Declining Woody Vegetation in Riparian Ecosystems of the Western United States

Robert A. Obedzinski, *USDA Forest Service, Gifford Pinchot NF, Vancouver, WA 98682;* **Charles G. Shaw III,** *USDA Forest Service, Research and Development, Washington, DC, 20024; and* **Daniel G. Neary,** *USDA Forest Service, Rocky Mountain Research Station, Flagstaff, AZ 86001.*

ABSTRACT: *Riparian ecosystems serve critical ecological functions in western landscapes. The woody plant components in many of these keystone systems are in serious decline. Among the causes are invasion by exotic species, stress-induced mortality, increases in insect and disease attack, drought, beaver, fire, climatic changes, and various anthropogenic activities. The latter include agricultural development, groundwater depletion, dam construction, water diversion, gravel mining, timber harvesting, recreation, urbanization, and grazing. This article examines the factors implicated in the decline and discusses the importance of interactions among these factors in causing decline. It also clarifies issues that need to be addressed in order to restore and maintain sustainable riparian ecosystems in the western United States, including the function of vegetation, silvics of the woody plant species involved, hydrologic condition, riparian zone structure, and landscape features, geomorphology, and management objectives. West. J. Appl. For. 16(4):169–181.*

Key Words: Insects, diseases, invasive species, dams, recreation, grazing, water diversion, mining, logging.

Riparian Zones and Their Importance

The ecological and economic importance of riparian ecosystems far exceeds their small footprint on the landscape (<2%, Svejcar 1997a). They are highly productive, providing water and essential habitat for various fish and fauna. Because they serve critical functions within and beyond their boundaries, these ecosystems must be examined in the broader context of the landscapes in which they occur.

Riparian refers to "the zone of direct interaction between terrestrial and aquatic environments" (Oliver and Hinckley 1987). It includes a perennial or ephemeral stream channel, a zone within the flood plain dominated by hydric woody plants or aquatic graminoids such as sedges, and a terrestrial area where vegetation and microclimate are influenced by perennial or intermittent water tables and by the ability of soil to store water (Figure 1). Riparian ecosystems range from southwestern arroyos that receive surface water only occasionally to the densely wooded banks of rivers in the Pacific Northwest. Their composition ranges from aquatic graminoid-sedge wet meadows to extensive deciduous, coniferous, or mixed-gallery forest.

Riparian zones are important not only because of the plants and animals that inhabit them but also because of their linkages with the natural characteristics and ecological health of streams and rivers (Naiman et al. 1992). Riparian vegetation influences light and temperature, supplies food for stream biota, and contributes large woody debris. Healthy riparian areas filter out large quantities of sediments, nutrients, pesticides, and animal wastes—nonpoint source contaminants responsible for more than 50% of the pollution in our nation's waters (Fail et al. 1988, Welsch 1991). Riparian ecosystems also can moderate flooding and provide a steady baseflow during dry periods.

In this article we discuss factors implicated in the decline of Western riparian ecosystems, examine interactions among these factors, and clarify issues that need to be addressed in order to restore and maintain their sustainability.

Sustainability and Health

Riparian health often is defined in terms of watershed-level functions—such as organic debris inputs, fluctuations in light and temperature and streamflow—that influence the physical, chemical, and biotic character of riparian and instream systems (Naiman et al. 1992). DeBano and Schmidt (1989) define riparian health as "a stage of vegetative, geomorphic, and hydrologic development, along with a degree of structural integrity…" This definition reflects equilibrium between aggradation and degradation processes and recognizes the importance of relatively static "stages" of vegetative, geomorphic, and hydrologic development while acknowledging the evolving nature of riparian zones.

NOTE: Charles G. Shaw III can be reached at (703) 605-5261; Fax: (703) 605-5133; E-mail: cgshaw@fs.fed.us.

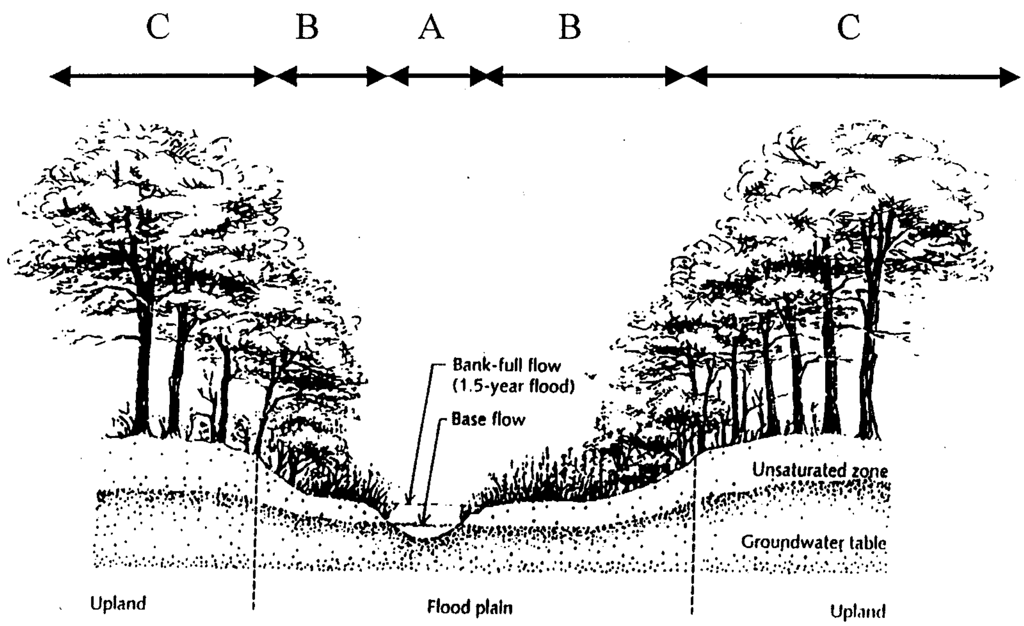

Figure 1. Riparian zone components: A: Perennial or ephemeral stream channel, B: First terrace flood plain dominated by hydric plants, and C: secondary and tertiary terraces influenced by perennial or intermittent water tables (adapted from Binford and Buchenau 1993).

