Reservoir
Fish Habitat Management

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Reservoir Fisheries Habitat Partnership

2017

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a project of the

RESERVOIR
FISHERIES
HABITAT
PARTNERSHIP

Suggested citation

Miranda, L.E. 2017. Reservoir fish habitat management. Lightning Press, Totowa, New Jersey. 306 pp.

ISBN 978-0-692-79872-0

Contents

List of Fish Taxa

Black bass	*Micropterus*
Black crappie	*Pomoxis nigromaculatus*
Bluegill	*Lepomis macrochirus*
Buffalos	*Ictiobus*
Catfishes	Ictaluridae
Common carp	*Cyprinus carpio*
Crappie	*Pomoxis*
Fathead minnow	*Pimephales promelas*
Freshwater drum	*Aplodinotus grunniens*
Gizzard shad	*Dorosoma cepedianum*
Grass carp	*Ctenopharyngodon idella*
Largemouth bass	*Micropterus salmoides*
Nile perch	*Lates niloticus*
Northern pike	*Esox lucius*
Perch	*Perca*
Pike	*Esox*
Rainbow trout	*Oncorhynchus mykiss*
Redbreast sunfish	*Lepomis auritus*
Redhorses	*Moxostoma*
Salmon	Salmonidae
Shad	*Dorosoma*
Smallmouth bass	*Micropterus dolomieu*
Smallmouth buffalo	*Ictiobus bubalus*
Spotfin shiner	*Cyprinella spiloptera*
Spotted bass	*Micropterus punctulatus*
Striped bass	*Morone saxatilis*
Sunfish	*Lepomis*
Temperate basses	*Morone*
Threadfin shad	*Dorosoma petenense*
Trout	Salmonidae
Walleye	*Sander vitreum*
Warmouth	*Lepomis gulosus*
White bass	*Morone chrysops*
White crappie	*Pomoxis annularis*
Whitetail shiner	*Cyprinella galactura*
Yellow perch	*Perca flavescens*

Preface

This document is organized into an introduction and 13 sections. Section 1 is based on a nationwide survey of reservoir habitats completed by the Reservoir Fisheries Habitat Partnership (RFHP). The RFHP canvassed agency professionals about various aspects of reservoir fish habitat degradation, and identified 12 comprehensive factors that captured most degradation. The sections developed in this document parallel these 12 factors to identify management practices that address the perceived degradation. Most sections are organized similarly: background information about the degradation factor is followed by a listing of alternative management practices. Deciding on appropriate management action can be complex because there are often several options for addressing degradation problems. Different options may have different cost, effectiveness, and consequences. A simplified decision model to choose among management options is suggested in section 12. Habitat management often goes far beyond the manpower available to agencies involved in reservoir habitat management, so suggestions on working with stakeholders are included in section 13. Ballpark costs of reservoir fish habitat management practices are given in the Appendix.

This project was supported by the Association of Fish and Wildlife Agencies' Multistate Conservation Grant Program through a grant to the RFHP.

Jeff Boxrucker, Coordinator of the RFHP, had substantial and meaningful input into various aspects of this document. Jeff wrote and administered the grant, constantly provided me new information to consider for inclusion, found reviewers, hounded reviewers, himself reviewed several drafts, and religiously stayed in touch to ensure that I stayed focused. In short, this document would not have started or been completed without Jeff's able guidance and administration.

Suggestions from reviewers were instrumental in refining the information presented. For reviewing one or more sections of the manuscript I am thankful to Michelle Balmer, Iowa Department of Natural Resources; Tim Birdsong, Texas Parks and Wildlife Department; Jeff Boxrucker, Reservoir Fisheries Habitat Partnership; Paul Brakhage, LakeTech Consulting; Chris Chizinski, University of Nebraska; Mike Colvin, Mississippi State University; Jeremy Crossland, U.S. Army Corps of Engineers; Dan Daugherty, Texas Parks and Wildlife Department; Brian Graeb, South Dakota State University; Reed Green, U.S. Geological Survey; Amberle Jones, Arizona Game and Fish Department; Rebecca Krogman, Iowa Department of Natural Resources; Kurt Kuklinski, Oklahoma Department of Wildlife Conservation; Tom Lang, Texas Parks and Wildlife Department; Chris Larson, Iowa Department of Natural Resources; Rick Ott, Texas Parks and Wildlife Department; Tim Patton, Southeastern Oklahoma State University; Mark Pegg, University of Nebraska; Kevin Pope, Nebraska Cooperative

Fish and Wildlife Research Unit; Mark Porath, Nebraska Game and Parks Commission; Janine Powell, U.S. Geological Survey; Hal Schramm, Mississippi Cooperative Fish and Wildlife Research Unit; David Weedman, Arizona Game and Fish Department; and Kevin Whalen, U.S. Geological Survey.

Various individuals contributed information included in this document. These include Shane Bush, Missouri Department of Conservation; Mike Colvin; Gene Gilliland, Oklahoma Department of Wildlife Conservation; Rebecca Krogman; Ron Marteney, Virginia Department of Game and Inland Fisheries; Jason Martin, Martin Ecosystems; Mike McGehee, Iowa Department of Natural Resources; Tom Mendenhall, U.S. Bureau of Land Management; Mike Mounce, Illinois Department of Natural Resources; Doug Nygren, Kansas Department of Wildlife, Parks, and Tourism; Rick Ott; Mark Porath; Michael Smart, U.S. Army Corps of Engineers; Dave Terre, Texas Parks and Wildlife Department; and Bobby Wilson, Tennessee Wildlife Resources Agency.

Eva Silverfine did the copy editing. Her attention to detail, meticulous comments, alternative options, and well-researched suggestions improved this document.

The Mississippi Cooperative Fish and Wildlife Research Unit is a cooperative effort of the Mississippi Department of Wildlife, Fisheries and Parks; Mississippi State University; U.S. Fish and Wildlife Service; U.S. Geological Survey; and the Wildlife Management Institute.

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Introduction

Thousands of large reservoirs have been constructed throughout the United States, nearly all during the twentieth century. The rate of large-reservoir construction has since declined almost to a halt as suitable construction sites already have been developed and as society's environmental sensitivities have shifted. While reservoirs are designed to address specific water-supply and water-control needs, they also provide habitat for fish, plants, and wildlife as well as extensive recreational opportunities for people. Reservoirs often are dismissed as unnatural, ephemeral, and ecologically disruptive, but they are a product of public policy and now prevalent features in our river basins. So long as society prizes the existence of reservoirs, they cannot be ignored if we are to conserve our aquatic biota effectively.

Much like natural lakes, reservoirs have traditionally been considered as standalone systems, separate from their surrounding watersheds and tributaries. Under this paradigm, fish habitat management approaches have focused on in-lake practices such as maintaining adequate water quality and enhancing structural habitat conditions. However, some of the properties of reservoirs are different from lakes. Reservoirs are usually not independent aquatic systems inasmuch as they have strong connections to the river upstream and downstream and, in serial reservoirs, to other reservoirs in the river basin. Reservoir systems exhibit longitudinal patterns both within and among reservoirs. Fish habitat management focused on the traditional in-lake scale may forego the potential benefits associated with considering reservoirs as part of the landscape. A scale broader than just the reservoir may provide the advantage of integrating abiotic, biotic, and socioeconomic characteristics active across the landscape.

Reservoirs are impounded rivers and, as such, have distinct habitat characteristics and aging patterns. Unlike natural lakes, reservoirs tend to have large watersheds and large tributaries because they were engineered to capture as much water as possible to serve diverse water storage purposes. This origin is manifested by relatively large inputs of inorganic and organic loads, nutrients, and contaminants. Depositional filling effectively has resulted in surface area, depth and volume reductions, backwater isolation, and habitat fragmentation. Artificial water-level fluctuations and wave action degrade riparian zones that were once uplands and are now unable to withstand continued flooding, resulting in erosion and ultimately homogenization of once-diverse nearshore habitats. Well-vegetated riparian zones and wetlands that provide key ecological services to natural lakes and the original river are missing in many reservoirs. Lack of woody habitat deposition, limited access to backwaters and wetlands, and unstable water levels that preclude establishment of native vegetation characterize barren littoral habitats in reservoirs.

Many fish habitat characteristics are expressed at the reservoir scale but are the upshot of broader scale factors operating outside the reservoir. Fish in reservoirs are shaped by conditions inside and outside the reservoir, and the relative importance of these internal and external factors often differs among reservoirs. Thus, fish in reservoirs may not respond to in-lake habitat improvements that fail to consider important elements of the entire watershed system. As a result, fish habitat managers may spend a large amount of resources with little benefit to fish.

Fish habitats show varying levels of degradation as reservoirs age. The intensity of habitat degradation differs among reservoirs depending on climate, physiography, land-use patterns, and a multiplicity of local conditions. The extent of degradation was investigated recently by the Reservoir Fisheries Habitat Partnership (RFHP). In a nationwide survey of about 1,300 large reservoirs, the RFHP study identified broad causes and levels of degradation.

Though the RFHP has quantified the prevalence of degradation, what is missing is practical guidance to address specific habitat degradation. Steered by the results of the nationwide survey, the RFHP developed this document to assist field biologists and administrators advance restoration and protection of fish habitats in reservoirs. The guidance provided in this document is deliberately general and nonprescriptive. Because this guidance is mostly unspecific it may not apply directly to local conditions. Instead, it is a starting point for development of site-specific prescriptions based on aspects such as goals, local site conditions, and socioeconomic circumstances. The aim of this document is to help identify "what to do," but local circumstances will likely dictate "how to do it."

Fish habitat management in reservoirs is in its early stages. Much of the methodology presented in this document has not been sufficiently tested, and its application is likely to have various amounts of uncertainty. The next stage in advancing reservoir habitat management is to organize a nationwide feedback system to assemble data on application successes, failures, and alternative actions. This system could involve standardized monitoring and reporting and use of the feedback to inform and refine the effectiveness of management activities.

Section 1

Sources of Reservoir Fish Habitat Problems

1.1 Introduction

Reservoirs have distinct habitat characteristics and degradation patterns due to their terrestrial origin and strong linkage to watersheds. Unlike natural lakes, reservoirs tend to have large watersheds and large tributaries because they were engineered to capture as much water as possible to serve flood control, water supply, navigation, or other purposes. This origin is manifested by relatively large inputs of inorganic and organic loads, nutrients, and contaminants. Depositional filling effectively results in surface area and volume reductions, habitat fragmentation, loss of depth, and associated changes in water quality. Unnatural water-level fluctuations interact with wave action to degrade shorelines that were once uplands and are unable to withstand continuous flooding, which promotes erosion and ultimately homogenization of once diverse littoral habitats. Well-established riparian zones and floodplain wetlands that provide key ecological services to natural lakes and the original river are mostly missing in reservoirs. Lack of woody debris deposition in the littoral zone, limited access to adjacent backwaters, and lack of seed banks and stable water levels to promote native aquatic vegetation characterize barren littoral habitats in many reservoirs, although in some cases there is excessive growth of nonnative aquatic vegetation. Because of their artificial origin reservoirs reveal unique fish habitat problems, exhibit senescing patterns not well correlated with chronological age, and can have major effects on habitats downstream from the dam.

1.2 A Nationwide Habitat Degradation Survey

Krogman and Miranda (2016) used an online survey to canvass resource managers about habitat degradation in reservoirs across the continental USA. The survey included about 75 questions regarding fish habitat in the reservoir and tailrace below the dam, recorded on a six-point degradation scale from 0 to 5: 0 = none, 1 = low, 2 = low to moderate, 3 = moderate, 4 = moderate to high, and 5 = high. The questions inquired about degradation to water quality and clarity, water fluctuations and flow through, submerged structure and vegetation, littoral and riparian zones, watershed uses, other habitat features of the reservoirs, and issues afflicting fish habitat in the tailrace. The survey also included questions about fish assemblages, fish populations, and fisheries. The fish data were collected on a five-point scale ranging from 1 to 5: 1 = low, 2 = below average, 3 = average, 4 = above average, and 5 = high. Respondents were fisheries biologists charged with managing fish in a specific reservoir, and the responses represented nearly 1,300 reservoirs 250 ac or more, approximately a 25% sample of U.S. reservoirs in that size class.

1.3 Geographical Patterns in Habitat Degradation

Factor analysis applied to responses to the nationwide habitat degradation survey conducted by Krogman and Miranda (2016) identified 12 major factors descriptive of habitat degradation (Table 1.1). The factors represented various degradations originating from within the reservoir or its watershed. Nationwide, degradation factors such as sedimentation, excessive nutrients, excessive mudflats and shallowness, water-level fluctuations, and limited submerged structure affected 10%–20% of reservoirs (Figure 1.1).

Intensity of degradation differed geographically among ecoregions, and ecoregions often were defined by a unique degradation or set of degradations (Figure

Table 1.1. Major reservoir habitat degradation factors in a sample of U.S. reservoirs ≥ 250 ac as estimated by Krogman and Miranda (2016). Each factor represents several observed variables scored by local fish biologists during an online survey.

Factor	Description
Point source pollution	Reservoirs with point source environmental problems stemming from watershed activities, thermal inputs, and contaminants
Nonpoint source pollution	Reservoirs with nonpoint source environmental problems stemming from broadly distributed watershed activities
Excessive nutrients	Reservoirs with excessive nutrient inputs originating from a broad area of the watershed
Algal blooms	Reservoirs with water-quality problems associated with variable oxygen, high temperature, and algal blooms
Sedimentation	Reservoirs with high suspended and deposited sediments and associated loss of habitat
Limited nutrients	Reservoirs that are often deep and oligotrophic or may be undergoing undesired oligotrophication
Mudflats and shallowness	Reservoirs that are excessively shallow particularly in the littoral zone and have extensive mudflats
Limited connectivity to adjacent habitats	Reservoirs with a lack or loss of connectivity to adjacent habitats, including backwaters and tributaries
Limited littoral structure	Reservoirs with insufficient physical structure and homogenized littoral habitats
Nuisance species	Reservoirs with aggressively expanding, typically non-native plant or animal species
Anomalous water regime	Reservoirs with frequent or poorly timed fluctuations or flushing
Large water fluctuations	Reservoirs with large or long-duration (or both) water-level fluctuations

1.2). For example, degradation due to large water-level fluctuation was most common in the drier areas of the contiguous USA, including the West (ecoregions WMT and XER; Figure 1.3) and large sections of the Great Plains (NPL and SPL). Water is scarcer in these areas and typically is collected for irrigation; water levels may fluctuate widely as incoming water is stored during the rainy season and released throughout the growing season. The water storage and allocation required to optimize water availability for irrigation often can conflict with the needs of fish by altering environmental cues or seasonal habitat availability (Ploskey 1986; Bunn and Arthington 2002; Dagel and Miranda 2012). Large water-level fluctuation was also the most important degradation in the Northeast (NAP); however, the extent of this degradation was relatively lower than in other regions.

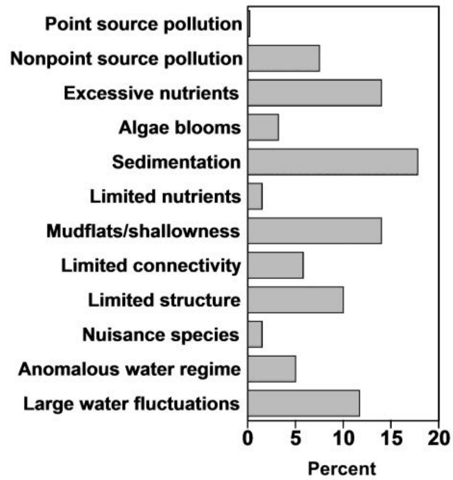


Figure 1.1. Percent of 1,299 U.S. reservoirs ≥ 250 ac scoring high (i.e., moderate-to-high degradation, and high degradation) in the survey for each of 12 habitat degradation factors defined in Table 1.1.

Unlike the West, most habitat degradation in the Midwest (TPL and UMW) and South (CPL) emphasized factors reflective of incoming water quality and land management in the reservoir's watershed rather than water storage. A reservoir's watershed is often the primary source of inputs into the reservoir, including nutrients, sediment, chemicals, and other pollutants (Kimmel and Groeger 1986; Kennedy and Walker 1990; Thornton 1990). Excessive nutrient input was the most important degradation in the Midwest, followed by sedimentation and nonpoint source pollution. Runoff from agricultural land contributes to all of these degradations; farm land covers more than 72% of Iowa (ISU 2013) and 60% of Illinois (IDNR 2013). In the South, sedimentation and mudflats and shallowness were the most important degradations, whereas excessive nutrient input was less important. Interestingly, this lowered importance of nutrients coincides with less land coverage by traditional agricultural land and greater land coverage by timber land. In the southeastern states of Florida, Georgia, North Carolina, South Carolina, and Virginia, about 60% of the land is forested, and over 95% of the forested land is considered timber land (Smith et al. 2009). This region includes about 12% agricultural land (USDA 2009). In the south central states of Alabama, Arkansas, Ken-tucky, Louisiana, Mississippi, Oklahoma, Tennessee, and Texas, about one-third of the land is forested, with over 90% considered timber land; this region also has about 20% agricultural land (USDA 2009). Commercial forestry

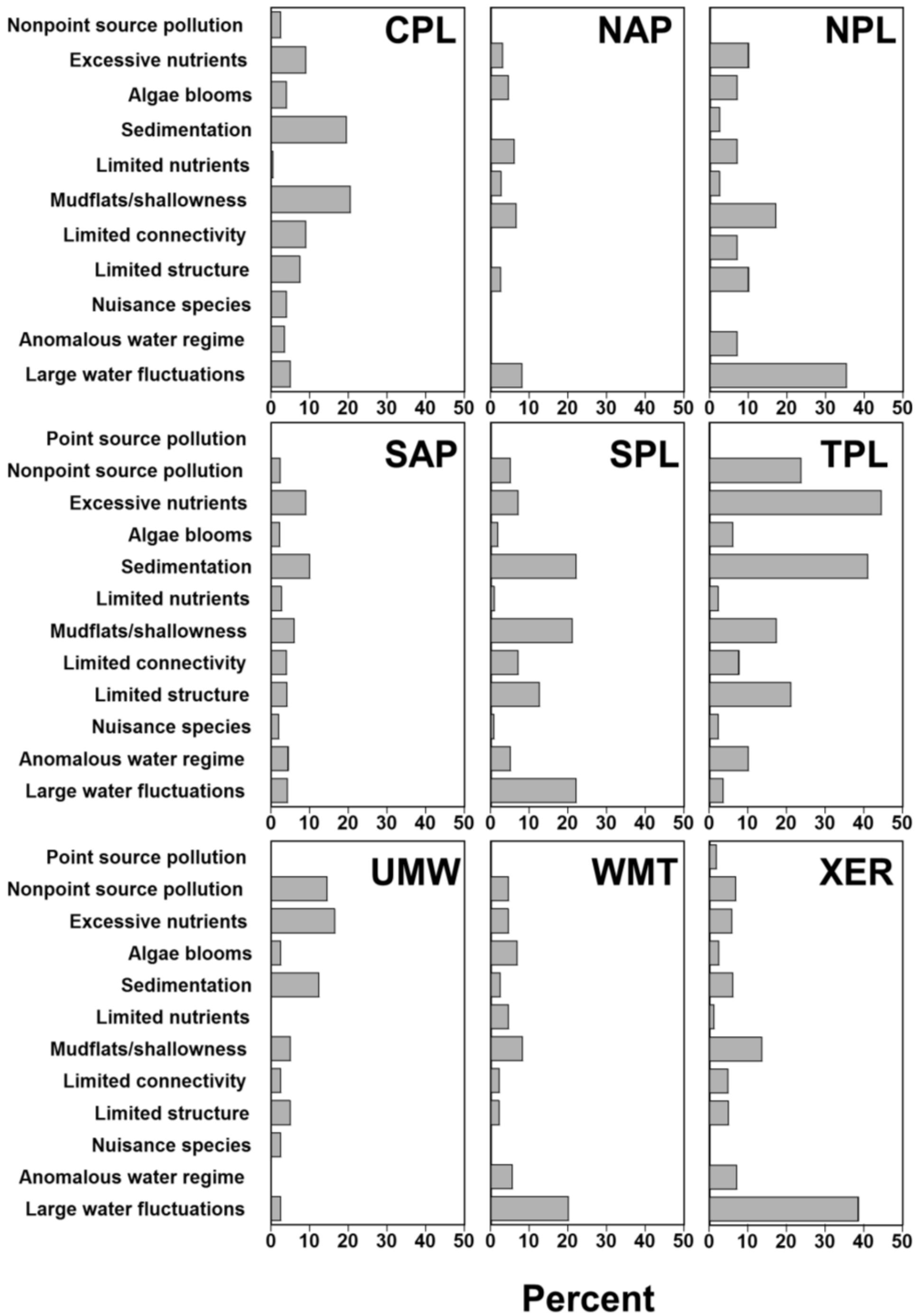


Figure 1.2. Percent of U.S. reservoirs ≥ 250 ac scoring high (i.e., moderate-to-high degradation, and high degradation) for each of 12 habitat degradation factors (Table 1.1) in the nine ecoregions identified in Figure 1.3.

practices such as roadbuilding and clear-cutting during harvest could export additional sediment and, to a lesser degree, nutrients to waterways directly (Ensign and Mallin 2001) or indirectly by altering streamflow (Troendle and Olsen 1994). Thus, reservoirs in the Midwest and South show faster rates of sedimentation and eutrophication than in other regions and hence faster functional aging as defined in section 1.4. The close ties among land use, eutrophication, and functional age were demonstrated effectively for Kansas reservoirs by Carney (2009).

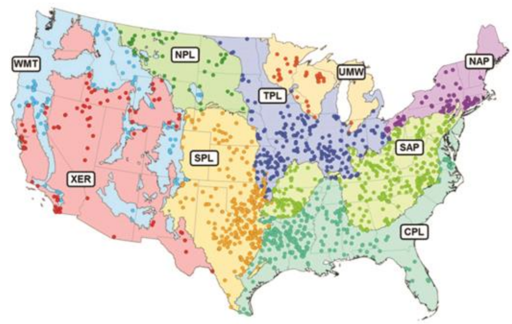


Figure 1.3. Ecoregions of the conterminous USA identifying position of reservoirs (points) of 1,299 reservoirs ≥ 250 ac included in the survey. Regions include Xeric (XER), Western Mountains (WMT), Northern Plains (NPL), Temperate Plains (TPL), Southern Plains (SPL), Upper Midwest (UMW), Coastal Plains (CPL), Southern Appalachian (SAP), and Northern Appalachian (NAP).

1.4 Reservoir Aging

The twentieth century was the golden age for construction of large reservoirs in the USA. Although some reservoirs were constructed in the nineteenth century, the rate of construction accelerated dramatically in the early 1900s, peaked near midcentury, and decreased to pre-1900s levels by the opening of the twenty-first century (Figure 1.4). Over the twentieth century unevenness in construction rates is apparent and linked to major events. For example, construction declined in 1915–1920 coinciding with World War I and then again in the early 1930s coinciding with the Great Depression. Construction accelerated to its fastest pace in the second half of the 1930s coinciding with the work programs developed to counteract the economic depression but declined again in the 1940s coinciding with World War II. Pace of construction picked up again in the 1950s to reach another peak in the late 1960s. Beginning in the early 1970s a steady decline in construction becomes apparent. The acceleration at the beginning of the twentieth century may be attributed to various engineering achievements (e.g., electrification, internal combustion engine, concrete design). The deceleration at the end of the century may be attributed to rising costs, depletion of suitable construction sites, and the strengthening of environmentalist views.

This roughly symmetric construction distribution has produced a counterpart distribution of ages. Examination of the age distribution reveals that as of 2016 the median age of U.S. reservoirs was 66 years, with 15^{th} and 85^{th} percentiles at 38 and 102 years, respectively. Similarly, the mean age was 71 years with ±1 standard deviation at 40 and 102 years. Projecting these distributions into the future, and assuming no or

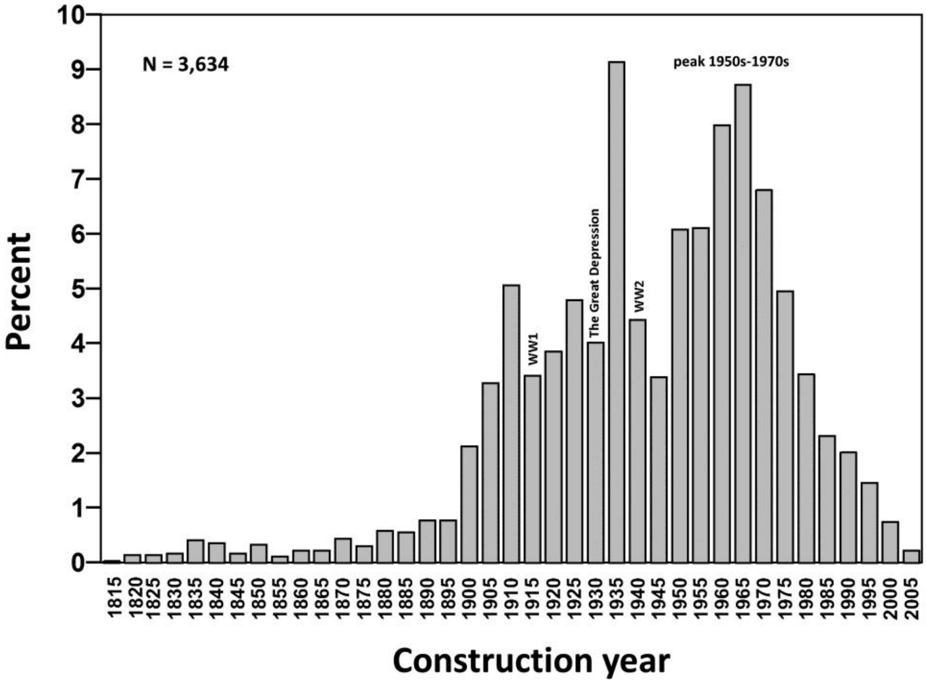


Figure 1.4. Distribution (percent by year) of reservoir construction in the USA, 1815–2010. N represents the sample size of reservoirs ≥ 250 ac. Selected historic events are identified within the plot (WW = World War).

few new reservoirs will be constructed, by the year 2050 the median age of U.S. reservoirs is expected to be about 100 years and the mean 105 years.

Gerontologists long have recognized that definitions of human age that focus exclusively on chronological age (years since birth) are incomplete because they are independent of human physiological and psychological factors (Baars and Visser 2007). Similarly, the rate at which reservoirs age may not be described best by chronological age. The rate of aging may depend on a diversity of attributes driven by climate and geography, watershed magnitude and composition, and reservoir hydrology and morphology. The crux of the problem with chronological age is that there are marked differences among humans and among reservoirs in the rate at which entities change over time. The implication of these differences is that chronological age and functional age (position along life span) may be related only weakly, and for many applications functional age may be more relevant than chronological age.

Reservoirs vary in their geographical distribution, physical characteristics, and operational scheme, potentially creating large variability in functional age. Reservoirs tend to have large watersheds and tributaries because they were engineered to capture as much water as possible to serve flood control, water supply, hydropower, or other purposes (Kennedy 1999). This unique hydrology can produce large input and retention of sediment and nutrients, although quantity may vary depending on climate, geology, and land cover. Thus, effects of inputs may differ depending on reservoir morphology. Depositional filling reduces depth and surface area and has been estimated to cause backwater isolation and habitat fragmentation (Patton and Lyday 2008). Wave action, coupled with unnatural water-level fluctuations dictated by operational goals, alter shorelines that were once uplands and are maladapted to continuous or seasonal flooding. Over time this promotes erosion and homogenization of once diverse littoral habitats (Allen and Tingle 1993). Well-established riparian zones and wetlands that provide key ecological services to natural lakes and the original river generally are limited to upper reaches near the entrance of tributaries but often degrade because of unnatural water-level fluctuations (Miranda et al. 2014). Lack of woody debris deposition in the littoral zone, limited access to backwaters and wetlands, and lack of seed banks and stable water levels to promote native aquatic plants characterize barren littoral habitats in many reservoirs (Miranda 2008). Woody materials flooded during impoundment disintegrate within a few decades (Agostinho et al. 1999). Inequalities in the manifestation of these and other key variables can reduce the correlation between chronological and functional age.

A limited number of published studies have included chronological age as a covariate in models designed to describe or predict reservoir biological characteristics, but chronological age has seldom been a reliable covariate. Jenkins and Morais (1971) examined various metrics descriptive of sportfishing effort and harvest and concluded that although, as expected, chronological age was inversely related to harvest, it accounted for less than 5% of the variability in harvest. Miranda and Durocher (1986) reported that growth of fish in reservoirs declined rapidly soon after impoundment, but subsequent reductions were minor, and Hendricks et al. (1995) reported that size of fish increased with reservoir age. In both of these studies, correlations with chronological age were unexpectedly low. Dolman (1990) reported that age did not help distinguish among distinct reservoir fish assemblages, and Carol et al. (2006) noted that chronological age surprisingly was not a primary factor governing nutrient levels or fish assemblages in reservoirs. These studies suggest that chronological age is not a good predictor of reservoir senescence (i.e., the process of growing old in a detrimental sense).

Miranda and Krogman (2015) evaluated functional age as an indicator of reservoir senescence. They used selected in-lake descriptors (Table 1.2) expected to change over the lifespan of a reservoir to construct a multimetric indicator of functional age. A scatterplot of functional age against chronological age showed no discernible pattern, and functional age scores were not correlated with chronological age. Reservoirs with the highest functional age scores generally occurred in the central USA

Table 1.2. Variables included in the online survey (Krogman and Miranda 2016) and used to index functional age. Data were collected with a six-point ordinal scale with 0 = no degradation, 1= low, 2 = low to moderate, 3 = moderate, 4 = moderate to high, and 5 = high degradation.

Metrics	Description
Mudflats	Seasonally flooded and exposed expansive soft sediments; unvegetated unless exposed for many months
Low connectivity to tributaries because of sediment	Sedimentation has decreased connectivity to tributaries during low flows, acting as a barrier to fish movement
Insufficient structural habitat	Lacking or deficient in structure such as large woody debris, gravel substrates, or diverse bottom relief
Excessive nutrients	Excessive nutrients, primarily nitrogen or phosphorous, which may result in excessive primary productivity and reduced water quality
Harmful algal blooms	Frequent occurrence of algal blooms that may be toxic to aquatic ecosystems or inhibit public enjoyment
Sedimentation	Settling of suspended sediments, which over time may reduce depth and homogenize substrates
Shore erosion	Removal of soil and terrestrial vegetation from the land–water interface, caused by weathering of banks or adjacent land slopes by water, ice, wind, or other factors
Loss of cove habitat resulting from depositional filling	Sedimentation has changed cove habitat including area reduction, isolation, and fragmentation and establishment of terrestrial vegetation in newly deposited land
Shore homogenization	Reduction of the shore's original habitat diversity by erosion or other processes
Homogenization of littoral substrates	Reduction of the substrate's original diversity by erosion and sedimentation

from North Dakota to Texas and in agricultural regions of the Midwest. Thirteen reservoirs shared the lowest possible functional age scores; these reservoirs were mostly at high elevations in the Rocky Mountains and Appalachian Mountains.

Functional age was highly variable relative to reservoir depth and watershed agriculture (Figure 1.5). Nevertheless, functional age did show a decreasing trend relative to reservoir mean depth. This pattern suggests that depth limited the maximum functional age scores in deep lakes, and although higher scores could be attained in shallow lakes, often other mitigating circumstances prevented shallow lakes from reaching high functional age. Conversely, functional age showed an increasing trend relative to extent of cultivated land in the watershed (Figure 1.5). This pattern suggests that watersheds with high levels of cultivated land almost always tend to have a high functional age and watersheds with low levels of cultivated land tend to have lower functional ages, although the latter sometimes they may have high functional age resulting from something other than the effects of a cultivated watershed. Other reservoir attributes likely temper functional age, but additional research is needed.

According to Miranda and Krogman (2015) factors representing fishing quality, fish size, fish recruitment, and fish mortality were related to functional age in various fashions. Fishing quality decreased relative to functional age and mortality increased. Size and recruitment showed hump-shaped patterns relative to functional age (illustrated in Miranda and Krogman 2015), suggesting they were optimized at intermediate levels of functional age.

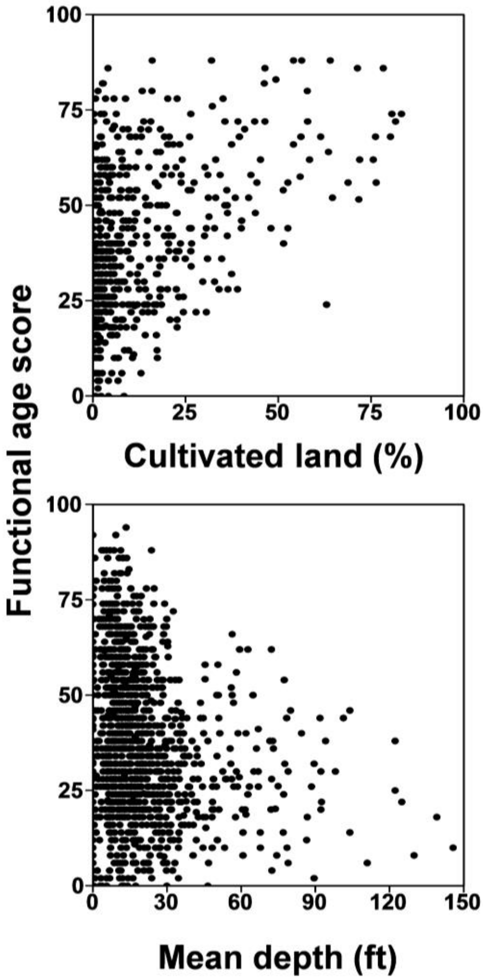


Figure 1.5. Functional age in relation to percent of cultivated land in the reservoir watershed (upper panel) and to mean depth of the reservoir (lower panel). Reservoirs in watersheds with high levels of cultivated land almost always have high functional age, whereas reservoirs with low levels of cultivated land can have low functional age, although the latter sometimes may have high functional age resulting from something other than cultivated land. Depth limits the maximum attainable functional age scores. In shallow reservoirs functional age scores can be much higher, although other reservoir attributes may prevent reaching high functional age.

The concept of functional age has advantages. Combining multiple metric scores to assemble an indicator of senescence presents the possibility for management intervention from multiple angles. If it is determined that a reservoir is functionally aging at an accelerated rate, action may be taken to remedy the conditions contributing most to functional age. Intervention to reduce scores of selected metrics potentially can reduce the rate of senescence and increase the life expectancy of the reservoir. This leads to the intriguing implication that steps can be taken to reduce functional age and actually make the reservoir grow younger (Miranda and Krogman 2015). The goal of habitat rehabilitation often is to alter the trajectory of the aging process such that the duration of a desired state is prolonged (Pegg et al. 2015). Slowing down the rate at which functional age increases, or even reversing functional age after decades of substandard watershed management practices, is challenging. Most of the information presented in the following sections is intended to facilitate management of functional age.

1.5 Downstream Concerns

A tailwater is the reach of a stream immediately below a dam that is hydrologically, physicochemically, and biologically altered by the presence or operation of the dam or both (Figure 1.6). A tailwater may be short or long, often persisting downstream to the confluence of a sizeable unregulated water source. The extent of stream alteration is related to the purpose of the reservoir, the design and depth of outlet structures (Walburg et al. 1981; Bednarek and Hart 2005), and the volume and schedule of water releases (Jager and Smith 2008). Although habitat management in the tailwater is beyond the scope of this document, sections 4 through 7 address issues that directly influence water releases into the tailwater.

Hydrologic alterations include changes to the natural timing and amount of discharge. Seasonal flows in tailwaters differ from natural flows and generally exhibit less temporal fluctuation (Johnson and Harp 2005). In regulated tailwater systems, flows outside a defined channel occur rarely and only during major floods. In some streams, particularly in the desert southwest, water may be present in a stream channel year-round below a dam, whereas the stream flowed ephemerally or intermittently before the dam was built (Sabo et al. 2012). Daily changes in flow resulting from peaking hydropower affect biota directly by stranding or indirectly by varying depth, temperature, and velocity (Gore et al. 1989; Nagrodski et al. 2012).

Physicochemical changes include shifts in various water-quality characteristics (Cushman 1985; Ashby et al. 1998; Olden and Naiman 2010). Turbidity in streams and rivers below impoundments is reduced because the reservoir above the dam acts as a settling basin for fine sediment. Water temperatures in the tailwater, and subsequently the fish community, may be changed relative to naturally occurring thermal regimes depending on the depth of water releases from the reservoir. Variables such as dissolved oxygen, pH, nutrients, and trace

Figure 1.6. Tailrace area immediately below the Nolin Lake dam, Kentucky. Photo credit: U.S. Army Corps of Engineers, Louisville District.

metal concentrations also may be affected by changes occurring within the reservoir and, in turn, may affect water quality in the tailwater reach.

Miranda and Krogman (2014) estimated the percentage of reservoirs with viable tailwaters in the USA. Viable tailwaters were those with sufficient flow to support a fish assemblage throughout the year. Overall, 42% of the sample reservoirs had viable tailwaters and, in general, the presence of a viable tailwater was related to the magnitude of the hydraulic system. Viable tailwaters commonly were associated with large, deep, high-storage-capacity reservoirs, whose basin had large catchments and substantial streamflows.

The survey also revealed that, in general, regions in the western USA tend to have longer tailwaters. Although the variability among tailwaters is high, and there are a large numbers of variables that interact to determine tailwater length, this effect is driven partly by reservoir characteristics. Tailwaters tend to be longer in large water-supply reservoirs with high storage capacity, which occur most commonly in the West. Length also was related directly to ratings assigned to degradations and differed among ecoregions. The principal stressors driving tailwater lengths were associated with flow (minimum flow, flow fluctuation, flow timing), fluctuating depth, bed scour, shore erosion, inadequate temperature, and overabundance of plants. Interestingly, variables related to gases and nutrients were not well associated with length of the affected reach. Thus, increasing the intensity of stress for flow-related variables can lead to affected reaches that are longer, but this effect may be weaker for chemical characteristics.

Major sources of environmental stress in viable tailwaters represent mostly issues associated with flow (Table 1.3). Flow is a major determinant of physical habitat in streams, which in turn is a major determinant of biotic composition (Bunn and Arthington 2002). Low base flows are pervasive below western reservoirs, especially in the Southwest. Flow changes and timing of flows, however, are important in most regions throughout the USA. Other issues, although occasionally important at the local level, afflict less than 20% of tailwaters. Base flow is one of the most widely studied aspects of regulated streams (Smakhtin 2001). During the dry season, minimum flows are often maintained in tailwaters to allow survival of biota, but the habitat they produce is usually of low quality and quantity (Walburg et al. 1981). During the wet season, storage reservoirs impound winter and spring runoff and as a consequence reduce tailwater flows. Below hydropower dams, large diel flow fluctuations can have destructive influences on the physical environment (Cushman 1985). The extreme variation in flow scours the tailwater, and high water velocities during power generation cause streambed instability, bank instability, and habitat degradation (Olden and Naiman 2010).

Table 1.3. Percent of viable tailwaters rated as moderate-to-high or high degradation with respect to 15 degradation variables by ecoregion and all ecoregions combined (Miranda and Krogman 2014). Percentages ≥ 20 are bolded. Values in parentheses are sample sizes. Ecoregion acronyms correspond to those in Figure 1.3.

Variables	WMT (60)	XER (24)	NPL (21)	SPL (61)	UMW (33)	TPL (105)	CPL (100)	SAP (130)	NAP (21)	All (555)
Erosion	10	0	19	17	3	14	11	0	8	9
Scour	13	4	19	11	3	18	10	0	10	10
Fluctuating water levels	10	**21**	14	11	12	15	12	11	12	13
Low base flow	**21**	**29**	**43**	**39**	12	16	17	14	16	**22**
Fluctuating flows	**25**	**29**	**38**	**34**	15	**32**	**25**	14	**22**	**25**
Untimely flows	**28**	**21**	**43**	**22**	15	**27**	**21**	4	18	**22**
Temperature	5	8	0	8	0	4	3	**21**	7	6
Dissolved oxygen	3	4	0	13	0	6	7	0	7	4
Other dissolved gases	3	8	0	0	0	2	2	0	1	2
Water chemistry	3	13	0	6	6	8	7	4	4	6
Nutrients	7	4	0	3	3	9	2	0	3	3
Algal blooms	5	8	0	9	3	5	1	0	2	4
Overabundance of aquatic plants	5	13	0	8	0	8	5	0	6	5
Lack of structure	11	8	10	13	3	16	14	14	10	11
Nonnative species	5	0	**29**	8	18	**37**	10	0	7	12

In view of the importance of flows, additional emphasis is needed on management of minimum flows in tailwaters. A single minimum flow level throughout the year does not provide adequate protection for streams. Tailwater reaches require seasonally adjusted flow regimes to maintain their full ecological function (Poff et al. 1997; Richter et al. 1997). Few states have laws to provide “full protection” of flow regimes, and some states don’t have the legal framework that allows even “threshold, or minimum, flow protection” (Annear et al. 2009). Thus, in many cases more or better laws, regulations, and policies are needed to maintain adequate flows in tailwater reaches.

Section 2

Partnerships for Watershed Management

2.1 Introduction

A watershed is the geographical area that drains into a reservoir, and thus a natural geographical unit for the management of water resources (Figure 2.1). Watershed land cover and land use is a major determinant of water quality, hydrology, and, thereby, fish community composition. A watershed contributes nutrients to a reservoir and subsequently influences primary production. Nutrients, especially phosphorus and nitrogen, flow to the reservoir from all parts of the watershed by way of streams, surface runoff, and groundwater. Typically, watersheds experience various levels of deforestation, agricultural development, industrial growth, urban expansion, surface and subsurface mining activities, water diversion, and road construction. These changes destabilize runoff, change annual amplitudes and distributions of flow, and increase downstream movement of nutrients, sediment, and detritus, which ultimately are trapped by reservoirs. Depending on their extent, inputs can regulate primary productivity, species assemblages, and food web interactions and control most all biogeochemical and ecological processes.

Figure 2.1. Watershed land cover and land use are major determinants of water quality, hydrology, and the fish community in the reservoir. Photo credit: Monroe County Soil Conservation System, Indiana.

Generally, reservoir fish managers lack expertise and jurisdiction to operate outside the reservoir and therefore have to partner with watershed-level organizations. Partnering with these organizations can provide the structure needed to plan, fund, and complete restoration work and may give reservoir fish managers the political clout they normally do not have outside the reservoir.

Over the last two decades watershed management organizations have shown unprecedented growth across the USA. Some of these are small and local, whereas

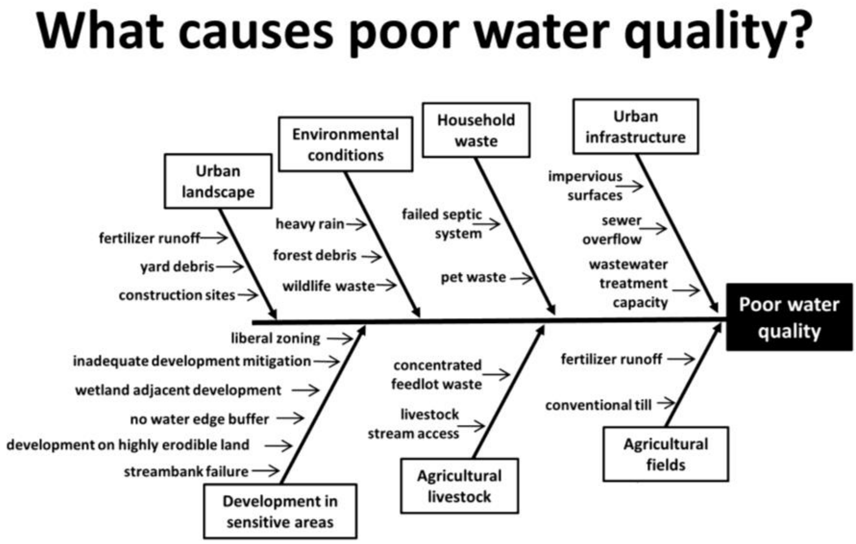


Figure 2.2. Conceptual model of watershed effects on the water quality of Geist Lake reservoir, Indiana, as developed by the Geist–Fall Creek Watershed Alliance. Image credit: Geist–Fall Creek Watershed Alliance.

others are basinwide or statewide. An example is the Geist–Fall Creek Watershed Alliance in central Indiana, which is focused on the improvement and protection of Geist Reservoir's water quality to alleviate fish kills (Figure 2.2). Its watershed management plan includes a watershed inventory, critical areas, goals, best management practices (BMPs), and effectiveness tracking (GWA 2011). Another example is the Cedar Creek Reservoir Watershed Partnership in north–central Texas, also formed to protect the reservoir (TWRI 2007). In this partnership reservoir fish managers participate as members of a technical advisory group. Watershed organizations differ geographically given the diversity of landscapes as well as parallel diversity in the cultural, political, and economic scene. Thus, it is unlikely that a standard model for participation by reservoir managers in watershed organizations is workable in all localities.

2.2 Reservoir Managers versus Watershed Management

It is pertinent to ask what strategic role reservoir fish managers might play in landscape partnerships. As partners, managers can be equipped to show the linkages between the reservoir and disturbances in the watershed and to be activists for change in the watershed that benefits fish in the reservoir. Managers may be prepared to contribute information suitable for developing restoration and protection plans, particularly relevant to how specific actions may affect sediment and nutrient inputs into the reservoir and subsequently fish communities.

As partners, reservoir management agencies contribute human resources with varied skills, abilities, experience, and technical expertise about reservoir management to the collaboration. These agency resources can have varying effects on the partnership's efforts, depending on the particular circumstances that brought the partners to the table. Sometimes agency technical expertise can help the group understand ecological processes and develop innovative plans for management. At other times, technical expertise could get in the way, such as when it is backed with an attitude that experts know best and others have little to contribute.

2.3 Common Watershed Problems

Sediment is a major watershed export into reservoirs that affects the water column through turbidity and, after settling to the bottom, through sedimentation (sections 3 and 5). Mean total suspended solids in 135 Missouri reservoirs ranged from 1 to 47 ppm and were related positively with the proportion of cropland in their watershed, negatively to forest cover, and weakly to grassland cover (Jones and Knowlton 2005). Sedimentation rates in reservoirs are higher in agricultural watersheds and show major shifts in relation to swings in agricultural land management (e.g., McIntyre and Naney 1990). Sedimentation of littoral areas in reservoirs (section 3) often results in loss of depth and associated water-quality problems (sections 5 and 6) and replacement of diverse substrates with fine, uniform particles that blanket existing habitats, fill interstitial spaces, and bury structure. Sedimentation not only affects the backwaters of the reservoir, but, as the backwaters fill, sedimentation extends upward beyond the reservoir into tributaries (section 9), disrupting the reservoir–river interface that supports the diversity of fish assemblages in the reservoir (Buckmeier et al. 2014; Miranda et al. 2014).

Nutrient inputs from the watershed are a leading cause of eutrophication (section 4). Many studies have quantified the interdependence of land cover and nutrient export from a variety of watersheds modified by human activity (Beaulac and Reckhow 1982). In general, nutrient levels in aquatic systems are related directly to the fraction of cropland and inversely related to the fraction of forest cover in the watershed. Row-crop agriculture with frequent tillage and fertilizer application represents a major disturbance to the watershed (Novotny 2003). Nutrient exports from croplands are several-fold that of grassland and forest (Beaulac and Reckhow 1982). Because phosphorus and nitrogen are the principal production-limiting nutrients in freshwater, excessive loading of these nutrients can affect receiving waters adversely. In 135 Missouri reservoirs, phosphorus and nitrogen levels were high in reservoirs surrounded by croplands and lower in those surrounded by forests, resulting in a 7-fold minimum difference in nutrients between a reservoir dominated by forest and one dominated by cropland (Jones et al. 2004). Similar relations were reported in Connecticut (Field et al. 1996), Iowa (Arbuckle and Downing 2001), and Ohio lakes and reservoirs (Knoll et al.

2003). The influences of grassland were less apparent in Missouri reservoirs, with reservoirs dominated by grassland watersheds having about triple the nitrogen and double the phosphorus levels of those dominated by forests. In Iowa, lakes in heavily cropped watersheds had higher nitrogen-to-phosphorus ratios than those found in highly pastured watersheds (Arbuckle and Downing 2001). Nutrient input from urban watersheds often equals or exceeds that from agriculture, as impervious surfaces increase runoff (Beaulac and Reckhow 1982).

Forests affect water quality, discharge, and the quality of reservoir sediment. Likens et al. (1970) showed a large rise in nitrate concentrations and transport following clear-cutting in New Hampshire. In paired watershed studies in northwest Montana, Hauer and Blum (1991) demonstrated increases in nitrogen and phosphorus mobilization and significant increases in algal growth in streams draining watersheds with up to 30% of the total forest area harvested. Vitousek et al. (1982) showed wide variation in nitrification and nitrate mobility in forested watersheds of North America. In shallow natural lakes in Alberta, after timber harvesting, chlorophyll-*a* and cyanobacteria increased and zooplankton decreased after edible phytoplankton biomass declined (Prepas et al. 2001). Woody debris is an important export from forested watersheds, but it is reduced substantially in managed forests relative to unmanaged ones (Duvall and Grigal 1999).

Livestock overgrazing can impair riparian zones and runoff (Belsky et al. 1999). Excessive consumption of vegetation in riparian zones reduces the vegetation's effectiveness at filtering nutrients, and excessive trampling by ungulates destroys the banks of reservoirs and their tributary streams, leading to increased sediment inputs and associated turbidity and sedimentation effects (Platts 1979). Compaction of soils in riparian zones decreases infiltration and thereby increases surface runoff and sediment supply. Magilligan and McDowell (1997) documented improved stream conditions in areas where cattle exclosures were installed. Livestock feeding facilities are major sources of nutrients; dissolved nitrogen inputs are sensitive to cattle densities and feeding rates, and nutrient inputs to aquatic ecosystems are related directly to animal stocking densities (Stout et al. 2000). Where livestock stocking rates are high, manure production exceeds agricultural needs for both nitrogen and phosphorus, causing surplus nutrients to accumulate in soils, later to be mobilized by precipitation into aquatic ecosystems (Carpenter et al. 1998). Intensive cattle, dairy, and hog-raising operations produce voluminous waste that rivals that of small cities, but the effect of livestock animals on aquatic systems is likely to differ across climates, geological settings, and hydrologic conditions.

Urban and suburban encroachments into reservoir watersheds and riparian zones contribute to point and nonpoint inputs of nutrients. Point sources can include wastewater effluent and leaching from waste disposal sites of municipal and industrial facilities and storm sewer outfalls. Nonpoint sources also can include runoff and seepage from animal feedlots and septic systems and from industrial, construction, and other sites. Over recent decades, point sources of nutrient inputs have been reduced partly because of the relative ease in their identification and control.

2.4 Links between Watersheds and Reservoir Fish

Increased nutrient inputs due to watershed practices stimulate aquatic plant growth and affect fish assemblages (section 4). Filamentous algae are favored under high nutrient and light availability conditions but are not readily incorporated into aquatic food webs by invertebrate consumers (Pusey and Arthington 2003). Consequently, fish may find their food base drastically altered in composition and abundance. Moreover, with increased nourishment, phytoplankton communities can shift from domination by green algae to cyanobacteria. Dominance also may shift seasonally, with cyanobacteria dominating for an increasingly longer portion of the year in highly eutrophic reservoirs (Smith 1998). In turn, zooplankton composition may be affected by phytoplankton availability. Macrofiltrators (usually large-bodied zooplankton) are more abundant in oligotrophic reservoirs but give way to low-efficiency, small-bodied algal and bacterial feeders as nutrients increase (Taylor and Carter 1997). In highly eutrophic reservoirs the food supply of zooplankton actually may decrease because of the dominance by cyanobacteria (Porter and McDonough 1984).

High levels of suspended solids reduce light penetration and photosynthesis (sections 4 and 5), reduce plant biomass, alter zooplankton communities, reduce visibility and possibly reduce fish growth, decrease fish size at first maturity and maximum size, and produce a shift in fish habitat use (Bruton 1985). Increases in turbidity, driven by sediment delivered by agricultural watersheds, tend to interfere with feeding by large zooplankton but not by smaller taxa such as rotifers (Kirk and Gilbert 1990). Thus, changes from vegetated to cultivated watershed might favor dominance by small zooplankton taxa and fish adapted to feed on these.

Subsidies from watersheds can promote selected components of reservoir fish assemblages (Figure 2.3). When nutrient subsidies are large, they stimulate phytoplankton and zooplankton production and, in turn, production of planktivorous fishes (Vanni et al. 2005). Similarly, when detritus subsidies are large, they stimulate production of detritivores (Gonzalez et al. 2010). Fish assemblages in reservoirs of agricultural regions of the eastern USA, where nutrient and detritus subsidies from watersheds are large, are often dominated by gizzard shad (Stein et al. 1995), a clupeid that as larvae rely on small plankton and can consume detritus in later stages (Yako et al. 1996; Miranda and Gu 1998). When at elevated densities, gizzard shad influence many functions of reservoir ecosystems, including nutrient cycling, primary production, and

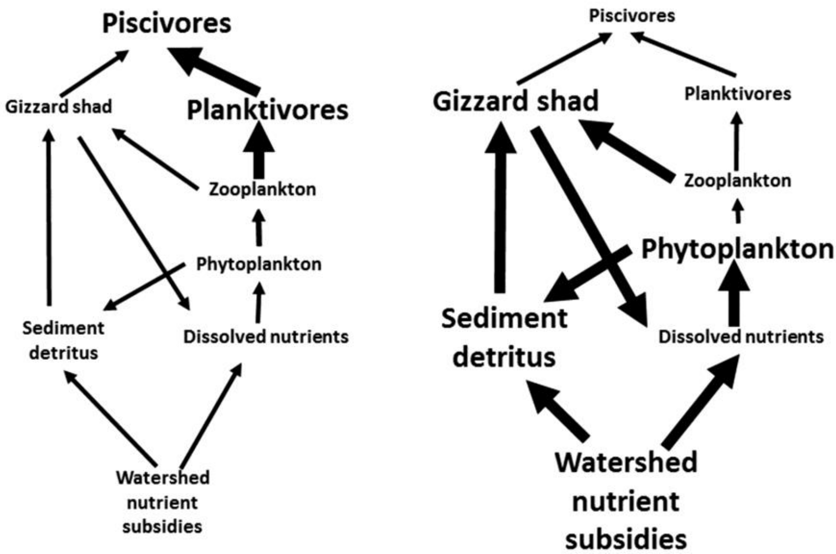


Figure 2.3. Linkages between watersheds and reservoir fish assemblages. The sizes of the arrows indicate the relative difference in a particular flux rate between least disturbed watersheds (left panel) and highly disturbed watersheds. Similarly, font size represents the relative difference in biomass between the watershed types. In less disturbed watersheds, subsidies from the watershed are lower and gizzard shad are scarce. This allows planktivores (e.g., sunfishes) to thrive; piscivores are abundant because they feed on these planktivores. In highly disturbed watersheds, subsidies of dissolved and particulate nutrients from watersheds are more substantial. These inputs stimulate phytoplankton productivity and also provide detrital resources for gizzard shad. Shad biomass therefore increases, leading to increased nutrient transport by shad. This transport leads to further increases in phytoplankton biomass and water-quality complications. Gizzard shad larvae are obligate planktivores and may outcompete other planktivores. Shad also may not be as vulnerable to piscivores as are other planktivores, and hence piscivores are less abundant in highly disturbed watersheds. A multiplicity of other variables can influence the outcome of trophic interactions. Image modified from Power et al. (2004).

composition and structure of the entire fish assemblage (Power et al. 2004). Conversely, when nutrient and detritus subsidies are small, phytoplankton production is reduced, water clarity increases, and zooplankton production is shifted toward grazing zooplankters such as *Daphnia* (Kirk and Gilbert 1990; Mazumder 1994). Lacking meaningful levels of detritus and increased water transparency, species composition shifts toward taxa that rely on visual selection of zooplankton prey or other invertebrates through some or all of their life stages (Power et al. 2004). In the eastern USA, those fish assemblages are often dominated by centrarchid species (Near and Koppelman 2009).

An analysis of fish assemblages in reservoirs within wooded and agricultural watersheds of the Tennessee River basin further suggests that entire fish assemblages are influenced by watershed land use (Miranda et al. 2015). In an agricultural basin, fish assemblages included a greater percentage of species that depend on diverse combinations of small zooplankton, benthic invertebrates, benthic algae, and detritus (e.g., threadfin shad, common carp, spotfin shiner, redhorses). In a forested basin, fish assemblages included a greater percentage of species that depend on large zooplankton and macroinvertebrates at some stage throughout their life cycle (e.g., redbreast sunfish, whitetail shiner, warmouth, spotted bass, largemouth bass). Thus, land-use differences in the study basins were associated with detritivore-based and invertivore-based fish assemblages.

2.5 Iowa's Lake and Watershed Management Program

Iowa leads the nation in disturbed land area: 72% of its land area has been converted to cropland, which combined with an additional 10% pastureland and 5% developed land results in 87% of Iowa's land area being directly disturbed (Heitke et al. 2006). As a result, many natural and constructed lakes in Iowa are impaired, with poor water quality, compromised fisheries, and reduced recreational value relative to their potential (R. Krogman, Iowa Department of Natural Resources, IDNR, personal communication). Over the years many lakes were renovated, some multiple times, resulting in improved fisheries that quickly degraded because underlying problems originating in the watershed (e.g., heavy sedimentation, excessive nutrients, legacy nutrient banks, and ensuing poor water quality) were not fully addressed. A lake scoring system was developed to rate lakes and reservoirs based on water quality, their potential for public benefit, and their feasibility for restoration. This rating, combined with socioeconomic factors, resulted in a priority ranking of lakes and watersheds for restoration. After local commitments are demonstrated and feasibility verified, comprehensive restoration is initiated to address both watershed and in-lake issues. Watershed models are used to simulate hydrologic processes and pinpoint the major sources of sediment and nutrient loading. These loads are reduced to acceptable levels through land-use changes and application of watershed BMPs.

Iowa fisheries managers work within partnerships composed of government agencies, landowners, and nongovernment organizations and invest 20%–30% of their efforts on watershed work associated with lake and stream projects (C. Dolan, IDNR, personal communication). Fisheries managers work in various capacities within partnerships, often as leaders in technical details of specific projects. This approach may be intimidating at first, but it does produce success stories and does garner the public support required to get the funding needed to work at the watershed scale. These restorations can be expensive and require years to complete but are an investment in the local economy, fishing quality, and natural resources as a whole.

2.6 Tennessee Valley Authority's Watershed Partnerships

The Tennessee River includes over 30 major reservoirs operated by the Tennessee Valley Authority (TVA) for navigation, flood control, power production, water quality, and recreation. In 1991, TVA adopted a reservoir-operating plan that increased the emphasis placed on water quality and recreation. This plan modified the drawdown of 10 tributary reservoirs to extend the recreation season and included a US$50 million program to improve conditions for aquatic life in tailwater areas by providing year-round minimum flows and installing aeration equipment at 16 dams to increase oxygen levels. In 1992, to prevent these improvements from being negated by nonpoint pollution, TVA launched an effort to protect watersheds by forging partnerships with governments, businesses, and citizen volunteers. The goal was to ensure that rivers and reservoirs in the basin were ecologically healthy and biologically diverse and supported sustainable uses. To accomplish this goal without regulatory or enforcement authority, TVA built action teams in each of 12 sub-basins within the Tennessee River basin (Poppe et al. 1997). These teams were responsible for assessing resource conditions and building partnerships to address protection and improvement needs.

The action teams represented a transformation of TVA's water management organization from a hierarchy organized around technical disciplines to an organization based upon cross-functional teams. These teams were unique in that they combined the skills of aquatic biologists, environmental engineers, and other water resource professionals with the skills of community specialists and environmental educators. Team members learned to communicate with the public in nontechnical language and to build partnerships with farmers, waterfront property owners, businesses, recreationists, and local and state government officials. Assigning teams to a geographical area for the long term allowed members to gain a better understanding of resource conditions, build community trust, and enhance the development of cooperative relationships with stakeholders. The teams were self-managed and empowered to decide how to focus resources and address protection and improvement needs, allowing a rapid response to evolving or newly discovered problems and opportunities.

The teams conducted watershed inventories so that TVA could rate sub-basins for their degree of degradation and identify areas needing remediation. This information helped focus resources and evaluate improvement activities. Team members shared monitoring information with key stakeholders (e.g., regulatory agencies, state and local governments, businesses and industries, citizen-based action groups, and watershed residents) and sought their support in developing and implementing protection and mitigation plans.

Team efforts to build partnerships paid off. In 1995 volunteers contributed 22,500 hours in monitoring, habitat enhancement, cleanup, and protection activities.

Acting as catalysts for change, action teams helped start or worked in partnership with many local coalitions to solve water-quality problems; conducted over 400 stream and reservoir assessments; established 20 native aquatic plant stands in reservoirs; installed 4,500 habitat structures; stabilized shorelines; and implemented watershed management practices, including construction of wetlands, fencing, and streambank revegetation. Team members also organized a variety of communication activities designed to educate people about water quality and involve them in solving pollution problems. By focusing on partnerships, action teams were able to accomplish what TVA could not have done acting as an independent government agency.

2.7 Watershed Management

2.7.1 The Necessity of Watershed Management

A fair question is whether watershed management is always necessary. The importance of watershed management is likely to increase with the level of disturbance experienced by the landscape. Reservoirs in relatively undisturbed landscapes

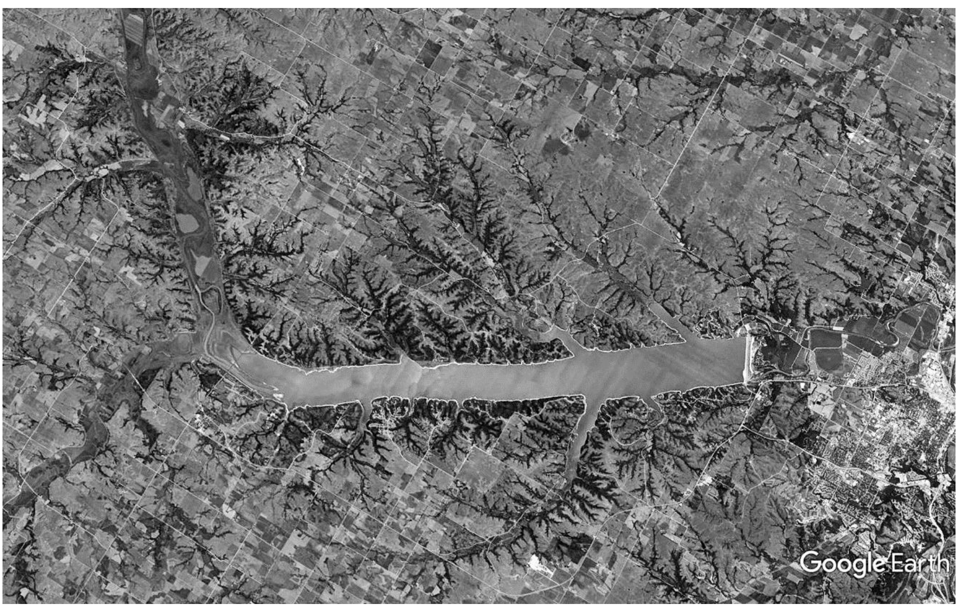


Figure 2.4. For most of the twentieth century fish managers viewed reservoirs as isolated lakes. With the arrival of easy access to satellite imagery, concepts about reservoirs have expanded to include the potentially large effects of watersheds and tributaries. The image shows Tuttle Creek Lake, Kansas surrounded by a mosaic of agricultural patches and fed by a large tributary with extensive backwaters. The reservoir is listed on the Kansas Section 303(d) as impaired by sedimentation and eutrophication. Extremely high suspended solids and nutrient loads enter the reservoir during storm events reducing its volume by approximately 40% and filling with sediment faster than other reservoirs in the region. Photo credit: Google Earth.

with high-quality tributaries and riparian zones are likely to require mainly watershed protection and traditional in-lake habitat management. In contrast, reservoirs in heavily disturbed landscapes with highly engineered tributaries may require considerable off-lake attention before in-lake habitat management becomes effectual (Figure 2.4). In this latter group, a focus on in-lake habitat management may provide only short-term fixes to complicated watershed issues that are the underlying problems to inadequate fish assemblages.

2.7.2 Goal of Watershed Management

The goal of watershed management usually is to facilitate self-sustaining natural processes and linkages among the terrestrial, riparian, and reservoir environment. It involves controlling the quantity, makeup, and timing of runoff flowing into the reservoir or tributaries from the surrounding terrain. The first and most critical step is halting, eliminating, or altering those anthropogenic practices causing reservoir degradation. Such approaches can involve a wide range of adjustments to human activities. For example, it may involve increasing widths of buffer strips around fields, altering livestock grazing strategies to minimize adverse effects, moving tillage operations farther away from riparian systems and water, changing tillage methods and timing, and stopping the release of industrial waste that causes water pollution. To this end, various management practices have been developed to target the diversity of potentially troublesome nonpoint sources in the watershed. Watershed management practices usually are applied as systems of practices because one practice rarely solves all problems, and the same practice will not work everywhere. There is a large body of literature about watershed management available to reservoir managers; nevertheless, in most cases, watershed management is not the direct responsibility of the fishery manager.

