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CHAPTER 8

# *Management of Natural Palustrine Wetlands*

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## Introduction

Palustrine wetlands, like all aquatic ecosystems, are inextricably linked to the upland habitats that surround them. Attempts to understand these systems become even more complex when issues of scale (e.g., spatial and temporal scales) are applied to ecosystem processes (Gelwick and Matthews 1990, Schramm and Hubert 1996). Aquatic systems act as integrators of terrestrial ecosystem characteristics (e.g., catchment size, slope, soil) and activities (e.g., land use) and thus can be indicators of changes in environmental quality. In this chapter we adopt a landscape-level approach that addresses causes, in addition to symptoms, of problems in management of palustrine wetlands. Because of the general aridity of the environment, wetland management is particularly important and problematic in the Intermountain West. However, the Intermountain West also offers unique opportunities for landscape-level management of wetlands because large-block land ownership patterns are conducive to manipulating landscape and regional inputs as well as local inputs to wetlands.

## Timber Harvesting and Road Building

### *Impacts of timber harvesting and road building on palustrine systems*

Timber-harvesting and road-building activities can have a substantial impact on aquatic systems, although research specifically focused on the Intermountain West lags behind that occurring in either coastal region of North America. Differences in watershed responses to timber harvesting among regions can be attributed to differences in climate, topography, soil and geology, and vegetation (Burns 1972, Bormann et al. 1974, Scrivener 1982, Swank and Crossley 1988). However, while there are region-specific differences in effects (i.e., specific organisms, degree



of perturbation), the fundamental physical processes that regulate environmental effects are manifested in similar ways among regions (i.e., operative physical processes are universal) and thus can be generalized at a coarse scale.

Timber harvesting involves the interaction of a complex set of anthropogenic activities and ecological and hydrological processes. The physical activities involved in timber harvesting include the felling, skidding, and transporting of logs; road construction; slash removal; site preparation (scarification, herbicide application, burning, etc.); and site rehabilitation including seeding or planting (Haupt and Kidd 1965, Miller 1984, Holopainen et al. 1991). The effects of timber harvesting actually result from many of the activities associated with logging beyond actual cutting and removal of trees. For instance, skidding logs can destroy other vegetation that helps prevent soil erosion, while heavy equipment compacts soil, increasing runoff rates and exacerbating erosion (Johnson and Beschta 1980, Clayton 1990). However, the effects of timber harvesting on aquatic systems often are discussed synonymously with other activities such as road building because the activities often occur together. Yet their operative processes differ in how they elicit effects in upland areas and how these effects are transferred to and manifest themselves in aquatic systems. In many cases, disturbances from these additional activities are the source of many environmental effects in watersheds and receiving waters.

Timber harvesting and road building are major influences on hydrology of catchments (Rothacher 1970, Likens et al. 1977, Swank and Crossley 1988); effects to aquatic systems cascade from these hydrologic responses. Increased runoff results in increased sediment, nutrient, and organic transport (Patric and Reinhart 1971, Hartman and Holtby 1982, Hall et al. 1987), which affects water chemistry (Bormann et al. 1974, Nicholson et al. 1982), productivity (Webster et al. 1988), fish habitat (Hall et al. 1987), and abundance and community composition of various taxa (Burns 1972, Murphy and Hall 1981, Hall et al. 1987). Other consequences of timber harvesting unrelated to hydrological processes include thermal changes, resource access, and habitat fragmentation (Franklin and Forman 1987) and changes in land use of large blocks of land near wetland complexes.

Timber-harvesting activities universally increase water yield in catchments, which is the most predictable response (Hibbert 1967, Harr et al. 1975, Likens et al. 1977, Hornbeck et al. 1986, Hetherington 1987, Bonell 1993). However, the actual water yield within a catchment differs locally based on differences in climate, topography, and native vegetation. Increased runoff after timber harvesting results from decreased interception of precipitation onto vegetation and into soil (Harr 1977), reduced evapotranspiration, and reduced landscape roughness. Timber harvesting exposes forest soils directly to erosive forces of precipitation and overland flow. Increased water yield increases the transport energy of moving water, which leads to increased erosion. Moreover, reduced vegetative cover increases the erosional power of water at the point of impact of water on bare soil where mobilization can occur. The link between water yield and erosion is clear: exposed or unvegetated soil in upland areas is a source of sediment during rainfall or spring runoff. Heavy rainfall saturates the soil, making it transportable via overland flow, while erosive forces increase considerably once flow becomes channelized in rills, ditches, and even streams. The riparian-wetland ecotone is critical in evaluating how specific sites will be affected, as it is the transfer point between upland and aquatic environments. Upland and riparian areas that have steeper slopes or more channelized drainage patterns increase transport potential of sites. The amount and location of natural areas positioned between these sites and wetlands reduces transport potential. As such, riparian zones buffer transport of terrestrial material (e.g., soil, leaves, needles, woody structure) to wetlands (see Chapter 5).

Harvesting in watersheds can directly alter soil structure through disturbance and compaction by heavy machinery. Moreover, tree cutting eliminates root structure that binds and stabilizes soils, while the rate at which precipitation is intercepted by the forest canopy decreases, thus increasing runoff. As runoff increases, the amount of sediment being transported also increases. Effects of increased suspended sediment include increased turbidity, reduced dissolved oxygen—particularly where high amounts of organic material are transported—and abrasion to gills and even suffocation of fish (Newcombe and MacDonald 1991, Bozek and Young 1994). Increased levels of sedimentation increase rates of eutrophication, increase turbidity, decrease oxygen levels, and increase nutrient loadings of aquatic systems that alter ecological, biological, and physiological processes. Organic soils can reduce oxygen by increasing the biological oxygen demand (BOD) and decreasing photosynthetic production of oxygen if transported sediment remains in suspension for prolonged periods.

Depending upon soil characteristics and chemistry, mobilized soils can transfer a variety of nutrients and contaminants (Bormann et al. 1974, Norris et al. 1991). While granitic soils of some intermountain drainages are nutrient-poor, thin topsoil layers and duff contain nutrients and organic matter that can be transported to wetland complexes. Increased nutrients can increase primary productivity and lead to nuisance blooms of noxious blue-green algae and macrophytes. In general, however, nutrient loadings in runoff can increase after timber harvesting, but effects are usually ameliorated in fewer than 8 years (Scrivener 1982).