Riparian Disturbance

Riparian areas are complex, diverse, and constantly changing because of their dynamic fluvial systems (Svejcar 1997a, Karr 1996). Unlike streamflow regimes and hydroclimate, riparian ecosystems exhibit local and regional distinctions (Lins 1997). When subjected to disturbance, resistant riparian systems do not change out of the range of their natural state. Resilient riparian ecosystems may shift to a disturbed, stable state, but eventually recover to a natural dynamic state

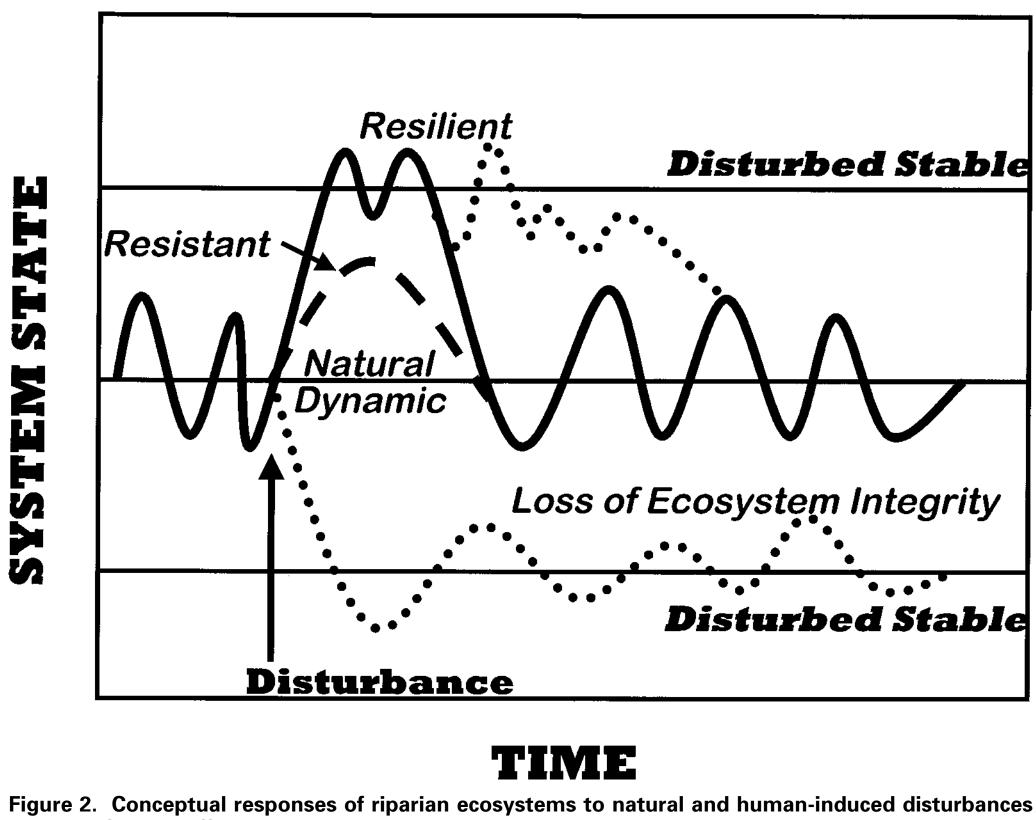

Figure 2. Conceptual responses of riparian ecosystems to natural and human-induced disturbances (adapted from Kauffman et al. 1997).

in the short term. However, if the frequency or severity of disturbance increases, then ecosystem health and sustainability can decline. The consequence can be a shift to another stable, but permanently disturbed state (Figure 2).

Short-term disturbances in riparian systems are relatively easy to comprehend and observe; however, short-term views can bias perception of the true dynamic state. The 100 yr “surprise” flood is but another disturbance in the long evolution of a riparian zone. Long-term disturbances, particularly those out of our current observational and historical range, may confuse our interpretation of the dynamic state of riparian ecosystems and the real magnitude of the disturbances we observe. For example, riparian zones in the Great Basin are still adjusting to the drying up of late Pleistocene lakes, and consequent lowering of base levels (Masters et al. 1991). Riparian zones in the Southwest have experienced four major sequences of aggradation and degradation at intervals of 150 to 400 yr over the past 2,100 yr (Grissino-Mayer 1996).

Properly Functioning Vegetation

Riparian ecosystems range from heavily wooded to dominance by aquatic graminoids such as *Carex* spp. and *Juncus* spp. (Medina 1996). Properly functioning riparian systems have: (1) stable stream banks, (2) good water quality, (3) a high water table, (4) high production of terrestrial biomass, (5) accretion of soil organic matter, (6) perennial vegetation, (7) native vegetation, (8) sustained aquatic fauna, and (9) a soil-geologic matrix that promotes water retention and baseflows. Woody and/or herbaceous vegetation provides these functions (Medina 1996).

A paucity of woody plants does not necessarily indicate a declining riparian system. Root density and depth are critical to streambank stability and some of the highest root mass develops not in wooded, but in sedge dominated riparian areas (Manning et al. 1989). These plants have the greatest ability to maintain streambank stability during high velocity flood flows. There can be good reasons (e.g., climate, hydrologic regime, fire history, natural herbivory levels, etc.) why woody plants are scarce or absent in a particular riparian system.

Woody Vegetation Decline

Woody riparian vegetation has been declining throughout the West (Boldt et al. 1978, Johnson and Haight 1984, Minckley and Rinne 1985, Green and Kauffman 1989, Kreuper 1993, Rood et al. 1995). These declines have resulted from combinations of natural and anthropogenic factors such as drought, water diversions, groundwater depletion, dams, exotic species, climatic changes, timber harvest, mining, urbanization, insect outbreaks, recreation, wildfire, and grazing (Minckley and Brown 1982, Svejcar 1997b). These declines are not all new; some riparian areas were already highly disturbed by the mid-1800s (Leopold 1951, 1959).

Green and Kauffman (1989) indicated that more than 70% of the riparian areas in the United States have been altered; less than 2% remain natural. Of the 400,000 ac of Sonoran riparian forests that grew along the Colorado River from Fort Mohave to Fort Yuma around 1900, only 768 ac remained by 1986 (Kreuper 1993). In the Salt River Valley of Arizona, dams led to down-channel decline of mesquite (*Prosopis* sp.) riparian systems (Minckley and Rinne 1985). Johnson and Haight (1984) estimated a 70–95% loss of natural riparian vegetation in the Southwest. In the Gunnison Basin of Colorado, half of the riparian systems are in poor condition (G. Kittel, Colorado Natural Heritage Program, pers. comm., 1994). On the Great Plains, many riparian woodlands have been replaced by grasses and forbs (Boldt et al. 1978). A 68% decline in cottonwoods along the St. Mary River in southern Alberta was related to water diversion (Rood et al 1995).