Since the late 1970s, many federal and state programs have been established in the USA to reduce soil erosion through implementation of BMPs in riparian and upland areas (Table 2.1). Federal agencies such as the U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS) and the Farm Services Agency (FSA) are responsible for reducing soil erosion and sedimentation problems. The NRCS provides technical assistance to farmers in the areas of BMP application, compliance with state water-quality standards, and voluntary efforts. Financial assistance is available to farmers from the FSA for efforts to control erosion and sedimentation problems. Technical, educational, and financial assistance is also available to eligible farmers through the Environmental Quality Incentives Program. This program addresses soil, water, and related natural resource concerns on farm lands in an environmentally beneficial and cost-effective manner. The purposes of the program are achieved through the implementation of a conservation plan that includes structural, vegetative, and land management practices on eligible land. Cost-share payments may be made to im-

Table 2.1. Watershed-related responsibilities of selected U.S. federal agencies (adapted from Graf et al. 1999). X = significant responsibility; O = some responsibility.

Federal agency	Estuaries	Wetlands	Research	Hydropower	Navigation	Recreation	Preservation	Wildlife	Fisheries	Flow regimes	Ecological diversity restoration	Erosion/sediment control	Water quality	Flood risk management	Water supply	Land use
Department of Agriculture																
Farm Services Agency		O									O	X				X
Forest Service		O	O			X	X	X	X		X	O	O	O	O	X
Nat Resources Conserv Services		X								O	O	X	O	O		X
Agricultural Research Service												O	O		O	X
Department of Commerce																
National Marine Fisheries Service	X	O			O			O	X				O			
National Oceanic Atm Admin	X								X				X	X		
Department of Defense																
Army Corps of Engineers		X	O	X	X					X	O	O	O	X	O	O
Department of Energy																
Federal Energy Regulatory Comm				X												
Department of the Interior																
Bureau of Land Management		X	O			O		X		X	X	O	O			X
Bureau of Reclamation			O	X		O		O	O		O	O	O	X	X	X
Fish and Wildlife Service		X	O			O	X	X	X		X		O			
Geological Survey	O	X	X					O	O	O	X	O	O	O	X	O
National Park Service	O	O				X	X	X	X		X	O	O			X
Bureau of Indian Affairs		O						O	O		O	O	X	O		O
Department of State																
International Boundary Comm	O												X	X	X	
Other Federal Independent Units																
Environmental Protection Agency	O	X	X			O	O	O	O	O	X	O	X		O	O
Tennessee Valley Authority		O	O	X	X	O		O	O	X	O	X	X	X	X	
Bonneville Power Administration		O		X	X			O	X	O	X	O	X	X	X	
Federal Emergency Mgmt Agency														X		

plement one or more eligible structural or vegetative practice, such as terraces, filter strips, tree planting, and permanent wildlife habitat. Incentive payments can be made to implement one or more land management practice. The U.S. Environmental Protection Agency (USEPA) is concerned with water-quality degradation caused by turbidity from soil erosion and sediment runoff. The USEPA is authorized under Section 319 of the Clean Water Act to work with states to develop management programs to solve water-quality problems, such as turbidity, and to provide matching grants for the implementation of approved nonpoint source management programs.

2.7.3 Watershed Inventory

A watershed inventory documenting features important to reservoir condition is essential. The inventory may focus on critical areas representing major sources of problems likely to have large effects on the reservoir, such as large stretches of channelized tributaries without adequate instream habitat or access to floodplain, agricultural ventures stretching down to the banks, and forest clear-cutting operations (Table 2.2). A focus on critica l areas would result in the greatest improvements and save time when gathering available information or conducting on-site surveys. Also, characterization of the watershed generally is limited to a geographic area or scale large enough to ensure that management opportunities will address all the major sources and causes of habitat degradation in the reservoir. Although there is no rigorous definition or delineation of this scale, the goal is to avoid a focus on narrowly defined scales that do not provide an opportunity for addressing watershed stressors in an efficient and economical manner. At the same time, the scale needs not be so large that it precludes successful implementation. A scale that is too broad might allow only cursory assessments and not accurately link effects to sources.

A visual watershed assessment may be one of the least costly assessment methods. By walking, driving, and boating parts of the watershed, one can observe water and land conditions, uses, and changes over time that might otherwise be unidentifiable from aerial surveys. These surveys identify and verify the source of pollutants, such as streambank erosion delivering sediment into the stream and illegal pipe outfalls discharging various pollutants. Visual watershed surveys can provide an accurate picture of what is occurring in the watershed and also can be used to familiarize local stakeholders, decision-makers, citizens, and agency personnel with activities occurring in the watershed. The survey may photograph key characteristics of the critical areas. These surveys may be greatly assisted by Geographic Information Systems (GIS) land-cover layers (Brenden et al. 2006) or simply by visual surveys in Google Earth. The GIS surveys can be used to identify major hazards and large nutrient and sediment sources, as well as help steer an on-the-ground survey.

Inventorying the watershed and its problems provides the basis for developing management strategies to meet watershed goals. Without an understanding of where pollutants are coming from, it is impossible to target control efforts effectively.

Table 2.2. Examples of the type of data to include in a watershed inventory and how they might be used. USGS = U.S. Geological Survey; NRCS = Natural Resources Conservation Service.

Data type	Data use
Watershed boundaries	• Provide geographic boundaries for evaluation. Depending on size of the watershed, boundaries might already have been delineated by an agency (e.g., USGS, NRCS). • Delineate drainage areas at desired scale.
Climate	• Organize precipitation data to provide insight into wet and dry seasons, which can help characterize watershed problems and sources.
Hydrology	• Identify the locations of water bodies. • Identify the spatial relationship of water bodies, including what segments are connected and how water flows through the watershed. • Identify flow gages in the watershed. • Gather information about loadings during storm events and high flow. • Identify any instream flow alterations or stream fragmentation. • Identify water rights (particularly in the West), use, and demand and how they affect the reservoir during drought years, upstream and downstream.
Topography	• Derive slopes of stream segments and watershed areas (e.g., to identify potentially unstable areas). Steep slopes might contribute more sediment than flat landscapes. • Evaluate elevation changes.
Soils	• Identify areas with high erosion rates or poor drainage. Soils can be grouped into hydrologic soil groups according to their runoff potential. Information is available from NRCS and local soil and water conservation districts. • Delineate subwatersheds based on soils. Soils have inherent characteristics that control how much water they retain, their stability, and water transmission. Understanding the types of soils in the watershed and their characteristics helps to identify areas that are prone to erosion or are more likely to experience runoff.
Land use and land cover	• Identify potential pollutant sources (e.g., nutrients, sediment). Sources are often specific to land uses, providing a basis for identifying sources. For example, land use for grazing livestock and agriculture potentially contribute mostly nutrients and sediment. Conversely, urban land uses typically have different signature pollutants (e.g., metals, oil, grease, point sources). • Identify potential sources of bacteria, such as livestock operations, wildlife populations and their distribution, and septic systems. • Identify known pollutant impairments in the watershed. These include wastewater treatment plants, industrial facilities, and concentrated animal-feeding operations. The discharge of pollutants from point sources, such as pipes, outfalls, and conveyance channels, is generally regulated. Point-source discharge information generally is available from state agencies. • Identify current control practices and potential targets for management.
Land ownership	• Identify land ownership. Many watersheds contain land owned by diverse parties, including private citizens and federal and state government agencies. Information on land ownership can provide insight into sources of information for characterizing the watershed or identifying pollutant sources. Ownership information can also be useful in identifying management opportunities.
Reservoir	• Describe long-term trends in sediment accumulation. • Describe seasonal patterns in clay turbidity or sediment intake.

This characterization may already be available, can be developed from existing information, or, rarely, is started from scratch. In the absence of existing data, rapid watershed assessment guidance is available (NRCS 2005; USEPA 2008) to serve as a framework for conducting the necessary surveys. However, research is needed to establish survey protocols specific to reservoir needs, develop and refine quantitative metrics to prioritize and measure progress, and establish how to integrate reservoir needs into landscape planning efficiently.

Some information may be available in 303(d) reports. Section 303(d) of the Clean Water Act requires states to develop lists of water bodies that do not meet water-quality standards (impaired) and to update lists every 2 years. The USEPA is required to review impaired water body lists submitted by states and approve or disapprove all or part of the list. For impaired water bodies on the 303(d) list, the Clean Water Act requires that a pollutant load reduction plan be developed to correct each impairment. This plan requires documentation of the nature of the water-quality degradation, determination of the maximum amount of a pollutant that can be discharged and still meet standards, and identification of allowable loads from the contributing sources. The elements of a plan include a problem statement, description of the desired future condition, pollutant source analysis, load allocations, description of how allocations relate to meeting targets, and margin of safety. However, the Clean Water Act is limited to waters with a significant nexus to navigable waters, and agricultural nonpoint discharges are generally exempted from regulatory oversight through the Clean Water Act.

Identifying existing information is critical to supporting the development of a watershed plan that is based on current or future planning efforts (e.g., zoning, development guidelines and restrictions, current and future land-use plans, road plans). This information will support the characterization of the watershed and identify any major changes expected to occur. To know what is available and how to get the information, it is necessary to become familiar with state-, county-, and city-level agencies. One may need to understand the authority and jurisdictions of the agencies in the watershed. For example, it is important that the watershed plan identify management practices that agencies in the watershed have the authority and jurisdiction to implement.

2.7.4 Partnership Building

Bringing together people and resources to address reservoir problems through a watershed approach blends science with social and economic considerations. Given reservoir fish managers often have no jurisdiction over the watershed, the very nature of working at a watershed level means managers may need to work with at least one partner. Watershed management is often too complex and too expensive for a fisheries management organization to tackle alone. Weaving partners into the process can strengthen the end result by bringing in new ideas and human resources and by increasing public understanding of the problems and commitment to the solutions. Partnerships also help to identify and coordinate existing and planned efforts. For example, a partner might be interested in implementing a volunteer monitoring program but is unaware that the local parks department is working on a similar program. Working with partners can help to avoid "reinventing the wheel" or wasting time and money in duplicated effort. Budgets can be unpredictable, and resources for watershed improvement efforts, such as fencing cows out or building retention ponds, are limited. Working with partners might provide resources not directly available to reservoir managers.

The makeup of partnerships will depend on the size of the watershed (to ensure adequate geographic representation) as well as the key issues or concerns. In general, there are at least four categories of participants to consider when identifying partners. These categories include (1) those partners responsible for implementing management practices; (2) those affected by the management practices; (3) those who can provide information on the issues and concerns in the watershed; and (4) those partners that can provide technical and financial assistance in developing and implementing practices. To function as a collaborative body effectively, the membership of the partnership might require balance in geographic and topical representation.

2.7.4.1 Levels of partnering

For simplicity as an example, two contrasting levels of partnering intensity may be identified. Each level represents a degree of involvement and sophistication in collaborative interactions between reservoir managers, other professionals, and the public. The levels refer both to the extent to which collaboration occurs and to the capacity for collaboration in a watershed setting as a whole. The extent of partnering on particular cases will be a function of the nature of the watershed problem and the collaboration capacity of watershed groups or agencies. The hierarchy of the two levels assumes that the greater the level of collaboration, the better the management of difficult watershed problems is likely to be. Conversely, difficult problems will generally challenge less collaborative settings beyond their ability to manage problems adequately. This model does not prescribe an optimal level of collaboration but rather describes the strengths and limitations of a variety of options. Moreover, agencies may use different levels for different problems, depending on the level of collaboration required.

At a basic level of partnering (i.e., project-based partnering), agencies, groups, or both continue working within their normal modes but with commitments to collaborate on one or more projects of mutual interest. Because of this collaboration and coordination, individual planning efforts are better aligned. Agencies and groups may pursue opportunities to co-fund top projects and promote policies to further their plans and implementation. Regular meetings are set between working groups to enhance communication on particular projects.

At a more complex level of partnering (i.e., systemic partnering), agencies, groups, or both establish shared goals, systems, and agreements to increase efficiency through collaboration within existing agency structures. Agencies may engage in a facilitated process to identify a shared vision and systemic way of bringing together their personnel—from the highest level of leadership in the agency to field managers—to evaluate short- and long-term opportunities for watershed management. Joint planning and decision-making may take place before study designs and budgets are locked in to allow for better use of available resources without unnecessary duplication of equipment, personnel, or effort. Agencies and groups may share some costs and pool resources to attract additional funding.

2.7.4.2 Development of collaborative know-how

Some organizations are accustomed to partnering with other agencies, but others may need to learn how to cooperate and work with organizations, agencies, and public groups that have different values, procedures, and processes. When organizations participate in collaborative processes, they begin a learning process that produces collaborative know-how (Imperial and Kauneckis 2003). Stakeholder collaboration develops as part of a learning process. Once the relationship between stakeholders is established, and collaborative projects are successful, it is much easier to take on additional collaboration. Learning how to partner effectively or, conversely, to identify and avoid ineffective partnerships takes time. However, the pace and scope of collaborative efforts can increase when partners gain experience implementing collaborative projects. Thus, reservoir managers over time gradually could scale up collaborative efforts in the watershed as they build on previous success.

2.7.4.3 Organizational structure

Partnerships between agencies and groups are likely to be facilitated by an organizational structure and culture that possibly may be different from that of many contemporary fishery management agencies. Depending on built-in flexibilities, agencies currently organized as isolated fish and game departments are likely to find it more difficult to collaborate with watershed agencies than those organized as departments of natural resources. Many agencies might require reorganization to develop the mission, mandate, resource authority, and skills required to manage reservoir habitats effectively at broader scales. In many cases, institutions that have served us well in the past outlive their intended missions and usefulness. Over time, existing agencies are reorganized to create new complexes of organizations to make decisions and meet new needs. This may mean rethinking the role and structures of natural resource management agencies.

2.7.4.4 Evaluation of partnering efforts

Partnering requires various levels of time and resource commitments. Thus, agencies and individuals may wish to evaluate periodically whether partnering efforts are achieving desired goals. Various aspects of the partnering effort may be considered before entering into a partnership or reconsidered periodically (e.g., annually) when involved in a partnership. An agency may wish to evaluate the effect and outcomes of the partnership, perhaps by asking what tangible outcomes involvement in the partnership had or whether any achievements might have been accomplished outside the partnership anyways. It also may be important to note what, if any, were the benefits for the agency and for its clientele and what changed as a result of participation in the partnership. Although the partnership might have produced important results, was what happened aligned with the goals set? It also would be useful to list any unintended positive or negative effects the partnership might have produced.

2.7.5 Watershed Management Practices

There are many types of individual management practices, from agricultural stream buffers, to urban runoff control practices, and to homeowner education programs. This section aims to familiarize the reservoir manager with three general classes of watershed BMPs, without an exhaustive review that is generally in the purview of land-based organizations. Management practices can be grouped into structural practices, nonstructural practices, and regulatory practices (Table 2.3). Structural practices are defined as something that is built or installed in the watershed. Examples may include sediment basins, filter strips, and drainage systems. Nonstructural practices usually involve changes in activities or behavior and focus on controlling pollutants at their source. Examples include developing and implementing erosion and sediment control plans, organizing education campaigns, and practicing good tidiness at industrial complexes. Regulatory practices include ordinances and permits.

2.7.5.1 Structural practices

Structural practices might involve construction, installation, and maintenance of existing structures. Structural practices can be vegetative, such as soil bioengineering techniques, or nonvegetative, such as riprap. Practices such as bank stabilization and riparian habitat restoration involve ecological restoration and an understanding of plant communities, individual species, natural history, and the vegetation’s ability to repopulate a site.

Table 2.3. Structural, nonstructural, and regulatory practices applied to watersheds. Structural practices involve something built or installed, nonstructural practices involve changes in activities or behavior, and regulatory practices involve ordinances and permitting.

Structural	Nonstructural	Regulatory
	Agriculture	
• Buffer/filter strips	• Brush management	• Tax incentives
• Grassed waterways	• Conservation coverage	• Pump logs
• Wind barriers/brush layer	• Conservation tillage	• Water use permits
• Mulching	• Educational materials	• Water management systems (ditches, culverts)
• Live fascines	• Erosion and sediment control	• Wetlands protection
• Live staking	• Nutrient management	• NPDES permits[1]
• Livestock exclusion	• Pesticide management	• Required training
• Revetments/riprap	• Prescribed grazing	• Pesticides storage/disposal
• Sediment basins	• Nutrient management training	• Surface water discharge permits
• Terraces	• Manure management system	• Waste disposal permits
• Waste treatment basins		• Conservation easements
• Cover crops/seeding		
• Fencing		
• Tree planting		
• Watering facility		
	Forestry	
• Broad dips	• Education of landowners and loggers	• Harvest/reforestation permits
• Culverts	• Forest chemical management	• Notification of intended harvest
• Riparian buffers	• Fire management	• Chemical permits
• Mulching	• Road layout	• Road construction methods
• Cover crops/seeding	• Preharvest planning	• Forest land conversion
• Windrows		• Management plans
• Road stabilization		
• Grade stabilization		
	Urban	
• Detention basins	• Reduction of impervious areas	• Land-use zoning
• Green roofs	• Educational materials	• Stormwater ordinances
• Stormwater ponds	• Lawn fertilizer management	• Wastewater treatment/discharges
• Sediment basins	• Pet waste programs	• Material storage/handling
• Tree revetments	• Shore setbacks	• Lawn care
• Wetland creation/restoration	• Storm drain stenciling	• Water setback requirements
• Water-quality swales	• Watershed zoning	
• Riprap	• Preservation of open space	
• Vegetated gabions	• Development of greenways	
• Silt fence/straw bales		
• Erosion control fabric		

[1] National Pollutant Discharge Elimination System permit program, authorized by section 402 of Clean Water Act.

2.7.5.2 Nonstructural practices

Nonstructural practices prevent or reduce runoff problems by reducing the generation of pollutants and managing runoff at the source. These practices can be included in a regulation (e.g., an open space, riparian stream buffer requirement, tilling method) or they can involve voluntary pollution prevention practices. They also can include education campaigns and outreach activities. Nonstructural practices can be subdivided further into land-use practices and source-control practices. Land-use practices are directed at reducing effects on receiving waters that result from runoff by controlling or preventing certain land uses in sensitive areas of the watershed. Source-control practices are aimed at preventing or reducing potential pollutants at their source before they come into contact with runoff or groundwater. Some source controls are applicable only to new watershed development, whereas others can be implemented after development occurs. Source controls include pollution prevention activities that attempt to modify aspects of human behavior, such as educating citizens about proper application of lawn fertilizers and pesticides.

2.7.5.3 Regulatory practices

Management practices required to manage the watershed can be implemented voluntarily or required under a regulatory program. Point sources are most often controlled using regulatory approaches and can work well if adequate mechanisms are in place to provide enforcement. For example, local stormwater ordinances may require development applicants to implement practices such as retention ponds or constructed wetlands to meet performance standards for the development set forth in the ordinance. Local development and subdivision ordinances may require development applicants to meet certain land-use (e.g., commercial versus residential versus undeveloped), development intensity, and site design requirements (e.g., impervious surface limits, open space, riparian buffers, setback requirements). Forestland owners often are required to develop and implement forest management plans. Federal or state lands that are leased to individuals often require permits that specify conditions and management practices adhered to for the term of the permit.

Point sources are regulated under the National Pollutant Discharge Elimination System (NPDES) permit program, authorized by Section 402 of Clean Water Act. Certain concentrated animal-feeding operations that meet a minimum threshold for number of animals require NPDES permits. Activities that take place at industrial facilities, such as material handling and storage, are often exposed to the weather. Operators of industrial facilities included in 1 of 11 (Jaber 2008) categories of stormwater discharges associated with industrial activity that discharge stormwater into a sewer system or directly to water bodies are regulated under a NPDES industrial stormwater permit.

Section 3

Sedimentation

3.1 Introduction

Sedimentation is a natural process in all water bodies. Sedimentation is relatively higher in reservoirs than in other water bodies because reservoirs impound a large volume relative to the area of their watershed. Sediment accumulation is accelerated by inadequate land-use practices that liberate soils, by the conversion of land into urban and suburban development that hastens runoff, or both. The rivers and streams deposit their sediment loads in the calmer waters of reservoirs, where sediment accumulation can have negative effects. Infilling with sediment can decrease water storage capacity and reduce the benefits of storing water in reservoirs. Shallower waters also may decrease the recreational value of a reservoir and the loss of access to parts of the upper reaches and embayments. Sedimentation also can result in the loss of habitat for fish, and sediment can carry pollutants including nutrients, which may act as catalysts for eutrophication.

Figure 3.1. Matilija Dam, on the Ventura River Basin, California, was constructed in 1947 and is nearly completely filled with sediment. When constructed, the dam was 190 ft high and impounded a volume of 7,000 ac-ft; by 2015 it was reduced to 400 ac-ft. Photo credit: P. Jenkin.

Reservoir sedimentation can change physical, chemical, and biological components of the ecosystem, which results in the degradation of beneficial uses such as drinking water supplies, navigation, electricity production, flood control, and recreation (Figure 3.1). Eventually the reservoir may have to be abandoned. In the USA more than 3,000 such dams have been retired (Marsh 2005). The effects of deposited sediment delivered from watersheds can have severe economic costs for downstream residents and may decrease property values for lakefront properties and those properties near the reservoir.

Table 3.1. Spearman correlations (r_S) between sedimentation and various watershed and in-lake characteristics. All correlations are statistically significant ($P < 0.01$; N = 1,271 reservoirs).

Variable	Description	r_S
Harmful levels of agriculture	Watershed around the reservoir has adverse row-crop agriculture practices	0.54
Harmful levels of livestock	Watershed around the reservoir has adverse grazing practices and/or feedlot production	0.45
Disturbances in upstream watersheds	Disturbances in watersheds upstream of the reservoir (not around) impairs habitat	0.55
Lack of connectivity due to sedimentation	Sedimentation has decreased connectivity to tributaries during low flow, acting as a barrier to fish movement	0.49
Shoreline erosion	Removal of soil and terrestrial vegetation from the land–water interface resulting from weathering of banks or adjacent land slopes by water, ice, wind, or other	0.63
Shoreline homogenization	A reduction of the shoreline's original habitat diversity by erosion or other processes	0.52
Homogenization of littoral substrates	A reduction of the substrate's original diversity by erosion and sedimentation	0.63
Excessively shallow	Reservoir is excessively shallow with no or few deep water refuges	0.54
Excessive mudflats	Seasonally flooded and exposed expansive soft sediment present; terrestrial vegetation seldom grows unless mudflats are exposed for many months	0.55
Excessively shallow littoral zone	Littoral zone is mostly shallow and heavily influenced by temperature, wind, and other atmospheric changes	0.56
Excessive nutrients	Excessive nutrients, primarily nitrogen or phosphorous, that may increase primary production and lead to excessive plant growth and decay and, lack of oxygen	0.55
Excessive suspended sediment or inorganic turbidity	Particulate inorganic matter and fine sediment in the water column that may inhibit primary production or foraging by fish and other aquatic organisms	0.72
Excessive organic turbidity	Particulate organic matter, other than algal blooms, suspended in the water column	0.56
Loss of cove habitat due to sedimentation	Sedimentation has changed cove habitat, including reduced surface area, fragmentation, and establishment of terrestrial vegetation in newly deposited land	0.73

Sedimentation is a major issue in many reservoirs in the USA. A survey of reservoir managers identified that approximately 28% of reservoirs ≥250 ac in the USA were of moderate-to-high or high concern relative to sedimentation (Krogman and Miranda 2016). These percentages vary regionally; for example, sedimentation afflicts as many as 51% of reservoirs in regions along the plains in the central USA. This same survey identified various correlations between sedimentation and both watershed and in-lake characteristics, particularly turbidity and loss of shallow reservoir habitats (Table 3.1).

3.2 Sources of Sediment

Reservoir sedimentation begins with soil erosion caused by rain and wind and with runoff that transports sediment particles into streams (Novotny and Olem 1994). Depending on composition, various types of land cover produce different runoff characteristics. Determining sediment sources is essential for designing cost-effective sediment management strategies that will achieve meaningful reductions in sediment loads and yields (Walling 2005). Overall, the sediment entering reservoirs originates from erosion of four general sources: (1) soil from overland flow, including farmed areas in the watershed; (2) streambank and channel erosion, including channel migration, bank widening, and avulsion; (3) remobilization of stored sediment through channel processes acting on floodplains or other storage sites; and (4) erosion of shorelines and shallow-water areas by wave action.

Streamflow slows as it enters a reservoir, and suspended particles begin to settle out. Eventually, most sediment will settle to the bottom of the reservoir, but heavier sediment particles are deposited first. Sedimentation does not occur uniformly; it is affected by many factors including the flow and volume of water produced by the incoming stream and the size and weight of sediment particles. The coarser portion of the inflowing sediment load is deposited where the main tributary and minor tributaries to embayments enter the reservoir. There, the tributaries form delta deposits that deplete reservoir storage, cause channel aggradation extending miles upstream from the reservoir, and fill in shallow coves and embayments that often represent some of the most diverse fish habitats in the reservoir (Williams 1991). Channel aggradation can change flood patterns and floodplain configuration upstream. If delta areas become heavily vegetated, the upstream flood levels can be elevated further because of increased hydraulic roughness, and the vegetation can trap sediment, thereby promoting additional aggradation (James and Barko 1990). In arid zones the transpiration from large areas of vegetation in delta areas can increase water losses from the reservoir significantly. For example, evaporative losses from the delta of Elephant Butte Reservoir on the Rio Grande in New Mexico were estimated at 140,000 ac-ft/year before they were reduced by 66,000 ac-ft/year by the construction of a low-flow conveyance channel through the delta in 1951 (BOR 2007).

The nature of the sediment accumulated in a reservoir depends on geology, topography, soil, and climatic conditions. Where parent materials in the watershed are shales or limestone, sand content of sediment is low. Where parent material is mainly sandstone, sand content may be high. Some igneous and metamorphic rocks produce fine sediment under some climatic conditions and coarse material under others. For example, sediment derived from Piedmont areas in the southeastern USA contains proportionally large amounts of clay and colloidal material. Sediment derived from loess-type soils in the Midwest has high silt content. The Cross Timbers area of Texas, with sandy soils and poorly consolidated sandstone substrata, provides sediment with high sand content. Environmental origin has a definite bearing on watershed sediment yield, transport, and deposition in a reservoir, and the nature of the sediment has a direct bearing on the percentage of total load deposited in the reservoir and on the ultimate volume of deposited material.

A river can carry sediment into a reservoir in two distinct modes: bedload and suspended particles. In bedload transport, the sediment particles move by rolling or sliding or through jumps the length of a few grain sizes (known as saltation), and they are thus in frequent contact with the channel bed. In suspended load transport, the weight of the particles is supported by turbulent forces in the water, and they can travel considerable distances without coming into contact with the bed. The total load is the sum of suspended load and bedload. Whether an individual particle is transported as suspended load or as bedload depends on particle size, weight, and shape and on hydraulic conditions. Whereas the bedload is often deposited in the upper end of reservoirs or the upper end of embayments, depending on current suspended loads can move farther into the reservoir. This fraction does not easily sink in the water column, and slight turbulent forces keep it in suspension for long periods. In rare instances, light dispersive colloidal clays delivered in suspended loads never settle out and remain in suspension aided by minimal wind-generated wave energy. In reservoirs, besides river inputs, fine suspended material originates from shoreline erosion and organic and inorganic material generated within the reservoir by biological activity. In eutrophic waters the latter source can be quite significant. Fine material can be resuspended repeatedly by currents and wave action until it eventually is deposited in an area where water movements are insufficient to resuspend or remobilize it.

3.3 Stages in Reservoir Sedimentation

Most natural river reaches are roughly balanced with respect to sediment inflow and outflow. Sediment may accumulate temporarily in some channel reaches but is mobilized and transported downstream by large flow events. No river reach is ever completely balanced with respect to sediment. Some reaches may experience long-term cycles of aggradation (increase in land elevation due to the deposition of sediment) and degradation (decrease in channel elevation due to erosion of the streambed) over time scales of centuries. Overall, the total amount of sediment transported through a reach is much larger than the rate of aggradation or degradation within the reach.

Figure 3.2. Sedimentation delta formed in the upper end of Sardis Lake reservoir, Mississippi. This reservoir impounds 32,500 ac at normal pool and is over 75 years old. Photo credit: Google Earth.

Dam construction dramatically alters this balance, converting the flowing stream into a pool characterized by low velocity and efficient sediment trapping. Coarse bed material loads are deposited as soon as stream velocity diminishes as a result of backwater from the dam, creating delta deposits at points of tributary inflow (Figure 3.2). Fine sediment is carried farther into the reservoir and accumulates downstream of the delta deposits. These fine sediments are deposited across the width of the reservoir. The slowed flows lose the ability to transport sediment, but wave energy can remobilize sediment or erode reservoir shorelines. Because reservoirs, with frequently changing water levels, rarely establish littoral zones stabilized by aquatic or terrestrial vegetation, their shorelines are especially vulnerable to erosion and transport of sediment. Over long periods, the impounded reach will accumulate sediment and lose storage capacity until a sediment balance is again achieved. This would normally occur after the impoundment has become filled with sediment and can no longer provide water storage and lacustrine benefits.

In advanced stages of sedimentation, the reservoir, or embayments within the reservoir, transition from continuous deposition to a mixed regime of deposition and scour, and the rates of sediment deposition are reduced compared with earlier stages when sediment trapping was continuous. In wide reservoirs, this stage is also characterized by the transition of sediment deposits from horizontal beds to a channel–floodplain configuration. This transition will occur naturally when sedimentation reaches the spillway crest; a main channel will be maintained by scour, and its base level will be established by the spillway. Sediment deposition continues on floodplain areas on either side of the channel, causing the floodplain elevation to rise above the spillway elevation (Figure 3.1). In narrow reaches, the scour channel may occupy the entire reservoir width and the floodplain may be absent.

Some older reservoirs have received substantial loads of sediment and are beginning to show advanced stages of sedimentation in their upper end and entrance to embayments. One example is Lake Texoma reservoir, Oklahoma–Texas. Patton and Lyday (2008) reported that extensive sedimentation and aggradation of sediment above water level effectively reduced reservoir area and led to embayment isolation, fragmentation of lacustrine habitats, morphometric changes, and establishment of terrestrial vegetation on newly emerged lands (Figure 3.3). Sedimentation led to the development of linear bars of deposition above normal pool elevation that have blocked mouths of embayments, bisected large areas of the reservoir, and fragmented several pools. Sedimented areas exhibited lower gradients and reduced habitat heterogeneity. The shorelines affected by sedimentation were homogenized and of low quality for all but a few species of fish, and their shallow nature made them especially susceptible to drying from even minor water-level reductions. Fish assemblages in isolated reservoir fragments appeared to be distinct from fish assemblages in nonfragmented habitats.

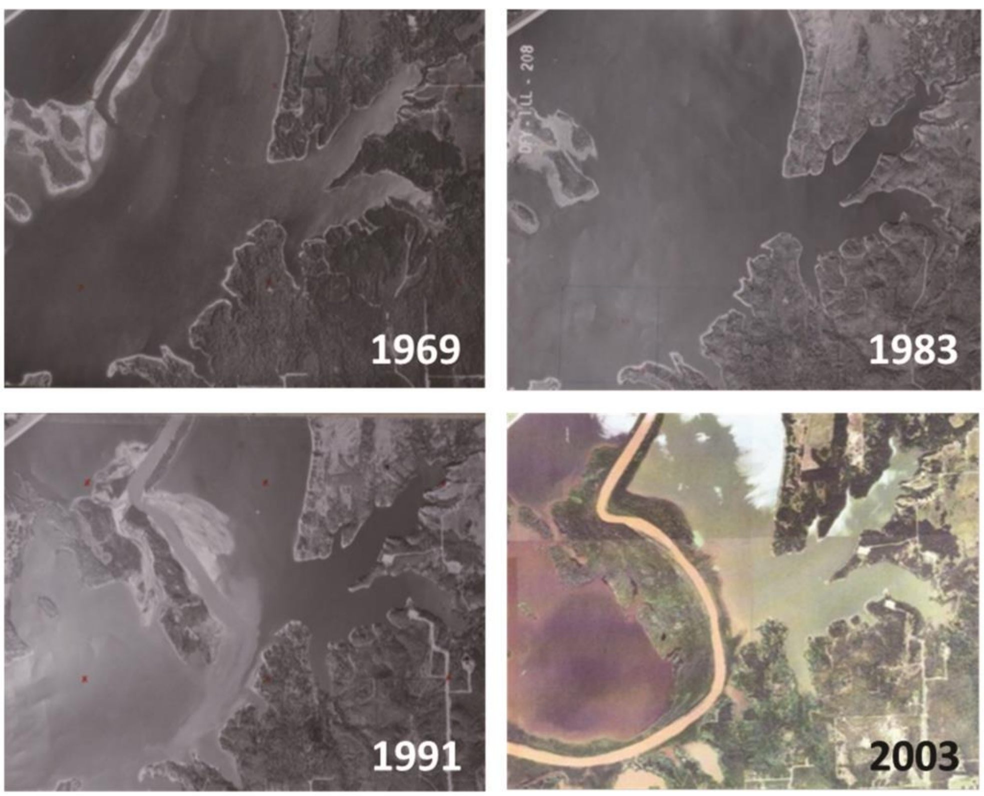


Figure 3.3. Time series photo of Widow Moore embayment (on the right of each photo) in Lake Texoma, Texas–Oklahoma. The time series begins in 1969, 25 years after impoundment. Note delta beginning to become visible in 1983, embayment beginning to become isolated in 1991, and embayment completely cut off from main body of Lake Texoma at normal pool in 2003. Photo credit: J. Boxrucker, Reservoir Fish Habitat Partnership.

This change in community structure conceivably was driven by a reduction of pelagic species from fragmented sites, as these sites had limited or no connectivity to the main body of the reservoir.

3.4 Rates of Sedimentation

Rates of reservoir sedimentation vary according to location of a reservoir within an impounded river basin. Upstream locations that are the outlets of small watersheds have different sediment yields than do downstream locations that are the outlets of larger watersheds. The sediment delivery ratio is the ratio between the amount of sediment produced from the surfaces in a watershed and the amount yielded at its outflow point (Neuendorf et al. 2005). This delivery ratio becomes progressively smaller for increasingly large watersheds. Generally, with all other factors being equal, the larger the watershed, the greater the internal storage of sediment (Graf et al. 2010). Thus, downstream reservoirs receive proportionally less sediment with respect to their drainage areas. This generalization does not hold true, however, if all other factors are not equal. For example, if major sediment-producing areas with highly erodible geologic materials are located low in the basin, low-basin reservoirs are likely to receive larger amounts of sediment. Variability in sediment contributions to downstream reservoirs is likely to result from regional climatic, geological, and land-use distribution.

Storms deliver large amounts of water to a river and downstream reservoir and potentially large loads of sediment. Fast-moving and high-flowing water can pick up, suspend, and move larger particles more easily than slow-moving normal flows (Olive and Rieger 1985). In fact, so much sediment can be carried during large storms that a high percentage of all the sediment moved during a year might be transported during a single storm period.

Data for rates of sedimentation in reservoirs of the Missouri River system exemplify the influence of scale and location in a drainage basin. The U.S. Army Corps of Engineers (USACE) has constructed six main-stem reservoirs and 16 additional reservoirs on tributaries. Sediment surveys have been conducted to define annual rates of storage loss through sedimentation for many of these reservoirs. The reservoirs with the highest rates of sedimentation were in small tributary streams, particularly in the Kansas and Osage river basins, where they receive runoff from drainage areas with high sediment yields. There are exceptions to this generalization, including the case of Lewis and Clark Lake reservoir on the Missouri River, which has lost more than 20% of its storage capacity resulting from sedimentation. This reservoir is at the downstream end of a series of six large reservoirs, so it receives little sediment influx from the Missouri River. Its primary source of sediment is the Niobrara River, a direct tributary to the reservoir. A similar example is Elephant Butte Reservoir on the Rio Grande in New Mexico. Although it is on the main-stem river, downstream from several other reservoirs, it has a very high sedimentation rate because it collects sediment from highly erodible basins drained by the Rio Puerco and Rio Salado rivers (Scurlock 1998).

The amount of sediment entering reservoirs responds to controlling factors that change over time as well as space. The primary controls on temporal change in reservoir sedimentation rates include climate, land use, geologic materials, fluvial system operation, and minor influences that may be locally or temporarily important. Geographical climate variation is especially important in the interior USA. Tucker et al. (2006) demonstrated that on the Great Plains the episodic timing of erosion and sediment yield were caused by climate oscillations between drought and wet decades. Based on nationwide data available in the Reservoir Sedimentation Survey Information System database, Graf et al. (2010) mapped mean annual loss of reservoir capacity through sedimentation. Mean annual loss ranged from <0.4% to >2%. A distinct national pattern was evident (Figure 3.4). The lowest sedimentation rates occurred in reservoirs of the Northeast and Tennessee Valley. The highest rates occurred in arid portions of the Columbia, Lower Colorado, Missouri, and Rio Grande rivers. High rates of sedimentation were also observed in the Lower Mississippi River basin.

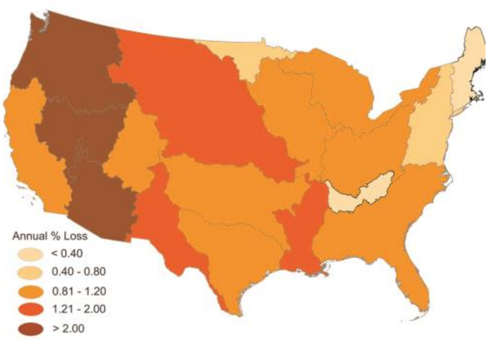


Figure 3.4. Distribution of mean annual loss of reservoir capacity in the continental USA according to hydrologic unit code (HUC)-2 units. Data from Ackerman et al. (2009).

No broad guidelines are available as to what are acceptable sedimentation rates. In Nebraska, the Department of Environmental Quality adopted methods to evaluate the severity of sedimentation in reservoirs (NDEQ 2008). This methodology uses the average annual loss of a reservoir's original conservation pool to index severity. Four volume-loss categories were defined for assessment purposes: substantial ≥0.75%, moderate ≥0.50% to <0.75%, slight ≥0.25% to <0.50%, and minimal <0.25%. These criteria are also used as the basis for placing reservoirs on Nebraska's Department of Environmental Quality Section 303(d) list for sedimentation. Any reservoir with average annual volume loss ≥0.75% falls into the "substantial" category and is placed on the Section 303(d) list. Although the volume loss cutoff of 0.75% is used to determine degradation, sedimentation goals for reservoir projects may need to be much more aggressive. Moreover, these guidelines may need to be modified geographically depending on local conditions.

3.5 Effects of Sedimentation

Bottom sediment is a critical component of reservoir systems. Sediment serves as habitat for benthic invertebrates (Peeters et al. 2004); can influence macrophyte distribution (Duarte and Kalff 1986); accumulates nutrients and regulate nutrient recycling rates (Søndergaard et al. 2003); controls concentrations of dissolved oxygen, hydrogen sulfide, and other constituents in bottom waters (Miranda et al. 2001; Reese et al. 2008); accumulates contaminants such as metals, pesticides, and other hydrophobic organic compounds (Karickhoff et al. 1979; Baudo et al. 1989); and provides a record of past conditions in the reservoir and its watershed (Wren et al. 2008). Sedimentation can clog interstices in substrate, thus reducing sediment–water exchanges and oxygen penetration and altering biogeochemical and microbial processes (Rehg et al. 2005; Nogaro et al. 2006). A large fraction of the nutrients deposited into a reservoir are stored within layers of sediment attached to organic and clay particles (Nürnberg 1988). Deposition of phosphorus bound to clays can play a large role in a reservoir's oxygen budget, particularly after the reservoir has lost depth. The interactions among the mineral properties of sediment and water chemistry determine whether sediment becomes a source or a sink of nutrients or other contaminants. Periodic anoxia in the hypolimnion can, for example, result in desorption of nutrients or other contaminants from sediment into the water column (Søndergaard et al. 2001; section 6). Physical and chemical interactions between water and influent sediment can, therefore, play an important role in determining the outcome of the effects of increased sediment loading on lake ecosystems (Nürnberg 1988).

Benthic invertebrates play an important role in the food web of many reservoirs and in recycling of materials (Underwood 1991). Increased sedimentation, by increasing inorganic turbidity of the water column and rates of sedimentation of inorganic particles into the reservoir basin, has a number of direct and indirect effects on benthic invertebrates, including reduced feeding and growth rates and increased mortality (Donohue and Irvine 2003). Sediment loading tends to reduce the abundance of benthic invertebrates (Donohue and Irvine 2004). Moreover, alterations to benthic invertebrate taxonomic composition can occur (Carew et al. 2007). These alterations frequently include reductions in species richness resulting from increased homogeneity of substrates. Input of fine sediment has been reported to be more detrimental to benthic invertebrates than coarse sediment because it is more likely to clog interstices and reduce oxygen penetration.

Spawning habitat of substrate-spawning fish is smothered by sedimentation (Muncy et al. 1979). If sediment blankets the substrate after spawning, oxygen supply to eggs and sac fry is decreased because of reductions in water circulation (Waters 1995; Argent and Flebbe 1999). Consequently, sedimentation decreases available spawning habitat, reduces spawning activity, and increases egg and larval mortality

(Alabaster and Lloyd 1982; Ryan 1991). Reproductive strategies that involve parental care, such as fin fanning and egg nipping and mouthing, appear to be more successful in habitats with intermediate levels of sediment (Berkman and Rabeni 1987).

In areas where sedimentation continues unabated, shallow aquatic habitats can transition relatively quickly to wetlands and eventually to uplands because of continued sediment deposition above the normal pool elevation during flood flows. Sedimentation of the littoral zone rather than the profundal zone, along with shore erosion, and reduced connectivity to embayment habitat through mouth sedimentation are likely to have the biggest effect on reservoir fish communities. Barren, homogeneous, windswept littoral areas are poor food producers, unsuitable habitat for nest builders, and poor refuges for littoral juvenile fishes. As the bank and littoral habitats degrade through sedimentation and erosion, and environmental conditions or reservoir operation prevent establishment of aquatic or wetland macrophytes, the density of fish that rely on the littoral zone during all or part of their ontogeny decreases. In such reservoirs, the fish community shifts toward dominance by species that can occupy pelagic niches and thus do not rely on substrates or substrate-based resources. Erosion and ensuing sedimentation and shallowing of reservoirs not only have been linked to reductions in benthic production but also to reductions in plankton production through reduced water clarity. In advanced stages of sedimentation, fish communities may consist of species that thrive in turbid, shallow systems with low oxygen and large fluctuations in temperature.

3.6 Monitoring Sedimentation

Various methods have been developed for estimating thickness of accumulated sediment in a reservoir. Three techniques are described here, including sediment cores, topographic contrast, and acoustic estimation. Each technique has limitations and strengths. Ideally, all three approaches may be applied concurrently to get a more complete representation and estimation of sediment thickness and distribution.

3.6.1 Sediment Cores

Cores typically are taken from a boat by means of a gravity corer or vibrational coring system. In either case, aluminum, plastic, or steel tubes are forced into the sediment, ideally until pre-impoundment substrate is reached. The tube is withdrawn and sliced longitudinally, or the sample is carefully removed from the tube, allowing for the measurement of sediment thickness and sample collection. The interface between pre-impoundment substrate and post-impoundment sediment is usually fairly distinct. Several companies manufacture sediment coring systems, including some small systems suitable for use in small boats in reservoirs (e.g., VibeCore Specialty Devices

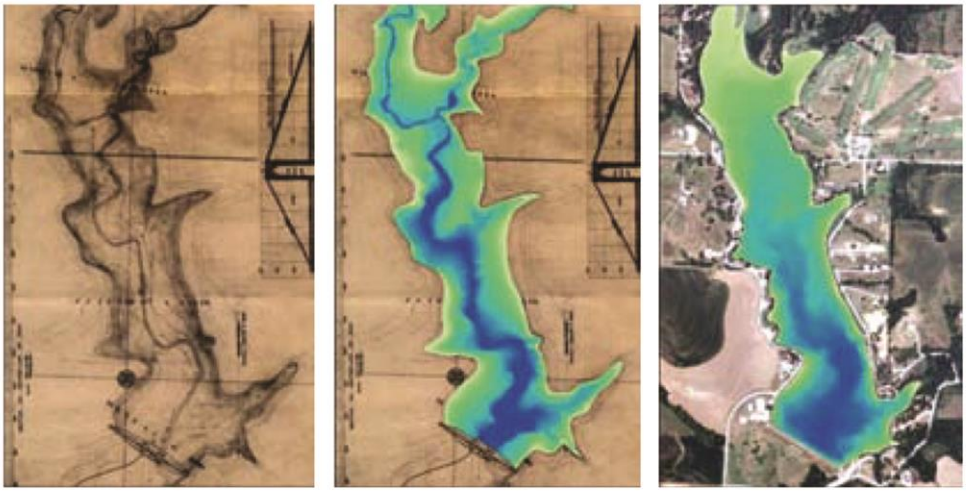


Figure 3.5. Topography of Mission Lake, Kansas. Left = 1923 engineering map; center = digital elevation model created from 1923 map; right = current bottom topography created from acoustic echo sounder data. Images credit: Kansas Biological Survey, Lawrence.

Inc., Wylie, Texas). A benefit of sediment cores is that they can be preserved and analyzed for sediment classification and chemical composition. However, core sampling and analysis is time and labor intensive.

3.6.2 Topography Contrast

This approach computes the difference between pre-impoundment and current bottom topography and creates a spatial representation of sedimentation (Figure 3.5). Data from pre-impoundment topographic surveys or reservoir blueprints are used to re-create a pre-impoundment surface, and data from recent bathymetric surveys are used to create a map of current reservoir bottom topography. Unlike spot sediment cores, topography contrast can represent sediment accumulations throughout a reservoir, facilitating estimates of sediment distribution and volume. The quality of data produced by this approach depends on quality of the pre-impoundment maps.

3.6.3 Acoustic Estimation

High-frequency and low-frequency transducers are operated simultaneously during a survey conducted from a boat (Anderson and Pacheco 2011). Differencing acoustic returns from high and low frequencies (reflecting off the current reservoir bottom and the pre-impoundment bottom, respectively) have shown promise for successfully mapping sediment thickness in inland reservoirs. Mapping the base of sediment acoustically may work best in reservoirs dominated by fine sediment (clay and silt rather than silt and sand), as illustrated in Figure 3.6. Reservoirs with fine-grain

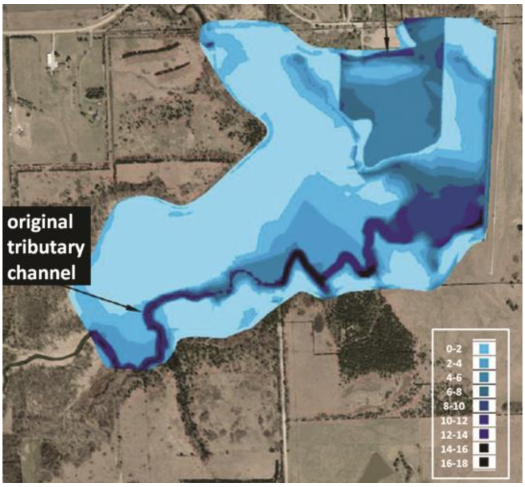


Figure 3.6. Acoustic survey of Conestoga Reservoir, Nebraska. Map shows sediment thickness over the reservoir and illustrates how sediment accumulation can occur throughout the reservoir rather than mostly near tributaries. Inset of sediment depth scale given in feet. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

deposition do not form significant deltas at tributary inlets. Coarse-grain-dominated reservoirs fill from the tributaries toward the dam and form deltas in their tributaries; therefore sediment may be surfacing and difficult to map unless water level is raised. In these cases the topography contrast method may be more appropriate.

3.7 Managing Sedimentation

Reservoir sedimentation management strategies can include one or more of the following techniques (Palmieri et al. 2003; Morris 2015): reducing sediment inflows, managing sediment once in the reservoir, and removing sediment accumulated in the reservoir. Successful sedimentation management may employ a combination of strategies, which may change over time as sedimentation becomes more advanced (Morris 2015).

The solution to external sedimentation problems is to control soil erosion in the watershed (section 2). However, controlling all soil erosion is not possible. Conservation farming practices significantly reduce amounts of sediment produced, although the sediment that is produced is of smaller particle size, which can efficiently carry some nutrients and chemicals. Additionally, streambank erosion is a source of sediment that is not easily controlled. Western parts of the USA also experience high rates of "geologic" erosion on lands that are not cultivated or disturbed by human activities. The Badlands of South Dakota are an example of very high natural geologic, or background, erosion rates.

Current management efforts focus on reducing sediment inputs from the watershed, streambanks, and streambeds. Much of the erosion from tillage practices has been greatly reduced since the 1970s, but there is a large amount of "legacy" material that has been deposited into stream channels and is now the primary source of sediment entering reservoirs in parts of the country, particularly the Midwest. It is also necessary to manage sediment already deposited in reservoirs. Reducing sediment will extend the useful life of reservoirs, reduce the amount of nutrients entering reservoirs, and improve water clarity and quality.

3.7.1 Reduction of Sediment inflows

Methods applicable to the watershed to control sediment before water enters the reservoir include watershed management (section 2) and channel structures such as sediment basins and dikes.

3.7.1.1 Sediment basins

A sediment basin (also referred to as a sediment trap, check dam, or detention basin) is an earthen or rock embankment suitably located to capture runoff and filter out sediment before they reach the reservoir (Figure 3.7). These basins alter the passage of flood waves, interrupt longitudinal movement of sediment, slow down turbulent flows into flows having lower energies, and may remove the majority of dense sediment within the water by settling (Boix-Fayos et al. 2008). Sediment basins are designed to provide an area for runoff to pool and settle out a portion of the sediment. Trapping efficiency is a function of sediment type and the ability of the basin to reduce the transport energy of the flows.

Basins may be dry or wet. Dry basins are empty most of the time but hold water for a few days during storm events. Detaining water for a few days allows settling of most of the sediment load (Cooke et al. 2005). Wet ponds retain a low volume most of the time and are often able to remove nutrients in addition to sediment. Because a permanent pool of water remains in a wet detention pond between storm events, microorganisms and algae flourish and provide additional removal of dissolved pollutants beyond that accomplished by sediment trapping (Harrell and Ranjithan 2003). Wet basins also can provide substantial aesthetic and recreational value and fish and wildlife habitat.

Figure 3.7. Sediment basin constructed above Wehrspann Reservoir, Nebraska, to trap sediment and improve water clarity. Note clear-water reservoir in background. Photo credit: Nebraska Game and Parks Commission, Lincoln.

Sizing of the basin relative to the watershed is important. A ratio of pond volume to mean storm runoff volume of 2.5 potentially can remove about 75% of suspended solids and 50% of total phosphorus (Schueler 1987). The National Urban Runoff Program (Athayde et al. 1983) recommended a wet pond with a surface outlet, a mean depth of 3 ft, and a surface area ≥ 1% of the watershed area. Urban wet detention ponds sized at 1% of runoff area had removal of solids up to 70% and of total phosphorus up to 45% (Wu et al. 1996). A surface overflow outlet improves sediment capture as the outlet removes the cleanest water. A chain of ponds, emphasizing biological removal of nutrients in the terminal pond, was recommended by Walker (1987). All ponds may require a dense perimeter of bank vegetation to provide protection from shoreline erosion. One problem in sizing ponds is the "short-circuiting" that occurs when storm water passes through the pond with little or no displacement of pond water (Horner 1995). A minimum length to width of 3:1 may eliminate this problem (Schueler 1987), but topography may prevent this design, forcing the use of groynes in the pond to divert inflowing water into the entire pond. Specific guidelines for sizing are usually available from regional natural resources conservation services or engineering offices.

All sediment basins require regular maintenance to remove sediment and trapped debris. When possible, basins may be designed to be drained for excavation, which is less expensive than dredging. Techniques to make sediment removal easier are to construct an accessible forebay that retains the largest particles, build a ramp for small-dredge access, and establish a watershed area for sediment disposal (Schueler 1987). Mowing may also be necessary to limit woody vegetation growth.

The size of the sediment basin depends on the size of the watershed, extent and composition of sediment runoff, amount of precipitation, and efficiency of sedimentation of a target sediment grain size. Various rules of thumb are used, but generally the amount of precipitation is taken as the largest 1-year, 24-h precipitation event. Coarse-to-medium size silt particles will settle out quickly whereas finer particles (i.e., clay and fine silt) will require a long time to settle. Thus, sedimentation basins tend to remove a high percentage of coarse-to-medium particles but a small percentage of clay and fine silt particles unless the size ratio of the sedimentation basin to watershed is increased. Pairing sediment basins with wetland cells to trap and treat the nutrients associated with the smaller particles can be an effective practice.

3.7.1.2 Sediment dikes

Installed at the upper reaches of reservoirs to form marshes, dikes (also reported as subimpoundments) trap sediment and agricultural runoff as water enters the reservoir (Figure 3.8). These structures slow water velocity during runoff events, allowing sediment to settle out before the water reaches the main reservoir. The water behind a dike develops an extensive wetland that increases filtration of sediment while

Figure 3.8. A sediment dike (subimpoundment) constructed at the upper end of a Nebraska reservoir. Also shown are groynes and jetties constructed to reduce wind-induced wave action that erodes shorelines. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

providing expanded habitat to fish species adapted to wetlands, as well as to shorebirds, waterfowl and furbearers. Sediment dikes and sediment basins serve the same purpose. Selection of one over the other often depends on site availability and ownership, access, hydrology, and potential for value added in terms of providing additional habitat for fish and wildlife.

These structures usually retain water above the normal operating level during reservoir drawdown periods, thereby creating small ponds or lakes. Besides trapping sediment, subimpoundments can contribute to natural resource management through the development and maintenance of wildlife habitats, wetlands, and dispersed recreation. While reservoir control authorities generally do not encourage the construction of subimpoundments, they will consider proposals from government agencies to provide public benefit. The Tennessee Valley Authority, for example, has worked with state and federal agencies to develop subimpoundments for natural resource management, such as those that create or enhance wildlife habitats.

A frequently used design is a notched, low-profile sediment dike spanning the width of a reservoir's headwaters. Although the dike converts a portion of the reservoir from open water into a large sediment basin and wetland, the benefits to the reservoir can be substantial. Because the dike is notched, water levels behind the structure are maintained at normal lake elevation unless a high runoff event occurs. This means practically no flood storage loss occurs. However, to maintain dike function, sediment trapped in the areas isolated by sediment dikes may have to be dredged periodically.

3.7.1.3 Bypass channel

When topographic conditions are favorable, a large-capacity channel can be constructed to bypass sediment-laden flow around a reservoir. By routing the sediment around the reservoir into the tailwater, sediment accumulation of bedload and suspended load is reduced. However, transport capacity in bypass channels may be limited for coarse sediment loads. Construction of bypass channels has been restrained by the high cost of construction and maintenance. Such channels may eliminate the need to construct and maintain a large-capacity spillway at the main dam because flood flow is diverted, possibly compensating for the high cost of building a channel.

3.7.2 Sediment Management in the Reservoir

Techniques for preventing sediment from settling once water enters the reservoir include sluicing, density current venting, and bypassing the reservoir via a channel (section 3.7.1.3). A major disadvantage of these techniques is that a substantial volume of water must be released to transport sediment. Therefore, they may be most applicable in reservoirs for which the water discharged by large sediment-transporting floods exceeds reservoir capacity, making water available for sediment release without infringing on uses. Moreover, these techniques may not be able to remove previously deposited sediment or pass the coarsest part of the inflowing load beyond the dam. Reservoirs constructed and operated as managed systems for water supply, irrigation, hydropower generation, or flood water capture may not be able to use these techniques and have to either reduce sediment inflows or remove sediment after deposition.

3.7.2.1 Sluicing

Figure 3.9. Sediment sluicing through a dam in the Yellow River, China. Photo credit: china.org.cn.

Sluicing is an operational technique in which a substantial portion of the incoming sediment load moves through the reservoir and dam before sediment particles can settle (Figure 3.9), reducing the reservoir's trap efficiency (ICOLD 1989; Morris and Fan 1998). In most cases, sluicing is accomplished by operating the reservoir at a lower level during the flood season to maintain higher flow velocity and sufficient sediment transport capacity of water flowing through the reservoir. Increased sediment transport capacity reduces the volume of deposited sediment. After flood season, the pool level in the reservoir is raised to store clearer water. Effectiveness of sluicing operations depends on availability of excess runoff, size of sediment, reservoir purpose, and reservoir morphology.

3.7.2.2 Density current venting

Density currents occur because the density of sediment-carrying water flowing into a reservoir may be greater than the density of clearer water already in a reservoir. The increased density, increased viscosity, and concomitant reduction in turbulence intensity result in a uniform current with high sediment concentration that dives underneath the clear water as it moves toward the dam. As the current travels downstream, it will generally deposit the coarser part of its sediment load along the bottom, and if enough load is deposited, the density current will dissipate along the way to the dam. If the current reaches the dam, it will accumulate to form a submerged "muddy lake," and the turbid water reaching the dam can be vented through low-level outlets.

Several observable phenomena indicate the presence of turbid density currents in a reservoir. A muddy flow that enters and disappears at the upstream limit of a reservoir, a phenomenon frequently observed from the air, is an indication of plunging flow. The plunge line may be observed as a sharp transition between clear and turbid water and by the accumulation of floating debris. Continuous turbidity monitoring immediately above the reservoir and at the dam can indicate the presence of turbidity currents and also establish their travel time to the dam. Bottom water can be discharged continuously through a low-level outlet and monitored below the dam to observe the arrival of turbid water. The presence of density currents also may be measured by sonic or other velocity-profiling methods or by monitoring of water-quality variables such as temperature and dissolved oxygen, which distinguish the inflowing and impounded waters.

In reservoirs with known density currents, installation and operation of low-level gates allows sediment currents to pass through the dam for downstream discharge. Density current venting is an attractive option because, unlike flushing operations, it does not require lowering the reservoir level. This approach results in increased downstream sediment loads that can aggrade stream habitats or possibly enhance sediment-starved reaches below the dam.

3.7.3 Removal of Sediment from the Reservoir

Techniques for removing sediment once it has accumulated in a reservoir include mechanical removal (e.g., excavation, dredging, and hydrosuction), consolidation, flushing, and aeration. The best-suited application will depend on the ability to manage reservoir water levels. In many situations, lowering water levels to remove sediment is desirable because removal options are usually less expensive, sediment can be removed in a manner that creates habitat features that are beneficial for fish habitat, and sediment can be placed at specific spoil locations. Excavating sediment from existing reservoirs to return to preconstruction volumes can require moving more material than originally was moved to construct the dam embankment and can be cost prohibitive. Also, significant control of water levels at the appropriate time for removal or flushing may not be practical. Choosing the best option for a reservoir may include a combination of techniques applied to different locations. A number of environmental concerns are associated with sediment removal. Sediment removal rarely will be cost effective where high sedimentation rates prevail.

3.7.3.1 Excavation

Figure 3.10. Excavation of accumulated sediment at a Nebraska reservoir. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

Excavation requires temporarily lowering the reservoir water levels, working within seasonal drawdowns, or working when reduced river flows can be controlled adequately without interfering with excavation work. Large-scale removal of sediment is possible with commonly used earth-moving equipment and in a manner that benefits fish habitat through the creation of ledges, trenches, and drop-offs (Figure 3.10). These irregularities in a reservoir's basin attract and concentrate fish. In Nebraska, excavation of sediment averaged US$4–5/yd^3 in 2014, although the cost went as high as $8. (M. Porath, Nebraska Game and Parks Commission, personal communication). This cost normally includes excavating, hauling to a designated spot (outside the basin) and dumping, and grading the dumped sediment. Distance hauled between the reservoir and spoil sites is the primary cost driver; therefore, nearby spoiling sites or spoiling within the basin (e.g., to create islands or other structures) may be a better option unless nutrient removal is also a goal. The cost cited usually does not include the cost to seed the spoil area or install and maintain soil erosion fencing (M. Porath, personal communication). Excavation and disposal costs can add up depending on the amount of sediment requiring excavation; therefore, this technique generally is used in relatively small reservoirs or in key embayments of large reservoirs.

Sediment spoil can be used to rebuild shorelines and create islands, if the sediment is not excessively nutrient rich and if it meets compaction standards for building in-lake structures. Shoreline erosion caused by wave action gradually enlarges reservoirs. However, the extra water created is shallow, absorbs the power of crashing waves, and supports few fish. By removing silt from areas close to shore and depositing it at the existing bank, deeper water more suitable for supporting fish can be created within casting distance of the bank. Sediment spoils excavated from the basin also can be relocated within the basin as islands if sediment is functional as per caveats listed. Creating islands creates more shoreline. Islands often are stabilized with rock riprap or other revetment materials to prevent bank erosion.

Depending on location, excavation can be costlier than dredging, but the comparative economics will depend on the job characteristics (Morris and Fan 1998). Excavation of dewatered sediment by means of heavy equipment and trucks eliminates the problem of dewatering the slurry generated by dredging, reduces sediment bulking compared with dredging, and can be used to deliver sediment to many small and widely dispersed containment or reuse sites. Environmental permitting requirements for excavation also may be simpler than those for dredging. Unit costs will vary widely and are region specific, depending on the volume of material, haul distance, and elevation difference between the points of excavation and disposal; the costs listed earlier for Nebraska are probably in the low end of the scale.

3.7.3.2 Dredging

The process of excavating deposited sediment from under water is termed dredging. This is a highly specialized activity used mostly for clearing navigation channels in ports, rivers, and estuaries. Dredging also can be used to reclaim reservoir storage capacity lost to sediment deposition and to open channels to restore connectivity with backwaters (section 9). However, dredging is often more expensive than excavation because of the amount of extra handling needed to move similar amounts of material. According to Hargrove et al. (2010) the cost of dredging ranges from \$2.50 to \$14.00/yd^3. To put this into context, the 2010 cost for removing sediment from a 7,000-ac reservoir nearly filled with sediment would be about \$1 billion (Hargrove et al. 2010). In addition, it would be necessary to clear and grub, haul, grade, and stabilize the spoil location. Ideally, a disposal location for the excavated material can be found close to the reservoir to reduce transportation costs. Costs associated with dredging are expected to vary widely geographically depending on a variety of local factors.

Smith et al. (2013) estimated the economics of dredging Tuttle Creek Lake, Kansas, versus implementation of cropland management strategies to reduce sediment runoff. They found that in this watershed if the marginal costs of agricultural best management practices (BMPs) implementation exceeded \$6.90/t of sediment reduction, then dredging would become the economically preferred alternative. Meeting this cost required that BMPs in the form of filter strips and no-till cultivation were implemented in a targeted, cost-effective manner, not in the random pattern of voluntary adoption that characterizes BMPs adoption in some watersheds. Although reservoir dredging is clearly expensive, Smith et al. (2013) showed that it is not entirely cost prohibitive on an annualized per unit basis.

There are two basic types of dredging equipment, mechanical and hydraulic. Mechanical dredges typically include backhoes, clamshells, and draglines (Figure 3.11). Mechanical dredges are capable of dredging soft and hard-packed material and also have the ability to remove debris. For the most part, these types of dredges can work in relatively tight areas and are efficient for side casting material from a dredge cut to a placement site or barge next to the dredging site. As compared with hydraulic dredging (see below), mechanical dredging does not have the issue of having to manage return water, but retaining fine or loose material in conventional buckets is difficult. Mechanical dredging can take longer to remove sediment when compared with hydraulic dredging, depending on the distance to the spoil location. Mechanical dredging is less efficient than hydraulic dredging when transporting material over long haul distances (>2 mi) and in areas that contain restricted width access points when barges are used to transport the dredged material.

Figure 3.11. Backhoe (top) and clamshell (bottom) mechanical dredges. Photo credit: Dredge Source, Kansas City.

Unlike mechanical dredging, hydraulic dredging allows almost continuous pumping, which results in faster completion than mechanical dredging. This method is very cost effective if the pumping site is within <2 mi from the disposal site. However, the dredging slurry is 80%–90% water and 10%–20% sediment, which can cause difficulties in obtaining and administering a water-quality permit. There are various types of hydraulic dredge heads that dislodge sediment to be pumped. Conical rotating heads mounted on a movable boom are used for soft sediment whereas cutter-style heads are likely to have viable applications in shallow backwater areas in reservoirs (Figure 3.12). Cutterhead pipeline dredges are sized based on the discharge pipe inside diameter and are typically available from 8 to 20 in with larger applications reaching 36 in. Cutterhead pipeline dredges are capable of excavating most types of material and can even dredge some

Figure 3.12. Hydraulic dredging at Decatur Lake, Illinois, part of a $91 million reservoir restoration. Photo credit: M. Honnold, Macon County, Illinois.

rock without blasting. Working depth is dictated by length of boom and water depth during operation, which limits the applicability of this technique.

Figure 3.13. Floating amphibious excavator dredger. Suitable for shallow applications that need to dig into a liquid environment where a traditional excavator cannot work because of the inconsistent substrate. Photo credit: B. R. T. Impianti.

A floating amphibious excavator is a mechanical excavator with an undercarriage that gives the excavator a very low ground pressure (Figure 3.13). This low ground pressure allows the excavator to work in marsh- and wetland-type environments where a normal excavator or typical dredge cannot reach. Floating excavators are ideal for those hard-to-reach places and are also highly mobile. However, they are not as efficient as mechanical or hydraulic dredges.