Changes in hydrographs associated with increased water delivery also affect depth, storage, and quality of water in wetland complexes. More overland flow results in quicker and more pronounced peaks in hydrographs and lower water levels during drier periods. Moreover, increased runoff decreases infiltration and recharge of aquifers that can result in lower discharge rates from springs that feed wetlands (Freeze and Cherry 1979).

The principal source of thermal inputs to most aquatic systems is direct solar radiation (Hynes 1970, Barton et al. 1985), although reduced groundwater discharge caused by declines in aquifer recharge rates (Freeze and Cherry 1979) also can influence temperatures. Temperature changes, particularly on smaller systems, can be dramatic as a result of timber harvest, while larger systems, with less initial shading by riparian vegetation and greater thermal mass, are affected less.

### *Impacts of road building on palustrine habitats*

Road building can have both direct and indirect effects on palustrine habitats. Roads are the largest source of sediment transported to aquatic systems (Swanston and Swanson 1976, Beschta 1978). In fact, erosion from roads contributes more sediment per unit area than all other forest activities (see Gibbons and Salo 1973). Sediment associated with road development and maintenance comes from road surfaces, drainage ditches along roads, and culverts or drainage crossings. Soil on roads is exposed continuously to hydrologic forces, making them highly susceptible to erosion. Moreover, road grading makes additional sediment available for transport, and drainage ditches along roads are among the most hydrologically active areas in forested environments. Where drainage patterns require culverts, proper design and regular maintenance are essential (McCashion and Rice 1983, Ontario Ministry of Natural Resources 1990). Improper drainage also can disrupt hydrological processes important to wetland functions, resulting in direct loss or impaired use of habitat. Other important effects of road building are the filling of wetland habitats during road construction, causing direct loss of wetlands, disruption of drainage patterns, and habitat fragmentation.

A road crossing is often the point where effects from upland land use activities are transferred to aquatic systems. Construction of roads accelerates erosional processes in watersheds by exposing soil on roads and adjacent ditches to direct erosion and by altering the natural drainage networks in watersheds (Beschta 1978, Reid and Dunne 1984). In particular, hydrologic runoff patterns are altered (i.e., from sheet to channel flow) and hydraulic forces are concentrated, both of which increase rates of erosion.

### *General best management practices for timber harvesting and road building*

In assessing effects of timber harvesting, attention needs to be paid to both scale issues and techniques to minimize impacts to aquatic systems (Heede and King 1990). In selecting best management practices (BMPs) that ameliorate the effects of timber harvesting on palustrine habitats, several principles apply. First, the best practices are those that are preventive (designed to reduce the incidence of a problem at its source) rather than remedial (intended to mitigate the effects after they occur). Second, techniques need to be based on site-specific conditions such as soil type, disturbance level, topography and specific slope, proximity to wetland complex, and overall management objectives. Third, several alternatives may produce the same result, and combinations of practices often are better than single approaches. Fourth, excluding timber harvesting or road building may be the best management practice for some areas.

Best management practices should be planned with resource managers. Careful attention should be paid to the design of any project at landscape and local scales. During planning, roads should be diverted away from sensitive environmental habitat or positioned to minimize environmental consequences. Often, less damaging alternatives can be found before plans are set, projects bid out, and activities begun. Changes to plans after activities start are costly and might be viewed as obstructionist in extreme cases. Sound evidence presented to resource managers in advance can reduce cost and protect resources without compromising the integrity of the aquatic environment.

Use of buffer reserves in riparian areas next to aquatic systems is the most widely recognized BMP (Ontario Ministry of Natural Resources 1988). This approach tends to be remedial by reducing sediment transport after it has begun rather than stopping transport before it starts. Many states and provinces have guidelines for design of riparian buffer widths, but in general, recommended reserve widths are a function of soil particle size, land slope, and vegetation. These guidelines stipulate that riparian buffer corridors along or around aquatic areas be extended to a distance (width) whereby transported sediment gets filtered by natural vegetation. These distances vary among governmental agencies because research has been equivocal in designating a "catch-all" buffer width. Climatological, geographical, and vegetational differences play a key role in observed differences in research results.

Some of the earliest and most convincing research on sediment transport (Trimble and Sartz 1957) showed that steeper slopes of undisturbed forest transported sediment farther and thus required greater buffer widths. Minimizing site disturbance, avoiding highly erodible sites, and



revegetating sites also help reduce the amount of sediment moving from harvested areas. In general, increased slope and decreased vegetation cover and soil particle sizes increase transport risk. Generalized buffer reserve widths are as follows: 30 m for slopes 0–15%; 50 m for slopes 16–30%; 70 m for slopes 31–45%; and 90 m for slopes 46–60%. The best guidelines to apply are those developed for local conditions. Of critical importance in protecting wetlands is determining what should be protected and where the buffer reserve should be placed so that all transported material is stopped before reaching the area of concern.

Thermal effects to wetlands resulting from timber harvesting are not well quantified. Most work has been conducted in small streams where the ratio of surface water to riparian interface is highest, and significant thermal buffering has been demonstrated (Barton et al. 1985, Campbell and Doeg 1989). But buffer reserves set up to minimize transport of sediment, nutrients, and organic matter to wetlands generally will provide adequate thermal buffering.

Constructing roads that have no effects on the environment is difficult. Rather, effort should be placed on designing roads that minimize their exposure to sensitive aquatic areas, using proper design and best management practices, and maintaining currently used roads or removing and rehabilitating discontinued roads. Most publications available from resource management agencies on implementing BMPs have not been researched or peer-reviewed (Ontario Ministry of Natural Resources 1990). Many prescriptions proposed in these documents have not been demonstrated to be effective and in some instances might cause more harm than good. This underscores the need to work directly with hydrologists or engineers who specialize in this type of work when undertaking these projects.

