissels, common yellowthroats (*Geothlypis trichas*), and red-winged blackbirds (Knot 2004). Occurrences of smooth green snakes (*Lioclonorophis vernalis*) in these grassed waterways were positively associated with litter cover, but eastern garter snake (*Thamnophis sirtalis*) occurrences were negatively correlated with litter.

Because grassed waterways are embedded in row crop fields, they are driven over by farm machinery, in contrast with most other strip cover practices. Farm equipment caused 9 percent of nest failures in Iowa grassed waterways, but the nest failure rate caused by such disturbance is small in comparison with the 80 percent of failures caused by predation in this study (Knot 2004).

CP33—Habitat Buffer for Upland Birds (Field Border)

Field borders are areas of managed, herbaceous vegetation, which can be planted along crop field edges regardless of the erosion potential of the border. In general, such buffers can be used to reduce erosion from wind and water, protect soil and water

quality, manage harmful insect populations, and provide wildlife food and cover. CP33 was recently created as part of a national northern bobwhite conservation initiative. This practice is typically narrow and can be planted to warm season grasses, legumes and forbs.

During a study of bird response to experimentally established field borders in Mississippi, Smith (2004) found that abundances of dickcissel (*Spiza americana*) and indigo bunting (*Passerina cyanea*) were double that of areas not planted with field borders. Overall bird abundance and species richness was greater in bordered edges than non-bordered edges, although diversity did not differ between treatments. Additionally, during winter, edges of fields with borders hosted a higher abundance of sparrows than those without field border buffers (Smith et al. 2005a).

The results presented by Smith et al. (2005a) also indicate that field borders will only contribute meaningfully to bobwhite quail conservation if they make up a significant proportion (5 percent to 10 percent) of the landscape. This is consistent with the principal outlined by Clark and Reeder (2005) that a coordinated, landscape-level approach to locating practices in the landscape stands to offer the most benefit for wildlife.

Terraces

Terraces are earth embankments built up across the field slope and thus have a steep profile and are not very wide. Terraces are so narrow that their effect on conservation of grassland birds is minimal, and changes in terrace management practices are unlikely to improve their habitat quality (Hultquist and Best 2001).

CP15—Contour Grass Strip,

CP24—Cross Wind Trap Strip

To our knowledge, these practices have not been the specific focus of wildlife research. However, they are similar to grassed waterways in a couple of key ways — they are areas of grass that are narrow, and they are embedded in a row crop matrix. Contour grass strips occur on slopes, however, they are often planted in an alternating pattern with crops and are generally wider than some linear practices, thus enhancing their value to wildlife.

Tree/Shrub Practices

In the Midwest, where the predominant historic vegetation is grassland, buffers with shrubs and small trees often have greater species richness than herbaceous buffers due to the increased heterogeneity of vegetation structure. But such woody plantings also chiefly host generalist species. For instance, studies in Iowa and Illinois showed that buffers with restored or existing trees hosted generalist birds such as red winged blackbirds, song sparrows (*Melospiza melodia*), and cowbirds (*Molothrus ater*) (Kammin 2003, Schultz et al. 2004).

In the Southeast, and elsewhere in the country, where the native vegetation was dominated by forests, buffers are more frequently planted with trees and shrubs (Table 2). However, most of the knowledge about wildlife response, especially that of birds, has come from general studies of riparian forest buffers in a variety of forest types (Dickson 1989, Haas 1994, Hodges and Krementz 1996, Machtans et al. 1996, Pearson and Manuwal 2001), rather than from practice-specific research. Whereas the presence of trees in riparian buffers in grassland landscapes often has important negative effects, common forest wildlife species are often better adapted to the edge effects of riparian corridors imbedded in forested landscapes. Often the diversity of birds is greater along forest corridors because of the interspersion of deciduous and evergreen species (Darveau et al. 1995, Dickson 1989, Hodges and Krementz 1996, Kilgo et al. 1998). As is the case with herbaceous riparian zones, wider forest buffers host more diverse and productive populations of birds and other wildlife (Hagar 1999, Hodges and Krementz 1996, Kilgo et al. 1998, Pearson and Manuwal 2001, Spackman and Hughes 1995, Rudolph and Dickson 1990, Semlitsch and Brodie 2003). Forest riparian buffers are used as movement corridors by birds, reptiles and amphibians, and presumably by small mammals (Burbrink et al. 1998, Haas 1994, Machtans et al. 1996).

When riparian forest practices are applied in the open grass or along croplands in the Midwest, tree and shrub buffers create “hard edges” so that edge effects are often more pronounced than with herbaceous practices. Some species such as regal fritillaries (*Speyeria idalia*), bobolinks (*Dolichonyx oryzivorus*), dickcissels (*Spiza americana*), and red-winged

blackbirds (*Agelaius phoeniceus*) also demonstrate behavioral avoidance of wooded edges (Ries and Debinski 2001, Fletcher and Koford 2003, Henningson 2003). In a study in Iowa, bobolink density was lower near wooded edges than other types of edges (road or crop), and breeding birds avoided placing territories near woody edges (Fletcher and Koford 2003). In addition to causing behavioral effects, woody edges can be a detriment to reproductive success. Winter et al. (2000) studied the effect of forested, shrubby, road, and agricultural field edges on artificial nests, and on real nests of dickcissels and Henslow’s sparrows (*Ammodramus henslowii*). Artificial nest survival was depressed within 30 m (98 ft) of woodland edges, and real nests suffered greater predation within 50 m (164 ft) of shrubby edges than at greater distances.

In northern areas, plantings that include trees and shrubs have special value during winter, providing both cover from severe weather and predators. For instance, when snow is deep, herbaceous buffers often act as drift fences that catch snow, but the presence of shrubs and trees provides additional structure that provides wildlife habitat value (Gabbert et al. 1999).

Riparian

CP22—Riparian Forest Buffer

Riparian forest buffers are plantings consisting of three zones – an unmanaged woody zone adjacent to the water body, a managed zone of woody vegetation, and a zone of herbaceous vegetation (grasses and sometimes forbs) adjacent to the cultivated field. Riparian forest buffer benefits are particularly focused on water quality, although they have important consequences as wildlife habitat. They are designed to reduce scour erosion on stream banks and reduce sediment and contaminant loads in surface runoff. They are also intended to create more favorable habitat for aquatic species by providing shade, lowering water temperatures, and creating a source of coarse woody debris.

Plant species diversity and associated structural heterogeneity provide a variety of perching and nesting sites for birds and lead to a greater variety of microhabitats for invertebrates and small mammals. In general, diverse vegetation structure and composition benefit a greater variety of wildlife, so the additional vertical structure provided in a riparian forest buffer should provide habitat for a greater number of species than an herbaceous strip. However,

in landscapes that were traditionally dominated by prairies and wetlands, the native wildlife may not be well adapted to the artificial introduction of trees (Naugle et al. 1999). Trees and shrubs provide perching sites for avian predators and species like cowbirds that parasitize nests of grassland birds. Large trees provide den sites for mammalian predators. Woody edges are associated with greater predation rates on nests (Winter et al. 2000) and some species exhibit an aversion to nesting near a woody edge (Henningson 2003).

In landscapes where cover is limiting, wooded riparian corridors provide important habitat and travel corridors for large mammals (Hilty and Merenlender 2004), such as white-tailed deer (*Odocoileus virginianus*) and larger predators such as red fox (*Vulpes vulpes*).

Information on how the width of riparian forest buffer plantings affects wildlife is lacking, but can be inferred from research on similar systems. For example, Keller et al. (1993) found that probabilities of occurrence of birds in riparian forests were positively associated with width, with the greatest increases occurring between 25 and 100 m.

A thorough understanding of how landscape context influences wildlife in riparian forest buffers is lacking. Research on riparian forests and riparian forest buffers in Missouri shows that they provide habitat for area-sensitive forest and grassland-shrub nesting species (Peak et al. 2004). However, nest success was lower than that needed to balance mortality, and the authors indicated that in predominantly agricultural areas, even wide riparian forest buffers (400–530 m) have limited potential to serve as high-quality breeding habitat for some forest bird species. Landscape context is thus an important consideration and as yet not fully understood.

In-field

CP5—Field Windbreak, CP16—Shelterbelt, CP17—Living Snow Fence

A windbreak is a strip of trees or shrubs planted in a field to reduce wind-caused soil erosion, conserve moisture, and protect crops and/or livestock. A shelterbelt is a type of windbreak that is used around buildings to provide a barrier against chemical drift (from hog confinements, for instance) or to protect a farmstead from wind, preserving energy and protecting livestock and plants. Living snow fences



Living snow fence designed to control snow deposition. (Photo courtesy of USDA NRCS)

are windbreaks that are placed by roads in order to control snow deposition. Windbreaks came into wide use after the Dust Bowl of the 1930s as a way of reducing the soil erosion that resulted from the transformation of the plains into a cultivated and grazed landscape. In Great Plains states where trees are scarce and naturally occur primarily along streams, windbreaks and shelterbelts make up a significant proportion of woody habitat. In Nebraska, for instance, which is less than 2 percent wooded, windbreaks make up 25 percent of the woody cover (Soil Conservation Service 1989). The bulk of the available research on wildlife response to these practices is centered on field windbreaks.

As with other linear practices, windbreaks are often small features on the landscape, thus influencing wildlife habitat quality. For example, Hess and Bay (2000) used a habitat suitability index (0–1 scale) created by the U.S. Fish and Wildlife Service to assess the value of Nebraska windbreaks for wildlife, including birds, small mammals, and deer. They found that 50 percent of windbreaks had a suitability of 0.25 or lower, and no windbreaks rated above a 0.6. They suggested that expanding the size of individual windbreaks will increase habitat suitability for such species.

Not only do windbreaks, shelterbelts, and fencerows attract birds and small mammals, they also provide habitat for mammalian predators and raptors. While the presence of these predators may cause direct mortality to birds such as pheasants, or limit their nesting success, the predators themselves are valuable additions to the diversity of wildlife on the agricultural landscape.

An indirect effect on wildlife that is easily overlooked is the influence of windbreaks and shelter-

belts on wind speed, which is particularly important to flying insects. Windbreaks and shelterbelts have a measurable impact on arthropod communities that function both as pests and prey in cropping systems and food for other wildlife (Bhar and Fahrig 1998, Dix et al. 1995). As width is increased and plantings are diversified, more microclimate conditions are created and insect communities are larger and more diverse (Pasek 1988). Woody vegetation within buffers and field borders may serve as a refuge for insect pests and beneficial predators and also inhibit movement of crop pests (Bhar and Fahrig 1998, Dix et al. 1995).

Conclusion

Linear practices were primarily designed to be efficient at reducing water flow, trapping sediment, and filtering harmful substances associated with wind and water erosion before they reach streams and lakes.

Buffers are useful in terms of soil and water conservation and certainly provide wildlife habitat improvements over crop fields, but they have limitations that are associated with the small size and isolated nature of most practices. Recent research has provided some direction about how to maximize the benefits of linear practice buffers to wildlife (Table 3). Positive effects are associated with longer and wider buffers, buffers associated with or connecting other habitat practices such as blocks of cover or food plots, and with practices that are grouped on the landscape.

From a wildlife conservation standpoint, even well-managed, strategically placed linear practices cannot replace the established benefits of the 28 million acres of CRP contracts slated to expire before 2010 (FSA 2004). But better understanding of how landscape context affects the value of linear practices for wildlife will provide some future alternatives for agricultural conservation policy. The recently available Conservation Security Program (CSP) takes a

Table 3. Information sources for linear conservation practices and often-cited benefits and concerns relevant to wildlife.

Practice	Information Available	Benefits/Concerns
CP21 Grass Filter Strip	Murray 2002, Henningsen & Best 2005, Kammin 2003, Reeder et al. 2005	Hosts small mammals, arthropods, birds, but few specialists
CP22 Riparian Forest Buffer	Peak et al. 2004, Henningsen & Best 2005	Provides habitat for greater variety of birds than herbaceous plantings, but nest success low for some species
CP8 Grassed Waterway	Bryan & Best 1991 & 1994, Knoop 2004	Provides perennial cover, but few species, high predation rates on birds, too small for area-sensitive species
CP15 Contour Grass Strip		
Terrace	Hultquist 1999, Hultquist & Best 2001	Bird abundance higher than row crops but lower than other buffers, few nesting species, high predation rates
CP24 Cross Wind Trap Strip		
CP5 Field Windbreak	Hess & Bay 2000, Brandle et al. 2004	Provides physical structure/shelter, but too small for area-sensitive species
CP16 Shelterbelt		
CP17 Living Snow Fence		
CP33 Field Borders (Upland Bird Habitat Buffer)	Smith et al. 2005b	

watershed-level approach, including incentives for agreements between neighbors partnering to achieve a common conservation goal (NRCS 2005). Conservation Reserve Enhancement Program (CREP) projects are using Geographic Information System (GIS) data to inform choices about target locations for conservation practices. Linear practices have a potentially important value in providing flexibility while implementing this extensive view of conservation on agricultural landscapes. ■

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Benefits of Farm Bill Grassland Conservation Practices to Wildlife

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ABSTRACT Various Farm Bill conservation practices apply to rangelands with prescribed grazing, prescribed burning, range planting, and restoration of declining habitats showing some of the greatest benefits to wildlife. Prescribed grazing has been shown to produce both positive and negative responses by wildlife. Prescribed burning has also been shown to have both positive and negative effects, but benefits generally outweigh detriments of this practice. Range planting and restoration of declining habitats have been shown to benefit wildlife, but determining appropriate comparisons can be problematic. Grassland ecosystems have been found to need greater heterogeneity and better representation of historical ecosystem diversity, challenges that make comparisons to “native” ecosystem conditions complex. Additional practices including fencing, brush management, tree planting and shelterbelts, and pest management can all be used to improve wildlife habitat, although each can also cause problems for wildlife in certain situations. Bird responses to practices have received the greatest attention, with generally inadequate information available for most other taxa. Even for birds, considerable information is lacking including effects of practices on many species, effects of surrounding landscape factors on wildlife responses, and responses in reproductive rates or survival rates to various practices. Yet, rangeland practices offer some of the greatest potential for conservation benefits to wildlife. Grassland ecosystems and wildlife are considered among the most at risk, and rangeland practices can be used to maintain, enhance, and restore needed plant communities and habitat conditions.

Programs of the Farm Bill contain a number of conservation practices that are either directly or partially directed at grasslands. There are many types of grasslands in the United States, some of which are transient successional stages in systems that quickly become shrub- or forest-dominated communities, while other grasslands, particularly in the Great Plains, were historically the dominant plant community. This diversity of occurrences and types of grasslands makes summarizing wildlife responses to grassland practices complicated, especially if generalized to all grasslands. Because of the historical and current importance of grasslands in the Great Plains to a wide array of wildlife species, we will focus this chapter on wildlife responses to grassland conservation practices in grasslands and associated shrub ecosystems of the Great Plains.

The grasslands of the Great Plains historically occurred across approximately 585 million acres of the United States and Canada. These grasslands displayed considerable variation from north to south and east to west, with shrub species such as sagebrush occurring on sites protected from frequent fire on the western fringes, eastern forests occurring on fire-protected areas on the eastern fringe, aspen parklands occurring on the fringes to the north, and ponderosa pine and juniper forests occurring in rougher (i.e., shallow or rocky soils) or higher elevation areas within the interior. Grasslands have been identified as among the most endangered ecosystems in the United States (Samson and Knopf 1994, Samson et al. 2004), and many grassland-associated wildlife species are considered species at risk. Maintaining and improving the condition and diversity of grasslands are therefore significant conservation objectives.

In the United States, the grasslands of the Great Plains have been divided by the USDA/NRCS into approximately 60 Major Land Resource Areas (MLRAs) that delineate areas with similar geo-climatic characteristics (USDA/NRCA 2006a). Within each MLRA, ecological site descriptors identify the types of ecological communities that occurred within each ecological site and the various states and transitions that have occurred under current management practices. These provide a reference and information base for planning and implementing grassland management practices.

Various conservation practices included under Farm Bill programs are directed at native grasslands and are directly applicable to the untilled portions of the Great Plains. While Jones-Jones-Farrand et al. (this volume) discussed grassland practices associated with tilled or converted lands, emphasizing those associated with the Conservation Reserve Program, our chapter discusses those conservation practices that target untilled landscapes and focus on improving or restoring grassland conditions.

Programs that Utilize Grassland Conservation-Related Practices

Grassland conservation practices are widely used within the Grasslands Reserve Program (GRP), Wildlife Habitat Incentives Program (WHIP), Conservation Security Program (CSP), and Environmental Quality Incentives Program (EQIP). These programs focus on enabling and maintaining stewardship on working lands, which are those lands that are used to produce agricultural products. With a focus on untilled lands, this chapter primarily addresses practices applicable to rangelands.

The Wetlands Reserve Program (WRP) focuses primarily on wetland restoration and improvement. However, upland habitat adjacent to wetlands is typically also restored and enhanced. Such uplands are recognized as providing critical breeding habitat for many wetland species. They also significantly benefit wetlands by acting as buffers and filters from soil erosion, human disturbance and noise, and pesticides and fertilizers. Uplands that are maintained or restored to grassland and shrub cover types commonly utilize the kinds of practices discussed here.

Commonly Used Grassland-Related Practices

The kinds of conservation practices commonly used on grasslands address five main functions:

1. Establish and maintain desired plant species and communities,
2. Suppress and control invasive or undesirable plants and/or animals,
3. Provide food, water, or cover for desired native wildlife or domestic animals,
4. Manage domestic animals to minimize adverse

impacts to water bodies, soil resources, and desired wildlife and plant communities, and

5. Reduce wildfire hazard.

The following practices are those most commonly applied on grasslands. However, most of them can be utilized with many other land uses.

Brush Management

Brush management includes removal, reduction, or manipulation of non-herbaceous plants to achieve a particular objective (USDA NRCS 2006b; Conservation Practices Standard 314). On grasslands, brush management is used to restore natural plant community balance, create a desired plant community, improve forage accessibility for livestock, maintain or enhance wildlife habitat, reduce wildfire risk, and restore desired vegetation cover to protect soils, control erosion, reduce sediment, improve water quality, and enhance stream flow. Most often, brush management is used to control undesirable and invasive shrubs and trees through mechanical, chemical, biological, or prescribed burning treatments. Although the primary objective of brush management is usually to increase herbaceous vegetation for livestock, increasingly it is prescribed and applied to thin or eliminate woody vegetation such as juniper (*Juniperus* spp.) and mesquite (*Prosopis* spp.) that have encroached into grasslands, or to thin stands of Wyoming big sagebrush that have become too dense or decadent to provide many desired wildlife benefits (Olsen and Whitson 2002).