Decline in riparian health has resulted from imbalances in aggradation and degradation processes. In some systems, management practices combined with various environmental changes (temperature, drought, flood, exotic species, etc.) have produced unstable vegetative structures and/or compositions. The results are increased stress, mortality of unadapted species, and consequent shifts in species composition to ones better adapted to the new conditions.

The forest decline model of Muller-Dumboise (1992) and Manion and Lachance (1992) provides a means to examine decline of woody vegetation in riparian ecosystems. The model contains four generic causal factors (simplified forest structure, edaphically extreme sites, recurring perturbations, and biotic agents) that fit into a three-step spiral of decline. The first step consists of such *Predisposing Factors* as genetic potential, age, urban disturbance, soil compaction, poor fertility, altered salinity, low soil moisture capacity, poor soil drainage, climate change, and air pollution. The second step consists of such *Inciting Factors* as defoliating insects, fungi, drought, salinity, frost, air pollution, and soil disturbance. The final step consists of *Contributing Factors* that are mainly biotic agents such as fungi, wood and bark boring insects, root rots, viruses, nematodes, and saprobic decay.

Predisposing Factors

Predisposing factors are stationary conditions resulting in stresses that remain relatively static within the current environment (Manion 1981). Climate, soils, age of woody species, and exotic plants are the major predisposing factors implicated in declines of woody vegetation in Western riparian systems.

Climate

Many Western climates are characterized by low precipitation, cold winters, and hot windy summers that produce high rates of evapotranspiration. Coupled with a short growing season, this creates a harsh environment for tree survival (Girard et al. 1987). Western climates also are characterized by wet and dry periods. Peterson (1979) reported 269 yr of drought and 469 wet years over a 738 yr period in Nebraska. Dry and wet periods averaged 13 and 24 yr and ranged from 5–38 and 3–69 yr, respectively. Thus, the general climate may periodically increase or decrease stress on riparian species. Grissino-Mayer (1996) reconstructed precipitation in New Mexico for the past 2,129 yr and found that there had been four major aggradation-degradation cycles. He documented droughts lasting nearly 400 yr and wet periods of up to 350 yr. The past 200 yr period is the wettest since the seventh century, and the past decade has been the wettest of the entire record. A similar shifting occurs in the Pacific Northwest, which generally has wet winters and dry summers (Spurr and Barnes 1980). Alterations in weather patterns owing to global climatic processes such as El Niño and La Niña can generate markedly drier or wetter conditions (Risser 1992).

These phenomena also can alter the runoff and flooding events that sustain many riparian systems. Shifts from aggradation to degradation cycles increase flood flows, stream downcutting, and changes in vegetation. In addition, along ephemeral or intermittent streams, nonriparian species such as ponderosa pine (*Pinus ponderosa*) often dominate because of limited moisture. During wetter periods, there are also fewer disturbances such as wildfire. This change allows shade-tolerant species to increase in abundance and density—increases that can lead to stress during later periods of drought.

These climatic factors may predispose some species to the debilitating effects of other factors (insects, diseases, air pollutants, mechanical injury, etc.), while coincidentally improving conditions for other species. For example, during dry periods, certain grasses may colonize sites previously dominated by moist-site species such as cottonwood (*Populus* spp.) and willow (*Salix* spp.). Fresh alluvium is important to the establishment of pioneer species such as cottonwood and willow (Johnson et al. 1976). If drier weather patterns prevent alluvial development, then species better adapted to nondisturbance such as ash (*Fraxinus* spp.), boxelder (*Acer negundo*), or oak (*Quercus* spp.) may begin to dominate.

Soils

Soil types can predispose riparian zones to decline as their physical, chemical, and biological characteristics affect moisture and nutrient availability (Powers 1991). Coarse soils (those with a high percentage of sands or gravels) limit moisture holding and can reduce the ability of plants, especially shallow rooted species, to obtain moisture when water levels decline. Moisture-holding capacity is particularly important for seedling and sprout establishment.

Compaction, erosion, and loss of the litter layer influence soil moisture and nutrient availability. Trampling by wildlife and livestock, wildfires, recreational traffic, and the impacts of increasing urbanization can alter physical properties of soil and influence the ability of woody plants to acquire water and nutrients—thus predisposing them to decline.

Age of Woody Species

As trees age, susceptibility to insects and disease increase and their ability to recover from damage decreases. The influence of age is evident in declining riparian woodlands of the Great Plains, where senescing trees became increasingly vulnerable to damage by harsh climate, insect attack, diseases, and mechanical stresses (Boldt et al. 1978).

Old trees have dominated certain riparian ecosystems through time; however, when such an overstory dies without an established understory, then other drought-tolerant species such as grasses, forbs, and exotics move in and reduce the potential for future natural woody vegetation. Lack of natural regeneration is due mainly to cattle grazing in the Great Plains (Severson and Boldt 1978). In the Southwest, water diversion and groundwater pumping are significant causes (Howe and Knopf 1991, Stromberg 1993a), as are cattle and elk grazing, and sand and gravel mining (Stromberg, 1993b).

Exotic Species

Russian-olive (*Elaeagnus angustifolia*), a native of Europe and western Asia introduced around 1900, and tamarisk (*Tamarix chinensis*), introduced in the early 1800s from Europe, have invaded many riparian sites because of a decrease in flooding events that create mineral seedbeds for disturbance-dependent native species such as cottonwood (Horton 1977, and Borell 1962). Also, tamarisk has a 5 month fruiting period, and Russian-olive has a fall to spring period of seed viability—life cycle features that give the exotics a longer period to exploit suitable germination conditions. In addition, shading by exotics minimizes the potential for successful cottonwood regeneration, and exotics are more competitive in shade (Howe and Knopf 1991) and under drought (Cleverly et al. 1997).

Horton (1977) found that phreatophyte species such as cottonwood and willow (*Salix* spp.) spread by releasing many wind-borne seeds that germinate quickly on water or moist soil. As these small seeds quickly lose viability, they need to germinate within 2 to 4 months. They require much moister soils for a longer time than does tamarisk and are poor competitors in shade, whereas Russian-olive grows well in shade beneath large cottonwoods (Borell 1962). Once established, these exotics outcompete cottonwood and willow for light and limited water.

Tamarisk's high tolerance to salt, coupled with its ability to increase the concentration of soil salts through autogenic processes, gives it a competitive advantage over many native species (Stromberg 1993a). It also can grow rapidly in wet years, yet tolerate years of drought and heat stress (Cleverly et al. 1997). Reduction in streamflow, stream regulation, or water diversion may give exotic species further advantage.