Best management practices for forest road construction include techniques that minimize proximity of roads to wetlands, route the increased runoff away from drainage ditches to reduce transport energy, and directly minimize sediment losses from erosion from roads (Rothwell 1978, Ontario Ministry of Natural Resources 1990). The following guidelines can be applied in most cases:

- Minimize proximity of roads, road crossings, and road drainages to wetlands, particularly higher-quality wetlands or sensitive areas
- Avoid road construction routes that occur in areas having high topographic relief or areas having highly erodible soils. Gentle grades <4% are most desirable, and shorter grades are better than longer grades
- Use large particle sizes for rip-rap in drainage ditches and along road crossings to minimize erosion
- Minimize the number of roads

- Design enough culverts and road crossings to handle runoff without causing erosion; the design should be adequate for runoff to pass without damming flows
- Include buffer reserve areas downslope of road construction and drainage
- Disperse runoff drainage rather than concentrate it, thereby minimizing soil transport power
- Encourage vegetative stabilization
- Conduct construction during times that minimize effects to soil movement and to organisms near the road

## Grazing

### *Impact of grazing on palustrine systems*

Grazing is the dominant land use in the Intermountain West, with about 70% of the 11 westernmost states grazed by livestock (see Fleischner 1994). Even though the effects of grazing have been extensively studied (see Vallentine 1990, Armour et al. 1994, Kie et al. 1994, Fleischner 1994, Payne 1998, Payne and Bryant 1998), its impacts are hard to generalize, as they vary with location, stocking rate, season, duration of grazing, topography, management practices, precipitation, and species of grazer. Grazing *per se* is not necessarily harmful, particularly in plant communities that evolved with herbivory, and the disturbance associated with limited grazing can increase diversity (Archer and Smeins 1991, Hobbs and Huenneke 1992). In addition, stock tanks and windmills built for cattle provide water and habitat for a vast array of wildlife (Crawford and Bolen 1976, Evans and Krebs 1977, Menasco 1986, Payne and Bryant 1998; also see Chapters 9, 10, and 11).

However, cattle spend a disproportionate amount of time in riparian areas (Ames 1977, Goodman et al. 1989), and the impact of grazing on wetlands can be devastating. These impacts can be direct (e.g., herbivory of wetland vegetation) or indirect (e.g., altered hydrologic regimes caused by reduced vegetative cover and soil permeability of grazed lands).

Grazing reduces emergent and wetland margin vegetation, as well as shrubs and trees associated with wetlands (Glinski 1977, Thomas et al. 1979, Kauffman and Krueger 1984). This reduces fawn-rearing and thermal cover for mule deer (Leckenby et al. 1982). Similarly, decreased nesting success of sandhill cranes in grazed wetlands in southeastern Oregon was attributed to reduced vegetative cover, easy predator access, altered predator communities, and altered prey base associated with high cattle stocking rates (Littlefield and Paullin 1990).

Wet meadows are preferred brood-rearing areas for sage grouse, as these meadows contain succulent forbs required by chicks (Klebenow 1982).



Excessive grazing reduces available vegetation in wet meadows (Call and Maser 1985), which are subsequently avoided by sage grouse (Klebenow 1982). In addition, wet meadows can also dry up from development of cattle-watering tanks at springs that feed the meadows (Thomas et al. 1979).

Management of surrounding uplands for cattle production can impact both water quality and quantity. Ethyl parathion, an organophosphorus insecticide often used to control grasshoppers on rangelands and croplands, can enter wetlands via runoff or spray drift. Ethyl parathion is toxic to waterfowl (Smith 1987) and aquatic invertebrates (Mayer and Ellersieck 1986, Tome et al. 1991). Water quantity is altered as grazing of uplands reduces ground cover and compacts soil, which decreases infiltration and increases surface runoff (Packer 1953, Sharp et al. 1964, Lusby 1970). Consequently, summer flow of water into wetlands is decreased, drying up wet meadows and gully seeps (Call and Maser 1985).

### *Best management practices for grazing*

Total removal of cattle is often recommended as the best, or even only, solution to habitat degradation caused by grazing (Fleischner 1994). But this solution often is not viable for political, social, and economic reasons. Several alternatives will reduce the impact of grazing (see Chapters 5 and 11), but BMPs will vary greatly among wetland types. For all cases, grazing fees that vary with wetland management and intensity of use could serve as incentives for improving range and wetland quality (Elmore and Beschta 1987).

The ecological health of the range and its accompanying wetlands will be influenced greatly by the stocking rate and type of grazing system. These topics are beyond the scope of this chapter but are covered by Holechek et al. (1989), Kinch (1989), Payne (1998), and Payne and Bryant (1998). For example, temporary high stocking rates implemented under short-duration grazing to benefit watershed ecology or vegetation can increase trampling of ground nests (Jensen et al. 1990). And even though grazing is typically considered harmful to wetlands, light to moderate cattle grazing can be used to open dense stands of tall emergent vegetation. This can be especially useful for creating an interspersion of vegetation and water suitable as brood habitat, and possibly increasing the density of invertebrates (see Kantrud 1986; see also Chapters 5 and 11).

Fencing wetlands to exclude cattle can prevent direct damage to wetlands but does not treat deteriorated watershed conditions (i.e., it may treat symptoms rather than problems [Davis 1986]). In addition, fencing is expensive to build and maintain and can impede movement of ungulates, particularly pronghorns. If fences are erected, the bottom strand should be smooth wire 46 cm above ground to permit movement of pronghorns (see Payne and Bryant 1998). When fencing, including a non-grazed buffer strip around the wetland will provide wildlife cover (Brown et al. 1990, Burke and Gibbons 1995) as well as reduce inflow of sediment and contaminants.

Cattle use of wetlands can be reduced by placing shade structures, water tanks, mineral blocks, feed supplements, and oilers at least 400 m and preferably 800 m away from riparian areas (Davis 1986, Kinch 1989). If stock tanks make use of natural springs or seeps, piping and fencing can be used to shift stock use away from wetland areas.