Prescribed Burning

Prescribed burning involves the application of controlled fire to a predetermined area (USDA NRCS 2006b; Conservation Practices Standard 338). Most grassland ecosystems in the United States evolved with frequent fire return intervals (Wright and Bailey 1982), which have largely been suppressed following extensive settlement (Seig 1997). Fire suppression in these areas has been linked with several ecological concerns, most notably the expansion of woody plants into areas in which they did not historically occur (Archer 1994). In grassland ecosystems, prescribed burning, as a conservation practice, is applied to control undesirable vegetation, prepare sites for planting or seeding, reduce wildfire hazards, improve wildlife



Prescribed burning in eastern Wyoming. (Photo by A. Ganguli, EMRI)



Aerial application of herbicide on a recently burned site to control invasive cheatgrass (*Bromus tectorum*). (Photo by A. Ganguli, EMRI)

habitat, improve plant productivity, remove debris or litter, alter distribution of grazing or browsing animals (Biondini et al. 1999, Fuhlendorf and Engle 2001), and to restore and maintain ecological sites.

Prescribed Grazing

Prescribed grazing is the act of managing the controlled harvest of vegetation with grazing animals (USDA NRCS 2006b; Conservation Practices Standard 528). Important components of developing grazing prescriptions are to specify the type of grazer, as well as the season, duration, and intensity of grazing that is needed to accomplish specific management objectives (Frost and Launchbaugh 2003). Prescription grazing is used in grassland ecosystems to improve or maintain the health and vigor of plant communities, control invasive plant species (Popay and Field 1996, Olsen et al. 1997), improve the quality and quantity of forage for livestock and wildlife

(Short and Knight 2003), maintain water quality and riparian area integrity (Sedgewick and Knopf 1991), improve wildlife habitat (Vavra 2005), reduce wild-fire risk, and reduce soil erosion.

Grazing Land Mechanical Treatment

Grazing land mechanical treatments utilize mechanical tools to modify soil and/or plant conditions with treatments such as pitting, contour furrowing, and ripping or subsoiling (USDA NRCS 2006b; Conservation Practice Standard 548). As a conservation practice it carries the restriction of only being applied to pastures where slopes are less than 30 percent. Mechanical treatments on grasslands are generally applied to fracture compacted soil layers to improve soil permeability, reduce water runoff and increase infiltration, break up sod-bound plant communities or thatch to increase plant vigor, and increase plant community productivity.

Range Planting

Range planting involves the establishment of adapted perennial grasses, forbs, legumes, shrubs, and trees (USDA NRCS 2006b; Conservation Practices Standard 550). Range planting is used as a conservation practice in grassland ecosystems to provide forage and habitat for livestock and wildlife, reduce soil erosion, improve water quality, increase carbon sequestration, and restore plant communities to a condition that is similar to historical conditions or to an identified desired plant community. Important considerations in developing range planting conservation practices include the economic feasibility, economic efficiency, and cost-effectiveness of the planting practice (Workman and Tanaka 1991), as well as an assessment of the potential competitive interactions of the species that will be used in the planting practice (Pyke and Archer 1991).

Stream Crossing

Stream crossings include stabilized areas or structures that are constructed across streams to provide crossing access for people, livestock, equipment, or vehicles that do not impede the natural passage of water, fish, or other organisms within the stream channel (NRCS NHCP; Conservation Practices Standard 578). The stream crossing conservation practice was established to reduce streambank and streambed

erosion, provide crossing for access to adjacent land units, and to improve water quality by reducing sediment, nutrient, organic, and inorganic stream loading. This practice is discussed by Knight and Boyer (this volume).

Water Development

Water development conservation practices include those that either collect, store, or deliver water. These include a variety of specific practices addressing water collection, watering facilities, creation of ponds or dams, water wells, and water distribution systems including irrigation, water conveyance, and



Windmill watering tank for livestock. (Photo by J. Haufler, EMRI)

pipeline practices. Water development practices are often aimed at protecting water sources and water supplies from contamination, as well as providing water for livestock and wildlife where water was previously unavailable. Water development practices for grasslands primarily serve to distribute livestock use evenly across pastures in order to maximize the use of forage resources without causing heavy grazing effects surrounding water source areas.

Pest Management

The conservation practice of pest management involves utilizing prevention, avoidance, monitoring, and suppression strategies in an environmentally sensitive manner to manage weeds, insects, diseases,

animals, and other organisms that cause damage or annoyance in a direct or indirect fashion (USDA NRCS 2006b; Conservation Practices Standard 595). Pest management is used to enhance the quantity and quality of commodities while minimizing any negative impacts to the environment or humans. Increasingly, pest management is applied as a part of Integrated Pest Management (IPM) programs which utilize chemical, cultural, and biological methods to control pests based on ecological, sociological, and economic factors (Allen and Bath 1980, Masters and Sheley 2001).

Tree and Shrub Establishment

Tree and shrub establishment includes the practices of planting seedlings or cuttings, direct seeding, and natural regeneration (USDA NRCS 2006b; Conservation Practices Standard 612). Tree and shrub planting in grasslands was initiated at settlement by pioneers from eastern states who longed for the trees they left behind in the East and needed timber for fuel, building materials, and aesthetics (Droze 1977). The United States government promoted tree planting through a number of programs including the Timber Culture Act of 1873, which granted homesteads of 160 acres, provided trees were planted on 40 of those acres (Droze 1977). In an effort to cope with the decline of soil and wildlife resources associated with unsustainable farming practices and droughts of the 1930s and 1950s, tree and shrub planting was promoted by federal action agencies (e.g., SCS), which culminated in modern state and federal planting programs for conservation (Glanz 1994). In grasslands, trees and shrubs are often planted to create windbreaks, shelterbelts, or hedgerows. The benefits associated with tree and shrub plantings include reduction of soil erosion, protection of plants from wind-related damage, retention of snow, enhancement of wildlife habitat, and provision of shelter for structures, animals, or recreational areas.

Fence Establishment

Fencing is constructed to form a physical barrier to animals or people (USDA NRCS 2006b; Conservation Practices Standard 382). As a conservation practice, fencing is intended to provide the means to control movement of animals and people to facilitate the application of other conservation practices. Examples of

fencing conservation practices include riparian zone exclusion (Keller and Burnham 1982), implementation of different grazing systems, modifications of fences to allow wildlife passage (Gross et al. 1983), and fencing to reduce livestock predation (Linhart et al. 1982, Nass and Theade 1988).

Restoration and Management of Declining Habitats

The conservation practice of restoring and managing declining habitats and associated wildlife is aimed at conserving biodiversity (USDA NRCS 2006b; Conservation Practices Standard 643). This conservation practice focuses on sites that either provide or previously provided habitat for rare and declining species. When compared to the other conservation practices reviewed in this chapter, this conservation practice involves incorporating several conservation practices to achieve objectives that may include restoration of lands degraded by human activity, restoration and conservation of native plant communities to provide habitat for rare and declining wildlife species, and increasing native plant community diversity.

Status of Great Plains Grassland Ecosystems

Grassland ecosystems of the Great Plains, like a majority of the ecosystems in the United States, have experienced considerable change. Historically, grasslands of the Great Plains covered vast tracts that were maintained and influenced by the interactions of fire and grazing in response to varying weather patterns. These grasslands have been generally classified into tallgrass, mixed grass, and short grass regions, depending upon the structure of the dominant species that historically occupied a site. Climate is a primary driver of where each type of grassland occurred, but fire and grazing played a role in determining the composition, structure, and function of grassland ecosystems as well (Knapp et al. 1999). While the extent and types of change affecting each category differ somewhat; all three types of grassland have undergone significant alterations. The extent of change has led some to consider grassland ecosystems among the most at-risk ecosystems in the country (Samson and Knopf 1994, Noss et al. 1995).

Grassland ecosystems have evolved with fire as a primary driver (Wells 1970, Brockway et al. 2002), particularly in the tallgrass and mixed grass ecosystems. Without fire as a disturbance process, many of these ecosystems would succeed to shrub or tree-dominated areas (Archer 1994). Fire was less of an influence in short grass ecosystems, but still played a critical role in shaping species composition, nutrient cycling, and discouraging the invasion of drought-tolerant shrubs or trees (Wright and Bailey 1982). The role of fire, with rare exceptions, has been largely eliminated in grassland ecosystems. This has modified the composition of species, altered nutrient cycling, and influenced grazing patterns of native herbivores, which in turn has influenced the structure of the vegetation. Grazing by native herbivores, especially bison (*Bison bison*), played a significant role in shaping and maintaining grass and shrub ecosystems in the Great Plains (Knapp et al. 1999, Hart and Hart 1997) and interacted with fire to create a shifting mosaic of conditions (Knapp et al. 1999, Fuhlendorf and Engle 2001). Although grazing by domestic animals is currently the primary use of grasslands, the foraging ecology of grazers that historically occupied the Great Plains differs from those used today, (Plumb and Dodd 1993) and the current grazing practices in grassland ecosystems have been found to dramatically differ from the historical role of herbivores (Fuhlendorf and Engle 2001). Existing livestock grazing practices have been focused on achieving even distribution of animals and even utilization, which produce relatively uniform or homogeneous vegetation conditions, a condition referred to by Fuhlendorf and Engle (2004) as “management to the middle.” Several grassland conservation practices of the Farm Bill, including water developments and grazing prescriptions, have been used to distribute grazing intensity relatively evenly across pastures, thus contributing to these uniform conditions.

Disruption of historical disturbance regimes has affected all types of grasslands in the Great Plains (Brockway et al. 2002). Conversion of grassland ecosystems to cultivation and other land uses has also had a significant influence. This influence has been greatest in the tallgrass ecosystems, where more than 99 percent conversion of sites with soils and topography favorable for cultivation has been reported (Samson and Knopf 1994). Conversion lev-

els in mixed grass and short grass ecosystems have been less than in tallgrass ecosystems, but can still have significant local impacts on wildlife species.

The net effect of the above impacts has resulted in serious concerns about reductions in grassland biodiversity. Grassland bird population declines are on a track to create a conservation crisis in these ecosystems unless current trends are reversed (Brennen and Kuvlesky 2005). Various studies have investigated mammals associated with Great Plains grasslands (see below for examples), but little information exists on current status of most mammals with respect to historical conditions and distributions. It is known that grizzly bears (*Ursus horribilis*) and wolves (*Lupus canadensis*) have been extirpated from the Great Plains. The black-footed ferret (*Mustela nigripes*), listed as a federally endangered species, was extirpated from the Great Plains but is currently in the process of being reintroduced to several locations (Dobson and Lyles 2000). Recent attention has provided considerable information on the status of the black-tailed prairie dog (*Cynomys ludovicianus*) and its role in creating ecosystems that help provide habitat for a number of associated species (Miller et al. 1994, Kotliar et al. 1999). Less is known about the effects of the above changes on other taxa. The current status of many species, including most grassland-supported reptiles, amphibians, insects, and many plants, remains largely unknown.

In addition to grassland species, sagebrush and other shrub ecosystems evolved in areas of the Great Plains that were not as heavily influenced by fire, although some shrub species in some areas are adapted to fire and quickly resprout following burning. Sagebrush-steppe ecosystems in the western United States, comprising some 44 million acres (Miller and Edelman 2000), are not specifically addressed in this chapter, although they share many of the concerns for sagebrush and other shrub systems associated with the Great Plains, including invasion by exotic species such as cheatgrass (*Bromus tectorum*). Concern exists for various species of wildlife associated with sagebrush ecosystems (Paige and Ritter 1999). Greater sage-grouse (*Centrocercus urophasianus*) have experienced significant declines (Schroeder et al. 1999) and have been considered for listing under the endangered species act. Major initiatives have been established to address the conservation of this species.

Wildlife Response to Grassland Conservation Practices of the Farm Bill

Grassland conservation practices can affect wildlife species in a number of ways. For example, conservation practices can affect the compositions, structures, nutritional quality, and other habitat features of specific sites. Wildlife use of an area is also influenced by the kinds of habitat features occurring in the surrounding area. This is particularly true where patchy or linear arrangements of grasslands occur, as discussed by Clark and Reeder (this volume). The overall status of a wildlife population in a given area will depend on the total availability of suitable habitat in a larger planning landscape or region. Thus, wildlife populations will be influenced by the overall types and arrangement of grassland ecosystems within a region as well as the occurrence of detrimental factors including barriers to movements, source areas for competing species, non-habitat related mortality factors, and other types of population threats.

Studies that specifically addressed Farm Bill-funded grassland conservation practices were not identified in the literature. However, considerable information on wildlife responses to grassland practices in general is available.

Grazing Practices

Great Plains grasslands, as discussed above, were historically dependent on grazing by native herbivores and fire as disturbance factors that shaped ecosystem diversity (Knapp et al. 1999, Fuhlendorf and Engle 2001). Current grazing by domestic livestock has been documented to create different responses in ecosystem diversity than historical conditions, but grazing can be used as an important management tool to achieve a variety of conservation objectives. Wildlife responses to grazing will depend on the type and intensity of grazing applied to specific ecological sites. Milchunas and Lauenroth (1993) conducted an extensive review of literature on effects of grazing. Among their findings was that grazing has a more significant effect on ecosystems that did not have an evolutionary history of extensive grazing. Great Plains grasslands have a well-documented history of grazing by native herbivores, while the historical role of grazing in sagebrush-steppe ecosystems is not well documented and is likely to have been a more minor influence.

The effects of grazing can be difficult to characterize because effects vary depending on the type of ecosystem and its evolutionary history, specific site differences, weather patterns during a study, surrounding land uses, intensity of grazing, response variable used to assess grazing effects, and other factors (Milchunas and Lauenroth 1993, Curtin 2002). Furthermore, studies often fail to account for many of these factors and may use quasi-experimental designs, so conclusions must be viewed cautiously (Jones et al. 2000).

Prescribed grazing as a Farm Bill conservation practice can be used to achieve a variety of conservation benefits. Two of the identified uses, improving or maintaining the health and vigor of plant communities and improving or maintaining the quantity and quality of food and/or cover available for wildlife, can have a wide range of interpretation. Current development of ecological site descriptions for grassland and sagebrush ecosystems identifies the range of specific states and their transitions that occurred under historical disturbances and current uses. Under historical disturbances, specific locations within a landscape may have experienced heavy levels of grazing by native herbivores, while other locations may have had light levels of grazing depending upon the landscape, proximity of water, history of fire events, surrounding topography, and other factors. Providing for all wildlife within a landscape may require that the full complexity of ecosystem diversity that occurred historically be represented within a landscape (Haufler et al. 1996, Haufler 2000). This makes understanding and specifically defining the desired health and vigor of plant communities — as well as the quantity and quality of food and/or cover for wildlife — complex.

Grazing effects on bird populations have received the most research relative to other taxa. Saab et al. (1995) summarized the findings of a number of studies on 43 grassland, shrubland, or riparian bird species. They reported that 17 species were negatively affected by grazing, 18 species were neutral, and eight species were positively affected by grazing. When compared with other grassland taxa, such as above- and below-ground macroarthropods, rodents, and rabbits, birds were found to be particularly responsive to grazing. Milchunas et al. (1998) and Brennan and Kuvlesky (2005) discussed how ecosystem diversity in grasslands must be maintained and restored

to address the needs of all grassland bird species. Milchunas and Lauenroth (1993) reported on results of a number of studies conducted on birds.

A number of studies have reported on the response of mammals to grazing. Phillips (1936) investigated use of sites receiving different levels of grazing by various rodents and lagomorphs in Oklahoma and found some species prefer heavily grazed areas while others were more abundant in “normal” areas. Grant et al. (1982) and Clark et al. (1989, 1998) studied the response of small mammals to grazing of grasslands and found that species respond differently to grazing. Deer mice (*Peromyscus maniculatus*) tended to increase on grazed sites, while species that require more grass cover or litter such as harvest mice (*Reithrodontomys* spp.) or voles (*Microtus* spp.) tended to prefer ungrazed areas. Matlack et al. (2001) compared deer mice use of areas grazed by both cattle and bison following burning in a tallgrass prairie in Kansas and noted different abundances in various seasons they investigated. This illustrates that providing habitat conditions for all species of native small mammals, as with birds, requires providing representation of the full range of ecosystem diversity. Prescribed cattle grazing has been successfully used to achieve more specific management objectives, such as improving forage quality on rough fescue grasslands for elk and deer (Short and Knight 2003).

Effects of grazing in grasslands on other taxa have not received extensive research. Kazmaier et al. (2001) reported on the response of Texas tortoises (*Gopherus berlandieri*) to moderate levels of grazing, and found no effects. Similarly, Ballinger and Jones (1985) reported no effects of grazing on a lizard community in western Nebraska. Joern (1982) and Quinn and Walgenbach (1990) investigated the response of grasshoppers to grazing.

Effects of grazing in sagebrush ecosystems have received less attention than in grassland ecosystems. Beck and Mitchell (2000) compiled available literature and discussed the influences of livestock grazing on sage-grouse. They found both positive and negative impacts of livestock grazing and related these impacts to both direct and indirect effects. They reported that indirect effects of livestock grazing (e.g., herbicide or mechanical reductions in sagebrush to increase forage production) have had greater impacts to sage-grouse than direct impacts.

include loss of food and cover for sage-grouse associated with livestock consumption of grasses and forbs. Crawford et al. (2004) reported that livestock grazing can have negative or positive effects on sage-grouse depending on the timing and intensity of grazing. However, judiciously applied livestock grazing prescriptions can be a valuable tool to help restore sagebrush ecosystems for sage-grouse (Vavra 2005).

In total, these studies indicate that wildlife responses to grazing can range from beneficial, to neutral, to negative. Great Plains grasslands evolved with considerable grazing pressure from bison and other herbivores. However, current grazing by domestic livestock is often conducted in intensities and durations across large landscapes that produce different conditions when compared with historical grazing by native herbivores. Prescribed grazing as a Farm Bill conservation practice can be used as an effective tool to produce desired plant community conditions, but can also produce negative effects.

Prescribed Burning

Fire, as discussed above, is an integral process to the maintenance and potential restoration of grasslands in the Great Plains and plays an important role in periodically setting back sagebrush ecosystems. Effects of the prescribed burning practice under the Farm Bill have not been specifically researched, however, a number of studies on wildlife responses to burning in grassland and sagebrush ecosystems have been conducted.

The influence of prescribed burning on wildlife varies by species, the season fire is applied, and by the fire return interval. Fire (and its exclusion from some areas) can be important to maintaining grassland heterogeneity. Several studies have reported on the importance of grassland heterogeneity to an area, as species with different habitat needs respond to the various conditions provided by this heterogeneity (Fuhlendorf and Engle 2001, Fay 2003, Bechtoldt and Stouffer 2005, Powell 2006). The season that fire is applied can influence wildlife species by altering habitat, forage, potential prey species, or by causing direct mortality. Spring burning has been found to have direct detrimental effects to several vertebrate species in grasslands (Erwin and Stasiak 1979). However, most often the effects of season of prescribed fire on wildlife are indirect, such as modification of

nesting habitat, insect populations, or forage availability (Towne and Ownesby 1984, Pyle and Crawford 1996, Fischer et al. 1996, Bechtoldt and Stouffer 2005). Prescribed burning programs that promote fire regimes that are not consistent with the historical fire regime of an area can be detrimental. This was demonstrated in the Flint Hills of Kansas where annual spring burning with intensive grazing was found to reduce the abundance of grassland birds (Powell 2006).