High-solids dredging, also known as Dry DREdge™, uses mechanical dredging to produce a slurry that is 50%–80% solids, thus resulting in a relatively clean effluent. This technique can be used to fill geotextile containers (e.g., geotubes), which can be used, in turn, to build the outer ring of islands (Figure 3.14) or restore eroded bank lines. High-solids dredging is one of the only techniques suitable for building islands out of a highly silty material.

Figure 3.14. The Dry DREdge™ lifts sediment from the reservoir bottom and uses a displacement pump to move it without adding water. The pump fills geotubes with the sediment and then pumps sediment with the consistency of toothpaste behind the tubes to form a small island. Photo credit: J. C. Marlin, Illinois Department of Natural Resources.

Disposing of dredged material can cause expensive environmental problems, and solutions have to be developed on a case-by-case basis (Skogerboe et al. 1987). Discharging high sediment concentrations generally associated with dredging slurry directly downstream from the dam can be environmentally unacceptable. Other than the destructive effects caused by excessive sediment downstream, sediment can contain trace metals and hazardous chemicals that make disposal problematic. Contaminants of concern include arsenic, chromium, copper, lead, nickel, zinc, cadmium, and mercury (Hargrove et al. 2010). In some limited cases, it might be possible to reduce the sediment concentration of dredge slurry discharged below a dam by concurrently releasing water from the reservoir. If dredged materials are disposed on land, consolidation with fly ash might be required. Although dredged material often can be a liability, in some cases it can be an asset (WOTS 2004). Uses for uncontaminated dredged sediment include habitat development, soil improvement for agriculture and forestry, and construction (e.g., brick making).

Dredging does not have to occur over the entire basin of most reservoirs. Tactical dredging of upper ends of the reservoir or major embayments removes sediment from where it is accumulating most rapidly and affecting the largest portion of the biota. Excavating upper basins deeper than their original contour creates settling basins that serve as sediment traps. Preserving an infrastructure that allows access to these settling basins will allow convenient redredging every 20 to 30 years, a possible long-term management strategy.

Figure 3.15. Mini-dredge for small sediment removal jobs Photo credit: Piranha, Albuquerque, New Mexico.

3.7.3.3 Small-scale removals

Mini dredges are available in the market for small sediment removal jobs (Figure 3.15). These dredges are effective in removing sand, silt, and organic sediment accumulated next to shore, around docks or boat slips, or from small but critical aquatic habitats in backwaters. The excavation capability of these units is in the neighborhood of 350–1,500 ft^3/h depending upon the nature of the sediment, depth operated, and distance pumped.

3.7.3.4 Hydrosuction

A hydrosuction removal system is a variation of traditional hydraulic dredging. Traditional dredging uses pumps powered by electricity or diesel. Hydrosuction uses energy from the hydraulic head available at the dam (Hotchkiss and Huang 1995). One end of a pipeline is situated over sediment at the bottom of the reservoir. The pipeline then extends through the dam to a discharge point downstream. Hydrosuction dredging does not rely on external power pumps to transport sediment (but may for mobility) and therefore avoids various problems associated with those operations. Where sufficient head is available, operating costs for hydrosuction are substantially lower than for other types of dredging.

Hydrosuction is most effective for transporting fine, noncohesive, unconsolidated sediment that collects in areas adjacent to the dam or that can be reached easily by the pipe inlet. Heavier materials such as sand may be transported but only at the expense of head loss and a higher head requirement. In a Nebraska reservoir, hydrosuction initially was capable of removing sediment at the annual rate it entered the reservoir, but efficiency dropped by 50% where sand bedload was encountered (M. Porath, personal communication). Whether hydrosuction is feasible for managing sediment at a particular reservoir depends on hydraulic, environmental, and operational factors at the dam (Hotchkiss and Huang 1995).

3.7.3.5 Consolidation

Sediment consolidation offers a cheaper alternative to excavation or dredging and can alleviate some of the environmental obstacles associated with excavation and dredging. Nevertheless, the effectiveness of consolidation is highly dependent on reservoir and sediment characteristics (Smith et al. 1972). Consolidation refers to a gradual decrease in the water content of water-saturated soils, with an associated rearrangement of the soil structure and a reduction in volume. In the case of reservoir sediment, the most practical method of consolidation is lowering the water level and consequently the water table below the sediment surface. Exposure and desiccation of sediment for purposes of consolidation can increase depth, temporarily arrest resuspension potential, and reduce turbidity after the water level returns to normal.

The water content of the organic-rich sediment in eutrophic lakes frequently exceeds 90% on a volume basis; complete dewatering could decrease sediment thickness by a corresponding amount. The water content of inorganic sediment is usually considerably lower, but if an appreciable amount of organic sediment is present, consolidation will still occur. Complete removal of water and 100% consolidation is not normally possible. At Snake Lake, Wisconsin, as much as 3 ft of consolidation took place during a 10-ft drawdown (Born et al. 1973). At Lake Tohopekaliga, Florida, consolidation of flocculent organic sediment in the nearshore areas ranged from 55% to 100% during drawdown (Wegener and Holcomb 1972). In Nebraska reservoirs, drawdowns to dry out sediment often produce a 6–8-in shrinkage in silt (M. Porath, personal communication). The potential for sediment erosion may need to be evaluated before exposure.

According to Dunst et al. (1974) the effect of drawdown on the physical characteristics of flocculent sediment may produce a largely permanent rearrangement of the structure of sediment, and no appreciable reswelling can be expected after lake refilling. However, sediment exposure and desiccation may result in chemical changes within the sediment that may have an undesirable effect on nutrient levels in the lake after reflooding. Sediment dewatering is often accompanied by marked increases in nutrient releases, particularly of phosphorus, which may stimulate unwanted algal growth after reflooding.

Drawdown, or sediment consolidation, can be a feasible technique for the improvement of shallow lakes if a number of conditions are satisfied. These include suitable lake basin morphometry (shallow slope so that a small vertical decline in water level exposes a maximum lake bottom), aesthetic and economic acceptability during extent of drawdown, management of water input to maintain drawdown and perform refill, and appropriate sediment characteristics.

3.7.3.6 Flushing

Flushing is potentially a tool if water operations can be controlled and manipulated. Flushing increases flow velocities in a reservoir to the extent that deposited sediment is eroded and resuspended and transported through low-level outlets in the dam (Figure 3.16). Flushing occurs in two ways: complete drawdown flushing and partial drawdown flushing (Morris and Fan 1998). Complete drawdown flushing occurs if the reservoir is emptied during flood season; this creates river-like flow conditions in the reservoir. Deposited sediment may be remobilized and transported through low-level gates to the river reach downstream from the dam. Low-level gates are closed toward the end of flood season to capture clearer water for use during the dry season. Partial drawdown flushing occurs when the reservoir level is partially reduced. Sediment transport capacity in the reservoir increases only enough to allow sediment from upstream locations to move farther downstream, closer to the dam. Partial drawdown flushing can remove sediment from shallow portions of embayments

Figure 3.16. The sediment built up in the upper end of Lake Aldwell, impounded by the Elwha Dam, Washington, was exposed by a drawdown, eroded by the river, and flushed downstream. The dam was eventually removed, and the Elwha River flowed freely through the site by March 2012. Photo credit: B. Cluer.

and transport them to a deeper location, where future complete drawdown flushing may remove them from the reservoir. A flushing operation is enhanced if there is access to additional water stored in reservoirs upstream and if timed with major rain events.

Flushing reservoirs can have unwanted effects on the receiving stream. If the dam is deep, water in the lower levels is frequently deoxygenated. Flushing the dam releases this often cold and highly turbid water into the receiving stream and can bring about fish kills. Hesse and Newcomb (1982) flushed sediment out of Spencer Hydro in the Niobrara River, Nebraska. They reported various negative effects to the biota above and below the dam. They recommended that (1) flushing should not be implemented during the spawning period of fish, (2) refill of the reservoir should be done in a way that avoids dewatering downstream, and (3) a mitigation program should be implemented for fish losses.

A potentially effective means to remove deposits in the inflow to embayments may be flushing via auxiliary channels (Tolouie et al. 1993; Morris and Fan 1998). One or more channels excavated parallel to the main channel are eroded by diverting water from the tributary, thereby eroding sediment deeper into the embayment or the main reservoir. Because sediment deposits slope laterally, pilot excavation is required to "train" the desired course of a longitudinal channel and to maintain the desired horizontal distance between channels. Diverted flow enlarges the pilot channel until the entire flushing flow passes through the auxiliary channel at the highest flow rate possible, thereby maximizing channel width. A fully developed system would consist of a series of longitudinal channels, submerged during normal pool impounding and exposed during flushing. Once the channels have been scoured, they would be maintained by rotating the diversion flow through each channel at regular intervals, possibly on the order of once every several years.

Flushing may have limited management applications based on the authorized purposes of the impoundment and nature of the sediment. Knowing the energy, sediment composition and distribution, and proposed flushing regime are critical components in the planning process.

3.7.4 Environmental Concerns

There are various potential environmental problems associated with sediment removal (Peterson 1982). Most of these problems center on the resuspension of sediment during its removal, particularly during dredging, but there are also problems associated with sediment disposal (not discussed here, but see USACE 1987 a, b). One of the most common problems is the freeing of nutrients attached to resuspended fine sediment. Phosphorus is of particular concern because of its high concentration in interstitial waters in eutrophic reservoirs and its affinity for finely divided particulate material. Dredge agitation and wind action tend to move the disturbed nutrient-laden sediment into the euphotic zone of the reservoir, producing the potential for algal blooms. The reverse of increased algal production problems can also be triggered by the resuspension of sediment. Reduced light penetration resulting from turbidity will have a tendency to inhibit algal production. A potentially more serious problem associated with fine sediment in the water column is oxygen depletion. If the sediment is highly organic, the particles quickly become bacteria-coated. The tremendous surface area of these particles permits rapid decomposition and possibly oxygen depletion.

Another problem associated directly with resuspended sediment is the liberation of toxic substances. Silts and clays transport metals, phosphorus, chlorinated pesticides, and many industrial compounds such as polynuclear aromatic hydrocarbons, polychlorinated biphenyls, dioxins, and furans. The most significant chemical transformation processes in dredging plumes may be the releases of ferrous iron and sulfides from oxygen-depleted resuspended sediment and their subsequent oxidation by the dissolved oxygen in the aerated water column (Jones-Lee and Lee 2005). The oxidation of sulfides to sulfate and of ferrous iron to iron oxides or hydroxides are the primary chemical processes driving dissolved oxygen reductions. Heavy metals occur mostly as sulfides in anoxic sediment. Upon resuspension of anoxic sediment into the oxic conditions of the overlying water, iron and manganese are rapidly oxidized and precipitate from the water column, forming fresh sediment layers. Compared with the rapid oxidation of iron and manganese, the oxidation of heavy metal sulfides is much slower, so they may remain in the water column for hours. There, they are available to fish via gill uptake or ingestion with food.

A relatively common concern with dredging projects is the destruction of the benthic community. If the lake basin is dredged completely, 2 to 3 years may be required to reestablish the benthic fauna (Carline and Brynildson 1977). However, if portions of the bottom are left undredged the reestablishment may be relatively fast (Wilber and Clarke 2007). In any case, the effect on the benthic community appears to be of relatively short duration compared with the longer-term benefits derived from sediment removal.

Section 4

Eutrophication

4.1 Introduction

Eutrophication is the process of increasing nutrient enrichment, especially phosphorus and nitrogen, in which the enrichment leads to phytoplankton blooms and deterioration of water quality and causes changes to the ecosystem (NRC 2000). Eutrophication of a reservoir and the resulting increases in phytoplankton growth can have various direct effects on the quality of water within the impoundment. Eutrophication increases phytoplankton, zooplankton, bacteria, fungi, and detritus. Phytoplankton production in the reservoir shifts from green algae to cyanobacteria (Smith 1998), which is a less desirable condition because cyanobacteria can produce undesirable tastes and odors as well as produce toxins (Figure 4.1). Cyanobacteria can outcompete green algae under low nitrogen to phosphorus ratios because of their ability to fix atmospheric nitrogen, they have nitrogen-storing heterocysts that help maintain buoyancy and thereby shade out other genera, and their ability to proliferate in hot and stagnant water, as well as other advantages (Scheffer et al. 1997). However, cyanobacteria may be less available as food for certain organisms (e.g., cladocerans) because of their larger size; also they may not provide sufficient nutrition (Smith 1998). As cyanobacteria blooms subside, the dead and decaying cells can reduce oxygen levels in the water, causing stress or death to aquatic animals and potentially prolonging stratification. Dissolved oxygen concentrations in eutrophic waters become more variable, reaching higher highs and lower lows (Bouvy et al. 1999).

Figure 4.1. Klamath River as it flows into the Copco Lake reservoir, California, and mixes with a cyanobacteria bloom. Photo credit: U.S. Environmental Protection Agency.

Although phosphorus and nitrogen occur in many different chemical forms in aquatic environments, it is the dissolved inorganic forms that are most readily available for assimilation by phytoplankton. Analyses of the inorganic species, ammonium (NH_4) and nitrogen oxides (NO_x), give reliable estimates of bioavailable nitrogen. Bioavailable phosphorus is more difficult to measure because of its high affinity to particles and because it is frequently used as soon as it enters the water column by algae and plants. Bioavailability of phosphorus varies depending on the source. Gerdes and Kunst (1998) showed that 72% of the total phosphorus in effluent from sewage treatment plants was bioavailable, but only 30% of the total phosphorus in eroded material entering a river was available. They also showed that this percentage increased to 59% when the soils from which the material was sourced were fertilized, suggesting that fertilizer introduced significant amounts of bioavailable material into the runoff.

Well-defined relationships between phosphorus and phytoplankton biomass (i.e., chlorophyll-*a*) have been identified in reservoirs (e.g., Hoyer and Jones 1983; Jones and Knowlton 1993). Consequently, phosphorus often has been considered the primary nutrient limiting phytoplankton production in reservoirs, and management efforts to control eutrophication generally have emphasized control of phosphorus loadings (e.g., Dodd et al. 1988). However, Elser et al. (1990) reviewed various studies and reported that co-limitation by nitrogen and phosphorus was a common response of phytoplankton to nutrient additions. In Kansas reservoirs, Dzialowski et al. (2005) reported that the addition of phosphorus or nitrogen alone rarely increased phytoplankton growth. Instead, growth was co-limited by both nutrients. Generally, reservoirs that were nitrogen limited had total nitrogen to total phosphorus ratios (TN:TP) <18; reservoirs that were co-limited by nitrogen and phosphorus had TN:TP between 20 and 46; and reservoirs that were phosphorus limited had TN:TP >65. Overall, these results suggested that in Kansas reservoirs management efforts might need to focus on both nutrients (Dzialowski et al. 2005).

In reservoirs, eutrophication is accelerated by a large watershed-to-lake-area ratio (Wetzel 1990). Croplands in the watersheds of reservoirs are usually the biggest contributors to eutrophication. The proportion of cropland cover in the watersheds of 126 Missouri reservoirs accounted for 60%–70% of the variance in long-term averages of total phosphorus and total nitrogen (Jones et al. 2004). Even among reservoir watersheds with >80% grass (including pasture) and forest cover, cropland accounted for most of the variation in nutrients. Reservoir nutrients showed a strong negative relation to forest cover. Relations between grass cover and nutrients were positive but weak, and grass had no detectable effect once the effects of croplands was taken into account. In this set of Missouri reservoirs, urban reservoirs had about twice the nutrient levels as reservoirs in forest and grass watersheds.

4.2 Trophic State Indices

Indices of trophic state based on readily obtainable water-quality data are used to describe the trophic state of lakes but have been modified for use in reservoirs. Indices assign trophic states according to the phytoplankton biomass present during summer (indexed by chlorophyll-*a* biomass), the concentrations of key nutrients

Table 4.1. Trophic state classification based on total phosphorus, total nitrogen, chlorophyll-*a*, and Secchi depth visibility for lakes (Forsberg and Ryding 1980) and for reservoirs (Jones and Knowlton 1993) in parentheses.

Trophic state	Total phosphorus (ppb)		Total nitrogen (ppb)		Chlorophyll-*a* (ppb)		Secchi depth (ft)	
Oligotrophic	<15	(≤10)	<400	(<350)	<3	(<3)	>13	(≥8.5)
Mesotrophic	15–25	(>10–25)	400–600	(≥350–550)	3 –7	(≥3–9)	8–13	(≥4–8.5)
Eutrophic	>25–100	(>25–100)	>600–1500	(≥550–1200)	≥9–40	(≥9–40)	3–8	(≥1.5–4)
Hypereutrophic	>100	(>100)	>1500	(>1200)	>40	(>40)	<3	(<1.5)

(phosphorus and nitrogen), and water transparency as measured with a Secchi disk (Table 4.1). This trophic classification of reservoirs results from the division of a trophic continuum into categories called trophic states. The trophic state of reservoirs is indicative of their biological productivity, that is, the amount of living material supported within them, primarily in the form of phytoplankton. The least productive reservoirs are classified as oligotrophic. These are typically deep and clear and have relatively low nutrient concentrations. The most productive reservoirs are classified as hypereutrophic and are characterized by high nutrient concentrations and shallow depth, which result in phytoplankton growth, cloudy water, and low dissolved oxygen levels.

4.3 Effects on Fish

Early stages of eutrophication may enhance fish growth and biomass and seem to be desirable from a fisheries perspective (i.e., more nutrients = more fish). However, water-quality changes associated with higher trophic states (e.g., hypoxia, denser phytoplankton blooms, reduced water clarity, and altered fish fauna) usually argue against promoting higher trophic states because of changes in fish food habits, spatial distribution, and community composition. In fact, extreme cases of hypereutrophication promote dense, noxious phytoplankton blooms that can cause fish kills (Figure 4.2). Moreover, phytoplankton communities in eutrophic reservoirs can shift from domination by green algae to potentially noxious cyanobacteria. While this dominance may shift seasonally in many reservoirs, cyanobacteria tend to

Figure 4.2. Oxygen depletions and toxins associated with phytoplankton blooms in hypereutrophic lakes can cause fish kills. Photo illustrates a fish kill at Possum Kingdom Lake reservoir, Texas. Photo credit: NBC 5–KXAS, Dallas–Fort Worth.

dominate for an increasingly longer segment of the year in eutrophic and hypereutrophic reservoirs (Smith 1998) and are considered "sentinels" of eutrophication (Stockner et al. 2000). In turn, zooplankton composition is affected by phytoplankton availability because macrofiltrators (usually large-bodied zooplankton) that are more abundant in oligotrophic reservoirs give way to low-efficiency, small-bodied phytoplankton and bacterial feeders as nutrients increase (Taylor and Carter 1997). In hypereutrophic reservoirs, the food supply for zooplankton actually may decrease because of the dominance by cyanobacteria.

Eutrophication can lead to undesirable shifts in fish community composition. Although early stages of eutrophication may enhance fish growth and fishery yield, later stages may force changes in food habits, spatial distribution, and community composition (Larkin and Northcote 1969). In Florida lakes, fish biomass increased with eutrophication status to a maximum in mesotrophic lakes and fluctuated around the maximum value in hypereutrophic lakes (Kautz 1982). In contrast, fish density increased to a maximum in mesotrophic lakes but declined in hypereutrophic lakes. Piscivorous fishes reached maximum biomass and optimum densities in lakes with a total nitrogen concentration of 1,200 ppb and a chlorophyll-*a* concentration of 11 ppb but suffered adverse effects with further enrichment (Bachmann et al. 1996). Nevertheless, planktivorous (Yurk and Ney 1989; Bachmann et al. 1996) and benthivorous fish have been observed to increase with eutrophication status (Persson et al. 1991; Jeppesen et al. 2000).

Trophic state reportedly has a major influence on gizzard shad population characteristics in reservoirs (Power et al. 2004; Gonzalez et al. 2010). Gizzard shad represent an important prey species for many piscivorous fish. In oligomesotrophic reservoirs in Alabama, gizzard shad abundances were relatively low, yet these populations contained faster-growing fish (after age 1) and a higher proportion of older fish (DiCenzo et al. 1996). Furthermore, populations in less productive reservoirs contained fewer gizzard shad but a higher percentage of larger gizzard shad. In eutrophic reservoirs, gizzard shad were more abundant, and the population was characterized by smaller, slower-growing fish that were more vulnerable to predation. Consequently, gizzard shad were more available as forage in eutrophic reservoirs because of their greater abundance, smaller size, and slower growth, which made them vulnerable to predation for a longer period of time.

Concerted efforts by government agencies and private citizens to reverse cultural eutrophication (e.g., promoting or mandating the use of phosphorus-free laundry detergents, building more efficient wastewater treatment plants, agriculture best management practices) have ocassionally led to unwanted consequences. Specifically, nutrient loading rates into some reservoirs were reduced at a time when many reservoirs were experiencing decreased internal nutrient-loading rates and trophic depression in the decades following impoundment. Rates of nutrient loading and trophic states have changed so abruptly in a few systems that a new word entered the lexicon of reservoir and lake managers: *oligotrophication*. Moving from eutrophy to mesotrophy or from mesotrophy to oligotrophy usually results in clearer water because of reduced phytoplankton biomass, which most citizens equate to "cleaner" water. However, the trade-off between "clean water" and productive fisheries began to be discussed by fisheries biologists (e.g., Ney 1996; Stockner et al. 2000). The tight linkage between phytoplankton standing crops or phosphorus concentrations and fish biomass and sport fish harvest means that fisheries can suffer in reservoirs that shift to a lower trophic state.

The trade-offs between cleaner water and popular recreational fisheries were investigated by Maceina et al. (1996). They showed that modest shifts in trophic state could achieve clearer water while still maintaining good fisheries. A reduction in chlorophyll-*a* concentrations in eutrophic reservoirs in Alabama to 10–15 ppb was projected to increase water clarity and improve aesthetics for other recreational users without adverse effects to recreational fisheries. If a shift in trophic state was to occur from eutrophic to oligomesotrophic, catch rates of major recreational species was unlikely to shift, but in some instances smaller fish could result.

4.4 Eutrophication Management

4.4.1 Monitoring Program

Identifying clear goals and means to achieve them is important in designing a eutophication monitoring program. Collecting data with vague goals and without a clear idea of how the monitoring data will achieve the goals rarely will produce good information. Possible goals include characterizing nutrient levels, identifying embayments where nutrients are excessive, determining whether management practices have forestalled eutrophication, and tracking water-quality changes or trends.

Monitoring eutophication can be complex and expensive and requires long-term commitments. Nevertheless, monitoring eutrophication may not always require field monitoring. Alternatives may include using existing data, partnering with another agency that is field monitoring, or documenting eutrophication with indicators that are not obtained through in-lake water-quality monitoring.

Whether or not monitoring is implemented, it is useful to gather and exame data from previous water monitoring in the reservoir. Often data are available from the reservoir controlling agency, from state water-quality agencies, from federal agencies such as the U.S. Environmental Protection Agency (USEPA) or U.S. Geological Survey, from local government agencies, or from universities. Existing data may not provide all the information needed but will help make an informed decision on whether it is necessary to implement a monitoring strategy and what information to target with monitoring.

One critical decision is what variables to monitor (Green et al. 2015). For monitoring eutrophication, obvious choices are total phosphorus, total nitrogen, chlorophyll-*a*, and Secchi depth. These variables have already been used to develop lake and reservoir classification schemes relative to trophic state (Table 4.1). Total phosphorus combines organic phosphorus (i.e., phosphorus bound to plant or animal tissue) and orthophosphate (PO_4; inorganic form of phosphorus). Although only orthophosphate is readily available to phytoplankton or aquatic plants, other forms of phosphorus can be converted to orthophosphate. Therefore, total phosphorus is the most complete indicator of eutrophication status. Total nitrogen combines nitrate (NO_3), nitrite (NO_2^-), ammonia (NH_3), and organic nitrogen. Total nitrogen can be analyzed in one step or calculated from the sum of nitrate + nitrite + total Kjeldahl nitrogen. Chlorophyll-*a* is the photosynthetic pigment that yields the green color in phytoplankton and can be used as an indirect estimate of phytoplankton biomass in water. Chlorophyll-*a* is generally correlated with levels of phosphorus and nitrogen, although the correlation is not always strong because other variables also influence phytoplankton production. Secchi disk may be used to index eutrophication when suspended sediments are not a large component of suspended solids (section 5.4).

These four eutrophication metrics have daily and seasonal cycles that change regionally and locally depending on precipitation, land use, and local effects. Moreover, the peak of cycles may not coincide among these four metrics. Thus, standardizing the best time to conduct measurements over a large geographical scale is ineffective. Nevertheless, if there is a best time it may be when rainfall is low (usually late summer to early fall in the eastern USA).

Eutrophication monitoring of the nation's aquatic resources is conducted routinely by the National Aquatic Resource Surveys (NARS), run by the USEPA's Office of Water. A collaborative program between the USEPA, states, and tribes, NARS is designed to assess the quality of the nation's lakes and reservoirs through statistically-based survey designs. The NARS program provides nationally consistent data on the nation's waters, although the number of lakes and reservoirs included in the surveys are limited. The NARS database (NARS 2016) and the National Lakes Assessment Field Operations Manual (NLA 2012) are useful resources that can provide the foundation for developing local monitoring programs.

4.4.2 Watershed Remediation

Most remediation techniques are directed toward reducing phosphorus flowing into reservoirs, moving nutrients through the reservoir with minimal retention, and neutralizing or removing nutrients already accumulated in the reservoir. Eutrophication control measures are aimed at reducing the levels of nutrients reaching a water body rather than treating the water body once a problem has occurred. Reduction of nutrients entering reservoirs from the surrounding watershed is a major emphasis of programs designed to control eutrophication. Reservoir managers can partner with watershed agencies and organizations (section 2) to reduce the amount of nutrients that enter the reservoir from point and nonpoint sources in the watershed.

When topographic conditions are favorable, bypass channels (section 3.7.1.3) may be constructed to route water around the reservoir during high-flow events with excessive nutrient concentrations. Infrequent large-flood events often contribute most to total loadings because of high concentrations and volumes (Morris et al. 2014). By routing nutrients around the reservoir into the tailwater, both nutrients and sediment accumulations are reduced. Conversely, bypass channels may be applicable during low flow when certain pollutants may flow in high concentrations.

4.4.2.1 Constructed wetlands

Constructed wetland systems (Figure 4.3) are designed explicitly to incorporate the functions of natural wetlands to aid in nutrient removal from inflowing water (USEPA 2000b). Constructed wetlands also can provide for quantity control of inflows by offering a temporary water storage above the permanent pool elevation. As runoff flows through the wetland, nutrient removal is achieved by settling and by biological uptake. Constructed wetlands are among the most effective practices in terms of pollutant removal and also offer aesthetic value (Moshiri 1993). A sediment forebay can be constructed for removal of coarse sediment that could degrade wetland performance. Construction costs may be relatively higher than the sediment basins described in section 3.7.1.1.

The processes of nitrogen and phosphorus removal in wetlands are different. Plants uptake inorganic nitrogen and phosphorus (e.g., nitrate, ammonia, and soluble reactive phosphate) through their roots, foliage, or both during warm seasons and convert them into organic compounds (USEPA 2000a). The majority of these assimilated nutrients are released back into the water and soils when plants grow old and decompose during the cold season. Roughly 10%–50% of the nutrients remain stored in hard-to-decompose plant litter and becomes incorporated in wetland soils.

Nitrogen removal involves a large suite of bacteria that mediate or conduct numerous chemical reactions (USEPA 2000a). These bacteria are found on solid surfaces such as soil, litter, and submerged plants. The main transformation processes are ammonification (organic nitrogen to ammonia), nitrification (ammonia to nitrate or

Figure 4.3. Constructed retention cells and wetland complexes in the 4,700-ac watershed of Iron Horse Trail Lake reservoir, Nebraska. The inset in the top left shows Lores Branch approaching the complex. The complex slows down flows and allows expansion into wetlands to remove sediment and nutrients approaching the reservoir. 1 = channel training and maintenance access berms; 2 = sediment retention dikes; 3 = islands created from in-lake sediment spoils; 4 = wetlands. The top of the berms are lower than the top of the sediment retention dikes, and are designed to train the normal stream inflows to meander during normal flows and small rain events, but will be overtopped at high flows. Wetland vegetation will grow around the berms. During drawdowns the berms provide access for heavy equipment to remove deposited sediment. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

nitrite), and denitrification, by which nitrate (NO_3) is converted to nitrogen gas (N_2), which composes 85% of the atmosphere.

Denitrification is the dominant, sustainable removal process in wetlands that receive high nitrate loadings from agricultural runoff (Hammer 1989). Denitrification primarily is performed by bacteria that are heterotrophic, meaning they require a carbon source for growth and energy. Wetland plants are a key source of this carbon. Because denitrification is facilitated by bacteria, the process is temperature dependent. Higher rates of denitrification occur during higher temperatures when the bacteria are more active. Therefore, wetlands designed for nutrient removal work hardest at removing nitrogen during the summer.

Conversely, phosphorus is removed primarily through physical and chemical processes (USEPA 2000a). Phosphorus typically enters wetlands attached to suspended material such as small soil particles or as dissolved phosphorus (PO_4). Particulate phosphorus is deposited in wetlands during sedimentation. The leaves and stems of vegetation help settle out particles by slowing the passing of water and allowing the particles to drop onto the substrates. The dissolved phosphorus accumulates quickly in sediment by sorption (to aluminum and iron oxides and hydroxides) and precipitation (to form aluminum, iron, and calcium phosphates).

There are several design variations of the constructed wetland, each design differing in the relative amounts of shallow and deep water and dry storage above the wetland. These designs include the shallow wetland, the extended-detention shallow wetland, and the pond–wetland system (Schueler 1992; Davis 1995).

In the shallow wetland design, most of the water-quality treatment volume is in the relatively shallow ($\leq$1 ft) marsh depths (Hammer 1997). The only deep portions (3–5 ft) of the shallow wetland design are the forebay at the inlet to the wetland and a small pool at the outlet. One disadvantage of this design is that because the pool is very shallow, a relatively large area is typically needed to store large volumes of water.

The extended-detention shallow wetland design is similar to the shallow wetland (Schueler 1992). However, the wetland is designed to hold deeper water (>1 ft) temporarily so that water can be held for a longer period. This design can treat a greater volume of water in a smaller space than the shallow wetland design. Plants that can tolerate longer and deeper flooding, as well as dry periods, are desirable in this design.

The pond–wetland system has two separate cells (Schueler 1992). These include a sediment basin (section 3.7.1.1) and a shallow marsh (Figure 4.4). The sediment basin traps sediment and reduces runoff velocities prior to the water's entry into the wetland for additional treatment. Less land area is required for a sediment basin–wetland system than for the shallow wetland or the extended-detention shallow wetland systems. Access to the sediment basin is desirable to remove sediment accumulation periodically. According

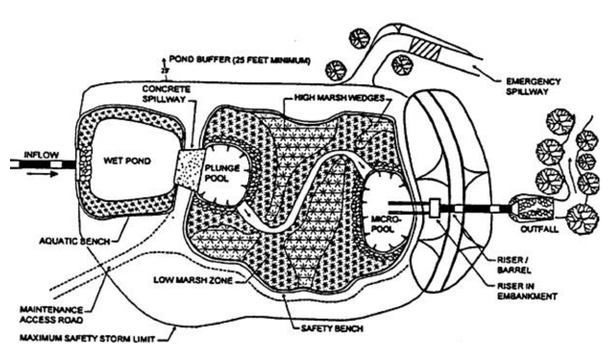


Figure 4.4. Shallow wetland schematic. Image credit: Center of Watershed Protection, Ellicott City, Maryland.

to Schueler (1992) approximately 70% of the volume should be deep storage and 30% marsh.

Constructed wetlands have a high suspended solids removal capability. Most wetland designs are able to remove roughly 50%–80% of the total suspended solids (Hammer 1989; USEPA 2000a, b). Removal of other pollutants is usually lower. Rough estimates of pollutant reductions derived from published data suggest that wetlands may reduce total phosphorus by about 30%–40%, total nitrogen by 20%–30%, fecal coliform by 50%–70% (if no resident waterfowl population present), and heavy metals by 40%–50%.

It may be beneficial to incorporate a cascade ponding system into the wetland layout to take advantage of an existing grade, provide depth diversity, and incorporate flow to provide aeration and increase oxygen levels in the water exiting the wetland. A cascade of wetlands would provide the ability to incorporate more than one wetland type to enhance different aspects of the overall treatment process (Kadlec and Wallace 2009).

Many reservoirs impounded over lowland rivers have extensive upstream floodplains associated with major tributaries. Wagner and Zalewski (2000) considered converting part of the natural floodplain of the Pilica River into constructed wetlands to trap phosphorus loads associated with major floods and normal flows. The river has an average discharge of 650 cfs and varies from 200 to 5,900 cfs. They estimated that for a reduction of 17%–27% of the total phosphorus load transported down to the 5,400-ac Sulejow Reservoir, wetland areas totaling 370 ac would have to be constructed in the floodplain upstream of the reservoir. These wetlands were estimated to average about 3-ft deep and extend about 15 mi upstream of the reservoir. They also predicted that if the wetlands had an area of 1,200 ac and a depth of 5 ft, the total phosphorus reduction achieved would be 21%–34%. In general, one large wetland may be more expensive to construct than many small ones, but a single wetland is easier and less expensive to operate and maintain (Hammer 1997).

4.4.2.2 Pre-dams

Pre-dams are small reservoirs having a relatively low retention time, usually just a few days. They are constructed immediately above the main reservoir in one or more of the major tributaries (Figure 4.5). Their objective is to trap nutrients to reduce the load into the main reservoir. Effectiveness depends on retention time. Because nutrient uptake by phytoplankton and sedimentation are the focal processes for nutrient removal in pre-dams, they are relatively shallow, have a surface outlet, and are of a

Figure 4.5. Rappbode Reservoir, Germany with two pre-dams (lower left corner). Photo credit: Google Earth.

size appropriate to optimize nutrient uptake. Complete draining is required for periodic removal of sediment. Benndorf and Putz (1987a, b) describe a method for estimating optimum size for optimal nutrient removal.

A pre-dam was constructed above Nielisz Reservoir, Poland, to improve water quality in the reservoir (Mazur 2010). Nielisz Reservoir is an impoundment of the Wieprz River. The reservoir has a watershed of 477 mi^2, an area of 2,200 ac, and an average depth of 9 ft. The pre-dam impoundment has an area of 442 ac and an average depth of 2.3 ft. A survey in 2008 revealed seasonal reduction of the majority of water-quality indicators at the outflow of the pre-dam. Within the study period the level of total suspended solids decreased by 78%, phosphates concentration by 47%, ammonia by 37%, nitrates by 34%, total nitrogen by 24%, nitrites by 17%, and potassium by 15% (Mazur 2010). Nevertheless, there is limited information about the value of pre-dams and whether pre-dams may be applicable for very large reservoirs.

4.4.3 In-Lake Remediation

Control of external sources may not be sufficient to return reservoirs to a desired state. In many cases the changes in the reservoir have been so dramatic—major shifts in biota, loss of habitat, physical changes in bottom sediment—that merely turning off the loadings is not sufficient to improve water quality and ecosystem performance. Therefore, in-lake restoration techniques may need to also be applied.

4.4.3.1 Guide curve revision

Operation of reservoirs often is guided by a water management plan that outlines the level at which the reservoir will be maintained on a daily basis, therefore dictating retention and discharge. This plan usually is known as a rule or guide curve (section 7). Depending on the purpose of the dam, the guide curve may permit large annual water-level fluctuations and may have some flexibility for modification.

Operation of a navigation reservoir requires a relatively stable water level, and retention time varies little from that of the river. Conversely, flood control or storage reservoirs usually fluctuate greatly over the year; water is stored in spring, held in summer, released in fall, and allowed to move through the reservoir in winter. This regime results in a large fluctuation of retention times and, therefore, the extent to which nutrients entering the reservoir are allowed to settle in the reservoir. Changing the guide curve to adjust the residence time may reduce long-term eutrophication of the reservoir. One strategy may be to drop the water level during the high-inflow season to allow undesirable inflows entering the reservoir to be flushed through (section 3.7). A few months later the reservoir is refilled to normal pool, when the nutrient concentrations in the inflow water are typically lower. Another strategy may be to maintain a large pool with a greater retention time, which may allow suspended material to settle uplake and thereby reduce nutrients in the main reservoir. Models are available to estimate the effect of discharge rates on nutrient retention (Park et al. 2008).

4.4.3.2 Inflow routing

Routing of undesirable major storm inflows is possible through control of discharges (WOTS 2004). Water from major storm events tends to have similar density and can be routed through the reservoir and past the dam to minimize nutrient and sediment settling within the reservoir. The inflow generally seeks and follows a layer of neutral density in a density-stratified reservoir, and thus a density current will develop and proceed through the reservoir. Because of the differences in density, currents can proceed toward the dam without mixing with most of the reservoir water. If the reservoir shape and bathymetry are highly irregular, with projecting features that can break up the flows, density currents may not sustain themselves. However, density currents occur in many reservoirs, and it is often possible to allow such currents to pass through the reservoir toward the dam. Existing outlets in the dam can then be operated to move the density current downstream. Selective withdrawal capability is required if inflow occurs at mid-depth levels. However, depending on the elevation of the inflow, it may be possible to use spillways, sluiceways, or other outlets to release the inflow (WOTS 2004). No structural modification or addition is involved, so costs are associated with only change of operation. This technique is applicable in small or large reservoirs, where other techniques may not be feasible (Kondolf et al. 2014). This technique is most applicable in reservoirs that stratify thermally (WOTS 2004).

4.4.3.3 Dilution

Dilution efforts direct a low-nutrient water source into and through a reservoir as a way to dilute and remove nutrients from the high-nutrient impounded water. The additional flow may wash out surface phytoplankton and replace high-nutrient impounded water with lower-nutrient dilution water. A disadvantage is that dilution requires large volumes of low-nutrient water that may not be accessible or available.

Moses Lake reservoir, Washington, was diluted with low-nutrient water from the Columbia River during summer. Annual volumes and timing of dilutions were highly variable depending on water availability. Average turnover rate in the lake was 0.3% per day, and dilutions increased it to 0.4%–2.2% per day (Welch and Weiher 1987). Notable reductions in total phosphorus from about 150 to 50 ppb were recorded in various parts of the lake, particularly in the arm where dilution water was inflowing. The oligotrophication of Moses Lake from hypereutrophic to mesotrophic was accompanied by a marked shift in the fish assemblages (Welch 2009).

4.4.3.4 Flushing

Flushing increases flow velocities in a reservoir to the extent that dissolved or suspended nutrients and nutrients concentrated in sediment are transported through low-level outlets in the dam (see section 3.7.3.6).

4.4.3.5 Selective withdrawal

This method is applicable to stratified reservoirs, where the highest phosphorus concentrations have accumulated in the hypolimnion resulting from the strong release of phosphorus from sediment during anoxic conditions. The method relies on selective discharge of hypolimnetic waters (low in oxygen and rich in phosphate, iron, and manganese) from a reservoir (WOTS 2004) instead of discharge of water from the epilimnion, which often has lower nutrients. Hypolimnetic withdrawals are most effective if done without affecting stratification and thus avoiding the transport of nutrients and anoxic water from the hypolimnion to the epilimnion. Effectiveness is also increased when the discharged volume can be replaced by sufficient inflow to maintain the lake level relatively constant (Cooke et al. 2005).

The advantage of hypolimnetic withdrawals is the relatively low cost. A disadvantage is the discharge of cold water, nutrients, and other toxic compounds downstream. Whereas cold water may allow development of specialty fisheries, water may require aeration or other treatments. Mixing discharges with epilimnetic water may improve discharge water quality, although increase temperature.

4.4.3.6 Hypolimnetic aeration and oxygenation

The basic concept of an aeration system is to maintain oxygen continually at the bottom of the reservoir so that phosphorus release from the sediment to the water column is reduced. The aeration also supports more rapid degradation of organic sediment by aerobic bacteria. Most commonly, aeration is achieved by compressors that introduce air near the bottom of the reservoir through perforated tubes. The rising bubbles push the anoxic water up to the surface where it is re-aerated with atmospheric oxygen. However, this method can break the stratified conditions in the reservoir and bring up nutrient-rich water to the epilimnion, which may trigger even more intensive phytoplankton growth. Additional details about hypolimnetic aeration and oxygenation are given in section 6.11.3.

Hypolimnetic aeration may not operate satisfactorily if the water body's maximum depth is <40–50 ft (Cooke et al. 2005). Aerators are usually turned on after the spring circulation and run throughout the summer until autumn circulation. Aerators also may be turned on during the winter under the ice cover if necessary. Hypolimnetic aeration has to be designed specifically for the conditions existing in a particular reservoir. Because of the need for a power source to operate the equipment, operation costs may be high, although solar systems are becoming available.

Hypolimnetic aeration and oxygenation is not always successful in controlling nutrients. An oxygenated hypolimnion does not necessarily assure that the sediment surface will be oxic enough to decrease phosphorus release sufficiently from the sediment. Also, in some cases diffusion of nutrients to the epilimnion from the hypolimnion has been observed even though stratification was maintained (Steinberg and Arzet 1984). Some side effects of aeration can be beneficial. Aeration allows zooplankton access to deeper, dark water that serves as refuge (McComas 2002). Additionally, the expanded aerobic environment can develop habitat for cold and coolwater fish.

4.4.3.7 Sediment removal

Sediment removal through dredging or excavation (section 3.7.3) could be an effective method for reducing nutrient availability in reservoirs. The advantage of this method is that the results are long lasting. Removal of upper layers of the reservoir bottom sediment is most effective in shallow water, and the upper layer is where phosphorus is often most available for plant production. Removal of sediment also may remove cyanobacterial inoculum (Drabkova and Marsalek 2007). Sediment removal is probably most applicable in small reservoirs (<2,000 ac) or when limited to carefully chosen embayments (Peterson 1982; Eiseltová 1994; Cooke et al. 2005). The decision of whether the sediment will be removed or treated and left in place (see 4.4.3.9) depends on local circumstances, including sediment amount and quality, nutrient content, content of toxic compounds, availability of a disposal area, and possibility of re-use. Disposal of the dredged materials can be especially problematic. If the sediment does not contain toxic compounds, it can be used for agricultural purposes as a fertilizer. In special cases, the dredged sediment can be applied directly on agricultural fields (Pokorný and Hauser 2002).

4.4.3.8 Sediment drying

Some reservoirs are characterized by a high degree of water-level fluctuations associated with their operational objectives (section 7). Deep portions of the reservoir may remain inundated under all but the most extreme drought conditions. Conversely, some of the shallower parts of the reservoir may be inundated for only a few days or weeks every few years. This wetting and drying can have a profound effect on the processes responsible for nutrient cycling in the regulated zone (i.e., the fraction of the reservoir dewatered in an annual cycle).

As sediment dries out, a decrease in bacterial biomass and activity is expected (Van Gestel et al. 1992; De Groot and Van Wijck 1993). It has been shown that bacterial activity declines linearly with soil water content (Orchard et al. 1992; West et al. 1992). At the extreme end of sediment desiccation a high bacterial mortality and release of nitrogen and phosphorus caused by cell lysis have been reported (De Groot and Van Wijck 1993; Qiu and McComb 1995), resulting in a flush of nitrogen and phosphorus upon rewetting of sediment. Thus, various studies have shown a net release of nutrients from sediment that has been exposed to air and subsequently rewetted.

4.4.3.9 Phosphorus precipitation and inactivation

This technique focuses on lowering the reservoir's phosphorus levels by removing phosphorus from the water column and retarding release of phosphorus from sediment (Holdren et al. 2001). This is achieved by application of coagulants. These compounds, when added into the water, precipitate into flocculates. During flocculation, the phosphorus is bounded and converted into a form unavailable to phytoplankton. Some coagulants also can bind small particles, including phytoplankton cells, into the flocculates. The flocculates then settle to the sediment, thus removing phosphate and cyanobacteria from the water column. At the bottom of the reservoir, the coagulum further increases the binding capacity of sediment for phosphorus (Holdren et al. 2001).

Binding of bioavailable phosphate into flocculates is stronger than binding of phosphorus in particulate form (e.g., organic matter). Therefore, this treatment works best when applied to reservoirs with long retention time during late fall to early spring, when free phosphate is highest before it is incorporated into intensively growing phytoplankton (Holdren et al. 2001). Interference with the binding process occurs in shallow reservoirs overgrown by macrophytes and when external loading exceeds the phosphorus binding capacity of the flocculate (Welch and Cooke 1995). Effectiveness of this treatment can be low in shallow reservoirs where wind and waves resuspend phosphorus in the sediment. Effectiveness increases in small reservoirs with long retention time when the major phosphorus input is from the sediment (Cooke et al. 2005).

Various compounds are available to use as coagulants, including aluminum, iron, calcium salts, and clay materials. The compounds vary in their effectiveness and are described below.

Aluminum.-- The most commonly used aluminum coagulant is aluminum sulfate (alum, $Al_2(SO_4)_3{\cdot}14H_2O$). When added to the water, alum quickly forms large, visible, nontoxic precipitates of aluminum hydroxide that settle to the sediment. Alum is extremely effective in controlling sediment phosphorus release rates, improving water clarity, reducing phytoplankton biomass, shifting population species composition from cyanobacteria dominance toward bacillariophytes and chlorophytes, increasing daphnid (Cladocera) biomass, and increasing usable fish habitat (Jorgensen et al. 2005).

To remove not only dissolved phosphorus successfully but also particulate phosphorus and to provide sufficient inactivation of sediment phosphorus, the goal is to apply as much alum as possible consistent with environmental safety. Several procedures to estimate a proper dose are suggested by Cooke et al. (2005) and are based on determination of mobile inorganic phosphorus in the sediment (Rydin and Welch 1998; Reitzel et al.

Figure 4.6. Applying alum at a Nebraska reservoir. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

2005), estimated rates of phosphorus internal loading from sediment (Kennedy et al. 1987), or lake water alkalinity (Kennedy and Cooke 1982). The inorganic phosphorus is removed more effectively than particulate organic phosphorus (cells, detritus), suggesting that the most effective timing of alum treatment would be in early spring when the content of soluble phosphorus is highest. On the other hand, coagulation is reduced at low temperatures. Treatments in early summer before cyanobacterial blooms occur are reportedly successful (Cooke et al. 2005). Alum treatments are often administered to only sections of reservoirs (Figure 4.6), but when applied to a whole reservoir, treatments are spread over several days, allowing organisms that are affected by the doses to escape to untreated areas (Cooke et al. 2005).

Barko et al. (1990) reported on the effects of a hypolimnetic alum treatment on sediment phosphorus availability in the 136-ac Eau Galle Reservoir, Wisconsin. Alum treatments over 5 years at batch doses of 100 lb/ac resulted in a substantial reduction in hypolimnetic total phosphorus and internal total phosphorus loading during the study. However, the frequency of major external total phosphorus loading events during that year (i.e., major precipitation events) negated the effectiveness of alum treatment in reducing epilimnetic total phosphorus mass as it remained essentially unchanged from pretreatment years.

Morency and Belnick (1987) reported on alum treatments at two relatively small (110 and 370 ac) and shallow (6 and 8 ft mean depth) hypereutrophic lakes in Oregon. Both lakes were treated with liquid alum at a dosage of 10.9 ppm based on their similar alkalinities (80 and 88 ppm as $CaCO_3$). This strategy allowed the highest alum application without decreasing pH below 6. In the smaller lake this treatment resulted in a dramatic reduction in total phosphorus from a mean summer concentration of 115 to 26 ppb and chlorophyll-*a* concentration from 58 to 5 ppb; water clarity increased from 5.6 to over 8.2 ft. Changes were also noted in the larger lake, although the changes were not as pronounced as in the smaller lake.

Alternative methods for alum applications have been developed (Harper et al. 1998). Continuous alum injection involves a flow-weighted alum dosing system designed to fit inside a storm sewer manhole. This method allows treatment of stormwater runoff (point sources). Continuous alum treatment is typically most applicable in unstratified lakes with short retention times to remove nutrients and sediment from the incoming waters at or near the lake inlets. It is also most applicable in reservoirs for which the locations of all the major stormwater inputs are known. Because of high installation and operation costs, alum injection is best applied to situations in which a large volume of water can be treated. To increase the efficiency and cost effectiveness, alum dosing may be designed to occur only during storm-flow conditions when nutrient and sediment concentrations are elevated to problematic levels. Alum dosing may not be necessary or may be reduced significantly during base-flow conditions when nutrient and sediment inputs are generally low.

There has been debate about the safety of alum to humans and the aquatic environment, particularly within the North American Lake Management Society (NALMS). The current position of NALMS is that alum is a safe and effective lake management tool, but that alum applications should be designed and controlled to avoid concerns with toxicity to aquatic life. Moreover, NALMS considers watershed management as an essential element of protecting and managing lakes. In cases in which watershed phosphorus reductions are neither adequate nor timely, alum is an appropriate tool to accomplish meaningful water-quality objectives (NALMS 2004).

Iron.-- Iron is applied usually in the form of $FeCl_3$ (iron chloride), but $FeCl_2$ or $Fe(SO_4)_3$ (iron sulfate) also may be used. In contrast to alum, the stability of iron flocculates is less dependent on pH, and iron does not appear in toxic form. Nevertheless, the sorption to $Fe(OH)_3$ (iron hydroxide) is greatest at pH 5 to 7, which is not common in eutrophic lakes especially if high phytoplankton densities are present. As with alum treatment, hydrogen ions are released, which may lead to a significant decline in pH and toxic effects to fish if pH levels decline below 6 (Søndergaard et al. 2002).

Further, the stability of Fe-P compounds is strongly dependent on changes in the redox state. As the dissolved oxygen in water above sediment drops below 1 ppm, iron is used as an alternate electron acceptor. Reduced ferrous ion (Fe^{2+}) is soluble, and iron-bound phosphorus is released. This change occurs rapidly, so that even brief periods of anoxia at the bottom of the reservoir lead to substantial phosphorus release. To prevent this effect, aeration is usually applied along with Fe application. Continuous Fe application during summer has been used combined with artificial destratification to prevent cyanobacterial blooms (Deppe and Benndorf 2002).

Calcium.-- Calcium carbonate (calcite, $CaCO_3$) or calcium hydroxide (lime, $Ca(OH)_2$) can be added to water bodies as phosphorus precipitants (Neal 2001). Calcite sorbs phosphorus especially when pH exceeds 9.0 and results in significant phosphorus removal from the water column. Phosphate adsorbs at the calcite surface or binds inside a crystal during $CaCO_3$ formation when calcium hydroxide is applied (Kleiner 1988; House 1990). Various calcite forms have been reported for potential use as active barriers in sediment caps designed to reduce phosphorus release from sediment (Hart et al. 2003).

The described application doses of lime are in a range of 25–300 ppm as Ca (Søndergaard et al. 2002). The advantage of lime is its low price and nontoxicity. However, adverse effects to aquatic organisms may occur because application of lime increases pH (Miskimmin et al. 1995). In soft-water lakes pH easily may exceed 11 (Zhang and Prepas 1996). The lime treatment also temporarily increases turbidity. As an additional benefit, lime and calcite also may be used to precipitate cyanobacterial cells from the water column (Zhang and Prepas 1996).

Clay materials.-- A range of clay materials can be used to bind phosphate from water, including zeolites, modified clays, and kaolins (Moharami and Jalali 2015). Phoslock™ is a commercially available specially modified clay made from bentonite clay in which the sodium or calcium ions (or both) are exchanged for lanthanum. The addition of this element allows it to bind with phosphates to form rhabdophane and thereby remove phosphorus permanently from the water column. Phoslock™ was reported to bind phosphorus successfully in the Canning and Vasse rivers in Australia (Robb et al. 2003). Phoslock™ applications require no buffer to protect water quality and aquatic life during and after application. Clay substrates with high phosphorus-sorption capacity may improve sustained phosphorus removal in wetlands (Mateus-Dina and Pinho-Henrique 2010).

4.4.4 Biomanipulation

4.4.4.1 Fish populations

Excretion of nutrients by benthic-feeding fish assemblages can be a substantial fraction of nutrient inputs, comparable to external loading or nutrient release from sediment (Schaus et al. 1997). Thus, eutrophication control may be more effective if it considers both external inputs of nutrients and the translocation of nutrients from the reservoir sediment to the water.

Biomanipulation refers to employing the service of secondary or tertiary aquatic producers to affect a community structure and ecosystem (Shapiro and Wright 1984). Limnologists traditionally have considered lake systems to consist of components linked through a unidirectional flow of influence from nutrients to phytoplankton and to zooplankton and finally to fish. Biomanipulation represents a shift in paradigm by considering the reverse effects. Thus, a reduction of planktivores and benthivores through predation by piscivorous fishes would be followed by an increase in the abundance of large zooplankton (predominantly cladocerans), an increase in water clarity, and a decrease in nutrient recirculation from sediment. As a consequence, the grazing pressure on phytoplankton by zooplankton is enhanced, bottom stirring is reduced, and bottom nutrients remain less disturbed because of the lack of stirring and the lack of recirculation of nutrients through feces. In theory, the reduction of planktivory or benthivory may be achieved either by managed removal of the zooplanktivorous and benthivorous fishes or by increasing predation by creating an abundant piscivorous fish community via stocking, introductions, or protection regulations.

Thus, manipulation of fish populations, especially through artificial enhancement of piscivore populations, could be a useful method for reduction of phytoplankton levels and eutrophication. Biomanipulation has the potential to combine eutrophication management and sustainable fisheries management. The strategy may be particularly successful in those regions where commercial and recreational fisheries target a broad scope of species. Nevertheless, biomanipulation can be unpredictable as there are many unknowns about community interactions (DeMelo et al. 1992).

As an example of an unintended biomanipulation, increased piscivory by introduced Nile perch caused a shift in the ecosystem of Lake Victoria, Africa (Ochumba and Kibaara 1989). Cichlids grazed on the lake's plant community. After the introduction of Nile perch, cichlid populations were depressed, which reduced grazing, and eventually phytoplankton biomass increased. Concurrently, water quality deteriorated, as measured by increased phytoplankton turbidity and anoxia in deep waters.

In Round Lake, Minnesota, Shapiro and Wright (1984) reported applying rotenone to eliminate the lake's fish community. The lake was then restocked with bluegill and a high population density of largemouth bass and walleye. After restructuring the fish community, water transparency increased and chlorophyll-*a* decreased. Zooplankton densities decreased, but the mean sizes of zooplankton increased. The shift in zooplankton size is important because the filtering rate and the size range of edible phytoplankton increases with zooplankton size. Changes in zooplankton were not only responsible for the decrease in phytoplankton but also appeared to be responsible for a reduction in nutrient concentrations in the epilimnion. Assimilation of nutrients by zooplankton occurs primarily in the epilimnion at night, while nutrient excretion occurs throughout the water column, possibly resulting in a net downward movement of nutrients during diel migration. Although Shapiro and Wright (1984) achieved a reversal of planktivore effects by stocking piscivores, the beneficial effects lasted for only 2 years. After the initial reductions, the water transparency and chlorophyll-*a* concentration began reverting to their pre-biomanipulation condition because of an expanding bluegill population.

4.4.4.2 Fish harvesting

In several lakes in Florida (e.g., Apopka, Dora, Griffin), biomanipulation programs have aimed to reduce nutrients by harvesting omnivorous gizzard shad (Schaus et al. 2010, 2013). These large-scale removals have reduced the biomass of harvestable (>12 in) gizzard shad by 40–60 lb/ac via subsidized commercial gillnet fisheries (Figure 4.7). These harvest rates represented about 75% of the harvestable gizzard shad. Given the size selectivity of the gear, the total population biomass of gizzard shad was reduced by <50% from an average pre-manipulation biomass. No major changes in total phosphorus or chlorophyll-*a* concentrations were detected following the biomanipulation. According to research in shallow lakes in the Netherlands, a biomanipulation must remove 75% of planktivorous and benthivorous fish before it can be successful (Hosper and Meijer 1993).

Figure 4.7. Gizzard shad harvest programs can remove nutrients from reservoirs. Photo credit: St. Johns River Water Management District, Palatka, Florida.

In practice, control of nutrients through biomanipulation and fish harvesting is not usually easy. Significant changes require substantial reductions in planktivores, often unachievable through fishing alone. Large reductions that rely on piscivores may be difficult to achieve in many reservoir communities because the prey communities are dominated by fish that grow beyond the reach of predators. For example, the adults of gizzard shad, perhaps the most abundant and common planktivore/benthivore in reservoirs of the eastern USA, grow to a size not available to gape-limited piscivores. Moreover, the effects of biomanipulation do not always last because populations of other fish species with a similar niche, or the same population, may expand. Ecosystem interactions are complex and difficult to predict, so it is also difficult to predict the results of manipulating a biological community (DeMelo et al. 1992). Despite this lack of predictability, the potential benefits of biomanipulation (e.g., low cost, absence of chemicals or machinery, fishery development) make the technique attractive.

4.4.4.3 Macrophytes

Macrophytes in reservoirs can control nutrients and prevent development of phytoplankton blooms. Macrophytes reduce wind and boat-induced resuspension of nutrients. They also absorb some of the nutrients and support periphyton communities, which further remove dissolved phosphorus (McComas 2002; Cooke at al. 2005). However, growth of macrophytes is limited in many reservoirs because of wave action, water-level fluctuations, low water clarity, fish and other vertebrates uprooting or eating plants, and other disturbances (section 11).

4.4.4.4 Floating wetland islands

Floating wetland islands are an emerging variant of constructed wetland technology that consist of emergent wetland plants growing hydroponically on structures floating on the surface of a pond-like basin (Headley and Tanner 2008). It is an artificial raft that houses native wetland plants (Figure 4.8). They represent a means of potentially improving the treatment performance of conventional pond systems by integrating the beneficial aspects of emergent macrophytes without being constrained by the

Figure 4.8. Floating wetland islands enhance removal of nutrients, provide habitat for fish and wildlife, and can enhance aesthetics. Photo credit: Cascade Meadow Wetlands and Environmental Science Center, Saint Mary's University, Rochester, Minnesota.

requirement for shallow water wetlands. An island consists of emergent wetland vegetation growing on a mat or structure floating on the surface of a pond-like water body. The plant stems remain above the water level, while their roots grow down through the buoyant structure and into the water column. In this way, the plants grow in a hydroponic manner, taking their nutrition directly from the water column in the absence of soil. Beneath the floating mat, a hanging network of roots, rhizomes and attached biofilms is formed. This hanging root–biofilm network provides a biologically active surface area for biochemical processes as well as physical processes such as filtering and entrapment.

Research has shown that floating wetland islands can reduce nitrogen and phosphorus levels in ponds (Stewart et al. 2008). One unpublished study found 32% removal of phosphorus and 45% reduction in nitrogen in lake water used in a mesocosm experiment. This is relatively new, not fully developed technology, and mesocosms are small-scale experiments that may not transfer directly into larger water bodies.

Floating wetland islands have the potential to upgrade the water-cleansing qualities of sediment basins (3.7.1.1), although more research is needed. Sediment basins are generally effective at attenuating hydraulics and removing coarse suspended sediments but are less effective at removing nutrients and dissolved contaminants. These floating wetlands are anchored but can rise and fall as the water level changes. The water depth typically has to be a minimum 3 ft to prevent the macrophyte roots from attaching to the benthic substrate.

Section 5

Water Clarity

5.1 Introduction

Water clarity refers to the transparency or clearness of the water and is influenced by turbidity and color. Turbidity is often used as a general term to describe the lack of transparency or the "cloudiness" of water resulting from the presence of suspended solids and colloidal materials such as clay, finely divided organic and inorganic matter, and plankton or other microscopic organisms (Davies-Colley and Smith 2001). Visibility through the water decreases as turbidity increases. The reduction in visibility is due to scattering of light by suspended particles (solids) in solution. Water clarity is influenced by water color. Pure water is transparent and colorless. Colored components in water absorb light energy, preventing it from penetrating as deeply as in colorless water and potentially altering temperature. The zone between the surface and the depth where light intensity is reduced to 1% of the intensity at the surface is defined as the photic zone. The rate at which light is attenuated in the water column is the light attenuation coefficient.

Solids that determine scattering of light and water clarity include inorganic and organic particulates, and suspended solids. Inorganic particulates are silt and sand that eventually settle to the bottom, resulting in sedimentation (section 3). The organic component may include dissolved organic matter and algae. Suspended solids are smaller particles that remain in suspension and generally account for most of the loss of water clarity. The sources of abiotic suspended solids include runoff from clear-cut or overgrazed watersheds, road or building construction, wave-induced sediment resuspension and shore erosion, and the bottom-stirring feeding activities of fish. There are other light-attenuating constituents of water besides suspended solids, most notably the water itself and its content of colored dissolved organic humic substances (Davies-Colley et al. 1993; Kirk 1994), but typically suspended solids are the dominant influence on light attenuation in natural waters.

Water clarity can be measured as concentration of suspended solids or indexed as turbidity or transparency (Davies-Colley and Smith 2001). Suspended solids measured as concentrations typically are measured in the laboratory. Conversely, turbidity and transparency measure optical qualities that can be measured on-site and more cheaply than solids in the laboratory. Both turbidity and transparency can be

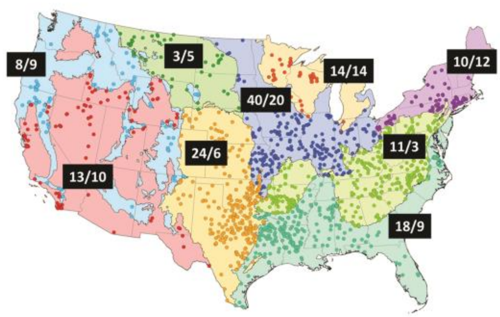


Figure 5.1. A survey of 1,299 reservoirs ≥250 ac across the continental USA identified that approximately 19% were of concern relative to inorganic turbidity and 9% relative to organic turbidity. Boxes show percentage of reservoirs according to regions (see Figure 1.3 for regions) scoring high (i.e., moderate-to-high degradation, and high degradation) on inorganic turbidity/organic turbidity. Data collected by Krogman and Miranda (2016).

calibrated to solids with reasonable predictive accuracy, although calibrations are spatially and temporally specific as solids' composition varies and affects relationships (Beschta 1980; Gippel 1995).

Water clarity is a major issue in reservoir fish habitat management, particularly in reservoirs of the central USA. A recent survey identified that the percentages of reservoirs considered impaired by turbidity vary regionally across the USA (Figure 5.1), with inorganic and organic turbidity distressing as many as 40% and 20% of reservoirs, respectively, in the temperate plains region (Krogman and Miranda 2016). The survey also identified the most important taxa in the recreational fisheries of these reservoirs. Catfishes, perches, crappies, and temperate basses provided the most common fisheries in reservoirs where inorganic turbidity was scored as moderate-to-high or high concern (Figure 5.2). Conversely, trout, salmon, pike, and black bass were less common in the fisheries of turbid reservoirs.

5.2 Total Solids

Total solids is a measure of the concentration of all solids in a water sample. They are measured by evaporating all of the water out of a sample at a standard temperature (103–105 °C) and weighing all the solids that remain (APHA 1998). Total solids can be classified into (1) suspended and dissolved solids, and (2) volatile and nonvolatile solids.

Suspended solids are that portion of total solids caught by a 0.45-micron filter (APHA 1998), or other pore size depending on standard method. The solids that pass through the filter and remain after the filtered water is dried are the dissolved solids. Suspended solids can be partitioned further into volatile and nonvolatile suspended solids. Nonvolatile suspended solids are those that remain after the suspended solids are ignited at 550°-600°C, whereas volatile suspended solids are those ignited. Volatile suspended solids are considered organics and nonvolatile suspended solids considered inorganic (APHA 1998).

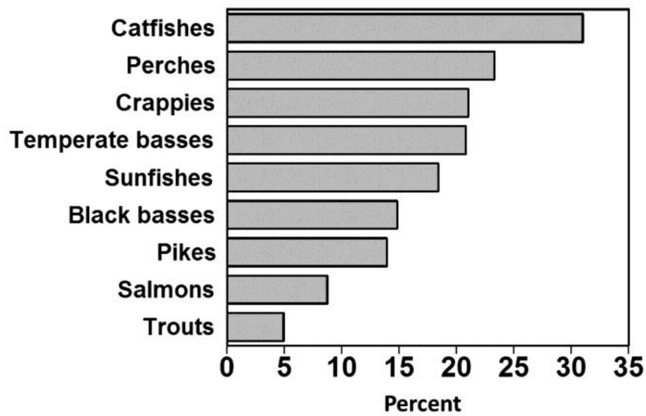


Figure 5.2. Taxa most targeted in fisheries of reservoirs where inorganic turbidity scored high (i.e., moderate-to-high degradation, and high degradation) on a survey of over 1,299 reservoirs ≥250 ac across the continental USA. Data collected by Krogman and Miranda (2016).

The particle size distribution is a key determinant of the effects of suspended solids on reservoir water clarity (Davies-Colley and Smith 2001). Particle size distribution controls not only the nature of the effects, by regulating the extent of water column turbidity and deposition rate, but also their spatial coverage. Fine particles stay in suspension for longer periods of time and for farther distances from their source, thus affecting considerably larger areas than coarse particles. Moreover, fine particles are more likely to become resuspended under windy conditions, particularly in shallow reservoirs or shallow embayments, and can adsorb more nutrients and other substances to their surfaces. The transport of clay particles in reservoirs is influenced greatly by wind-induced wave action and by influent tributaries. Effects of turbidity are, therefore, often manifested most strongly in the upper regions of a reservoir and in shallow embayments (Thornton 1990).