Proper timing can reduce the impact of grazing on wetlands. Because cattle prefer wetland areas through much of the year, continuous grazing usually results in overuse (Kinch 1989). In spring, upland vegetation is similar in succulence to wetland vegetation, and disproportionate use of wetlands is reduced (Clary and Webster 1989). Stocking rate and location of animals must be monitored and adjusted as necessary, as bank damage and soil compaction might be greatest in spring when soils are moist. Removal of cattle following spring grazing allows vegetation regrowth before fall dormancy. Winter grazing reduces soil compaction if soil is frozen, although excessive removal of vegetation before spring runoff can be detrimental to water infiltration (Kinch 1989) and remove nesting cover for birds. Grazing in summer should be avoided, as cattle concentration around wetlands is usually greatest then (Clary and Webster 1989).

Animal type also can affect the impact of grazing on wetlands. For example, yearling steers tend to be wider ranging and use wetlands less than cow-calf pairs (Kinch 1989). Herded sheep offer advantages over cattle in some areas, as herders can control timing, frequency, and duration of grazing (Kinch 1989).

The impact of grazing will vary greatly among sites and even among years for a given site. Therefore, grazing recommendations must be flexible, incorporating local and regional land use, wetland type, precipitation, landscape patterns, stocking rate, type of animal stocked, and timing of grazing. In all situations, wetlands and surrounding range conditions should be monitored and appropriate changes made if water quality or quantity is altered by grazing practices (see also Chapters 5 and 11).

## Development and Recreation

### *Impacts of development and recreation on palustrine habitats*

Development of palustrine habitats in the Intermountain West is exacerbated by the predominantly private ownership of riparian areas and wetlands. For example, even though 86% of Nevada's land area is publicly owned, at least 85% of Nevada's lowland meadow habitat is in private ownership (McAdoo et al. 1986). Also, roads often are developed along drainages because of the relatively gentle topography (Thomas et al. 1979). Because of private ownership and good access, wetlands and adjacent uplands are susceptible to development and, in areas with high human densities, urbanization.



Development has dramatic impacts on palustrine wetlands. Habitat is lost as vegetation is destroyed (Liddle 1975) and wetlands are filled. Habitat can be altered as wetlands are converted to pastures or hay meadows through removal of woody vegetation (often willow) and exotic plant species are introduced (Forman 1995). Wildlife travel corridors are fragmented, and increased human presence can disrupt animal activity patterns (see Boyle and Samson 1985), which can be particularly important in winter when wetlands are heavily used by ungulates. Urbanization is also characterized by 4 "standard pollutants" in runoff: (1) sediment and other solid particles, caused mainly by erosion associated with construction; (2) oxygen-demanding components; (3) nitrogen and phosphorous; and (4) traces of heavy metals (U.S. Environmental Protection Agency 1983).

### *Best management practices for development and recreation*

Ideally, zoning should be enacted to prohibit development in and adjacent to wetlands. Unfortunately, zoning is often ineffective due to social and economic pressures and even due to the difficulty of clearly defining and delineating wetlands. But the impact of development can be lessened by clustering impacts, where development (housing, for instance) is aggregated rather than dispersed across the landscape. This concentrates human impacts in one area, reducing impacts on the rest of the landscape.

Buffer zones can also be required around development. Vegetated buffer strips function as horizontal runoff filters, reducing or stabilizing sediment, nutrient, and pollutant loads entering water bodies (Nieswand et al. 1990, Osborne and Kovacic 1993). Characteristics of vegetative buffer strips necessary to protect wetland qualities will vary depending on local conditions. Buffer zones also are used to reduce the impact of human disturbance and development on wildlife. In Florida, a 100 m buffer minimized the flushing of 16 species of waterbirds and shorebirds from walking or vehicular disturbance (Rodgers and Smith 1997). A buffer zone of 275 m was recommended to protect 100% of turtle nest and hibernation sites in uplands surrounding a wetland in South Carolina; 90% of sites would have been protected with a 73 m buffer zone (Burke and Gibbons 1995).

Quality of runoff flowing into wetlands from urban areas can be improved by constructing retention ponds, which are highly effective at trapping and biologically incorporating pollutants (U.S. Environmental Pollution Agency 1983). Similarly, filter fabric placed along the perimeter of construction sites and vegetative buffer strips next to roads can reduce flow of sediment into wetlands. Wetlands themselves also are effective at removing suspended solids, heavy metals, and biological oxygen demand, but at the cost of increased sedimentation of the wetland.

## Agriculture

### *Impacts of agriculture on palustrine habitats*

Agriculture has many direct and indirect impacts on palustrine habitats, the most conspicuous of which are habitat conversion (see Mitsch and Gosselink 1993) and altered hydrology. Irrigation projects require that vast amounts of water be retained, stored, moved, and distributed, draining some wetlands and altering the hydrology of others. Typically, water regimes are stabilized by irrigation projects, changing the wetland hydroperiod. These changes can (1) influence composition and species richness of wetland vegetation; (2) reduce primary productivity as water flow and pulsing hydroperiod are reduced; (3) alter accumulation, decomposition, and export of organic material; and (4) influence nutrient cycling and nutrient availability (see Mitsch and Gosselink 1993; see also Chapter 6 of this volume).

Altered hydrology can have local and regional impacts on wildlife. Water impoundments in Colorado and Wyoming have stabilized water regimes and reduced spring flooding along the Platte River in Nebraska, allowing encroachment of woody vegetation onto sandbars that formerly served as roosts for whooping cranes and sandhill cranes (Aronson and Ellis 1979). Wet meadows along the Platte River, which serve as foraging areas for cranes, also have been reduced because of upstream water regulation (Aronson and Ellis 1979). But irrigation drains, ditches, and accompanying seeps also create wetland habitat used by wildlife (Mustard and Rector 1979; Ball et al. 1989; see also Chapter 6).

Irrigation does not completely remove water from local aquatic systems. Excess irrigation water, sometimes called drainwater, might return to local wetlands, often carrying a burden of salts and contaminants leached from agricultural soils. Closed aquatic systems (those with no outlet) are particularly susceptible to degradation, since contaminants become concentrated as water evaporates (see Lemly et al. 1993).