For management of biological diversity in the Northern Great Plains, Sieg (1997) recommended applying fire at different times of the year and at intervals that vary to better mimic how fire historically occurred on the landscape. The concept of increasing habitat heterogeneity through patch burning, which creates a shifting mosaic of vegetation successional stages, has been tried in tallgrass prairies (Fuhlendorf and Engle 2001, Fuhlendorf and Engle 2004) and has been recommended as an appropriate strategy to manage biological diversity in systems that were historically maintained by fire and grazing (Bechtoldt and Stouffer 2005, Fuhlendorf et al. 2006, Powell 2006, Wilgers and Horne 2006).

Range Planting and Restoration and Management of Declining Habitats

As stated previously, providing representation of the full ecosystem diversity that occurred in an area historically may be a desirable objective for grassland management, which may be addressed using various conservation practices and their combinations. Many grassland areas lack this representation as a result of past management practices that have produced relatively uniform conditions in terms of ecosystem diversity. Management of declining habitats is a grassland conservation practice that directly addresses the need to maintain or restore plant communities that are lacking in some way and are suspected of causing a decrease in populations of desired species. Restoring these plant communities on existing grasslands may require the use of a number of other specific conservation practices including range planting, prescribed burning, prescribed grazing, control of invasive or undesirable species, brush management, and others.

A number of studies have investigated wildlife responses to grassland restoration, where restoration was from croplands back to grasslands (Blankespoor

1980, Fletcher and Koford 2002, 2003, Farrand et al. this volume). One investigative approach used in these studies was to compare restored grasslands with wildlife use of croplands, as used in many of the studies cited in Jones-Farrand et al. (this volume). This approach has demonstrated conservation benefits from CRP programs, but does not provide good insights into grassland restoration efforts applied to working rangelands where grasslands currently occur but may be in relatively uniform or undesirable compositions or structures. A second approach used in restoration studies is to compare wildlife, typically birds, in the restored areas with wildlife in “native” prairies. Two questions arise in such investigations; how did the “restored” area compare with historical conditions, and what was the condition of the “native” area that was being used as a comparison. CRP practices, discussed by Jones-Farrand (this volume) restore croplands to permanent grass cover (usually for a 10-year commitment). Many of these restored sites use native seed mixtures. But were these seed mixtures designed to restore the compositions of specific plant communities that would have occurred on a site historically and, if so, under what type of grazing and fire regime? It is known from ecological site descriptions (USDA NRCS 2003) that various historical states occurred on each ecological site, depending on fire and grazing effects. Restoration needs can be prioritized to target those states most lacking in the landscape (see Thunder Basin case study in Franklin et al. this volume), and appropriate seed mixtures and other practices utilized to restore these needed plant communities. For comparative purposes in restoration studies, “native” communities selected for comparison should be identified to represent specific historical states appropriate for a site and not assumed to represent all “native” communities in the landscape. Thus, studies of grassland restoration have a number of key questions to address to accurately reflect the measurement of restoration.

Evaluation of range planting within working landscapes (e.g., rangelands) needs additional research. Studies are needed that compare a “restored” site with an established baseline condition. Range planting has been identified as a successful method of reducing weed species in tallgrass prairie and as having the potential to reduce the ability of invasive plant species to successfully invade a plant community (Blumenthal

et al. 2003). Range planting has also been successfully used as a component of IPM to accomplish multiple management objectives such as suppression of an invasive species, establishment of desirable native species, and to increase forage productivity (Masters et al. 2001, Masters and Shelley 2001).

Recent efforts to improve specific habitat for declining species have shown successes. EQIP funding was specifically targeted for sage-grouse habitat improvements, and various projects were initiated to improve sagebrush ecosystems for this species. Practices have included control of cheatgrass, mechanical treatment of decadent stands of sagebrush, range planting with species utilized by sage-grouse, and prescribed grazing. While most of these efforts are ongoing, and information on their effectiveness has not been reported to date, they indicate the ways that restoration of declining habitat can be implemented.

Tree and Shrub Establishment

Tree and shrub establishment in the Great Plains grasslands has provided a form of wildlife habitat enhancement, especially in areas that have experienced higher levels of conversion to production agriculture. Several species of birds and mammals have been documented to use tree and shrub plantings for habitat (Johnson and Beck 1988, Schroeder et al. 1992). Characteristics such as size, width, height of the tallest tree or shrub, snag density, and foliage height diversity of shelterbelts have been identified as important determinants of the diversity of avian species that use shelterbelts (Schroeder et al. 1992).

While a form of wildlife habitat enhancement is accomplished by tree planting in prairies, many species that use planted trees and shrubs for food and cover are habitat generalists that thrive at the expense of native prairie habitat specialists (Henzlick 1965, Coppedge et al. 2001a, 2001b, Clark and Reeder this volume). In fact, tree planting and woody plant expansion are associated with loss of grassland biodiversity including the recent decline of grassland birds (O'Leary and Nyberg 2000, Coppedge et al. 2001a), the fastest declining bird guild in North America (Knopf 1994, Herkert 1995). Furthermore, nesting success has been shown to decrease in some species that use trees and shrubs established along fencelines, which are similar to the linear habitats provided by windbreaks and shelterbelts, indicating

that these linear habitat features can act as habitat sinks because they attract higher rates of predation (Yosef 1994).

Several species that have been planted for conservation practices are either non-native to the United States, such as Russian olive (*Elaeagnus angustifolia*) and multiflora rose (*Rosa multiflora*), non-native to the region in which they are planted, or native but invasive in the absence of historical disturbances such as eastern redcedar (*Juniperus virginianus*) expansion in the absence of fire. To avoid further degradation of grassland ecosystems, it is critical to select species for conservation planting practices that are listed in the historical climax plant community within ecological site descriptions that are appropriate for the site and are not likely to invade. Ecological risk assessment can provide a valuable tool to screen and evaluate the invasive potential of species currently used in planting programs, as well as prevent the introduction of new invasive species (Lodge and Shrader-Frechette 2003).

Fencing

Fencing is used in grasslands to keep livestock in designated areas and out of others. This allows areas to be protected from grazing, trampling, and other impacts from livestock. Benefits include development of better habitat for various species as well as protection of stream banks, water quality, and aquatic habitat (Knight and Boyle this volume). However, fencing can also have detrimental effects on wildlife. Poorly designed fencing can create barriers to animal movements, keeping animals from important habitat areas, and can ensnare wildlife (Jackson Hole Wildlife Foundation no date).

Research Needs

Considerable information, as identified in this chapter, is available on wildlife responses to many of the conservation practices applicable to grasslands. However, due to the complexities of wildlife responses, interactions among practices, varying responses in different locations, and temporal differences due to varying weather patterns, much more information and monitoring are needed. For example, Winter et al. (2005) pointed out that unlike forest ecosystems, veg-

etation structure in grasslands can vary dramatically from year to year. They noted that no large scale studies have been conducted that have evaluated grassland bird densities and nesting success as responses to vegetation dynamics across large areas or long time spans. They also noted differences in responses to vegetation dynamics of three species they examined.

One of the greatest needs is establishing definitions and understanding of what are “native” grasslands. This term is loosely used, often referring to unplowed areas that support some mix of predominantly native plant species. However, do such areas actually represent native grassland conditions in terms of compositions, structures, and processes, or do they represent the conditions resulting from the “management to the middle” (Fuhlendorf and Engle 2004) that has caused reductions in grassland heterogeneity and declines in many wildlife species? Until a better baseline is established and recognized that describes an appropriate range of states for ecological sites across delineated planning areas such as Major Land Resource Areas delineated by NRCS, references to native grassland ecosystems will be problematic.

Many of the practices described in this chapter result in mixed responses by wildlife species. The literature review clearly documented this for prescribed burning and prescribed grazing. With various species benefiting while others are impacted by any specific practice, it is clear that a mix of practices must be utilized to maintain and increase grassland heterogeneity. Research is needed that addresses the most effective and efficient ways of creating this heterogeneity in different grassland ecosystems.

Most available information has examined responses by wildlife species to changes in habitat conditions at specific sites. Information is lacking on landscape influences that can result in varying responses by a wildlife species to conditions at a specific site. While some studies, especially a number of the more recent investigations, often include measurements of these factors, complexities in experimental designs required to effectively address landscape factors make these studies difficult. With annual differences in weather often confounding results, as noted by Winter et al. (2005), larger scale and longer term studies are needed.

Grassland birds are the most studied of the grassland taxa. Considerably more information is needed

on all of the other taxa. However, as noted above, even many basic questions about grassland birds still remain unanswered.

As conservation practices are applied, they should be monitored, and when feasible, use an adaptive management approach (Franklin et al. this volume). Providing replicated application of practices can be challenging, but is important to incorporate if the deficit of information on grassland responses to conservation practices is to be corrected. ■

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Fish and Wildlife Benefits Associated with Wetland Establishment Practices

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ABSTRACT Efforts to establish wetlands through restoration and creation actions have increased in recent decades in response to regulatory and voluntary incentive programs. This paper summarizes the findings of studies conducted to document fish and wildlife response to these practices. The majority of published studies describe bird response to wetland restoration, with most reporting bird communities in restored wetlands to be similar to those of natural reference wetlands. Studies also indicate that invertebrates and amphibians generally respond quickly to and colonize newly established wetland habitats. Key factors reported as correlated with wildlife species richness include wetland size, availability of nearby wetlands habitats, diversity of water depths and vegetation, wetland age, and maintenance and management activity. Key knowledge gaps in our understanding of fish and wildlife response to wetland establishment practices are identified, including the need for studies on biota other than birds and long-term monitoring of wetland condition and wildlife response over time.

Wetlands have been shown to provide a variety of ecological, biological, and hydrologic functions that provide economic, aesthetic, recreational, educational, and other values to society (Mitsch and Gosselink 1986, National Research Council 1992, Heimlich et al. 1998). However, these values were poorly recognized in the United States during the 19th and most of the 20th centuries. Numerous federal incentives encouraged wetland drainage, ranging from direct support for wetland “reclamation” under the Swampland Acts of 1849, 1850, and 1860, to agricultural subsidies that

indirectly supported conversion of wetlands to crop production (U.S. Department of the Interior 1988, Heimlich et al. 1998).

Conversion of wetlands to agricultural production has greatly impacted fish and wildlife habitats throughout the world (Lemly et al. 2000). In North America at the time Europeans arrived, there were approximately 221 million to 224 million acres of wetlands in what is now the conterminous United States (Dahl 1990). By 1992, 45 percent to 50 percent of the original wetland area in the lower 48 states had been converted to agricultural and other

uses, with losses approaching 90 percent in some states (Heimlich et al. 1998).

The 1985 Food Security Act's Wetlands Conservation (Swampbuster) provision and the 1986 Tax Reform Act largely eliminated indirect government support for wetland conversion (Heimlich et al. 1998). Since 1985, the Conservation Title of the 1990, 1996, and 2002 Farm Bills has supported the protection and restoration of wetland resources through a variety of U.S. Department of Agriculture (USDA) conservation programs.



Example of wetland conversion (i.e., draining) for agricultural production. (Photo courtesy of USFWS)

Wetland Conservation Practices

A variety of conservation practices that affect wetlands are implemented through USDA conservation programs and technical assistance provided by Natural Resources Conservation Service (NRCS) conservation planners to owners and operators of agricultural lands and other USDA clients. For the Conservation Reserve Program (CRP), similar wetland-related conservation practices with slightly different codes and definitions are applied by the Farm Service Agency. For the purpose of this chapter, practices that are typically viewed as directly affecting wetland function have been selected for treatment. While other conservation practices relating to land treatment and management can and do affect wetland functions in a variety of ways (Lowrance et al. 2006), those practices are addressed in other chapters of this publication. Practices addressed here are those listed and defined in Table 1. There are a number of other practices that are typically used in wetland restoration and management activities (e.g., dike, structure for water control, tree/shrub establishment, etc.). However, these practices are also used in a

Table 1. USDA conservation practices with direct connection to wetland function.

Practice (Acres)	NRCS Practice code	FSA Practice code (CRP)	Definition ¹
Wetland Creation	658		The creation of a wetland on a site that was historically non-wetland.
Wetland Enhancement	659		The rehabilitation or reestablishment of a degraded wetland, and/or the modification of an existing wetland.
Wetland Restoration	657	CP23 ² CP27 ³ CP31 ⁴	The rehabilitation of a degraded wetland or the reestablishment of a wetland so that soils, hydrology, vegetation community, and habitat are a close approximation of the original natural condition that existed prior to modification to the extent practicable.
Wetland Wildlife Habitat Management	644		Retaining, developing, or managing wetland habitat for wetland wildlife.
Shallow Water Development and Management	646	CP9	The inundation of lands to provide habitat for fish and/or wildlife.

¹Definitions are from the NRCS National Conservation Practice Standards from the National Handbook of Conservation Practices (www.nrcs.usda.gov/technical/Standards/nhcp.html).

²Includes CP23 (floodplain wetland) and CP23a (non-floodplain wetland) restoration.

³Wetland restoration through the CRP Farmable wetland program, including buffer areas (CP28).

⁴Tree planting associated with wetland restoration on land enrolled in CRP.

Table 2. Practices related to wetlands planned in FY 2004 under a variety of USDA conservation programs.¹

Practice	NRCS Practice code	Conservation Program (acres)					All programs ²
		WRP	WHIP	EQIP	CTA	CRP	
Wetland Wildlife Habitat Management	644	75,102	36,769	15,100	178,538	30,877	444,474
Shallow Water Development and Management	646	4,461	4,922	6,549	8,399	1,408	26,759
Wetland Restoration	657	98,613	9,316	1,088	38,829	71,862	220,878
Wetland Creation	658	3,493	119	205	3,389	1,118	8,324
Wetland Enhancement	659	5,026	601	827	30,586	710	37,795

¹WRP Wetlands Reserve Program; WHIP Wildlife Habitat Incentives Program; EQIP Environmental Quality Incentives Program; CTA Conservation Technical Assistance; CRP Conservation Reserve Program.

²Total includes acres planned under programs not listed.

Source: USDA System 36 database.

wide variety of other applications that do not have to do with wetlands and therefore are not included in this chapter.

Cost-share and technical assistance is available through several USDA conservation programs. Table 2 provides acreages of wetland conservation practices planned during FY 2004 under various USDA conservation programs. Table 2 is intended to give readers an idea of the types of wetlands conservation activities under way during a single planning year, rather than a comprehensive cumulative total of all wetlands affected across all programs.

Documented Fish and Wildlife Response

This paper compiles available literature that describes fish and wildlife response to conservation practices applied to wetland systems. Documented effects are grouped by major taxa reported in the literature. Much of the literature relates to a combination of practices. In many instances, wetland restoration and creation are indistinguishable in terms of fish and wildlife response. In other cases, wetland enhancement measures studied are indistinguishable from wetland management actions, and many wetlands that are managed for wildlife have been previously subject to wetland restoration (e.g., see

Marburger 2002, Bryan et al. 2003). For this reason, it is difficult to sort the literature by NRCS defined conservation practices listed in Table 2. Where possible, distinctions are made between two broad categories of wetland conservation activity: 1) wetland establishment (including Wetland Restoration and Wetland Creation) and 2) wetland management (including Wetland Wildlife Habitat Management, Shallow Water Development and Management, and Wetland Enhancement). This paper focuses primarily on summarizing the literature on fish and wildlife response to wetland establishment practices.

Rewa (2000) summarized the literature related to the fish and wildlife response to the Wetlands Reserve Program by examining reported effects of wetland restoration and creation reported in the literature and extending these findings to the WRP where applicable. Information contained in that review related to wetland practices and the fish and wildlife response reported is included here, along with additional results reported since the 2000 report was completed.

Invertebrates

Several studies have shown that soon after wetlands are restored or created, they are quickly colonized by a variety of aquatic invertebrates and other animals (Reaves and Croteau-Hartman 1994, Juni and Berry

2001). Brown et al. (1997) found similar invertebrate taxa between natural wetlands and restored wetlands in New York. Insects with aerial dispersal colonized restored wetlands more rapidly than less mobile invertebrates. In recently constructed coal surface mine sediment ponds, Fowler et al. (1985) found 66 and 44 invertebrate taxa in the first and second years sampled, respectively, indicating rapid invertebrate colonization.

The invertebrate fauna of restored wetlands is typically characterized as very similar to natural wetlands with similar vegetation structure (Brown et al. 1997, Zimmer et al. 2000, Juni and Berry 2001). Mayer and Galatowitsch (1999) found diatom species richness and composition in restored prairie wetlands in North Dakota to be similar to that of natural wetlands. LaGrange and Dinsmore (1989) found a total of 18 wetland invertebrate species in four formerly drained prairie wetland basins several years after the basins were reflooded. In a survey of 156 restored seasonal and semi-permanent wetlands of 12 different ages in Minnesota and South Dakota, Sewell and Higgins (1991) found 31 taxa of aquatic macroinvertebrates in restored wetlands, 12 of which occurred in wetlands the first year following restoration. Restored prairie pothole wetlands are generally believed to be readily and adequately colonized by invertebrates, although invertebrate community differences between restored and natural wetlands may have gone unnoticed due to the low taxonomic resolution at which most invertebrate communities are sampled (Knutsen and Euliss 2001).

Benthic invertebrate communities are strongly associated with wetland vegetation (Streever et al. 1995). In a created freshwater herbaceous wetland in central Florida, Streever et al. (1995) found three of five common Chironomid genera were more abundant in areas with greater than 50 percent herbaceous cover than more open areas and greater abundance of all five common genera in areas with greater than 80 percent vegetation cover. Transplantation of remnant wetland soil that increases the rate of wetland plant growth can also increase overall invertebrate abundance in restored wetlands (Brown et al. 1997).

Invertebrate taxa can be used to assess biotic response to restored wetlands (Brown et al. 1997). However, significant spatial and temporal variation

must be considered. Dodson and Lillie (2001) found zooplankton taxon richness in restored wetlands in Wisconsin mimicked that of least-impacted reference wetlands within six to seven years after restoration. Ettema et al. (1998) found spatial distribution within a restored wetland in Georgia varied substantially among nematode taxa, with substantial temporal variation within taxa. Distribution of nematode taxa did not correlate well with soil resource patterns. In a rehabilitated wetland in northern Spain, Valladares Diez et al. (1994) found that a diverse community of Coleoptera had developed, but most species found belong to early successional groups or are ubiquitous. In the same restored wetland, Gonzales Martinez and Valladares Diez (1996) found aquatic Heteroptera and Odonata communities to be similar to natural immature wetlands (ubiquitists and pioneers). In general, the communities of beetles, dragonflies, and aquatic heteroptera are representative of recent wetlands, with evidence of changes toward a more stable and mature environment.