In the Southwest, tamarisk has invaded many mesquite bosques and cottonwood/willow forests (Stromberg 1993a, b). Along lower reaches of the Colorado River, increasing tamarisk invasion limits cottonwood regeneration (Snyder and Miller 1992). In Colorado, dying riparian cottonwoods are being replaced by Russian-olive, raising concern about how a species shift will affect these riparian systems (Hibbert 1979). In New Mexico, the Rio Grande riparian system will likely be dominated by exotic species within the next century (Howe and Knopf 1991).

These shifts toward exotic woody plants could significantly influence wildlife habitat. Carothers and Johnson (1975) and Hubbard (1977) noted reductions in diversity, faunal-plant assemblages, uniqueness, and critical habitat in riparian systems dominated by exotics. Besides influencing terrestrial wildlife and vegetation, loss of native riparian species also may affect invertebrates within streams. Changes in the chemical nature of organic material entering streams coupled with changes in light may affect food available to micro- and macro-invertebrates (Sweeney 1992).

Other exotics also may stress native species in riparian systems. In the Pacific Northwest, Himalayan blackberry (*Rubus procerus*) and (*Scenecio* spp.) turned a tree-dominated riparian zone into a monoculture of shrubs (Newton 1989). Tree species exotic to the Great Plains, such as Siberian elm (*Ulmus pumila*), may be invading riparian woodlands with poorly understood consequences. The recent explosion of annual, biennial, and perennial weeds such as diffuse knapweed (*Centaurea difusa*) is further stressing native riparian species across the West (De Waal 1994).

Inciting Factors

Incitants may be biotic or abiotic, are generally short in duration, and cause relatively severe damage or injury. A plant's ability to recover from this damage or injury is limited because of the unchanging stress created by predisposing factors. Major inciting factors in riparian decline are: grazing by domestic and wild animals, wildfire, recreation, drought, floods, water diversion, insect defoliators, vegetation removal, and beaver (*Castor canadensis*). Their primary impact has been to reduce plant vigor through mechanical damage, soil compaction, disturbance, and increased susceptibility to insects and disease-causing organisms.

Grazing

Past livestock grazing has exacerbated decline in many riparian ecosystems. However, as livestock numbers recede across the West, expanding populations of native ungulates, particularly elk (*Cervus elaphus*), may cause similar damage. Significant concerns with grazing include: (1) soil compaction from trampling; (2) herbage removal that lowers vigor and alters competitive advantages among species; (3) physical damage to plants from trampling, rubbing, and browsing; and (4) changes in fluvial processes that may lower water tables and eliminate germination sites for woody vegetation (Kauffman 1988).

Trampling breaks down soil aggregates and associated soil pore space (Spurr and Barnes 1980). Compaction decreases infiltration rates and water available to plants while increasing surface runoff and erosion (Kauffman and Kreuger 1984). Compacted soils may hold less water in spring and dry out earlier in fall and winter (Steinbrenner 1951). By contrast, hoof action can assist recovery of disturbed streambanks as it creates more microsites for seed germination and reduces the angle of repose of incised streambanks that can aid plant colonization (Rosgen 1996).

Herbage removal can alter vegetation structure and composition (Kauffman and Kreuger 1984) through selective feeding by different herbivores (Spurr and Barnes 1980) and the greater sensitivity of certain plants to trampling and grazing. Kauffman and Kreuger (1984) noted that overgrazing and site deterioration resulted in the replacement of native bunch grasses with Kentucky bluegrass (*Poa pratensis*) in the Pacific Northwest. In the Custer National Forest in North Dakota, grazing and trampling by livestock lead to reproduction failures in woody draws (Hansen et al. 1984). Grazing of cottonwood seedlings limited regeneration along a southern Arizona creek (Glinski 1977). In Yellowstone National Park, a 95% decline in the numbers of tall willows (*Salix bebbiana*, *S. boothii*, *S. lutea*, and *S. geyeriana*) and virtual elimination of willow seed production was due to repeated elk browsing (Kay and Chadde 1991). Large increases in the number and range of elk in the Southwest during the past century could substantially change vegetation in riparian zones (Truett 1996).

Herbage removal also can affect seed production by removing flower buds (Kay and Chadde 1991), reducing plant size, and causing photosynthate to be diverted to replacing needed tissue (Oliver and Larsen 1990). In addition, herbage removal can slow development of riparian systems. Kauffman et al. (1983) noted that grazing slowed succession in riparian systems dominated by willows and black cottonwood (*P. trichocarpa*), with tree densities being consistently lower in stands grazed late in the season. In the Pacific Northwest, livestock and game herds degraded willow-dominated sites, decreased forage production, and reduced or eliminated willows from many suitable habitats (Kindschy 1985, Kovalchik 1987).

Rubbing and trampling breaks or wounds stems and branches, creating entryways for insects and disease-causing organisms. Cytospora canker increased in aspen in Colorado due to heavy grazing (W. Shepperd, USDA Forest Service, 1994, pers. comm.). Stem wounding is required for *Cytospora* sp. to infect, and grazers and browsers were the primary agents of such wounding in northeastern Oregon (Filip et al. 1992). In response to damage, plants divert energy from growth and development to callus (Oliver and Larsen 1990).

Grazing has exacerbated downcutting of stream channels during the past 100 yr (Trimble and Mendel 1995). Cattle, sheep, and elk have weakened streambanks and disrupted substrates that armor channel bottoms, as hoof action initiates channel downcutting (Neary and Medina 1996, Medina 1996). Eventually, channels lose riffle areas, streams migrate laterally, pools become shallow, the water table lowers, and riparian vegetation shifts from hydric to more mesic species.

Riparian Vegetation Removal

Native Americans impacted trees and shrubs in riparian areas of the West, as did settlers removing woody vegetation. There were about 775,000 ac of riparian woodlands in California *circa* 1850. By the late 1800s, these woodlands had been reduced significantly by settlers seeking wood and converting land to agricultural use. By 1977, only 12,000 ac (1.5%) remained (Sands and Howe 1977).

Mechanical equipment, grazing, and water regulation have reduced riparian vegetation by 95% in lowlands of the Southwest (Johnson and Haight 1984). In the northern Black Hills, numerous riparian areas were cleared of willow and shrubs for agricultural use (Froiland 1962). In the Northwest, timber harvesting in riparian zones before the 1970s showed little regard for objectives other than removal of timber volume (Anderson 1985).