The most famous case of drainwater contamination occurred in the Central Valley of California at Kesterson Reservoir, which suffered extensive inflow of selenium from subsurface drainwater in the late 1970s and early 1980s (Zahm 1986). Selenium bioaccumulates in plants and animals and at high concentrations can cause teratogenesis, reduced hatching success, and mortality of birds (Ohlendorf 1989). Although typically associated with Kesterson Reservoir, symptoms of selenium toxicity and high concentrations of selenium in wetlands are widespread throughout the West (Lemly et al. 1993). Similarly, salts can occur naturally or can enter wetlands from irrigation drainwater. Excessive levels of sodium and magnesium sulfate in drinking water can alter growth and development and even cause death of mallard ducklings (Mitcham and Wobeser 1988*a,b*).

Agrichemicals (pesticides and fertilizers) can enter wetlands directly through overspray, drift, cultivation, and treatment of dry wetland basins or indirectly through volatilization and runoff (Grue et al. 1986). Herbicides and fertilizers can cause mortality or increased growth, respectively, of wetland vegetation (Solberg and Higgins 1993, Mitsch and Gosselink 1993); other pesticides can cause direct (Grue et al. 1986, Tome et al. 1991, Dieter et al. 1995) and indirect (Hunter et al. 1984, Grue et al. 1986) mortality of wildlife.

Erosion of agricultural lands is a primary cause of sediment loading of wetlands. In some areas of the Pacific Northwest where conventional tillage is used, up to 12 bushels of topsoil are eroded annually for each bushel of wheat produced (Michalson and Papendick 1991). Loss of nutrients accompanies soil erosion; combined erosion losses of nitrogen, phosphorus, and potassium in the United States exceeds 69 million metric tons annually (Larson et al. 1983). Ultimately, high levels of erosion entering wetlands can cause increased biological oxygen demand, altered hydrologic regimes, sedimentation, eutrophication, and loss of open water habitat (Mitsch and Gosselink 1993).

Fragmentation and degradation of uplands next to wetlands can impact wetland wildlife negatively. Increased edge can lead to increased predation of bird nests (see Paton 1994) and turtle nests (Temple 1987). In addition to loss and degradation of habitat, tillage of uplands causes substantial nest loss for waterfowl and other species requiring uplands for nesting (Higgins 1977, Cowan 1982, Rodgers 1983).

### *Best management practices for agricultural development*

Many wetland problems associated with agriculture can be avoided or reduced by using alternative management techniques in adjacent uplands. Foremost among these is the use of conservation tillage techniques. These techniques, which vary in purpose and effectiveness, are also known as no-till, minimum-till, reduced-till, mulch-till, and stubble-mulching. The main purpose of conservation tillage is to reduce erosion by leaving protective crop residue on the soil surface rather than turning residue under the soil as with a moldboard plow. In addition to reducing soil erosion and downstream pollution, conservation tillage can increase soil organic matter, reduce fuel use, reduce soil compaction, and increase storage of soil moisture (Ritchie and Follett 1983). Conservation tillage is not a panacea, however, as weed control and insects might require increased use of chemical pesticides. Conservation tillage can reduce soil and nutrient losses dramatically (Langdale et al. 1983), although implementing conservation tillage widely might require integrating new crop management practices, plant types, pest control methods, and socioeconomic principles (Michalson and Papendick 1991).

Conservation-tillage fields typically have higher nesting densities of upland-nesting birds relative to conventionally tilled fields (Cowan 1982, Basore et al. 1986), although numbers of birds are low relative to undisturbed natural grasslands. Nest losses can be high from machinery disturbance and high rates of predation, however, and conservation-tillage fields can become ecological traps if nesting success is inadequate to sustain populations without immigration from other sources (Basore et al. 1986, Best 1986). Nest loss in conservation-tillage grain fields can be reduced by using undercutters with mulch treaders removed rather than disks, plows, or treaders for the first spring tillage (Rodgers 1983). Where practicable, delaying cultivation and haying until after fledging also will reduce nest loss. If fieldwork is necessary, avoidance of known nests can reduce nest losses, and cutting hay from the center of the field outward will allow some wildlife to escape.

Vegetative buffer strips surrounding wetlands can be extremely effective at reducing sediment, nutrient, and pollutant loads entering wetlands (see Nieswand et al. 1990, Osborne and Kovacic 1993) in addition to providing cover for wildlife (see Payne 1998, Payne and Bryant 1998). Because the effectiveness of vegetative buffer strips varies with slope, topography, vegetation density, vegetation type, soil characteristics, subsurface drainage, upslope land use, and overland flow (see Nieswand et al. 1990, Osborne and Kovacic 1993), universal guidelines for buffer width are inappropriate. In addition, buffer zones of any width are not a remedy for poor land-management practices, and special restrictions might have to be placed on some lands upslope of the buffer zone (Nieswand et al. 1990). Slopes >15% and impervious surfaces such as roads should not be considered part of a buffer zone, and width of buffer zones should increase with time of travel of overland flow and steepness of slope (Nieswand et al. 1990). The effectiveness of vegetative buffer strips will be maximized if they are placed along smaller headwater streams, the lengths of which dominate drainage networks (Osborne and Wiley 1988).



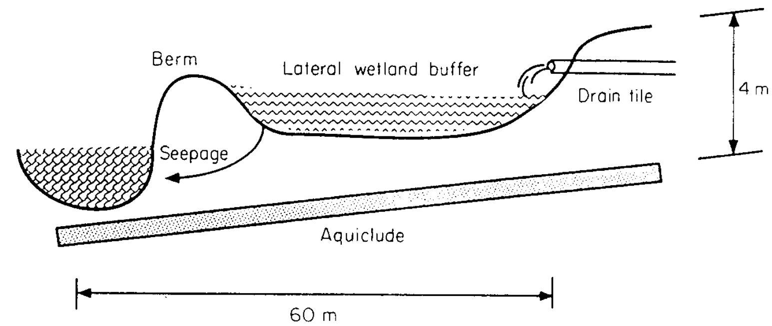


FIGURE 8.1. Lateral wetland buffer for removing nutrients and sediment from water carried by drain tiles or runoff ditches (after Kovacic et al. 1990).