The presence of fish in restored wetlands may also influence how invertebrates respond to restored wetland conditions. Zimmer et al. (2000, 2002) found the presence of fathead minnows (*Pimephales promelas*) to have a major influence on the invertebrate community structure in restored prairie wetlands in Minnesota. However, Dodson and Lillie (2001) found no influence of the presence of fish on the zooplankton community of restored wetlands in Wisconsin.

Fish

The effect of wetland establishment on fish communities has not been extensively investigated. Wetland geomorphic and geographic setting appears to have a significant influence on how the fish community responds. Within two years of development of a constructed wetland in east-central Florida, Langston and Kent (1997) observed a rich and abundant fish community that was similar to natural wetlands in the area. They surmised that in this geographic setting, fish may have been introduced to the wetland through irrigation or transport by local fauna.

In other settings, such as shallow prairie wetlands that are typically isolated from deeper water bodies, fish have not played a significant role in the development of biological communities inhabiting these wet-

lands. Recent studies have shown that introduction of fish into historically fish-free prairie wetlands can negatively affect native fauna such as invertebrates, amphibians, and waterbirds (Knutson and Euliss 2001). Likewise, agricultural ponds in Minnesota free of fish have been found to be more likely to support diverse populations of amphibians than those with fish (Knutson et al. 2004).

Herpetofauna

Several studies illustrate rapid amphibian colonization of constructed and restored wetlands. Lehtinen and Galatowitsch (2001) found restored wetlands in Minnesota to be rapidly colonized by eight amphibian species, all of which established breeding populations. Fowler et al. (1985) documented 12 species of breeding amphibians in newly constructed coal surface mine sediment ponds in western Tennessee, and all nine ponds surveyed contained at least one breeding amphibian species. Anderson (1991) found American toads (*Bufo americanus*), green frogs (*Rana clamitans*), and leopard frogs (*Rana pipiens*) using recently restored wetlands in Wisconsin. Lacki et al. (1992) found that a wetland constructed for treatment of mine water drainage in east-central Ohio supported greater abundance and species richness of herpetofauna than surrounding natural wetlands. This was primarily due to the large number of green frogs and pickerel frogs (*Rana palustris*) and numerous species of snakes found using this site.

Stevens et al. (2002) found a greater number of anurans calling from restored wetland basins on Prince Edward Island than from similar reference wetlands. This may have been due in part to the greater amount of microtopography in restored wetlands resulting from the actions of removal of fill material from these sites as the primary restoration action.

Landscape condition and surrounding land use appear to be critical components that influence amphibian colonization and use of restored wetlands. In glacial marshes in Minnesota, Lehtinen et al. (1999) found amphibian species richness was lower with greater wetland isolation and road density at all spatial scales in both tallgrass prairie and northern hardwood forest ecoregions. Limited dispersal capability likely contributes to slow colonization of restored wetlands by amphibians in fragmented landscapes

(Lehtinen and Galatowitsch 2001). Likewise, elimination of small wetlands that are relied upon by reptiles and amphibians can have a devastating effect on habitat availability and populations of these animals (Gibbs 1993).

Although studies have shown rapid amphibian colonization of restored and created wetlands accessible by dispersing individuals, there remains significant uncertainty concerning the long-term viability and population dynamics in these sites (Petranka et al. 2003).

Birds

The response of birds to wetland conservation practices is better documented than for other wildlife taxa (Knutson and Euliss 2001). Numerous studies have documented extensive bird use of restored freshwater wetlands (Guggisburg 1996, Sleggs 1997, Muir Hotaling et al. 2002, Stevens et al. 2003, Brasher and Gates 2004). LaGrange and Dinsmore (1989) found a total of 11 bird species in four formerly drained prairie wetland basins several years after the basins were reflooded. Anderson (1991) monitored wildlife use of small restored wetlands in Wisconsin and documented ducks and duck broods and nesting marsh wrens (*Cistothorus palustris*), sandpipers, and woodcock (*Scolopax minor*) using these habitats. Fletcher and Koford (2003) observed an increase in many bird species of management concern in response to restoration of prairie-wetland complexes in Iowa. Although no quantitative data were collected, Oertel (1997) noted substantial increases in wetland-associated wildlife use following restoration of a 55-acre wetland in northern New York. Dick (1993)



High density waterfowl use of a wetland. (Photo courtesy of W. Meinzer, USFWS)

observed wetland-dependent birds using an 80-acre restored wetland site in south-central Pennsylvania during the first year after restoration. Bird groups observed included winter raptors, wintering and migrating ducks, geese and tundra swans (*Cygnus columbianus*), foraging wading birds, waterfowl and shorebirds, and other birds. Breeding mallards (*Anas platyrhynchos*), wood ducks (*Aix sponsa*), sora (*Porzana carolina*), sedge wrens (*Cistothorus platensis*), common snipe (*Gallinago gallinago*), spotted sandpiper (*Actitis macularius*), and pied-billed grebe (*Podilymbus podiceps*) were documented. Restoration of the wetland increased bird diversity by 60 percent during the first year. In restored wetlands in central New York, Kaminski (2005) found survival probabilities for female nesting mallards to be comparable with those of mallard populations in natural wetland systems.

In most situations, birds rapidly colonized restored wetlands, usually in the first year after restoration. Delehanty and Svedarsky (1993) found breeding black terns (*Chlidonias niger*) using a restored prairie wetland during the second and third breeding seasons after restoration. As many as 40 adults were present in the marsh during the third breeding season, and a minimum of seven young were fledged. Sewell and Higgins (1991) found 12 species of waterfowl using restored wetlands of varying ages in Minnesota and South Dakota. During the first five years after restoration, White and Bayley (1999) documented 50 shorebird species, 44 waterfowl species, 15 raptor species, and 28 other new bird species using a 1,246-ha formerly drained northern prairie wetland that was restored and flooded with municipal wastewater. In the case of bottomland hardwood wetland restoration, studies have shown that birds associated with grasslands and scrub-shrub communities readily use these sites as they transition from open field to forested habitats (Twedt et al. 2002, Twedt and Best 2004). These studies show how quickly wetland-associated birds respond to restored wetland habitats. However, bird response to created bottomland hardwood wetlands may be somewhat less predictable due to the variability of wetland (or non-wetland) conditions established. For example, Snell-Rood and Cristol (2003) found that created bottomland hardwood wetlands in Virginia had significantly lower bird species richness and diversity

than similar reference wetlands. The authors of this study hypothesized that the lack of bird response was likely due to unnatural patterns of hydrology and poor vegetation development in created wetland sites.

In most studies in the literature, bird use was found to increase with the size of restored wetlands examined. Brown and Dinsmore (1986) found more diverse bird communities in larger prairie marshes. Among restored emergent wetlands in Wisconsin, Guggisberg (1996) found that large restored wetlands had greater non-game bird species richness than did small wetlands. In restored herbaceous wetlands in northern Iowa, Hemesath and Dinsmore (1993) found that breeding bird species richness increased with wetland size, regardless of how long the wetlands were restored or the duration of prior drainage. Analysis of data collected on bird use of wetlands restored in central Iowa under the Farmable Wetlands Conservation Reserve program imply a strong correlation between wetland size and bird species richness (R. N. Harr, Iowa State University, unpublished data). However, others have documented changes in the bird community with the amount of time following wetland restoration in response to changes in vegetation (Wilson and Twedt 2005). Vanrees-Siewert and Dinsmore (1996) found that total bird species richness increased with the age of restored prairie wetlands in Iowa, while waterfowl use (breeding and total) was influenced more by restored wetland size, regardless of age.

Habitat structure in restored wetlands appears to be a primary element that determines bird use of individual wetland sites. Density of waterfowl breeding pairs was lower in borrow ponds constructed along a highway in North Dakota than in natural basins of similar size (Rossiter and Crawford 1981, 1986). This was attributed to lack of a shallow water area and emergent wetland vegetation in borrow area wetlands. During drought conditions, Ruwaldt et al. (1979) found spring waterfowl pair use in South Dakota was greater in semi-permanent natural wetlands and artificial stock ponds than in other wetland types, indicating the importance of surface water availability to breeding waterfowl.

Bird use of restored wetland systems has been shown to be similar to that of natural wetlands with similar habitat structure. Ratti et al. (2001) did not detect any difference in bird abundance, species rich-

ness, or species diversity between 39 natural prairie wetlands and 39 restored wetlands in North and South Dakota. Brown and Smith (1998) found that the number of bird species and individuals did not differ between restored and natural wetlands in New York for the three bird groups studied (wetland-dependent, wetland-associated, and non-wetland birds). They found bird communities were more similar among restored sites than between restored and natural wetland sites. Thompson (2004) found similar bird species richness and diversity among restored and natural wetlands in Michigan, with restored sites supporting higher densities of wetland dependent birds. Delphey and Dinsmore (1993) found species richness of breeding birds was higher at natural wetlands than restored prairie wetlands. However, duck species richness and pair counts did not differ between natural and restored wetlands. Drought during the study may have influenced results.

Brown (1999) found more plant species valuable as food sources for wetland birds and greater coverage of these species occurred in restored wetlands than in natural wetlands in New York. Differences in bird similarity between natural and restored wetlands may disappear as restored wetlands develop over time (Brown and Smith 1998).

While bird use is related to the size of restored wetlands, it is also influenced by the proximity to other wetland habitats (Reaves and Croteau-Hartman 1994). The condition of upland habitats adjacent to wetlands and the surrounding landscape greatly influences use of restored wetlands by many bird species. Local wetland conditions dictate habitat suitability for some wetland bird species that are relatively sedentary, while wide-ranging species are greatly affected by the condition of the landscape surrounding wetland habitats. Naugle et al. (1999) found that while pied-billed grebes and yellow-headed blackbirds (*Xanthocephalus xanthocephalus*) used wetlands in South Dakota based on the condition of the habitat within wetlands, use of wetlands by black terns, a wide-ranging species, was dictated more by the use and condition of the surrounding landscape.

Habitat diversity within individual wetlands is associated with bird use. Fairbairn and Dinsmore (2001) found bird diversity to be positively associated with the percentage of wetland area with emergent vegetation within wetland complexes, total wetland area

within three km, and total area of semipermanent wetlands within three km of wetland complexes. Likewise, McKinstry and Anderson (2001) found the presence of emergent and submersed wetland vegetation and the presence of nearby wetlands to be important factors in determining waterfowl use of created wetlands on mined lands in Wyoming. Naugle et al. (2000) found black tern use of prairie wetlands was largely correlated with wetland area, amount of semi-permanent wetland area within the wetland, and grassland area in the surrounding upland matrix. Black tern use was associated with large wetland basins located in high-density wetland complexes, illustrating the importance of considering entire landscapes in habitat assessments and conservation efforts.

Landscape Factors

Wildlife response to wetland restoration may be as much a function of the presence of other wetlands nearby and overall landscape condition as the state of wetland habitats evaluated (Griffiths 1997, Haig et al 1998). Fairbairn and Dinsmore (2001) found the percent of emergent vegetation in wetland complexes in Iowa and the total area of wetland in the surrounding landscape to be important predictors of bird species richness. Likewise, Ratti et al. (2001) speculated that the higher avian density they observed in restored prairie wetlands was likely due to the presence of upland cover adjacent to restored sites, which provided superior habitat for upland nesting waterfowl and other birds compared with existing remaining wetlands, many of which were surrounded by active cropland. Whereas studies have shown the use of restored wetlands in the Prairie Pothole Region of the North American upper Midwest by waterfowl for migrating, breeding, and rearing young, wetland complexes providing a variety of wetland conditions are more beneficial than isolated restored basins (Knutsen and Euliss 2001).

Amphibians are particularly sensitive to landscape factors (Lehtinen et al. 1999, Guerry and Hunter 2002). Linkages between wetland habitats and adjacent uplands and the condition of those upland habitats are important aspects determining the value of wetland habitats for semi-aquatic amphibians (Semlitsch 1998). Midwestern landscapes that include a complex of habitat types, including wetlands, have

been shown to be beneficial to amphibians (Knutson et al. 1999). In agricultural ponds in Minnesota, Knutson et al. (2004) found amphibian species richness to be highest in smaller ponds with low nitrogen concentrations resulting from minimal livestock access. They concluded that small farm ponds, properly managed, may help sustain amphibian populations in landscapes that lack natural wetland habitats.



Recently restored wetland in Ohio enrolled in the Wetlands Reserve Program. (Photo by K. Schneider, USDA NRCS)

Wetland establishment activities are intended to put in place features that support development of wetland functions over time. Short-term and long-term changes in physical conditions over time result in shifts in habitat suitability for a wide variety of species.

For example, Braile and Dunning (2003) noted high shorebird use of a restored wetland complex in Indiana shortly after restoration—associated with an abundance of mudflats and open, shallow water habitats—and a dramatic decrease in shorebird use as the site became vegetated. Likewise, Wilson and Twedt (2005) noted the use of restored bottomland hardwood wetlands by forest-dwelling land birds as soon as trees established on the site grow tall enough to begin to provide the necessary habitat structure.

Practice Application Principles

Several key factors driving fish and wildlife response to wetland establishment practices are apparent within the knowledge base provided by the literature.

Wetland Size

In general, larger restored wetlands and wetland complexes have been shown to be associated with greater wildlife species richness (Hemesath and Dinsmore 1993, Guggisberg 1996). Waterfowl use has been shown to increase with wetland size (Vanrees-Siewert and Dinsmore 1993). However, small prairie wetlands have been shown to be extremely

important for migrating and breeding waterfowl (Krapu et al. 2000).

Wetland Age

Wildlife use of established wetlands is in part dictated by the amount of time since the physical restoration or creation action was taken. Whereas bird species richness has been shown to increase with wetland age (Vanrees-Siewert and Dinsmore 1993), wildlife response is highly species-specific. Shorebirds, wading birds, and some waterfowl species have been noted to heavily use mudflats and open water habitats in recently restored wetlands (White and Bayley 1999). Use of recently restored wetlands by shorebirds and other species associated with open areas generally declines with wetland age and emergent vegetation growth (Braile and Dunning 2003). In bottomland hardwood wetland restoration, use by species associated with early successional habitats declines as forest landbird use increases with wetland maturation (Twedt and Best 2004, Wilson and Twedt 2005).

Hydrologic and Topographic Features

The condition of habitats provided in established wetlands is greatly influenced by the water depth and periodicity as well as surface microtopography and other surface features. Although there has been limited effort expended on quantifying how various microtopographic features influence wildlife response in restored and created wetlands, evidence is emerging that indicates that restored wetlands with greater diversity of surface features, supporting a wider variety of water depths and vegetation, are associated with greater wildlife species richness (Tweedy et al. 2001).

Proximity to Other Wetland Habitats

Wetlands established in the vicinity of other wetland habitats typically have greater value for many wildlife species. Amphibian habitat value is particularly influenced by the availability of nearby wetlands (Lehtinen et al. 1999). Greater wildlife response has been observed in complexes of restored wetlands than in isolated basins (Reaves and Croteau-Hartman 1994, Beyersbergen et al. 2004).

Surrounding Landscape Features

Land use, vegetation type, and overall condition of upland habitats surrounding established wetlands typically has a direct affect on the value of these wetland habitats for many species. For example, restored prairie wetlands established in unfragmented prairie landscapes have greater value for wetland birds than those established in intensively managed agricultural landscapes (Naugle et al. 2000). The amount of wetland habitat within several km of examined prairie wetland sites has also been observed as a predictor of wetland bird species richness (Fairbain and Dinsmore 2001).

Regional Water Conditions

Regional water conditions can have a dramatic effect on the quality of wetland habitats, both natural and established (Austin 2002). Seasonal and long-term climate variation is of particular significance in prairie wetlands where cyclical drought and deluge patterns are common (Euliss et al. 2004).

Sources of Population Recolonization

Wetlands that are established in areas that are far removed or otherwise isolated from source populations for recolonization may be of lesser value to many species. This is particularly true for some aquatic invertebrates (Knutson and Euliss 2001) and amphibians (Lehtinen and Galatowitsch 2001) with limited ability to traverse significant distances across non-wetland habitats.

Maintenance and Management

Establishment of appropriate wetland hydrology and vegetation are important factors in determining fish and wildlife value. However, maintenance of established wetland conditions and management of water regime and vegetation are equally important. Whereas wetlands managed to enhance wildlife value have been shown to generate increased use by target species (Kaminski 2005), others that are not properly maintained limit restoration success (Hicks 2001).

Knowledge Gaps

Wetland establishment through restoration and creation actions has become a common practice in wetland management and regulatory activities (National Research Council 1992, 2001). While there has been considerable improvement in our understanding of the effectiveness of these activities and in our ability to effectively establish a suite of wetland functions through these actions, controversy remains regarding what should be considered successful wetland establishment (Malakoff 1998, Middleton 2001).

In many instances, it is difficult to directly discern the effects of specific wetland conservation practices on wildlife use of the affected areas from broader population changes or temporal shifts in landscape conditions (Naugle et al. 1999). For example, Fletcher and Koford (2003) found only two of six wetland-nesting bird species populations increased in response to restoration of wetland complexes in Iowa, likely due to the high variability among restored sites and years, or lag time in recolonization. They also recognized that temporal dynamics of bird populations can affect estimates of population change at individual wetland sites. Wide-ranging and highly mobile species such as waterbirds pose a particular challenge for resource managers, where the presence of numerous wetlands on the landscape is more likely to influence local habitat use of individual restored sites than the local habitat conditions in those sites (Haig et al. 1998).

These issues illustrate some of the challenges resource managers face in enumerating fish and wildlife response to wetland establishment and management practices. Numerous gaps in our understanding remain to be filled before a more complete picture may be assembled. Some of the more significant data gaps apparent in the literature include:

- Most of the studies conducted have focused on breeding birds. Much less is known about bird use of these habitats during migration, wintering, and other non-breeding periods.
- The paucity of studies on wildlife other than birds is apparent in the literature. Additional work is needed on general response of fish and other non-bird biota to wetland establishment practices during all life stages.
- The literature contains numerous studies indicating that many wildlife species, primarily wetland

habitat generalists, are able to exploit habitats made available through wetland establishment practices (Knutson and Euliss 2001). Much less is known about how wetland habitat specialists may be affected by these practices.

- The widespread practice of wetland restoration and creation is a relatively recent trend; most of these wetlands have been established within the last 20 years. Whereas age seems to be an important factor in dictating fish and wildlife habitat value, greater effort is needed to gain a better understanding of the long-term viability and condition of the habitats provided.

- There is great variety in the types of activities undertaken to restore and create wetland habitats and a wide variety of wetland types in various hydrogeographic settings that are established. It is difficult to generalize the findings among these diverse wetland habitats. Greater understanding is needed on the primary factors that influence wildlife value among the habitats established.

- The presence of invasive plants or animals can greatly influence the condition of wetland habitats. This is particularly the case in created wetlands where vegetation establishment is less predictable and invasive plants are more likely to become established in response to greater disturbance and challenges of establishing wetland vegetation (Snell-Rood and Cristol 2003). Additional study is needed to better understand how invasive and non-native species influence habitat use and suitability.