5.3 Turbidity

Turbidity is an optical property of the water and a general term that describes the cloudiness of water. It measures light scattering and absorption by suspended sediment, dissolved organic matter, plankton, and other microscopic organisms (APHA 1998). Consequently, turbidity is a key water-quality parameter in aquatic systems in that it has a major influence on the depth to which photosynthesis can occur and is therefore a critical determinant in the distribution of aquatic plants. Turbidity can be caused by many substances, including microscopic organisms (phytoplankton and zooplankton), bacteria, dissolved organic substances that stain water, suspended clay particles, and colloidal solids.

Although turbidity is easier to measure than suspended solids, there are limitations when using turbidity as a surrogate measure of suspended solids because the relationship between turbidity and suspended solids is confounded by variations in particle size, particle composition, and water color (Gippel 1995). Turbidity responds to factors other than just suspended solids concentrations: turbidity readings are influenced by the particle size and shape of suspended solids and the presence of phytoplankton, dissolved humic substances, and dissolved mineral substances. Consequently, a high turbidity reading can be recorded without necessarily involving a high suspended solids concentration or similar turbidity measurements from different locales may represent different concentrations of suspended solids. There is no universal relationship between turbidity and suspended solids. Site-specific relationships can be developed (e.g., Kunkle and Comer 1971; Beschta 1980; Gippel 1995), but even these relationships can vary from storm to storm, seasonally, and from year to year (Beschta 1980). When relying solely on turbidimeter data, it is not easy to know exactly what is causing the turbidity.

Turbidity traditionally has been measured as the absorption and scatter properties of light when it passes through water and reported in terms of two units of measure. The unit most frequently encountered in older reports is Jackson Turbidity Units (JTU), measured by a Jackson candle turbidimeter. The APHA (1998) no longer recommends the measurement of turbidity using this technique. More recently, turbidity is measured using a nephelometric turbidimeter that measures light scattering relative to a standard suspension, usually of formazin. Turbidity, as measured by this type of turbidimeter, is reported in Nephelometric Turbidity Units (NTU). The APHA (1998) currently recommends that NTU be used as the standard of measure for reporting turbidity.

Increased turbidity also can influence the heat budgets of reservoirs through the absorption of heat by suspended particles (Kirk 1985) or by increased reflection of sunlight back to the atmosphere (Clarke et al. 1985). Therefore, depending on the nature of the suspended sediment, mineral turbidity can cause water temperatures to increase or decrease. Alterations to heat budgets may, in turn, affect other abiotic processes indirectly. By modifying water temperature and therefore water density, temperature alters the settling velocity of suspended particles, especially those with densities close to that of water (Kerr 1995).

5.4 Transparency

Historically, water transparency has been measured in standing water bodies with a Secchi disk, a black-and-white disc that is lowered into the water by a graduated line until the image is judged to disappear from view. The depth of disappearance, the Secchi depth, is a useful index of visual water clarity. Secchi depth provides a simple and inexpensive indicator for the clarity of natural waters (Preisendorfer 1986). Secchi depth can vary depending on the reflectance of the white face of the disk and the reflectance of the water. Secchi depth readings are thus dependent on light conditions

(Davies-Colley and Smith 2001). Standardization of observations can increase precision (Smith 2001). Standardization can be achieved by (1) keeping constant the size and design of disk; (2) consistently measuring just above disk disappearance, at disk disappearance, at disk reappearance, or the mean of the latter two; (3) collaborating between more than one observer to arrive at the numbers; and (4) measuring with the sun behind the person taking the measurement, except when the sun is directly overhead (Hambrook-Berkman and Canova 2007).

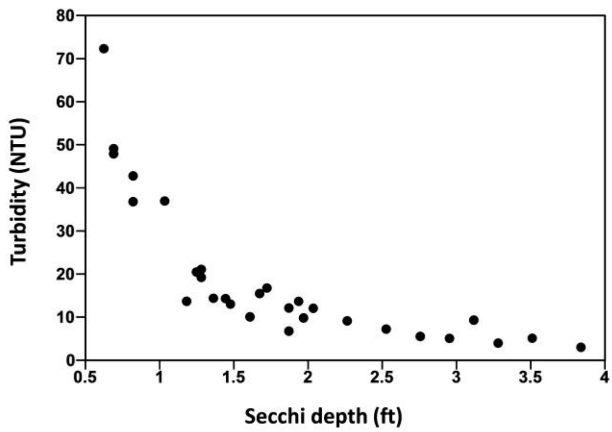


Figure 5.3. Relationship between Secchi depth and turbidity in Mississippi reservoirs. Mississippi Department of Environmental Quality, unpublished data.

Secchi depth transparency is correlated with turbidity, but they measure different things (Effler 1988). These two measures differ in their sensitivity to the light attenuation processes (i.e., absorption and scattering), and therefore measurements are affected by different substances that determine attenuation. Secchi depth becomes increasingly more insensitive to changes in turbidity and scattering at high values of turbidity and even more insensitive to changes in absorption. Absorption becomes progressively more important in influencing Secchi depth at low turbidity. Because of these relationships, Secchi depth does not respond linearly to turbidity, and the relationship between NTU turbidity values and Secchi depth is curvilinear (Figure 5.3).

5.5 Sources of Suspended Solids

The weathering and decomposition of rocks, soils, and dead plant material and their transport into streams and reservoirs represent the natural background of solids washed from a watershed (Sorensen et al. 1977). However, rates of transport have been augmented by anthropogenic disturbances to landscapes surrounding reservoirs and their tributaries. In agricultural and grazing areas, removal of vegetation and compaction of soil can cause runoff to carry eroded topsoil into rivers. Fertilization practices also may increase loads of nutrients that result in turbid algal growths. In areas with forestry operations, timber-harvesting practices, road construction, slash disposal, and site preparation can increase inputs of solids. Overall, impervious surfaces created by urbanization prevent rain from penetrating into the soil, and causes water to run off more quickly and at greater velocities, resulting in the pick up and transport of materials into streams and reservoirs directly or in stormwater outfalls. Erosion of soils at construction sites without proper controls also can increase solids and cause associated reductions in water clarity. Mining operations expose soils and can result in chronic turbidity issues. Industrial effluents and storm water can directly input solids into streams.

5.5.1 Water Flow

Turbidity generally increases as flow increases. High flow velocities keep solids suspended instead of letting them settle to the bottom. Thus, in reservoirs with major tributaries turbid waters are often present throughout the rainy season. Heavy rainfall also affects water flow, which in turn affects turbidity. Rainfall can increase stream volume and thus stream flow, which can resuspend settled sediment and erode riverbanks, loading the reservoir with suspended solids and sediment. Rain also can directly increase the level of total suspended solids through runoff. If the flow rate increases enough during major rain events, it can resuspend bottom sediment, further raising suspended sediment concentrations.

5.5.2 Wind

In areas of dry, loose soil or in earth-disturbed sites (e.g., mining or construction areas), wind can blow dust, sediment, and other particles into the reservoir. The addition of new particles will increase the suspended solids concentration. However, wind-blown dust alone generally will not increase turbidity levels in the water. Wind and water depth interact to influence turbidity in reservoir. Factors such as wind velocity, duration, direction, fetch length, and water circulation patterns interact with sediment compaction, reservoir-bottom roughness, and depth to ultimately determine the extent of sediment resuspension (Howick and Wilhm 1985). Wave-induced water movement across the surface of sediment results in resuspension of sediment. Waves are a function of the amount of wind energy impinging on the lake surface, which in turn is a function of wind velocity and fetch length. The amount of resuspension caused by waves is also a function of water depth, as the amplitudes of these movements decrease with increasing water depth.

5.5.3 Point-Source Pollution

Point-source pollution can increase turbidity through the addition of suspended solids and colored effluent (wastewater). Common examples of point-source pollution include discharge pipes from factories and wastewater treatment plants. In addition, farms and timber operations can also fall under the category of point-source pollution. These sources can release suspended solids into selected tributaries and reservoir embayments. Sometimes this water is treated or filtered before it is discharged, but sometimes it is not. Although most wastewater treatment plants include a settling period in the treatment process, this settling period does not remove nonsettable solids (Drinan and Spellman 2012). When this wastewater is discharged, these suspended solids still may be present unless treated with additional filters. In addition, colored effluent cannot be trapped by a filter (Drinan and Spellman 2012). While dyes and colored dissolved organic material are not included in a suspended solids measurement, they will contribute to turbidity readings because of their effects on light absorption.

5.5.4 Land Use (non-point pollution)

A major factor in increased turbidity and total suspended solids concentrations is land use. Agriculture, construction, logging, mining, and other disturbed sites have an increased level of exposed soil and decreased vegetation. Land development disturbs and loosens soil, increasing the opportunities for runoff and erosion. The loosened soils can then be carried away by wind and rain to a stream or reservoir.

Sediment runoff also can originate in urban areas. When it rains, soil, tire particles, debris, and other solids can get washed into a water system. This often occurs at a high flow rate because of the amount of impervious surface areas (e.g., roads, parking lots). Water cannot penetrate these surfaces, so sediment cannot settle out. Instead, the stormwater runoff flows over the pavement, carrying the suspended solids with it. Even in areas with storm drains, drains can lead to a local water source without filtration (Hamel et al. 2013). Stormwater retention ponds allow suspended particles to settle before water drains downstream (Hamel et al. 2013).

5.5.5 Boat Traffic

Similar to wind-induced waves, the action of both propeller-induced turbulence and wakes from boat traffic may resuspend sediment (Garrad and Hey 1987). These types of boat-induced turbulence have been correlated to rapid increases in dissolved solids and turbidity. Nedohin and Elefsiniotis (1997) calculated the mixing depths of 10, 28, and 50 horse power engines to 6, 10, and 15 ft, respectively. These authors also determined that motorboats had sufficient effects in a study lake to disrupt the bottom sediment and release phosphorus and other nutrients into the overlying water. Anthony and Downing (2003) concluded that although it was likely that recreational boating contributed to sediment resuspension in Clear Lake, Iowa, the correlation between boat traffic and sediment resuspension was weak. In Allatoona Lake reservoir, Georgia, regular increases in turbidity and decreases in pH occurred each weekend during the summer, suggesting increased mixing by increased boat traffic (Dirnberger and Weinberger 2005). Increases in turbidity on weekends became greater after initiation of drawdown as the reservoir became shallower. The effect of boat traffic on resuspension is likely site specific, even within the same reservoir.

5.5.6 Water-Level Fluctuations

The dewatering and flooding of soils associated with water-level fluctuations, especially winter drawdown, represent a major disturbance to reservoir ecosystems. Heavy rain on exposed soils produces migration and resuspension of sediment. Lowered winter water levels together with wind and wave action can resuspend sediment once it is well below the surface. High winds, associated with the passage of weather fronts, resuspended deposited sediment from as deep as 3 ft in Lake Carl Blackwell, Oklahoma (Norton 1968). Alternating periods of flooding, dewatering, and resuspension may result in significant movement of sediment in reservoirs.

5.5.7 Fish Feeding

Although resuspension of sediment is associated mostly with wave action, the bottom-feeding activity of fish also contributes to resuspension (Meijer et al. 1990). Benthivorous fishes such as common carp, buffalos, and gizzard shad ingest sediment, from which food particles are retained by filtering through gill rakers (Lammens and Hoogenboezem 1991). The fine sediment particles that are not retained by the fish may become suspended in the water. Given that some fish species may process up to five times their body weight of sediment per day, the effect on turbidity can be considerable in waters with high fish densities (Breukelaar et al. 1994). Moreover, there may be an interaction between wave action and fish foraging on sediment resuspension. Foraging benthivores leave small pits in the sediment surface (Lammens and Hoogenboezem 1991). Observations of sediment in lakes where benthivorous fish are abundant often have shown the sediment surface to be almost entirely covered by foraging craters (Scheffer 1998). These disturbances to a consolidated top layer of sediment would facilitate the stirring effect of wave action by reducing the erosion resistance of the sediment. In an experiment conducted by Scheffer et al. (2003), the critical water velocity needed for resuspension roughly doubled two weeks after fish removal. Matsuzaki et al. (2007) demonstrated that common carp could have a dramatic influence on sediment and nutrient dynamics, resulting in a modification of the littoral community structure and triggering a shift from a clear-water state dominated by submerged macrophytes to a turbid-water state dominated by phytoplankton. Similarly, Schaus et al. (2010, 2013) reported large populations of benthic-foraging gizzard shad had a substantial effect on an entire lake ecosystem.

5.5.8 Suspended Organic Matter

Increased solids and associated increases in turbidity and reductions in water clarity also are caused by organic materials such as suspended organic matter and plankton. Unlike inorganic turbidity, organic turbidity can be driven by nutrient loading, warm water temperature, and the decomposition of dead plant material on the bottom that gives the water a brownish tint. Large quantities of allochthonous organic matter are washed into aquatic systems, and accumulated organic matter may be resuspended during floods and storms or washed from the floodplain (Bonetto 1975). Organic materials have lower density and lower refractivity relative to water, with the result that their light attenuation cross section peaks at larger particle sizes. Because of this size dependence of light attenuation by organic particles, phytoplankton cells contribute appreciably more light attenuation in natural waters than the often more numerous, but much smaller, bacterial cells.

5.6 Longitudinal Gradients

Reservoirs often may exhibit longitudinal turbidity gradients (Kennedy et al. 1982). High concentrations of suspended materials are imported from all tributaries

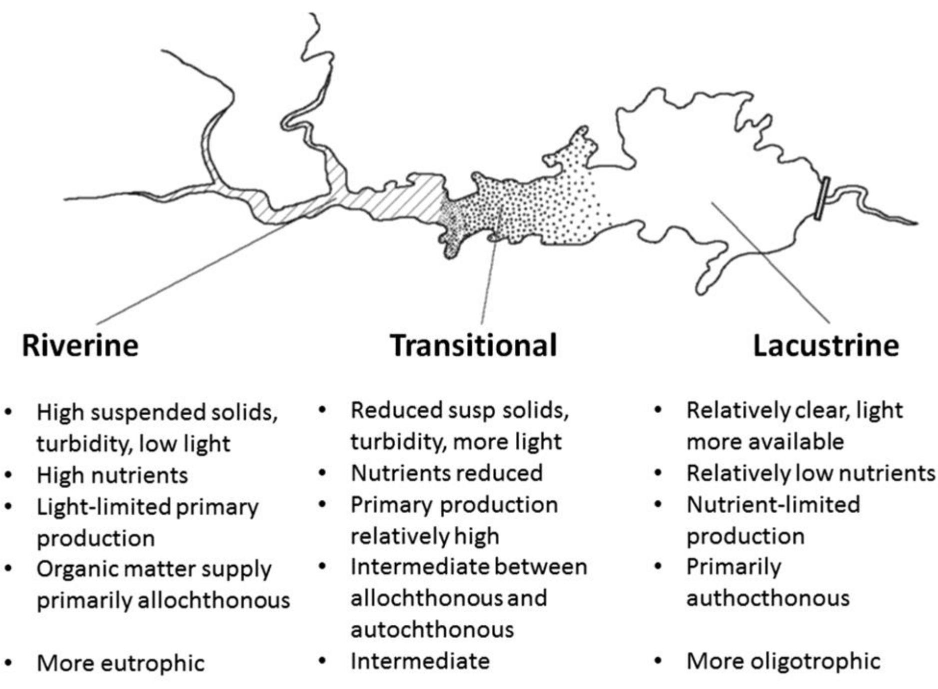


Figure 5.4. Riverine, transitional, and lacustrine sections in a reservoir as defined by Kimmel et al. (1990).

but especially the main river impounded by the reservoir. As these materials are deposited, a gradient of turbidity is established along the longitudinal axis of the reservoir. This process also can occur within single embayments. The length and strength of the gradient depends upon the hydrologic regime, season, interval since the last storm pulse, and the operation of the outflow at the dam. In West Point Reservoir, Alabama–Georgia, turbid waters entered the reservoir following storm events and were evident as surface plumes for up to 18 mi into the reservoir (Kennedy et al. 1982). These plumes often continued farther downstream as underflows or interflows. This longitudinal gradient in turbidity has direct effects on primary production along the longitudinal axis of many reservoirs.

Kimmel et al. (1990) described reservoirs as consisting of three regions along their longitudinal axis: riverine (uplake), transition, and lacustrine (Figure 5.4). Each of the regions is characterized by different water clarity, different causes of light attenuation, different nutrient regimes, and different biota. Kimmel and Lind (1972) also showed that the spatial differences within a reservoir not only occur longitudinally but also laterally because of differences in tributaries and associated embayments.

5.7 Biotic Effects

Reduced water clarity and transparency resulting from suspended solids has three main types of biotic effects: improved conditions for development of bacterial food webs, reduced penetration of light for photosynthesis (Kirk 1994), and reduced visual range of sighted organisms (e.g., Vogel and Beauchamp 1999). Loss of clarity also can have effects on human perception of the aesthetic qualities of water bodies (e.g., Smith et al. 1995a, b) and of their fishability. Effects of increased solids levels on aquatic life vary with the magnitude, duration and frequency of exposure, and the physical characteristics of the solids. These factors can result in decreased clarity and increased turbidity and affect the biotic composition of a reservoir.

5.7.1 Bacteria

The adsorption of dissolved organic carbon onto suspended mineral particles can subsidize a reservoir's food web and mediate the effects of suspended sediment in reservoirs with sufficient sources of autochthonous or allochthonous organic matter (Baylor and Sutcliffe 1963; Arruda et al. 1983; Gliwicz, 1986; Lind et al. 1997). Lind et al. (1994) attributed the greater productivity of fish in a highly turbid Mexican reservoir to subsidies to the food web provided by the bacteria-clay-organic aggregate pathway. The concentration of dissolved organic carbon adsorbed onto suspended clay particles as a consequence of their relatively large surface areas can create a concentrated food source for bacterial colonization and growth (Lind and Davalos-Lind 1991;

Lind et al. 1997). This deviation from the traditional heterotrophic microbial loop makes dissolved organic matter available to higher planktivores as particulate food, bypassing the intermediate link through heterotrophic nanoflagellates and larger protists (Lind et al. 1997), and thereby increasing use of dissolved organic carbon. Under these conditions, aggregate-associated bacteria may represent an important fraction of the total energy available to higher trophic levels.

5.7.2 Photosynthesis

Primary productivity, which includes mostly the growth of phytoplankton, periphyton, and aquatic plants, provides the base of the food chain in reservoir systems, influencing food available for invertebrates and fish. Primary productivity depends on the availability of light and nutrients, both of which interact with mineral or clay turbidity to influence primary productivity. Clays not only attenuate light needed for photosynthesis but also can deprive algae of nutrients by absorbing phosphorus from the water column and ultimately carrying it out of the photic zone into sediment (Heath and Franko 1988). Moreover, clays form complexes with dissolved organic materials and prevent microbial degradation (Lind and Davalos-Lind 1991; Tietjen et al. 2005). Phytoplankton composition reportedly varies among reservoirs with different turbidity levels and even within a reservoir along a turbidity gradient (Søballe and Threlkeld 1988). In Belton Reservoir, Texas, the phytoplankton assemblages at five sites from headwaters to dam were all taxonomically dissimilar with one another, with dissimilarity increasing progressively with distance (Lind 1984). Turbid reservoirs often fall short of expected levels of primary production and algal biomass predicted by nutrient loading models (Jones and Knowlton 2005). When trophic state indexes (section 4.2) were applied to Texas reservoirs, 44% were misclassified when chlorophyll-*a* and phosphorus data were used—phosphorus overpredicted chlorophyll-*a* (Lind et al. 1993). The shortfall is attributable to an unfavorable light climate or competition by clays for phosphorus.

Turbid reservoirs tend to have less submerged macrophytes and periphyton. High turbidity has been shown to reduce the density, growth, photosynthetic activity, and maximum depth of colonization of aquatic plants as well as causing physical damage to leaves (Chandler 1942; Robel 1961; Lewis 1973; Canfield et al. 1985; Kimmel et al. 1990). High mineral turbidity also has been shown to reduce the standing crop of periphyton, although high nutrient loadings can alleviate the effects of increased turbidity (Burkholder and Cuker 1991). Considering that increased mineral turbidity can promote flocculation and sinking of phytoplankton (Avnimelech et al. 1982; Guenther and Bozelli 2004), the importance of periphyton to lake primary production may increase in shallow reservoirs with high loading of sediment and nutrients.

5.7.3 Zooplankton and Invertebrates

High suspended sediment concentrations alter zooplankton assemblage composition and reduce abundance and biomass (Jack et al. 1993; Donohue and Garcia-Molinos 2009). Moreover, reduced population growth reportedly is a consequence of decreased survival and fecundity associated with increased mineral turbidity (Kirk and Gilbert 1990; Kirk 1992). Suspended sediment can reduce rates of feeding and the incorporation of carbon into zooplankton tissue (Hart 1988; Bozelli 1998), although this effect varied with the size of suspended particles and among zooplankton taxa. Interference of suspended sediment with feeding behavior seems to be the primary mechanism producing these patterns (Kirk 1992). Cladocera appear to be among the most susceptible zooplankton to high concentrations of suspended sediment. High filtration rates and greater size ranges of food generally enable cladocerans to outcompete rotifers in clear-water conditions (MacIsaac and Gilbert 1991). Large-bodied cladocerans, are commonly the dominant herbivores in clear-water lakes, but increased suspended sediment concentrations can reduce their feeding efficiency because of overlap between the sizes of their algal food and inorganic particles in suspension. Increased turbidity has been shown thereby to enhance the dominance of rotifers over cladocerans as rotifers are generally more selective feeders and can avoid ingesting large volumes of suspended sediment (Kirk 1991).

Nevertheless, dissolved organic carbon associated with suspended clay particles can be a source of food for zooplankton (Arruda et al. 1983; Gliwicz 1986). This source may compensate for the loss of phytoplankton due to light attenuation. Although the quantity and quality of organic matter available in association with mineral particles vary greatly depending on mineral composition and on environmental characteristics, for many reservoirs the ambient mineral turbidity is sufficient to at least provide food in excess of the starvation level (Donohue and Garcia-Molinos 2009). Gliwicz (1986) concluded that when there is a seasonal low in phytoplankton because of high turbidity, the organic carbon associated with suspended sediment is essential to zooplankton maintenance although less than the threshold concentration necessary for population growth.

5.7.4 Fish

Whereas massive fish mortality has been reported as a result of anoxic conditions associated with the resuspension of deposited sediment in shallow water, relatively high concentrations of suspended sediment and long exposures are required to cause direct mortality (Bruton 1985). However, exposure to high sediment loads over time may result in reduced feeding rates, reduced growth rates over several days, reduced biomass and population over months and years, and potentially indirect changes in community composition (Figure 5.5). Species associations in a large data set of Texas reservoirs were related to turbidity gradients (Dolman 1990). High turbidity limited standing stocks of large daphnids, resulting in food limitation of a planktivorous fish making up the majority of the fishery in a South African reservoir (Hart 1986). Larval shad and freshwater drum shifted their distribution and food intake within Lake Texoma when zooplankton density dropped during turbidity surges, potentially driving the fish population dynamics (Matthews 1984). The primary effect of turbidity on feeding by planktivorous fish may be to reduce water clarity and thus limit the depth at which fish are able to feed effectively (De Robertis et al. 2003).

Because many game fish are visual predators, much attention has been given to the effects of turbidity on their visual perception and foraging activity. In addition to reducing ambient light intensity, turbidity can impair visibility by degrading apparent contrast. Lythgoe (1979) hypothesized that increased turbidity and associated light scatter reduce the visual range of fish by degrading target brightness and contrast. High turbidity levels thus diminish feeding efficiency and, consequently, growth rates of visually predatory fish by reducing the reactive distance between predators and their prey at the time of detection (Barrett et al. 1992; Miner and Stein 1993). Decreased reactive distance in turbid waters thus results in smaller volumes of water searched per unit time and reduced encounter rates of both small and large prey (Utne-Palm 2002). Under moderate turbidity and high ambient light conditions, feeding performance and growth rates are frequently higher than those in clear water (Miner and Stein 1993; Bristow and Summerfelt 1994; Utne-Palm 2002). Moderately turbid water may increase the contrast of prey against their background and thus improve detection under sufficient light conditions (Hinshaw 1985).

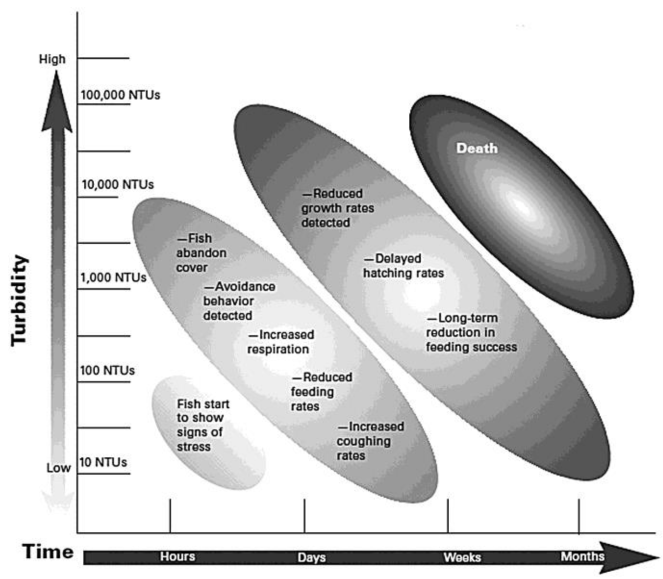


Figure 5.5. Relationship between turbidity (NTU = Nephelometric Turbidity Units) and fish activity relative to time. This model is based on Newcombe and Jensen (1996).

Turbid water may provide a refuge from potential predators (De Robertis et al. 2003). Visual fish predators tend to avoid turbid areas because of their lowered foraging ability and greater physiological stress whereas fish with good sensory adaptations to low light become predominant (Rodrıguez and Lewis 1997). This, in turn, reduces predator avoidance behavior in turbid areas (Gregory 1993; Lehtiniemi et al. 2005), and the consequent reduction in energy expenditure can then be invested in foraging for food, resulting in increased rates of feeding and growth. Consequently, turbid lakes may exhibit a reduction of visual predators such as black bass and an increase of prey species such as small sunfishes (Alferman and Miranda 2013).

Reductions in prey selectivity as turbidity increases have been reported for black bass. Reid et al. (1999) reported that juvenile largemouth bass selected small fathead minnows in laboratory studies at low turbidity, but selectivity disappeared as turbidity increased. Changes in turbidity can also affect the type of prey selected by piscivorous fish. At low turbidity levels (0–5 NTU), largemouth bass selected fish prey (i.e., showed neutral or positive electivity with respect to them) and avoided crayfish (Shoup and Wahl 2009). As turbidity increased to moderate levels (10 NTU), selection for gizzard shad declined and selection for crayfish increased. At the highest turbidity level tested (40 NTU), bluegills were selected. Carter et al. (2010) found that prey consumption by smallmouth bass decreased substantially as turbidity increased from 0 to 40 NTU. Hueneman et al. (2012) reported that higher turbidity levels reduced the ability of largemouth bass to capture prey and increased the time taken to locate and interact with prey.

A few studies indicate that turbidity does not affect some fish species. Rowe et al. (2003) found that the feeding rates of rainbow trout in New Zealand lakes did not decrease compared with controls at 160 NTU. However, the study found that in clear water, rainbow trout ate primarily larger prey, whereas this selectivity decreased as turbidity increased. In another study, growth rates of juvenile white crappie and black crappie were not affected by turbidity ranging from 7 to 174 Formazin Turbidity Units (FTU), and growth rates of adult crappie were not affected in 13–144 FTU treatments in 25-week studies (Spier and Heidinger 2002). Crappie generally are thought to be tolerant to changes in turbidity and other measures of water quality (Buck 1956).

5.7.5 Aesthetics

The various recreational services provided by reservoirs—fishing, swimming, boating, picnicking, and nature appreciation in general—are expectedly enhanced by the body of water's natural beauty. Egan et al. (2009) reported water clarity was a key variable shaping visitation to Iowa lakes. Nevertheless, whereas angler surveys often have identified aesthetics as an important component of the overall angling experience, surprisingly little information is available about the effect of water clarity on angler attraction. Perceptions of what is acceptable in a water body will depend upon the use to which it may be put and likely vary regionally depending on user expectations. Smith et al. (1995a) investigated the water clarity criteria for bathing waters based upon user perception. They found that bathing water-quality assessment was strongly related to visual cues, in particular water clarity. Minimum water clarity of about 5-ft Secchi disk depth is required before water is perceived, on average, as suitable for bathing. The National Technical Advisory Committee (NTAC 1968) recommended that a Secchi disk should be visible at a depth of 4 ft. This value subsequently has been included in several water-quality compilations (CCREM 1987). No such aesthetic targets have been established for fishing in reservoirs as fishing success can be high in low or high turbidity but with shifts in catch composition.

5.8 Water Clarity Management

Improving water clarity in reservoirs has focused on limiting inflows of turbid water, controlling shore erosion induced by wave action, and inducing flocculation of suspended sediment. Shore erosion and flocculation are considered below; limiting inflows of turbid waters and passing turbid water through the reservoir are discussed in sections 2 and 3. Where managing water clarity is not possible or control can take a long time, reservoir fish managers may focus on species that thrive in turbid conditions. The benefits and appeal of clear water generally are assumed, but some reservoir fisheries in fact may benefit from turbid water. The interactions between aesthetic values and fishery values have not been studied adequately.

5.8.1 Monitoring Considerations

Regular monitoring of turbidity can help detect trends that might indicate increased erosion developing in the watershed. Traditionally, methods used to monitor water clarity in reservoirs have been based on in situ measurements with meters or a Secchi disk or by collecting water samples and transporting them for laboratory analyses. These approaches, while generally accurate, are time consuming and do not easily lend themselves to understanding the spatial and temporal dimensions of water clarity within a reservoir. More recently, technology has been developed for continuous, sensor-based monitoring. This technology can allow for an increase in the number of sites monitored and an improved understanding of temporal patterns. Additionally, turbidity may be monitored through remote-sensing technology, which is evolving rapidly (Choubey 1997; Nellis et al. 1998).

Spatial aspects of sample allocation are important from a sampling design perspective. Given that reservoirs often receive a majority of their inflow from a single tributary located a considerable distance from the dam, the sampling design may include multiple stations located longitudinally along the reservoir. At a minimum, stations include the tributary and the dam. Another spatial aspect of reservoirs that may need to be considered is the pelagic versus littoral zones. These zones could have substantially different water clarity levels resulting from wind action, erosion, resuspension of bottom sediment, and possibly currents. Sampling stations also may need to be allocated to areas of special interest, such as key embayments or littoral areas potentially affected by riparian disturbances.

5.8.2 Shore Erosion Control

A major source of turbidity in many reservoirs is shoreline erosion (Figure 5.6). Not only does erosion contribute to reduced water clarity and increased sedimentation, but it also reduces suitability of shoreline habitat for vegetation and wildlife (Keddy 1983). Although the shallow aquatic zones may produce suitable habitat for some aquatic plant species, wave energy limits density, diversity, and distribution of aquatic vegetation on unprotected shorelines (Collins and Wein 1995; Luken and Bezold 2000), which, in turn, degrades habitat for invertebrates and fish. Erosion rates in reservoirs can be up to 20–30 ft/year (Khabidov et al. 1996) and can vary from <1 to 5 ft/year in small reservoirs and lakes (Vilmundardóttir et al. 2010). Saint-Laurent et al. (2001) found erosion rates of 3–5 ft/year with fetch distances of 7.5 mi. Rates on the order of 1–2 ft/year are common (Kirk et al. 2000).

Figure 5.6. Shore erosion is a major source of fish habitat degradation in reservoirs. Erosion not only destroys bank habitat but also blankets substrates with sediment and increases turbidity of the water. In the photo erosion in Brownlee Reservoir, Idaho-Oregon, was caused by seasonal water level fluctuations. Seasonal reservoir operations scour a thin layer of soil from the reservoir "walls." Photo credit: C. Welcker, Idaho Power Company.

Shore erosion control commonly has relied on protecting vegetation and installing structures to armor the shoreline. Vegetation not only prevents erosion but also has value for aesthetics, shade, and fish and wildlife habitat. Traditionally there have been two general types of installed structures: those that reduce the strength of water smashing against a shoreline, such as breakwaters and groynes, and those that increase the shoreline's resistance to erosive forces, such as revetments and seawalls. Breakwaters and groynes are similar, but they are each unique in their location and function. Breakwaters are typically found surrounding a shore, embayment, or harbor facility as they are primarily designed for limiting wave action. Groynes are structures positioned perpendicular to shore and are intended to trap sediment as a means of erosion control. More recently, a third type of installed structures, soft structures, is gaining popularity. Soft structures, or living shorelines, are an approach to shoreline stabilization that preserves vegetation in shorelines.

Figure 5.7. Breakwaters positioned at the mouth of an embayment to reduce wind-induced wave action originating in the reservoir. Photo credit: M. Porath, Nebraska Game and Parks Commission, Lincoln.

5.8.2.1 Vegetation protection

The protection of vegetation in riparian zones often is advocated as an environmental management tool for reducing effects of land-use activities on aquatic resources. The buffer zone generally is regarded as the belt of land that separates an upland or hillslope area from the reservoir. Land-use activity often is modified in this zone to prevent adverse effects on water. Management of riparian zones is a management tool used to perform many functions, including stabilizing shorelines and filtering sediment and nutrients—all of which improve water clarity. Section 8 discusses details about managing riparian zones.

5.8.2.2 Offshore breakwaters

Breakwaters are commonly rock or concrete block structures that cause approaching waves to break prematurely, creating a calm environment landward of the structure. Breakwaters can be attached to shore and built at an angle from the shore or detached and built nearshore and parallel to shore (Figures 5.7, 5.8). Built near erosion-prone shorelines in water about 3 ft deep, these structures create quiet water nearshore. They can be particularly effective when placed at the mouth of an embayment to stop exposure to wave action originating in the main reser-

Figure 5.8. Offshore detached breakwater. Photo credit: J. Sullivan, Getty Images.

voir. Structures are normally connected to the shore at intervals to exclude boats, and culverts are included to allow fish passage (section 9.2.7). In many cases these structures are a better alternative for stabilizing eroding shorelines than simply dumping rock riprap. Aquatic vegetation may grow between the structures and the shore, allowing both shore and boat anglers access to productive fishing water. Breakwaters also can serve as habitat for many fish species attracted to structures, and those connected to shore can be designed so they provide safe access to bank fishers.

The idea is to create something analogous to a barrier reef nearshore. This breakwater dissipates a wave's energy in deeper water before it can pick up bottom sediment and before it reaches shore and causes erosion. The protected water and shoreline then may be able to develop into a transitional wetland containing emergent and submergent aquatic vegetation. The tops of offshore breakwaters usually are constructed at a reservoir's normal pool elevation, notched to allow fish and fresh water to move between the protected and open areas of the lake, and marked with floating buoys or large individual rocks to alert boaters. Because winds also may blow parallel to shore and cause erosion to the shorelines behind the breakwaters, the breakwaters periodically are connected back to the nearest adjacent bank with low profile groynes. Some of these groynes also can be constructed to provide access and fishing opportunities.

The distance a breakwater is located offshore is varied depending on the distance wind can blow uninterrupted, which controls wave amplitude and therefore the depth of the base of the structure. Usually, a 35 mph sustained wind is used in the calculations. For example, on a reservoir with 2 mi of open-water fetch, a 35 mph wind produces waves that would begin affecting bottom sediment at a depth somewhere around 3 ft. Consequently, in most reservoirs ≤1,000 ac, an adequate depth for breakwater placement is usually 4 ft. Breakwater construction becomes more complex and costly when large water-level fluctuations occur, such as one might find in a flood control or irrigation reservoir.

Figure 5.9. Rock-log breakwater at Peterson Lake, Pool 4, Upper Mississippi River. Photo credit: USACE, Rock Island District.

5.8.2.3 Rock-log structures

In protected areas with minimal ice effects, rock-log structures provide an economical alternative to offshore rock mounds (Figure 5.9). These structures protect existing shoreline while providing woody structure for fish.

Figure 5.10. View of a floating breakwater, Lake Allatoona reservoir, Georgia. Photo credit: J. Chulick.

5.8.2.4 Floating breakwaters

If water is too deep or fluctuates substantially, a floating breakwater may be a better choice if wave action is not excessive (Figure 5.10). Floating breakwaters are typically used on limited-fetch water bodies where wavelengths are relatively short. Materials used for floating breakwaters include wood, barges, scrap tires, logs, and steel drums, as well as floating wetlands.

Several advantages of floating breakwaters have been identified over fixed structures. In deeper water (>10 ft) floating breakwaters are less expensive to install than fixed structures (Hales 1981). They effectively can attenuate moderate wave heights (<6 ft; Tsinker 1995). Floating breakwaters produce minimal interference on water circulation, sediment transport, and fish migration (Kelly 1999). They can be moved easily and rearranged in different layouts or transported to another site or away from icy conditions (Hales 1981). Floating breakwaters are not as obtrusive as fixed breakwaters and can be more aesthetically pleasing (McCartney 1985).

However, there are disadvantages to floating breakwaters. These floating structures are less effective in reducing wave heights for slow waves than fixed structures (a practical upper limit for the design wave period is in the range of 4 to 6 sec; Tsinker 1995). They are susceptible to structural failure during catastrophic storms (Tsinker 1995), and if the structure fails and is detached from its moorings, the breakwater may become a hazard (Kelly 1999). Relative to fixed breakwaters, floating breakwaters require a high amount of maintenance (Tsinker 1995).

There are an extensive number of different types of floating breakwaters. The various types may be seen as combinations of variations of materials, breakwater shape, mooring system, and function. These combinations generate a large list of permutations that can be divided into three basic groups: box, pontoon, and inflatable. A fourth type, mat breakwaters constructed out of discarded tires, is being used less often because of aesthetic issues and concerns about leachates.

Figure 5.11. Hex-box breakwaters in Potomac River. Photo credit: A. Russo.

Most box-type breakwaters are reinforced concrete, rectangular-shaped modules that may be flexibly or rigidly connected to other modules to make a larger breakwater (Figure 5.11). They are either empty inside or, more frequently, have a core of light material to promote flotation. Box breakwaters also may be constructed of steel. These structures have proved to be effective and have several uses, including recreational boat moorage. The main disadvantages for these structures are that they are expensive and require high maintenance.

Figure 5.12. Pontoon-type breakwater. Photo credit: Horseshoe Bend Docks and Rip Rap, Lake Ozark, Missouri.

Pontoon types (Figure 5.12) often serve multiple uses. These structures are ideal for uses such as floating walkways, boat moorings, and fishing piers (Hales 1981). Pontoon types are generally less expensive than box types and have similar advantages and disadvantages to the box type.

There are potential advantages to using inflatable structures as breakwaters. As opposed to a rigid breakwater, which absorbs wave energy by its mass and mooring system, inflatable breakwaters may absorb energy through the structure's deformations as well. When the breakwater is not needed, it may be deflated and stored. Some disadvantages may include the need for inflating and towing and the possibility that the structure will be punctured.

5.8.2.5 Groynes

Groynes (jetties, hardpoints) are piles of riprap, boulders, or concrete built perpendicular to shore to control littoral drift and arrest its effects on erosion (Figure 5.13).

Figure 5.13. Groynes (sometimes referred to as jetties or hardpoints) along a shore of a Nebraska reservoir. Groynes are positioned to reduce nonperpendicular wave energy and allow growth of macrophytes. Photo credit: Nebraska Game and Parks Commission, Lincoln.

They interrupt, slow, or redirect longshore currents and waves to accumulate sediment along shore on the updrift side. Because of their shore orientation, groynes function best in areas with stronger alongshore waves, such as when the fetch is parallel to shore. Groynes are commonly straight, linear structures, but they can have various shapes including a T or L shape. Newer groynes typically are constructed with armor stone, concrete blocks, or concrete modules. Older groynes used timber. A series of groynes may be preferred at an expansive location, creating a groyne field.

After groynes are constructed, shoreline reshaping occurs, with deposition near the groynes and erosion in the reach between two groynes. This continues until a stable scalloped shape is formed. To maintain a groyne or groyne field, periodic monitoring of the structure(s) is necessary. Repositioning or replacement of the armor units may be necessary to ensure the structure functions properly because excess sediment may build up on the updrift side of the groyne. A groyne can extend 40–50 ft offshore and have a top elevation of as much as 1-2 ft above the mean high water line. The ratio of groyne spacing to groyne length varies from 4 to 6. The advantage of groynes is cost savings (if in shallow water), creation of littoral and beach habitat, and an aesthetically pleasing shoreline.

Figure 5.14. Jetties protecting large embayments in Branched Oak Reservoir, Nebraska. Photo credit: Nebraska Game and Parks Commission, Lincoln.

At the mouths of embayments, large groynes (also often referred to as jetties) can be built from opposing shorelines, extending toward one another so that the opening between them is just wide enough to allow boat passage (Figure 5.14). These structures reduce wave action and shoreline erosion in the embayment and provide anglers access to clearer and calmer water.

5.8.2.6 Rock vanes

Figure 5.15. Rock vanes at Lost Island Chute, Pool 5, Upper Mississippi River. Photo credit: USACE, Rock Island District.

Rock vanes are effective on shorelines that experience moving current. Vanes extend upstream from the shoreline and feature a sloping top elevation with a sharp 45°–60° angle (Figure 5.15). As vanes are overtopped by the water, they function as weirs and redirect flow away from the shore. In many situations, vanes also function as groynes by reducing littoral drift due to wind-driven wave action.

5.8.2.7 Revetments

These are protective structures of rock, concrete, or other materials constructed with a sloping surface to break waves more gradually (Figure 5.16) than the vertical walls of bulkheads and seawalls (section 5.8.2.9). Revetments are constructed by grading the shoreline to an appropriate slope and installing layers of suitably sized rock or rock-like materials to maintain property landward of the structure. Revetment is typically installed high enough to withstand waves in extreme conditions and incorporate enough large stones that will maintain their position over time. Revetments are better wave barriers than vertical structures and generally cause less toe scour than do vertical walls. However, the need for a sloping surface generally creates a wide footprint that extends farther into shore.

Revetments are flexible and do not require special equipment. Damage or loss of rock is easily repaired, but the construction can be complex and expensive. The slope of the shoreline is typically 2:1 or flatter. Revetments are particularly useful in shaded areas where vegetation may be difficult to establish. Revetments protect only the land immediately behind them and provide no protection to adjacent shores. Erosion may continue on adjacent shores and may be accelerated near the revetment by wave reflection from the structure.

Figure 5.16. Shore revetted with riprap in Lake Lanier reservoir, Georgia, to prevent bank erosion induced by wave action. Photo credit: Marine Specialties, Inc., Gainesville, Georgia.

Rock riprap revetment consists of stones used to stabilize and protect the shoreline. The amount and size of the stones are dependent on the site and shoreline characteristics. Rock riprap may be used in conjunction with vegetation and soil bioengineering techniques to create an efficient, cost-effective, and more appealing alternative. By leaving exposed soil between the rocks on the shoreline, vegetation may grow and appearance of the shoreline can be enhanced. Revetments are inspected periodically for signs of scour at the top, base, or sides and repaired as needed.

An alternative revetment technique may be suitable in situations where waves approach the shoreline at an angle. Riprap may be placed in discrete piles at diverse spacing. This pattern of rock placement will provide hard points interspersed with eroded areas, producing a scalloped effect that will increase shoreline length while improving diversity of shoreline habitat. It produces an effect similar to that of groynes (section 5.8.2.5). This type of rock placement often requires less rock and less labor to spread the rock, resulting in cost savings. Because erodibility of soils, fetch, and other factors vary among sites, appropriate spacing between rock piles is a consideration.

5.8.2.8 Natural stone revetment

Riprap, the standard method of shoreline revetment, is effective but can be cost prohibitive, especially when long shorelines are involved. On many eroding shorelines, significant quantities of rock are left behind on the "beached" area of the shoreline as the vertical bank erodes. This process, referred to as natural armoring, leaves rock too heavy to be washed away by wind and wave forces. Unfortunately, this natural armoring is often inefficient and does not protect the eroding vertical bank sufficiently.

The rock remaining on the beach area of an eroding shoreline can be used to protect the banks of reservoirs whose pool elevations fluctuate too much to make vegetative methods practical. In addition, such rock is cheaper than quarried rock, which is purchased and hauled sometimes considerable distances, making standard riprapping too expensive. The rock remaining as a result of erosion and natural armoring can be used to protect eroding banks. The scattered rock can be collected and piled using a rock picker, a conventional piece of farm machinery (Figure 5.17). The equipment is available from farm implement dealers and can be purchased for $15,000-30,000 depending on size. A tractor is required to operate the

Figure 5.17. Rock often found scattered along the regulated zone can be collected with a rock picker and piled to construct stone reefs and/or shore reinforcements. Photo credit: Highline Manufacturing Ltd., Vonda, Saskatchewan.

rock picker. This method is practical only in regions of the country or areas of a reservoir where an eroding shoreline contains significant quantities of appropriately sized rock.

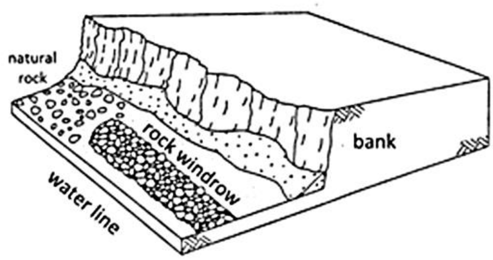


Figure 5.18. Natural rock collected with a rock picker and arranged into a windrow pattern. Image credit: USACE.

The scattered rock collected with the rock picker is arranged in a windrow pattern (Figure 5.18) on the beach area of the eroding shoreline. This method of bank protection can be timed to coincide with low pool elevations to allow collection activities. The windrow can be located some distance from the vertical bank because continued erosion, bank failure, or slumping may result in sediment accumulating behind the windrow. Thus, the windrow is typically located far enough from the vertical bank so that the weight of sediment accumulating behind the windrow will not force or push the windrowed rocks out of position.

5.8.2.9 Bulkheads and seawalls

Bulkheads and seawalls are terms often used interchangeably to describe similar shoreline protection structures. Both bulkheads and seawalls are vertical structures placed along the shoreline that retain soil behind the structure (Figure 5.19). Bulkheads are generally smaller and less expensive than seawalls. Bulkheads typically are made of wood and often provide minimal protection from severe wave action. Bulkheads are retaining walls whose primary purpose is to prevent bank slumping. Although they also provide some protection from wave action, large waves are usually beyond their design capacity. In contrast, seawalls are generally made of concrete or steel (or both) and are designed to withstand the full force of waves. They are usually constructed so that wave energy will not overtop the structure. Bulkheads are most applicable at locations where water depth is needed

Figure 5.19. Bulkhead designed to control wind-induced shore erosion. Photo credit: M&M Marine Construction, Queenstown, Maryland.

directly at the shore and a sloping revetment is not feasible.

Scour is a problem with vertical bulkheads and seawalls. As waves break against the structures, the wave energy is reflected both upward and downward, increasing current velocity around the structure and leading to scour at the base. The extent of the scour depends on the substrate, the orientation of the shore and the structure, the fetch length, the frequency of storms, and other factors. Generally, the scoured area deepens the shore in front of the structure. Because damaging scour can undermine the base and cause failure, toe protection is necessary for stability. Typical toe protection consists of rocks large enough to resist movement by wave forces, with an underlying layer of granular material or filter cloth to prevent the soil from being washed through voids in the scour apron. Also, groundwater percolating through the soil may build up pressure behind the wall and cause it to fail. Weep holes are spaced along the bottom of the structure to relieve the pressure.

5.8.2.10 Sills

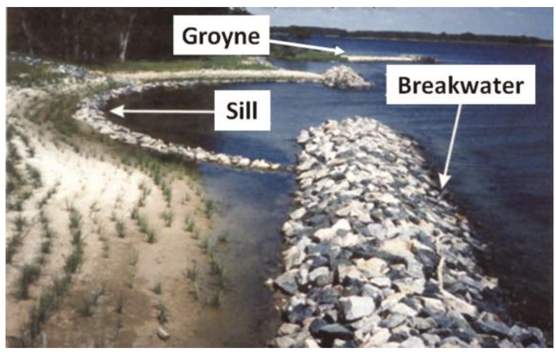


Figure 5.20. Three structural additions to maintain a mostly natural shore. Sills, breakwaters, and groynes are placed to enhance substrate buildup along the shoreline and allow development of wetlands. Photo credit: Wetlands Watch Inc., Norfolk, Virginia.

Sills combine elements of rock revetments, breakwaters, and living shorelines and are used in conjunction with natural or planted marshes. Sills are designed to maintain a wide marsh fringe, which acts as the primary erosion control device in the system (Figure 5.20). They are similar to breakwaters but are smaller and constructed closer to shore. Sills are low profile structures, generally no more than 6–12 in over the normal water level. Wind-induced waves pass over sills, and transport sediment into the structure. They can be constructed out of riprap or other natural materials. By trapping sediment and water between the edge of the structure and shore, sills can create marsh systems that protect shores from erosion.

5.8.2.11 Living shorelines

The term "living shorelines" encompasses a wide variety of environmentally friendly erosion control devices. When properly installed, living shorelines reduce and control eroding sediment. Living shorelines act as natural buffers, filtering pollutants and upland runoff and improving water clarity in the surrounding aquatic waters.

Living shorelines are designed to function as living space for fish and wildlife. They provide additional foraging and nesting areas for native species and often replace areas that were previously lost to erosion. Living shorelines also provide aesthetic value, enhancing views and creating wildlife viewing opportunities for landowners and the general public. Relative to costs, living shorelines can be competitive or cheaper in low wave energy environments compared with traditional armoring approaches to shoreline protection.

5.8.2.12 Tree felling

Felling large trees is discussed in section 8.9.5 as a method to provide woody structure along a shoreline. In areas where trees in the riparian zone are numerous, the hinge-cutting method of felling selected trees can provide shore erosion control. The technique involves cutting selected trees near their base just deep enough so that the tree can be pushed into the water but remain attached to the trunk. Hinge-cut trees cut about two-thirds of the way through the trunk may continue to live for a period of time. Alternatively, a steel cable may be used to secure the tree to the stump and keep it from floating away. A cable may be required by the reservoir control authorities.

The upper section of trees lying in the water will reduce wave action and catch debris, which will slow down shoreline erosion. Not all tree species will live long under such conditions; however, most willows will flourish and produce thick clumps of new growth with this treatment. This method protects the shoreline from erosion and can also provide desirable shallow-water fish habitat. This method is low cost and does not require a large amount of labor. However, removing trees from a shoreline can also enhance erosion, so candidate trees are selected with care and limited to those that occur in high density.

5.8.2.13 Boat traffic ordinances

Besides disturbing sediment by creating turbulence and wave action (section 5.5.5), waves created by boat traffic also can erode banks along the shore. No-wake zones in shallow areas of reservoirs can help reduce these effects by reducing the overall amount of boat activity in these areas and by limiting the effects from high-speed boats. In certain cases it may be beneficial to restrict boat activity altogether, such as in extremely shallow waters where boats can generate substantial waves even at no-wake speeds.

The types of ordinances that may be enacted include (1) restrictions on speed; (2) restrictions on certain types of boating activities on all, or in specified parts, of the reservoir; and (3) restrictions on certain types of boating activities during specified seasons or water levels. Speed restrictions designated in miles per hour are difficult to enforce; a slow-no-wake restriction is preferable where appropriate. Ordinances controlling boating activity may be subject to review by the various agencies with jurisdiction over the reservoir, law enforcement organizations, and governing entities. Regulation by boat size, type of boat, or horsepower has been considered an unwarranted restriction of public rights in previous court rulings (WDNR 2016). Use restrictions in general can be unpopular with some user groups. As a result, use restrictions can lead to litigation, requiring consultation with staff attorneys before enactment and careful adherence to due process.

5.8.3 Flocculation

Flocculation is a way of controlling clay turbidity by adding substances to water that facilitate the formation of bridges between particles, allowing them to combine into groups of small particles that precipitate. Colloidal soil particles are negatively charged and repel each other so that they do not settle out. Introduction of positively charged electrolytes partially neutralizes the electrical field around the colloids, reducing the strength of repulsion between particles (Boyd 1979). In general, the effectiveness of electrolytes increases with the number of positive charges in the electrolyte.

Hay, cottonseed meal, and other organic matter have been used to remove clay turbidity, but their effects are not highly predictable and several weeks often elapse before a treatment may have an effect. These organic substances biodegrade when introduced into the water and release carbon dioxide, which combines with water molecules to produce positively charged ions. No method is available for determining the amount of organic matter to apply per unit volume. The large amount of organic matter often needed is expensive, and considerable labor is required for its application. Organic matter added for turbidity removal decomposes, exerts an oxygen demand, and lowers dissolved oxygen concentrations.

Electrolytes have been used widely in environmental engineering to remove clay turbidity from water supply reservoirs, potable waters, and settling basins (Ree 1963; Sawyer and McCarty 1967). Aluminum sulfate (alum) is a common source of positive electrolytes often employed for turbidity removal (Figure 4.6). Other coagulants, although not nearly as effective as alum, include ferric sulfate, calcium hydroxide (hydrated lime), and calcium sulfate (gypsum). More details about use of electrolytes are given in section 4.4.3.9.

Although treatment with electrolytes will clear water of clay turbidity, applications are probably practical only in small reservoirs or small sections of large reservoirs. Also, these treatments do nothing to correct the cause of turbidity. Unless the sources of the turbidity are eliminated, results of treatments may be short lived.

5.8.4 Fishery Management

As reservoirs become turbid their fish communities may shift. Fish species that rely on clear water for feeding or reproduction may decline or are limited to selected clear-water refuges in the reservoir. Stocking these waning species to boost their abundance may have few lasting benefits if quiet, clear water with aquatic vegetation is being replaced by wind-swept, muddy water. Reservoirs in this stage are best suited for species that do well in turbid waters such as catfishes, perches, crappies, and temperate basses (Figure 5.2). Thus, dealing with turbidity may require changing the emphases placed on various fisheries and possibly careful consideration of nonnative species.

Although reduced water clarity associated with sediment resuspension is caused mostly by input from tributaries and wave action, the bottom-feeding activity of fish also may cause a resuspension of sediment (Meijer et al. 1990). Benthivorous fishes such as gizzard shad, common carp, and smallmouth buffalo ingest sediment, from which food particles are retained by filtering through gill rakers. The fine sediment particles that are not retained by the fish become suspended in the water. Given that these fish may process up to five times their body weight of sediment per day, the effect on turbidity can be considerable in waters with high fish densities (Breukelaar et al. 1994). In the absence of benthivorous fish, lake sediment may consolidate rapidly during periods with little wave action, and the sediment may become firm enough to resist the shear stress caused by waves during windy periods (Scheffer et al. 2003). Extensive removal of benthivorous fish (Barton et al. 2000; Søndergaard et al. 2008) coupled with fish barriers to limit movements can be efficient tools to create clear water (Bulow et al. 1988; Meronek et al. 1996; Barton et al. 2000; Søndergaard et al. 2008). However, periodic fish removal presumably is required to obtain long-term effects.

5.8.5 Aesthetics

The aesthetic value of clear-water reservoirs is often assumed, yet no robust study of this value has been reported in the reservoir literature. Many turbid reservoirs with seemingly low aesthetic value produce excellent fisheries that attract large number of anglers despite high turbidity. Additional research is needed to dissect the interaction between loss of aesthetic value due to reduced water clarity and high catch potential.

Section 6

Water Quality

6.1 Introduction

Flowing water stored in a reservoir undergoes various physical and chemical transformations that can change the quality of fish habitat within the reservoir and the river downstream. The extent of the transformations is related to the water retention time in the reservoir, which is controlled by the reservoir's storage capacity in relation to size of the watershed and extent of precipitation. Water in small and shallow navigation reservoirs in a large river generally undergo little or no transformation because of short retention time; conversely, water stored for many months or even years behind a deep storage reservoir in a minor tributary may undergo major transformations that can affect most life in the reservoir and in the river below the dam.

Water-quality degradation can vary widely in reservoirs. Dissolved gases and temperature often receive the most attention. Dissolved oxygen is necessary to sustain aquatic life, and temperature regulates biotic growth rates. Both temperature and dissolved gases control other physical characteristics of the water as well as chemical reactions and define the biotic, and more specifically the fish, assemblage that develops in the reservoir. Nutrient enrichment principally with phosphorus and nitrogen promotes excessive primary production, which can deplete oxygen (section 4). Contaminants including organic chemicals and trace metals are of concern because they accumulate within sediment and move through food chains and food webs (Erickson et al. 2008; Stahl et al. 2009). Total dissolved solids concentrations may be of interest for water supply and other uses. Turbidity is also a key water-quality characteristic because its effects on light transmission and water clarity define habitat characteristics (section 5). A major aspect of turbidity is total suspended solids, which are also a major transport and deposit mechanism for nutrients and contaminants in reservoirs (section 4). Water pH regulates aquatic chemistry, which can affect water use and habitat. The reservoir hypolimnion is often low in dissolved oxygen and can accumulate dissolved phosphorus, iron, manganese, and sulfide, which can produce water-quality problems within the reservoir and downstream if hypolimnetic water is upwelled or discharged. Upwelled phosphorus (internal loading) can cause summer-time algal blooms. Iron and manganese affect water color and can produce water treatment problems when water is withdrawn for municipal uses. Sulfide causes odor problems when it escapes during reaeration.

Water quality in reservoirs often can be controlled by depth-selective withdrawals. However, in many cases discharges create water-quality problems downstream in the tailwater (Miranda and Krogman 2014). Water-quality management alternatives include management techniques that can be implemented in the watershed before potential pollutants reach the reservoir (sections 2 and 8) and management techniques that may be applied within the reservoir. Other in-lake management practices include phosphorus inactivation and sediment oxidation, biomanipulation, hypolimnetic aeration, artificial circulation, and sediment removal and are discussed below.

6.2 Temperature

A reservoir's annual temperature regime is perhaps the most important water-quality attribute, capable of influencing various other water-quality characteristics. Thus, knowledge of the temperature regime is key to water-quality management. Vertical thermal variation in the reservoir produces density stratification (Elçi 2008). In the less dense epilimnion, temperature decreases slowly with depth as this upper layer is usually influenced by the depth to which light penetrates. The metalimnion separates the epilimnion from the more dense deepwater hypolimnion. In the metalimnion there is a thermocline where temperature changes rapidly with depth. Below the metalimnion, in the hypolimnion, temperatures again change slowly with depth.

The physical properties of water contribute to this temperature-induced density stratification (Wetzel 2001; Elçi 2008). Because water warmer than 39 °F (4 °C) is less dense, the warmer waters usually exist near the surface. But because water colder than 39 °F is also less dense, some temperate reservoirs have colder waters at the surface mostly during the winter. Under both of these conditions, a reservoir may be thermally stratified. During the fall, the reservoir cools at the surface and experiences cooler inflows. At the surface the cooling produces shallow instability, and mixing can occur at progressively greater depths. This process continues through early winter. When the reservoir finally achieves, thorough cooling and mixing, a uniform temperature surface to bottom, the reservoir is said to be isothermal. This condition usually occurs in winter and continues until spring. These reservoirs experience one season of mixing each year (i.e., monomictic reservoir).

Besides water temperature, water density is also determined by dissolved substances in the water (Wetzel 2001). Increased dissolved solids increase water density. In reservoirs, dissolved solids are influenced by natural watershed characteristics and by anthropogenic activities in the watershed. In some reservoirs the density of deep waters is high enough that complete mixing does not occur (i.e., meromictic reservoir). Meromictic reservoirs are often deep (Wetzel 2001).

Many northern reservoirs stratify during winter (Rahman 1978). Their surface temperatures will be at or near freezing, and often the reservoir may be covered with ice, with slightly warmer and denser water toward the bottom. Because these reservoirs mix in the fall until they experience winter stratification and then mix again in the spring thaw until they restratify in summer, they have two seasons of mixing each year (i.e., dimictic reservoir). This condition also may occur occasionally during winter in shallow, isolated embayments of reservoirs in more southern temperate latitudes.

However, some reservoirs may experience short-term thermal stratification followed by frequent mixing (polymictic), or not stratify at all. Shallow depths, short retention times, and extensive wave-induced mixing may prevent stratification. If no stratification occurs, then the reservoir may consist exclusively of an epilimnion with possibly a weak thermal gradient, and the reservoir consists primarily of warm-water conditions.

6.3 Dissolved Oxygen

Next to temperature, dissolved oxygen is another key indicator of reservoir water quality. The concentration of dissolved oxygen in the reservoir affects the capacity of inorganic substances to reduce other substances as well as the distribution of aerobic and anaerobic organisms. The dissolved oxygen demand may be classified into sediment oxygen demand and water column demand (Cross and Summerfelt 1987).

Sediment oxygen demand is the rate of oxygen consumption by bacteria and other organisms that metabolize organic matter in the sediment. Sediment oxygen demand is usually greatest in the uplake region of the reservoir, or in the entrance to embayments where most sediments are deposited and water is warm and shallow (section 3). Organic matter in sediment in these upper regions is high, and hypolimnetic dissolved oxygen can be depleted, particularly given that the volume of water is typically small because the uplake regions of a reservoir are often shallow (USACE 1987c). Nevertheless, these upper regions do not always stratify because their shallow depth may allow mixing by wave action and sometimes by limited flows.

Water column oxygen demand is the rate of oxygen consumption by bacteria and other organisms that metabolize organic matter in the water column above the sediment. Density flows transport oxygen-demanding material into the metalimnion and hypolimnion and entrain reduced chemicals from upstream anoxic areas to augment water column oxygen demand. Organic compounds, including those in dead organisms from the epilimnion, settle more slowly in the metalimnion and hypolimnion because of increased water density caused by lower temperatures. Because this organic matter can remain in the metalimnion and hypolimnion for a long time, decomposition occurs over a long period, exerting long-term oxygen demands (USACE 1987c).

A threshold concentration of 4–5 ppm often is used to set dissolved oxygen water-quality standards. Hypoxia results when dissolved oxygen concentrations fall to less than 2 ppm, which is generally accepted as the minimum level required to support most animal life. When dissolved oxygen levels become severely depressed or anoxic (near 0 ppm), anaerobic conditions occur. Although anaerobic conditions typically occur in near-bottom waters, it can extend upward through the entirety of the water column. Hypoxia has been shown to be an endocrine disrupter in fish, which impairs fish reproduction (Wu et al. 2003).

6.4 Carbon Dioxide

Inorganic carbon derived from carbon dioxide is used by plants to produce organic matter. Inorganic carbon also can control the pH and buffering capacity of aquatic systems. Inorganic carbon occurs in equilibrium in three forms (Figure 6.1): carbon dioxide, bicarbonate ions, and carbonate ions (USACE 1987c). When plants use inorganic carbon to create organic compounds through photosynthesis, the pH goes up, and the concentrations of carbon forms shift from carbon dioxide to bicarbonate ions and to carbonate ions (Wetzel 2001). The scope of this pH rise and shift in the forms of carbon indexes the buffering capacity of the water. A system with low buffering capacity (i.e., with low alkalinity) is likely to have larger fluctuations in pH and shift quicker from carbon dioxide to bicarbonate ions and carbonate ions than systems with higher alkalinity (Wetzel 2001).

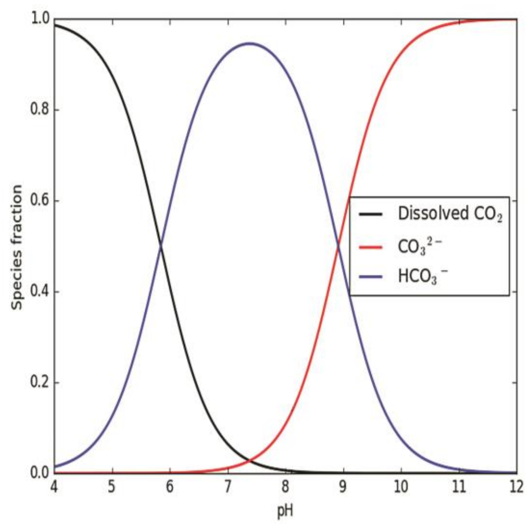


Figure 6.1. Proportion of concentrations of carbon dioxide (CO_2), bicarbonate ions (HCO_3^-), and carbonate ions (CO_3^{2-}) relative to pH.

6.5 Phosphorus and Nitrogen

Phosphorus is needed by plants and animals to build enzymes and to store energy in organic compounds, whereas nitrogen is needed to build protein. As a gas, nitrogen is important to water quality mostly when there is too much of it (i.e., supersaturated). Supersaturation can cause injury or death of aquatic organisms, including fish. This is not a common problem in most reservoirs, but supersaturation can harm fish below some hydroelectric facilities (Weitkamp and Katz 1980). Phosphorus and nitrogen are typically the key nutrients controlling primary production and are considered in detail in section 4.

6.6 Sediment Contaminants

Sediment plays a significant role in shaping the water quality of reservoirs. Benthic habitats are an environmental sink for many contaminants, as many pollutants sink through the water column to bond with sediment particles (Horowitz 1985; Mulligan et al. 2001). Because sediment is also an important biological habitat, uptake of toxicants into the food web is influenced by toxicant concentrations in sediment. Pollutants that settle out into sediment exist in an equilibrium state with the water above, but this equilibrium may be altered by natural and anthropogenic environmental disturbances (Theofanis et al. 2001; Jaglal 2009).