Logging affects riparian zones through tree falling and skidding, road construction, and vegetation removal (DeBano and Schmidt 1990). These factors compact and disturb soil, which increases erosion, reduces growth, and further stresses residual vegetation. Vegetation removal also can alter temperature, species composition, and structural diversity (e.g., change an uneven-aged canopy to even-aged).

In the Northwest, fire and conifer logging have left many riparian areas dominated by red alder (*Alnus rubra*), with dense salmonberry (*Rubus spectabilis*) understories, and all but a few scattered conifers have been suppressed and outgrown (Bacon and McConnell 1989). Large conifers were historically a part of these systems.

Soil compaction from logging can minimize the potential for riparian vegetation to use water and nutrients or to regenerate. Soil compaction and associated root damage on skid trails can hinder aspen sprouting (Shepperd 1993), whereas skidding throughout an area can stress residual plants and new regeneration. Splash dams, commonly built to float logs out of harvested areas prior to the 1930s, markedly altered many streams and their associated riparian areas (Sedell et al. 1988).

Water Diversion and Regulation

Dams and diversion canals have significantly altered habitat for woody riparian vegetation. Stromberg (1993b) surmised that altering water resources is among the greatest threats to Sonoran cottonwood-willow ecosystems. Rood and Mahoney (1993) placed damming and water diversion right after livestock grazing and agricultural clearing as causes of decline in cottonwood riparian systems. Crouch (1979) speculated that water management and diversion were the most important factors in cottonwood decline on the South Platte River.

Many riparian species are sensitive to changes in hydrologic regimes that affect flooding periodicities and water table depth. Cottonwood, like many riparian species, is adapted to spring flooding and high water tables. Water diversion has significantly reduced the number of floods, minimized the area available for potential seedbeds, and reduced water tables to depths that prevent established cottonwoods from drawing water. In some areas, dams and diversions have shifted flow patterns from spring or early summer to midsummer or later. Conditions of stress similar to those of extreme drought have occurred, causing “xylem cavitation” in some cottonwoods, which reduces stem hydraulic conductivity and may contribute to their decline (Tyree et al. 1994). In some cases, root systems may not be able to grow fast enough to keep up with the lowering of water tables (Groeneveld and Griepentrog 1985).

Johnson et al. (1976) found less radial growth, low seedling recruitment of cottonwood and peach leaf willow (*S. amygdaloides*), and low survival of boxelder and American elm (*Ulmus americana*) seedlings following reservoir construction along the upper Missouri River. Furthermore, construction of the Garrison Dam altered flood patterns from April-July to February-March and decreased erosion and deposition rates (Johnson 1992). These changes are expected to cause the cottonwood-dominated forest to give way to a forest lower in species diversity and vertical structure and dominated by green ash (*Fraxinus pennsylvanica*).

Water diversion and regulation also have fostered growth of exotics such as Russian-olive and tamarisk that can outcompete cottonwood and willow for available seedbeds. In the Southwest, tributaries of all major rivers have been affected by diversion of some type. For example, by the mid-1900s, water diversion on the Gila-Salt-Verde River system left less than 50% of the original length free flowing (Johnson and Haight 1984).

Development of Fresno Dam in Montana so reduced flood magnitude and intensity that the density of cottonwoods decreased significantly over 45 yr (Bradley and Smith 1986). Even though recruitment of cottonwood seedlings continues, survival is low. Not enough seedlings are growing in the critical zone; that is, far enough above water level to root before flooding disturbs their establishment, yet near enough to draw sufficient moisture to endure. St. Mary Dam in southern Alberta caused similar problems (Rood et al. 1995).

By contrast, a tributary of the South Platte River experienced a striking shift from sparse grasses to dense woody vegetation after dam construction (Hadley and Emmett 1998). The rapid change was linked to channel deepening, reduced sediment- and vegetation-scouring peakflows, and increased dry-season low flows. A primary effect of water diversion is on hydrologic regimes and geomorphic conditions. In some parts of the West, riparian woody vegetation is probably more abundant owing to human-induced disturbances. Thus, some declines that we currently observe may actually be slow recovery to a natural dynamic state (Figure 2). In evaluating whether or not riparian “decline” is really occurring, understanding the nature and trends of past disturbances is important.

Drought

The 1930s drought in the Great Plains killed innumerable trees within riparian zones. In the Pacific Northwest, drought, together with stand densities and species compositions created during more mesic periods, has led to increased mortality, lowered productivity, and insect and disease outbreaks (USDA 1992). Drought minimizes a plant’s ability to photosynthesize that can result in stress or death (Spurr and Barnes 1980). It also affects reproduction by minimizing moisture available for germination, or by altering patterns of flooding that can minimize the creation of seedbeds for flood-adapted species. A flood pattern that maximizes development of sediment beds for seedling establishment and growth is critical to the cottonwood life cycle (Crouch 1979).

Drought also influences native riparian species because they cannot compete with exotics better adapted to drier conditions and minimal flooding. Stromberg (1993b) noted that summer flooding, no flooding, or reduced or altered water tables allowed tamarisk to invade cottonwood-willow systems in Arizona.

Groundwater Decline

Urban and agricultural use has markedly lowered groundwater levels in some areas. Groundwater is important to riparian vegetation during periods of drought or in areas of low seasonal precipitation (Groeneveld and Griepentrog 1985, Stromberg et al. 1992). For example, stress on mesquite woodlands increased with increasing groundwater withdrawal from an ephemeral creek in Arizona (Stromberg et al. 1992). Although summer rains and seasonal surface flow temporarily reduced stress, effects of a decreased water table were not offset and led to continued stress and decline. Along the Carmel River in California, vegetation decline and severe bank erosion were linked to groundwater lowering (Groeneveld and Griepentrog 1985).

Flooding

Flood events can be major inciting factors of riparian decline in certain streamflow regimes. Riparian systems along streams having episodic, high volume, high velocity storm flows can suffer major reductions in woody vegetation. Trees that establish in the floodplain between major floods or on the first terrace can be scoured out by high velocity flows, particularly when canyon walls confine streamflows within narrow channels. In the upper Verde River of Arizona, peak stormflows increased eightfold over a 30 yr period, culminating in a 70 yr return period storm in 1993 (Neary and Rinne 1998). In reaches of the Verde River confined by steep bedrock walls, most sapling and larger cottonwoods and willows were removed by high flows. Most old-growth cottonwoods occur on second and third terraces in wide reaches of the channel.