Conclusion

There are a number of studies that imply that restored wetlands provide wildlife habitat value similar to natural reference wetlands. Fewer studies are available describing wildlife response to created wetlands. Most studies focus on bird response to wetland restoration. These studies reveal that while wetland-associated birds respond positively to the habitats established, species composition and community structure are highly variable and depend on local wetland conditions and landscape factors. Many researchers conclude that wildlife species richness is expected to increase over time with the expected increase in vegetation complexity in most restored wetland sites. Long-term monitoring is necessary to gain

a better appreciation for how restored and created wetlands develop over time and how various groups of wildlife respond to the habitats provided. Long-term and cyclical weather patterns, regional population trends, management activities, and landscape and surrounding land use changes must be factored into these monitoring efforts.

Wetland conservation practices supported by USDA programs and technical assistance are tracked under broad categories of wetland establishment (Wetland Restoration and Wetland Creation) and management (Wetland Wildlife Habitat Management, Wetland Enhancement, Shallow Water Development and Management). A wide variety of activities and wetland types are established and managed through these practices. A better understanding of the diversity of these practices is needed in order to directly relate findings in the literature on wetland restoration and creation to USDA conservation practices. ■

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Effects of Conservation Practices on Aquatic Habitats and Fauna

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ABSTRACT A major goal of both state and federal agricultural and environmental agencies in the United States is sustainable management of watersheds where agriculture is a dominant land use. Because watershed processes and conditions directly and indirectly affect soil, water, air, plants, animals, and humans, USDA NRCS encourages a watershed approach to management of agricultural operations in the United States. This requires a suite of approaches or practices that address natural resource concerns in uplands and stream corridors. Land clearing, leveling, draining, tilling, fertilizing, and harvesting together create prolonged perturbations manifested in the ecological and physical conditions of streams and rivers. Regardless of the cause of a problem in a watershed, its effect on aquatic habitats and their biological communities is dramatic. Physical damage due to channelization, erosion, sedimentation, and altered hydrological regimes coupled with ecological damage due to excessive nutrients, pesticide contamination, and riparian clearing cumulatively diminish the quality of aquatic habitats and threaten their biological communities. In general, the primary goals for farmers and ranchers in agricultural watersheds are (a) control of non-point source pollutants such as nutrients, sediments, and pesticides, (b) adequate water supplies for crop and animal production, and (c) stream/river channel stability. As indicators of watershed conditions, aquatic species and their habitats play a pivotal role in how we manage watersheds, with the ultimate goal of sustaining water quality and ecological integrity. Conservation planning identifies resource concerns within watersheds and what practices should be implemented to address them. If such practices are applied according to USDA standards, habitats will benefit as will the species that inhabit them. This paper examines the effects of NRCS-defined conservation practices used as conservation measures for aquatic species and their habitats.

Rivers and streams historically have served as sources for human development. The Tigris, Euphrates, and Nile Rivers were “cradles of civilization” because of the resources they offered. Rivers and streams provided a seemingly endless supply of water, first for agricultural development and later for industrialization. As natural sculptors of landscapes, rivers and streams carved away mountains and uplands while annually renewing the fertility of croplands downstream. These valuable systems were not only conduits for water and sediments but also human settlement, trade, and transportation. Rivers were the first highways, capable of transporting tremendous quantities of both raw materials as well as finished products. However, human waste products also became a passenger on the world’s rivers (Knight et al. 1994).

While rivers and streams have great capacity to rapidly recover from anthropomorphic influences, this capacity is not without limits. Degradation of lotic systems worldwide is pervasive. While some rivers and streams of the United States are still biologically diverse, many species are imperiled (Williams et al. 1989, Williams et al. 1993, Ricciardi and Rasmussen 1999, Warren et al. 2000). The causes of these declines are numerous and cumulative, including habitat and water quality degradation associated with erosion and sedimentation, watershed development, deforestation and subsequent agricultural or urban development and other human activities (Lenat and Crawford 1994, Allan et al. 1997, Harding et al. 1998). Of all the large- to medium-sized rivers in the lower 48 states, only the Yellowstone River remains unregulated by dams or channelization (Gore 1985). According to the 1994 National Water Quality Inventory of 617,000 miles of rivers and streams, only 56 percent fully support their designated use of supplying drinking water, supporting fish and wildlife, providing recreation, and supporting agriculture (FISRWG, 1998). Simon and Rinaldi (2000) reported that in the loess area of the midwestern United States, thousands of miles of unstable stream channels are undergoing system-wide channel-adjustment processes as a result of 1) modifications to drainage basins dating back to the turn of the 20th century, including land-clearing and poor soil-conservation practices, which caused the filling of stream channels, and, consequently, 2) direct, human modifications to stream channels such as dredging and straighten-

ing to improve drainage conditions and reduce the frequency of out-of-bank flows.

River and stream corridors are dynamic ecosystems that function across different spatial scales over time. Most rivers interact at various times and locations with agricultural operations. River and stream ecosystems provide a number of landscape functions, including transport of materials such as sediments, large wood and storm runoff, transfer of energy, cycling of nutrients, and distribution or redistribution of plants and animals. Although agricultural watersheds are controlled and restricted by human manipulation, they depend on the same underlying processes and therefore they function in the same ecological framework as natural ecosystems. Agricultural watersheds are superficially simple in that crops are typically a monoculture grown in parallel rows, soils are homogeneously broken and mixed through tillage, and landscape grade has been uniformly smoothed. This apparent simplicity belies the complex interactions between soil, crops, beneficial and pest flora and fauna, agrochemicals, weather, and adjacent non-cultivated lands and receiving bodies of water. Because of the often close association of farming operations within river and stream ecosystems, agriculture has the opportunity to strongly influence whether aquatic ecosystems can effectively perform their myriad functions.

Conservation practices may improve or protect the ability of rivers and streams to function in a number of ways. Conservation practices, which may be either agronomic or physical measures, may prevent an agricultural operation from interfering with stream ecosystem function (such as reducing sediments in runoff or protecting stream banks from failing) or directly restore that function (such as improving stream habitat). Ecological response to watershed management practices may be detected in three major areas—stream and riparian/floodplain habitat, water quality and quantity, and biota. Due to the complexity of aquatic ecosystems, no single area will provide a true measure of ecological changes in a watershed. For example, changes in habitat may be immediately detectable, while biological response to perturbations may take longer to become evident. Although quicker to detect, habitat changes may or may not indicate an ecological problem. Moderately disturbed habitats are often the most productive and

have higher species diversities, which may or may not indicate good ecological conditions. In general, water quality is useful in detecting acute problems. Water quality monitoring can easily detect dissolved oxygen concentrations that fall below the threshold to support aquatic life; however, many species of aquatic life are adapted to survive short-term declines in water quality (Cooper and Knight 1990b).

Effects of Conservation Practices on River and Stream Biota

This paper compiles available literature that describes fish and wildlife response to USDA conservation practices applied directly or indirectly to river and stream systems. While USDA Farm Bill programs offer increasingly attractive financial incentives to farmers and ranchers for conservation of aquatic resources, the degree to which aquatic habitat restorative actions are implemented and monitored for effectiveness at local scales is challenging to report and evaluate. This is apparent by the poor rate at which completed restoration projects have been evaluated (Bernhardt et al. 2005). This lack of evaluation is a result of limited dollars allocated for such efforts. Monitoring designs are necessarily intricate and expensive to implement due to the ecologically complex nature of stream, river, floodplain, and upland processes. Stream project evaluations are more prevalent in the “gray literature” and case files of USDA field offices, some of which are referenced in this document.

The success of restoration actions targeted to improve habitats for aquatic species is also difficult to evaluate because effects can be manifested by physical, biological, and chemical responses at multiple scales and time periods of catchments and their biological communities (Minns et al. 1996, Lammert and Allan 1999, Fitzpatrick et al. 2001, Vondracek et al. 2005). Moreover, suites of practices installed either sporadically or strategically in a watershed will differentially influence the breadth and timing of response of stream or wetland species

and their physical habitats. Thus correlations between a specific practice and the ecological response of an organism or its habitat are not easily discerned. These limitations aside, recent studies that focus on the effects of agricultural practices on conservation of aquatic species and their habitats are beginning to be reported and offer insights into which of these are effective at arresting the decline in aquatic species in North America. In most cases, management practices that retain or improve connections among ecological processes and/or different aquatic habitats contribute to the quality of those habitats and the well-being of the aquatic species that inhabit them.

Management actions to address aquatic habitats and their species vary according to the overall conditions of the sites where they are employed. While site-specific actions may improve bank stability along a reach of stream, a suite of practices designed to minimize soil erosion, conserve vegetation along

Table 1. National Conservation Practice Standards Relevant to Aquatic Species and their Habitats

Practice Name	Practice Code
Channel Bank Vegetation	322
Clearing & Snagging	326
Dam, Diversion Dam	402/348
Fence/Use Exclusion	382/472
Filter Strip	393
Fish Passage	396
Fish Pond Management	399
Forest Stand Improvement	666
Grade Stabilization Structure	410
Grassed Waterway	412
Irrigation Water Management/Structure for Water Control	449/587
Nutrient Management	590
Pond	378
Prescribed Forestry	409
Prescribed Grazing	528
No-till Residue Management	329
Riparian Herbaceous Cover	390
Shallow Water Management for Wildlife	646
Streambank and Shoreline Protection	580
Stream Crossing	578
Stream Habitat Improvement and Management	395
Wetland Enhancement	659

streams, and maintain ecological processes over a broader landscape are likely to improve water quality and aquatic habitats not only at a site but also throughout a larger portion of the watershed.

While not all-inclusive, this work is an attempt to provide pertinent information currently available. Documented effects are grouped by NRCS defined conservation practices listed in Table 1. Many conservation practices either serve multiple purposes, or due to their design and location on the landscape, have benefits beyond their original design considerations. Use Exclusion, for example, may be recommended to prevent bank erosion resulting from animal trampling; however, water quality may also be improved when animal waste is prevented from entering a stream, thus providing a secondary benefit. Furthermore, the distinction between one practice and another may be subtle; for example, diversions, grade control structures and dams all incorporate structures to impound water to some degree, with consequent responses by aquatic species.

The following paragraphs summarize major findings in the literature regarding the documented effects of the major conservation practices affecting stream habitats and associated aquatic biota.

Channel Bank Vegetation

There are a number of conservation practices developed to improve streambank condition and function (i.e., stability, habitat for wildlife, filtering capacity, shading of stream), including riparian buffer practices (see below). When implemented in concert with stabilization measures and considerations for aquatic species, this practice indirectly benefits aquatic habitat conditions (Sedell and Beschta 1991, Sweeney 1993, Washington Department of Fish and Wildlife 2003). Bank vegetation provides additional roughness to dissipate energy along streambanks or lakeshores while improving habitat and water quality by providing shade and plant material to the stream. A study by Shields and Gray (1992) of the Sacramento River near Elkhorn, California, suggests that allowing woody shrubs and small trees to be planted on levees would provide environmental benefits and would enhance structural integrity without the hazards such as wind throwing associated with large trees.

Clearing and Snagging

Clearing river and stream channels of wood and wood debris reduces hydraulic resistance and thus contributes to lowering the risks of flood flows. Logs, limbs, branches, leaves, and other debris transported during flooding often become lodged against bridges, hydraulic structures, and vegetation, particularly in and near overbank areas (Dudley et al. 1998). This practice helps prevent accumulations of in-channel wood that can deflect flows toward streambanks, resulting in bank erosion. While these objectives are beneficial for maintaining stable banks and minimizing flooding, they also result in a homogeneous channel that lacks habitat complexity important to aquatic species. Large wood, woody debris, and leaf litter are essential sources of carbon for stream ecosystems (Malanson and Kupfer 1993). While wood and debris removal may reduce channel and bank erosion by reducing debris-induced scour, experimental removal of wood from a small, gravel-bed stream in a forested basin resulted in dramatic redistribution of bed sediment and changes in bed topography (Bilby 1984, Shields and Smith 1992, Smith et al. 1993, USDA Natural Resources Conservation Service 2001). Removal of woody debris changed the primary flow path, thereby altering the size and location of bars and pools and causing local bank erosion and channel widening (Shields and Nunnally 1984, Smith et al. 1993). In a study of coarse woody debris removal on streams damaged by the eruption of Mount St. Helens, Lisle (1995) found total debris removal from selected stream reaches caused additional scour and coarsening of the bed surface compared with segments with no or partial debris removal. Total wood debris removal caused pools to become shallower, and in segments of low sinuosity, decreased the frequency of major pools. Habitat complexity decreased after total debris removal, as indicated by a decrease in the standard deviation of residual depth and an increase in the size of substrate patches. Myers and Swanson (1996) also found that pool quantity and quality decreased on streams subjected to coarse woody debris removal.

The importance of in-stream large wood as a component of stream habitat in forested ecosystems is well-documented (Gregory et al. 2003). As such, the practice of clearing and snagging is not without controversy and should be used with serious consideration for aquatic species of concern.

Dam/Diversion Dam

It is estimated that more than 60 percent of the fresh-water flowing to the world's oceans is blocked by some 40,000 large dams (>15 meters high), and more than 800,000 smaller ones (Petts 1984). Negative effects of large and small dams on aquatic fauna relate to creating barriers to migration (Bramblett and White 2001 Morrow et al. 1998, Helfrich et al. 1999, Neraas and Spruell 2001, Zigler et al. 2004), which disrupt spawning and rearing of fish, modify population structure, and create slow water habitat unsuitable for many native stream/river species (Ligon et al. 1995, Brouder 2001, Marchetti and Moyle 2001, Dean et al. 2002, Schrank and Rahel 2004, Tiemann et al. 2004). Impoundment of rivers by dams has been implicated as one of the leading causes of native mussel declines (Williams et al. 1993). Small impoundments generated by dams are implicated in the demise of some native prairie fishes (Mammoliti 2002).

Of broader significance, dam construction and maintenance dramatically alter the hydrological regime of streams and rivers, which in turn affects riparian-floodplain processes, aquatic community dynamics and structure, flood-pulse regimes important to many native aquatic species, and geomorphic conditions of stream/river channels that contribute to the dynamic complexity of stream and riparian habitats (Rood and Mahoney 1990, Bergstedt and Bergersen 1997). As such, use of this conservation practice should take into account the effects of dams on watersheds as a whole, and more specifically the migratory needs of aquatic species. Solutions to the problems dams present to aquatic species include the construction of fish ladders or elevators, trapping and transporting fish around the dam, or removal of the dam (see section on Fish Passage). These features do not, however, mitigate the effects of dam construction on riverine processes.

Positive effects of dams on aquatic species include creation of lake habitats suitable for recreational angling, increased processing of nutrients and agrichemicals such as pesticides and trapping of sediments (Dendy 1974, Griffin 1979, Dendy and Cooper 1984, Dendy et al. 1984, Bowie and Mutchler 1986, Cooper and Knight 1990a, Cooper and Knight 1991). Additionally, dams constructed with low flow releases that may sustain instream flows in first-order tributary streams during dry periods of the year (Cullum and Cooper 2001).

As dams age, consideration must be given to the consequences of decommissioning dams to water quality and downstream ecology (Smith et al. 2000, Bednarek 2001, Doyle et al. 2003).



Restored stream channel in Montana. (Photo by K. Boyer, USDA NRCS)



Streambank stabilization with stream bars and riparian re-vegetation in Oregon. (Photo by K. Boyer, USDA NRCS)



Example of streambank erosion in Missouri. (Photo courtesy of USDA NRCS)

Fence/Use Exclusion

Use exclusion is most often employed to prevent livestock use from causing bank and channel erosion as they cross a stream or enter to drink. Myers and Swanson (1996) found that bank stability, defined as the lack of apparent bank erosion or deposition, decreased on streams where banks were grazed by livestock. Overhanging banks are important fish habitat, and grazing of banks was implicated in loss of fish habitat in western U.S. streams (Duff 1977, Marcuson 1977). Use exclusion has also been shown to improve water quality by preventing livestock wastes from contaminating streams (Line et al. 2000). Few studies have addressed direct effects of use exclusion methods on aquatic flora and fauna. Trout abundance was found to be higher in Sheep Creek, Colorado, after cattle were excluded (Stuber 1985). Benthic macroinvertebrates less tolerant of poor water quality were more abundant in streams with exclosures, although the study design did not rule out other factors that may have led to the same result (Rinne 1988). In New Zealand, the types of aquatic insects in small streams with exclosures were different from those without exclosures, where riparian vegetation damage resulted in decreased shading and increased bank erosion (Quinn et al. 1992). In other studies, riparian vegetation condition improved subsequent to fencing cattle out of previously damaged areas (Schulz and Leininger 1990, Kauffman et al. 2004).

Filter Strips

Filter strips are installed on cropland and pastures to minimize the amount of chemicals, nutrients, or sediments in runoff to surface waters such as streams. Studies have validated the effectiveness of filter strips in improving the quality of surface waters (Lenat 1984, Dillaha et al. 1989, Lim et al. 1998, Krutz et al. 2005). Care must be taken to design filter strips in concert with riparian areas to avoid development of concentrated flows (Schultz et al. 1995a).

Fish Passage

Dams, culverts, and other barriers present fish and other aquatic species with a wide range of challenges including blocking dispersal or migration, as well as

changes in flow rates, water velocity, depth of spawning habitat, water temperature, predator-prey relationships, and food supplies. Fish passage facilities have been used in the United States since the 1930s; however, extensive research on fish passage did not begin until the 1950s (Ebel 1985). Literature on fish passage structures ranges from studies of design criteria (Eicher 1982, Moffitt et al. 1982, White 1982, Bunt et al. 1999) to usage and efficiency (Downing et al. 2001). Successful designs take into consideration optimal velocities to accommodate fish swimming abilities, light conditions, placements of entrances and exits, and use of air jet sounds and lights to guide fish through the structures (Ebel 1985).

Additional passage research has examined the ability of riverine fishes to migrate through large impoundments (Trefethen and Sutherland 1968). Raleigh and Ebel (1968) found that mortality of juvenile salmonids significantly increased for fish passing through impounded rivers. While early fish passage research focused primarily on large riverine systems, Anderson and Bryant (1980) provide an annotated bibliography of fish passage associated with road crossings. In agricultural systems, installation of fish passage structures such as fish ladders or culverts, which simulate stream substrates and velocities, is important for reconnecting different types of habitats used by fish during their life history stages. Studies in the Pacific Northwest demonstrate the value of reconnecting migratory routes and their habitats for anadromous salmonids (Scully et al. 1990, Beamer et al. 1998, Pess et al. 1998). Simply maintaining physical connectivity between intermittent stream channels used as drainage ditches and main-stem rivers has been shown to increase the amount of winter habitat for native fish, benthic invertebrates, and amphibian species in the grass seed farms of the Willamette Valley of Oregon (Colvin 2006). Similarly, maintaining open drains on agricultural lands in Ontario provides fish habitat for fish assemblages identical to nearby streams (Stammler et al. in press).