In polluted waters, pollutants mainly are found adsorbed by particles and bound to organic sediment (Chapman 1992; Baldwin et al. 2002). Changes in environmental conditions alter the various phases of pollutants found on particulates, sometimes causing the pollutants to be released into solution. Various forms of organic matter can be degraded under oxidizing conditions, releasing bound pollutants (Chapman 1992). Particulate pollutants may also become soluble within the acidic digestive tracts of detritivores such as gizzard shad, releasing these substances within the animal and making bioaccumulation through predation possible (Eagles-Smith et al. 2008).

As a result of changing environmental conditions, including dissolved oxygen and temperature, there is an internal recycling of pollutants in the sediment and hypolimnion (Baldwin et al. 2002). The processes are complex and poorly understood. The greatest amount of information is available for mercury and phosphorus. The transfer of mercury from sediment is mediated by bacteria that convert sediment-bound mercury to soluble mono-methylmercury or to volatile di-methylmercury, depending upon the pH (Erickson et al. 2008). The recycling of sediment-bound phosphorus (i.e., internal phosphorus loading) is particularly important because it may increase the rate of eutrophication within a reservoir. Many environmental and physical processes are involved in the release of phosphorus. A common process is the release of phosphorus bound to iron oxide under reducing conditions found in sediment. When bottom waters are anoxic, interstitial phosphate diffuses to the overlying water, increasing the rate of eutrophication.

6.7 Metals

Natural waters normally contain low concentrations of metals, but anthropogenic sources increase the concentrations above natural levels (Rosales-Hoz et al. 2000). Heavy metals often are discharged or leached from industrial point sources, mining operations, municipal wastewater, landfills, and the atmosphere. The problem is aggravated by the fact that there is no natural degradation process for eliminating metals from the environment. Metals shift from one compartment within the aquatic ecosystem to another, including biota, often with detrimental effects (Hart and Lake 1987). Where sufficient accumulation of metals within biota occurs through food chain transfer, there is an increasing toxicological risk through fisheries. As a result of adsorption and accumulation, the concentration of metals in sediment may be much higher than in the water above.

The behavior of metals in natural waters is influenced by the substrate sediment composition, suspended sediment composition, and water chemistry. Sediment composed of fine sand and silt will generally have higher levels of adsorbed metal than will quartz, feldspar, and detrital carbonate-rich sediment (Yu et al. 2012). Metals also have a high affinity for humic acids, organo-clays, and oxides coated with organic matter (Connell and Miller 1984). The water chemistry of the system controls the rate of adsorption and desorption of metals to and from sediment. Metals may be desorbed from the sediment if the water experiences decreases in pH, such as when anoxic conditions develop. Desorbed metals return to the water, where they recirculate and may be assimilated by the biota.

6.8 Organic Contaminants

Thousands of organic compounds enter water bodies as a result of human activities. These compounds have a wide variety of properties and many may be toxic. Common organic pollutants include mineral oils, petroleum products, phenols, pesticides, polychlorinated biphenyls, and surfactants (Perelo 2010). Although some of these degrade rapidly in the environment, others accumulate in bottom sediment and bioaccumulate to toxic concentrations within the food web.

6.9 Longitudinal Gradients

The water quality in the reservoir is related to various longitudinal, morphological, and hydrological processes. In their uplake regions, reservoirs are shallower and narrower and may maintain higher levels of flow, all of which may affect water chemistry. Moreover, these upper sections tend to trap large organic matter and debris particles that generally settle in uplake deltas with coarse inorganic particles such as sands and gravels. Finer particulate inorganic and organic matter tends to settle farther down the reservoir. Overall, the smaller the particle size, the greater the surface area to volume ratio. This increased ratio increases the sorptive capacity for transporting phosphorus, organic carbon, metals, and contaminants (USACE 1987c). Clays have a high sorptive capacity, whereas sand has essentially no sorptive capacity. Consequently, nutrients, metals, and contaminants may move into and out of the reservoir joined to fine silts and clays. Longitudinal patterns in hydrology, morphology, and settling can cause longitudinal water-quality clines.

Development of anoxic conditions within the reservoir often follows a longitudinal pattern. The pattern is dictated by local morphometry conditions and nutrient deposition patterns. Anaerobic processes may begin in the uplake portions of the reservoir if organic matter accumulations from the inflow are high and progress downstream (USACE 1987c). Conversely, anaerobic processes may develop in deep water by the dam and progress upstream (USACE 1987c). However, both upstream and downstream movement patterns can co-occur.

6.10 Depth Gradients

As dissolved oxygen concentrations decrease in the hypolimnion to about 1–2 ppm, the oxygen conditions at the water–sediment interface can become anoxic, and anaerobic processes begin to emerge in the sediment interstitial water (USACE 1987c). Nitrate denitrification to NH_4 (ammonium), N_2O (nitrous oxide), and N_2 occurs first (Bonin 1996; Wetzel 2001). As a result, ammonium-nitrogen can build up in the hypolimnion. Denitrification is the major mechanism for transferring nitrate out of the hypolimnion.

After denitrification, manganese compounds in the interstitial water are reduced to soluble forms able to mix with water in the hypolimnion. Thus, nitrate reduction eventually allows manganese reduction. As the system becomes further reduced, iron is transformed from ferric form to soluble ferrous forms and diffuses into the hypolimnion. As the iron is transformed, phosphorus associated with ferric compounds is released. Thus, sediment is typically a major phosphorus supplier when the hypolimnion is anoxic. During this anaerobic period, bacteria decompose organic matter into acids and alcohols, such as acetic, fulvic, humic, and citric acids and methanol (USACE 1987c).

The potential benefits of maintaining an oxic and cool hypolimnion are numerous. In some cases, cool, oxic hypolimnetic water is essential to satisfy the needs of in-lake and downstream biota. Maintenance of oxic conditions generally decreases sediment release of phosphorus and ammonia, thereby slowing eutrophication. Primary sources of phosphorus include iron complexes and microorganisms that release orthophosphate during metabolism under anoxic conditions. Oxic conditions can stimulate sediment nitrification and subsequent denitrification, resulting in a net loss of nitrogen from the system (Ahlgren et al. 1994; Rysgaard et al. 1994). Oxic conditions can also stimulate bacterial growth, resulting in increased rates of nitrogen assimilation (Graetz et al. 1973).

6.11 Water-Quality Management

Water-quality enhancement opportunities occur in the watershed and in the reservoir. In the watershed, protection techniques focus on agricultural and livestock farming, forestry practices, and other human activities. Watershed management is discussed in section 2.

There are in-lake enhancement techniques suitable for improving water quality. These focus on reducing the effect of an anoxic hypolimnion and management of contaminants. The technologies available to manage an anoxic hypolimnion can be separated into (1) those that prevent an anoxic hypolimnion by mixing hypolimnetic and epilimnetic waters to avoid stratification, and (2) those that prevent an anoxic hypolimnion through aeration (or oxygenation) but still maintain a distinct hypolimnion. Mixing hypolimnetic and epilimnetic water will break temperature stratification and reduce the habitat available for coldwater fishes but increase the habitat for warmwater fishes. Maintaining an oxic hypolimnion through aeration will preserve temperature stratification and provide low-temperature habitat to coolwater or coldwater fish species, prevent fish kills potentially caused by rapid turnovers that mix the epilimnion and hypolimnion, and also prevent discharge of anoxic water into the tailrace.

Hypolimnetic aeration can be beneficial to fish populations in the reservoir. During summertime in many deep reservoirs, coldwater or coolwater fish have inadequate habitat and survive between a layer of anoxic bottom water and warm surface water (Coutant 1985). By aerating the hypolimnion, fish are provided an oxygenated coolwater summer refuge. Oxic hypolimnia also may provide fish and zooplankton a dark daytime refuge in which to avoid predation (Fast 1971; Field and Prepas 1997). Also, benthos diversity and density tends to increase with oxic conditions in the sediment (Pastorok et al. 1981; Doke et al. 1995). A decrease in internal nutrient loading combined with improved zooplankton habitat under aerated conditions may cause a decrease in algal biomass or a shift to more desirable phytoplankton species (section 4).

6.11.1 Monitoring Considerations

Monitoring a reservoir's water quality can be an expensive and time-consuming task (Bartram and Balance 1996; Green et al. 2015). Whenever possible, water quality data may be obtained by partnering with agencies whose mission is water-quality monitoring. These may include the agency that controls the water stored by the reservoir, local and state departments of the environment, federal agencies including the U.S. Environmental Protection Agency (USEPA) and the U.S. Geological Survey, and universities. Section 305(b) of the Clean Water Act requires states to develop an inventory of the water quality of all water bodies in the state and to submit an updated report to the USEPA every 2 years. This process was established as a means for the USEPA and the U.S. Congress to determine the status of the nation's waters. The 305(b) report includes an analysis of the extent to which water bodies comply with the “fishable/swimmable” goal of the Clean Water Act; an analysis of the extent to which the elimination of the discharge of pollutants and a level of water quality achieving the fishable/swimmable goal have been or will be attained, with recommendations of additional actions necessary to achieve this goal; an estimate of (1) the environmental effects, (2) the economic and social costs, (3) the economic and social benefits, and (4) the estimated date of such achievement; and lastly, a description of the nature and extent of nonpoint sources of pollutants and recommendations of programs needed to control them—including an estimate of the costs of implementing such programs. However, the Clean Water Act is limited to waters with a significant nexus to navigable waters, and agriculture nonpoint discharges are generally exempted from regulatory oversight through the Clean Water Act.

In the absence of existing data or collaboration opportunities, agencies tasked with managing fish habitat may opt for narrowly focused monitoring programs, such as a program focused on temperature and oxygen. Suitable temperature and oxygen conditions generally will limit problems associated with toxic conditions and associated compounds, including ammonia, sulfide, manganese, and metal compounds. Monitoring of nutrients and water clarity is considered in sections 4.4.1 and 5.8.1.

Monitoring temperature and dissolved oxygen is probably most effective in May through September as reservoirs begin to warm and potentially stratify. Sample stations may be established at the deepest part of the reservoir (usually near the dam) or at the midpoint of the reservoir. Additional sample locations may be needed if the reservoir is long, there is interest in the status of various embayments in the reservoir, major inflows occur within the reservoir at various locations, or the lake is divided into significant subunits by causeways (Green et al. 2015).

Water column profiles can be taken with multiparameter sondes or other field meters (Green et al. 2015). Variables including dissolved oxygen concentration, percent oxygen saturation, and temperature can be recorded at regular depth intervals (OEPA 2010; Green et al. 2015). Other useful data often available from sondes may be recorded. The first reading may be taken at the surface (1 ft) and subsequent readings at suitable intervals proportional to depth of the sampling site.

6.11.2 Destratification

Monitoring may identify the need to avert stratification. Mixing of the reservoir to destratify layers or prevent stratification is accomplished with three general procedures: aeration, pumping, and hypolimnetic withdrawals. Mixing will help produce nearly even oxygen and temperature conditions throughout all depths. These conditions will affect in-lake water quality and that of water released through the dam. Mixing can limit phytoplankton blooms by reducing the amount of sunlight reaching phytoplankton by causing plankton to recirculate below the photic zone. Destratification will change the amount of habitat available for warmwater and coldwater fishes. Destratification will also warm the release waters, which may affect the use downstream. Lorenzen and Fast (1977) outline some additional benefits and consequences for reservoir destratification.

Johnson (1984) and Singleton and Little (2006) provide overviews of the various types of aeration–destratification systems, how they operate, and guidelines for system selection based upon the reservoir characteristics and the goals of the system to be installed. Whereas equipment technology continues to improve, the general approaches have remained the same. In general, the system design depends on the volume of water to be mixed and the temperature and oxygen profile. Equipment alternatives are available regardless of project scale (e.g., area, depth, oxygen demand) requirements (Johnson 1984).

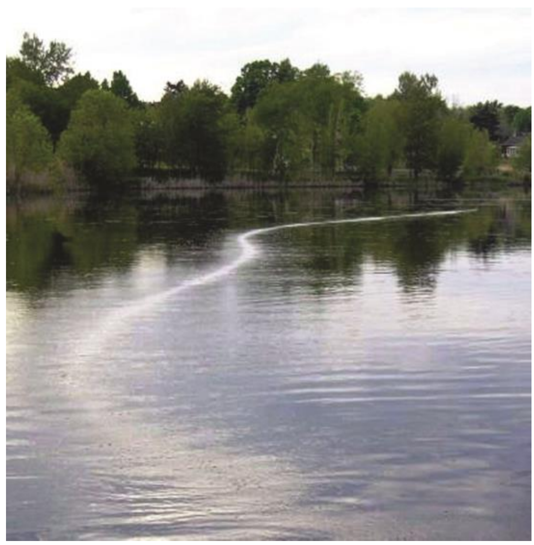


Figure 6.2. Oxygenation with a linear diffuser system. Photo credit: Diversified Pond Supplies LLC, Wapakoneta, Ohio.

6.11.2.1 Diffused aeration

Destratification of impounded waters may be achieved with diffused air aeration. The diffusers are normally flexible tubes that are installed on the bottom of the reservoir (Figure 6.2). The air bubbles move upward, creating an upwelling of cold water that spreads out laterally upon reaching the surface while carrying anoxic water upward. Once cold, aerated water reaches the upper layers, it sinks back again, bringing oxygen to the hypolimnion. This action causes the reservoir to destratify. Normally, compressed air is used. In general, linear or circular diffusers strategically positioned on the reservoir bottom are supplied by a compressor located on shore or fixed within the reservoir in a floating system (Singleton and Little 2006).

6.11.2.2 Mechanical flow pumps

Mechanical flow pumps provide enough mixing in a local area to reduce or eliminate thermal stratification (Mueller et al. 2002; Gafsi et al. 2009). Three general types of mechanical flow pumps have been used (Gafsi et al. 2009). The first one employs a water pump located on a floating platform or on shore. A pipe extends into the hypolimnion. Water is drawn from the hypolimnion, passes through the pump, and is discharged into the epilimnion or back into the hypolimnion (Figure 6.3) where it mixes (Hooper et al. 1953; Fast 1994). The second type of mechanical water pump consists of a motor located on a moveable float. A tube extends from the float into the hypolimnion, and a propeller and shaft extend into the tube (Symons et al. 1967). The propeller draws water into the bottom of the tube, and it is forced to the surface and discharged. Symons et al. (1967) compared the efficiency of this system with diffused aeration and found the latter more efficient. A third type of pump jets surface water down into the hypolimnion to achieve a locally uniform vertical temperature profile. A limitation is that the jet may strike the bottom and cause resuspension of sediment and erosion.

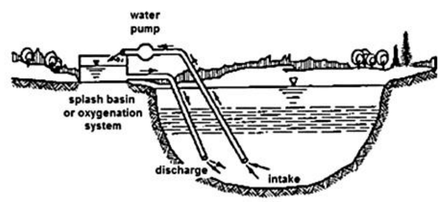


Figure 6.3. Sketch of a mechanical flow pump system. Modified from Fast et al. (1975).

6.11.2.3 Solar and wind technology

Because of the energy consumption and high costs associated with running electrical aerators or pumps, several commercial companies have developed solar-powered or wind-powered aerators for use in reservoir aeration or oxygenation. The advent of these technologies has reduced the cost of treatments by eliminating the need for an electrical grid and for power—and without greenhouse gas emissions.

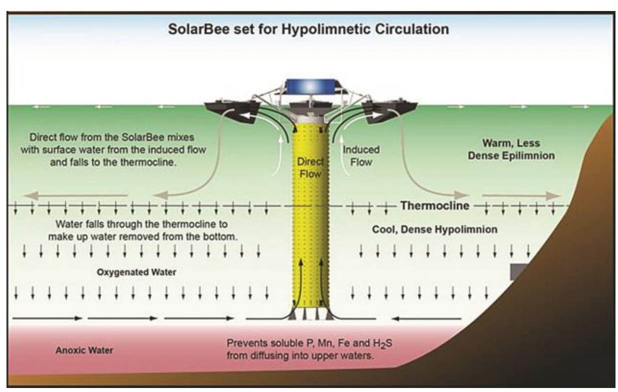


Figure 6.4. Sketches of solar-powered SolarBee® mixer for destratification. Image credit: Medora Corporation, Dickinson, North Dakota.

Solar units operate by capturing solar energy via an array of solar panels, converting the solar energy to electrical energy, and then delivering the electricity to a small motor to drive either an aerator or a pump (Figure 6.4). One of the companies currently supplying solar technology produces the SolarBee® mixer (Medora Corporation, Dickinson, North Dakota). The mixer is designed for simple handling and deployment with a small crew. Maintenance of the mixer is minimal and typically consists of removing weeds and debris from the impeller and cleaning the solar panels as needed.

SolarBee® mixers have been installed to enhance water quality in many reservoirs (Figure 6.5). For example, several units were installed in Jordan Lake, North Carolina, to assist with improving lake water quality in the vicinity of water supply intakes. The aeration systems were expected to reduce dissolved manganese and iron concentrations, the proliferation of cyanobacteria blooms in the vicinity of the intake and associated taste and odor issues, and the overall reservoir water quality in the area in which the intakes are located. Follow-up monitoring suggested the mixers were only partially successful.

Figure 6.5. SolarBee® lowered into Mt. Dell Reservoir, Utah. Photo credit: Salt Lake City Department of Public Utilities, Salt Lake City, Utah.

Windmills resemble smaller replicas of the stately old windmills that, at one time, were common across rural landscapes. The windmill blades harness winds to power a crankshaft that operates a diaphragm that pumps air or oxygen into the reservoir hypolimnion. Windmill aerators have been installed in various reservoirs operated by the U.S. Bureau of Land Management. They are more difficult to install than solar aerators, making them less mobile and requiring careful planning before installation. This technology has been extensively applied to pond management (Koenders Windmills Inc., Saskatchewan) but is just recently being applied to reservoirs and intensive evaluations are not available.

6.11.3 Hypolimnetic Aeration–Oxygenation

Hypolimnetic aeration–oxygenation is achievable through a variety of approaches, ranging from pumping the hypolimnetic water to the surface for aeration and returning it to the hypolimnion, to fine-pore pneumatic diffusers placed in the hypolimnion for the introduction of oxygen (Figure 6.6). These systems introduce oxygen into the hypolimnion as air or as pure oxygen. Pneumatic diffuser systems are designed such that the rising bubble plume does not mix hypolimnetic water into the epilimnion and thus prompt destratification (Singleton and Little 2006). Systems that transfer hypolimnetic water to the surface for reaeration are designed to minimize the speed at which the water is transferred so that mixing does not induce unwanted destratification (WOTS 2004).

There are a number of potential problems associated with aeration. The oxygen transfer efficiencies of most hypolimnetic aeration techniques are low, ranging from about 10% (Smith et al. 1975) to 50% (Bernhardt 1967). Thus, aeration units may need to operate at high recirculation rates, which could produce turbulence within the hypolimnion and thereby increase sediment oxygen demand (Smith et al. 1975; Singleton and Little 2006). Large reservoirs may require the installation of numerous aeration units, which potentially could produce enough turbulence to cause destratification (Heinzmann and Chorus 1994). Moreover, the introduction of compressed air that predominantly consists of atmospheric nitrogen may elevate levels of dissolved nitrogen gas in the hypolimnion and the formation of gas bubble disease in fish (Beutel and Horne 1999).

The primary advantage of hypolimnetic oxygenation over aeration is that the solubility of pure oxygen in water is roughly five times that achievable via aeration because air is about 20% oxygen. A second advantage of hypolimnetic oxygenation systems is their high transfer efficiencies (percentage uptake of delivered oxygen), which generally range 60%–80% (Speece 1994; Mobley and Brock 1995). As a result of higher oxygen solubility and higher system transfer efficiencies, size of the mechanical devices and recirculation rates needed to deliver an equivalent amount of oxygen using pure oxygen instead of air are greatly reduced. This scale reduction avoids a number of the disadvantages associated with traditional aeration systems (Singleton and Little 2006). Lower recirculation rates minimize turbulence introduced into the hypolimnion, thereby minimizing induced oxygen demand (Moore et al. 1996). High oxygen delivery rates and low induced oxygen demand allow for the maintenance of suitable levels of dissolved oxygen in oxygenated hypolimnia (Thomas et al. 1994; Prepas and Burke 1997). Smaller oxygenation systems also may be able to oxygenate large bodies of water with a reduced risk of accidental destratification. Additional advantages of hypolimnetic oxygenation include avoidance of hypolimnetic dissolved nitrogen supersaturation (Fast et al. 1975) and substantial decreases in energy use (Speece 1994).

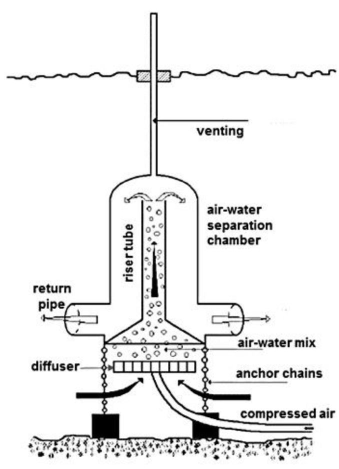


Figure 6.6. Sketch of a hypolimnetic aerator. After I. McAliley, HDR Inc., Charlotte, North Carolina.

Three general types of systems can be used: bubble plume oxygenation (Singleton et al. 2007), deepwater oxygen injection through linear or circular diffusers (Mobley and Brock 1995; Prepas and Burke 1997), and submerged down-flow bubble contact chambers (Speece 1994).

6.11.3.1 Bubble plume

Bubble plume oxygenation works by injecting pure oxygen through a dense group of porous diffusers at the bottom of the reservoir, creating a gas–water mixture that rises and gains momentum due to a positive buoyancy flux (Singleton et al. 2007). Oxygen bubbles dissolve into a surrounding plume of rising water. The oxygenated plume then detrains and spreads out horizontally below the thermocline. Bubble plumes are generally linear or circular and inject oxygen at a relatively low gas-flow rate (Schladow 1993). These systems are most suitable for deep reservoirs where the bulk of the bubbles dissolve in the hypolimnion and the momentum generated by the plume is low enough to prevent significant erosion of the thermocline. Bubble plume oxygenation may have trouble maintaining a well-oxygenated sediment–water interface because most of the oxygen rises to the upper levels of the hypolimnion.

6.11.3.2 Diffusers

Linear or circular diffuser oxygenation systems consist of an extensive network of linear diffusers that release fine oxygen bubbles that rapidly dissolve into the overlaying water column (Singleton and Little 2006). The diffuser system has a few advantages over other systems. In contrast to contact chambers (section 6.11.3.3) the system does not require the pumping of water. In addition, unlike the bubble plume oxygenation system (section 6.11.3.1), at low gas-flow rates the system does not induce a large-scale vertical current of water. Thus, dissolved oxygen tends to stay deeper in the reservoir. A system installed at Douglas Reservoir, Tennessee, successfully oxygenated the hypolimnion and improved water quality of turbine discharges (Mobley and Brock 1995).

6.11.3.3 Submerged contact chamber

The submerged contact chamber oxygenation systems consist of a submerged cone-shaped contact chamber installed on the bottom of the reservoir. A submersible pump draws water from the hypolimnion into the top of the cone. Oxygen supplied from an onshore facility is injected at the top of the cone. The oxygenated water is discharged through a horizontal diffuser pipe. Speece et al. (1971) observed oxygen transfer efficiency in the range of 80%–90% in an experimental chamber. With the proper horizontal dispersion of reoxygenated water, a submerged chamber system can overcome potential limitations of a bubble plume or a diffuser system. These limitations include accidental destratification caused by oxygen bubbles rising through the thermocline (Speece 1994) and localized anoxia as a result of limited oxygen dispersion within the hypolimnion (Fast and Lorenzen 1976). In addition, in contrast to bubble plumes and line diffusers, horizontal dispersion sends reoxygenated water out over the sediment, thereby keeping highly oxygenated water in direct contact with the sediment and promoting a well-oxygenated sediment-water interface.

Submerged contact chamber systems have been operated successfully in various water bodies (Figure 6.7). Camanche Reservoir is a large, multipurpose reservoir in the foothills of the Sierra Nevada Mountains in Northern California. A fish hatchery just downstream of the reservoir experienced large fish kills due to hypoxic withdrawals from the reservoir. After a contact chamber oxygenation system was installed, no fish kills occurred. Spatial monitoring of dissolved oxygen showed that a well-oxygenated plume of deep water migrated about 2 mi longitudinally up the reservoir 40 days after oxygenation (Speece 1994). In Newman Lake, Washington, low oxygen levels in bottom waters during the summer resulted in a severely degraded coldwater fishery. A contact chamber oxygenation system dramatically improved bottom water quality for fish during the summer by maintaining a well-oxygenated hypolimnion (Doke et al. 1995; Moore et al. 1996).

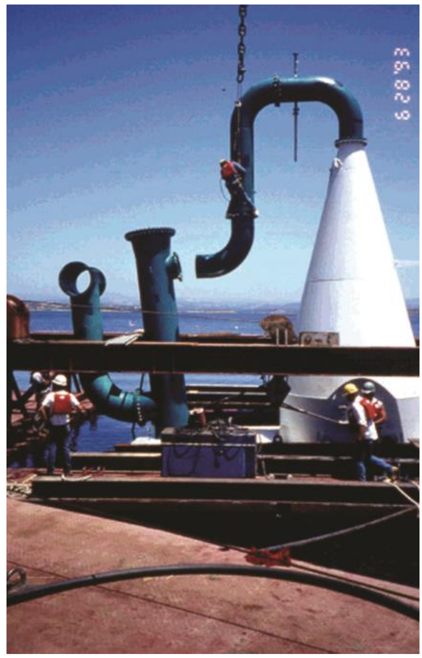


Figure 6.7. Speece cone installation at Camanche Reservoir, California. Photo credit: R. Jung, East Bay Municipal Utility District, Orinda, California.

6.11.4 Hypolimnetic Withdrawal

Hypolimnetic withdrawal (Figure 6.8) is a form of selective withdrawal through the dam but with the release of water from only the hypolimnion (Nürmberg 1987; Dunalska 2001; Hueftle and Stevens 2001). Epilimnetic water, which maintains adequate concentrations of dissolved oxygen, is retained in the reservoir. The major objective is the reduction of anoxic conditions in the hypolimnion which, in turn, will limit the release of phosphorus from the sediment and reduce the cycling of nutrients to the epilimnion. On an annual basis, the volume of water released remains unchanged, but the thermal stability may be reduced by withdrawing water from the hypolimnion. The effectiveness of this approach would depend upon the reservoir's morphology and inflow water-quality characteristics. Nürmberg (1987) evaluated hypolimnetic withdrawals in nearly 50 lakes and reservoirs and reported that withdrawals decreased summer average epilimnetic phosphorus and chlorophyll concentrations, increased Secchi disk transparency, and decreased hypolimnetic phosphorus concentration and anoxia.

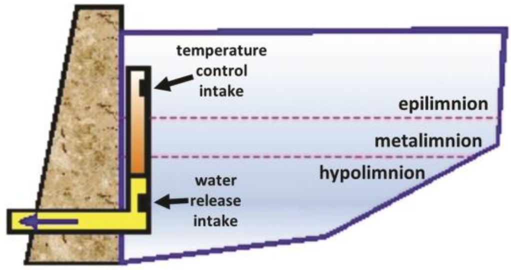


Figure 6.8. Sketch of a hypolimnetic withdrawal. Modified from D. Harpman, U.S. Bureau of Reclamation, Denver, Colorado.

Hypolimnetic withdrawal does not involve mixing the epilimnion with the hypolimnion, which is possible with aeration methods. Therefore, hypolimnetic withdrawal helps control eutrophication of the reservoir. Nevertheless, hypolimnetic withdrawals release nutrient-rich water of poor quality (i.e., low dissolved oxygen, low temperature, high dissolved solids) downstream and potentially increase eutrophication of downstream water bodies.

6.11.5 Guide Curve Management

Many reservoirs operate under some type of water-level management plan, or guide curve, that directs seasonal change in water storage. Modification of the reservoir's guide curve may enhance water quality. By modifying the annual distribution of retention time, undesirable water quality may be avoided or flushed out. Similarly, by modifying annual storage distribution, the intensity of stratification may be controlled. Guide curve modification for water-quality purposes is not common because such action usually imposes on other objectives of the reservoir. However, reservoir guide curves have been adjusted for other purposes, such as ensuring minimum downstream flows during low-flow periods, and adjustment for water quality is identified as an option by U.S. Army Corps of Engineers (USACE 1995). See section 7 for more on guide curve management.

6.11.5.1 Modification of guide curve to maximize flow

Modifying the guide curve may allow the flexibility to route inflows through the reservoir to release low-quality inputs or to flush low-water-quality conditions developed within the reservoir. Low-quality inputs may include highly turbid water during the wet season. This water can be moved through the reservoir more quickly by maintaining a low reservoir volume. Undesirable water-quality conditions may develop in late summer and fall when the reservoir stratifies. Increasing flow during this period may reduce the intensity of the stratification process, although extra flows during this period may be hard to get unless water is available from reservoirs upstream.

6.11.5.2 Modification of multi-reservoir operation

It is also possible to operate multiple reservoirs within a river basin to meet water-quality targets at downstream points within the basin. For example, releases of good quality from one reservoir can be blended with poor-quality releases from another reservoir in an adjacent feeder stream to neutralize the negative water-quality attributes. The USACE has used multi-reservoir releases to neutralize releases with low pH and to dilute highly turbid releases. The HEC-5Q model (USACE 1986) contains algorithms to calculate release requirements from multiple reservoirs to satisfy a downstream water-quality target, but other models are available (reviewed by Labadie 2004).

6.11.6 Contaminants Management

Management practices applicable to improving water quality may sometimes also be applicable to management of contaminants (WOTS 2004). Heavy metals that are mobilized when dissolved oxygen concentrations are low may be managed with aeration practices. Mobility may also be controlled by adding chemical materials (amendments) to the water or sediment. A "do-nothing" alternative that allows contaminated sediment to be buried with time may be feasible if contaminants remain bound to the sediment. Other alternatives may involve water management practices such as employing suitable water residence times to allow dilution or water drawdowns to allow aeration and drying. For contaminants in sediment that are primarily cycled by biotic organisms, remediation may be possible through separation of the biota from sediment by means of capping, dredging, or isolation (WOTS 2004; Jaglal 2009).

6.11.6.1 Amendments

Amendments are add-ons, usually possessing high cation exchange capacity, which can lower mobility and bioavailability of contaminants in sediment, thereby decreasing their solubility. In situ immobilization using inexpensive amendments such as minerals (e.g., apatite, lime, zeolites, beringite) is considered promising (Peng et al.

2009). Compared with the amendments used in terrestrial soils, those used in submerged sediment usually have higher sorption capacity, lower water solubility, higher stability under reducing and oxidizing conditions, and lower cost (Raicevic et al. 2006).

6.11.6.2 Capping approach

Decreasing the direct contact area between water and the contaminated sediment can lower the release of contaminants. Therefore, capping the contaminated sediment with sandy materials, such as clean sediment, sand, or gravel, could be an effective remediation technique (Peng et al. 2009). If properly designed, the placement of a relatively coarse-grained cap does not disturb or mix with an underlying soft, fine-grained sediment. Theofanis et al. (2001) indicated that a good cap may be about 20-in thick and that capping the sediment with sandy materials can reduce heavy metal concentration in water by about 80%. Compared with other in situ remediation methods, the capping approach is low cost. To further immobilize contaminants and enhance the cap quality, amendments (section 6.11.6.1) may be mixed into the sand cap. Jacobs and Förstner (1999) reported that fixation capacity for heavy metals and organic contaminants increased sharply after adding zeolite into a sand cap.

6.11.6.3 Phytoremediation

Phytoremediation is the use of plants to extract, sequester, or detoxify pollutants. This technology is widely viewed as an ecologically responsible alternative to environmentally destructive chemical remediation methods (Meagher 2000). Phytoremediation comprises two tiers, one by plants themselves and the other by root-colonizing microbes that degrade the toxic compounds further to nontoxic metabolites. This technology popularly is applied in terrestrial soil remediation and also shows some potential value for remediation in shallow rivers, lakes, and wetlands. At present, this technology reportedly presents good immobilization effects for zinc, iron, manganese, and cadmium in sediment (Peng et al. 2009).

Section 7

Water Regime

7.1 Introduction

Water regime, which includes water-level fluctuations and water retention, is an important characteristic of reservoir environments. Water-level fluctuations can be large (Figure 7.1) and fluctuation patterns influence physical, chemical, and biological features of reservoirs, especially in shallow reservoirs and nearshore zones of all reservoirs because these are exceptionally sensitive to changes in water level. Aquatic habitats and feeding and spawning grounds are gained and lost as water levels fluctuate. The littoral zone is the most likely habitat to be affected by reservoir water-level fluctuations, which affect some fundamental processes, such as decomposition, production, and trophic interactions among organisms. Similarly, a diversity of water retention rates makes reservoirs range in characteristics from river-like to lake-like. Therefore, water regime may have an overriding effect on the ecology, functioning, and management of reservoirs.

Reservoirs are engineered to meet well-defined operational goals. Several important physical attributes, including depth, area, and flushing rate, are dictated by reservoir location, topography, and hydrology. These attributes are controlled further

Figure 7.1. The high water mark in Lake Mead reservoir, Nevada. Photo credit: U.S. Bureau of Reclamation.

by structural (e.g., dam height, outlet depth) and operational (e.g., water-level targets, release rates) characteristics. Operation of reservoirs generally is constrained by existing legalities in project authorizations, water control goals, and local physiographic and climatological features.

Operational goals of a reservoir dictate water regime (Kennedy 1999). Navigation uses, which usually occur in shallow channels, require that reservoirs be maintained near 100% of their storage capacity, thus water-level fluctuations are generally small. Hydropower reservoirs are deeper so that they can attain efficient hydraulic head. They usually exhibit diel fluctuation patterns associated with intermittent water releases to generate electricity, annual fluctuation patterns associated with precipitation cycles, and, in some areas, multiyear cycles associated with long-term drought. Although there is great variability, water level in reservoirs whose main purpose is water supply for agriculture often varies on a scale of several years, as the reservoir may be used to store water during wet years to compensate for dry years. Conversely, flood control reservoirs can experience seasonal drawdowns to <30% of capacity because of their requirement to store seasonally excessive runoff. Water levels in flood control reservoirs generally follow marked annual patterns.

As much as possible, notwithstanding the vagaries of rainfall, reservoir operators follow pre-established guide curves, which dictate what the water level should be on each day of the year (Figure 7.2). Guide curves are formal (often legal) descriptions of where the water level should be on any given day of the year to meet the reservoir's primary purposes (e.g., flood control, power generation, navigation) within the constraints posed by droughts, floods, and other unforeseen circumstances (see sections 7.4.8-7.4.10). Regional authorities or regulatory agencies develop guide curves to optimize water storage benefits among reservoirs in a river basin. For instance, the Tennessee Valley Authority manages 41 reservoirs on the main-stem Tennessee River and its numerous tributaries, and the operation of each dam affects the operation of other dams in that system.

Hydraulic retention time influences various abiotic and biotic characteristics of reservoirs—in particular, primary production. Retention time can range from a few days to a few years (Søballe and Kimmel 1987) and tends to be lower in reservoirs low in a river basin than those high in the basin (Miranda et al. 2008). Phytoplankton communities reach their full production potential in reservoirs with high retention times, which behave more like lakes and less like rivers. In a study of reservoirs across the USA, Søballe and Kimmel (1987) noted that algal communities needed about 60–100 d of retention time to realize their full potential at any given level of nutrients. In Alabama reservoirs, the relationship between retention time and algal production was confirmed, but the threshold retention time was thought to be closer to 30 d (Maceina et al. 1996). The retention time in any reservoir is driven by storage capacity and the amount of precipitation in the watershed, which varies seasonally and annually. Thus,

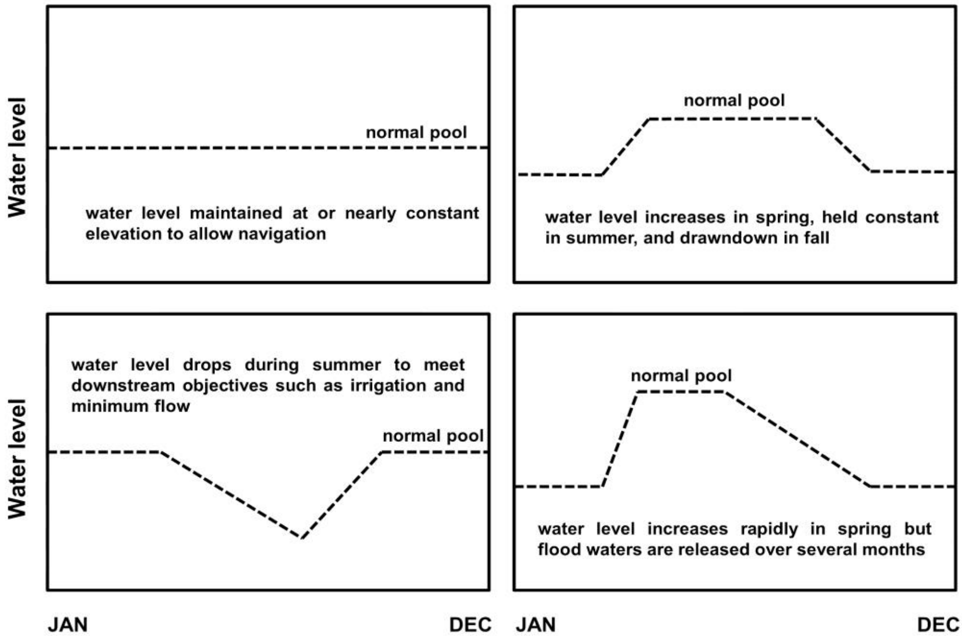


Figure 7.2. A generalized selection of guide curves applied to reservoirs. The dashed lines represent the water levels applied relative to time of the year.

retention time and primary production varies within and between years. In hyper-eutrophic reservoirs, long retention times can be problematic as they may foster cyanobacteria blooms that can cause human and animal health concerns and limit fish production (section 4).

7.2 Reservoir Water Levels Relative to Flood Pulse Concept

Inundated floodplains in river systems contribute habitats that provide fish space for reproduction, feeding, and refuge (Junk et al. 1989; Bayley 1995). When waters recede, wetland and upland vegetation thrive (Casanova and Brock 2000; Leyer 2005), rejuvenating the floodplain until the next flood pulse. This effect is particularly evident in large lowland rivers where floodplains are sizeable and make up a large part of the river ecosystem (Tabacchi et al. 1998; Kingsford 2000). As a consequence, fish communities in large rivers frequently exhibit high biodiversity, attributed to the structural diversity and habitat richness of floodplain environments (Schiemer 2000).

Similarly, reservoirs that experience water-level fluctuations as part of their operational objective periodically inundate and dewater nearshore areas. However, nearshore areas in reservoirs are different from river floodplains. Within the main stem and throughout most embayments, water levels fluctuate over elevation contours that were once uplands. These uplands have slopes, soils, and seed banks that are different from those of floodplains. A true floodplain does not occur in reservoirs except in the upper reaches where tributaries enter the reservoir. Size of floodplains in reservoirs is usually small in small tributaries and in reservoirs high in a river basin; conversely, floodplains are usually large in large tributaries and in reservoirs impounded over lowland rivers.

7.3 Effects of Water Regime

The effects of water-level fluctuations on nearshore reservoir environment are influenced by the amplitude, frequency, periodicity, and timing of the fluctuations (Ploskey 1986). The effect of water-level fluctuations may be expected to be directly related to the amplitude of the fluctuations, as large fluctuations are likely to affect a larger portion of the nearshore zone. The frequency of fluctuation determines the duration of the effect (exposure or inundation) and the time available for the responses of the biota. The effect of reservoir drawdown may be either beneficial or detrimental to the littoral ecosystem depending on the duration of the drawdown (Godshalk and Barko 1988). The frequency of fluctuations may vary from diel (e.g., pump storage, hydropower reservoirs), to seasonal (flood control), to long-term (irrigation) fluctuations. Timing of water-level fluctuations can promote habitat integrity, encourage or suppress plant growth, and stimulate fish access to key habitats during critical periods of their life cycle (Miranda et al. 2014).

Water retention time influences several water-quality constituents directly and many more indirectly (Søballe and Kimmel 1987; Straškraba 1999). Temperature, dissolved oxygen, and the production of algae are affected directly by water retention time (sections 4 and 6). The timing and degree of thermal stratification (the separation or layering of colder and warmer waters within the reservoirs) also is related directly to water retention time. Dissolved oxygen concentrations in reservoirs are related to thermal stratification, oxygen demand (biological, chemical, and sediment), and the timing and depth of water releases. Water retention time and the availability of nutrients and light affect the dynamics of phytoplankton growth. In turn, phytoplankton play a critical role in the dissolved oxygen balance of the system. Water retention time changes seasonally and is usually higher in summer and early fall, although this varies geographically.

Figure 7.3. Drawdown in deep Causey Reservoir, Utah, showing a bare band in the regulated zone. Photo credit: Utah Division of Wildlife Resources, Salt Lake City.

A survey of reservoir managers identified that approximately 12% of reservoirs ≥250 ac in the USA had moderate-to-high or high complications associated with water retention time (Krogman and Miranda 2016). Quick flushing of the reservoir maintained high turbidity and precluded development of plankton communities. This percentage was as high as 22% in reservoirs managed for hydropower. In terms of water levels, seasonally mistimed water-level fluctuations were problematic in 16% of reservoirs. Timing of annual filling and emptying was inconsistent with the life-history requirements and habitat needs of fish. This percentage was as high as 27% in reservoirs managed for flood control and for irrigation.

7.3.1 Aesthetics

The exposure of nearshore substrates and stumps in reservoirs is the most visually obvious effect of water-level fluctuations, although the extent of the effect varies with slope of the reservoir basin (i.e., steepness of nearshore zone; Stamou et al. 2007). When the reservoir is near normal pool (Figure 7.2), the trees on the reservoir banks are seen as though emerging from the water, giving a pleasant aesthetic view. Progressive lowering of the water level results in the appearance of an area of barren, uncovered land (i.e., the regulated zone). The regulated zone interrupts the continuity of the landscape and gives the impression of an empty reservoir. In reservoirs where shore slopes are steep, regulated zones appear as bare bands (or rings; Figure 7.1 and 7.3). In reservoirs with mild shore slopes, regulated zones give the impression of desert areas, often called mudflats (Figure 7.4). The regulated zone tends to produce muddy water, large areas of exposed lake bottom, and eroding shoreline, which are primary detractors from scenic quality. Reservoir rings, mudflats, or exposed objects can detract from site attractiveness at low water levels.

7.3.2 Physiography

The extent to which flood pulses affect nearshore habitats depends on physiography. For example, in steep terrain such as mountain reservoirs, the nearshore area is only scarcely developed, and the surface affected by water-level fluctuations is small even during large fluctuations. Conversely, shallow reservoirs with low-gradient zones provide more space for flood expansion. Large, shallow reservoirs with a strong riverine

Figure 7.4. Drawdown in shallow Grenada Lake, Mississippi, showing desert-like mudflat areas in the regulated zone. U.S. Army Corps of Engineers, Vicksburg District.

influence will have sharper hydrographs than those fed mainly by surrounding runoff, minor tributaries, or diffuse inflow from groundwater. Thus, shallow systems with gradually sloping shorelines are more affected by water-level fluctuations than lakes with steep nearshore slopes because larger areas are flooded and exposed. In general, fluctuating water levels tend to produce straighter shorelines, gentler slopes, and softer homogeneous substrate.

7.3.3 Sediment

Sediment distribution and composition along the nearshore zone depends on water regime (Furey et al. 2004). The shore zone that is exposed to waves for the longest time of the year has the coarsest sediment, while the average grain size decreases with decreased wave exposure (i.e., downward on the depth gradient). During drawdown or rising water levels, the moving surf zone touches different types of sediment. Sediment at different depth zones may be chemically different, so that the surf zone mobilizes variable quantities and qualities of dissolved substances as water levels change (Figure 7.5).

During the rising water-level phase, several processes facilitate the deposition and temporal storage of organic matter in the nearshore zone of reservoirs. Apart from drowned terrestrial biomass, rising water collects the organic matter that has become deposited during a drought and carries it to the surf zone. This terrestrial leaf litter becomes degraded by waves and is then contributed to the nearshore zone as organic sediment.

7.3.4 Bottom Exposure

Periodic long-term exposure of sediment by fluctuating water levels may consolidate flocculent sediment and potentially increase reservoir capacity. In experiments with sediment dredged from Lake Apopka, Florida, Fox et al. (1977) noted that dewatering and drying for various periods of time shrank bottom sediment. Also, water had the same or lower nutrient concentrations, reduced turbidity, and higher dissolved oxygen tensions after bottom

Figure 7.5. Zonation of sediment types caused by fluctuating water levels at Stevens Creek Reservoir, California. Photo credit: J. Sullivan, Getty Images.

exposure. Similarly, reduced water levels and concomitant compaction and aerobic decay of organic matter in Lake Tohopekaliga, Florida, reduced the depth of organic sediment by 50%–80% (Wegener and Williams 1974).

The positive benefits that a drawdown can have on sediment desiccation may result in some undesirable effects such as enhanced rates of nutrient release after refill (Fabre 1988). The outcome of enhanced rates of internal nutrient loading from consolidated sediment could be the development of a large pulse of nitrogen and phosphorus to the water column and stimulation of excessive algal growth. Positive and negative effects of sediment desiccation on macrophyte and phytoplankton growth need to be considered before implementing lake drawdown and refill as a management technique.

7.3.5 Erosion

A less obvious effect of water-level changes is shoreline alterations due to erosion and redeposition of sediment from nearshore substrates and shore banks (Figure 5.6). Waves driven principally by wind but also by navigation distribute sediment vertically along the nearshore zone according to particle sizes (Furey et al. 2004). Large rocks and gravel remain nearshore, whereas progressively smaller particles are displaced gradually toward deeper contours. Wind and rain also erode substrates exposed by drawdown.

The area modified by erosion is largely determined by the magnitude of water-level fluctuations and the morphometry of the reservoir. Rates of shoreline changes depend on characteristics of the fluctuation zone—its slope, degree of exposure, and composition (Gasith and Gafny 1990; Hofmann et al. 2008). When steep shores of impoundments are exposed to large fluctuations in water level, they generally erode rapidly and, depending on local geology, leave a zone of barren rock interspersed with gravel.

In contrast to the relatively permanent rapid changes in steep-sided impoundments, the shorelines of shallow reservoirs undergo slower long-term erosion (Reid 1993). Alluvial soils in these exposed, wind-swept, low-slope littorals are easily eroded. However, because of the gradual slope, erosion is slow and involves large areas even when vertical changes in water levels are small but especially when they are large. Rising turbid waters redeposit sediment at higher elevations to form terraces at various elevations that are dictated by the water levels (Figure 7.6).

Terrestrial vegetation that develops in dewatered areas helps attenuate the effects of erosion. Cleared areas of Beaver Reservoir, Arkansas, were subjected to greater and more rapid erosion than areas with vegetation (Aggus 1971). Breakup and decomposition of flooded herbaceous vegetation in Beaver Reservoir resulted in a conspicuous increase in erosion and redeposition. Erosion also was noticeably slowed in several Kansas reservoirs by flooded herbaceous vegetation established during drawdowns in the previous growing season (Groen and Schroeder 1978).

Figure 7.6. Terracing created by sediment depositions at various elevations in Boca Reservoir, California. Photo credit: L. Driscoll, Getty Images.

7.3.6 Ice Formation

Ice formation affects nearshore habitats, and the effect depends on water level (Reid 1993). When reservoirs freeze and undergo cold contraction, cracks are formed into which water flows and freezes. Subsequent warming will expand the ice, forcing the ice to buckle, and to be shoved onto the shores. Sediment in the nearshore can be moved shoreward by this force. By the time the reservoir freezes, the pool level often has dropped below bank levels, reducing direct impacts on the shoreline. However, many reservoirs freeze when the water level is high; thus weakening of the banks may occur, resulting in massive bank failures in the following spring and summer.

7.3.7 Water Clarity

Sources of colloidal turbidity in reservoirs include inflowing tributaries, erosion of banks and mudflats (via waves, wind, and rain), and suspension of bottom sediment by waves or currents (section 5). Water-level fluctuations affect the gradient and water retention time in the reservoir and thereby determine rates of sedimentation and contours of where sedimentation occurs (Lara 1973). At low water levels, sediment previously deposited near inflow areas may be sluiced farther down the reservoir. Turbidity may increase or decrease as water levels change because fluctuating water levels expose substrates of different composition or cover to erosion. Reservoir drawdown may increase turbidity by resuspending previously eroded sediment. Increased water levels often reduce turbidity, especially if inundated areas are covered with terrestrial vegetation. Nevertheless, fluctuating water levels limit the growth of macrophytes, which bind soils and dampen waves in the nearshore zone, thereby resulting in increased turbidity (Judd and Taub 1973).

In flood control reservoirs that experience substantial annual water-level fluctuations, resuspension of fine sediment deposited in deeper contours can greatly reduce water clarity (Dirnberger and Weinberger 2005). Wave action induced by wind and boat traffic tends to have greater effects on water clarity after drawdown begins than before. Fine particles that have been transported gradually and have settled in deep water, where they normally are not affected by turbulence when pool levels are high, become accessible for resuspension at low water levels. Resuspension likely begins as soon as drawdown begins but should be most noticeable when water level drops to conservation pool. Also, resuspension is greater in uplake segments because they have proportionally more shallow area, more nearshore area relative to total area, and greater tributary sediment deposition. Thus, the effects of drawdown on turbidity is observed sooner uplake and in the upper ends of major embayments.

7.3.8 Mixing

Water-level changes that significantly alter depth, area, or fetch length may change depth of mixing or patterns of stratification. A shift from stable to fluctuating water levels could reduce the tendency for much of a reservoir to stratify (Turner 1981). This possibility is even more likely if changes in water levels result from selective discharge from the hypolimnion or from rapid rates of discharge (sections 4.4.3.5 and 6.11.4). Temperature of inflowing water tends to dominate the thermal regime of reservoirs as the retention time of water decreases (Carmack 1979). Cooper (1980) reported that high water levels and insignificant drawdowns in late summer effectively prolonged thermal stratification in Grenada Reservoir, Mississippi. In Lake Hume reservoir, Australia, because the depth of the reservoir was relatively shallow because of an extreme drawdown, wind-driven events led to a substantial deepening of the thermocline, which allowed periodic pulses of nutrients into the warm surface layer (Baldwin et al. 2008).

7.3.9 Biotic Characteristics

Water regime affects species composition and abundance as well as biotic interactions (Ploskey 1986). Phytoplankton community composition and density can be affected indirectly by shifts in nutrients availability and light that result from water-level changes. Direct effects include the physical removal of phytoplankton by water releases (Perry et al. 1990). Periphyton can be affected directly by changes in water levels, as it desiccates when exposed and may not get sufficient light when flooded deep (Gasith and Gafny 1990; Brauns et al. 2008). Aquatic macrophytes generally are discouraged by drawdowns of lake level because of the reduction in habitat availability, increased erosion of substrates, deterioration of light conditions, and desiccation (Gafny and Gasith 2000). However, moist-soil and terrestrial macrophytes, mainly her-

Figure 7.7. Mudflats in Barren River Lake reservoir, Kentucky. Photo credit: E. Cummins, Kentucky Department of Fish and Wildlife Resources.

baceous plants, can proliferate in exposed areas given the right conditions. Fluctuations in water level produce conspicuous zonation of moist-soil and terrestrial macrophytes on reservoir shorelines. As different plant species tolerate different degrees of flooding, and are able to colonize and develop in drawdown zones at different times of the year, the range of flooding regime creates distinct vegetation zones. Where water levels are stable, there is generally a two-zoned system with emergent aquatic macrophytes and possibly submerged macrophytes nearshore and woody plants inshore. If water levels fluctuate widely, plants are limited by their tolerance limits in terms of wetness and dryness, which results in the loss of some or all species and produces mudflats that are devoid of plants or dominated by only a few opportunistic species (Figure 7.7). Uniformity of water regime driven by unnatural retention and release schedules can promote low macrophyte diversity or even monocultures adapted for the unique water regime for the reservoir.

Some invertebrate groups may be affected by water regime. Zooplankton community composition and density are unlikely to be affected directly by changes in water levels. Direct effects are limited to flushing resulting from shortened water retention time. Indirectly, increases in water level may be associated with higher turbidity, which may shift the zooplankton community structure toward rotifers (section 5.7.3). Benthic invertebrates also are discouraged by water-level fluctuations, which cause a reduction in species diversity and abundance in nearshore invertebrate communities (Brauns et al. 2008). Most species have little ability to retreat with lowering water levels and, therefore, become stranded in substrates and desiccate. Species able to migrate with the changing water level (e.g., chironomids) or independent of the substrate (e.g., mobile benthos) may increase in abundance and representation. Water-level changes in shallow, gently sloping reservoirs likely affect more benthic organisms per unit of depth change than such changes do in steep-sided reservoirs.

Water-level fluctuations affect various aspects of fish ecology (Zohary and Strovsky 2011). Effects on fish spawning include limiting or expanding availability of suitable spawning habitat, increasing egg mortality by stranding eggs, forcing adult fish to abandon nests, constraining fish spawning to less favorable areas, and exposing eggs to increased predation. Water-level fluctuations often change food availability as they may force changes in feeding habits of fish and change predator–prey relationships, frequently by increasing piscivory during drawdowns. As is the case for invertebrates, drawdowns may force small or juvenile fish to move out of the nearshore region, exposing them to pelagic predation. Changes in food availability and trophic interactions can lead to changes in community composition. Fluctuations in population densities in many reservoir fish species have been linked to water-level fluctuations (more below).

7.4 Water Regime Management

Water regime management in reservoirs traditionally has focused on managing storage, which indirectly controls retention time. In fact, the goal of many reservoirs is to alter retention time to rearrange and smooth the annual discharge cycle by holding water during periods of high flow and releasing it during periods of low flow. As a result, there is a reduced seasonality in the river below the reservoir (Pegg et al. 2003). The dynamic nature of water regime in reservoirs and diversity of reservoir purposes make it difficult to develop generalized water regime management practices. However, there are general principles common to water regime management.

7.4.1 Monitoring Program

Long-term daily water-level records are often available online from the reservoir controlling authority or can be requested. For example, daily records for many reservoirs managed by the U.S. Army Corps of Engineers (USACE) are available online (USACE 2016). If records are not already available, they can be monitored by placing on-site a long-term water-level logger. Loggers often record other variables along with water level (e.g., water temperature). These loggers are available from various companies (e.g., HOBO, Bourne, Massachusetts; YSI, Yellow Springs, Ohio). Also available are off-the-shelf telecommunications options that include radio, telephone, and cell phone (e.g., Campbell Scientific, Logan, Utah; Bentek Systems, Calgary, Alberta). Systems can be programmed to send alarms or report site conditions by calling out to computers and phones. Real-time data can also be exported for further processing by spreadsheets or data analysis programs.

7.4.2 Timing of Flood

7.4.2.1 High water in spawning–growing season of fish

Water-level manipulation has been recognized as a useful tool for managing fish populations. Raising the water level above summer pool creates cover and a favorable environment for shoreline-dwelling fish (Keith 1975; Martin et al. 1981). A high water level during the spawning and growing seasons generally increases the abundance of age-0 fish of various species (Aggus and Elliott 1975; Shirley and Andrews 1977; Miranda et al. 1984; Mitzner 1991; Kahl et al. 2008). If the dam has the infrastructure needed to control water discharges, the management of water levels provides one of the most cost-effective means of managing a fish community. Depending on the species, strong year classes may not be needed every year to maintain a healthy fish population. The effects, however, are not always sufficiently predictable with a comfortable degree of confidence.

Extent of spring water-level fluctuations did not significantly affect the density of juveniles in main-stem reservoirs of the Tennessee River (Miranda and Lowery 2007). These aging reservoirs are characterized by barren regulated zones where repeated annual flooding prevents the establishment of most perennial and annual plants, which usually require 2–3 years to re-establish. Unlike the wetlands in a floodplain, the regulated zone does not normally support plants that thrive or can readily re-establish after flooding. Thus, the benefits of flooding may manifest only after a drought that has allowed vegetation enough time to expand in the mudflats or during a flood that inundates vegetation at elevations above the regulated zone.

7.4.2.2 Low water in growing season

Drawdowns during the growing season (i.e., frost-free season) are often an integral component of the operation of reservoirs, especially those operated for flood control and irrigation. Water is released slowly prior to rainy periods to open up storage capacity to hold potential floods and to provide water for the irrigation season. Guide curves generally have some flexibility for accommodating slightly different water regimes, flexibility that is needed given the randomness in annual rainfall patterns. Less flexibility exists in reservoirs operated for purposes other than flood control or irrigation.

Herbaceous terrestrial plants that become established on suitable substrates after a drawdown during the growing season are beneficial. These plants provide spawning and nursery sites for many species of fish when inundated at the appropriate times (Figure 7.8). They also provide food and refuge for bacteria, zooplankton, benthos, fish, and birds; substrates for attached algae; and nutrients for aquatic primary production. Establishment of herbaceous terrestrial vegetation after drawdown is also important for erosion control, aesthetic purposes, and nutrient retention. When reflooded, areas with a vegetation cover are less apt to contribute to turbidity.

An earlier drawdown time in fall will result in a greater colonization and abundance of emergent species (Kadlec 1962; Casanova and Brock 2000). The longer the drawdown persists within the extent of the growing season, the greater the vegetation growth. Slow plant colonization and growth may occur in years of drought, when not enough rain wets the exposed regulated zone. Once the growing season ends, annual plant species may decay, although slowly because of low temperatures. If flooded, disintegration is accelerated. Thus, in many cases plant growth from the previous season may be best used in the current year if not flooded until just before fish spawning begins (Dagel and Miranda 2012). Depending on soils and geography, longer drawdowns caused by long-term droughts can encourage growth of woody vegetation that has longer durability under prolonged flooding.

According to Jenkins (1970) a drawdown during the growing season is potentially the most effective management tool for the reservoir biologist. Large drawdowns during the growing season can be used to restructure fish assemblages, although conflicts with other reservoir uses may occur (Keith 1975). Such conflict is minimal with flood control functions, so a drawdown is most applicable in flood control reservoirs. Based primarily on experience in the southeastern USA, Keith (1975) summarized the effects of a fall drawdown on forage populations. Predation reduces populations of overabundant forage fish when they are forced from cover in shallow water and concentrated in open water. Enhanced predator–prey interactions may increase predator growth rates. However, early summer drawdown can lead to year-class failures of some fish species, although responses to refilling the following year may more than compensate for current year inadequacies. Effects of drawdown on fish populations are frequently temporary, so the drawdown may need to be repeated every few years.

Figure 7.8. Herbaceous plants growing in the back of an embayment during low water in Dodd Reservoir, Colorado. Photo credit: Colorado Parks and Wildlife, Grand Junction.

7.4.2.3 Winter drawdown

Lowering the water level in the winter exposes the sediment to both freezing and loss of water. Effects of winter drawdowns probably are most severe on aquatic macrophytes because prolonged exposure to freezing temperatures can be fatal (Cooke 1980). Thus, drawdowns can be used as a tool to control undesirable plants (section

11.5.2), although some nuisance plants are resistant to or are stimulated by the drawdown (Cooke 1980). Winter drawdowns also have been identified as a tool for reducing densities of zebra mussel (*Dreissena polymorpha*), although not total elimination (Grazio and Montz 2002). A negative effect of winter drawdowns is overall benthos reduction through exposure and freezing, which reduces the potential for high production in the following spring and early summer because of the time it takes to recolonize newly flooded substrates.

Warmwater fishes are more susceptible to entrainment and discharge from impoundments in winter than in other seasons because they are less active (Layzer et al. 1989). In reservoirs with extensive and rapid drawdowns in winter, mortality of fish can be reduced by limiting the rate and extent of drawdown. Survival of eggs or juveniles hatched in fall may be significantly reduced by fall and winter drawdown, as a result of stranding or loss of habitat. Development of eggs and fry is slow in cool water; consequently, their vulnerability to potentially adverse conditions is prolonged. The maintenance of stable water levels until juvenile fish are able to escape receding water levels is essential. If water levels must be lowered in late winter, it is best to lower them slowly.

Additional benefits of winter drawdowns are that they may provide an opportunity to repair and improve fish habitat, docks, and other structures. Moreover, loose, flocculent organic sediment can become consolidated after drawdowns, potentially solidifying sediment and reducing turbidity.

7.4.3 Periodicity

7.4.3.1 Variability in timing

Varying water regimes have dissimilar effects on different fish species (Miranda and Lowery 2007). Therefore, a diversity of water regimes, including years with extreme low and years with extreme high water levels, rather than a rigid guide curve repeated annually, is likely most beneficial to the long-term permanence of diverse fish assemblages. Flooding within reservoirs is often highly engineered, with inflexible guide curves that allow only limited annual variation. Steady annual fluctuation patterns produce regulated zones devoid of vegetation and consisting primarily of mudflats of limited ecological value to most fish. Maintenance of water levels within the margins stipulated by the narrow range of a guide curve forces operational drawdown and flooding that adversely affect nearshore spawners. Operational flexibility to produce diverse water-regime patterns may be found in reservoir cascades such as those in the Columbia, Tennessee, and Missouri rivers (Hesse and Mestl 1993; Zagona et al. 2001). In developing water management plans, regulatory agencies may consider incorporating managed randomness into guide curves.

7.4.3.2 Extended flood

For some species year-class strength can be influenced more by survival after spawning than by spawning success (Martin et al. 1981; Miranda et al. 1984). Survival may be improved by maintaining high water levels for as long as possible after spawning or at least until juveniles have reached some milestone (e.g., a certain length, a shift in diet, a change in habitat) that would increase the chances of recruitment into the next life-history phase. This is particularly true if lush vegetation is present in the regulated zone. The provision of high water during summer is often restricted by peak demands for water usage and the need to begin reducing volume before the upcoming wet season. Thus, opportunities for enhancing recruitment of juveniles by maintaining high water in summer and fall until low water temperature reduces fish activity are generally limited. However, water management agencies are more likely to entertain such requests if they come spaced out every few years.

7.4.3.3 Long drawdowns and floods

Long drawdowns are more effective than short ones. To have a meaningful effect, according to Ploskey (1986) drawdowns should last several months during the growing season and substantially reduce the surface area to expose all or a large fraction of the regulated zone. Such drawdowns would allow for rejuvenation of moist-soil and terrestrial vegetation along nearshore areas and concentrate fish to facilitate predation and energy transfer, potentially enhancing growth and condition of all fish species. Similarly, Ploskey (1986) suggested that floods should be large enough to inundate terrestrial vegetation for several weeks during key reproduction and development periods. Generally, sporadic large and long drawdowns or floods are more effective than frequent ones in altering and biotic assemblages.

In reservoirs with extensive drawdowns, some embayments and inlets can be prevented from complete dewatering by the construction of submerged barriers across their mouths, which will help retain a shallow depth of water in them. Nevertheless, such barriers may become navigation hazards at some depth ranges. Alternatively, excavated depressions (section 9.3.3) also can hold back water in pools during drawdown.

Excessively long drawdowns can be detrimental to fisheries. Multiple sequential years of low water can limit use of boats, fishing effort, catch rates, and fishery harvest (Chizinski et al. 2014). Increased feeding activity and growth by piscivores has been reported during and immediately after reservoir drawdowns (reviewed by Ploskey 1983). However, following extended drawdown, and after an initial growth burst, growth of piscivores may diminish as prey densities are depleted.

7.4.4 Rate

7.4.4.1 Slow drawdowns

In general, slower drawdowns produce better conditions for a wide variety of plants, benthic invertebrates, and juvenile fish species. For plants, slow drawdowns allow the development of a shoreline zonation with different stages of plant growth; rapid drawdown tends to provide a wide homogenous area with lower diversity. For benthic invertebrates, slow drawdowns allow downward migration (Aroviita and Hämäläinen 2008); for benthic invertebrates and for fish rapid drawdown tends to increase stranding in substrate or isolated pools. In an Illinois reservoir, hatching success of largemouth bass was disrupted during rapid water-level rises and drops (Kohler et al. 1993); peak hatching success occurred when water levels were relatively stable. Gently rising water levels may be beneficial to many reservoir fish species if they evolved as floodplain species benefitting from rising water levels during the spawning period. Slow rises may produce a moving littoral within the reservoir as described for floodplains by the flood-pulse concept of Junk et al. (1989).

Rapid water-level drawdowns also can reduce the quality of shoreline habitats. Bank failures due to sudden drawdown often occur in slopes composed of clay materials, in which the excess pore water pressures do not have enough time to dissipate, thereby reducing the overall shear strength of the clay materials (Abramson et al. 2002). If the water level against the slope face is suddenly drawn down, the stabilizing pressure is removed quickly, creating an unbalanced condition. Thus, the banks may lose strength and stability, and segments may slide down into the reservoir reducing, depth and slope and homogenizing the bank.