Recreation

Stresses created by camping, hiking, and off-road vehicles have affected riparian systems. Heavy recreational use can mimic damage caused by grazing. Foot and vehicle traffic compacts soils and impacts vegetation by: (1) reducing density and diversity of herbaceous ground cover, (2) lowering tree vigor, (3) eliminating seedlings and saplings, (4) increasing infection of mature trees by microorganisms, (5) shifting species to plants adapted to disturbance, and (6) increasing the potential for invasion by exotics (Johnson and Carothers 1982).

Younger trees can be reduced or eliminated on sites receiving moderate or heavy recreational use (Manning 1979). In areas of concentrated use, aspen stands suffer from carving, vandalism, trampling, firewood cutting, sucker removal, and soil disturbance (DeByle 1985). This damage reduces advance regeneration and tree vigor while increasing susceptibility to insects and disease-causing organisms.

Insects

Although an array of insects is associated with riparian hardwoods in the western United States (Table 1), their roles in decline are largely undocumented. Most studies address upland environments experiencing large-scale outbreaks. The overall effect of insects within riparian zones may be subtler.

For example, the poplar and willow borer (*Sternochetus lapathi*), introduced from Europe in 1882, feeds on the inner bark and bores into the center of the stem to pupate. In the Black Hills, this borer was common but did not kill many willows; rather the weakened trees could not withstand other damaging factors (Froiland 1962).

Certain insects set the stage for attack by others. The forest tent caterpillar (*Malacosoma* spp.) defoliates many riparian woody species (Johnson and Lyon 1988, Furniss and Carolin 1977). Typically this defoliation alone is not significant; however, it may leave oaks vulnerable to serious outbreaks of the two-lined chestnut borer (*Agrilus bilineatus*) (Johnson and Lyon 1988).

Table 1. Insect species and types associated with hardwood riparian species of the western United States.

Tree species	No. of insect species	Insect type*
Acer negundo	2	1, 2
Betula papyrifera	5	1, 2
Fraxinus pennsylvanica	4	1, 2
Populus fremontii	2	3
P. deltoides v. *occidentalis*	14	1, 2, 3
P. tremuloides	7	1, 2, 3
P. trichocarpa	3	1, 2
Quercus macrocarpa	1	2
Salix spp.	5	1
Tilia Americana	18	1, 2, 3, 4
Ulmus americana	6	1, 2, 4

* Insect Type: 1 = defoliators, skeletonizers, and leaf miners; 2 = borers; 3 = gall

Inciting insects also may provide entryways for disease-causing organisms. Egg-laying by cicadas may have contributed to the high number of cottonwood trees (*Populus fremontii*) infected with bacterial wetwood in central Arizona (Brickler 1993). Jackson and Resh (1989) identified relationships between aquatic insects in riparian zones and the birds and bats that utilize these areas. They also discussed the possible effects of changes in riparian zones on aquatic insect populations.

Disease

Even though many diseases affect riparian hardwood species (Table 2), little is known about their influence on riparian function. As with insects, most of our knowledge is inferred from upland environments. Diseases are primarily inciting factors in riparian decline because they tend to weaken rather than to kill, making plants more susceptible to other factors. An example is infection by true mistletoe (*Phoradendron macrophyllum*) in Arizona and New Mexico, which lowers the vigor of (and occasionally kills) riparian species such as cottonwood, ash, sycamore (*Platanus* spp), and walnut (*Juglans* spp.) (Sinclair et al. 1987, Dahms and Geils 1997).

In Arizona, a rust (*Melampsora* sp.) occurs on six species of willow, some of which may be placed on the U.S. Fish and Wildlife Threatened, Endangered, and Sensitive plant

Table 2. Number of diseases, organisms, and disease types of riparian hardwood trees of the western United States.

Tree species	No. of diseases	No. of organisms	Types*
Acer negundo	5	13	2, 3, 9
Betula papyrifera	10	16	1, 2, 3, 4
Fraxinus pennsylvanica	7	13	2, 3, 4, 5, 6
Populus fremontii	3	3	1, 3, 4
P. deltoides v. *occidentalis*	14	20	1, 2, 3, 4, 7
P. trichocarpa	13	14	1, 3, 4, 7
Quercus macrocarpa	7	8	1, 2, 3, 9
Salix spp.	16	18	1–4, 6
Tilia americana	5	5	1, 3
Ulmus americana	14	18	1–3, 6, 7, 9

* Disease type: 1 = cankers; 2 = stem, limb, and root rots; 3= leaf and foliage diseases; 4 = rusts; 5 = wilts; 6 = nematodes, 7 = bacteria; 8 = virus; and 9 = vascular disease.

species list (Fairweather 1993). Arizona willow (*Salix arizonica*) and Rocky Mountain willow (*S. monticola*) are highly susceptible, with the former having suffered both dieback and mortality. Arizona willow may have little genetic resistance to this rust, or perhaps this rust is predisposing Arizona willow to a secondary pathogen or to environmental stress.

Decline of Mountain alder (*Alnus incana*), a dominant species along many water courses in Oregon and Washington, has been associated with a *Cytospora* species (Filip et al. 1992). The wounding required for infection by Cytospora resulted primarily from cattle, deer, elk, and the mass movement of river ice (Filip et al. 1989, 1992). *Perrenipora fraxinophila*, a heart-rot fungus, is a significant factor in the decline of green ash in shelterbelts in the Great Plains, as is the Digger nematode (*Xiphinema* spp.), which has been identified in the decline of green ash, cottonwood, Russian-olive, and elm (Sinclair et al. 1987). In these cases, disease affected a riparian system's sustainability by increasing stress on specific vegetation components.

Disease levels can serve as a measure of stress. Page (1981) found that the intensity of true mistletoe infection increased with increasing water stress. With increases in disease levels following stress, the trees become highly susceptible to other insect and disease-causing organisms.

Fire

The role of fire in riparian zones dynamics is poorly understood. Riparian areas along the Gila River in the Southwest periodically burned during dry periods (Stromberg 1993a). Riparian woodlands in the Great Plains may be constrained by fire (Severson and Boldt 1978), whereas aspen seedlings were concentrated in riparian zones that had burned to mineral soil during the 1988 wildfires in Grand Teton and Yellowstone National Parks, (Kay 1993).

Fire's effect on competitive advantages among conifers in the Northwest is another example of how fire, or its suppression, influences riparian decline. Without fire, Douglas-fir (*Pseudotsuga menziesii*) and ponderosa pine are encroached upon by shade-tolerant species such as grand fir (*Abies grandis*). The resulting high tree density stresses all species, increasing their susceptibility to pests.

There may be interacting relationships among native and exotic species and fire. For example, Busch and Smith (1992) found that tamarisk had a competitive advantage in water uptake over willow coppice sprouts on burned sites in the Southwest. Accumulation of flammable tamarisk leaf litter also may contribute to the occurrence of episodic fires that destroy willow.