Dam removal is a viable option, albeit not without controversy, for restoring riverine habitats and reconnecting different habitat types. In the Pacific Northwest and New England, where anadromous salmon, steelhead, lamprey, shad, and herring utilize all or part of entire river systems to complete their

life cycles, dam removal is often the focus of stream restoration projects. Inland fish communities also require well-connected habitats to pass between habitats that change seasonally or provide elements for specific life-history stages. Dam removal is a relatively new practice and thus the effects on downstream habitats have not yet been widely addressed. Potential problems with sediment transport, contaminated deposits, and interim water quality are of concern, as are the economic impacts. Sethi et al. (2004) found that while benefits of dam removal included fish passage and restoration of lotic habitats in a former millpond, the mussel community downstream of the project was impacted by sediments freed when the dam was breached. Kanehl et al. (1997) evaluated the removal of a low-head dam and determined that both stream habitat and desired fish assemblage were improved by the action. Stanley et al. (2002) detected no negative effect on aquatic macroinvertebrates as a result of dam removal.

Fish Pond Management

Ponds managed to raise fish for non-commercial uses provide aquatic habitat for aquatic insects, waterfowl, and possibly amphibians. The location of the pond dictates the precautions managers should take to protect receiving waters in the catchment from a potential introduction of an exotic species or fish disease, should the pond overflow or breach. Introductions of non-native fish species are a significant threat to the native aquatic biodiversity of watersheds (Fuller et al. 1999).

Forest Stand Improvement

This practice has applications in the management of riparian forest buffers. When the forestry objectives are to improve or maintain the number of trees available for recruitment to the stream channel for stream habitat, models and prescriptions are available to meet this objective (Berg 1995). For a review of specific riparian forest stand improvement considerations relevant to stream habitats, see Boyer et al. (2003).

Grade Stabilization Structure

This practice has been used for several decades to control the grade and head cutting in natural or artifi-

cial channels. Grade control structures may be designed to stop or minimize head cutting both within river and stream channels as well as at the edge of fields where gully formation is a concern. Grade stabilization structures typically consist of a low dam, weir or berm constructed of earth, stone riprap, corrugated metal, concrete, or treated lumber (Abt et al. 1991, Jones 1992, Becker and Foster 1993, Rice and Kadavy 1998). Additionally, rock chute channels are occasionally used as grade control, embankment overtopping, and energy flow dissipation structures (Ferro 2000). Water either passes over the structure and into an armored basin typically with an energy dissipation structure or into a pipe in front of the dam where it is discharged downstream. Grade stabilization structures modify in-channel flow regimes and thus the effects of these structures on stream species can be similar to those documented for low-grade dams (see above section on dams).

In degraded systems, pools associated with these structures have been compared with naturally occurring scour holes. Cooper and Knight (1987a) found that grade control pools supported a higher percentage of lentic game species than did natural scours. This was attributed to the more stable, self-cleaning nature of grade control pools. In habitat-limited streams such as those affected by channel incision and bank failure where depths are limited, grade control structures can provide stable pool habitat (Cooper and Knight 1987b, Knight and Cooper 1991). Shields et al. (2002) established minimum size criteria for habitat benefits.

Smiley et al. (1998b) documented fish use of habitat created both above and below field level grade control structures. These structures are designed to control gully formation where fields drain into deeply incised stream channels. Low dams and L-shaped pipes are constructed and installed along the top of the stream bank to divert water from field runoff through the pipe to the stream channel rather than over the bank. Depending upon their design and local conditions, field level grade control structures may be constructed either with or without small impoundments. These temporary or shallow pools of field level grade control structures have been shown to provide important transient aquatic habitats, particularly in stream reaches that have lost stream channel flood plain interactions due to

channel incision (Cooper et al. 1996a, Smiley et al. 1997, Smiley et al. 1998a). Knight and Cooper (1995) and Knight et al. (1997a) documented water quality improvements in larger field level control structure pools where water residence time was sufficient to allow sediment to deposit and nutrients and pesticides to be processed.

Grassed Waterway

As is the case with filter strips, grassed waterways are used to minimize the amount of sediments, chemicals, and nutrients from cropland and pastureland. Recent studies validate their efficacy (Fiener and Auerswald 2003), and indirect benefits to aquatic habitats and their species are likely. These include minimizing sediment delivery from surface water run-off to stream habitats and protecting water quality.

Pond

Farm ponds are usually constructed to provide water for livestock or for aquatic habitats. Livestock ponds in some areas of the country are referred to as dugouts and they are often constructed in the floodplain of stream channels or in the stream channels themselves. Recent studies evaluated the effects of these ponds or dugouts on native prairie fishes in South Dakota. Researchers determined that if dugouts were constructed out of the stream channel, but within the floodplain, they provided important off-channel refuge habitat for Topeka shiners (*Notropis topeka*) (Thomson et al. 2005).

Other studies in the Midwest have indicated that with proper management, farm ponds help sustain amphibian populations in landscapes where natural wetland habitat is rare and where livestock access to the pond is limited and no fish are planted in the pond (Knutson et al. 2003).

Prescribed Grazing

Grazing management regimes influence both upland and aquatic habitats. Recent studies demonstrate how grazing management can contribute to the ecological connections between riparian and aquatic habitats. Riparian vegetation structure influences the terrestrial insect community. By altering graz-

ing management regimes to favor vegetation where terrestrial insects thrive, fish benefit from seasonally important food sources derived from riparian zones. Grazing regimes that allow cattle to graze for only short durations increase terrestrial insect production. This has recently been shown to be strongly correlated to fish condition and survival on Wyoming ranchlands (Saunders 2006, Saunders and Fausch 2006).

Riparian Forest Buffer

Riparian areas play an important role in all landscapes, serving as ecotones or transitional habitats. Ecotones support a greater diversity of plants and animals because they bridge two different ecosystems. Hald (2002) assessed the impact of agricultural land use of the bordering neighbor fields on the botanical quality of the vegetation of stream border ecotones. While the importance of ecotones has been well documented in ecological research, little work has focused on the effects of field borders on riparian habitats and stream ecosystems, particularly in the United States. Riparian and floodplain forests are important components of stream corridor systems and their watersheds. Riparian forests are major sources of in-stream wood that is an important structural component of habitat for fish and other aquatic species (Bilby and Likens 1980, Angermeier and Karr 1984, Benke et al. 1985, Bilby and Ward 1991, Flebbe and Dolloff 1995, Beechie and Sibley 1997, Cederholm et al. 1997—reviewed in Boyer et al. 2003, Vesely and McComb 2002, Dolloff and Warren 2003, Zalewski et al. 2003, Shirley 2004). Effects of riparian forest buffers on water quality are well documented (Lowrance et al. 2000). Riparian forests protect stream banks from erosion, thereby reducing sediment loads (Neary et al. 1993, Sheridan et al. 1999), and help process nutrients (Lowrance et al. 1995, Hubbard and Lowrance 1997, Hubbard et al. 1998, Snyder et al. 1998, Meding et al. 2001) and pesticides (Hubbard and Lowrance 1994, Lowrance et al. 1997). Schultz et al. (1995b) and Schultz (1996) demonstrated how riparian buffer systems may be incorporated or integrated into cropping systems in such a way as to improve runoff water quality and improve fish and wildlife habitat concurrently.

Because of the complexity of the interactions between riparian forests and streams and rivers,

it is difficult at best to identify direct relationships between riparian forests and aquatic species. It is well documented that riparian ecotones are among the most biologically diverse habitats known. As discussed in other sections of this manuscript, riparian forest buffers affect river and stream ecosystems by providing shade, cover, bank stability, and allochthonous materials essential to system productivity (Wallace et al. 1997). Curry et al. (2002) showed that the thermal regimes in streambed substrates used by brook trout (*Salvelinus fontinalis*) were significantly impacted by harvest of riparian forest buffers. Oelbermann and Gordon (2000) documented the quantity and quality of autumnal litterfall into an agricultural stream that had undergone riparian forest restoration. Wider buffers provided litterfall with higher levels of essential nutrients. Kiffney et al. (2003) demonstrated the importance of riparian buffers in forest streams to periphyton and aquatic macroinvertebrate production. Kondolf and Curry (1984) and Robertson and Augspurger (1999) also demonstrated that geomorphic processes related to river planform promote spatially complex but predictable patterns of primary riparian forest succession. Studies in Minnesota further support the importance of riparian corridor conservation/restoration to aquatic species because it contributes to in-stream habitat and geomorphic features at multiple scales of catchments (Stauffer et al. 2000, Blann et al. 2002, Talmage et al. 2002).

Riparian Herbaceous Cover

Effects of riparian herbaceous cover on terrestrial wildlife and birds are well documented and covered in depth elsewhere (Anderson, et al. 1979, Rubino et al. 2002, Blank et al. 2003, and Crawford et al. 2004). Riparian herbaceous buffers tend to have indirect effects on aquatic organisms by affecting channel morphology and erosion control, and as a source of organic materials. Forestation of riparian areas has long been promoted to restore stream ecosystems degraded by agriculture in central North America. Although trees and shrubs in the riparian zone can provide many benefits to streams, grassy or herbaceous riparian vegetation can also provide benefits and may be more appropriate in some situations. Lyons et al. (2000) reviewed some of the positive and

negative implications of grassy versus wooded riparian zones and discussed potential management outcomes. When compared with wooded areas, grassy riparian areas result in stream reaches with different patterns of bank stability, erosion, channel morphology, cover for fish, terrestrial runoff, hydrology, water temperature, organic matter inputs, primary production, aquatic macroinvertebrates, and fish.

Shallow Water Management for Wildlife

Shallow water management for wildlife primarily affects upland game and waterfowl (Maul et al. 1997, Maul and Cooper 1998, 2000, Elphick and Oring 2003). Shallow water management such as that created by flash board risers may affect stream or river fauna indirectly by improving water quality (Verry 1985, Knight et al. 1997b) or providing refuge for riverine species during seasonally high flows (see Wetland Enhancement).

Streambank and Shoreline Protection

Stream banks and shorelines are valuable habitat features to fish and invertebrates (Newman 1956, Wickham 1967, Butler and Hawthorne 1968, Blades and Vincent 1969, Chapman and Bjornn 1969, Lewis 1969). For example, Hunt (1971) found a direct relationship between bank cover and the trout-carrying capacity of streams. Giger (1973) demonstrated that stream banks form shallow water refugia, allowing fish to rest in areas of lower water velocity.

In some regions of the United States, streambank erosion is the number one source of sediments in rivers and streams (Grissinger et al. 1981). Streambanks and shorelines may be protected by a number of methods including bank shaping, board fences, bank revetments, stone toe, bank paving, spur dikes or groins, and bendway weirs (Galeone 1977, Davidson-Arnott and Keizer 1982, Pennington et al. 1985, and Johnson 2003). Some methods employing living materials include the planting of dormant willow posts, branch packing, brush mattresses, coconut fiber roll, joint plantings, live cribwalls, live stake, live fascines or gabions, and stiff grasses while other methods use dead or dormant plant material such as root wads and tree revetments (Sherman 1989, Evans et al. 1992, Siefken 1992, Geyer et al. 2000, Shields et

al. 1995a, Shields et al. 2000b). An appendix of bank protection methods may be found in FISRWG (1998). Modest changes in design can turn bank erosion control measures into habitat improvement. Modification of existing structures with additional stone or wood structure may improve habitat or contribute to rehabilitation or restoration of habitat (Shields et al. 1992, Shields et al. 1993, Shields et al. 1995a, Shields et al. 1997, Shields et al. 2000a).

Effects of stream bank protection on fish and macroinvertebrates have been documented for some specific practices such as lateral stone paving, spur dikes, bendway weirs, and chevron weirs (Knight and Cooper 1991, Knight et al. 1997a, Shields et al. 2000b). Knight and Cooper (1991) reported that stone spur dikes provided better habitat as indicated by large and more species-diverse catches when compared with unprotected banks and banks armored with stone toe and stone paving. Often, a combination of hard structures such as stream barbs with revegetation of the streambanks provides protection while enhancing riparian processes. Loss of cropland due to streambank erosion has encouraged new interest in riparian management that includes replanting of herbaceous and woody riparian buffers, often coupled with in-stream rock or rock/wood barbs to deflect the flow away from raw banks. Preliminary investigations in western Oregon indicate this streambank stabilization practice encourages in-stream processes important to aquatic species, such as retention of detritus and large wood for fish cover and macroinvertebrate food sources (S. Gregory, Oregon State University, unpublished data).

Stream Crossings

Stream crossings can be designed to serve as grade control structures to prevent head cutting and reduce suspended bed sediments resulting from traffic. Logging operations are particularly damaging to stream channels without some consideration for specifically designed stream crossings. Most research on stream crossings addresses effects on water quality (Milauskas 1988, Grayson et al. 1993, Blinn et al. 1998, Aust et al. 2003). However, like dams or diversions, stream crossings may form barriers to fish movement. Gibson et al. (2005) found 53 percent of culverts posed problems to fish passage, due to poor

design or poor installation. Additionally, Miller et al. (1997) found that stream bed fine sediment levels were higher, basal area lower, and herbaceous cover higher in the immediate vicinity of some crossings simply due to the presence of the road and fill banks associated with crossings using gravel culverts. Myers and Swanson (1996) studied two Nevada streams and found that road crossings increased sedimentation.

Stream Habitat Improvement and Management

Modifying streams to improve habitat has been ongoing for decades (Alabaster 1985), albeit with numerous changes in philosophy. The U.S. Bureau of Fisheries (1935) reported the effects of adding rock-boulder deflectors to improve fish habitats as early as the mid 1930s. Effects of stream habitat improvements including effects on food-producing areas, velocity, substrate, depth, drift, spawning area, and cover are extensively reviewed by Wesche (1985). Methodologies may be found in Seehorn (1985, 1992), Hunter (1991) and Cowx and Welcomme (1998). While most research on stream habitat modification has focused on salmonids (Roni et al. 2002), Shields et al. (1995b), Shields et al. (1995c) and Cooper et al. (1996b) documented the effects of various in-stream modifications on fish and macroinvertebrates in unstable warmwater streams. In-stream structural improvements have met with some success in improving local fish habitats. In-stream structures placed in western Washington and Oregon streams revealed significantly higher densities of juvenile Coho salmon, (*Oncorhynchus kisutch*), steelhead, (*Oncorhynchus mykiss*) and cutthroat trout, (*Oncorhynchus clarki*) (Roni and Quinn 2001). While placement of in-stream log structures has shown to be successful in the Northwest (Abbe and Montgomery 1996, Thom 1997, Roper et al. 1998), reported failures in the southeastern United States indicate the re-introduction of large wood to drastically altered systems is often unsuccessful when placed in stream reaches unable to retain them (Shields et al. 2006).

River and stream food webs are dependent upon the interactions between aquatic, riparian, and terrestrial environments (Goulding 1980, Insaurrealde 1992). Organic materials such as leaf litter and large wood (Benke et al. 1985, Junk et al. 1989) are most often deposited in channels during floods; flood-

ing stimulates both detrital processing and primary production within inundated terrestrial components of the ecosystem (Bayley 1989, 1991). These dynamics in turn establish the energetic foundation supporting secondary production and ultimately the fish production potentials associated with the ecosystem. The extent and duration of flooding strongly influence fish production (Welcomme 1976, 1979, 1985, 1986, Goulding 1980) because fish utilize floodplains as spawning grounds, food sources, and refuges (Robinette and Knight 1981, Knight 1981, Risotto and Turner 1985). Thus habitat improvement designs that enable streams to re-connect with their floodplains are warranted.

Stream habitat improvement is at its pinnacle when it crosses into stream restoration. Restoration is a complex endeavor that in one sense turns ecological theory into an applied science (Culotta 1995, Wagner and Pluhar 1996, Dobson et al. 1997, Purkey and Wallender 2001). Because it can be defined rather broadly, it may include other practices such as bank protection, stream habitat improvement, and riparian zone practices. The National Research Council (1992) defined restoration as the re-establishment of the structure and function of ecosystems. Thus ecological restoration is the process of returning an ecosystem as closely as possible to pre-disturbance conditions and functions. Rehabilitation, which is related to restoration, is usually understood as returning some level of ecological function but not necessarily to some pre-disturbance condition (FISRWG 1998). River and stream restoration has been extensively researched and several definitive works are available (Gore 1985, Anderson 1995, Brooks and Shields 1996, FISRWG 1998).

Several case studies of stream restoration cover all aspects of the subject including planning, implementation, and evaluation (Bassett 1988, Anderson et al. 1993, Rinne 1994, Myers and Swanson 1996). While most research covers specific restoration practices or target organisms, Amoros (2001) and Ebersole et al. (1997) examined habitat and capacity diversity. Nunnally (1979) explored habitat restoration from a landscape perspective.

Structure for Water Control

Water control structures such as irrigation diversions can entrain or entrap fish and other aquatic species.

Keeping fish and water in streams is an objective of an increasing number of ranchers and farmers in the arid West and has triggered development of sophisticated fish screens for irrigation diversions (Zydlewski and Johnson 2002, McMichael et al. 2004).

Wetland Restoration and Enhancement

Floodplain wetlands play an important role in the life histories of many riverine fishes (Killgore and Baker 1996). As such, the practice of floodplain wetland restoration has great potential for improving habitats for aquatic species and the survival of declining species. The connections between floodplain wetlands and stream systems and other permanent water bodies has been shown to be a dominant factor influencing fish assemblages inhabiting floodplain wetlands (Baber et al. 2002). Floodplain inundation during high water flows provides riverine species access to floodplain wetlands and other off-channel habitats for spawning, nursery areas, and other life-history functions (Junk et al. 1989). Individual species' life-history adaptations to hydrologic regimes such as duration and timing of flooding and the geographic position of floodplain wetlands in relation to the channel typically dictate the response of river fish fauna to flooding (Pearsons et al. 1992, Snodgrass et al. 1996, King et al. 2003).

Lateral movement between river channels and floodplain habitats is an important component of many species' life history, particularly for juveniles, and these species are adapted to seek backwater and other habitats attached to stream channels as flood flows recede (Kwak 1988). Restored and created off-channel wetlands and ponds have been shown to provide habitat values for juvenile fishes similar to natural high-flow floodplain habitat (Richards et al. 1992).

Entrapment of individuals in off-channel habitats and irrigation ditches has been documented, and a variety of fish screens have been designed to minimize negative effects of irrigation water withdrawals (McMichael et al. 2004). Installation and active management of water control structures in constructed or restored wetlands have been shown to be effective in preventing entrapment, allowing fish to migrate out of floodplain wetlands entered during seasonal high flows (Swales and Levings 1989, Henning 2005).