Little information is available on the effectiveness of buffer strips in the Intermountain West, but studies from across North America have shown reduced nitrogen and phosphorous inputs with grass strips as narrow as 5 m and forest strips as narrow as 30 m (see Osborne and Kovacic 1993). Recommended minimum buffer widths to protect water quality in the eastern United States are 91 m for terminal reservoirs with water supply intakes and 15 m for perennial streams (Nieswand et al. 1990). Widths of up to 223 m are recommended to preserve wildlife habitat (Brown et al. 1990). Local conditions must always be considered when determining appropriate width of vegetative buffer strips (see also Chapters 5 and 11).

Vegetative buffer strips are much less effective in areas that have drain tiles, where subsurface flow directly enters water bodies. This can be remedied by having tiles (or runoff ditches) enter small artificial wetlands separated from natural wetlands or water bodies (Figure 8.1). These small artificial wetlands are separated from the natural water body by a vegetative buffer strip that can filter sediment and nutrients (Osborne and Kovacic 1993). Similarly, sediment ponds can be excavated and seeded in the terminus of small gullies in cultivated fields. These impound runoff, which loses much of its silt load because of reduced velocity and the surrounding vegetative buffer strip.

Salt loading of streams from irrigation water can be reduced by closing open drains, reducing the amount of irrigation water used on croplands, and lining canals and laterals with concrete or running them through pipes (Mustard and Rector 1979). However, these control methods also can reduce artificial wetland habitats by eliminating seepage of irrigation water (Mustard and Rector 1979; also see Chapters 5 and 6).

Integrated pest management, IPM (Huffaker 1980), on crop fields adjacent to wetlands can reduce the amount of pesticides entering wetlands, as well as reduce pesticide costs to farmers. IPM reduces insect populations using crop rotation, biological controls, physical barriers, and precise application of chemical pesticides in conjunction with close monitoring of insect populations. If chemical pesticides are used, applicators should (1) use the smallest necessary amount of the least toxic and persistent pesticides; (2) avoid using chemicals during the nesting season; (3) avoid treatments near riparian zones; (4) avoid using pesticides in irrigation systems because of drift potential and lack of control over application rate; and (5) avoid chemical use in nontilled areas, which serve as buffer strips and wildlife habitat (Payne 1998, Payne and Bryant 1998).

## Mining

Mining provides products such as coal, sand, gravel, crushed stone, quartzite, limestone, granite, gypsum, bentonite, phosphate, oil shale, uranium, and precious and semiprecious minerals including gold, silver, iron, tin, copper, nickel, lead, and cadmium. All mining disturbs habitat directly with alteration of landscape and vegetation (Moore and Mills 1977, Matter and Mannan 1988, Hart 1992). Such alterations can change the water table and groundwater flow (Parizek 1985). Surface-water diversions and aquifer dewatering can reduce water inflows. In addition, during and following mining operations, runoff and leachate from abandoned tunnels and shafts and front tailings dissolve trace metals, thus contaminating nearby surface and even downstream water, especially from mining coal and metals. During coal mining, iron pyrite and other metal-bearing minerals are exposed to percolating water, leading to release of acidic leachates (acid mine drainage), producing acid water conditions that can kill invertebrates in wetlands. Mine leachates especially have variable and high concentrations of dissolved iron, sulfate, calcium, magnesium, aluminum, copper, manganese, zinc, cadmium, lead, and arsenic, which can lead to toxic concentrations in fish and wildlife (Parizek 1985). And, of course, turbidity from siltation reduces photosynthesis and dissolved oxygen content.

### *Impacts of mining on palustrine habitats*

Mining can impact wetlands on site or off site by extending to wetlands downstream or upstream by changing water levels due to channelization or blockage (Cardamone et al. 1984; also see Table 8.1). Impacts can be acute from removing vegetation or chronic from alteration of seasonal flows of surface water and groundwater and from chemical factors



**Table 8.1.** On-site and off-site impacts of mining on wetlands (Cardamone et al. 1984)

| Impacted Resource | Impact On-site | Off-site |
|---|---|---|
| *Water* | | |
| Quality degradation | X | X |
| Aquifer disruption | X | X |
| Flood control disruption and storage loss | X | X |
| Alteration of seasonal flow | X | X |
| *Land* | | |
| Erosion | X | X |
| Alteration of land use | X | X |
| Soil redistribution | X | |
| Alteration of soil productivity | X | |
| Alteration of soil stability | X | |
| *Vegetation* | | |
| Vegetation removal | X | |
| Alteration of species composition | X | X |
| Reduction of diversity | X | X |
| *Wildlife* | | |
| Habitat destruction | X | X |
| Wildlife displacement | X | X |
| Creation of wildlife barriers | X | |
| *Other* | | |
| Alteration of recreational use | X | X |
| Alteration of esthetic value | X | X |
| Alteration of scientific/educational/ historical/archeological value | X | X |

such as mine drainage. Impacts may take place during any stage of mining, including exploration, dewatering and diversion, topsoil removal, blasting, overburden removal, spoil placement, hauling, soil storage, maintenance, reclamation, and post-operation (Cardamone et al. 1984). Detrimental impacts of mining on wetlands can be chemical and physical (Table 8.2).

Wetlands vary in ability to adapt and recover from disturbance (Payne 1998). Wetland characteristics are measurable (Evans and Krebs 1977, Beule 1979, U.S. Fish and Wildlife Service 1981, Verry 1985, Adamus