Quick fluctuations of water levels are common in some reservoirs (Kennedy et al. 2002). Some of these fluctuations are innate to the reservoir's purpose, such as hydropower reservoirs that experience small diel water-level fluctuations because of their power generation objective. Other reservoirs, such as those designed for flood control, experience fluctuations that may last a few months prompted by the need to increase discharge in anticipation of intense precipitation events, followed by quick rises as rainfall is stored. These quick fluctuations are less common in storage reservoirs, which are driven by the necessity to manage water level within a narrow-bounded guide curve. In reservoirs of the Tennessee River, quick spring fluctuations were more relevant to fish than spring water level (Miranda and Lowery 2007).

7.4.4.2 Long-term view

Long term, water-level management may produce more benefits if focused on producing a year with exceptional water-level conditions every few years than trying to provide acceptable water levels every year. A high water level only every 4–5 years would allow enough time for development of a lush vegetation community along shores and floodplains in the intervening years, whereas flooding yearly may not allow the plant community to develop. Even worse, high water levels that come too frequently may expand the width of the mudflats ring and worsen littoral habitat. Moreover, many fish species that inhabit reservoirs have developed life-history strategies that enable them to exploit periodic increases in resources, resulting in episodically strong year classes, sometimes described as cycles.

7.4.5 Vegetation Protection and Establishment

Where substrate conditions and other factors are suitable during drawdown, natural regeneration will take place from the existing seed bank or from seed blown in from surrounding upland areas. However, the shores on reservoirs often can remain bare or with patchy vegetation because of unfavorable environmental conditions. In these situations, it may be desirable to establish vegetation by artificial means (section 11.6). Conversely, excessively high and prolonged water levels kill trees and other terrestrial plants. Therefore, while high water levels inundate vegetation and that benefits fish, recurrent flooding can eliminate the vegetation.

7.4.6 Enhanced Aesthetic Value

Aesthetic value is not a major emphasis because most reservoirs were constructed for pragmatic applications such as flood control or hydropower applications. The water-level fluctuations associated with these applications prevent the growth of vegetation, create a ring of barren shoreline, and are therefore in direct conflict with aesthetics (section 7.3.1). Revegetation of this zone is possible and has been implemented (section 11.6). However, the vegetation does not last long (usually until the following flood), and recurrent plantings are necessary.

Aesthetics is a major concern in some reservoirs, or over time aesthetics might have evolved into a concern as economic and environmental values shift. In these reservoirs, options for preserving aesthetic value include limiting the magnitude of the drawdown as well as restricting the timing of the drawdown to seasons with reduced visitation. These changes are likely to conflict with the original objective of the reservoir. A study of Tennessee Valley Authority reservoirs revealed that large drawdowns, earlier in the summer and through the fall, decreased recreation suitability both through reduction of aesthetic quality and reduction of opportunity capacity (Cordell and Bergstrom 1993). These authors considered the effects on visitation under various alternative water-level management scenarios. For their study reservoirs, they estimated that gains in the economic value of recreation were greater than the losses in power generation. Results from the study eventually led to changes in the operating schedule to balance aesthetic values, recreation, and power generation.

7.4.7 Drought-Related Problems and Opportunities

Droughts and associated water-level reductions can present significant challenges to fish and water users. As a result of reduced freshwater inflow during droughts, reservoirs may experience increased salinity due to evaporation. Moreover, droughts can exacerbate algal blooms with the potential for production of the taste-and-odor compounds cyanotoxins, formation of sulfides, increased resuspension of nutrients from sediment, and depletion of oxygen. Absence of freshwater inflow, a major drought-related effect, aggravates anoxia in the hypolimnion. In dry years, the littoral zones of reservoirs are exposed to erosion resulting from reduced water levels, which changes nutrient and phytoplankton dynamics in the years following drought.

Nevertheless, droughts also provide the opportunity to improve fish habitat in the littoral zone. During low water various activities become possible, including sediment removal (section 3.7.3), renovation of connectivity passages (section 9.2), shoreline stabilization and improvements (section 5.8.2), introduction of structure (section 10.4), breakwater installation (section 5.8.2), and revegetation of mudflats (section 11.6).

7.4.8 Interagency Cooperation

Above all, efforts to enhance fish habitat in reservoirs through improved water regime would benefit most from intensive collaboration between the fish management and water management agencies. These efforts may include yearly consultations on timing of water retention and discharges. Whereas the water management agency is often mandated by law to adhere to an established guide curve, the curve has flexibility that can be exploited to improve fish habitat. However, when unable to find common ground, agencies have resorted to more vigorous request as exemplified in Table 7.1. These requests are often most effective if they are supported by data.

7.4.9 Guide Curves

The allocation of water storage volume to meet the operational purposes of a reservoir commonly is regulated through a water control plan that includes schedules to guide reservoir volume and water level (Kennedy et al. 2002). These schedules are often called guide curves or rule curves because they govern water levels throughout the year and indirectly guide releases. Guide curves can be designed to regulate storage for flood control, hydropower production, or other operating objectives, as well as a combination of objectives (Figure 7.2). Most reservoirs are operated according to

Table 7.1. Resolution submitted by the Tennessee Wildlife Resources Agency in 2002 to the Tennessee Valley Authority to request a change in water regime at Douglas Lake, Tennessee.

WHEREAS, the Tennessee Wildlife Resources Commission and the Tennessee Wildlife Resources Agency are established by the legislature of the State of Tennessee for the purpose of placing first and foremost the welfare, management, and conservation of wildlife and habitat resources; and,

WHEREAS, fishing, boating, hunting and wildlife viewing are recreational activities vital to the public and economic interest of the great State of Tennessee; and,

WHEREAS, The State Boating Safety Act was given solely to the Wildlife Resources Agency for administration and coordination; and,

WHEREAS, an estimated 392,027 fishing hours were expended in 2000 on Douglas Lake and 40% of this effort was expended by crappie anglers; and,

WHEREAS, the value of crappie angler daily expenditures in 2000 at Douglas Lake was estimated at $379,390 with an overall economic effect $811,895; and,

WHEREAS, the Tennessee Valley Authority holds authority to control water levels on the reservoirs of the Tennessee River and its tributaries; and,

WHEREAS, adequate and stable water levels throughout the peak boating and fishing seasons are crucial to the public's enjoyment and critical to the financial success of the marinas and lakeside resorts; and,

WHEREAS, water level management by the Tennessee Valley Authority has been shown to adversely affect the spawning success of several sport fish species, including crappie, in tributary reservoirs; and,

WHEREAS, the spawning success of crappie in 1999, 2000, and 2001 was low to non-existent in Douglas Lake due to low spring lake levels; and,

WHEREAS, angler catches of crappie have declined 74% between 1998 and 2000 as a result of the failed crappie spawning success,

BE IT THEREFORE RESOLVED the Tennessee Wildlife Resources Commission on the 31st day of January, 2002, and in furtherance of the unanimous Resolution of the 31st day of January, 2002, petitions the Tennessee Valley Authority Board of Directors to raise water level at Douglas Lake to full pool prior to April 1, 2002 and hold water level stable until October 1, 2002. Further, we petition the Board of Directors to provide and maintain the current minimum flow schedule in the tailwater below Douglas Dam.

BE IT FINALLY RESOLVED, the Tennessee Wildlife Resources Commission and the Tennessee Wildlife Resources Agency are committed to working with the Tennessee Valley Authority to conserve and protect the natural resources of Tennessee.

guide curves established at the planning stage to provide long-term operation guidelines for engineers. The guide curves generally do not account for year-to-year hydrological variability. Most guide curves prescribe reservoir daily target water level throughout the year as a relatively simple model that reservoir managers can apply and the public can understand. Guide curves usually are established based on analyses of historic hydrological conditions through a complicated and data-intensive process. Traditionally, guide curves were set based on the simulation of hypothetical curves compared with documented historical floods, but more recently they are based on optimization (Lund 1996) or other models (Rani and Moreira 2010). Mower and Miranda (2013a) describe a risk-based procedure for evaluating guide curves that uses long-term daily water-level records, which are often available online.

7.4.9.1 Amendment of guide curves

Over time the purpose of the reservoir may change or new purposes may be added, regulatory requirements may proliferate, and public interests for the management of the reservoir may intensify. Most of these changes may originate from increased environmental demands, but significant changes also may be instigated by recreational and water supply demand. These changes in societal objectives and demands have prompted re-examination and re-regulation of many reservoir systems across the country, often accompanied by heightened levels of controversy and technical scrutiny. Examples include guide curves at Lake Lanier and John H. Kerr Lake in the southeastern USA, both involved in litigation regarding water allocation; Lake Heron and other reservoirs on the Rio Grande River in the southwestern USA, involved in litigation concerning endangered species; and main-stem reservoirs in the Missouri River currently in litigation involving competition between navigation and environmental requirements.

Federal and state agencies as well as nongovernmental organizations administer water storage, use, and discharge in U.S. reservoirs. Water management goals depend on each reservoir's authorized purpose. Although reservoir administrators often are mandated to consider additional factors such as fish and wildlife habitat and recreational opportunities, the original authorization of the reservoir (e.g., navigation, flood control) drives the management and operation, with additional emphases sometimes added after construction.

The process used first to establish and subsequently to amend guide curves is not well understood by the public that uses the reservoirs or by the personnel from natural resources agencies charged with overseeing water quality, fisheries, and recreational needs. Often, questions and controversies arise as to how the guide curve might be amended. Fisheries managers, for example, may want to change seasonal water levels to inundate specific elevations for spawning habitat at certain times of the year. Recreational users may like to have a specific elevation at certain times of the year to provide the maximum enjoyment. Waterfowl managers may require reduced water levels at certain times of the year to provide forage for migratory waterfowl.

Environmental managers may require increased discharge to maintain water quality in tailwaters. It is difficult for user groups to ask for a change in a guide curve because they often do not understand what is required to implement change, nor are they aware of all the constraints imposed on the allocation of the resource. These constraints are sometimes embedded in the authorizing legislation, but in some cases they are part of interstate agreements, contracts, or treaties.

Conversely, it is not possible for water management agencies to implement changes that go beyond the authorization, compromise or conflict with an authorized purpose, or violate one or more of the legal agreements that constrain reservoir operations. Sometimes the structure and the water do not necessarily belong to the agency that manages storage in the reservoir. Particularly in the west, local sponsors may own the structure (or a share of it) and control all or part of the storage volume under contract or other agreement. The agency then cannot reallocate volume for other purposes without agreement from the local sponsor(s) and often from the U.S. Congress if the reservoir was constructed by the federal government.

All parties need to understand the ramifications of a requested change or of maintaining an outdated guide curve. A clearer understanding of the process could promote productive cooperation among water managers, natural resource managers, and the public.

Mower and Miranda (2013b) reviewed frameworks for amending reservoir water management in USACE reservoirs. They identified three frameworks, each with a unique process, scope, and varying degrees of flexibility. The *general investigations framework* is used to obtain congressional authorization for a new USACE project or to recommend modifications to an existing water development project to the U.S. Congress, to the extent that such an amendment exceeds the Chief of Engineers' discretionary authority. Under this framework, congressional authorization is required to make guide curve amendments. Many USACE activities and projects are not large enough in scope for congressional attention. Those changes can be accomplished through the *continuing authority program* that enables small-scale projects to move more quickly. A third framework for amending rule curves is *updating the water control plan*. This type of action is acceptable to optimize the project for general authorities passed subsequent to the original authorizing act. The framework used depends on the degree of flexibility afforded to the USACE by the authorized purpose(s) of the project. To be safe, most districts and higher-level USACE officials may be hesitant to consider the possibility of permanently amending reservoir operations without congressional approval.

In most cases operational flexibility exists for many reservoirs, and water levels can be manipulated to enhance fish production or at least to reduce substantially major adverse effects on fish communities. The key to finding flexibility is to understand the operational constraints. Amending guide curves often involves many stakeholders with many competing interests regarding old water development projects. Tradition and original purposes require serious consideration. Having clear alternatives that consider the major purposes for which the reservoir was constructed promotes productive communication and cooperation among resource management agencies, water managers, and stakeholders.

7.4.9.2 Guide curves in Kansas reservoirs

A general approach for managing water levels in Kansas reservoirs (Figure 7.9) was described by Willis (1986). It consists of (1) a spring rise and hold to flood terrestrial vegetation; (2) a summer drawdown to allow regrowth of vegetation and concentrate predators and prey to encourage predation; (3) a small autumn rise to flood terrestrial vegetation and attract waterfowl; and (4) a winter drawdown to once again concentrate predators and prey and protect remaining vegetation from water damage. Variations of this basic plan, with regard to magnitude, duration, and timing of water-level changes generally meet specific needs of most fish and wildlife conservation agencies.

According to Willis (1986) the generalized Kansas water regime successfully promoted various fish populations, including walleye, white bass, and white crappie, if it exposed >20% of the reservoir basin. In reservoirs where no effect was obvious, the basin area exposed was <13% in essentially all of the cases. Largemouth bass did not seem to benefit from this regime, apparently because the July drawdown moved juveniles away from the shore structure necessary for cover and protection from wind, waves, and predation. Diversity in timing of drawdown, rather than a rigid guide curve, could benefit a broader spectrum of the fish assemblage.

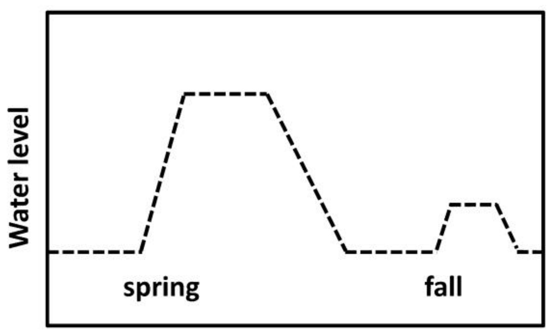


Figure 7.9. General guide curve applied to water levels in Kansas reservoirs. Modified from Willis (1986).

Section 8

Riparian Zone

8.1 Introduction

Riparian zones are areas of biological, physical, and chemical interaction between terrestrial and aquatic ecosystems and typically have high abiotic and biotic diversity. Riparian zones represent the strip of land immediately bordering rivers and streams, generally beginning at the bank and moving inland a loosely defined distance. Riparian zones have been defined as encompassing the terrestrial landscape from the high-water mark toward the uplands to the limit of where vegetation may be influenced by elevated water tables and flooding (Naiman and Decamps 1997). The riparian zone may be narrow in small streams; larger in creeks, where it is represented by a distinct band of vegetation whose width is determined by long-term channel dynamics and the annual discharge regime; and large in rivers, where it is characterized by well-developed and physically complex floodplains with long periods of seasonal flooding, lateral-channel migration, and oxbow lakes.

The characteristics of riparian zones adjacent to reservoir shorelines are somewhat unique and different from those associated with rivers, but they can be managed in a way that offers similar functional values (e.g., shade, bank stabilization, water quality). Reservoir riparian zones resemble those of rivers only at the mouth of tributaries. Near their main body, reservoirs often lack a true riparian zone because the original river channel has been submerged and the new shoreline contour is at a higher elevation and fringed by upland vegetation that is not adapted to regular flooding. The upland vegetation along this new water line is also exposed to a higher water table, and the upland trees and shrubs that cannot adapt to wetter conditions do not survive. Without the root systems to stabilize soils, shorelines become vulnerable to water fluctuations, wave action from wind and boat traffic, and overland runoff. Shoreline erosion increases sedimentation and reduces habitat quality for invertebrate production and fish that depend on shore environments during some stage of their life cycle. Consequently, depending on water level, extent of water-level fluctuations, and shoreline slope, the riparian zone along many reservoirs can vary from upland terrestrial vegetation to nonvegetated mudflats.

Riparian zones play a critical role in linking aquatic and terrestrial systems (Naiman and Decamps 1997). In river systems, beneficial functional roles of riparian zones include shading, thermal buffering, providing woody debris and bank stability, and intercepting nutrients and sediment (Pusey and Arthington 2003). In reservoirs these roles are similar, but the importance of riparian zones shifts toward protection of the shore from strong fetch, bank stabilization (by armoring banks against wave-induced erosion), and interception of sediment and nutrients. In addition to the important biotic and abiotic roles, riparian zones provide an aesthetic visual barrier (Pusey and Arthington 2003) that helps maintain quality of recreational experiences, particularly in urban and agricultural settings.

8.2 Shade

Shade provided by trees within the riparian zone is a feature of habitat structure and diversity (Figure 8.1). Fish use shade as a refuge from predation and as cover to launch predatory attacks (Helfman 1981). Shade also regulates thermal aspects of water quality by helping to moderate against extreme temperature fluctuations in the summer and winter (Quinn et al. 1992; Amour et al. 1994), and by moderating daily temperature fluctuations (Quinn et al. 1992).

Figure 8.1. A well-developed canopy along the riparian zone regulates light intensity, thermal aspects, and other environmental characteristics that promote fish species diversity. Additionally, the riparian zone controls wind fetch, is a source of leaf litter and woody debris, and can intercept sediment and nutrients. Photo credit: C. Watts, Mississippi State University.

In most reservoirs, the extent of shade provided by riparian zones in the nearshore environment is small relative to unshaded open-water areas, and the large open-water volume tends to neutralize effects of shading on water temperature and associated water quality (section 6). As a result, shade provided by riparian zones may not influence water temperature and water chemistry significantly in reservoirs, and its primary effect may be a reduction in light intensity in the nearshore environment. For example, in Columbus Lake reservoir, Mississippi, average light intensity in summer was 66% lower in shaded sites, but average temperature and dissolved oxygen were <5% lower in shaded sites (Raines and Miranda 2016).

The decrease in light intensity in shaded nearshore environments has the potential to influence biotic assemblages through competitive mechanisms associated with finding food, avoiding predation, and other aspects associated with visibility rather than through physiologic effects via water quality. In Columbus Lake, clupeids and most centrarchids were represented better in terms of abundance in unshaded sites, and percids were represented better in shaded sites (Raines and Miranda 2016). Shaded sites also tended to include intolerant species whereas unshaded sites did not. The diversity in light intensity and spectral composition of light produced at shaded and unshaded sites can create diverse mosaics of light-based habitats in nearshore environments that attract different species or life stages (Raines and Miranda 2016). This patchwork of light characteristics can enhance fish species richness and diversity and the variety of species associations. Damage to vegetation in riparian zones, or water-level reductions that move the shore away from riparian vegetation, can cause the diversity of light in nearshore environments to decrease and, generally, to become dominated by lighted habitat that lose nearshore biodiversity.

8.3 Wind Breaks

Limited information exists on the effect of riparian zones on wind fetch and its concomitant effects on reservoir environments. In Canadian natural lakes, removal of trees within reservoir riparian zones through wind blowdown and wildfires tripled the overwater wind speeds and caused deepening of the thermocline (France 1997). Moreover, in a set of lakes where riparian trees had been removed a decade earlier through either clear-cutting or by a wildfire, thermocline depths were over 6 ft deeper per unit fetch length compared with lakes surrounded by mature forests. Therefore, changes in fetch caused by tree removal in riparian zones potentially can have substantial effects on reservoir water quality. However, these relationships have not been studied adequately and are likely to vary greatly with reservoir morphometry, water retention, and operation plan.

8.4 Interception of Sediment and Nutrients

Nutrients and sediment enter reservoirs through tributaries and runoff from the surrounding landscape. A vegetated riparian zone precludes development activities that generate sediment and nutrients. Moreover, a vegetated riparian zone can trap sediment and nutrients in surface runoff, reducing sedimentation and eutrophication of the reservoir. Vegetated riparian zones also may reduce the velocity of sediment-laden storm flows, allowing sediment to settle out of water and be deposited on land (Magette et al. 1989; Daniels and Gilliam 1997).

8.5 Source of Nearshore Habitat

Riparian zones are a significant source of nearshore structural habitat in the form of leaf litter and woody debris. Leaf litter can constitute a major source of organic matter to benthic organisms. The importance of this input depends on the characteristics of the riparian zone (e.g., development and vegetation composition), the degree of shoreline complexity, and the overall productivity of the aquatic system (Gasith and Hasler 1976). Most of these organic inputs are deposited as litterfall from the vegetated riparian zone. However, in some instances inputs of terrestrial insects can produce substantial subsidies of prey for aquatic predators and for reservoir nutrient cycles (Carlton and Goldman 1984). Large woody debris from riparian zones enters reservoirs as trees that have fallen into the nearshore environment, often resulting from natural forest succession, bank sloughing, or other causes. Large woody debris can trap sediment, cushion the effects of wave action on shorelines, and reduce or prevent scouring of the banks, which help maintain diverse nearshore habitat for aquatic biota. Woody debris also adds structural complexity to aquatic habitats, and habitat complexity is an important determinant of fish species richness along reservoir shorelines (Barwick 2004). The structural complexity of woody debris itself may be important in determining the degree to which it supports fish (Wagner et al. 2015). Woody debris provides protection from fish piscivores and avian predators. Woody debris may be used as cover for ambush predators and may be an important determinant of the growth rates of piscivorous fish, potentially influencing production (Bolding et al. 2004).

8.6 Effects on Fish

Without suitable riparian zones along the reservoir periphery and tributaries, fine sediments are transferred from the watershed to shallow reservoir environments where they can affect littoral fish species. Increased turbidity due to suspended sediment and sedimentation alter food availability (e.g., algae and benthic invertebrates; Berkman and Rabeni 1987), affecting fish foraging behavior and efficiency (Bruton 1985) and altering interspecific interactions. Other detrimental effects include a reduction in habitat suitability for substrate spawners (Walser and Bart 1999), including increased egg mortality and reduction in rates of larval development and survival (Jeric et al. 1995). As the banks and associated littoral habitats degrade, density of fish that rely on the littoral zone during all or part of their ontogeny is likely to decrease.

Submerged woody habitat originating from the riparian zone influences composition of lacustrine fish assemblages. In reservoirs of the southern USA, species richness and centrarchid abundance are generally higher in coarse woody habitat (Barwick 2004) contributed from vegetated riparian zones surrounding the reservoirs. In a lake in Wisconsin, experimental removal of coarse woody habitat contributed from the riparian zone resulted in largemouth bass consuming less fish and more terrestrial prey and growing more slowly (Sass et al. 2006). Moreover, in the same lake, yellow perch declined to extremely low densities as a consequence of predation and little or no recruitment.

Deforestation in riparian zones can expose lake surfaces to strong winds. Excessive wind may deepen thermoclines and reduce habitat for cold stenotherms such as some salmonids (France 1997). Moreover, wind simply may mix the hypolimnion and epilimnion, resulting in loss of thermal refuge for species that rely on them during warm months, such as striped bass (Coutant 1985). Mixing may also cause periodic declines in water quality that could affect a subset of the fish assemblage negatively while favoring others. Excessive wind associated with deforestation of riparian zones has been linked to increased turbidity through resuspension of sediment produced by the interaction of fetch and depth in shallow reservoirs.

8.7 Aesthetics

Riparian zones increase the aesthetic value of waterscapes (Brown and Daniel 1991; Emerson 1996). In agricultural areas, riparian zones can provide a buffer between the reservoir and the cultivated terrain. In urban areas, riparian zones can create park-like areas or natural areas that buffer the water body from the urban environment. In residential and campground areas, vegetated riparian zones provide visual contrast and relief and buffer the noise from nearby highways. Also, diverse types of vegetation in riparian zones provide further enhancement of aesthetic qualities and possibly enhance the filtering value of riparian buffers.

8.8 The Reservoir Manager in the Riparian Zone

Ownership or control of the riparian zone along reservoirs varies by reservoir and among regions of the USA. In states west of the Mississippi River, riparian zones surrounding large reservoirs are often federally owned, whereas in states east of the Mississippi River, there is a greater percentage of land in private or local-government ownership. Similarly, riparian zone laws vary among states and are generally different between eastern and western states.

Most commonly, reservoir managers lack jurisdiction to manage reservoir riparian zones actively and thus must become partners in broader land management partnerships (section 2). These partnerships often include a combination of local, state, and federal government agencies, local municipalities, nongovernmental organizations, private landowners, and various reservoir users and stakeholders. A diverse group of partners can provide the capacity needed to plan, fund, and complete restoration or management of riparian zones. The makeup of these partnerships will vary and is often influenced by the cultural, political, and economic landscape and societal values of the region.

As partners, reservoir managers can demonstrate and promote the linkage between the aquatic environment in the reservoir and the riparian zone. Reservoir managers can contribute technical guidance and planning assistance in development of restoration and management plans for riparian zones. Furthermore, reservoir managers can offer science-based expertise as to the effects that specific actions or management scenarios may have on reservoir water quality and biotic communities as management options are considered. To this end, a reservoir-specific riparian zone inventory documenting features important to reservoir condition is essential, focusing on critical areas representing major sources of problems likely to have large effects on the reservoir, such as long segments of agricultural ventures stretching down to the banks, periodic forest clear-cutting operations, developments in residential or commercial construction, and eroded shorelines. A focus on critical areas would result in the greatest improvements. Help in gathering this information can be enlisted from within the partnership and from reservoir associations and stakeholder groups (section 13).

8.9 Riparian Zone Management

8.9.1 Width

Studies comparing multiple width riparian zones have shown that effectiveness increases with increasing width. Grass filter strips in particular have been shown to be very effective at trapping sediment particles. Neibling and Alberts (1979) estimated that 91% of incoming sediment to a grass filter strip was deposited in the first 2 ft of grass filter. Much of the larger particles of sediment may be removed in 15 ft of grass buffer, but finer particles may require 30 ft (Gharabaghi et al. 2002). The width required to optimize nutrient removal is less clear. Generally, 30-ft forested bands have achieved >70% retention of nitrogen, and almost 100% of nitrogen can be removed by bands 65–100 ft wide (Fennessy and Cronk 1997).

Slope is a key factor in determining sediment entrapment within vegetated riparian zones (Young et al. 1980; Peterjohn and Correll 1984; Dillaha et al. 1989; Magette et al. 1989; Phillips 1989). In general, riparian zones need to be wider when the slope is steep to allow more time for the velocity of surface runoff to decrease and deposit sediment (Barling and Moore 1994; Collier et al. 1995). In steep terrain, overland flow tends to concentrate in channelized natural drainage ways, giving rise to high-flow velocities.

The pattern and intensity of rainfall are important factors in determining the effectiveness of riparian zones. It is expected that in regions where rainfall is uniform and light, narrower riparian zones may manage most of the sediment and nutrients that enter them effectively. In areas that experience seasonal storms of high intensity, even if few such events occur, wider riparian zones may be necessary because water residence time in the buffer is decreased.

There is no one-size-fits-all width for riparian zones appropriate for all reservoirs. Width depends on needs and objectives, on the intensity of the land use surrounding the reservoir, and local climate conditions. As a rule of thumb, a 75-ft riparian zone may be sufficient for a low-intensity land-use area and a 150–ft riparian zone for a high-intensity land-use area, but all recommendations are site specific. As the width of riparian zones increases, their buffering effectiveness may reach a point of diminishing returns compared with the investment involved. Therefore, managers may develop guidelines that remain flexible to site-specific needs to achieve the most benefits as is practical (Figure 8.2).

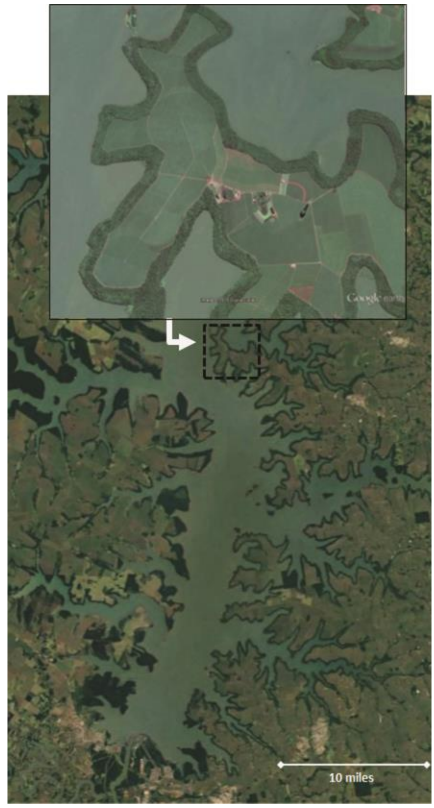


Figure 8.2. Itaipu Reservoir on the border of Paraguay (left) and Brazil (right). Note the well-developed managed forested riparian zone on the Brazilian side. The riparian zone is a minimum of 500 ft wide. Image credit: Google Earth.

8.9.2 Three-Zone Buffers

Riparian zones with multispecies strips may best protect water bodies against effects associated with land disturbances because of the different modes of particulate and dissolved contaminant transport (e.g., Schultz et al. 1995). This concept is based on three interactive buffer strips within the riparian zone that are in a consecutive upslope order from shore: a strip of permanent forest, a strip of shrubs and trees, and a strip of herbaceous vegetation (Figure 8.3; Table 8.1). Width and vegetation composition of this basic model is adapted to the geographical variability of terrestrial plant communities and riparian zone condition (Sparovek et al. 2002). The first strip of forest influences the aquatic environment directly (e.g., temperature, shading, bank stability, wind break, source of coarse woody debris). The second strip incorporates shrubs and

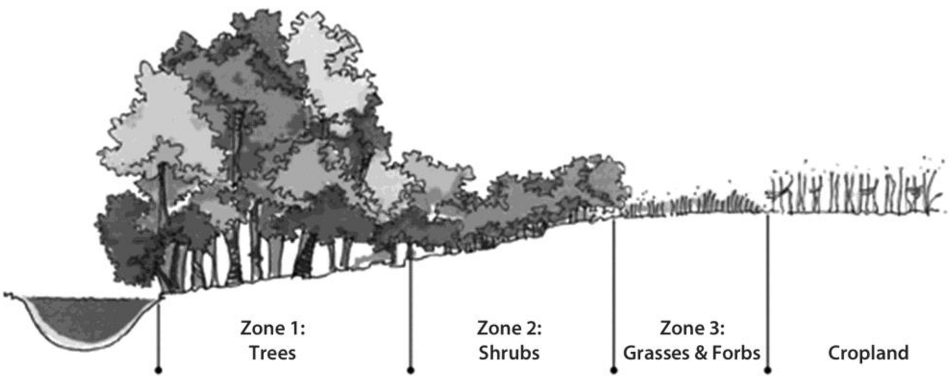


Figure 8.3. Three-zone engineered riparian buffer (see Table 8.1). Image credit: University of Kentucky Cooperative Extension Service, Lexington.

trees to control pollutants in subsurface flow and surface runoff; this strip is particularly important because this is where biological and chemical transformations, storage in woody vegetation, infiltration, and sediment depositions are maximized. The first two strips contribute to nitrogen, phosphorus, and fine-sediment removal. The third strip consists of grasses to spread the overland flow, thus facilitating deposition of coarse sediment. Grassy riparian areas trapped more than 50% of sediment from uplands when overland water flows were <2 in deep (Magette et al. 1989). Grassy buffer strips are effective at filtering sediment and sediment-associated pollutants (particulate phosphorus and nitrogen) from surface runoff. However, they are less effective in removing soluble nutrients such as nitrate, ammonia, and dissolved phosphorus. Nitrate removal from subsurface flows is considered to be greater in forested buffers, partly through uptake by plants (Fennessy and Cronk 1997; Martin et al. 1999). Wetlands and soils in riparian zones have a high capacity for denitrification compared with terrestrial and aquatic soils.

Riparian zones accumulate nutrients and absorb them into plant biomass, thus serving as nutrient filters. In North Carolina, riparian zones removed up to 80%–90% of the sediment leaving agricultural fields (Daniels and Gilliam 1997). In Vermont, reductions of approximately 20% in mean total phosphorus concentration and 20%–50% in mean total phosphorus load were observed (Meals and Hopkins 2002). In Lake Rotorua, New Zealand, riparian zone management reduced sediment loads by 85%; particulate phosphorus and soluble phosphorus by approximately 25%; and particulate nitrogen and soluble nitrogen by 40% and 26%, respectively (Williamson et al. 1996). These reductions were predicted to reduce the chlorophyll-*a* concentrations in the lake by approximately 5 ppb and help shift the lake's trophic state from eutrophic to mesotrophic. The effectiveness of riparian zone restoration in sediment and nutrient reduction is diminished during periods of high runoff and outside the growing season, which, depending on geography, is often when the highest discharges occur.

Table 8.1. Three-zones buffer model for riparian areas. Width and vegetation composition of this basic model is adapted to the geographical variability of terrestrial plant communities and riparian conditions.

Purpose	Vegetation	Management considerations
	Zone 1	
Creates a stable ecosystem adjacent to water's edge; reduces runoff nutrient levels; provides shade; contributes organic matter and large woody debris	Native riparian trees, shrubs, forbs, and grasses suited to a wet environment and of value to wildlife; use fast-growing tree species where banks must be stabilized	Exclude heavy equipment; remove trees only for hazard reduction; discourage livestock; avoid concentrated surface runoff through use of flow spreaders
	Zone 2	
Provides contact time for carbon and energy sources to stabilize and store nutrients	Predominantly native riparian trees, shrubs, forbs, and grasses	Avoid gullying by maintaining vegetation and grading; management for timber or wildlife is encouraged, but leaf litter and shade levels should be maintained
	Zone 3	
Provides area to convert concentrated overland flow to uniform sheet flow; promotes deposition of sediment, infiltration of runoff, and uptake of nutrients by vegetation	Dense perennial grasses and forbs; an ungrazed grassland may serve as Zone 3	Maintain vegetation in vigorous growth stage; weed control may be needed; periodic reshaping may be necessary to prevent gully formation

Phosphorus accumulates in the soils of riparian zones and can be taken up by plants, but there is no process similar to denitrification that removes phosphorus to the atmosphere. Therefore, riparian zones potentially could become saturated with phosphorus, and their ability to trap phosphorus may decline with age unless sediment or organic matter is removed from the riparian zone (Barling and Moore 1994). Thus, harvesting trees or plants from the riparian zone can provide a mechanism whereby phosphorus is removed.

Figure 8.4. Fencing installed to manage grazing along a reservoir in the Missouri River, Montana. Photo credit: Missouri River Conservation Districts Council, Helena, Montana.

8.9.3 Livestock

Usually, livestock grazing adversely affects water quality, hydrology, riparian zone soils, and bank vegetation and stability

(Kauffman and Krueger 1984). Livestock damage to riparian vegetation and soils destabilizes the banks and leads to mobilization of fine sediment, which in turn causes sedimentation in shallow-water areas adjacent to shore and causes reduced water clarity. The resulting increased sediment load is accompanied by particulate nutrients that may contribute to eutrophication. Further, livestock contribute nutrients directly to riparian areas through feces and urine. Fecal material deposited in damaged riparian zones may readily wash overland into the water with little opportunity for filtration.

Nevertheless, most riparian zones evolved with animals feeding on the lush vegetation and stepping on banks while accessing water. Although the original grazers were bison, moose, and deer rather than cattle, sheep, and goats, this evolutionary pressure resulted in regrowth of native riparian plant species following a period of grazing (Ohmart 1996). When farmers and ranchers displaced these occasional grazers with continuously grazing livestock, quality of riparian zones decreased. Provided with limited grazing area and little stimulus to move from one area to another, continuous grazers trample banks, congregate in the shade and cool breezes next to water, and overgraze the lush vegetation in these fertile areas. Occasional grazing of riparian zones may be unavoidable, particularly in the western USA. Strategies for attracting livestock away from riparian zones include providing alternative watering systems; planting palatable forage species on adjacent upland areas; using prescribed burning on upland areas to enhance forage production and palatability; and placing feed supplements such as salt, grain, hay, or molasses in upland areas of paddocks away from the riparian zone. Fencing often is required to prevent overgrazing and can be cost-shared with government agencies (Figure 8.4). Sometimes fences may not be intended to create riparian exclosures but rather to create riparian pastures that can still be grazed according to the goals for balancing livestock and natural resources.

Figure 8.5. General approaches to bank stabilization include live planting, bioengineering, and hard armoring. Photo credits: U.S. Army Corps of Engineers, Savannah District (upper and lower photos), and The Nature Conservancy, Rhode Island (middle).

Brush removal by cattle can maintain grassy buffers that help protect water. Light grazing in riparian zones may be managed to mimic the activities of native wildlife by grazing small herds for a limited time and by grazing at different times of the year. In Wisconsin fisheries managers often contract with farmers to graze riparian zones rotationally (Lyons et al. 2000). Goats are used to control noxious weeds and nonnative brush species in riparian zones, allowing for the growth of plants that provide healthy riparian conditions. Detailed guidelines for managing grazing within riparian zones are available (e.g., Clary and Webster 1990; Leonard et al. 1997; Swanson et al. 2015).

8.9.4 Bank Stabilization

Overall, there are three general approaches to bank stabilization: live planting, bioengineering, and hard armoring (Goldsmith et al. 2013; Figure 8.5). Live planting involves sowing vegetation appropriate for the site and region. Bioengineering relies on a combination of structural components and plant material to produce a dense buffer of vegetation that serves as a "living system" to protect shorelines. Hard armoring involves installing structures such as breakwaters, revetments, and bulkheads (section 5.8.2). Shorelines with steeper slopes tend to experience greater erosion, so reducing the slope of a shoreline dissipates wave energy and lessens erosion. In reservoirs, a fourth approach may be to establish no-wake zones in nearshore, shallow areas.

8.9.5 Facilitation of Interactions between Reservoir and Riparia

Of the methods used to provide woody structure in reservoirs, felling large trees most closely duplicates natural processes. In areas where trees in the riparian zone are numerous, the "hinge cutting" method of felling selected trees can accelerate development of nearshore fish habitat (section 5.8.2.12). The technique involves cutting selected trees near their base just deep enough so that the tree can be pushed into the water but remain attached to the trunk (Figure 8.6). Hinge-cut trees cut about two-thirds of the way through the trunk may continue to live for months to several years. Trees may be cut as clumps of two to three to maximize structural complexity. Younger trees work best because older trees may tend to break when felled. In Smithville Lake reservoir, Missouri, managers hinge-cut over 6 mi of shoreline and documented that hinge-cut trees were sheltering large concentrations of juvenile fish.

Figure 8.6. Hinge-cut trees in Smithville Lake, Missouri. Photo credit: Missouri Department of Conservation, Columbia.

8.9.6 Residential Development Management

Residential development of lakeshores is associated with changes in key ecosystem characteristics, including nutrient levels (Moore et al. 2003), aquatic vegetation (Radomski and Goeman 2001), and the spatial distribution (Scheuerell and Schindler 2004) and growth rates (Schindler et al. 2000) of fishes. Residential development increases the area composed of impervious cover such as driveways, parking lots, rooftops, sidewalks, and lawns and decreases the area composed of native plants and undisturbed soils (Figure 8.7). These artificial surfaces collect pollutants such as oil and gas, heavy metals, fertilizers, and pesticides, which can be washed into the reservoir. These surfaces also prevent water from infiltrating the ground. When storm water cannot infiltrate the ground, it is collected and discharged through storm sewers, drainage ditches, or some other means of conveyance, often into downstream reservoirs. Pavement can also lead to increased temperatures of the storm water entering the reservoir. Mitigation of residential development may be as simple as not mowing within a riparian zone or changing land management and yard care practices, or as complex as changing zoning ordinances or widening riparian zones through buyouts.

Figure 8.7. Unmanaged residential development can reduce the ability of the riparian zone to perform its natural functions. Photo credit: D. Fogia.

Residential developments in rural areas surrounding reservoirs generally depend upon septic systems as public sewer systems may not be available. On-site septic systems can be safe and efficient if designed, installed, and maintained properly. A septic system relies on natural bacteria in a tank to break down solid matter. A drain field then transfers the liquid waste into the soil for treatment. Malfunctioning septic systems can leak effluent with high concentrations of nutrients and bacteria into the reservoir. Nutrients entering the reservoir can cause algal blooms and excessive growth of unwanted aquatic plants.

8.9.7 Effects of Drawdown on Connectivity to Riparian Zone

Drawdowns in reservoirs move the edge of the water away from the riparian zone. The regulated zone in storage or flood control reservoirs may be very wide in some cases and may be devoid of vegetation unless the drawdowns last several years. Frequent flooding within the regulated zone discourages establishment of terrestrial vegetation both by surface erosion and the physiological effects of periodic inundation on terrestrial plants. During periods of low water, the exposed regulated zone is colonized by herbs and seedlings of shrubs and trees. The extent of their development in the regulated zone reflects timing and length of the drawdown.

Frequency and duration of inundation of the regulated zone diminishes laterally away from the reservoir. At normal pool elevation, riparian vegetation in areas close to the main reservoir is characterized by younger stands, commonly composed of deciduous shrubs and trees. Portions of the riparian zone farther from the reservoir may contain older plant communities composed of either typical riparian species where tributaries enter the reservoir or upland species away from tributaries and in upland areas flooded by the reservoir. Thus, reservoirs with fluctuating water levels may have a riparian zone only part of the time; the rest of the time the riparian zone may be represented by a barren band or ring that follows the contour of the regulated zone. Providing diverse fish habitat within this contour is challenging. Reservoir managers have resorted to artificially introducing some of the features (e.g., large woody debris, and aquatic and terrestrial plants; sections 10 and 11) normally provided naturally by a functional riparian zone.

Reservoirs impounded low in a river basin over floodplain rivers, such as flood-control and navigation reservoirs, are unique because they may include within their upper reaches extensive shallow water stored over the original floodplain. Moreover, reservoirs that experience large seasonal water-level fluctuations as part of their operational objective periodically inundate and dewater floodplains associated with their upper reaches, partly mimicking the natural inundation of river floodplains. Because of their relatively flat topography and riverine origin, floodplains in the upper reaches of reservoirs provide broad expanses of facultative wetland vegetation within a narrow range of reservoir water-level elevations. Vegetation in these floodplains is important because wetland plant species can provide suitable vegetated habitats for fish at elevations below normal pool. These may be the only flooded vegetated habitats in the reservoir early in the spawning period when water levels may not be high enough elsewhere in the reservoir to flood vegetated habitats. Moreover, access to vegetated habitats below normal pool in the upper reaches of these reservoirs precludes the need to flood upland vegetation above normal pool every year, vegetation that would inevitably be impaired by regular flooding. Thus, management of riparian zones may also include management of floodplains in the upper reaches of reservoirs with the goal of preserving or restoring, through judicious water-level management, key vegetated habitats.

8.9.8 Conservation Easements

Conservation easements can be a valuable tool when the manager perceives the need for the permanent conservation of a reach of riparian zone that is directly or indirectly providing quality fish habitat (Figure 8.8). A conservation easement is a section of land where the right to develop has been donated or sold by a landowner to a government entity or a nonprofit land trust. The purpose of a conservation easement is to preserve property in its predominantly undeveloped, natural, scenic, or forested condition and to prevent any use of the property that will impair or interfere significantly with the identified conservation value. The Nature Conservancy is one of the largest nonprofit holders of conservation easements in the USA (Kiesecker et al. 2007). The landowner retains ownership and pays property taxes. Although taxes continue to be paid, a landowner who donates a conservation easement may be eligible for a reduction in federal income taxes and a reduction in the value of the property for the purposes of property tax valuations and estate taxes (Morrisette 2001). Indeed, these tax factors can be a significant component of a landowner's motivation to donate the easement.

Figure 8.8. Conservation easements are a legal tool to preserve riparian habitats of value to reservoir fish. Photo credit: U.S. Natural Resources Conservation Service.

Conservation easements are privately initiated land-use restrictions designed to protect and preserve private lands from development. They commonly are used to protect open space and scenic sites or preserve wildlife habitat and historical structures or cultural sites. The owner retains title to the land and may continue to use the land subject to restrictions imposed by the easement. Thus, the owner retains all rights to the property that the owner possessed prior to the easement subject to the restrictions imposed by the easement. The owner may continue to exclude the public from lands protected under a conservation easement, unless the easement provides for public access. The owner also may sell the property or pass it onto heirs, but the property remains bound by the terms of the conservation easement—conservation easements convey with the land and are usually perpetual unless the easement stipulates otherwise. Typically, a conservation easement prohibits any further development of the land unless it is related to a use of the land that is permitted by the easement. For example, a conservation easement on a large section of land along a reservoir may prohibit the owner (current or future) from subdividing the property; the owner, however, is permitted to continue using the property for its current use and is allowed to make improvements to the property that are related to its current use. Ensuring that the property remains in the current use may be the primary reason behind the conservation easement.

Section 9

Lateral Connectivity

9.1 Introduction

The extent of backwaters adjacent to reservoirs varies and controls their influence on reservoir fish assemblages. Backwaters may include wetlands, sloughs, and oxbow lakes within the floodplain adjacent to the reservoir and may be inundated or connected periodically and temporarily by the reservoir. Reservoirs have been constructed for many purposes, including flood control, navigation, water supply, hydropower, and recreation (Kennedy 1999), and differ in the extent of backwater availability based on where they were built and how they are operated. For example, reservoirs positioned lower in a basin, such as flood control and navigation reservoirs, tend to have more extensive floodplains than do deeper hydropower reservoirs that are generally located higher in a basin where backwaters tend to be more limited. Thus, lateral connectivity tends to be more relevant in lowland reservoirs.

The rehabilitation of large rivers and reconnection of isolated floodplains and their associated habitats has become a critical component of river ecosystem restoration (Galat et al. 1998; Holmes and Nielsen 1998; Sear et al. 1998; Buijse et al. 2002; Florsheim and Mount 2002). Techniques used to reconnect river–floodplain systems are at the early stages of development and have for the most part received limited or no attention in reservoirs. Reservoir backwaters are often ignored because they are considered to be drastically transformed by the effects of impoundment. To various extents backwaters have been permanently submerged by the reservoir, but in many river systems, particularly lowland rivers, abundant backwaters remain principally in the upper end of reservoirs (Oliveira et al. 2005; Buckmeier et al. 2014; Miranda et al. 2014). These backwaters, if accessible, can benefit reservoir fishes and riverine fishes that use the reservoir seasonally.

Lateral connectivity has been suggested as a major determinant of species richness and species composition for many taxonomic groups (Tockner et al. 1999). Influenced by flood-pulsing of the river and by artificial flood pulses imposed by the management of water levels in the reservoir, floodplains, sloughs, backwaters, wetlands, and tributaries contiguous to reservoirs can provide essential habitat for lacustrine and riverine fishes. These water bodies are used by many reservoir species for spawning and nursery sites, by permanent residents, and by species that live in the reservoir or tributaries seasonally or during specific life stages. Connectivity also benefits riverine species that depend on floodplains and backwaters to complete life cycle processes (Miranda et al. 2014).

Connectivity between a river and its floodplain is a time-dependent occurrence linked to the hydrological dynamics of the river (Tockner et al. 1999). The occurrence of a river–backwater connection depends on prevailing hydrologic conditions within the river and the surface elevation of the floodplain. As river stage exceeds floodplain elevation thresholds on the ascending limb of the hydrograph, a connection occurs and floodplains and backwaters are inundated. There is a large body of literature about the interaction between rivers and floodplains (reviewed by Ickes et al. 2005), but there is a scarcity of data about the interaction between reservoirs and floodplains. For reservoirs, lateral connectivity is dependent on the water level in the reservoir and on adjacent topography. Dams may have strict temporal release schedules dictated by the operational goals of the reservoir. Such artificial hydrographs tend to make connection to backwaters more disciplined and possibly temporally inharmonious with the movement, reproduction, feeding, and refuge needs of floodplain species.

Figure 9.1. Lake Texoma, Oklahoma, where the Washita River feeds into the reservoir. The pale green line indicates the original river. The red arrows indicate areas that are nearly continually isolated from the main reservoir by sediment deposition and connect only at very high water levels. The yellow arrows indicate areas that are frequently isolated from the main reservoir. The dashed line indicates where inflow is depositing sediment following the pattern of natural river-levee deposition; these depositions are the ones primarily associated with isolation of areas and may be areas in which to target connectivity efforts. Photo credit: Google Earth; photograph annotated by T. Patton, Southeastern Oklahoma State University.

Sedimentation can cause contiguous water bodies to become physically separated from the reservoir. The hydrodynamics of many reservoirs require storage of water high in suspended sediment that generally settles near the mouth of tributaries as water enters the reservoir (section 3) or in pockets with reduced flows anywhere in the reservoir. Coincidentally, in many reservoirs, backwaters occur near the entrance of tributaries. Over time (often a few years or decades) tributaries and associated backwaters become isolated from the reservoir

(Figure 9.1) except during peak flows (Patton and Lyday 2008). Similarly, pockets of water next to channels trap sediment and develop sediment plugs near their entrance, eventually becoming seasonally or permanently isolated from the reservoir (Figure 9.2; Slipke et al. 2005). These areas may become inaccessible to seasonal fish use, trap adult and juvenile fish requiring access to the reservoir or tributaries, prevent utilization by anglers or other users, or even go dry from lack of connectivity to surface or ground water.

Loss of connectivity also develops in shallow embayments and major tributaries through the fragmentation created by the combination of sediment deposition and accretion. As these areas of the reservoir become filled with sediment, water that flows into the reservoir helps to form channels by depositing sediment on both sides of the flow channel. Although this process is occurring within the reservoir basin, it is similar to the formation of a natural river levee. As discharges exceed the banks, water spills out of the channel, losing much of its energy and allowing sediment to fall out of the water column and deposit adjacent to the channel. Over time, this process tends to separate the channel from the backwaters and isolate backwaters from each other. The resulting landscape resembles a series of reservoir fragments bisected by a riverine channel (Figure 9.1; Patton and Lyday 2008). It is not clear what short and long-term effects this isolation has on fish assemblage composition. Nevertheless, it is likely to affect fisheries negatively by reducing access to floodplain fishing sites.

Figure 9.2. Bendway (channel of the original river) isolated from the uplake section of Columbus Lake reservoir, Mississippi. The river was channelized to facilitate navigation, and bendways were cut off on both sides of the new navigation channel. Cutoff bendways trap sediment that creates sediment plugs (SP) near their entrance and eventually become seasonally or permanently isolated from the reservoir unless weirs are installed to increase flow through the bendway or closing structures prevent flow into the bendway. Photo credit: Google Earth.

Connectivity to backwaters may also be reduced by the effects of upstream dams. Chains of dams reduce flow variability and thus attenuate floods that otherwise would have inundated side channels and floodplains to connect isolated backwaters. Reduced floods in floodplains in the upper reaches of reservoirs may no longer flush fine sediment that can accumulate and may exacerbate loss of connectivity to backwaters.

9.2 Maintenance of Lateral Connectivity

Connectivity between floodplain aquatic habitats and the main reservoir can be maintained, restored, or created through several procedures (Roni et al. 2005). Submersed check dams, connecting channels, water-level manipulation, and levee setbacks can restore connectivity to existing backwaters. Closure structures can reduce back-flooding and sedimentation and thus avert loss of connectivity. The use of notched dikes and culverts may also provide opportunity for creating additional connectivity.

9.2.1 Submersed Check Dams

Submersed check dams are installed in uplake riverine stretches of reservoirs to raise the bed elevation of the main channel when it has been incised, often by channelization to support navigation. Raising the bed elevation in riverine sections raises water levels and permits reconnection to the former connecting channels and floodplain. For example, in the Danube River in Slovakia, check dams in lateral artificial channels have been used to aggrade the river high enough so that older channels are now reconnected, improving the retention time of water in the reach (Cowx and Welcomme 1998). Similarly, in the Kissimmee River, Florida, channel filling has been used to reconnect old meanders and floodplains (Toth et al. 1993). The meanders had been cut through by a channel designed for navigation. The channel was filled with levee material at points in the river where the meander crosses the main channel to raise water level and connect the channel to the adjacent meander.

Figure 9.3. Dredging may be necessary to maintain or create connections between the reservoir and water bodies in the floodplain. Photo credit: U.S. Army Corps of Engineers (USACE), Rock Island District.

9.2.2 Connection Channels

Channels dredged within the floodplain allow connection between the main reservoir and key backwater aquatic habitats (Figure

9.3). Even if no aquatic habitats exist in the floodplain, or these habitats are inaccessible, channels within the floodplain provide entrance and exit routes that facilitate use of the floodplain when it is inundated. Channels are often dredged with fingers, which are subchannels that extend out and away from the main dredge cut (USACE 2012).

The depth, length, and width of the channels depend on several site-specific factors. Some of these factors relate to biological concerns, logistics of dredge equipment mobilization, sediment–substrate characteristics, and hydrology and hydraulics (USACE 2012). Determination of the desired dredging depth includes assessment of typical water-level elevations present at the site of channel construction, desired maintained water depth, and the projected sedimentation over the expected life of the project. In northern latitudes, the maintained water depth is determined from the anticipated maximum ice depth and the desired maintained water depth below the ice. A desired water depth of 2–4 ft below the ice is typically optimal and translates to a maintained water depth of 4–6 ft. Nevertheless, flow conditions can alter the formation of ice, so shallower channels may be adequate if flow is present during winter. Width of the channel will be determined by existing channel conditions, project requirements, and project funding. Typically, dredge cuts are designed based on desired bottom width and a side slope of 2–5:1. The side slopes depend on the type of material that is being dredged. The channel may also be cut with vertical slopes, thereby allowing bank sloughing until the natural angle of repose is achieved and minimizing project cost by reducing dredging volume and time.

The location of the channel intake relative to incoming flow is critical for controlling sediment introduction into connecting channels (USACE 2012). Typically, a channel may require a dike or control structure and bank armoring at the entrance to protect it from bank erosion. Nevertheless, sedimentation is inevitable if the area is flooded periodically and inflow contains high levels of suspended sediment. If the water table is high, groundwater-fed channels can be excavated. Groundwater-fed channels offer stable year-round water flows with limited suspended sediment and stable water temperatures. Regardless of how channels are constructed, it is important to ensure that they are connected to the reservoir at least seasonally but preferably consistently throughout the year.

9.2.3 Water-Level Manipulations

Water-level management involves increasing water level to achieve a connection between the reservoir and isolated or poorly connected water bodies in the floodplain. Water-level manipulation is perhaps the easiest and least costly method for achieving periodic lateral connectivity. However, the feasibility of water-level manipulation as a tool for connecting isolated backwaters depends on reservoir attributes and operational characteristics (section 7). Lowland reservoirs are likely to show a more noticeable response to increased connectivity through water-level manipulation because smaller changes in elevation produce relatively larger increases in connectivity and because lowlands are likely to have more backwaters. Flood control reservoirs experience some of the greatest annual vertical fluctuations, so they are more likely to reconnect to water bodies in their floodplains.

9.2.4 Levee Setbacks

A levee setback is an earthen embankment placed some distance landward of the bank of the main-stem reservoir or tributary stream (Roni et al. 2005). Setbacks are applicable on available government land or land that can be acquired from private ownership through conservation easements (section 8.9.8). Setbacks allow for the development of bypasses for large tributaries, flooding a land area usually dry but subject to flooding at high stages. Levee setbacks allow the reservoir to spread by creating a wider, connected floodplain with increased conveyance capacity of the floodway. Levee setbacks provide floodplain storage benefits and sustain dynamics of the river system, which depends on recurring flood events. The passage of water and sediment in the channel, and their exchange between the channel and the floodplain, characterizes the physical environment and effects of habitat, biodiversity, and sustainability of the river. Levee setbacks would also permit an active, natural meander belt on tributary rivers that do not need to be maintained for navigation, thereby improving the floodplain habitat.

9.2.5 Closure Structures

Closure structures are constructed across the entrance to backwaters to reduce sediment conveyance through back-flooding (USACE 2012). There are two types of closure structures, the submerged closure structure and the emerged closure structure. Submerged structures may take the form of underwater rock sills built higher than the bed of the channel. Safety for recreational boats is a consideration because the location of submerged structures is not visible. Usually an elevation resulting in a depth of at least

Figure 9.4. Partial closure structure at the Weaver Bottoms secondary channel, Pool 5, Upper Mississippi River. Photo credit: USACE, Rock Island District.

4 ft during low flow conditions is specified based on recreational boating concerns. Emerged closures (i.e., those with a top elevation higher than the water-surface elevation) are generally constructed to the bank-full flood elevation (Figure 9.4). If built to close the entrance fully, a low-flow notch may be included to allow boat access and continuous two-way flow of water during low-flow conditions. Because most closure structures are designed to be overtopped, they can experience significant hydraulic forces during flood events and therefore are usually constructed of rock (e.g., riprap).

9.2.6 Notched Dikes

To maintain adequate flows within channels, and to reduce sedimentation rates within channels, engineers construct dikes to train channels (Roni et al. 2005). Some dikes are perpendicular to river channels to direct flow toward the middle of the channel, and some dikes are longitudinal running parallel to channels. The longitudinal dikes are often installed to keep the flow within the channel and off adjacent backwaters. Notching a longitudinal dike or other closing structure allows fish exchange between the main channel and backwaters, as well as access to boat anglers. Notches are large enough to accommodate recreational boat traffic. The notch's bottom elevation is typically at least 3.5 ft below normal pool elevation.

9.2.7 Culverts

Culverts are structures used to convey surface runoff through embankments. Embankments are often installed around the reservoirs to support roads or other structures. These embankments can isolate the back end of coves or embayments as well as small wetlands, bogs, or backwaters adjacent to the reservoir (Figure 9.5).

The installation of appropriate-diameter culverts or pipes under the roads in normal seepage channels can provide connectivity between the reservoir and the isolated backwater. This connectivity can facilitate exchange of water and dispersal of biota. In most situations, state or county roads departments are able to provide and install culverts.

Figure 9.5. Culvert installed on an embankment built across the back of a cove. Photo credit: www.slideshare.com.

9.3 Maintenance of Adjoining Habitat

Adjoining backwaters, whether connected to the reservoir all the time or during occasional high-water periods, may be maintained or created through several procedures. Weirs and pump systems maintain water in backwaters to avoid detrimentally shallow depths or desiccation. Depression pools and deep holes can be constructed in floodplains to create new backwaters to compensate for losses due to disconnections.

9.3.1 Weirs

Reservoir backwaters are important spawning and rearing grounds for many species and consequently are inhabited by a diversity of fish species and life stages. However, premature or excessive dewatering of backwaters can occur as a result of channelization, channel incision, water diversion, loss of storage caused by sedimentation, and other modifications to tributaries that result from impoundment. Low water contributes to hypoxia and high temperatures in isolated backwaters, concentrates fish, and may result in fish kills. Increasing depth in these backwaters can improve environmental conditions during key periods. Increases in depth may be achieved by installing rock weirs, earthen embankments, or earthen embankments covered with riprap (Figure 9.6). For additional water-level control, a water-control structure may be installed at the embankments (Ickes et al. 2005). The water-control structure may be operated to retain water during low-water conditions, to allow more frequent connection to the reservoir during high-water periods, and to promote fish accessibility. Whereas these weirs may increase isolation, they do help retain suitable depths and water quality in backwaters until seasonal high waters can reconnect the backwater to the main body of the reservoir.

Figure 9.6. Weirs installed to retain water and prevent premature dewatering of backwater areas. Photo credit: USACE, Rock Island District (upper), and R. Kröger, Mississippi State University (lower).

9.3.2 Pump Systems

During periods of low water in the reservoir, water may be maintained in backwaters by a system of pumps (USACE 2012). Pumps can provide either groundwater or surface water. The volume of water required generally will dictate whether a groundwater well will be feasible. Water is pumped as needed to maintain the desired elevation. Permanent pumping stations (Figure 9.7) require a pump(s), housing, revetment reinforcement, and flood protection structures and have an annual (seasonal) pumping cost.

Temporary pumping stations (Figure 9.8) may be established if only occasional pumping is required. In many situations pumping may be cost prohibitive, although a cost–benefit analysis may be conducted. A concern is the possibility of translocation of fish eggs or larvae during the pumping process, so adequate filters may need to be installed if translocation is not an objective.

Figure 9.7. Permanent pumping station at the Andalusia Refuge, Pool 16, Upper Mississippi River. Photo credit: USACE, Rock Island District.

Pumps may be electric, diesel, or propane driven depending upon the availability of utility power and user needs. Electric-d riven pumps have the advantage of being quieter to operate (little vibration), providing easier automation, and requiring less routine maintenance. Some of the disadvantages are that the electrical equipment has to be protected from flooding, the available utility power can limit capacity, there can be a costly high-demand charge, and usually larger, more elaborate structures are required to house electrical equipment. Diesel-driven and propane-driven pump stations are suitable where utility power is unavailable. They have a large capacity, can be permanently mounted with submersible gear drives, or can be trailer-mounted or tractor-mounted for rapid redeployment if there is a threat of flooding. Disadvantages to diesel and propane driven pumps are that they are noisy to operate, require more routine maintenance, and are difficult to automate. Also, capacity and availability of on-site fuel supply can be restricted.

Figure 9.8. Portable pump used in Lake Odessa, Pools 17-18, Upper Mississippi River. Photo credit: USACE, Rock Island District.

9.3.3 Excavated Pools

Excavated pools provide habitat diversity by creating artificial backwaters within the floodplain (Figure 9.9), either connected or disconnected from channels carved within the floodplain. These pockets fill with water and may allow for growth of aquatic vegetation. If deep enough (depending on location of the water table) the depression pool may have continuous access to groundwater. Otherwise, to prevent desiccation these pools may depend on surface flows and connection through carved channels.

Figure 9.9. Borrow pits (left side of the photo) along the Tennessee–Tombigbee Waterway in Columbus Lake reservoir, Mississippi. Connectivity between these waterbodies and the reservoir varies depending on distance from the reservoir, elevation, and levees but can often be engineered. Photo credit: Google Earth.

Size of the depression pool may be important, but there are limited data available. In borrow pit lakes along the Mississippi River fish assemblages were linked to engineered morphologic features, suggesting that diversity in engineered features can contribute to diversity in fish assemblages (Miranda et al. 2013). The deeper borrow pits had fish assemblages similar to those in riverside oxbows, whereas the small and shallow borrow pits included a higher representation of fish species that inhabit small, palustrine waterbodies and are adapted to periodic hypoxia and shallow conditions. Nevertheless, more research is needed to match engineering designs with fish assemblages that meet management needs. Larger pools at least 5 ft deep are less likely to desiccate, and pools may need to be a minimum of 6–8 ft deep to prevent them from freezing solid in colder latitudes.

Floodplain soils are very diverse. Therefore, prior to constructing a depression pool, a detailed soil analysis can determine soil type, permeability, and compaction. The desired soils to hold water within the depression pool are clays, which have the lowest permeability. The site will also need good compaction in order to improve the impermeability of the clay.

If borrow material is needed for a proposed project, project designers may consider incorporating depression pool designs into the project, thereby gaining habitat benefits through beneficial use of borrow and placement of excavated material.

Section 10

Artificial Reefs and Structures

10.1 Introduction

Submerged structures are often lacking in reservoirs because of removal during construction, decomposition over time, lack of recruitment potential from the riparian zone, or little structural material in the landscape prior to impoundment. Deficiency of submerged structures can have negative effects on fish abundance, ecological diversity, and fisheries (Bolding et al. 2004; Wills et al. 2004). The lack of structure has been identified as a major habitat degradation in U.S. reservoirs, although its preponderance varies across geographic regions (section 1).

Installing reefs and spawning structures in reservoirs has been a common practice. The overarching goal of these installations has been to enhance the naturalness of the artificial aquatic environment created by the impoundment and to aggregate fish to facilitate predator–prey interactions, including fishing. Pragmatic goals of artificial reefs include creating new fishing sites, improving angling efficiency, providing more food for fish, increasing growth rates, improving reproductive success, improving juvenile survival, providing protection from predators, and in general improving fish production. The addition of structural habitat may increase carrying capacity and biomass, at least in the areas where the structures are placed (Bortone et al. 1994; Polovina 1994; McCann et al. 1998). Enhancements with reefs may increase species diversity, complexity of trophic interactions, and ecosystem stability (McCann et al. 1998; Neutel 2002).

Installing supplementary habitat is a common habitat management activity amongst freshwater fisheries management agencies, with 80% of state agencies in the USA having installed some type of supplementary habitat enhancements (Tugend et al. 2002). However, habitat addition is far from a ubiquitous management strategy, as less than 20% of regulated lakes and reservoirs have received habitat enhancements (Tugend et al. 2002). Common types of artificial structures include woody materials (e.g., trees, brush, lumber), stone materials (e.g., gravel, riprap, boulders), and synthetic materials (e.g., plastics). The effectiveness of supplementary habitat varies greatly depending on objectives, materials, structure size and morphometry, target species, and existing habitat.

10.2 Expected Benefits

Fish habitat programs are a popular and effective way to increase catch rates. However, attraction is not a big concern where fishing effort is low, and attraction can promote overharvest where fishing effort is high. Moreover, whereas fish attractors can concentrate a fish population, too many fish attractors could again dilute the local density of target species. Other benefits of artificial reefs and structures may include increased production through reduced mortality and increased growth, although direct links to increased production are difficult to document.

10.2.1 Increased Catch Rates

In Lake Havasu, Arizona, angler success more than doubled after an extensive program of reef installation (Jacobson and Koch 2008). Studies have demonstrated that habitat enhancement structures concentrate fish (Prince and Maughan 1979; Brown 1986; Moring et al. 1989; Rogers and Bergersen 1999). However, the extent of concentration varies for several reasons, including species-specific characteristics (Hubbs and Eschmeyer 1938; Rodeheffer 1939, 1945), diel fluctuations in fish distribution (Moring and Nicholson 1994), age of the reef (Moring and Nicholson 1994), and the reef's physical attributes. In general, the average number of fish and species attracted increases with the structural complexity, which is achieved by increasing the volume and surface area of the reef, although the benefits are likely asymptotic (Wickham et al. 1973; Rountree 1989). Further, fish abundance, life stage, and species composition vary with structure interstice size (the space within structures) in a complex fashion (Wege and Anderson 1979; Johnson et al. 1988; Lynch and Johnson 1989; Walters et al. 1991).