Knowledge Gaps

A number of studies, discussed in this chapter, have addressed the conservation effects on fish and aquatic fauna of fish passage around dams and road crossings (culverts), and stream habitat improvement and management. In addition, there has been considerable research on the effects of riparian forest buffers and herbaceous cover on water quality. For all of these topics, however, the complexities of effects on fish and macroinvertebrates leave many questions unanswered and requiring additional research. Snagging and clearing is generally considered detrimental to aquatic fauna because of the important role large wood plays in providing habitat and carbon. However, removal of some material may prevent bank erosion and failure, thus reducing suspended sediment loads. Field borders are often too far removed to have a significant impact on aquatic fauna; however, additional research may be necessary to explore off-site impacts of these practices. Stream crossing, bank protection, and exclusions improve water quality and intuitively should have a positive impact on aquatic fauna; however, documentation remains a significant gap. Effects of bank or shoreline protection have focused primarily on cool water species. Shallow habitats such as those created with flash board risers provide valuable habitat for waterfowl, however, like field borders, they may be too far removed from the stream channel to significantly impact aquatic fauna other than through improvements in water quality. Cumulative effects of multiple practices, and the time scale at which effects of practices on aquatic communities can be demonstrated, have not been reported. The degrees to which aquatic habitat restorative actions are implemented and monitored for effectiveness at local scales are challenging to report and evaluate. This is apparent by the poor rate at which completed restoration projects have been evaluated (Bernhardt et al. 2005). This lack of evaluation is likely a result of limited dollars allocated for such efforts. Monitoring designs are necessarily intricate and expensive to implement due to the ecologically complex nature of stream, river, floodplain, and upland processes. Determining key indicators relevant to the appropriate time scale in the continuum of restorative actions is critical.

Conclusion

A considerable body of work exists on the effects of anthropogenic activities on river and stream ecosystems and much of this research may be linked to specific management practices. Historically, it appears that management practices were designed to affect a specific target such as sediment, pesticide or nutrient reduction, and which secondary ecological impacts or improvements were intuitively assumed to occur. Few research projects have been specifically designed and conducted to definitively relate practices to ecological effects. This review highlights some of the ancillary research that relates to specific practices; however, it also demonstrates the need for research that specifically documents the ecological impacts of management practices. ■

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Using Adaptive Management to Meet Conservation Goals

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ABSTRACT Natural resource professionals should know whether or not they are doing an effective job of managing natural resources. Their decision-making process should produce the kind of results desired by the public, elected officials, and their agencies' leadership. With billions of dollars spent each year on managing natural resources, accountability is more important than ever. Producing results is the key to success. Managers must have the necessary data to make enlightened decisions during program implementation—not just at the conclusion of a program. Adaptive management is described as an adapt-and-learn methodology as it pertains to implementing Farm Bill conservation practices. Four regional case studies describe how adaptive management is being applied by practicing fish and wildlife managers. Indicators were identified to monitor and evaluate contributions to fish and wildlife habitat for each of the case studies. Data collected at each stage of the studies were used to make mid-course adjustments that enabled leadership to improve or enhance ongoing management actions.

As a natural resource professional with a federal or state government or conservation non-governmental organization (NGO), how do you know that you are doing the best job of managing natural resources? You have a responsibility to inform your constituents about how well your programs are contributing to conservation goals and objectives. Sounds like common sense, but in today's world of tightening budgets, constant change, unpredictable political environments, and high expectations by the public, we often fail to demonstrate results. Decision-makers may want monitoring and evaluation of programs and use of adaptive management in program implementation, but they often allocate too few resources to make it happen.

Since both elected officials and the public are now focused on accountability, we have to produce results. If you haven't been asked to provide information on the effectiveness of your projects and programs, you soon will be. The key lies in having the necessary data both to make decisions and to communicate the information to your constituents. Adaptive management, including monitoring and evaluation, is critical to successful conservation. After reading this chapter, we hope that you will be inspired to integrate adaptive management into your decisions and management activities.

Billions of dollars are spent each year on managing our natural resources. As accountability becomes more important, we'll need to make better decisions not just on how we use those dollars, but also on helping the public understand how they benefit from the work of natural resource professionals. The responsibility lies with leadership and management to make good decisions. Those decisions should be based on the best science, and that science comes from research that should include a monitoring and evaluation component. Adaptive management enhances the quality of the data. With better information, better decisions can be made.

Adaptive Management and Monitoring/ Evaluation Basics

Adaptive management, focused on monitoring and evaluation, can help you improve your natural resource management decisions. This section answers the basic question on how these concepts apply to your work.

What Is Adaptive Management?

Adaptive management is a relatively new concept that has begun to gain popularity in the mainstream conservation community. Adaptive management incorporates research into conservation action. Specifically, adaptive management is the integration of design, management, and monitoring to systematically test assumptions in order to adapt and learn (Salafsky et al. 2001). Adaptive management is the process of hypothesizing how ecosystems work, monitoring results, comparing them with expectations and modifying management decisions to better achieve conservation objectives through improved understanding of ecological processes (Lancia et al. 1996).

An adaptive management approach deals with the uncertainty inherent in managing natural ecosystems by treating policies or practices as experiments. Below is a definition of the concept:

Adaptive management is an approach to natural resource policy that embodies a simple imperative: policies are experiments; learn from them. In order to live we use resources of the world, but we do not understand nature well enough to know how to live harmoniously within environmental limits. Adaptive management takes uncertainty seriously, treating human interventions in natural ecosystems as experimental probes. Its practitioners take special care with information. First, they are explicit about what they expect, so that they can design methods and apparatus to make measurements. Second, they collect and analyze information so that expectations can be compared with actuality. Finally, they transform comparison into learning—they correct errors, improve their imperfect understanding, and change action and plans. Linking science and human purpose, adaptive management serves as a compass for us to use in searching for a sustainable future (Lee 1993).

Adaptive management incorporates research into conservation action. In a conservation project context, adaptive management is about systematically trying different actions to achieve a desired outcome. It is not, however, a random trial-and-error process. Instead, adaptive management is a cycle that involves several specific steps:

START: Establish a clear and common purpose

STEP A: Design an explicit model of your system

STEP B: Develop a management plan that maximizes results and learning

STEP C: Develop a monitoring plan to test your assumptions

STEP D: Implement your management and monitoring plans

STEP E: Compare result to hypothesis

ITERATE: Use results to adapt and learn

Adaptive management encourages research and management to be conducted simultaneously to reduce uncertainty and improve management and ecological understanding. Administrators can benefit from funding sound management experiments because they can gauge the effectiveness of various management scenarios and can improve understanding of why a particular action succeeds or fails (Lancia et al. 1996).

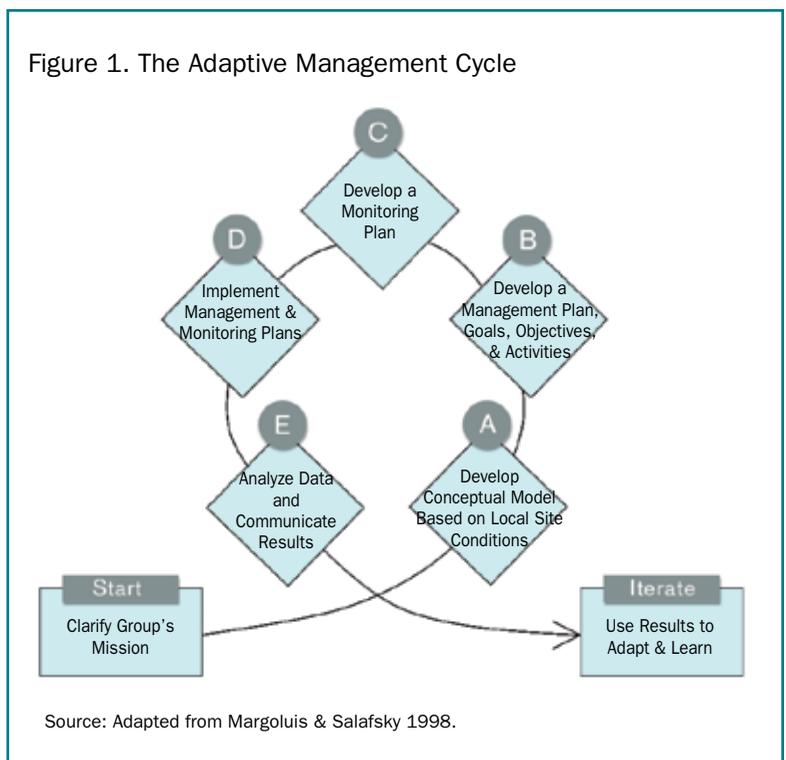
Why is Adaptive Management Important?

Adaptive management is a tool that enables natural resource agencies or organizations to evaluate how they are meeting their short-term and long-term natural resource goals. It allows us to answer basic questions: Is our management of the land working? Are our management actions having the desired effects? Are we contributing to the expansion of desirable/targeted habitats and subsequent increases in fish and wildlife?

In order to use these tools effectively, natural resource organizations will have to improve coordination and collaboration with each other. This collaboration will lead to the development of more comprehensive data and more efficient use of resources. Data sets can be expanded and shared. Funding can be leveraged. Key spatial and temporal indicators or benchmarks can be jointly developed that can be used to provide a better understanding of variation in performance over a range of conditions, supporting better analysis. Better decisions on future directions should result from the evaluations. The evaluation will also allow better communication with the public on the effectiveness of the programs.

Who will Benefit from Adaptive Management?

Three significant groups will benefit from adaptive management. Agencies and organizations will be



able to provide better information and a more efficient use of resources. The improved information will help the organizations in their outreach efforts with constituents and elected officials. These improvements could result in increases in budgets due to improved performance on accountability measures (indicators/benchmarks). The public benefits from an improved natural resource base at a net savings. Most importantly, natural resources will benefit. With better data, better decisions can be made. Corrections or adjustments in project and program design and implementation can be made early with more data and improved coordination that are part of adaptive management.

When and Where Is it Appropriate to Use Adaptive Management?

Adaptive management is appropriate for all programs. The following case studies illustrate the benefits. Coordination between federal, state, and conservation NGOs can build on successes. Regional applications can be better met via this process by minimizing replication. Partnering with others and sharing data can allow you to use scarce resources more efficiently.

How Can You Gain Efficiency with Adaptive Management?

Adaptive management is a better process for making better decisions. Better decisions should lead to better project implementation and results. Through more effective management and programs, you will be in a position to establish a record of success and communicate that success to both your constituents and your political leadership.

Better trend data enhances the science and better documents result. This allows for better accountability of programs. You may be able to clarify the cause and effect relationship between management actions taken and responses in habitat conditions and population enhancements.

So, if you successfully seek to employ both adaptive management and monitoring and evaluation, you will have to be able to answer these questions:

1. Do I do my monitoring and evaluation alone as an agency/organization?
2. Do I coordinate with other federal and state agencies and conservation NGOs in monitoring and evaluation activities?
3. Does the public understand my research goals?
4. Is there a relationship between information, management decisions, and monitoring and evaluation data and the changes in public attitudes toward the agency?
5. Is the monitoring information used adaptively and linked to agency policies?

Indicators/Benchmarks—How Do You Utilize Indicators to Evaluate Progress?

In order to evaluate projects and to make midstream corrections if necessary, you need to develop and institutionalize a system of tracking a set of indicators that monitors soil, water, air, and wildlife. These four indicators are interrelated. The information can be used to inform decision-makers of the status of each program or project.

Once indicators are identified, you'll be in a better position to answer the question: "Are fish and wildlife conditions stable, declining, or improving over time?" The answer can then be connected to policies, laws, and goals established by fish and wildlife agencies.

There should be a correlation between the agencies' goals and the indicators you chose. Remember, there are multiple audiences that you need to be working with so how you select the indicators often will determine their acceptance by targeted audiences. Since we are focusing on Farm Bill conservation programs, it would be appropriate to also look at the social and economic implications of indicators.

Case Studies

These case studies describe how adaptive management is being applied on the ground. The Thunder Basin of Eastern Wyoming case study and the Monitoring and Evaluation Plan for Habitat Buffers for Upland Birds (Northern Bobwhite Quail Buffers) case study apply adaptive management principles to specific Farm Bill conservation practices. The other case studies, The Tidelands of the Connecticut River case study and the Oregon Salmon/Watershed Project case study, while not Farm Bill-specific, describe projects that demonstrate how adaptive management can and should be applied to Farm Bill conservation practices.

Thunder Basin of Eastern Wyoming

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The Thunder Basin Grasslands Prairie Ecosystem Association (Association) is a non-profit organization established to provide private landowner leadership in developing a responsible, common sense, science-based approach to long-term management of private lands. Members in the Association consist of private property owners, primarily ranchers and energy production companies, within a designated 931,000-acre mixed-ownership landscape in eastern Wyoming. This landscape is recognized as one of the most ecologically significant grasslands in the United States.

The Association was formed in 1999 to address growing concerns about land management with particular interest in activities related to ranching, coal mining, coalbed methane development, and oil and gas production, and the influences of these activities on a number of wildlife species of concern. The Association's goal is to maintain responsible economic use of the land while demonstrating how effective

stewardship of natural resources can be provided through voluntary, privately led, collaborative efforts.

The Association recognized that each landowner working independently would not be as effective as a collaborative effort that considered the cumulative contributions of all lands within the landscape for ecological, economic, and social objectives. Consequently, the Association focused its efforts on developing an ecosystem management plan that addressed the habitat needs of all species of concern while balancing those needs with sustainable economic and social activities. The ecosystem management plan will provide the science-based information and integration needed to meet these objectives and will provide the basis for landowners to implement appropriate strategies.

The Association obtained a pooled Environmental Quality Incentives Program (EQIP) grant, with additional funds from the Wyoming Wildlife and Natural Resources Trust Fund and Wyoming Department of State Lands and Investments to restore and manage the declining habitat of a number of species of concern. These species included the long-billed curlew (*Numenius americanus*), upland sandpiper (*Bartramia longicauda*), chestnut-collared longspur (*Calcarius ornatus*), lark bunting (*Calamospiza melanocorys*), McCown's longspur (*Calcarius mccownii*), mountain plover (*Charadrius montanus*), short-eared owl (*Asio flammeus*), plains sharp-tailed grouse (*Tympanuchus phasianellus*), and swift fox (*Vulpes macrotis*). The Association is applying specific conservation treatments to 3,250 acres spread across 13 pastures in an active-adaptive management design. These treatments are designed to restore specific grassland conditions within the Thunder Basin that are in decline relative to the historical record.

Treatments were designed to produce specific plant communities across three different types of ecological sites. Three treatments will be used in combination: prescribed fire; inter-seeding with selected native species; and herbicides to control cheatgrass (*Bromus tectorum*), an exotic invader. In addition, several grazing regimes are being applied to pastures following these treatments. The Association expects to produce the desired plant community conditions through responses to the treatments. However, it is not well known how the plant communities will respond to the specific combination of practices.

Therefore, treatments will be replicated and monitored to provide information for adjustments to future treatments.

The Association selected three sets of pastures that averaged approximately 1,000 acres in size to replicate a desired range of ecological sites: five pastures were composed of primarily of clayey sites; five pastures were composed of primarily of loamy sites; and three pastures were dominated by saline conditions. The treatment portion of each pasture was left ungrazed prior to treatment to build up fuels for prescribed burning. In each pasture, prescribed burning is being applied to 240 acres in late summer/early fall. The burned areas will receive rangeland planting on two-thirds of the area (approximately 160 acres) as inter-seeding with a native seed mixture appropriate for that ecological site that emphasizes species known to have decreased in occurrence and dominance due to past grazing and other factors. Approximately 80 acres of each burn will remain unseeded to allow for the determination of the response of native plants to fire without the inter-seeding. In addition to seeding, half of each burned area (approximately 120 acres of each pasture) will be treated with an herbicide in fall to control cheatgrass.

The Association will apply varying levels of prescribed grazing as an additional treatment, with an entire pasture being the treatment unit. The treatments, with the varying levels of grazing, should result in different vegetation responses in both the treatment areas as well as areas of each pasture outside of the treatment area.

In each pasture, five exclosures of approximately one-half acre will be constructed, with one exclosure placed in the burned/planted/herbicide treated area, one exclosure in the burned/planted area, one in the burned/herbicide treated area, one in the burned-only area, and one in the untreated area of the pasture that is open to the specific grazing treatment. These exclosures will provide for an ungrazed control for each treatment combination in each pasture for monitoring purposes.

Monitoring, beginning in 2006 with pre-treatment measurements, will document the response of each pasture for vegetation conditions and wildlife use (plot sampling of bird use) to determine if the desired conditions for ecosystem diversity and associated habitat conditions for species of interest are

obtained. Monitoring for each treatment combination (Figure 2) will be continued for a number of years post-treatment to identify the vegetation and wildlife responses.

The pooled EQIP grant will support conservation needs at a landscape scale and will also improve rangeland productivity for each of the producers involved in the project. The treatments are designed to produce a significant acreage of desired conditions to meet the management objectives. By pooling the funds and using an adaptive management framework, the results will allow for an evaluation of the effectiveness of each practice and its combination applied across different ecological sites. This design will allow future treatment programs to focus efforts on those practices that produce the best results in this landscape and increase the effectiveness and efficiency of future Farm Bill funding. Monitoring associated

with the project will document the responses of the plant communities and selected wildlife populations.

Monitoring and Evaluation Plan for Habitat Buffers for Upland Birds (Northern Bobwhite Quail Buffers)

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<http://teamquail.tamu.edu/publications/HabitatBuffersforUplandBirdsCP33.pdf>

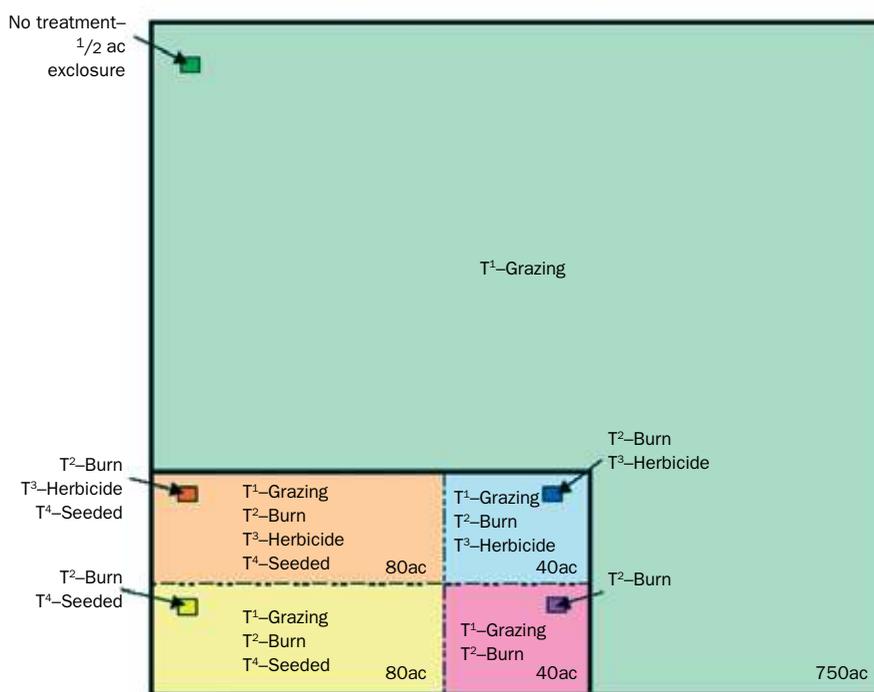
The U.S. Department of Agriculture's Farm Services Agency (FSA) Notice CRP 479 required development and implementation of a monitoring program as a precondition for states receiving their Habitat Buffers for Upland Birds (CP33) allocation. Specifically:

"A monitoring and evaluation plan must be developed in consultation with the state technical committee, including the U.S. Fish and Wildlife Service, State Fish and Game agencies, and other interested quail parties. The plan must provide the ability to establish baseline data on quail populations and estimate increasing quail populations and impact on other upland bird populations as a result of practice CP33, Habitat Buffers for Upland Birds, including the following:

- verification that suitable Northern Bobwhite quail cover is established
- verification that appropriate cover management practices are implemented on a timely basis
- states must control acreage within their allocation
- implementing a statewide sampling process that will provide reliable estimates of the number of quail per acre (or some other appropriate measure):
 - before practice CP33, Habitat Buffers for Upland Birds, is implemented (baseline)
 - resulting from the established CRP [Conservation Reserve Program] cover."

