



Effects of Conservation Practices on Aquatic Habitats and Fauna

Scott S. Knight, USDA ARS National Sedimentation Laboratory

PO Box 1157
Oxford, MS 38655
Email: sknight@ars.usda.gov

Kathryn L. Boyer, USDA NRCS West National Technology Support Center

1201 NE Lloyd Blvd, Suite 1000
Portland, OR 97232
Email: kathryn.boyer@por.usda.gov

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 up-land 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 barbs 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, Insaurralde 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 boarders, 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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