**Table 8.2.** Chemical and physical impacts of mining on wetlands (Cardamone et al. 1984)

| Impact |
|---|
| *Chemical* |
| Addition of large amounts of chemicals |
| Addition of large amounts of chemically reduced materials, especially sulfides |
| Addition of metallic oxides and hydroxides |
| Addition of large quantities of sulfuric acid |
| Drastic lowering of pH |
| Reduction and elimination of carbonates |
| Placing of heavy metals into solution |
| Reduction of free oxygen |
| Contamination of groundwaters that feed wetland areas |
| *Physical* |
| Drainage of wetlands |
| Filling of wetlands with spoil and tailings |
| Alteration of stream courses by channelization, diversion, and impoundment |
| Widening of stream beds |
| Covering of wetland bottoms with spoil and tailings |
| Increased silt loads |
| Increased turbidity |
| Decreased light penetration |
| Reduced habitat diversity |
| Removal of natural cover |
| Removal and burial of topsoil |
| Exposure of vast bare rock surfaces |
| Creation of long highwalls that might seep |
| Creation of open pits, quarries, and spoil depressions that might fill up with seepage |
| Creation of large areas of spoil piles that seep, erode, and are unstable |
| Acceleration of surface runoff |
| Increased erosion |
| Watercourse modification from spoil and tailing impoundments |
| Lowering of groundwater |
| Inadequate buffer zones or refugia |

et al. 1987, Bartoldus 1992, McKinstry and Anderson 1994). Sensitivity of vegetation, or ability to resist structural and functional perturbances, is called inertia and can be used to indicate wetland health (Cardamone et al. 1984). Inertia is determined by redundancy in landforms and functional factors, existence of vegetation accustomed to variable conditions, chemical characteristics of water, mixing or flushing capacity, amount of previous disturbance, and management potential of the region. Elasticity (i.e., adaptability of a wetland) is determined by habitat condition or toxin levels at disturbed areas, existence of seed banks to repropagate the disturbed wetland, dispersal ability of seed types, and management capability to control damaged sites (Cardamone et al. 1984). Resilience, or recovery capacity of a wetland, is not well understood, but a wetland system can recover from disturbance a limited number of times before being critically damaged. Extent of reclamation is influenced by natural means of reducing toxins in topsoil through leaching and dilution to allow volunteer seeding. Impact of mining is influenced by methods, timing, and quality control of mining, as well as site conditions, wetland sensitivity, and extent of reclamation (Cardamone et al. 1984).

### *Mitigation*

To alter wetlands, a permit is needed from the U.S. Army Corps of Engineers (Corps) under Section 404 (b)(1) of the Clean Water Act, following guidelines established by the U.S. Environmental Protection Agency (Bean 1983, Salvesen 1990, Cylinder et al. 1995, Studt and Sokolove 1996; also see Chapter 1). The U.S. National Environmental Policy Act (NEPA) of 1970 established the Council on Environmental Quality, which promulgates rules that govern compliance with NEPA. Among those is the definition for mitigation: ". . . (1) avoiding the impact altogether by not taking a certain action; (2) minimizing impacts by limiting the degree of the action; (3) rectifying the impact by repairing, rehabilitating, or restoring the affected environment; (4) reducing or eliminating the impact over time by preservation and maintenance operations; and (5) compensating for the impact by replacing or providing substitute resources . . ." (Marsh et al. 1996).

The Corps generally will grant a permit to alter wetlands if the applicant first takes all practicable steps to avoid adverse impacts to the wetlands, then minimizes unavoidable damage to the wetlands, and then compensates for permanent destruction of wetlands. With compensation, the Corps and EPA prefer that the applicant create the same kind of wetland on the same site as the one being altered, preferably connected to the same water source, and as close to the altered wetland as possible. (See Chapter 1 of this volume for more detail on CWA requirements.)

### TYPES OF MITIGATION

Five basic types of compensatory mitigation are available as options for wetlands damaged or destroyed by draining or filling activities: (1) restoration, (2) creation, (3) enhancement, (4) exchange, and (5) preservation (Kruczynski 1990). But preservation of existing wetlands through acquisition is not good compensatory mitigation unless the acquired wetlands have substantially higher quality, because a net loss of wetland area and functions has occurred, and the new area already is regulated through the Section 404 program. Any preservation agreement should be in perpetuity via title transfer to a responsible conservation organization. Preservation is a last mitigation option. Enhancement means increasing or improving at least one of the functions of an existing wetland. Unlike restoration sites, enhancement sites already provide some wetland functions. Extreme enhancement is called *exchange* because it merely exchanges wetland types, such as placing fill material in open water of a deepwater marsh to replace a submerged habitat type with an emergent one. Exchange can improve ecological diversity, but it is generally considered inferior to preservation as a mitigation option.

Restoration of degraded wetlands should be the first option, because it reestablishes what had evolved as a natural part of the landscape and because it often is easiest to do. Sometimes restoration merely involves plugging a drainage ditch or breaking drainage tile (Payne 1998). Other times it might mean using heavy equipment to grade an area to or just below the water table and reestablishing wetland soils and vegetation. Creation and enhancement are intermediate mitigation options. Creation is preferred to enhancement because creation adds to the total wetland area of a site, assuming created wetlands function as natural wetlands being lost.

### TIMING AND LOCATION

The best timing for mitigation projects is "up front" (i.e., before the impact occurs), for then success can be ascertained in advance and adjustments made (Kruczynski 1990). Mitigation up front should be required for all projects with substantial risks due to size and complexity. Otherwise, mitigation should proceed concurrently with project construction. It should be discouraged after project completion, when little incentive remains to complete the mitigation phase timely and satisfactorily. On-site and off-site mitigation should occur in the same ecosystem, best defined as the same watershed. If no adequate mitigation sites exist in the same watershed, the construction permit should be denied (Kruczynski 1990).



### RATIOS

For each 90% reduction in area, a habitat will lose, on average, 30–50% of its species (Diamond 1975). Species richness is greater in large habitat patches than small patches in terrestrial (Whitcomb et al. 1981) and aquatic systems (Brown and Dinsmore 1986, Gibbs et al. 1991), although density of breeding ducks is higher on small wetlands than large (Cowardin et al. 1995). General ratios exist for the amount of area to be mitigated relative to the amount of area damaged or destroyed, especially for on-site, in-kind (type for type) mitigation (Kruczynski 1990, King and Adler 1992).