The morphometric characteristics of reservoirs also influence the effectiveness of attraction. Enhancement structures may be less effective in systems with bathymetrically complex bottoms (Pardue and Nielsen 1979) or with adequate natural habitat (Madejczyk et al. 1998; Rogers and Bergersen 1999). Depth at which habitat enhancement structures are placed controls variables such as temperature, dissolved oxygen, and light availability (section 10.4.2.4) and can influence the effect of the habitat structure (Walters et al. 1991; Johnson and Lynch 1992).

10.2.2 Increased Cover for Spawners and Juveniles

Habitat enhancement structures have been associated with increased recruitment by means of providing cover for nest spawners (Vogele and Rainwater 1975; Hoff 1991; Hunt et al. 2002), thereby increasing nest density. Nest success also may increase if structures provide habitat that allows adults to protect their young more effectively (Hoff 1991). By increasing cover, structures also can offer juveniles refuge from predation (Bohnsack and Sutherland 1985; Johnson et al. 1988; Moring and Nicholson 1994), provide shade that serves as cover (Helfman 1979, 1981; Johnson and Lynch 1992; Raines and Miranda 2016), and provide sites for orientation and schooling (Klima and Wickham 1971; Bohnsack and Sutherland 1985).

10.2.3 Increased Food Availability

Prey abundance in and around the reef may be enhanced (Wege and Anderson 1979; Moring et al. 1989), in turn increasing the feeding efficiency and growth of predators (Wege and Anderson 1979; Bohnsack 1989). Many studies have reported observations of fishes feeding within artificial reefs. The added substrate provided by reefs provides additional periphyton and associated biota (Figure 10.1) (Van Dam et al. 2002), although there is a debate about how much new fish biomass subsequently is produced and whether the added biomass is a significant contribution to a population or assemblage. Attraction of prey and predator fish to the reef facilitates predator–prey interactions. Improved feeding efficiency implies faster growth rates in artificial reefs, but this has not been demonstrated on a general basis.

Figure 10.1. Submersed reefs provide substrate for periphyton and associated biota. Photo credit: A. Norris, Department of Agriculture and Fisheries, Agri-Science, Queensland, Australia.

10.2.4 Increased Production

An underlying rationale for structure deployment is the production hypothesis, i.e., reefs provide additional critical habitat that increases the environmental carrying capacity and eventually the abundance and biomass of fish in the entire system. Thus, barren, unproductive substrate may be transformed into highly productive environments through the addition of artificial reefs (Stone et al. 1979). Mechanisms suggested for this transformation include (1) providing additional food; (2) increasing feeding efficiency; (3) providing shelter from predation; (4) providing recruitment habitat for individuals that would otherwise have been lost; and (5) attracting fish, whereby reefs help vacate space elsewhere, space that is eventually colonized with new biomass (Randall 1963; Ogawa 1973; Stone et al. 1979; Matthews 1985).

Although the production hypothesis has been recognized for a long time, progress toward resolving its propositions has been slow. Attraction and production are not mutually exclusive and can be considered opposite extremes along a gradient. While artificial reefs may merely attract and concentrate some fishes, they may promote the production of others. Most fishes probably lie somewhere between these two extremes. Demonstrating attracting mechanisms does not refute the possibility of increased production. Attraction behavior in fish presumably evolved because of some selective advantage such as faster growth and increased survival, both of which promote production. Nevertheless, a better understanding of the relative importance of attraction and production is critical for wise fisheries management and the effective construction and deployment of artificial reefs.

10.3 Possible Drawbacks

10.3.1 Overfishing

Artificial reefs can be used to increase public access to fish by making it easier for anglers to locate fish and also to increase catch rates by concentrating fish (Bohnsack 1989). Under heavy fishing pressure, structures that attract fish may promote overfishing by increasing fish catchability. Fishes normally dispersed over a wide area would instead be concentrated in a smaller area around reef structures and possibly be depleted more rapidly by fishing. In waters where stocks are relatively low, the addition of structure may improve catch rates but intensify problems associated with overharvest. These concerns may not be applicable if the target species draws primarily anglers that practice catch and release, such as many black bass fisheries.

Additional concerns come from the possibility of removing top-level predators that may concentrate in artificial reefs. Removal of these predators may influence predator–prey dynamics and shift fisheries toward less desirable conditions. Alternatively, overharvest of a population could cause a shift in fishing efforts toward more susceptible species. Ironically, the lowered catch rates caused by overfishing are often cited as the primary reason for creation of artificial reefs (Polovina 1991); in such cases, reefs would be detrimental.

10.3.2 Navigation Hazards

Structures installed in reservoirs to enhance fish habitat can become hazards to commercial navigation or recreational boating. Trees, brush piles, buoys, or other structures can shift positions because of wind and wave action or float to the surface if not properly anchored. Artificial structures also can become a navigation problem in reservoirs where water levels fluctuate, bringing near the surface structures that might have been installed well under the surface of the normal pool elevation.

10.3.3 Leachates

Materials used to construct reefs may produce harmful leachates that create water-quality and aquatic health concerns, either immediately after deployment or as the structures age. Some of these materials include plastics and treated wood.

10.4 Management Practices

Responsibilities related to installing reefs and other structures in reservoirs are not limited to deployment activities. Project managers may need to establish that there is a need for these structures, identify the management goals for installing the structures, select appropriate sites, identify suitable materials, notify permitting agencies, and conduct post-deployment evaluations. This section addresses such considerations and tasks.

10.4.1 Justification and Evaluation

Structures are often added to reservoirs without clear objectives or realistic expectations about benefits (Bolding et al. 2004). Lacking justification, it is difficult to evaluate the effectiveness of a program.

10.4.1.1 Justification of needs

Justification of the program's goals is the foundation for artificial reef projects. A written plan can have various benefits and can be viewed as the cornerstone of the program. Some of the benefits of a written plan include providing continuity to the program regardless of personnel changes; establishing strategies for reaching goals; serving as a handbook for the program; supporting and simplifying the decision-making process; providing a basis for adaptive management; and providing leverage for funding. Moreover, many agencies with jurisdiction over a reservoir may require that the objectives of the structure are defined clearly before any permits are granted.

Artificial reefs are worth considering only after physical and biological surveys of the reservoir have been conducted by trained personnel. These surveys can establish the extent of structure availability and whether the scarcity of structures and bottom reliefs is potentially limiting fishing opportunities, fish population characteristics, or fish assemblage structure. Although a detailed evaluation of structures as a limiting factor may be difficult, or prohibitively expensive, the general availability of structures in a body of water usually can be determined by visual examination of the littoral, particularly during low water. Alternatively, various side-scan sonar devices are available to conduct underwater surveys (Kaeser et al. 2013; TPWD 2016). The side-scan features on newer sounders can capture images of structure in wide swaths on both sides of the boat, and associated software can link these images to map the underwater topography.

10.4.1.2 Clearly stated goals

Goals and objectives are the driving forces behind structure enhancement programs (section 12.2). Examples of goals may include the enhancement of recreational fishing, conservation of fish populations, and restoration of diversity. Structures can enhance recreational fishing by increasing catch rates where exploitation is not high. Conservation of fish populations may be achieved if the structures reduce juvenile mortality or increase growth rates. Structures may increase habitat diversity and at the same time promote species diversity (Kovalenko et al. 2012). These goals are appropriate in reservoirs where lack of structures clearly can be identified as a factor limiting fish assemblages, where this habitat has been lost because of aging processes or anthropogenic disturbances, or where it is necessary to relieve life-history bottlenecks, such as increasing survival of juvenile fishes. In some cases, adding supplementary habitat may be a more efficient method for increasing juvenile recruitment than is stocking. If habitat is limiting, stocking is likely to be ineffective as stocked fish will not have suitable habitat. Inappropriate goals include (1) aiming to increase attraction without proper protection from overharvest, and (2) aiming to increase production of certain fish species when there is no evidence that the scarcity of structure is a limiting factor.

10.4.1.3 Evaluation of performance

It is crucial to evaluate whether the program is achieving the stated goal(s). An evaluation may require monitoring of whether the reef is influencing fishery harvest, enhancing production of selected species, or creating economic benefits. The extent of monitoring and variables monitored is guided by the project objectives, available resources, and apparent knowledge gaps. A preconstruction baseline evaluation may include existing fish assemblage and angler use data and form the basis against which a program's success is measured. Some or all of these data already may be available from routine monitoring in previous years. Preconstruction estimates of economic effects can be extremely powerful in attracting funding for the venture. Postconstruction evaluations may monitor fish assemblages and angler use to evaluate any changes in the fishery, fish population dynamics, or species assemblage composition. A performance evaluation also can detect whether the reef is having any unexpected negative consequences as well as provide insight into the need for future modifications. Without this step, or without a suitable study design, a learning opportunity would have been missed and future reef construction efforts may be wasted.

10.4.2 Reef Site Selection

10.4.2.1 Incompatible sites

Conflicting uses need to be evaluated when planning where to deploy structures. Possible conflicts include presence of power lines; oil, gas, or sewer pipelines; alternative energy projects; restricted areas for civilian or military activities and other rights of way; and shoreline real estate developments. Exclusion of known commercial fishing areas and navigation lanes also need to be considered. If the purpose is to provide recreational fishing opportunities, the reef might not be used to its fullest potential if fishers have to travel a long way to get to it or environmental conditions at the site (e.g., wind, wave action) are such that preclude a pleasant experience. Alternatively, if the structures are established to enhance spawning and recruitment, distance from access sites may not be a concern (unless cost of deployment is high) or sites may need to be placed far from popular fishing areas. Attraction of fish to an extent that attraction interferes with access to natural spawning habitats such as tributaries, floodplains, and adjacent wetlands may need to be prevented. Last, reefs may need to maintain a reasonable distance from the dam and associated water intakes, discharges, or energy-production facilities.

10.4.2.2 Compatible sites

Compatibility depends on the goal of the reef. For example, habitat enhancement may target a section of a reservoir where littoral fish habitat is composed primarily of barren mudflats, with the goal of increasing fish densities in these areas. Installing reefs would attract and potentially retain fish that would otherwise not stay long and move on to other areas with more desirable habitats. The reef provides the fish with a domain that includes the reef and the surrounding waterscape. The reef serves as a place to rest in cover, wait in ambush, or feed on periphyton or other small prey or from where to launch feeding or spawning forays into neighboring areas. Without the reef some of these basic life-history necessities may not be available, and therefore the fish would not remain in the area for a prolonged period of time. Thus, in this case the reef actually may create new biomass and enhance production. At the same time, anglers may benefit by having new areas to fish and an improved chance at catching fish in an area that in the past had been mostly unrewarding.

Alternatively, reefs may be installed in areas known to attract large numbers of fish with the goal of attracting even more fish, retaining fish at the site for longer, or both. Areas such as submerged rocky ledges, points, creek beds, and roads often tend to attract fish. These preferred fish staging areas potentially can get a boost when a reef is installed.

If the goal is to encourage shoreline-based angling, then it may be beneficial to locate some reef habitat under or adjacent to fishing piers, close to them, or within casting distance of shoreline areas that can be accessible to shore-based anglers.

10.4.2.3 Substrate

The characteristics of the sediment need to be known or evaluated prior to deployment of the structure. Appropriate substrate conditions are required to prevent the reef or spawning bed from sinking beneath the sediment surface. Inappropriate placement can cause significant or total loss of exposed material and greatly reduce the utility of the structure. Although some settling of deployed material is expected in unconsolidated sediment, and can actually assist stabilizing a reef, conditions of the reservoir floor have to support deployed materials sufficiently to allow long-term success of the structure.

In general, areas with soft sediment such as soft clay, fine silt, or loosely packed sand may need to be avoided because they increase the likelihood of the structure sinking or subsiding. These areas are more common in the headwaters of the reservoir where major tributaries deposit their sediment loads but may also occur in major embayments where smaller tributaries enter the reservoir (section 3). If sedimentation is a major issue in the headwaters, reefs quickly may become covered with sediment; instead they could be installed in areas farther downlake where sedimentation may be less severe. In contrast, deployment on a hard substrate may increase the structure's susceptibility to slide during storm events. The ideal substrate for structure placement would be a thin layer of soft sediment over a harder layer of soil.

10.4.2.4 Depth

Reefs can create navigational hazards. The depth of the reef needs to be sufficient to allow for safe navigation over the reef. The required clearance (i.e., minimal water depth above the reef) depends on the location and anticipated type of traffic that would traverse the area. Generally, the depth clearance of a permitted artificial reef should not exceed the shallowest depth of surrounding natural features, and the top may be installed at least 3 ft below the water surface at mean annual low water. Installation in remote coves of the reservoir usually limits navigation risks. On a case-by-case basis, specific buoys or other markers may be needed to designate the reef area.

The depth of the structure is also a consideration based on the goal of the structure. For example, if the goal is to provide habitat for a given species, the preferred depth range of that species can be factored into where the structure is placed. Physicochemical variables may influence use of structures, particularly in deeper water where temperature, dissolved oxygen concentration, and light intensity are reduced. Deeper

Figure 10.2. Brush (upper panel) and stone (lower panel) linear reefs designed to cope with fluctuating water levels. Photo credit: North Carolina Wildlife Resources Commission, Raleigh, and Missouri Department of Wildlife Conservation, Columbia.

reefs may have less periphyton and associated invertebrate communities. Water temperatures below about 10°C can cause centrarchids to leave structures in shallow littoral areas (Prince and Maughan 1979). Similarly, structures placed in cold, deeper water in summer may not attract many species. Prince et al. (1985) reported that some reservoir reefs at depths of 20 ft or less were virtually devoid of fish in the winter. For optimal value, the structure is placed above the summertime thermocline, particularly if the hypolimnion goes anoxic (section 6). High light levels often result in increased fish use of structure that produces shade (Helfman 1979). Fish may find diversity in light levels by using deeper structures, provided that temperature and oxygen needs are met in deeper water.

Siting reefs in reservoirs with large water-level fluctuations can be problematic. Ideally, reefs are installed below the lowest pool elevation to avoid endangering recreational boaters. Nevertheless, at such low water levels the structure may not be available to most fish during periods of high water, which is usually a time of year when fish are most active. Where boating traffic is not high, speeds are low, and reefs can be marked, linear reefs may be appropriate. A linear reef may run from low to high water perpendicular or at an angle to shore (Figure 10.2). Such an arrangement is possible with any reef material (section 10.4.3). A linear reef provides access to cover at multiple water levels and enables fish to select a preferred depth during high water, which is often in summer when water-quality conditions differ most over depths. The wetting and drying cycles in these linear reefs are likely to deteriorate some reef materials more quickly. A marker buoy (section 10.4.8) can be positioned at the deepwater end of the reef to alert boaters, and a sign at the shallow end would tip off anglers as to how the reef is positioned so they can fish it effectively (Figure 10.3).

Figure 10.3. Fish attractor sign posted on a tree near the shoreline. Photo credit: Arkansas Game and Fish Commission, Little Rock.

10.4.2.5 Floating reefs

Floating reefs can provide structure that will rise and fall with the water level, eliminating the problem of the structures being too deep, too shallow, or out of the water for large parts of the year (Brouha and von Geldern 1979). When adequately buoyed and securely anchored, floating reefs offer year-long utility as well as possibly providing wave attenuation between the structure and the reservoir shoreline. Floating reefs may be combined with other existing floating structures (e.g., fishing piers, breakwaters, floating docks, buoys) or they may be constructed independently. However, floating reefs pose a risk to navigation. Their use may need to be limited to areas of the reservoir where (1) boating is primarily for fishing and (2) boating speeds are generally lower. Reduced speed limits could be put in place in areas with floating fish attractors to reduce collision risks further.

10.4.2.6 Waves

Areas of consistently high wave energy may not be suitable for reef installations. High wave energy will decrease the durability and stability of a reef because of constant exposure to wave surge. The wave energy also may limit the settlement potential of periphyton if water is too turbulent and may exclude some fish species that would otherwise be attracted. High wave energy zones often are close to shore. Placing structures in these areas also may affect longshore sand transport, which may not be desirable and may need to be evaluated thoroughly. Analyses of seasonal weather patterns and wave fetch can be used to assist in the selection of most appropriate sites.

10.4.3 Reef Design

The morphology and complexity of habitat can be one of the more important factors influencing the effectiveness of structure as fish habitat. Relationships between habitat and species associations may best be summarized by the habitat diversity hypothesis, which states that species diversity increases with increasing availability of habitat types (Kovalenko et al. 2012). Fish abundance, richness, and diversity along shorelines of reservoirs are generally directly related to the structural complexity of available habitats (Barwick 2004; Newbrey et al. 2005). Higher species diversity can result in more complex food webs and longer trophic loops. Increased complexity of food webs creates a more stable fish assemblage that is less susceptible to chaotic dynamics and is made up of resilient interspecific interactions.

Interstitial space affects the size of fish attracted. Prey fish prefer small- or medium-size interstices when in the presence of predators (Crook and Robertson 1999). Early life stages of bluegill occupied small interstices within habitat, as close to body size as possible (Johnson et al. 1988), reducing danger from predation. However, even at younger life stages, largemouth bass were more likely to choose medium interstices rather than small ones. Potentially, this choice provides a balance between avoiding predation and having sufficient opportunity to ambush prey. Thus, diversity of interstitial space may best promote diverse fish assemblages within artificial reefs.

Besides complexity, the effect of structure on populations may depend on size of the reef. Rountree (1989) reported that fish abundance and diversity were related positively to structure volume and surface area. Bohnsack et al. (1994) reported that large reefs may have higher biomass densities than do small reefs but are oftentimes composed of fewer but larger individuals. Daugherty et al. (2014) reported that largemouth bass exhibited greater percentage occupancy in large structures but higher densities in small structures. Thus, depending on size, some artificial reefs may support fewer and larger fish, whereas others may support more and smaller individuals.

Reef arrangement may also be important. The responses of largemouth bass and bluegills to reef arrangement in a Texas reservoir suggested that cluster-shaped reefs (roughly circular to minimize the amount of edge and maximize the amount of interior cover) provided greater protection from predation than did a linear design (trees organized in a line to maximize amount of edge; Daugherty et al. 2014). Greater protection from predation likely is related to the increased interior space provided by the clustered design, which reduces visual encounters with predators and excludes predators from the reef interior. The percent of reefs occupied and catch rate of bluegills was highest in cluster-shaped structures, but size of bluegills was smaller, suggesting that the clustered design was used as protective cover (Daugherty et al. 2014). In Ruth Reservoir, California, Bryant (1992) organized brush into three arrangements (Figure 10.4). The discrete open-center structure was the most used by juvenile and adult largemouth bass and smallmouth bass. However, both the continuous open-center and dense design structures were used by largemouth bass and smallmouth bass more than shoreline areas with no woody structures.

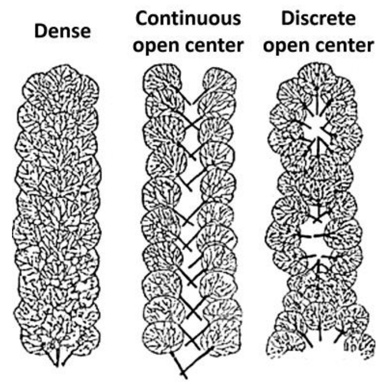


Figure 10.4. Different reef arrangement designs often have different attraction rates and attract different species. Figure reproduced from Bryant (1992).

Size, morphology, and complexity of reefs influence the species and life stages attracted. Thus, as a general strategy to benefit as much of the fish assemblage as possible, a broad diversity of reef sizes, morphologies, and complexities may optimize the value of an artificial reef program. In some cases, it may be much easier to replicate a single standard design. Nevertheless, size, morphology, and complexity of the reefs may need to be linked to the goal(s) of the reef program.

10.4.4 Reef Area

The amount of structure needed depends on the goals of the program. If the goal is fish attraction to improve catch rates in an underutilized fishery, this can be accomplished with a limited number of reefs distributed strategically near fishing areas or in areas targeted by managers for increased effort. Alternatively, if the goal is to alter population dynamics noticeably by possibly shifting rates of recruitment, growth, and mortality, and even maybe change community structure through large changes in habitat composition, then the amount of structure needed may be extensive and possibly unrealistic. A definition of "extensive" has been researched by a few investigators, but the research has not been sufficient to pin it down.

Crowder and Cooper (1979) offered a conceptual model of predator–prey interactions relative to structural complexity. They suggest that prey-capture rates per prey available decline with increasing structural complexity. However, prey density (diversity and abundance) is positively correlated with structural complexity. These counter currents can lead to maximal feeding rates at intermediate structure levels. Thus, at low levels of structure availability, fish can feed most efficiently, but few reefs are available, and thus overall utility of reefs is relatively low. At high levels of structure availability, despite relatively high prey densities, prey capture rates are low because of reduced feeding efficiency. Thus, high levels of structure availability are also a low-utility habitat. At intermediate structure levels, feeding efficiency is optimized as prey is relatively more available than in either high or low levels of structure availability. These authors also hypothesized that the actual level of structure that maximizes feeding rate is a function of fish size. A large predator would reach maximum feeding rates at a lower level of structure than would a smaller predator.

Field surveys linking aquatic vegetation density to characteristics of largemouth bass populations have suggested that densities in the 20%–50% range optimize some population characteristics (Durocher et al. 1984; Wiley et al. 1984; Miranda and Pugh 1997). Aquatic plants may have greater effects on water quality (Miranda and Hodges 2000) than do artificial reefs, but this target range could be used as a guideline for coverages of artificial structure needed. Clearly, 20%–50% coverages would be practicable in only small reservoirs or embayments of large reservoirs. In large reservoirs these levels of cover may be attempted through introduced structure combined with plant growth (section 11).

10.4.5 Permits and Regulations

The permitting process, if any, for installing artificial reefs and other structures varies on a lake-by-lake basis depending on the agency that manages the facility. It is the responsibility of the organization installing the structures to obtain the necessary permits. For permitting, some agencies may require information such as location of the structure, including latitude and longitude; purpose and need for the structure; description of type, quantity, and composition of material to be placed in the water; and provisions for installation, monitoring, and managing the life of the structure.

Depending on the scope of the project, a permit may be needed from the agency managing water storage. For example, the U.S. Army Corps of Engineers (USACE) holds authority under Section 10 of the Rivers and Harbors Act and under Section 404 of the Clean Water Act to regulate structures and placement of materials into the waters of the USA (the U.S. Environmental Protection Agency [USEPA] has delegated the "404 process" to the USACE). Most USACE districts do not require special permits for adding small-structure projects in reservoirs, but it is always good to check with the local office with jurisdiction over the reservoir.

The USACE may coordinate with other federal agencies such as USEPA and U.S. Fish and Wildlife Service through the public notice process. For example, endangered species surveys may be requested from the reviewing agencies in order to note the presence and prevent damage or destruction to hard-bottom or endangered species. Surveys also may be requested to identify historical sites or artefacts to avoid. Applicable authorizations include the National Environmental Policy Act, which provides a mandate and framework for federal agencies to consider all reasonably foreseeable environmental effects of proposed actions and to involve and inform the public in the decision-making process by considering environmental effects and reasonable alternatives. It requires federal agencies to conduct an Environmental Assessment or Environmental Impact Statement for each project. The National Historic Preservation Act provides for evaluation of direct and indirect effects of the project on historic resources in the area. The Endangered Species Act provides a consultation requirement for any federal action (e.g., USACE issuing a permit) that may affect a listed species to minimize the effects of the action.

10.4.6 Reef Construction

A diversity of reefs and spawning structures has been constructed and installed in reservoirs. These can be classified into three general types: (1) tree, brush, and lumber structures; (2) structures constructed from stone materials; and (3) structures constructed from synthetic materials such as plastics. Reefs concentrate fish to increase angling catch rates by anglers, promote predator–prey interactions, or provide cover for various species or life stages. Spawning structures provide spawning substrate for various species whose spawning habitat is limited in reservoir environments or has been degraded by long-term environmental changes.

10.4.6.1 Tree, brush, and lumber reefs

Wood piles provide excellent habitat for a broad range of species and life stages. Wood piles include brush, shrubs, tree limbs, and entire trees (Figure 10.5). Common sources of brush have been conifers and hardwoods. Conifers tend to have smaller interstitial spaces but decompose quicker than hardwoods; stumps may last for decades. There are many advantages to using brush piles and woody debris, beginning with the wide range of interstitial spaces that provides diverse microhabitats for fish of various sizes. Perhaps the major allure of brush and woody debris is their naturalness and that, in time, they biodegrade and leave no trace. Brush jams form naturally in streams, so many fish species instinctively are attracted to brush piles and often colonize them quickly after installation. Brush and other tree remnants remain the most frequently used materials to build reefs because in most geographical areas they are abundant, inexpensive, are relatively easy to install with volunteers, and closely resemble the natural log and brush jams and root wads experienced by fish in their native environments.

Brush can be organized in various configurations to conform to the local waterscape or to optimize perceived fish habitat requirements. The size of the brush structure may depend on how many individual units are included in a cluster and how units are placed in relation to each other. Diversity of size, shape, proximity to each other, and depth is probably best to meet the various needs of diverse species and fish sizes. A range of depth contours may be targeted if water-level fluctuations are a concern (section 10.4.2.4). A quick way to introduce brush into shoreline habitats is by felling trees along the shoreline (sections 5.8.2.12 and 8.9.5).

Brush is often available locally, precluding the expense and logistical problems associated with long-distance transportation. Installing brush and woody debris lends itself well to recycling resources that otherwise would go to waste and to including volunteer groups in the gathering and installation processes (section 10.4.11). If the brush pile has to be sunk, "green" recently cut brush sinks easier. Brush that has been stockpiled for 1–2 weeks loses weight through desiccation and requires more weights to sink and secure. Depending on source and size, brush piles and woody debris may be longer lasting than Christmas trees (Bolding et al. 2004). Conifers lose much of their interstitial space (i.e., small branches) within 2–7 years, although the thicker branches remain. Most hardwoods lose their interstitial space in 10–12 years, but again thicker branches remain. Large trees and stumps may last 15–25 years or longer, but large trees may be hard to install (section 10.4.7). As interstitial space is lost to decay, or if larger trees or stumps are installed, the fish assemblage occupying the reef probably shifts toward larger individuals. Because of the hodgepodge nature of brush reefs, they need to be tied together and anchored to the bottom (section 10.4.7).

Reefs also may be constructed from commercially available lumber, scrap lumber left over at sawmills, or repurposed lumber such as wooden pallets (Figure 10.6). Reefs made with lumber can be costlier, but unlike brush piles they can be assembled off-site. Treated wood increases durability of reefs constructed with lumber. Commercially treated wood most commonly is preserved with creosote or copper products. Creosote is a distillate of coal tar and is a variable mixture of 200–250 compounds, with over a dozen inventoried in the USEPA's List of Priority Pollutants (NOAA 2009). Exposure to creosote reportedly produces reproductive anomalies and immune dysfunction and impairs growth and development in fish exposed to sufficiently high concentrations over long periods of time (reviewed by NOAA 2009). Treatment with copper includes zinc, chromium, and arsenic, but copper is the focal point because it leaches from treated wood at rates that can affect aquatic resources. However, use of treated wood products is unlikely to cause detectable effects in aquatic environments unless used in excessively large quantities (NOAA 2009). To minimize risk, copper-treated wood is preferred over creosote-treated wood, and maximum prefabrication can be done before the structure is placed in the water to lessen the releases of treated debris associated with cutting and drilling (NOAA 2009).

Various studies have evaluated aspects of constructing and installing brush reefs (reviewed by Bassett 1994; Bolding et al. 2004). Results have differed depending on brush type, reef size, configuration, depth, local conditions, and many other variables. These studies suggest that brush reefs benefit fish in multiple ways, and providing diverse brush reefs is likely to optimize benefits.

Half-logs or spawning benches reportedly attract various substrate-spawning species. These structures provide overhead cover in areas that already have favorable spawning substrate for nest spawners such as centrarchids and percids (Bassett 1994). The structures consist of a log sawed longitudinally in half, or hardwood slabs, fastened with the flat surface down on two to three concrete blocks. These structures are 8–12 ft long, 10–15 in wide, and 5–10 in thick. Half-logs generally are placed on firm substrate, preferably gravel, at depths of 3–10 ft.

10.4.6.2 Stone materials

Stone materials including boulders, riprap, and gravel are natural, durable, and familiar to many species inhabiting reservoir environments. These materials may exist already in tributaries and within the reservoir basin but may need to be supplemented because the reservoir might have submerged them at depths where they are no longer available to many species or might have been blanketed by sediment. Rocks may be placed singly (if large boulders), in piles to form patches, in long reefs along shore, in wood cribs, or in other containment structures (Grove et al. 1991; Kelch et al. 1999; Houser 2007; Figure 10.7). Riprap is commonly applied on reservoir shorelines

Figure 10.5. Examples of tree and brush reefs. Photo credits: (Left column, top to bottom) U.S. Army Corps of Engineers (top two), Aquatic Environmental Services, and *Billings Gazette*; (Right column, top to bottom) Missouri Department of Conservation (top two), *Denver Post*, and Virginia Department of Game and Inland Fisheries.

Figure 10.6. Examples of lumber reefs. Photo credits: (Left column, top to bottom) Pond Boss Magazine, Long Lake Fishing Club, Kids And Mentors Outdoors Northwood Chapter, and Pennsylvania Fish and Boat Commission; (Right column, top to bottom) U.S. Army Corps of Engineers (top two), Kentucky Department of Fish and Wildlife Resources, and Pennsylvania Fish and Boat Commission.

to control erosion and wave action (section 5.8.2) but can also be applied with the explicit purpose of providing fish habitat. Riprap additions also may be applicable in key areas where fish may concentrate for spawning, such as the mouth of tributaries. A mixture of stone sizes may work better than size-graded stones because this creates a diversity of interstitial voids among the stones that fish can exploit.

Another source of stone materials may be construction supplies and demolitions. This source may include concrete blocks, rock, brick rubble, fractured concrete and slabs, and ceramic and concrete pipes. Concrete blocks provide intermediate-size interstitial space, and after colonization by algae and sessile organisms they appear natural (Moring and Nicholson 1994). Large concrete slabs produce large interstitial spaces that tend to attract large predators. Variable rock sizes within a structure create greater habitat diversity through more diverse interstitial spaces. Oftentimes stone construction material can be installed in the form of jetties, serving as fish habitat and points of access to shoreline anglers. Hernandez et al. (2001) found that rock jetties act as a refuge area for larval and juvenile fish.

Gravel beds attract substrate spawners and likely improve spawning success (Irwin et al. 1997). Gravel often can be distributed along shoreline areas with a habitat barge or in smaller operations from a 4 × 8-ft plywood sheet mounted on the bow of a flat-bottom boat. Shores with hard clay substrate are preferred or gravel will sink or be covered with sediment rather quickly. Gravel attracts spawning centrarchids and other species and makes for good areas to fish. Gravel substrate also can be introduced in spawning boxes. These are square boxes approximately 3 ft wide by 1 ft deep. The top of the box is open, and the bottom consists of 0.5–0.75-in hardware cloth reinforced with wood braces. The box is filled about halfway with 1–3-in gravel and placed in 3–10 ft of water on soft substrate. Centrarchids and other substrate spawners nest inside the box. Multiple boxes may be installed near brush or other installed reefs to increase the density of adults spawning within a concentrated area.

10.4.6.3 Synthetic materials

Plastics including polyethylene and polyvinyl chloride (PVC) have been used when constructing artificial reefs. Plastics can be convenient because they are lightweight and easy to work with and handle. Some can be extremely durable, are inexpensive and readily available, and, unlike brush, some will not snag an angler's lure. Various vendors may be found by searching online for "artificial fish habitat."

Various plastics are available to construct reefs (Figure 10.8). Relatively inexpensive plastic crates, and plastic netting for snow and safety fencing, can be used as building blocks to assemble reefs. Although convenient to obtain and simple to assemble, the strength of reefs built from plastic crates would need to be ensured. Snow fencing has been used in various designs where it is wrapped around a frame; however, when it becomes untied from its frame it can float and become a navigational hazard or a hazard to wildlife. Pipes of PVC are readily available at hardware stores and are safe to use in water as they are used for residential plumbing. Their round shape will not snag most lures. Corrugated pipe is also readily available and can be easily shaped into diverse forms. Some of these pipes also can be filled with sand, gravel, or concrete to weigh them to the bottom, and some may sink if drilled with holes to allow water to infiltrate the structure. The larger-diameter pipes also may harbor small fish inside the pipe. In summary, these products are relatively cheap, readily available, and can be used safely to make simple, reproducible designs by people with limited construction skills.

A variety of plastic artificial structures are manufactured commercially. They vary widely in size, design, cost, and materials. Common designs emulate aquatic plant material. Structures made of synthetic materials are advantageous because they have greater longevity than small-diameter brush, are light, are easy to transport and assemble on site for quick installation, and do not require special equipment for assembly. However, they do have disadvantages. Four issues that have been identified are (1) some lack complex structure and small interstitial spaces; (2) they can be expensive when compared with brush; (3) they have been found to attract fewer fish (Rold et al. 1996; Magnelia et al. 2008); and (4) they have had low satisfaction rates among state fisheries agencies when the objective is to increase angler catch rates (Tugend et al. 2002).

Structures made of synthetic materials seem to be less efficient than brush at attracting fish. However, they have been shown to concentrate largemouth bass successfully, with some designs having better attracting qualities than others (Rogers and Bergersen 1999). When natural materials are not readily available, as for example in reservoirs constructed in desert regions, reefs fabricated from commercial synthetic structures may be a good option.

Plastics can break down over time and be hazardous to fish (Rochman et al. 2013). Risks come from the material itself and from chemical pollutants that absorb into the materials. Some of these compounds are added during plastics manufacture, whereas others adsorb from the aquatic environment (Eerkes-Medrano et al. 2015). Polyethylene accumulates more organic contaminants than do other plastics such as polypropylene and PVC. Overall, the hazards associated with the complex mixture of plastic and accumulated pollutants are largely unknown (Free et al. 2014; Driedger et al. 2015).

Another limitation of plastic reefs is that they are not detected by some of the more inexpensive sonar devices available to recreational anglers. Some devices may

Figure 10.7. Examples of reefs constructed from stone materials. Photo credits: (Left column, top to bottom) FISHBIO, Missouri Department of Conservation, Texas Parks and Wildlife Department, and Nebraska Game and Parks Commission; (Right column, top to bottom) Pennsylvania Fish and Boat Commission (top two), Georgia Department of Natural Resources, and Pond Boss Magazine.

Figure 10.8. Examples of reefs constructed from synthetic materials. Photo credits: (Left column, top to bottom) U.S. Bureau of Land Management, Georgia Department of Natural Resources, Arkansas Game and Fish Commission, and Mossback Fish Habitat; (Right column, top to bottom) Pond Boss Magazine, Fishiding Reclaimed Artificial Fish Habitat, Porcupine Fish Attractors, and Pond King Inc.

detect them but not very clearly. Thus, plastic structures more than brush or stone may require identification with buoys for maximum benefit to recreational anglers. Alternatively, if the goal of the structure is something other than to concentrate fish to promote angler catch, unmarked, hard to find, plastic structures may be a good option.

10.4.7 Reef Installation

Artificial reefs can be installed from boats, land if water levels fluctuate, the air if helicopters are available, or ice if the reservoir surface freezes substantially. Small brush piles and synthetic structures easily can be deployed from agency work boats or volunteers' boats. Brush piles can be staged on a shore near the destination site or boat ramp, expediting loading and accelerating deployment. Large reefs may require a specially designed and equipped habitat barge (Figures 10.5, 10.7). Such barges typically are equipped with a winch, dump bed, or both to facilitate deployment of large or heavy wooden structures as well as stone materials. In addition to being able to install larger structures, barges can facilitate more rapid installations. Habitat barges can cost \$50,000–75,000. However, some large trees may be hard to install even with a habitat barge. Large trees may have to be towed by the habitat barge or installed when the water level is low enough to access the site with heavy machinery.

In reservoirs where water levels fluctuate annually or over multiyear cycles, reefs may be built on-site or dragged to the site with all-terrain vehicles. Deployment during low water level allows for a faster, safer, and more precise installation. At latitudes where the reservoir surface freezes sufficiently to support vehicles, reefs may be constructed on the ice or off the ice and dragged to the desired position. Once positioned, the ice may be cut, or the reef left on the ice until the ice melts and the wooden or stone reef sinks into position. Occasionally agencies may have access to helicopters to hoist and transport reefs to a desired location (Figure 10.5).

Brush reefs and synthetic reefs generally need to be tied and permanently anchored to the bottom in such a way to prevent floating to the surface, floating of any of its member pieces, and excessive chafing of its lashings or anchor lines. Tying and anchoring can be accomplished with a combination of galvanized wire, polypropylene rope, nails, rods through drilled holes, stakes, concrete blocks, rock-filled mesh bags, sand bags, or other materials of suitable weight. In some synthetic reefs, such as those made from PVC or corrugated pipes, gravel may be incorporated into the pipes to ensure the structures remain on the bottom at the location intended. Heavy-weight cable ties are used (150-lb breaking strength or greater).

Strict safety guidelines are required to avoid overloading boats or snagging the reef on personnel or equipment as the reef is being deployed. Providing global positioning system (GPS) waypoints or installing markers prior to reef deployment or on-site construction is desirable to ensure the reef is sited at the exact location planned.

10.4.8 Reef Markers

Figure 10.9. A marking buoy used to identify a reef. Photo credit: U.S. Army Corps of Engineers, Nashville District.

In navigable waters, artificial reefs need to be marked clearly with permanent buoys, as required by the U.S. Coast Guard (Figure 10.9). Nevertheless, even in non-navigable waters, reefs that are near the surface may pose potential navigational hazards to recreational boaters and may require buoys. Markers are also useful to inform fishers where the attractors are located. The buoy shape, color, or both are usually different from buoys with navigational significance. Buoys that are colored plastic all the way through are generally best as opposed to painted buoys because the paint will degrade from sun exposure and wave action. Nevertheless, the authority with jurisdiction over the reservoir may need to be consulted about the need for markers and the required marking and color system. Moreover, in some states if a boat strikes a buoy the possibility for litigation may be high, so consultation with agency counsel may be necessary. If markers are a risk, signage for reefs may be installed on shore in areas next to the reef.

Markers also can be placed to serve as moorings for recreational fishers to avoid anchor damage to the artificial reef. Mooring buoys typically have a metal ring protruding from the top that can be used as a tie-off point for boats. Mooring buoys are ready-made, commercially available spherical structures (about 18-in diameter) of polyethylene plastic. They are usually filled with polyurethane foam and treated with ultraviolet inhibitors so as to endure strain after continual exposure to sunlight.

Expenses include not only the buoys and anchors but also maintenance. Occasionally, a stray buoy has to be retrieved and redeployed and may create navigational problems. Anglers also may try to move buoys for various reasons. Buoy monitoring and maintenance programs can be oriented toward preventing failures in the field. Ultimately the use of buoys or other markers is at the discretion of the program manager, unless markers explicitly are required through the permitting process.

10.4.9 Maintenance and Monitoring

Long-term stability and durability is an important consideration in siting an artificial reef and in selecting materials. The shallower the water on a high-energy shoreline, and the greater the water-level fluctuation, the more severe the physical conditions a shallow-water structure will experience. This severity may lessen the durability of the structure and may compel more frequent maintenance. Most wood materials are less stable and durable at shallower depths where they are more exposed to solar radiation, changes in weather conditions, wave action, and exposed to the atmosphere during low water levels. Reefs are designed to survive prevailing physical conditions to prevent large parts from breaking free or compromising overall structural integrity. Ideally, materials used need to be resistant to degradation due to air exposure and the chemical forces of the aquatic environment. Most commonly, some reefs degrade and collapse and may not provide the original benefits but continue to provide some habitat benefits.

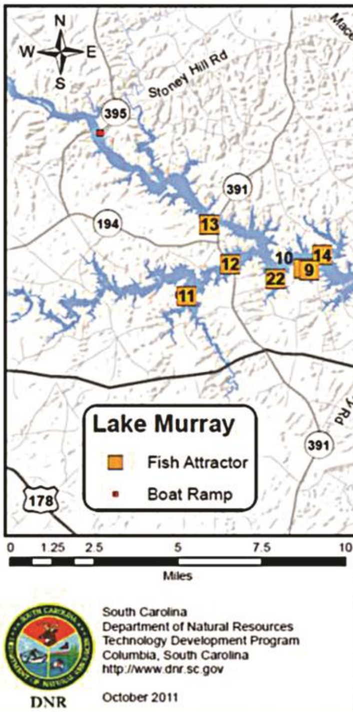
The longevity of reefs will vary depending on multiple variables associated with architecture, construction materials, workmanship, siting within the reservoir, and local environmental and climatic conditions. Nevertheless, all reefs tend to deteriorate over time, and their effectiveness may change linearly, although some more rapidly than others. Thus, all reefs need to be monitored periodically to determine whether maintenance is needed to preserve or upgrade their effectiveness and safety or whether they need to be removed. Reefs built from natural materials such as wood and stone generally may remain in the reservoir as they degrade. However, reefs made from synthetic materials may need to be removed once their useful life has been exceeded.

Monitoring reefs may include a general evaluation of several aspects associated with the reef's attraction or production capacity or whether the reef is still safe (e.g., has not degraded to the extent that it may affect navigation). Monitoring may be as simple as visual inspections during low water or more demanding inspections with divers or side-scan sonar technology. Annual surveys with side-scan sonar (TPWD 2016) may be sufficient to detect reef degradation.

Scoring of reef status may be achieved with a qualitative scale (e.g., functional, moderately functional, nonfunctional) or a semiquantitative scale (e.g., 0 to 10). A subjectively selected threshold may be used to initiate reef restoration or removal. In many cases, volunteers may be enlisted and trained to perform inspections (Halusky et al. 1994).

10.4.10 Brochures and Electronic Media

Budgets and available resources dictate whether printed maps may be made available. A simple brochure can be created using any word-processing software and reproduced on the office copier. In fact, this avenue may be preferable to an expensive, quickly out-of-date, full-color brochure. Plain or extravagant, the main objective is to



SITE	Latitude	Longitude	SITE	Latitude	Longitude	SITE	Latitude	Longitude
1	34.063711	-81.22274	9	34.078799	-81.521174	17	34.080656	-81.393477
2	34.029074	-81.293718	10	34.079559	-81.524014	18	34.088716	-81.405866
3	34.028014	-81.320327	11	34.066831	-81.584923	19	34.129726	-81.434806
4	34.016825	-81.339007	12	34.081289	-81.562326	20	34.076054	-81.314928
5	34.025996	-81.383486	13	34.09923	-81.573223	21	34.085635	-81.320779
6	34.044985	-81.360687	14	34.085679	-81.514444	22	34.074606	-81.538791
7	34.056015	-81.374647	15	34.090696	-81.415396	23	34.04771	-81.429605
8	34.069608	-81.485204	16	34.082006	-81.406166	24	34.113368	-81.346495

Figure 10.10. An example of how information about reefs may be made available online in agency websites, phone apps, and social media. Source: South Carolina Department of Natural Resources, Columbia.

satisfy the public need for basic data on distribution of artificial reefs. Fishers mainly want to know the reef location and possibly the configuration and depth. A list of GPS coordinates will do, but a brochure can be greatly enhanced with a chart showing site descriptors. The information may also be made available online in agency websites, phone apps, and social media (Figure 10.10).

This informational tool can increase the success rate of anglers who may not be familiar with a reservoir, like first-time anglers or anglers from outside the region. These anglers can use the information to find the areas where fish habitat improvements have been installed and have a more pleasant experience with the confidence they are fishing at a tactical location.

10.4.11 Volunteer Assistance

Volunteers can expand greatly the manager's ability to develop a reef program (Jacobson and Koch 2008). However, one significant hurdle in the use of volunteers is the availability of agency personnel to provide training and oversight of activities. Because volunteers are generally available only on weekends, the manager or coworkers need to also be available on weekends. This type of management is time intensive, and the manager needs to consider the full scope of the task before committing to relying on volunteer labor. The manager provides sufficient training to the volunteers and plans and organizes the activities. Most volunteers are well intentioned, but they have other demands on their time and may not be available to complete the activity, so turnover may be another challenge. Wilson et al. (1996) worked extensively with volunteers in developing habitat in Norris Lake, Tennessee, and provided useful advice for working with volunteers (Table 10.1).

Table 10.1. Suggestions for working with volunteers in habitat enhancement projects. Modified after Wilson et al. (1996).

Keys to successful volunteer program
• Design projects that address real problems and that have a high probability of success. Projects should include activities that volunteers believe are worth their efforts.
• Make sure that a project leader is present at all activities to answer questions and be an example.
• Identify and support a leader among the volunteers who will take responsibility for recruiting and notifying other volunteers to fulfil work schedules.
• Have all equipment and materials present at the site on time and in good working order. Start on time. The volunteers are there to work and will not want to waste time waiting.
• Give clear instructions as to what will be done, how long everyone will work, the importance of safety, and agency policies.
• Give the volunteers as much responsibility as allowable (e.g., driving boats, backing trailers, operating posthole diggers).
• Make the project fun but have goals that require hard work. At the end of the day, everyone should feel that a significant amount of work was accomplished.
• Be sensitive to when it is time to quit for the day. Avoid the tendency to work volunteers a little longer than was agreed.
• Be sensitive to varying abilities and health of the volunteers and their tendency to work beyond their limits.
• Let volunteers know they are appreciated. Keep an accurate account of time they work. Throughout the project, express gratitude for their efforts. After the project is completed, reward them with appropriate recognition and awards (e.g., a banquet, caps, personal letters).
• Invite the media to feature volunteers at work. Encourage the media to emphasize the role of volunteers in the project.
• Keep in touch with volunteers between activities. Let them know the results of other activities undertaken for the project.
• Always speak positively about projects, agencies, and volunteers. Do not let projects become a forum for negativism toward user groups and agencies. Rather, let them be opportunities for building understanding and respect among the diverse user and management groups.

Section 11

Aquatic and Terrestrial Plants

11.1 Introduction

Plants are an important part of healthy, diverse aquatic ecosystems. Specific roles of aquatic plants and terrestrial plants that colonize reservoirs include producing and consuming of oxygen, stabilizing temperature and light, recycling nutrients, controlling turbidity, and providing food, spawning substrate, and habitat for invertebrates and fish. Plants also protect shorelines from erosion, and plant roots stabilize lake-bottom sediment to protect it from the stirring effect of wave action. Additionally, plants are valued for their aesthetic qualities and help provide a more "natural" buffer between the riparian zone and the open water.

Aquatic macrophyte abundance in reservoirs often exhibits two contrasting problems: too many or too few macrophytes. A survey of 1,299 reservoirs ≥250 ac in the USA identified that excessive macrophytes was a concern in nearly 10% of the reservoirs surveyed, and not having enough macrophytes was a concern in over 25% of the reservoirs (Krogman and Miranda 2016). These percentages varied regionally, with excessive macrophytes afflicting nearly 25% of reservoirs in the Coastal Plains ecoregion, and not enough macrophytes troubling over 40% of reservoirs in the Southern Plains and Temperate Plains ecoregions (Figure 11.1).

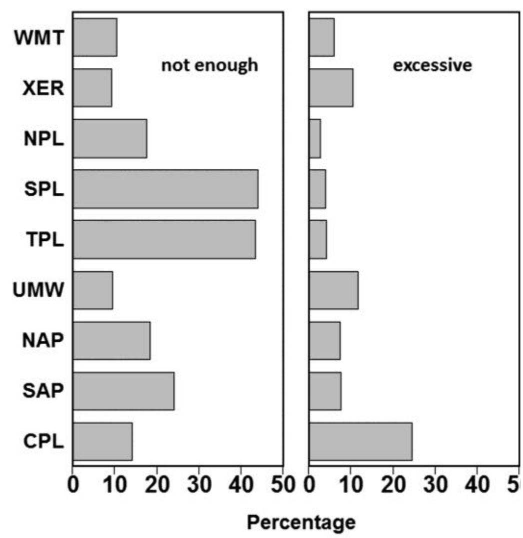


Figure 11.1. Percentage of 1,299 U.S. reservoirs ≥250 ac scoring high (i.e., moderate-to-high degradation, and high degradation) in the survey for not having enough aquatic macrophytes or having excessive aquatic macrophytes according to the ecoregions shown in Figure 1.3. Regions include Xeric (XER), Western Mountains (WMT), Northern Plains (NPL), Temperate Plains (TPL), Southern Plains (SPL), Upper Midwest (UMW), Coastal Plains (CPL), Southern Appalachian (SAP), and Northern Appalachian (NAP).

Three factors commonly preclude development of adequate aquatic plant densities in reservoirs. First, aquatic plant communities may take hundreds or even thousands of years to develop in natural lakes (Doyle and Smart 1993). Because most reservoirs are <100 years old, there has not been enough time to allow the development of a seed bank that can support suitable plant assemblages. Moreover, suitable seed banks may not exist in the reservoir's watershed. Second, the abiotic conditions in many reservoirs may be too harsh for many aquatic plants. These include high turbidity and large and rapid water-level fluctuations. Third, herbivores, including various fish species, reptiles and amphibians, mammals, birds, crayfish, and insects can prevent the survival of pioneer aquatic plant colonies that eventually may colonize the reservoir.

Macrophytes occasionally can become a nuisance, but how much is too much depends on the reservoir and its use. Some uses of a reservoir are more affected by macrophytes than others, and some types of plants interfere with boating or recreational activities more than others. Generally, plants are not considered to be a problem unless they interfere with desired uses for the reservoir. Some plants have capabilities to become very abundant and are thus apt to become a nuisance. An example is the nonnative hydrilla (*Hydrilla verticillata*), which is found in a wide range of environments. This plant has a broad tolerance in its environmental requirements and is capable of flourishing under what seems to be difficult conditions. Recreational boaters unwittingly contribute to the spread of hydrilla and other macrophytes by carrying fragments of the plant on their boats, trailers, or fishing gear to other water bodies.

Various problems frequently are attributed to the excessive growth of macrophytes in reservoirs. Oxygen deficiencies due to plant respiration and to decay of deceased plants often are identified as a major problem for various water uses. Excessive protection of prey fish to the extent that normal predator–prey interactions are substantially diminished and alter population dynamics, fish assemblage composition, and possibly fish production are major fishery concerns. Another common complaint is the interference with recreational activities such as boating, water skiing, swimming, and bank angling. Additionally, unsightly and odoriferous accumulations of plant material can develop on the water surface, on beaches, and along property fronts.

Terrestrial plants in regulated zones of reservoirs can provide important habitat to spawning adult fish and juveniles. The regulated zone often turns into bare shorelines or mudflats because of the die-off of flood-intolerant plants, which is caused by annual or semi-annual flooding, wave action, or both (section 7). Some reservoirs, particularly in the West, have steep, bare banks with 100-250-ft drawdowns. Conversely, shallow reservoirs with smaller drawdowns can expose extensive areas encompassing hundreds or thousands of acres and representing a large fraction of the reservoir. These large areas of bare mudflats exposed during drawdowns may be recolonized by terrestrial plants during drought years when water levels remain low but otherwise remain mostly bare and provide low-quality habitat.

11.2 Structure Provided by Plants

Aquatic vegetation increases the habitat complexity of reservoir ecosystems. An overabundance of plants, however, can interfere with fish feeding. In waters with no aquatic macrophytes, there may be insufficient cover to allow survival of structure-oriented small fish. As vegetation increases to intermediate levels, habitat becomes more complex, invertebrate densities increase, small prey and young predator fish find more refuge from predators, and recruitment into older age groups increases (Dibble et al. 1997; Miranda and Pugh 1997). At high levels of vegetation, especially dense monocultures formed by invasive aquatic species, it is more difficult for fish predators to forage because of the visual barrier or inaccessibility. This lack of access to prey causes overall slower fish growth, favoring small-size fish and reducing the larger fish that commonly make up a fishery. Fish assemblage composition may also shift. Reservoirs with low vegetation densities tend to include a higher abundance of fish species adapted to open-water habitats, whereas reservoirs with a high abundance of aquatic vegetation tend to be dominated by fish species adapted to cover (Bettoli et al. 1993). In addition, many fish that live among aquatic plants are visual feeders, and the shade produced by overhanging leaves and plant canopies improves visual acuity so fish can find prey and avoid becoming prey (Helfman 1981).

Researchers have suggested that a moderate amount of vegetation is optimal for fish production. Vegetation coverage of 20%–80% encourages the formation of stable fish assemblages, and 20%–40% has been reported as optimal (Durocher et al. 1984; Wiley et al. 1984; Miranda and Pugh 1997). This is a relatively wide range, which meets diverse goals of management including maintaining adequate fish and wildlife habitat.

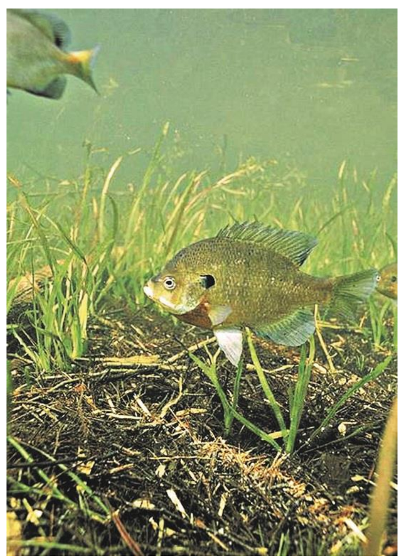


Figure 11.2. Bluegill using aquatic vegetation for protection and spawning. Photo credit: E. Engbretson, U.S. Fish and Wildlife Service (USFWS).

11.3 Influence of Plants on Fish Spawning

The structure provided by aquatic plants provides important habitat for fish reproduction (Petr 2000). Many fish are obligate plant spawners, directly or indirectly requiring aquatic plants to reproduce. Various fish families use vegetation as nurseries for their young, and reproductive success of nest spawners is improved when they have access to sites with aquatic vegetation, other forms of structure, or both. Fish can derive a number of benefits from nesting near aquatic plants. For example, vegetation can protect nest sites from wave action and sedimentation, which can harm eggs and larvae. Also, parents often use aquatic plant patches or edges as backing to protect nests from predators (Figure 11.2).

Figure 11.3. Reservoir managers can partner with local organizations to staff and fund plant establishment. Photo credit: Texas Parks and Wildlife Department, Austin.

11.4 Aquatic Plant Establishment

Aquatic vegetation is often lacking in reservoirs because of the unnatural fluctuations in water levels and the lack of an established seed bank. A seed bank may take several hundred years to develop in flooded lowlands and may take even longer along reservoir shores where soils originate from uplands (Godshalk and Barko 1985). Also, these upland shores may not be initially suitable for the growth of rooted aquatic plants if soils are hard-packed clay or rocky. To accelerate establishment of aquatic plants in reservoirs with little or no water-level fluctuations, efforts have been directed at planting native vegetation (e.g., American pondweed *Potamogeton nodosus* and wild celery *Vallisneria americana*) in exclosures to produce founder colonies (Smart et al. 1996). Exclosures are critical because small patches of transplanted plants or propagules growing along a barren shoreline quickly will be grazed by terrestrial, aquatic, amphibian, and avian herbivores (Smart et al. 1998). Although the prospect of establishing native plants is appealing because of their potential to transform fish habitat, such programs have had mixed success. Exclosures are prone to failure when they are forcibly entered by turtles and other grazers, and plants that expand outside the exclosures are often cropped by herbivores. Nevertheless, successful establishment of aquatic macrophytes in some reservoirs may be possible (Webb et al. 2012).

Establishment of aquatic plants has had some success through the establishment of a small, protected start-up of high-quality propagules, such as mature transplants, at strategic locations in the reservoir (Webb et al. 2012). This founder colony provides propagules that may allow expansion of the vegetation into a large section of the reservoir. The founder colonies expand through direct vegetative spread and through formation of new founder colonies from fragments or seeds (Smart et al. 1996, 1998; Webb et al. 2012).

Staffing and funding plant establishment programs can be difficult. Often the only way to accomplish all the steps in the establishment process is to partner with local communities, fishing clubs, lake associations, and schools (Figure 11.3). However, involving partners on plant-establishment programs is not a tough sell (Webb et al. 2012). The public can understand the benefit and eventually can see the product of their work. Fishing clubs often benefit directly through increased catch rates.

11.4.1 Plant Selection

Various plants have demonstrated potential for establishment in reservoirs (Table 11.1), and many others have been considered with limited or no success (e.g., sago pondweed *Stuckenia pectinata*, coontail *Ceratophyllum demersum*, muskgrass *Chara* spp., three-square bulrush *Scirpus americanus*, wild blue iris *Iris missouriensis*, swamp dock *Rumex verticillatus*). However, this aspect of reservoir habitat management is still in its infancy and relatively little is known about how to establish plants successfully. Perhaps an initial strategy may be to plant a diverse group of plants to explore which one(s) does best in the target reservoir (Webb et al. 2012). Species could be selected based on anticipated environmental conditions. For instance, in a reservoir known to fluctuate in water level, focusing on drought-tolerant or flood-tolerant species may be a sensible initial strategy. Planting aquatic vegetation or wetland species in reservoirs with extreme water-level fluctuations is unlikely to be successful. Established emergent plants can tolerate temporary inundation for weeks, but submersed species tolerate exposure and desiccation for only days or hours.

11.4.2 Source of Propagules

Although commercial suppliers may be a source of propagules, local production may often be preferred (Smart et al. 1998). Only a limited selection of aquatic plant species is available from commercial sources. Additionally, propagules available commercially are often marketed as seeds, tubers, winter buds, or root crowns but seldom as mature plants. These commercially available propagules can be used to culture plants and produce mature plants for establishment in the target reservoir. Commercial propagules are often available only seasonally, and availability may not be timely for a planting project. Moreover, even though a species may be distributed throughout the USA, genetic variability among plants associated with climatic diversity may require finding local sources (Webb et al. 2012).

11.4.3 Propagule Production

Occasionally, stands of plants suitable to provide propagules may be available from local wetlands. More commonly, propagules may need to be produced in

Table 11.1. Native aquatic plants potentially suitable for introduction into reservoirs (adapted from Webb et al. 2012).

Plant	Planting season	Planting depth (in)	Tolerance to	
			Desiccation	Herbivory
Wild celery *Vallisneria americana*	Early spring– early fall	12–48	Low	Low
American pondweed *Potamogeton nodosus*	Spring–late summer	12–48	High	Low
Illinois pondweed *Potamogeton illinoensis*	Early spring–midsummer	12–48	Low	Low
Water stargrass *Heteranthera dubia*	Early spring–late summer	12–48	High	Low
White water lily *Nymphaea odorata*	Late spring–midsummer	20–36	High	Moderate
Spatterdock *Nuphar advena*	Late spring–mid summer	20–36	High	High
Softstem bulrush *Schoenoplectus tabernaemontani*	Early spring–midsummer	0–36	High	High
Water willow *Justicia americana*	Early spring–midsummer	0–36	High	High[1]
Common spikerush *Eleocharis palustris*	Spring–midsummer	0–12	High	High
Squarestem spikerush *Eleocharis quadrangulata*	Spring–midsummer	0–24	High	High[1]
Pickerelweed *Pontederia cordata*	Early spring–late summer	0–36	Moderate	Moderate
Bulltongue arrowhead *Sagittaria platyphylla*	Early spring–late summer	0–48	Moderate	Moderate
Broadleaf arrowhead *Sagittaria latifolia*	Early spring–late summer	0–24	Low	Low
Creeping burhead *Echinodorus cordifolius*	Early spring–late summer	0–12	Moderate	High
Water hyssop *Bacopa monnieri*	Early spring–late summer	0–12	High	Low

[1] Particularly resistant to grass carp herbivory.

controlled environments (Figure 11.4). Propagule production for establishing founder colonies has focused on rooted plants. Culture of rooted aquatic plants depends on providing adequate light, adequate nutrients through sediment, and adequate levels of inorganic carbon via the water, all of which can be controlled under culture conditions. Nonrooted submersed aquatic plants obtain light, nutrients, and carbon via the water, so they are more difficult to culture. For this reason, establishment of nonrooted aquatic plants depends on access to specimens in natural populations. Detailed requirements for developing the infrastructure needed for propagule production are listed by Webb et al. (2012).

Figure 11.4. Growing propagules in nursery boxes prior to planting in a reservoir within exclosures. Photo credit: C. Bonds, Texas Parks and Wildlife Department, Austin.

11.4.4 Plant Establishment in Reservoirs

Once enough propagules are available, it is time to transplant them into founder colonies in the reservoir. Founder colonies are reportedly most successful when they are well protected from wind and wave action and are initiated in water <6 ft deep with a gradually sloping bottom. Sites with minimal wave action usually are associated with clearer water, are near the back of coves, and tend to have fine-textured substrate. Soft substrate allows for deep rooting of plants (about 6 in). If a protected shore is not available, wave action may be buffered with hay bales or other wave breaks (section 5.8.2). Depth is critical during plant establishment (section 11.4.5), and depth can be affected by water-level fluctuations. Protection of the founder colony from herbivores (e.g., deer, turtles, fishes) is critical. Various exclosures have been designed to keep herbivores out (see examples in Smart and Dick 1999; Webb et al. 2012). Within 1–2 years, founder colonies are expected to be expanding beyond the exclosure, although herbivores may halt or slow down expansion (Figure 11.5).

In general, planting may begin as early as practicable before or during periods of active growth to ensure establishment. Depending on latitude, planting may range from mid-spring to late summer (Table 11.1). In reservoirs that experience spring floods, planting can be delayed until water levels return to normal levels. Mature propagules can be planted over a wider range of time. Establishment of a viable population from mature propagules is possible in late summer, but late planting reduces the length of the growing season and may decrease the likelihood of success.

Figure 11.5. Founder colony developing within exclosures in Lake Gaston reservoir, North Carolina. Photo credit: A. Potter.

11.4.5 Multiple Depths Planting

Many reservoirs experience water-level fluctuations. Founder colonies planted at a single depth level may spend much of the year out of water or in water that is too deep and little time at ideal depths. Moreover, the period of ideal depth may not always coincide with the optimum growing period for a particular species. Establishing founder colonies at multiple depths increases the likelihood that plants will be actively growing and producing new propagules throughout the growing season. Webb et al. (2012) suggest that emergent species should be planted in less than 1 ft of water, floating-leaved species at 2 ft, and submersed species at 2–3 ft depth. To address fluctuating water levels (±2 ft), multiple exclosures may be constructed to track the water. In many reservoirs water levels may fall throughout the growing season, and establishing three or more depth tiers of plants is possible. As water levels change, plants exposed to desiccation or in water too deep generally decline but may recover when water levels return to suitable depths. Construction of exclosures often involve wire mesh and steel posts that can become a navigation hazard during high water if not marked or installed in isolated areas. Obtaining appropriate permits from the reservoir controlling authority before installation is a good practice.

11.4.6 Post-Planting Monitoring

Monitoring the results of plant establishment efforts is critical for long-term evaluation of the benefits (Smart et al. 1996). Without information on the possible causes of failed efforts—or successes—progress is slower. If a few propagules are planted in an exclosure, post-planting monitoring simply can involve counting the number of clumps within the exclosure. As the clumps begin to grow together, visual estimates of the percent cover of the plants within the plot or exclosure can be made. Line transects can estimate density and species composition as colonies expand outside the exclosures. Monitoring may be continued even after establishment is certain to track the species composition in the community, and desirable species that are missing or present in low quantities can be added selectively.

11.5 Control of Aquatic Plant Growth

Control of aquatic plants can rely on various strategies that often can be used in combination (an integrated management plan). Selection of the best treatment or combination of treatments depends on the species of plant, the extent of the problem, economic considerations, and local environmental conditions. Frequently, integrated management can provide more efficient control for less cost with superior results by matching individual controls to the goals and resource limitations of the individual situation. Major classifications of controls used in an integrated plan are outlined below.

11.5.1 Biological Control

Figure 11.6. Grass carp can help achieve various control goals when stocked at appropriate rates, but escapees can reproduce and can feed on nontarget aquatic vegetation. Photo credit: S. Miyazono, Mississippi Cooperative Fish and Wildlife Research Unit, Mississippi State University.

Biological control involves the introduction of a parasite, predator, or pathogen into the environment to suppress an unwelcomed plant species (Madsen 1997). Biological control operates by reducing the target population to lower, desirable densities suitable to maintaining fish habitat and recreational use of the reservoir. Therefore, the goal of biological control usually is not complete eradication of a plant from a water body. Biological control frequently is considered as one of the most environmentally acceptable goals for managing overabundance of aquatic plants.

Several broad types of biological control approaches can be recognized (Madsen 1997). These include introduction of host-specific organisms from the native range of the target plant, the use of opportunistic native or nonnative pathogens or insects, conservation or augmentation of native herbivores, and the use of general feeders or non-host-specific organisms. An example of the last is the Asian grass carp that is used to control most types of submersed aquatic vegetation (Figure 11.6). Biological control is typically a long-term approach for the suppression of a target plant species. A disadvantage of using biological control alone is that results can be unpredictable: control can take too long, overrun target levels, or produce undesirable side effects. This long-term method of suppression is best suited in low-priority areas, at sites where the use of other control strategies would be cost prohibitive, or where the goal is maintaining a lower level that has already been achieved. Biological control is a potentially effective long-term control practice when used in conjunction with the short-term chemical or mechanical options (sections 11.5.2 and 11.5.3).