The research committee of the Southeast Quail Study Group (SEQSG)

Figure 2. Treatment applications within a schematic 1,000 acre pasture. Practices to be applied include prescribed burning, rangeland planting, pest management-chemical, prescribed grazing, and fencing. In combination, these practices are designed to provide restoration and management of declining habitats to restore desired ecosystem conditions as described by ecological site descriptions. Exclosures (1/2 acre in size) will be placed in each treatment area to monitor the effects of each treatment combination in the absence of livestock grazing.



developed a suggested national protocol for monitoring northern bobwhite (*Colinus virginianus*) response to CP33 that could be deployed through a combined effort of state offices of USDA-FSA/Natural Resource Conservation Service (NRCS) and state resource management agencies to: 1) provide statistically valid estimates of northern bobwhite density (or some other appropriate measure) on fields enrolled in CP33 at state, regional, and national levels and 2) provide a measure of the relative effect size of the CP33 practice. The protocol suggested a framework for monitoring breeding bobwhite and grassland songbirds using point transect methodology and fall bobwhite density using distance-based fall covey counts. The FSA national office, SEQSG, Southeastern Association of Fish and Wildlife Agencies (SEAFWA) directors, and Association of Fish and Wildlife Agencies (AFWA) have endorsed this protocol in concept. Furthermore, Southeast Partners in Flight (SEPIF) has expressed a commitment to assist in breeding season songbird monitoring and dovetail winter grassland bird monitoring on this sample of contracts. SEPIF has already provided much needed guidance regarding non-game bird monitoring in the CP33 monitoring protocol. A grassland songbird monitoring protocol also is available at <http://teamquail.tamu.edu/publications/HabitatBuffersforUplandBirdsCP33.pdf>.

The team initiated monitoring in 2006. AFWA is assisting states with carrying out the monitoring. Mississippi State University coordinated sample selection and sampling packet assembly, and is assisting with data analysis.

The Tidelands of the Connecticut River

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Paul Capotosto, Wetland Habitat and Mosquito Management (WHAMM) Program, Connecticut Department of Environmental Protection, N. Franklin, CT

The Tidelands of the Connecticut River Habitat Restoration Project is a cooperative effort to restore the ecologically unique habitat for a diverse group of organisms in the landscape where the Connecticut River meets Long Island Sound. The wetlands, ranging from fresh to saline, provide many ecosystem

services, including flood storage, upland buffering, water quality improvement, resource production, recreation, transportation, and aesthetics. Native biological diversity and the integrity and health of this system are threatened by an invasive species, the common reed [*Phragmites australis* (Cav.) Trin. Ex Steud.]. *Phragmites* has spread unchecked, achieving near exclusive dominance in many tidal marshes along less saline reaches [See Figure 3.] Management of the threat and recovery of the system requires *Phragmites* control.

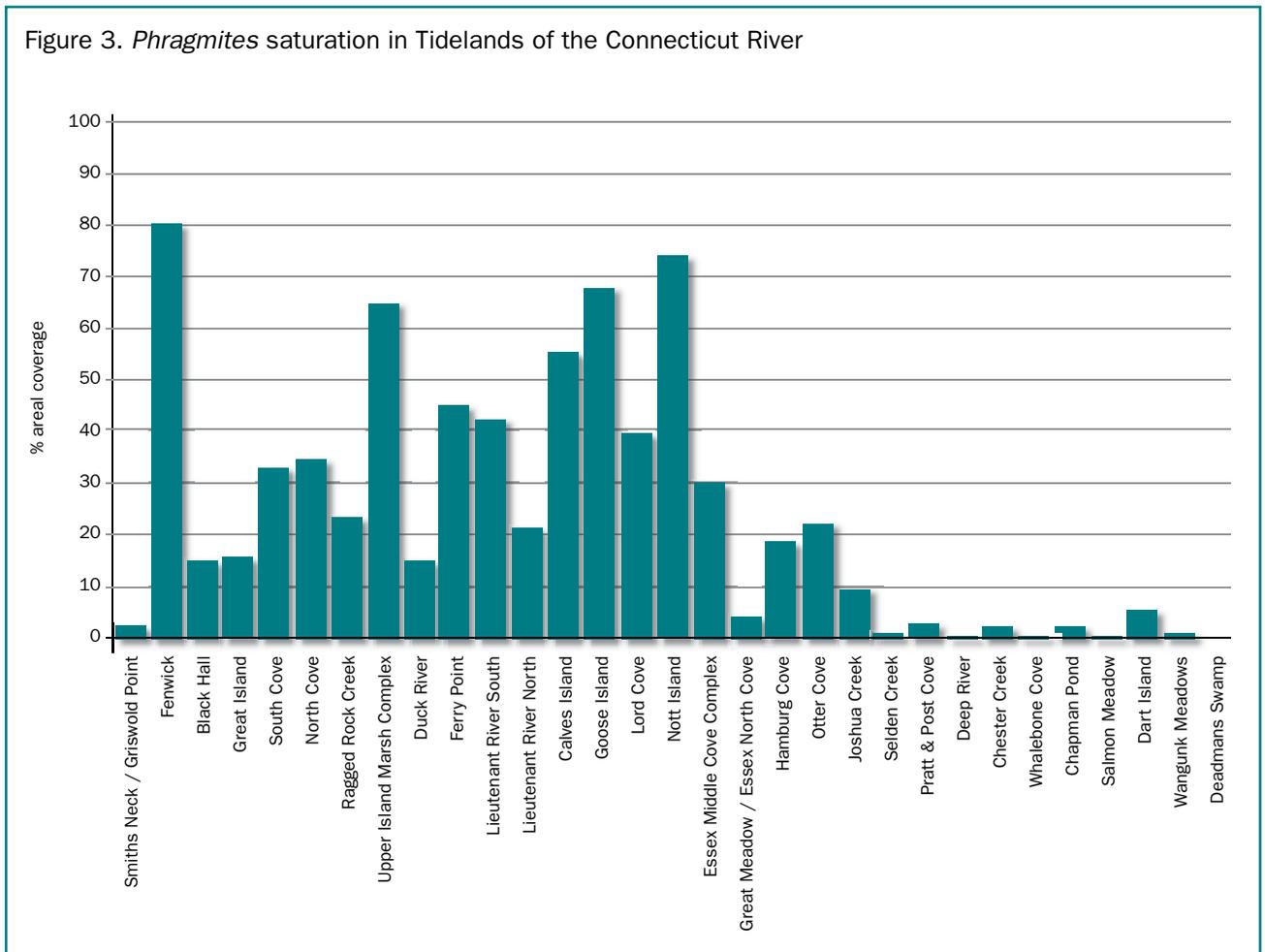
Numerous governmental and non-governmental organizations came together to create a partnership-based institutional structure, the Habitat Restoration Initiative Committee, and to establish a common vision of success. The partnership required a commitment of resources from modeling to on-the-ground restoration activities, monitoring, and outreach. Cooperation required clarification of restoration issues and needs, clear goals and objectives, a means for facilitating partnering, and a peer-review process. The assumption is that once *Phragmites* is controlled, the native vegetation will return. A key milestone was the development of the restoration project plan. The partnering structure facilitated participation and peer review. The effort formally began with work assessing biophysical and social realms, developing a conceptual model, and explicitly stating the assumptions underlying the goals of restoration and identifying social values.

The Habitat Restoration Initiative Committee decided to proceed sequentially so that, as restoration practices and treatments were completed at one site, new project sites were initiated. To date, three sites have been completed, one is in process, and six have been planned.

Regular monitoring of *Phragmites* and of rare plants was incorporated into the plan to determine the effectiveness of on-the-ground efforts and to identify areas of uncertainty that could affect the long-term success of the effort. Monitoring was necessary because *Phragmites* tends to re-invade and may require repeated control measures. Monitoring was also necessary to ensure that rare plant species were not adversely affected by the treatments.

Scientists and managers involved in the projects used the data from monitoring to re-evaluate previous steps and thereby establish a feedback loop on the effectiveness of treatments. Monitoring data were

Figure 3. *Phragmites* saturation in Tidelands of the Connecticut River



also used in performing outreach with the public to engage their interest and to continue the momentum toward achieving the project goals.

Representatives of the following groups partnered in monitoring—Related Activities Conservancy, Tidelands of the Connecticut River, Potopaug Gun Club, Connecticut Department of Environmental Protection, Migratory Bird Stamp Program of Connecticut, Stewart B. McKinney National Wildlife Refuge, Silvio O. Conte National Fish and Wildlife Refuge, the U.S. Fish and Wildlife Service, the Connecticut state office of NRCS, and the National Fish and Wildlife Foundation.

The Tidelands Plan employs a sequential landscape-scale management strategy as the most effective way to eradicate *Phragmites* and restore the biological integrity of the wetland systems. The sequential treatment of discrete sections was decided upon as a means for “learning from doing” and for improving the cost-effectiveness of efforts to restore Tidelands ecosystems. Data gathered

were geo-referenced into a Geographic Information System (GIS).

The adaptive management (AM) approach has led to changes in how the project is implemented and the longer-term effort to control *Phragmites* is conducted. Eradication efforts now focus on treating one section at a time, evaluating the effectiveness of the treatment from monitoring data and then making adjustments to the treatment practices at subsequent sites. This sequence of treatment, monitoring and evaluation, and adjustment is repeated at each subsequent site. The cost of treatment at each new site declines. The result has led to steady improvements of the control practices at each site with a concomitant increase in overall cost-effectiveness of the effort to eradicate *Phragmites* and restore the Tidewater ecosystem.

Lessons are still being learned on how to restore Tidelands ecosystems. Experience with AM up to now has shown that the assessments improve ecological understanding. Similarly, the partnering and out-

reach components of AM can help to communicate this understanding to scientists and managers and the general public, to redeem social value, and to foster an organizational culture of responsiveness.

Oregon Salmon/Watershed Project

**Stan Gregory, Oregon State University,
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The Oregon Plan for Salmon and Watersheds (Plan) is a cooperative effort to restore salmon runs, improve water quality, and achieve healthy watersheds and strong communities across the state. To contribute to this vision, the Plan relies on volunteers, creating a combination of voluntary and regulatory actions to conserve and restore watersheds and stocks of Pacific salmon. This cooperative paradigm drives the effort and remains the cornerstone to achieving success. This effort began with the creation of an implementation team that reviews and coordinates watershed restoration priorities. Members from federal, state, and local governments and tribal agencies have responsibility for activities contributing to watershed protection and restoration. A charter was endorsed by representatives of Oregon's state agencies who agreed to support the Plan.

With a formal infrastructure in place, the critical component of a monitoring and evaluation plan was established in March 1997. Its purpose was to 1) establish a structure and identify responsibilities for the development of monitoring teams, 2) coordinate and evaluate the monitoring efforts of the state agencies, federal agencies, and citizen groups and 3) annually review the progress of the monitoring program and explore the information emerging from the joint efforts. An independent multi-disciplinary science team provides an ongoing review of the scientific foundations of the Plan to the state. The monitoring program solidified the interagency commitments to the Plan, including coordination of public and private monitoring activities.

Representatives of the following groups participated in monitoring-related activities:

State: Departments of Agriculture, Environmental Quality, Fish and Wildlife, Forestry, State Lands, Transportation, and Water Resources; the Governor's Natural Resource Office; Oregon Watershed Enhancement Board; and legislative committees on natural resources.

Federal: National Marine Fisheries Service, U.S. Fish and Wildlife Service, U.S. Environmental Protection Agency, Forest Service and Bureau of Land Management.

Tribal: Columbia River Intertribal Fish Commission.

Partners: Oregon State University, Dept. of Land Conservation and Development, Watershed Councils, some soil and water conservation districts, landowner groups, environmental community and individuals.

Monitoring is a systematic collection of information used to assess the current conditions and trends in critical resources, ecological processes, or environmental conditions. Factors that affect the status and trends in salmon populations such as habitat conditions, water quality, watershed health, fisheries harvest, fish hatcheries, predation by birds and mammals, and ocean conditions are also monitored. The Plan's monitoring was designed to measure those factors needed to describe relationships between populations, habitats, restoration actions, natural processes, human activities, and management actions.

Because salmon require well-connected and intact habitats from headwaters of watersheds to ocean feeding grounds, the Plan endorses management with a landscape perspective as the most effective way to accomplish meaningful contributions to long-term salmon recovery in Oregon and the Pacific Northwest. The Plan's focus on habitat restoration at multiple scales across watersheds encourages voluntary land-use practices known to effectively improve not only local conditions but also watershed conditions critical to sustained salmon populations. The major land use and geographic areas considered in planning efforts included virtually all parts of Oregon with watersheds that drain into the Pacific Ocean. This area includes eastern Oregon drainages of the Columbia and Klamath basins.

Successful implementation of the Oregon Plan for Salmon and Watersheds depends on partnerships between state agencies and stakeholders in specific sub-basins and watersheds. Thus, in October 2002, a charter agreement for regional team coordinators was created to develop biennial work plans identifying key objectives, priorities and collaborative actions to support implementation of the Plan.

Coastal Coho Project and Assessment (coastal watersheds)

The Coastal Coho Assessment is the starting point for more effective future restoration investment, monitor-

ing, and adaptive management action. The objective of this effort is to assist in the recovery of one of the species of salmon that depends on Oregon watersheds. This assessment includes: viability analysis, population bottlenecks, evaluation of conservation efforts, monitoring, evaluating current threats, and lessons learned with a commitment to adaptive management.

Key conclusions of the assessment points can be found at www.mtjune.uoregon.edu/website/OWEB/Assessment. One of the key findings related to adaptive management included “maintaining a comprehensive monitoring program to allow adaptive management of conservation efforts as new information is gained.”

Actions Taken as a Result of Adaptive Management

In reviewing the factors for coho salmon decline, it was determined that changes were needed in the fishery harvest, hatchery management, and habitat protection and restoration in forest, agricultural, and urban lands. Major modifications of fishery harvest and hatchery management were implemented. Direct commercial harvest of coho salmon was totally eliminated from 1998 to 2002, followed by low rates of harvest to the present. Several hatcheries were closed and brood stock management and release practices have been modified to minimize the potential for adverse impact on coastal coho salmon. Now reduced numbers of hatchery coho salmon are released in only seven of 19 populations. This decrease in released fish and attention to locations of hatchery releases are intended to lessen genetic interactions, competition, and predation. Enhanced habitat management included protection, riparian restoration with extensive tree planting and fencing, in-stream improvements, development of additional forest management plans, improvement of culverts and bridges, confined animal feeding operation programs, total maximum daily load plans, and weed and invasive species control.

Lessons Learned

The assessments demonstrated Oregon’s responsiveness to new information and a willingness to implement needed changes in management programs. Examples included extensive restoration efforts of watershed councils, improved forest practice rules, improved water quality management plans by agriculture, reduc-

tions in fishery harvest rates, and redesign of hatchery management policies. These changes represent significant departure from historic practices, based on data and analysis. The state reviewed the status of coho salmon in 2005 and concluded that the coho salmon stocks of coastal Oregon were minimally viable. Based on the quantitative data developed collaboratively through the Oregon Plan for Salmon and Watersheds, the state recommended that the federal government remove coho salmon from the endangered species list. Both state and federal reviewers of the assessment noted that this assessment would not be possible in most states or for many resources and applauded the coordination of the monitoring program with the management actions of the Oregon Plan for Salmon and Watersheds.

A Reality Check—Adaptive Management: Myth and Reality

Jay Nicholas, Oregon Department of Fish and Wildlife, Salem, OR

The Oregon Department of Fish and Wildlife used adaptive management to assist in its decision-making process. Adaptive management is not just tweaking around the edges of natural resource issues; it implies significant course corrections. Under adaptive management, theoretically, monitoring provides data, data generates information, and agencies learn from the information and generate changes to management programs that are more effective in producing desired natural resource outcomes. In theory, adaptive management is just that simple. It is logical. It is timely.

Nonsense.

Here’s the reality. Adaptive management (change) can be achieved, but it can only be achieved slowly, in the proper time, and it requires some key ingredients. These are:

- leadership
- data
- patience
- public support

Of these four ingredients, data are possibly negotiable, the others are not. Leadership can come from elected officials, agency directors, charismatic individuals, or the public. Depending on the circumstances of the issues, leadership may be bold or timid.

Leadership may truly be out in front of the public or it may actually be following public sentiment. But someone, somewhere, has to lead, or create the appearance of leading the change.

Data should be a crucial ingredient in adaptive management but, in reality, it may or may not be. Sometimes, the data to support change in natural resource policy or programs are overwhelming and indisputable—yet it will be ignored, minimized, or disputed. This is where patience comes in. The facts may signal a need for change, but the time may not be right for the change to be implemented. Under these circumstances, those who see the need for change must be patient and not throw themselves unnecessarily or prematurely under locomotives that are not yet ready to be moved. Under these circumstances, one must wait for the leadership and public support to achieve sufficient momentum—then adaptive management can be implemented. At this moment, whatever data are available (from scant to extensive) may be cited as evidence for the needed change.

Examples? Over the course of my career I have seen extremely significant changes in management of fishery harvest and hatchery practices in Oregon. These changes were needed and valid well before they were actually implemented, by perhaps two or three decades. A shortage of data did not slow implementation of change; neither was change ultimately achieved solely on the strength of new data. Society and the leaders were not ready to accept or push for the change.

The Oregon Plan for Salmon and Watersheds is an example of timely, effective leadership that produced a new approach to natural resource management in Oregon. The Oregon Plan incorporates many recently changed management philosophies and practices, including fishery management, forestry management, water quality management, and restoration management. These changed philosophies and practices, together, reflect genuine examples of adaptive management and offer real hope for more effective and sustainable management of natural resources.

The time was right to initiate this plan when it was conceived and launched. Success was achieved because the agency was ready to accept adaptive management as a strategy to make better natural resource decisions. As a result, the effectiveness of conservation practices was enhanced. ■

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