Success in restoring most damaged or destroyed herbaceous wetlands is good because a wetland existed at the site and herbaceous vegetation grows fast (Kruczynski 1990). But because success is uncertain and the restored wetland needs time to become fully functional, a 1.5 : 1 ratio is recommended, that is, an area is restored that is 1.5 times larger than the area lost. If natural colonization of desirable plants is unlikely, planting is necessary. If successful restoration occurs up front (in this case, before initiating a filling or draining activity) or if the restored wetland will be an improvement ecologically, the ratio can be 1 : 1. Creating wetlands from uplands is risky. Thus a ratio of 1.5 : 1 or 2 : 1 is recommended. If success is demonstrated up front, the ratio can be 1 : 1. Enhancing wetlands also is risky because although some functions will be improved, other existing functions could be degraded. Thus, a 3 : 1 ratio should be required unless enhancement is performed up front, when a 2 : 1 ratio will suffice.

Exchanging wetland types is undesirable, because one wetland type is merely replaced with another. Still, an exchange could be an improvement if it is an abundant wetland for a rare one. Areal size is determined case by case.

Acquiring wetlands for preservation through donations, conservation easements, restrictive covenants, etc., generally should not be considered as mitigation because of the net loss in overall area and function.

### COMMUNITY TYPE

In-kind mitigation, that is, replacement with the same community type, is best (Kruczynski 1990). Out-of-kind mitigation is used only when an improvement is clear, such as when a mining company is granted a 404 permit to mine ore under a wetland dominated by a monotypic stand of cattail, and mitigation calls for out-of-kind replacement to create a more diverse herbaceous community at the same site. Maintaining wetland complexes is also important in providing wildlife habitat and ensuring wetland function (Fairbairn and Dinsmore 2001).

### MITIGATION BANKING

Wetland mitigation banking is a form of off-site compensatory mitigation (Kruczynski 1990, Cylinder et al. 1995, Marsh et al. 1996). It has evolved as an alternative to traditional approaches of compensating for "unavoidable" wetland losses under Section 10 of the Rivers and Harbors Act and Section 404 of the Clean Water Act (Short 1988, Kelly 1992). McElfish and Nicholas (1996) described 6 functions of a mitigation banking program: (1) existence of a client (use of credits), (2) permitting the project to proceed, (3) credit production, (4) long-term property ownership, (5) credit evaluation, and (6) bank management. As of mid-1992, the Environmental Law Institute (1993) identified 46 existing mitigation banks in 17 states; almost 75% of all banks provided mitigation solely for public works projects. Brumbaugh and Reppert (1992) listed existing and planned wetland mitigation banks.

Mitigation banking is generally used to aggregate small impacts on wetlands. Variations occur, but generally, someone such as a government agency or an investor buys and restores a degraded wetland, which becomes the bank. Values of this wetland are quantified and used as "credits" that can be withdrawn at a price paid in compensation by a developer who unavoidably damages wetlands elsewhere. The bank should occur in the same watershed and consist of wetland types similar to wetlands where impacts eventually will occur. Much legal, scientific, and administrative complexity is involved, with potential for misuse. Hammer et al. (1994) listed positive and negative effects that mitigation banking has on wildlife, wetlands, and society. Positive effects include: (1) opportunities for restoring degraded wetlands, (2) potential for a net gain in wetlands, (3) improving involvement and cash flow for owners of lands that have little market value, and (4) facilitating conflict resolution. Negative effects include: (1) loss of terrestrial habitats to wetlands conversion, (2) altering the natural distribution of wetlands, (3) altering types and functions of wetlands, and (4) increasing regulatory requirements. Hammer (1994) also listed considerations for a wetland mitigation banking policy, such as including national guidelines in a national wetland policy, state or regional decision making, ecological equivalency as a specific goal of compensatory mitigation, and a monitoring and evaluation plan and funding.

### VALUES, EVALUATION, MONITORING

Wetlands are tremendously valuable to society and the environment for a variety of reasons, including hydrology, water quality, food-chain support and nutrient cycling, habitat, and socioeconomic considerations (Sather and Smith 1984). Foster (1986) recognized intrinsic wetland values and 4 categories of socioeconomic wetland values: (1) economic, (2) scientific and educational, (3) experiential (ethical, spiritual, physical, aesthetic, recreational, inspirational), and (4) ecological (groundwater recharge or discharge, flood control, erosion control, water purification, primary production, secondary production, threatened and endangered species, wildlife refuge).



Wetlands must be evaluated before and monitored during and after mitigation. Acceptable procedures include the Habitat Evaluation Procedure, HEP (U.S. Fish and Wildlife Service 1981) and the Wetland Evaluation Technique, WET (Adamus et al. 1987). Bartoldus (1992) described the Wetland Replacement Evaluation Procedure (WREP) as a better alternative to HEP; other wetland evaluation techniques were listed by Verry (1985) and Dobie (1986). Photographic documentation also is helpful, with permanent photographic stations located to provide a visual record of the mitigation site before, during, and after construction and during monitoring (Cylinder et al. 1995). The system must be given time, about 15 to 20 years for freshwater marshes (Mitsch and Wilson 1996), and monitoring can be one of the conditions of a 404 permit (Hart 1992). Moore and Mills (1977) provided details for mitigation and monitoring its success from impacts of western surface mining.

## Conclusion

Management of palustrine habitats must consider large-scale influences. Palustrine wetlands are affected by a wide range of inputs from uplands, and management of surrounding uplands will dramatically affect palustrine wetlands. The hydrology and quality of palustrine wetlands can be directly affected by off-site conditions and uses (e.g., irrigation, drainage projects, water diversion, mine drainage), some of which may be hundreds of kilometers away. The effects of altered habitat can also be far-reaching on wildlife that uses palustrine wetlands for part of the year but migrates to other areas for breeding or wintering. Finally, temporal scale must be included in management decisions. Annual variation in precipitation is particularly important in the arid Intermountain West, and management prescriptions should be appropriate for the worst conditions that might be encountered, not the best. Similarly, timber harvest, mining, and grazing pressures will vary depending on social and economic pressures, and again, management prescriptions should consider all possible conditions.

Management must also consider local variation. No management prescriptions are universal, as they will vary with soil types, precipitation, land use, cattle stocking rates, slope, and economic conditions. In all cases, BMPs should be adapted for local and regional conditions. Human impacts and the quality of palustrine habitats should be monitored and management prescriptions changed as necessary. In some cases, the BMP will be to reduce or eliminate human uses that adversely affect wetlands.

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