A concern regarding the use of grass carp as a biocontrol agent is the potential of escaped fish to reproduce in the wild or feed on aquatic vegetation the manager wants to preserve. The development and aquacultural production of sterile triploid grass carp has provided a solution to the reproduction problem. As to the latter problem, grass carp stocking may not be prudent in open systems that are connected to a stream or river because grass carp are attracted to moving water and will leave the stocked water body. Grass carp stocking rates in closed systems typically range between 2 and 50 fish/ac. There is no “magic number” of grass carp to stock to achieve a specific percentage of submersed weed control because optimum stocking rate is dependent upon the type and quantity of aquatic plants present, water temperature, lake morphometry, and desired speed of control. Grass carp remain illegal in many states, and most other states require permits for use of the fish.

Various introduced and native insects (e.g., beetles, weevils, moths, mites) have been used for the control of alligator weed (*Alternanthera philoxeroides*), water hyacinth (*Eichhornia crassipes*), hydrilla, purple loosestrife (*Lythrum salicaria*), Eurasian watermilfoil (*Myriophyllum spicatum*), giant salvinia (*Salvinia molesta*), and water lettuce (*Pistia stratiotes*) (Newman 2004). The use of insects as biological control agents for aquatic plants has yielded mixed results, which is typical and expected of biocontrol programs. However, a few aquatic plants, including alligator weed and purple loosestrife, have been controlled successfully by insects released as biocontrol agents. Control of other plants—including water hyacinth, hydrilla, Eurasian watermilfoil, and giant salvinia—has been less successful. Multiple factors often play a role in the failure of some biocontrol agents to reach their full potential. Insects can be an effective tool in the manager's toolbox since host-specific biocontrol agents allow management of populations of undesirable species while leaving nontarget plants unharmed (Newman 2004). A major factor that limits insect utility is that unless a potential control agent is species specific, it cannot be introduced into the USA. Therefore, it is unlikely that a plant control program can rely on biocontrol alone.

Biological control also can involve introduction of desirable native plant species to fill the vacant niche resulting from disturbance due to other control measures. If the native species can preempt recovery or reduce the probability of reintroduction of nuisance species, the temporal benefit of the original control measure can be prolonged and the need for additional control inputs may be minimized.

11.5.2 Mechanical and Physical Control

Mechanical and physical control practices have been used to control many aquatic plants, especially invasive and exotic species.

11.5.2.1 Hand pulling

Hand pulling is similar to weeding a garden. The whole plant, including the roots, is removed while leaving any desired accompanying plant species intact. This procedure works best in soft sediment, with shallow rooted species, and in small (discrete) areas. Hand pulling can be a highly selective technique, provided the target species can be identified easily (Kettenring and Adams 2011). The process has to be repeated often to control regrowth. When hand pulling nuisance species the entire root system and all fragments of the plant are pulled; even small root or stem fragments could result in additional growth. Once the bottom substrate is disturbed, suspended sediment often greatly reduces visibility, which results in the need to make multiple passes over the same area. The time required by hand-pulling operations varies widely depending on the degree of infestation. Hand pulling usually is used as a component of invasive species management programs to target new infestations with low plant density (generally <500 stems/ac). Hand pulling is often an important follow-up strategy to an herbicide treatment program to extend the duration of plant control.

11.5.2.2 Hand cutting

Hand cutting can be used for localized removal of invasive aquatic plants. The removal of small patches of vegetation can be accomplished by cutting with hand tools while wading along the shoreline or floating on a small boat in shallow water. This approach is feasible only in areas where water level allows access, usually less than about 4 ft deep. Various commercial companies have developed power and nonpower hand tools specifically designed to remove submersed aquatic plants. Because many submersed aquatic plants spread by fragmentation, hand cutting may exacerbate the problem, but that depends on the plant. If the plant spreads by fragmentation, hand-cutting operations may be appropriate only in lakes where the plant has expanded to most of the littoral zone. Cutting pioneer colonies could accelerate the spread of the plant to noninfested areas.

11.5.2.3 Hand rakes

Hand rakes of varying sizes and configurations are available for aquatic weed control. Many of these hand rakes are lightweight aluminum with rope tethers and are designed to be thrown out into an area and dragged back onto shore. Some are designed to cut the weeds instead of raking them back to shore. While these may be cost-effective strategies to manage small areas, there is a risk that these rakes will make the problem worse by creating weed fragments that can escape and infest other portions of the reservoir.

11.5.2.4 Mechanical harvesters

Mechanical harvesters are machines that cut and collect aquatic plants (Figure 11.7). These machines can cut the plants 5–10 ft below the water surface and may cut an area 6–20 ft wide. Most mechanical harvesters are highly maneuverable around docks and boat houses and can operate in as little as 12–18 in of water. The plants are cut and then collected by the harvester, stored within the harvester or accessible barge, and then transferred to an upland site (Madsen 1997). The advantages of this type of weed control are (1) cutting and harvesting immediately opens an area, such as boat lanes; (2) plants are removed during harvesting and do not decompose and reduce

Figure 11.7. An aquatic plant harvester on Fairview Lake reservoir, Oregon, working to remove excessive underwater plant growth. Photo credit: B. Greene, *The Oregonian*.

dissolved oxygen in the water column as they do after herbicide application; and (3) the habitat remains intact because most harvesters do not remove submersed plants all the way to the lake bottom, i.e., clipped plants remain rooted in the sediment and regrowth can begin soon after the harvesting operation. However, there are disadvantages (WDFW 2011). These include (1) the equipment is fairly expensive; (2) harvesting may have to be repeated several times per growing season to maintain control of nuisance aquatic plants; (3) mechanical harvesting leaves plant fragments floating in the water, which if not collected may spread the plant to new areas; (4) harvesters may affect nontarget organisms such as insects, amphibians, and fish, removing them with harvested material; (5) cutting plant stems too close to the bottom can result in resuspension of bottom sediment and nutrients; (6) harvesters are not species selective; (7) harvesters cannot be used where abundant timber was left in the reservoir basin at impoundment; and (8) a crew operating a harvester can generally clear <5 ac/d, whereas a crew applying herbicide can cover 10–15 ac/d.

11.5.2.5 Track hoes and draglines

Track hoes are large shovel machines, and draglines use a large cable system to cast and drag a shovel that collects plants and organic material (Figure 11.8). Track hoes have claw shovels that can reach 25–30 ft over the water body, dig down, and pull plants back to shore. Shore-based track hoes or draglines are best suited for channel maintenance, in areas where

Figure 11.8. Pushing aquatic plants to shore for harvesting by track hoe. Photo credit: Center for Aquatic and Invasive Plants, University of Florida, Gainesville.

plants accumulate, or in locations where plants can be pushed to an established collection point. Barge-mounted track hoes or draglines can be used for transportation to off-shore work sites. In that case, plants are loaded on an attending barge and hauled to a disposal site.

11.5.2.6 Legalities of collection and transportation

Plant collection and transportation in most states is subject to various state and federal regulations. Various permits may be necessary. The laws and regulations in this regard are complex, subject to frequent change, and vary among states. Some species may be under special legal protection because of their conservation or nuisance status. It usually is prohibited to transport certain noxious plants within a state or across state lines. Some collection sites, such as aquatic preserves or parks, may be off limit. Thus, several types of regulation may need to be considered when transporting plants to disposal sites or when collecting specimens for establishing plant colonies.

11.5.2.7 Water drawdown

Water drawdown can be an effective aquatic plant management method (Cooke 1980). It is used for control of submersed species, and it is most effective when the drawdown depth exceeds the depth of invasion of the target plant species. In northern areas during the winter, drawdown will result in plant and root freezing for an added degree of control (Beard 1973). Drawdown is typically inexpensive and has effects that last two or more years. Drawdowns can have various other environmental effects and interfere with other functions of the water body (section 7).

Plants that are controlled by drawdowns usually include many submersed species that reproduce primarily through vegetative means, such as root structures and vegetative fragmentation. Some invasive submersed species most commonly targeted by drawdown include Eurasian watermilfoil, fanwort (*Cabomba caroliniana*), *Egeria* spp., and coontail. However, opportunistic species like hydrilla may expand rapidly following drawdown.

Figure 11.9. Drawdowns to control nuisance aquatic vegetation on B. A. Steinhagen Lake reservoir, Texas, were successful only when coordinated with prognostic of bitterly cold temperatures. Photo credit: U.S. Army Corps of Engineers, Town Bluff Project.

A general rule of thumb is to maintain drawdown conditions for 6–8 weeks to ensure sufficient exposure to freezing and drying conditions (Figure 11.9).

11.5.3 Chemical Control Practices

The use of herbicides for the control of aquatic plants represents one of the most effective management options available. Herbicide control is often the first step in a long-term integrated control program (Madsen 1997). Federally approved compounds for aquatic plant use are summarized in Table 11.2. The cost of testing and registering aquatic herbicides limits the number of available herbicide options.

11.5.3.1 Herbicide use and classification

There are approximately 300 herbicides registered in the USA, but only around a dozen are registered for use in aquatic systems (Getsinger and Netherland 1997). Herbicides labelled for aquatic use can be classified as either contact or systemic (Table 11.2). Contact herbicides act immediately on the tissues contacted, causing extensive cellular damage at the point of uptake. Typically, these herbicides are faster acting, but they may not have a sustained effect, in many cases not killing root crowns, roots, or rhizomes. In contrast, systemic herbicides are translocated throughout the plant. They are slower acting but often result in mortality of the entire plant.

In treating submersed species, application is made directly to the water column as concentrated liquids, granules, or pellets, and the plants take up the herbicide from the water. The applicator needs to know the water exchange rate to determine the appropriate exposure time and concentration of the herbicide required to control a specific target plant. This value may be different for each target species. Species with significant above-water vegetative surfaces, such as floating and emergent species, can be treated with direct application to the surface of the actively growing plant (Figure 11.10). For these species, care is taken to avoid application if rain events are likely.

Figure 11.10. A Texas Parks and Wildlife Department crew sprays an herbicide on giant salvinia detected in Lake Fork. Photo credit: K. Storey, Texas Parks and Wildlife Department, Tyler.

Instructions for the use and application of herbicides change often. Whether an herbicide is appropriate for a water body or aquatic system with a particular water use

Table 11.2. Characteristics of U.S. Environmental Protection Agency approved aquatic herbicides including trade names, formulation, and whether they target submersed, floating, or emersed plants. Listed in parentheses is whether the herbicide is contact (acts immediately on the tissues contacted) or systemic (translocated throughout the plant).

Herbicide	Some trade name(s)	Formulation	Target form		
			Submersed	Floating	Emersed
Copper (systemic)	Komeen, Nautique, Copper Sulfate, Cutrine, Cutrine Plus	Liquid, granular	•	•	
Endothall (contact)	Aquathol K, Hydrothol 191	Liquid, granular	•	•	
Diquat (contact)	Reward, Harvester, Tribune, Tsunami DQ, Weedtrine D	Liquid	•	•	•
Carfentrazone (contact)	Stingray	Liquid		•	•
2,4-D (systemic)	Aqua-Kleen, Weedar 64, Navigate	Liquid, granular	•	•	•
Triclopyr (systemic)	Renovate 3, Renovate OTF	Liquid, granular	•	•	•
Glyphosate (systemic)	Rodeo, Shore-Klear, Aquapro	Liquid		•	•
Imazapyr (systemic)	Habitat, Ecomazapyr 2sl, Imazapyr 2sl, Polaris AC	Liquid			•
Fluoridone (systemic)	Avast, Sonar, Whitecap	Liquid granular	•	•	
Penoxsulam (systemic)	Galleon SC	Liquid	•	•	
Imasamox (systemic)	Clearcast	Liquid	•	•	•

is specified on the product label. Always follow the instructions on the label and check with the appropriate regulatory agencies in your state before applying herbicides to any body of water.

11.5.3.2 Selectivity

Herbicide activity can be characterized as species selective or nonselective (Getsinger and Netherland 1997). Nonselective or broad-spectrum herbicides control all or most vegetation because they affect physiological processes common to all plant species. Because nonselective herbicides can kill all vegetation they contact and not just the problem species, care is taken that they do not affect desirable plants. Selective herbicides will control only those groups of plants that carry the biological pathways targeted by the chemical active ingredient.

11.5.3.3 Control of specific plants

A wealth of information is available online about invasive species in general, common nuisance species in particular, and relevant species-specific treatments. Because these are changing often, details are not considered here. The following websites are excellent reference sources and updated often:

The University of Florida Center for Aquatic and Invasive Plants
http://plants.ifas.ufl.edu/
U.S. Department of Agriculture Natural Resources Conservation Service, Plants Database
http://www.plants.usda.gov/

U.S. Army Corps of Engineers, Aquatic Plant Management Information System
http://el.erdc.usace.army.mil/apis/

Texas AgriLife Extension Service Aquaplant: A Pond Manager Diagnostics Tool
http://aquaplant.tamu.edu/

11.5.4 Cultural Control Practices

Cultural control techniques focus on a large array of social methods used to prevent or reduce the entry or spread of unwanted aquatic plant species. Cultural control practices can be an essential component of long-term management and prevention of aquatic plant infestations.

11.5.4.1 Prevention

Prevention is one of the best and most cost-effective methods to avoid aquatic plant infestations (Figure 11.11). A commitment of volunteer time to a plant control program can save thousands of dollars in invasive plant management costs. Volunteer boat cleaning, inspections, and temporary quarantine during transfer of watercraft are all components of prevention programs. However, this type of program does require management, education, and planning. Because nutrients and sediment influence the presence and growth of macrophytes, curtailing their flow into the reservoir is important. Preventive maintenance or actions that can be taken include curtailing fertilizer use; using a phosphorus-free fertilizer on established lawns; developing landscaping practices that do not require nutrients and instead will trap nutrients running into the reservoir; maintaining septic tank systems to prevent failures and supporting laws aimed at preventing construction of septic tanks in unsuitable soil types; and supporting the adoption of ordinances designed to minimize surface water runoff and unnecessary land clearing during construction.

Boat ramp monitoring programs are used to inspect boats and trailers for the presence of invasive species. These are largely volunteer or summer intern positions that try to staff boat ramps during peak-use periods. Inspections can be either mandatory or voluntary and usually only take a matter of minutes. The interaction with boat ramp monitors also provides an opportunity to distribute educational materials.

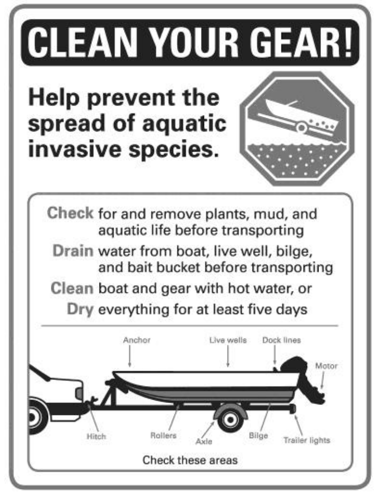


Figure 11.11. Pennsylvania is enlisting the help of the public to fight the spread of aquatic invasive plant species. Signs such as this one are posted at boat ramps to help increase public awareness and encourage boat inspections. Photo credit: Pennsylvania Fish and Boat Commission, Harrisburg.

11.5.4.2 Education

Education is a key component of prevention. Educating reservoir users and the general public about the threat of invasive species is necessary to prevent new infestations and to sustain effective aquatic plant management programs. Education involves creating public awareness of the problem and familiarizing people with possible solutions.

Education facilitates involvement of both volunteer labor and other resources to accomplish a management goal. Many activities can be used for education, including workshops, public meetings, press conferences, news releases, posters and flyers, popular articles, postings at boat ramps, videos for interest groups, development of publicized web sites, and involvement of recreation associations, fish and wildlife groups, and social media. Well-educated citizens and technically informed agency biologists are essential components in the successful control of invasive aquatic plants. Educational efforts may focus on preventing the spread to new water bodies by educating the nursery and aquarium trade, recreationists and boaters, the general public, and policy makers. A lake association or friends of reservoir chapter can help with these activities (section 13.5).

11.6 Promotion of Terrestrial Plants on Barren Shorelines

Barren shorelines in reservoirs are caused by water-level fluctuations and their negative effect on flood-intolerant plants (section 7). In contrast to other engineered environments, attempts to establish and improve the vegetation of bare reservoir shores have been few (Allen 1988; Fraisse et al. 1997). Gill and Bradshaw (1971) proposed three explanations to account for those reservoir shorelines that are devoid of vegetation: (1) the environment is so extreme that plants are incapable of colonizing or growing; (2) the environment in the margins is not extreme, but suitable plants for colonization do not grow nearby; and (3) the environment in the margins is sufficiently extreme to prevent natural colonization, but not vegetative growth—if plants were introduced by artificial means, they would flourish. The first explanation is plausible if the extreme environment is caused by water levels that fluctuate relatively quickly or drop too late in the growing season so that time available for establishment is minimal. If so, artificial plant establishment could mitigate this deficiency. The second explanation is unlikely because plants generally have high dispersal ability. To be sure, well-vegetated riparian zones may be encouraged above normal (summer) pool elevation. These can act as "source" sites for colonization of the drawdown area. The third explanation suggests that if suitable seeds are absent from the substrate, or are unable to germinate, then the introduction of propagules may be needed to attain basic vegetation cover (Brock and Britton 1995). Indeed, it has been found that the use of appropriate species and management techniques can create plant communities that will survive and benefit from flooding and exposure (Allen and Klimas 1986; Allen 1988).

11.6.1 Well-Vegetated Riparian Zones

Properly managed riparian zones are advocated to filter potential pollutants from inflowing runoff, to provide a source of shade and woody debris for the littoral zone, and to maintain desirable aesthetics (section 8). However, a riparian zone also can be thought of as a source of seeds for plant colonization of the fluctuation zone of a reservoir. To be effective as such, the riparian zone needs to include a plant community consisting of upland and wetland species capable of colonizing the fluctuation zone at various times of the year, depending on timing of drawdown. This flexibility can be achieved by maintaining a diverse natural plant community including a mix of aquatic grasses, sedges, and rushes along with upland plants growing on shore. Riparian zone management is discussed in section 8.

11.6.2 Seeding

Seeding of herbaceous terrestrial plants in dewatered fluctuation zones can succeed if done during the growing season, although historically seeding has had mixed success (Figure 11.12). Candidate plant species for seeding mudflats in regulated zone of reservoirs are suggested in Table 11.3. Hulsey (1959) successfully planted rye in Arkansas reservoirs in late September. In Kansas reservoirs, Groen and Schroeder (1978) planted rye (30–60 lb/ac), ryegrass (10 lb/ac), and wheat (30–60 lb/ac) during September or October. In drawdowns before August, Japanese millet and hybrid sudan–sorghum were planted in Kansas and Arkansas, leading to lush stands (Groen and Schroeder 1978). However, summer drought conditions can lead to poor survival (Ploskey 1986). In some situations it may be possible to raise water level slightly to "irrigate" the seeded vegetation, but this has not been tried. Strange et al. (1982) planted rye, fescue (*Festuca* spp.), a sudan–sudan hybrid, and a sudan–sorghum hybrid (45 lb/ac) from July to September on the exposed mudflats of Lake Nottely, Georgia. Grasses grew poorly in unfertilized sites but did well when fertilized. The numbers of aquatic insects, small sunfish, and age-0 black basses were higher in seeded areas. Ratcliff et al. (2009) planted barley at Shasta Lake, California, and observed that juvenile black bass abundance over 50 times higher in planted grass.

Figure 11.12. Seeding mudflats at Enid Lake reservoir, Mississippi, with an all-terrain vehicle and a food plot implement. Photo credit: L. E. Miranda, Mississippi Cooperative Fish and Wildlife Research Unit, Mississippi State University.

Various site factors are considered in planning a shoreline revegetation effort (Allen and Klimas 1986). These include water-level fluctuation range and time of year; bank morphometry (i.e., steepness and shape); extent of wave action; animal depredation potential; and soil texture, fertility, and moisture status. Success rates are likely to be highest on sites that are gently sloping (i.e., bank slopes not greater than 1:3 vertical to horizontal), are protected from extreme wave action, have soils conducive to plant growth, and do not support high populations of potentially destructive animals, e.g., beavers, muskrats, and cattle. Sites with adverse characteristics such as steep or vertical banks can be vegetated but will require more effort and expense. Soils consisting predominantly of shrinking and swelling clays or those having high concentrations of sodium salts are less likely to produce plants. Soil analyses can identify such prohibitive characteristics and aid in choosing sites to be revegetated and target plant species, as well as determine if soil amendments will be needed.

11.6.3 Timing

As a general rule seeding is conducted when favorable soil-moisture and temperature conditions are going to occur. Often, it is best to seed or plant in the fall just after water levels drop so the planting substrate is still moist. Nevertheless, if reservoirs are at their lowest level during December or January and rise very slowly during the spring, seeding or planting could occur during winter to early spring, depending on rainfall availability, temperature conditions, and plant species. Some grass and herbaceous species can be seeded or transplanted in either the spring or the fall, while others establish better in a particular season.

11.6.4 Seeding Methods

The methods of seeding are determined by location, size, and topography of the reservoir shoreline; time of drawdown; water level; seed mixture; and soil conditions. If the revegetation site will be subjected to fluctuating water levels or wave action soon after planting, seeding is probably not the best plant establishment alternative because the seeds are likely to wash out. If reservoir water levels are lowered long enough for seeds to germinate and plants to grow, seeding will be the most cost-effective means of establishing plants, particularly grasses and forbs. Fowler and Maddox (1974) and Fowler and Hammer (1976) were successful in seeding mudflats in Tennessee reservoirs by means of various techniques, some of which are described below.

11.6.4.1 Broadcasting

The most common method of seeding on large areas is to disperse seed from a tractor-mounted or all-terrain vehicle (ATV)-mounted broadcast seeder. Broadcasting by hand with a knapsack seeder usually is restricted to small areas or inaccessible sites such as steep slopes. Broadcasting by hand is labor intensive and used only when no other method is applicable. Because of the relatively harsh growing conditions on reservoir shorelines, three to five times the normally recommended amounts of seed may need to be mixed thoroughly with fertilizer and sawdust or sand and broadcasted over the site. The sawdust or sand serves as an indicator of areas already seeded and promotes a more even distribution of seeds. Broadcast seeding is rapid and easy but is typically not recommended for large or fluffy seeds that may plug the equipment, blow away, or be lost to scavenging animals.

11.6.4.2 Drill seeding and cultipacking

Drill seeding (Figure 11.13) and cultipacking (Figure 11.12) are generally preferred over broadcast seeding. Both of these methods will place seeds in the soil at the desired depth and cover them with soil for germination. Tractors or ATV-mounted seeders may be used. These seeders often have one or several seed boxes designed to seed various seed sizes and mixtures (small and dense, light and fluffy, or medium-heavy seeds) with fertilizer at the time of seeding.

Figure 11.13. A drill seeder positions seeds in the mudflats at the proper depth and covers them, preventing predation by birds and improving germination rate over seed broadcasting. Photo credit: M. Lashley, Mississippi State University.

Drills have coulters that will lay open the surface soil for seed placement, leading to better seed–soil contact. Drill seeding has been successful on some reservoirs and can be done cost effectively if terrain and soil conditions permit. The South Dakota Game, Fish, and Parks successfully drill-seeded reed canary grass (*Phalaris arundinacea*) on a shoreline of Lake Oahe reservoir. Reed canary grass provides spawning substrate for northern pike.

Cultipackers cover the seed with a minimum amount of soil to ensure proper seed-to-soil contact. It resembles a large rolling pin with evenly spaced ridges and dimples. The cultipacker's primary functions are to break up clods, remove excess air spaces from loose soil and smooth the soil's surface. This method consists of heavy-duty, smooth, spoke or crowfoot rollers that provide clod-breaking and smoothing capabilities. As with any tillage, it is important not to overwork the soil or work it when it is too wet.

A diversity of equipment is available to spread and bury seeds with ATVs. Up-to-date information can be obtained at web sites that specialize in food plot seeding implements.

11.6.4.3 Hydroseeding

Hydroseeding involves spraying a slurry of seed, fertilizer, mulch, and water onto a site (Figure 11.14). It is commonly used for seeding steep road banks or the uneven terrain of surface-mined lands. It may be used to vegetate reservoir shorelines by mounting the equipment on a barge that can be towed to otherwise inaccessible sites. Fowler and Hammer (1976) described modified hydroseeding equipment, the aquaseeder, which was developed for the Tennessee Valley Authority and was tested

Figure 11.14. A team of Alabama Power and U.S. Forest Service biologists used a pontoon boat equipped with a spray rig and 150-gallon tank to hydroseed banks of Smith Lake reservoir, Alabama. Photo credit: Alabama Power, Birmingham.

successfully along reservoir regulated zones. Hydroseeding has the advantages of using a one-step application of seeding materials and the ability to seed large areas of rough terrain. Disadvantages are that it can damage seeds, and broad mudflats may be inaccessible to floating hydroseeding equipment. Because of potential soil erosion associated with steeply sloping reservoir shorelines, mulching over the seeds is often required to protect the surface soil. However, mulching is used only if water levels will remain down until the plants have reached a desired size.

11.6.4.4 Aerial seeding

Seeding from aircraft is a specialized technique and can be quite expensive unless it is applied to large areas (i.e., >100 ac). It is often used where site features prevent conventional methods from being used. In 1973 and 1974, the Tennessee Valley Authority successfully used this technique with a helicopter and a hopper-spreader unit to vegetate >1,000 ac of mudflat on an experimental basis. The helicopter operated 20 ft above the ground over a 30-ft swath at the speed of 30 mph and spread 20 lb/ac of annual ryegrass. A possible disadvantage of using helicopters for aerial seeding on reservoirs, particularly where drawdowns are erratic, is the difficulty of scheduling the service (Fowler and Hammer 1976). Also, steep shorelines may be difficult to seed with this method because of the inability to achieve a uniform spread and obtain good seed–soil contact.

11.6.5 Transplanting

In contrast to seeding, transplanting uses one or more of several kinds of planting stocks, including bare-root seedlings, rooted or uprooted cuttings, balled-and-burlapped plants, containerized plants, sprigs, plugs, rhizomes, and tubers. Transplanting is generally more effective than other establishment techniques because root system development and height growth are maximized during the growing season prior to inundation of the site. Nevertheless, transplanting requires substantially more labor than seeding and may be impractical in large areas. Moreover, plants are seldom available from commercial growers and have to be either removed from wetlands or grown in an in-house nursery.

11.6.6 Grasses and Other Herbaceous Plants

This group of potential transplants may include wetland species of the genera *Carex*, *Cyperus*, *Eleocharis*, *Juncus*, *Panicum*, *Polygonum*, *Phragmites*, *Sagittaria*, *Scirpus*, *Spartina*, and *Typha*. There are four forms of propagule types commonly used to establish grasses and other herbaceous plants as transplants on reservoir shorelines.

11.6.6.1 Sprigs

This propagule is the entire plant dug and removed from its natural habitat and transplanted to the new site. The term "sprig" generally refers to smaller transplants that are obtained by breaking multistemmed plants into smaller clumps containing one to five stems. It is best to leave soil on transplant roots when they are dug to minimize root loss and disturbance. Plants dug during the dormant season usually suffer less from stress and shock than those dug in the late spring and summer.

11.6.6.2 Rootstocks and plugs

Rootstocks consist of the root system of a plant, including that portion of stem normally growing below ground. The propagule may be divided into sections or clumps for planting; new growth will generate from the old root systems. Plugs are obtained by extracting rootstocks with some type of coring device. This approach was applied to planting marsh in western New York; cores of wetland soil were transplanted in a grid pattern on 3-ft centers and subsequently flooded (Allen and Klimas 1986). The cores contained various types of propagules that were present in the source wetland, including rootstocks, rhizomes, seeds, and whole plants. Plugs can be carried in plastic bags to a shoreline to be vegetated and planted in or out of water. Planting in water, however, is very time consuming and more costly. Using plugs and the coring method described would have its greatest utility in reservoir areas shallowly covered by water, such as some mudflats and shallow-sloped shorelines.

11.6.6.3 Rhizomes and tubers

Rhizomes are similar to rootstocks but refer to underground stems that often grow horizontally. The rhizomes are dug and divided into sections, taking care to keep at least one viable growth point (node) on each to ensure new growth. Tubers are large, fleshy underground stems often associated with rhizomes. They are usually available to be dug near the end of the growing season.

11.6.7 Trees and Shrubs

This group of potential transplants may include wetland species of the genera *Salix*, *Cornus*, *Morus*, *Nyssa*, *Populus*, and *Taxodium*. Four propagule types may be used to establish trees and shrubs in the drawdown zone of reservoir shores: bare-root seedlings, cuttings, and balled-and-burlapped and containerized plants. These four types exhibit various advantages and disadvantages.

11.6.7.1 Bare-root seedlings

Bare-root seedlings are young plants with exposed root systems that are transplanted from nursery beds or from natural stands to the planting site. Seedlings of trees and shrubs are usually hand planted, using either a mattock or planting bar (dibble) for preparing a hole. Bare-root transplants are successful for many tree and shrub species, but because site conditions can be restrictive, survival will probably be higher with container-grown stock (Allen and Klimas 1986). The advantages of using bare-root stock are that seedlings are easier to handle, are less costly, and are easier to plant. These characteristics make bare-root materials appropriate for planting larger areas.

11.6.7.2 Cuttings

Cuttings are sections of the shoots of a plant and include nodes in the section cut. Cuttings may be unrooted or rooted. To obtain rooted cuttings, roots have to develop in an appropriate rooting soil, possibly treated with a root stimulator. If planted as unrooted cuttings, the cut section can be placed in the substrate at the planting site. The size of cuttings may vary from thin slips (<0.5-in diameter) to large poles (4-in diameter, 10 ft long) (Allen and Klimas 1986). When cuttings are planted, they need to extend deep enough into the soil to be firm and relatively difficult to pull out; only 1–2 in may be left above ground to prevent moisture loss, with any excess pruned off. Cuttings may be pushed directly into the soft soils of recently dewatered areas (Gray and Leiser 1982). Not all trees and shrubs will reproduce from cuttings; only those that sprout readily from the stem are likely to grow. Examples of woody species that readily sprout from the stem include all willows (*Salix* spp.), some poplars (*Populus* spp.), river birch (*Betula nigra*), swamp privet (*Forestiera acuminata*), and some alders (*Alnus* spp.). Use of unrooted cuttings could be an economic method of plant establishment, so some pilot testing plots may be considered.

11.6.7.3 Balled-and-burlapped propagules

Propagules that are balled-and-burlapped refer to large trees and shrubs >5–7 ft tall that have been nursery grown with balled-and-burlapped root systems. These propagule types are normally too expensive for most shoreline revegetation projects, except in recreation areas that are subject to periodic inundation and for which higher planting costs can be justified.

11.6.7.4 Containerized propagules

Containerized tree and shrub propagules are those that have been grown in pots or similar containers. Plants grown in gallon-sized or larger containers are often available for tree and shrub species used in regular commercial landscaping but are limited in variety. Consequently, they may not be best for use on reservoir shorelines that are periodically inundated, unless a nursery has been contracted to grow flood-tolerant species. Survival frequently is reduced because of limited root systems in relation to size of the tops of the plants (Allen and Klimas 1986). The main advantage of containerized plants is that they have developed root systems and stems that are ready to grow when they are placed into the ground. However, containerized plants cost considerably more than other propagule types. Therefore, they are often reserved for high-priority recreation sites or other such sites requiring greater assurance of success.

At Lake Fork, Texas, containerized buttonbush (*Cephalanthus occidentalis*) was introduced successfully on the exposed littoral zone during a prolonged drought (R. Ott, Texas Parks and Wildlife Department, personal communication). These specimens persisted when the reservoir level recovered. However, similar introduction of additional containerized specimens in the same area following water-level recovery was unsuccessful because wave action uprooted the specimens before establishment could occur.

11.6.7.5 Spacing

Spacing of plantings generally ranges from 2 to 15 ft centers. Nevertheless, spacing is influenced by the project goal. Spacing for aesthetic improvement of a project area may be different than when the goal is to improve fish habitat. Other factors that may be considered in selecting the plant spacing includes growth patterns, survival rate, time of planting, and propagule type.

Section 12

Habitat Management Decisions

12.1 Introduction

Habitat management requires clearly identifying the habitat problem, goals for remediation, and objectives to attain the goals and choosing alternative management actions to reach the objectives. Decisions about habitat management are sometimes simple. Some problems have an obvious solution, and given the same objectives, two decision makers probably could choose similar alternative actions. More often, the scope and complexity of some of the habitat problems can make choosing difficult. Thus, decisions can be complex, and multiple decision makers easily can disagree on alternative actions (Figure 12.1). Furthermore, the process by which alternative actions are chosen can be difficult to explain, which in turn makes it hard to communicate to administrators, partners, stakeholders, and the public. A structured approach to choosing among alternative actions can facilitate and clarify the decision-making process by decomposing a problem into elements that are easier to rationalize and convey.

Figure 12.1. Decisions about how to approach fish habitat management are rarely simple and often can be complex. Photo credit: L. Passuello.

Decision science is being applied increasingly in management of natural resources (Conroy and Peterson 2013). This section describes a basic structured approach to arriving at decisions about implementing management to address perceived habitat problems. More complex models for arriving at optimal decisions are beyond the scope of this section but are available from Conroy and Peterson (2013).

The approach considered here includes three major components. First, identify goals and objectives to address the perceived problem. Second, recognize alternative actions to achieve the objectives. Third, score alternative actions relative to meaningful criteria that estimate the utility of each action. Once utilities are quantified, the manager can select one or more actions to address the habitat problem.

12.2 Problem Statement, Goals, and Objectives

The first step is a problem statement that includes a clear declaration of the fish habitat problem and the underlying problem to be solved. This first step may be completed in-house by agency personnel but may require participation by key stakeholders (section 13), ensuring that collectively key stakeholders have a say and ultimately concur on the problem statement. This first step guides the process toward clearly stating the problem to be solved and ensures that ultimately the right objectives are established to solve the right problem. In many cases the problem is poorly stated, if stated at all. This can lead to actions that may not address the problem adequately, may waste time and resources, and even may create new problems and conflict with partners and stakeholders.

Defining the problem can be more complicated than it seems. This is often where managers may struggle because they react to what they think the problem is. Trying to better understand why one thinks there is a problem may help elucidate the problem more clearly. This search may include questions such as (1) what are we observing that causes us to think there is a problem; (2) where is the problem happening and how; (3) when and why is it happening; and (4) how are fish affected by the problem. Based on the answer to these or similar questions it would be appropriate to write down a paragraph starting with something such as, "The following should be happening, but isn't." If the problem seems overwhelming, it may be necessary to break it down into several problems by repeating the process. If the problem consists of several related problems, it may be necessary to prioritize which problems should be addressed first.

A goal is the end that the manager wishes to achieve through the management action(s). Goals are broad-brush statements that identify the overall management purposes or a desired end state. For example, generic statements such as "maintain adequate habitat connectivity" or "improve water quality" are general statements (goals) about why management actions are undertaken. Habitat management goals should identify the desired state of the system once the identified problem is resolved. A goal itself does not have to include a measure or target necessarily but rather should provide a focus for steering toward a solution to the problem.

Objectives are more specific than goals and are ways of achieving the goals. There may be multiple objectives necessary to achieve a goal, and some objectives may include subobjectives. Whereas goals relate to the "big picture" or desired end result, objectives should be specific and measurable. An objective is not just a subgoal but provides a level of specificity necessary to implement broad-based goals fully. Structuring goals and objectives in a hierarchy can crystalize the nature of the problem and reveal any key objectives that may be missing or redundant.

12.3 Alternative Actions That Focus on Objectives

Defining alternative actions involves identifying feasible management actions that have the potential to address the objectives. Various alternative actions are reviewed in earlier sections. However, managers should not be constrained by what has worked in the past; this is the time to be creative in crafting alternative actions. This is an exercise in which alternative actions are considered in terms of how well they might meet the objectives relative to various criteria. Some actions may work well to meet some criteria (e.g., feasibility criterion) but not meet other criteria (e.g., affordability criterion). Similarly, some actions may work well in some reservoirs to meet some objectives but not work at all in others; for many actions it may not be known how well they will work. Defining alternative actions may need to involve input from partners and stakeholders with a diverse expertise base.

12.4 Scoring the Utility of Alternative Actions

Scoring is desired to narrow down the actions to a single action or to a set of actions that has the highest potential utility to meet the objectives. Scoring could be based on criteria considered important in the selection process. Table 12.1 lists a set of criteria that may help score the utility of alternative actions, but a smaller set or a different set of criteria may be more applicable depending on objectives and local conditions. The utility of each alternative action may be scored on, say, a 1 to 5 scale for each criterion, and a global utility value for each alternative action may be generated as the sum of the utility scores over all criteria scores. This global utility is then used to evaluate and select alternative actions. Some criteria for choosing among alternative actions may have different importance, as often occur with the affordability of an action. To account for such differences, summation over criteria to compute the global utility score may be weighted by differences in importance pre-assigned to each of the criteria (example in section 12.5).

Scoring each management action according to each criterion involves predicting the performance and consequences of each action, based on an understanding of the treatments and of the ecological and social systems affected, in terms of the objectives and local potential for success. Sometimes models, whether conceptual, quantitative, or expert based, are used to predict outcomes and consequences. Most commonly the utility score is based on the opinion of a group of agency personnel and outside experts, with input from partners and stakeholders if pertinent.

This evaluation in some cases will lead clearly to an optimal alternative action. But in most cases the predicted outcomes may display a complex mix of trade-offs so that no one action is clearly optimal. The complexity of the decision can be reduced by

Table 12.1. Example criteria for scoring utility of habitat management actions. Each criterion may be scored in a scale of 1-5 or other scale. See example in Table 12.2.

Criteria	Description
Applicability	Is the management action applicable for addressing the problem it seeks to address? Some management actions may work well in all settings, but others may be site specific. For example, habitat management through water level manipulation may be applicable in flood control reservoirs but difficult in navigation reservoirs
Efficacy	How well does the management action works? The fundamental question in evaluating efficacy is whether a beneficial effect of a management action has been demonstrated, and is better than doing nothing. The efficacy of interventions can be supported by observations by trained, knowledgeable, and experienced individuals. Consensus, among experts in a particular area, can always add information
Reliability	Does the management action produce consistent results? Reliable management actions are dependable and unfailing
Feasibility	Is the management action easy to administer and acceptable to stakeholders? Interventions that are difficult to administer may jeopardize successful outcomes and waste resources. The complexity of implementation will have implications for the form and cost of administration. Staff expertise and training may need to be considered
Affordability	Is the management action affordable? Affordability includes direct, indirect, short-term, and long-term costs, and should weigh the costs of withholding or delaying management intervention. Cost effectiveness associated with a management action may include prevention of future habitat problems, as when an early intervention with watershed erosion averts the need for sediment removal later on
Value-added	Does the management action produce other benefits? Value may be added when a management action makes other management actions unnecessary. For example, cost incurred in habitat management to improve cover for juvenile fish may also reduce the need for stocking, and thus cut hatchery costs. Moreover, some significant gains cannot be reduced to monetary savings, including the enjoyment that anglers and other users may gain from a shoreline stabilization or nutrient reduction program
Safety	Reservoir operators and boating program administrators may reject certain reservoir habitat enhancement actions (e.g., manipulation of reservoir elevation, installation of fish habitat structures, planting of macrophytes) because they may present a safety concern to boaters or reservoir operation
Durability	A concern may be how long-lasting a management action may be and how often it will have to be repeated (e.g., installation of fish habitat structures). Moreover, degradation of certain structures may result in fouling of outlet works or navigation hazards in the event that degradation causes debris to be released into the reservoir

identifying evaluation criteria over which alternatives do not differ and removing those criteria from consideration (e.g., if all alternatives are equally affordable, affordability can be removed as an evaluation criterion). Still, managers may be faced with a range of alternatives that have similar global utility, although they differ broadly in the utility score assigned to each criterion.

12.5 Example: Choosing Action for Dealing with Sediment

The problem in this example is the excessive sediment that has accumulated in the embayment of a reservoir that receives a major tributary draining an agricultural watershed. The sediment accumulation has resulted in loss of depth (section 3), has increased turbidity and other associated water quality problems (section 5), and has isolated backwaters (section 9). Altogether, sedimentation has reduced the ecological and aesthetic value of the embayment and the quality of recreational fishing opportunities. The goal is to improve fish habitat by regaining depth and to reduce influx of new sediment to curb future sedimentation. Specific objectives to achieve this goal include (1) remove sediment in shallow nearshore areas and the delta that has formed in the upper reaches of the embayment at the mouth of the tributary, and (2) keep sediment out of the upper reaches of the embayment. Additionally, the agency will intensify efforts to participate in watershed partnerships to try to influence long-term relief through structural, nonstructural, and regulatory land management practices, but those efforts are not part of this example.

Alternative actions being considered to address objective 1 include (1) excavating the sediment with heavy equipment (section 3.7.3.1); (2) dredging sediment with a hydraulic dredge (section 3.7.3.2); and (3) consolidating sediment through an extended drawdown (section 3.7.3.5). Actions being considered to address objective 2 include (1) constructing sediment basins (i.e., retention ponds) in the immediate approach to the embayment (section 3.7.1.1); (2) constructing a subimpoundment in the upper reaches of the embayment essentially to partition the embayment and create a separate marsh (section 3.7.1.2); and (3) constructing a channel that during high discharge bypasses the upper shallow reaches of the embayment and discharges into deeper water farther down the embayment (section 3.7.1.3).

Criteria were those listed in Table 12.1. Not all the criteria were assigned equal weights. Affordability was given a higher weight (0.3) than the other seven criteria (0.1 each). Utility of the six actions for reaching the objectives, given local conditions, was scored in an ordinal scale from 1 to 5, where 1 = least effective and 5 = most effective. Utilities were assigned based on expert elicitation with a team of agency and outside experts. Variability in expert opinion could be assigned to each utility score and considered in the decision but is not included in this example.

The six alternative actions selected to meet the two objectives had different strengths and weaknesses relative to the eight selection criteria (Table 12.2). It is noted that strengths and weaknesses are site specific because costs, applicability, and other criteria for scoring the alternative actions vary with locality. In this example, the best method for achieving objective 1 was excavation with heavy equipment and for objective 2 was development of retention ponds. Conceivably more than one method may be implemented, or Table 12.2 may be rescored with a combination of methods as the treatment. Table 12.2 provides the transparency necessary to document to administrators, partners, and stakeholders how decisions were reached and provides a vehicle for amending decisions as upgraded knowledge for rescoring criteria becomes available.

Working through an organized and agreed-upon process for choosing an action provides a transparent framework by which decisions on alternative actions can be made. In most cases, it is the process of bringing the entire decision team along the same educational route that permits consensus building. The actions table and associated utility scoring provide a framework by which to organize discussions. Invariably, some members will become engulfed by the scoring process. It is important that the facilitator keeps a perspective on the level of uncertainty associated with selecting alternative actions.

Table 12.2. Scoring the potential utility of management actions. Each criterion (see Table 12.1) was scored from 1 to 5 by a team of experts relative to the six management actions considered to achieve the two objectives. The unweighted global utility represents the average score over the eight criteria; the weighted global utility represents the average score weighted by the importance weights listed in parentheses (weights were derived through consultation with agency personnel, partners, and stakeholders).

Criteria (weight)	Objective 1 (regain depth)			Objective 2 (keep sediment off)		
	Excavate	Dredge	Consolidate	Retention pond	Sub-impoundment	Bypass channel
Applicability (0.1)	5	3	2	4	5	2
Efficacy (0.1)	5	4	1	5	5	3
Reliability (0.1)	5	4	2	2	3	2
Feasibility (0.1)	3	4	5	5	3	1
Affordability (0.3)	3	2	5	4	3	2
Value-added (0.1)	2	2	1	4	5	1
Safety (0.1)	5	5	4	5	5	4
Durability (0.1)	5	5	1	5	5	5
Unweighted global utility	4.1	3.6	2.6	4.3	4.3	2.5
Weighted global utility	3.9	3.3	3.1	4.2	4.0	2.4

Section 13

Stakeholder Engagement

13.1 Stakeholders as Partners

Many habitat management issues usually involve public resources, so the general public and various organizations have a vested interest in the outcomes of decisions. Natural resources also have multiple uses that can lead to competition among user groups and potentially conflict among groups. By implementing a stakeholder-driven process, managers can help those with vested interests understand how decisions are arrived at and participate in the decision-making process.

Stakeholders are defined here as groups, organizations, or individuals with some level of vested interest (Watt 2014) in the effects of fish habitat management. These groups generally include consumers (e.g., anglers, boaters, citizens interested in natural resources, lake associations); nongovernmental organizations (e.g., Bass Anglers Sportsman Society, The Nature Conservancy); natural resource management agencies (e.g., state or federal agencies charged with managing some aspect of a reservoir resource); political representatives, elected or appointed, to local, state, or federal governments; and economic interests such as local business or landowners potentially affected by activities associated with a reservoir. This list can change or shift in emphasis depending on the nature of habitat alteration. For example, changes to water regime may be of interest to a different set of stakeholders than installation of reefs and structures.

Stakeholders can be a vital part of the process of identifying management goals. While often it is possible to anticipate and identify goals without including stakeholders, involving them will typically assure a more comprehensive analysis of goals and also will help managers anticipate and, possibly, resolve or at least minimize potential conflicts among competing user groups. Finally, and perhaps most importantly, participation builds public support and ownership of the decision when the public is explicitly involved in the decision-making process (Watt 2014).

13.2 The Pitfalls

Working with stakeholders can involve controversy as different stakeholders may have different objectives (Townsley 1998). Often stakeholders may be members of more than one group and, hence, are likely to have multiple objectives. For example, an angler with recreational objectives may also be a fishing guide or a bait-shop owner with economic objectives. Similarly, an agency staff member may also be an angler. The potential downside of including stakeholders with multiple group memberships is that a stakeholder may portray him or herself as representing one stakeholder group when many of his or her objectives may reflect the other group, which can lead to conflict. The benefit is that such an individual may facilitate communication and understanding among different stakeholder groups.

All stakeholders are not necessarily equal. For example, all decision makers are stakeholders in that they have a vested interest in the outcome of a management decision. However, not all stakeholders are decision makers. Decision makers often have the legal authority or accountability for carrying out a management action, such as providing funds or personnel and equipment. Thus, decision makers generally have greater responsibility and accountability than other stakeholders. So the question is, what stakeholders should be involved in the process and how much weight should the group be granted?

13.3 Need for Gaging Stakeholder Importance

A preliminary stakeholder analysis may be used to prioritize people, groups of people, or organizations that significantly may influence or be influenced by the management decision. One approach is to develop a matrix to evaluate stakeholder importance (Freeman 2010). Potential stakeholders can be scored in a matrix based on the relevance of the potential management action to the stakeholder (i.e., likelihood that the stakeholder will be affected by a decision) and the perceived ability of the user to affect policy decisions (i.e., likelihood that the stakeholder will affect the decision) (Allen and Miranda 1997). Potential stakeholders that are strongly affected by a decision or have a strong effect on the decision are essential to involve in arriving at a decision. Conversely, it is not important to involve those that are not affected by the decision or have little or no influence on the decision.

The first step in creating such a matrix is to develop a list of potential stakeholders, including as many as possible without limiting the list to those initially perceived to be important. A useful set of questions one may ask to identify stakeholders might include (1) what groups potentially will be affected by the decision; (2) what groups often are involved in these decisions; (3) who has the knowledge of how the system works (e.g., biologists, engineers); (4) who has the legal authority to approve or implement management actions; and (5) who can potentially overturn the decision. Once the matrix is created, the rankings are examined to distinguish the stakeholders that are essential to involve in the decision-making process from those that are desirable (Table 13.1).

Table 13.1. Example stakeholder analysis matrix for a situation in which a state fishery conservation agency is considering sediment removal from a large arm of a reservoir as a way to improve fish habitat. To remove sediment, water level in the lake will be lowered for about 1 year. The matrix lists potential stakeholders and their likelihood of affecting or being affected by the sediment removal. Three groups ranked high on both counts, and four others ranked high on one count. These seven groups may be included as stakeholders and the others excluded or their input down-weighted. NGO = nongovernmental organization.

Potential stakeholder	Likelihood that stakeholder will affect decision	Likelihood that stakeholder will be affected by decision
Local anglers, boaters	Medium	High
Local business (bait shops, restaurants, campgrounds)	Low	High
Local fishing clubs	Medium	High
Lake association	High	High
Agency with control over reservoir facilities	High	High
U.S. Fish and Wildlife Service	Medium	Low
Department of Environmental Quality	High	High
Department of Tourism	Low	Low
Relevant NGOs (e.g., The Nature Conservancy, watershed groups)	Low	Low
Key local or state politician	High	Low

13.4 Disclosure of Who Decides

An important consideration when working with multiple stakeholders is identifying clearly beforehand to all participants the way stakeholders interact in the process. This establishes ground rules by which everyone agrees to abide. Participants need to know before they commit to participate whether the final decision will be made by full consensus or simple majority or whether the participant is just playing a consultative role and the final decision will be made by an administrator after hearing from all stakeholders involved. The latter is the most common form of decision making in natural resource management. The advantages of this form of decision making is that the input can help the decision makers better understand the issues and stakeholders generally appreciate an opportunity to voice their opinions. The disadvantage is potential conflict as some stakeholders may expect that their wants and needs will be included in the final decision, and they may not be.

13.5 Stakeholders as Supporters

Rarely can a single fishery management agency alone maintain or enhance fish habitat in a large reservoir. Even if it could, the agency will never be as successful as it would by enlisting help, such as from a reservoir association. The resources available to fishery management agencies are limited and unlikely to be sufficient to provide the comprehensive habitat maintenance or rehabilitation many reservoirs require. Access to local contacts can support and strengthen the manager's ability to navigate local political processes and enhance the odds of success. Managers seeking help in conducting habitat enhancement projects can obtain it from a reservoir association, which generally include homeowners, landowners, business owners, concerned anglers, and conservation-minded people. Increasingly, attention is being given to reservoir associations or "Friends of Reservoirs" groups. Having such an organization set up for each reservoir, cluster of reservoirs, or key reservoirs can enhance project success, expand the scope of projects, provide fundraising capability, and provide influence when needed. Such an organization also may serve as an informal advisory board if such a function is needed. An agency may need advice on how to implement a habitat management plan, particularly when various options are available. In many cases, habitat management could benefit from the diversity of opinions and experiences that a supporting group can contribute. Greater support, and thus likelihood of success, will occur when including reservoir associations in the planning process and incorporating their vision into the final product.

If such a group is not already in place, a valuable management practice is to help organize an effective group of supporters (IDNR 2010). With such an organization available, the reservoir manager creates the ability to initiate projects and programs of long-term significance that will lead to healthier fish habitat. An organized group can have more credibility and clout than any one individual. Such a group creates opportunities for public education and information. Also, the circle of influence and contacts of the individual members of homeowners' associations can be invaluable and make all the difference.

13.6 Reservoir Association Establishment

To get started the manager may identify individuals or groups that may have an obvious connection to or interest in the reservoir's fish habitat, reach out to their leadership, gauge their interest, and try to get them to sign on as supporters. Then, if fitting, a public meeting may be organized to recognize supporters already on board and to garner more supporters (IDNR 2010). The public meeting may be advertised to target local groups that fish the reservoir, property owners, and other local groups interested in protecting the reservoir. Invite lakeshore residents, local fishing clubs, and conservation groups. It is also helpful to enlist the help of members of other reservoir associations in your area, if they exist, as members can provide valuable organization information based on their own experiences. Discuss the issues that are important to the group with a focus on habitat. Try to hone in on a goal that most participants can support and that will become the foundation for the organization. Assistance in creating a citizens' group can be obtained through contacting the Reservoir Fisheries Habitat Partnership (www.reservoirpartnership.org) and the Friends of Reservoirs Foundation (www.waterhabitatlife.org). Membership in these organizations can help with fundraising (offering a membership in a 501(c) (3) corporation) by providing a tax deduction for locally raised funds, technical assistance with projects, networking with other Friends of Reservoirs groups, and access to grants.

13.7 Reservoir Association Structure

Ideally the organization will manage itself with only limited and occasional input from the reservoir manager. To this end the organization may elect a board of directors that will serve as a point of contact. For a large reservoir, the board of directors may come from different parts of the region. This strategy can help increase membership and can be a time saver when getting out information or organizing work projects. A set of bylaws may be drawn, including a mission statement. This will allow for smoother operation as the association becomes involved in important issues and meaningful projects (Lyden et al. 2006). The organization may wish to file the necessary papers required for a nonprofit organization, as this will allow the organization to become involved in fundraising activities and be eligible for grants and other funding opportunities. While funding can be important, volunteerism and commitment to a task are usually the most valuable assets (Lyden et al. 2006).

As needed, the organization may form committees to tackle the various functions, goals, and projects identified as important (Lyden et al. 2006). Most associations will have committees dealing with membership and different projects. It is important that committees are headed by devoted and enthusiastic people with a track record of getting the job done. A newsletter, website, and social media platforms are key to keeping everyone informed about activities, upcoming events, and other issues and topics that are of importance to the group, ultimately helping to maintain the group's support and passion for the mission. These vehicles are also great ways to educate the group about habitat management issues.

13.8 Reservoir Association Projects

Various needs listed throughout this document require a greater workforce than the reservoir management crew has available. For example, assistance may be needed with mapping problem areas in the riparian zone and adjacent watershed to expand the watershed inventory outlined in section 2 and Table 2.1. Some basic monitoring may be conducted by members of the organization who spend a great deal of time at the reservoir, such as adjacent property owners. Projects such as installing fish attractors, transplanting aquatic vegetation, and seeding mudflats require workers and equipment not always available within fish management organizations. Last, reservoir associations may be able to organize the public and political support necessary to accomplish some habitat projects.

To be able to identify potential sources of sediment, nutrient loading, and pollution it is vital to have a good map of the riparian zone and adjacent watershed (sections 2 and 8). A geographic information system map may be obtained from or created by a government organization, but it is important to have the help needed to go out in the field to verify its important features, identify land uses, and update the map. It is valuable to note key sources of direct and indirect surface runoff and nutrient loading, such as wastewater treatment plants, septic systems, storm sewers, drainage ditches, agricultural drain tiles, parking lots, new construction, road building, agricultural row-crop fields, and feedlots. The assessment also may identify areas of special concern around the shoreline that need remedial action. Items may include eroding shorelines, straight pipes into the reservoir, areas of dense vegetation, or areas devoid of vegetation. Note well-manicured lawns that may be heavily fertilized and areas of greenness that may reveal failing septic systems. Ongoing construction, dumps, livestock access to the water, open burning, erosion, and any other activity associated with sediment or nutrient loading could be noted in this assessment. Ground-truthing watershed inventories is an excellent activity for getting organization members involved in meaningful projects.

Figure 13.1. Volunteers assisting with a habitat enhancement program in Grand Lake O' the Cherokees, Oklahoma. Photo credit: Grand River Dam Authority, Langley, Oklahoma.

A simple means of assessing a reservoir's water-quality status is by monitoring the water clarity with a Secchi disk (section 5.4). Over a period of years, temporal and spatial trends of water clarity within the reservoir will develop and will reflect changes in water quality over time. Other water-quality needs may be handled by the organization if provided with equipment and training. A "lake watch" program can be set up in which volunteers monitor water clarity, temperature, and dissolved oxygen. Often with a little training and provision of proper equipment volunteers can provide a valuable service and free up agency staff to focus on other tasks.

Figure 13.2. Volunteers assisting with husbandry of aquatic plants. Photo credit: J. Boxrucker.

Most reservoirs benefit from fish habitat enhancement or restoration programs if suitable habitat is not available or is degrading. Such programs sometimes require introduction and maintenance of a large volume of structure. Access to volunteers already organized into a reservoir association can facilitate and even expand existing programs (Figure 13.1).

Aquatic plant management requires control of undesirable aquatic plants (e.g., nuisance, invasive, or excessive aquatic plant growth) or establishment of desirable species. Both of these activities may require substantial help from outside groups (Figure 13.2). This is particularly true for projects designed to establish aquatic plants because facilities and personnel are needed to grow propagules, and a large number of volunteers may be needed to plant them and build exclosures (section 11).

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Appendix

Ballpark costs of reservoir fish habitat management practices of three types: N = non-structural, S = structural, and R = regulatory (defined in section 2.7.5). Costs are listed over categories. The least expensive ($) treatment may cost nothing or under 1 thousand U.S. dollars. The most expensive ($$$$$) treatment may cost over 1 million U.S. dollars. Treatments in between represent approximately 1-10 thousand ($$), 10-100 thousands ($$$), and 100 thousands to 1 million ($$$$). ID = identification number for cross reference within Appendix. Additional cost estimates specific to reducing nutrient pollution at its sources were compiled by the U.S. Environmental Protection Agency (USEPA 2015).

ID	Management practice	Type	$	$$	$$$	$$$$	$$$$$
	Watershed						
1	**Watershed management to reduce sediment and nutrients**						
2	watershed inventory	N	●	●	●		
3	partnering with watershed organizations	N	●	●			
4	structural practices	S		●	●	●	●
5	non-structural practices	N	●	●	●		
6	regulatory practices	R	●	●			
	Sedimentation						
7	**Monitoring sedimentation**						
8	sediment cores	N	●	●			
9	topography contrast	N	●				
10	acoustic estimation	N	●	●			
11	**Reduction of sediment inflows**						
12	sediment basins	S		●	●		
13	sediment dikes	S		●	●		
14	bypass channel	S			●	●	
15	**Sediment management in the reservoir**						
16	sluicing	N		●			
17	density current venting	N		●			
18	**Removal of sediment from the reservoir**						
19	excavation	N		●	●	●	●
20	dredging	N		●	●	●	●
21	hydrosuction	N		●	●		
22	consolidation	N	●				
23	flushing	N	●				
24	small-scale removals	N	●	●			
	Eutrophication						
25	**Monitoring program**	N	●				

ID	Management practice	Type	$	$$	$$$	$$$$	$$$$$
26	**Watershed remediation**						
27	watershed management (see 1)	N					
28	constructed wetlands	S		•	•		
29	pre-dams	S			•	•	
30	**In-lake remediation**						
31	guide curve revision	N	•	•			
32	inflow routing	N	•				
33	dilution	N	•				
34	flushing (also 23)	N	•				
35	selective withdrawal	N	•				
36	hypolimnetic aeration and oxygenation	S		•	•		
37	sediment removal (see 18)	N					
38	sediment drying (see 22)	N	•				
39	phosphorus precipitation and inactivation	N		•	•		
40	**Biomanipulation**						
41	fish populations	N	•	•			
42	fish harvesting	N	•	•			
43	macrophytes (see 150, 153, 159)	N	•	•	•		
44	floating wetland islands	S	•	•			
	Water Clarity						
45	**Monitoring considerations**	N	•				
46	**Shore erosion control**						
47	introduction of shore vegetation (see 145, 170)	N	•	•			
48	vegetation protection	N	•	•	•		
49	offshore breakwaters	S		•	•	•	
50	rock-log structures	S	•	•			
51	floating breakwaters	S		•	•		
52	groynes	S	•	•			
53	rock vanes	S	•	•			
54	natural stone revetment	S		•	•	•	
55	bulkheads and seawalls	S		•	•	•	
56	living shorelines	S		•	•		
57	tree felling	S	•	•			
58	sills	S		•	•		
59	boat traffic ordinances	R	•				
60	**Flocculation to improve water clarity**						
61	organic matter	N	•	•			
62	electrolytes (also 39)	N		•	•		

ID	Management practice	Type	$	$$	$$$	$$$$	$$$$$
63	**Fishery management to improve water clarity**						
64	harvest of benthivores	N		•	•		
65	eradication of benthivores	N		•	•		
66	manage for turbid-water fish	N	•	•			
	Water Quality						
67	**Monitoring considerations**	N	•	•			
68	**Destratification**						
69	diffused aeration	S		•	•		
70	mechanical flow pumps	S		•	•		
71	solar and wind technology	S		•	•		
72	**Hypolimnetic aeration-oxygenation**						
73	bubble plume	S		•	•	•	
74	diffusers	S		•	•	•	
75	submerged contact chamber	S			•	•	
76	hypolimnetic withdrawal	N	•				
77	**Guide curve management for water quality**						
78	modification to maximize flow	N	•				
79	modification for multi-reservoir operation	N	•	•			
80	**Contaminants management**						
81	amendments	N		•	•	•	
82	capping approach	S		•	•		
83	phytoremediation	S		•	•		
	Water Regime						
84	**Monitoring program**	N	•	•			
85	**Timing of flood**						
86	high water in spawning-growing season of fish	N	•				
87	low water in growing season	N	•				
88	winter drawdown	N	•				
89	**Periodicity**						
90	variability in timing	N	•				
91	extended flood	N	•				
92	long drawdowns and floods	N	•				
93	**Rate**						
94	slow drawdowns	N	•				
95	long-term view	N	•				
96	**Vegetation protection and establishment** (see 170)						
97	**Enhanced aesthetic value**	N	•				
98	**Drought-related problems and opportunities**	N	•				

ID	Management practice	Type	$	$$	$$$	$$$$	$$$$$
99	**Interagency cooperation**	N	•				
100	**Guide curves**	R	•	•			
101	**Amendment of guide curves**	R	•	•	•		
	Riparian Zone						
102	**Width**	R	•				
103	**Three-zone buffers**	S		•	•		
104	**Livestock**	R	•				
105	**Bank stabilization**	S		•	•	•	
106	**Tree felling**	S	•	•			
107	**Residential development management**	R	•				
108	**Drawdowns**	N	•				
109	**Conservation easements**	R	•				
	Lateral Connectivity						
110	**Maintenance of lateral connectivity**						
111	submersed check dams	S		•	•		
112	connection channels	S		•	•		
113	water level manipulations	N	•				
114	levee setbacks	S			•		
115	closure structures	S		•	•		
116	notched dikes	S		•	•		
117	culverts	S		•	•		
118	**Maintenance of adjoining habitat**						
119	weirs	S		•	•		
120	pump systems	S		•	•		
121	excavated pools	S	•	•			
	Artificial Reefs and Structures						
122	**Justification and evaluation**						
123	justification of needs	N	•				
124	clearly stated goals	N	•				
125	evaluation of performance	N	•	•			
126	**Reef site selection**						
127	incompatible sites	N	•				
128	compatible sites	N	•				
129	substrate	N	•				
130	depth	N	•				
131	floating reefs	S	•	•			
132	waves	N	•				

ID	Management practice	Type	$	$$	$$$	$$$$	$$$$$
133	**Reef design**						
134	reef area	N	•				
135	permits and regulations	R	•				
136	**Reef construction**						
137	tree, brush and lumber reefs	S	•	•			
138	stone materials	S	•	•	•		
139	synthetic materials	S	•	•	•		
140	**Reef installation**	S	•				
141	**Reef markers**	S	•	•			
142	**Maintenance and monitoring**	N	•	•			
143	**Brochures and electronic media**	N	•				
144	**Volunteer assistance**	N	•				
	Aquatic and Terrestrial Plants						
145	**Aquatic plant establishment**						
146	plant selection	N	•				
147	source of propagules	N	•				
148	propagule production	S	•	•	•		
149	plant establishment	S	•	•	•		
150	multiple depth planting	N	•				
151	post-planting monitoring	N	•				
152	**Biological control of aquatic plants**						
153	grass carp	S		•	•		
154	insects	S		•	•		
155	**Mechanical and physical aquatic plant control**						
156	hand pulling	N	•				
157	hand cutting	N	•				
158	hand rakes	N	•				
159	mechanical harvesters	N		•	•		
160	track hoes and draglines	N		•	•		
161	legalities of collection and transportation	R	•				
162	water drawdown	N	•				
163	**Chemical control practices**						
164	herbicide use and classification	N		•	•		
165	selectivity	N	•				
166	control of specific plants	N	•				
167	**Cultural control practices**						
168	prevention	N	•	•			
169	education	N	•	•			

ID	Management practice	Type	$	$$	$$$	$$$$	$$$$$
170	**Promotion of terrestrial plants in barren shorelines**						
171	well-vegetated riparian zones as colonization sources	N	•				
172	seeding	S		•	•		
173	timing	N	•				
174	**Seeding methods**						
175	broadcasting	N	•	•			
176	drill seeding	N		•	•		
177	hydroseeding	N		•	•		
178	aerial seeding	N		•	•		
179	**Transplanting**	S	•	•			
180	**Grasses and other herbaceous plants**						
181	sprigs	S	•	•			
182	rootstocks and plugs	S	•	•			
183	rhizomes and tubers	S	•	•			
184	**Trees and shrubs**						
185	bare-root seedlings	S	•	•			
186	cuttings	S	•	•			
187	balled-and-burlapped propagules	S	•	•			
188	containerized propagules	S	•	•			
189	spacing	N	•				
	Habitat Management Decisions						
190	**Problem, goals, and objectives**	N	•				
191	**Alternative actions**	N	•				
192	**Scoring the utility of alternative actions**	N	•				
	Stakeholder Engagement						
193	**Stakeholders as partners**	N	•				
194	**Gaging stakeholder importance**	N	•				
195	**Disclosure of who decides**	N	•				
196	**Stakeholders as supporters**	N	•				
197	**Reservoir associations**	N	•				

Abbreviations

ac	**acre**
ac-ft	**acre-foot**
cfs	**cubic feet per second**
d	**day**
ft	**foot**
h	**hour**
in	**inch**
lb	**pound**
mi	**mile**
min	**minute**
mph	**miles per hour**
ppb	**parts per billion**
ppm	**parts per million**
sec	**second**
t	**ton**
yd	**yard**