Fire also affects sediments reaching streams (DeBano et al. 1998). Prescribed fires and wildfires of low to moderate severity usually do not lead to much sedimentation, whereas severe fires often produce enough sediment to tax the transport capacity of a stream, leading to channel aggradation (Neary 1995). In the West, long-term sediment production that is fire-related ranges from 70% in mountains of the Southwest to 30% in the Cascade Mountains. In lower relief forests, less than 5% of the stream sediment load can be attributed to fires (Swanson 1981).

Fires add large quantities of coarse woody debris to riparian zones (DeBano et al. 1998). While this debris can benefit aquatic habitats in the short term, it also can cause long-term disruptions of channel morphology, streamflow, and sediment transport. For example, a 1985 fire in a riparian area in southern California killed more than 90% of the alder, all shrubs, and the tops of oaks, sycamores (*Platanus*), and cottonwoods (Barro et al. 1989). After fall rains and wind, sediment and fallen trees and branches clogged channels. Spies et al. (1988) reported a four-fold increase in woody debris in streams of a Douglas-fir/western hemlock (*Tsuga heterophylla*) forest after a catastrophic wildfire, but only a two-fold increase after a partial burn.

Beaver

Beaver have markedly altered the hydrology of riparian systems by expanding floodplains, reducing effects of flooding, and creating depositional areas (Elmore and Beschta 1987, Stromberg 1993a). Beaver dams can change base flow, lateral movement of water, and the water table. They also assist in the accumulation of organic matter and the aggradation of riparian areas, and help in nutrient cycling (Stromberg 1993b). However, beaver also flood and remove established trees, and their dams can fail during high streamflows, increasing peakflows.

In western Nevada, beaver colonies have eliminated black cottonwood from certain riparian zones (Clements 1991). In South Dakota, beaver nearly decimated willows in some areas (Froiland 1962). Wherever beaver built dams, willows were killed for some distance above and below due to drowning or cutting. While secondary, this mortality was a factor in the decline of willow (Froiland 1962). The historical development of this problem is interesting to contemplate considering the likely reduced abundance of beaver in the West today versus that prior to extensive land alteration and trapping that began in the 1800s.

Contributing Factors

Contributing factors include organisms that produce actual symptoms and are often identified as causing decline, but whose attack is related to some stress resulting from predisposing and inciting factors. For example, western pine beetle (*Dendroctonus brevicomis*) killed many ponderosa pine in riparian zones in northwest California (D. Schultz, USDA Forest Service, 1993, pers. comm.). Although mortality was attributed to the beetle, periodic severe drought and high tree densities were inciting factors. Certain bark beetles attack stressed trees; such as ambrosia beetles (*Typodendron retusum*) invading the sapwood of declining aspen and possibly contributing to further decline (Jones et al. 1985).

Drought may have increased the incidence of two-lined chestnut borer, which is causing extensive dieback of oak in woody draws of the Black Hills (R. Dorsett, S. Dakota Forestry, 1993, pers. comm.). Similar observations were noted in the Sulleys Hill National Game Preserve (Antrobius et. al. 1991). Attacks by this insect are highly correlated with prior outbreaks of defoliators such as forest tent caterpillars (*Malacosoma* sp.), cankerworms (*Alsophila pometaria*, *Paleacrita vernata*), and oakworm (*Phryganidia californica*) (Johnson and Lyon 1988). However, little is known about what stress-related interactions within the system trigger decline, and we cannot quantify effects of these interactions.

Rehabilitation of Declining Riparian Systems

We have been discussing factors implicated in the decline of western riparian ecosystems. Now we turn to examining the potential for returning these degraded systems to sustainable states. While a laudable goal, restoration is not necessarily easy or achievable in a short timeframe. Restoration can require decades and even then may be difficult to achieve given the unknown mix of future change in conditions and events.

To rehabilitate declining riparian systems we need to: (1) understand their differing vegetative capabilities, (2) identify and recognize decline symptoms, (3) define and understand their causes, (4) define the value we place on each system (shade, recreation, wildlife etc.), and (5) develop management regimes that address sustainability. We also must recognize that riparian areas are but components of larger watersheds. Thus, sustainability within and around riparian zones involves landscape features, geologic formations, and hydrologic development as well as vegetation (DeBano and Schmidt 1990).

While current declines are predominantly based on direct human and animal impacts, climate and drought also may be involved. Interactions among these factors, not any specific one, frequently lead to decline. Understanding these interactions will help us identify pathways to achieve and maintain sustainability. In some cases, we may be able to reverse decline by removing unnatural conditions such as overgrazing, timber harvest, or recreation (Stromberg 1993a), thus allowing the system to restore itself naturally (passive restoration). In other cases, we may need to alter conditions proactively (Kauffman et al. 1997). For example, reducing stand densities may help restore the moisture needed for continued regeneration. In cases of damming or diversion, we need to accept that returning to a natural, sustainable condition may not be possible. In such instances, we may need to design management techniques to maintain riparian *function* despite changed *condition*. Such techniques might include timed interval flooding, artificial seedbeds, planting to reestablish native vegetation, or thinning and planting to promote species diversity. We also may need to learn the silvics of exotic species so as to utilize them where they best fit into riparian systems.

Treatment can return riparian systems to sustainable conditions. For example, before the 1930s, a riparian system on the Three Bar wildlife area in Arizona was grazed and dominated by chaparral (Brown et al. 1977). Grazing was eliminated in the mid-1930s, and the chaparral was burned and treated with herbicides in 1959. These treatments led to the development of a dense, mixed-broadleaf riparian community that not only provided habitat for black bear (*Ursus americanus*) and turkey (*Melagris gallopavo*) but also changed streamflow from ephemeral to nearly perennial.

Elsewhere in the West, altering the season of grazing or its intensity has significantly improved riparian condition (Chaney et al. 1993). In the northern Great Plains, thinning and light grazing is improving the condition of native woodlands (Boldt et al.1979, Uresk and Boldt 1986). Butler and Goetz (1984) noted the high density of shrubs and trees in lightly grazed areas as contrasted with the more open canopy and better shrub development in moderately grazed sites. In the Pacific Northwest, riparian areas that developed during wetter periods with little fire are being thinned to reduce the potential for insect outbreaks, thereby maintaining forest health (USDA 1992). Systematically analyzing these types of treatments will help to determine which contribute to decline and which may aid recovery.

Recognizing Decline

Determining the status of a woody riparian system requires identification of decline symptoms. These include:

1. Large numbers of mature to overmature trees that are dead or have many dying branches,
2. Little natural regeneration of desired species. Seeds do not germinate or die as germlings because of desiccation, insects, or disease. Established seedlings are severely damaged or killed by grazing and trampling,
3. Reduced vigor and growth of woody vegetation,
4. Increased levels of parasites and diseases such as mistletoes and rusts,
5. Increased populations of defoliating insects such as the forest tent caterpillar,
6. Increased numbers of boring insects such as the two-lined chestnut borer, the poplar and willow borer, and various bark beetles,
7. Above-normal mortality in native species and takeover by exotic species associated with alteration in flooding patterns or competitive interactions for light and moisture, and
8. Significant removal or alteration of native vegetation by grazing, agricultural practices, vegetation harvest, or use of campgrounds and trails.

These obvious visual symptoms of riparian decline may be coupled with less obvious ones (Antrobius et al. 1991). For example, we know little about how degeneration of roots, mycorrhizae, reduction of root food stores, and parasitic fungi influence riparian areas. We also must remember that some woody decline is natural and that certain riparian systems dominated by sedges, grasses, and forbs can be fully functional and sustainable from ecological, hydrological, and geomorphic standpoints.

Studying inventories that include historical information can help us characterize declining and sustainable riparian systems. From inventories, we can characterize the biotic and abiotic conditions within declining and healthy systems. Inventories can identify riparian characteristics while treatment studies can clarify alteration effects. Analyzing this information may highlight specific pathways to both decline and sustainability.

Determining the capability of riparian systems will help us to identify potential products that they can provide on a sustainable basis. It requires understanding aggradation and degradation processes that have resulted in their vegetative structure, composition, and stress level. Determining capability requires the understanding that riparian zones are spatially and temporally diverse (Oliver and Hinckley 1987). Thus, riparian sustainability must be evaluated from the stream reach, stand, watershed, and landscape levels.

Long-term strategies for vegetation management are key to returning declining woody riparian systems to conditions of health and sustainability. Past management regimes have typically been consumptive (i.e., trapping, grazing, gravel mining, and timber harvest) with little consideration for sustainability. In some places, they have left systems in decline. At a minimum, management strategies need to address: (1) the function of vegetation in riparian systems, (2) silvics of the species present, (3) hydrologic condition of the stream, (4) associated landscape and stand features, (5) stream geomorphology, and (6) management objectives. By knowing these factors we can better understand how components of vegetation interact, how abiotic factors affect vegetation, and to what level objectives can be met. Further, we can begin to understand how activities such as grazing interact with other system stressors.

The Function of Vegetation

Structures created by plant communities can stabilize riparian systems. For example, the fibrous and dense root systems of grasses and sedges help prevent streambank erosion (Elmore and Beschta 1987). Their stems provide protective cover for stream banks during periods of high flow and allow for sediment deposition. Coarse, woody roots of trees and shrubs buffer against high stream flows, but can also contribute to bank scouring (Elmore and Beschta 1987). Aboveground parts of woody plants contribute to evapotranspiration, provide food for fauna, modify light quality and quantity, affect groundwater chemistry, and supply leaf litter and debris (Sweeney 1992).

Ecology of the Species

Riparian decline is integrally associated with the plant species present. The plants' adaptability to changing conditions and disturbances must be considered in rehabilitating declining systems. For example, Plains cottonwood (*P. deltoides var. occidentalis*), which typically maintains its vigor for 80–90 yr (Burns and Honkala 1990), is in decline due to drought, excessive withdrawal of groundwater, heavy grazing, gravel mining, invasion by exotics, and damming.

Impacts are often cumulative. For example, damming alters the extent and timing of floods that minimizes areas available for seedbeds. This change favors invasion by exotics that are better adapted to post-damming flood regimes. Heavy grazing tramples or damages trees, all but eliminating young cottonwood; the few that do survive must compete with exotic species for light and moisture. The result is cottonwood under high stress and subject to attack by an array of pests. Knowing how riparian species grow, reproduce, respond to environmental change, and react to attack by insects and disease-causing organisms can help us minimize disturbance effects.

Hydrologic Condition

Erosion and sedimentation caused by flooding can damage or kill some species while creating new sites for establishment of others (Naiman et al. 1992). These processes allow a variety of species to develop into communities that benefit other plants and animals (Oliver and Hinckley 1987).

Dam construction, grazing, and timber harvest have changed hydrologic disturbance patterns. Those, in turn, have altered vegetation structure and composition. Although amelioration of human-caused disturbance has fostered recovery in some riparian areas, it is not always necessary or practical (Elmore 1992). Altering management practices together with understanding stream system hydrology can return some riparian systems to sustainability.

Riparian Zone Structure and Landscape Features

Riparian structures have evolved through both disturbance and adaptive mechanisms that favor colonization by certain species. Learning how these structures developed is key to restoring sustainability. Water has probably been the most significant natural disturbance agent, although fire in ephemeral stream channels during dry periods also has been important.

As riparian vegetation develops, it competes for moisture, nutrients, and light, leading to changes in species composition and structure. The resultant dynamic interaction between growth and disturbance creates openings that provide growing space for new plants. This activity leaves riparian zones with varying compositions of structures, ages, and species.

Management practices have altered these disturbance processes and increased stability in some case. This changes overall system functioning that can alter structure and lead to decline (Kauffman 1988). Knowing the structural conditions in healthy riparian systems and why they exist are key to returning declining systems to sustainability.

Geomorphology of the Stream/Riparian System

Understanding stream geomorphology in a riparian system is important to evaluating proper functioning, condition, trends, and effects of disturbance. Rosgen's (1996) stream classification system allows for a quick determination of channel conditions that depart from normal and can lead to declines in riparian health. Oliver and Hinckley (1987) defined two broad types of riparian zones in the Pacific Northwest: (1) steep slopes surrounding streams containing fast moving water, and (2) flat areas around streams containing slow moving water. Microsites within these types include steep slopes and fast water, eroding straight channels with flat, slow-moving water, stream bars, local flooding, high water, and potentially warm pools. Geomorphologic relationships can either add to or reduce the stresses placed on a system.

For example, if a riparian zone is located within a developing sediment bar, then only species adapted to increased siltation will survive over the long term. Stress to these species may be created by decreases in sedimentation owing to dams and irrigation diversions (Bradley and Smith 1986). Dams will irreversibly alter the hydrologic and morphologic characteristics of a river and thus influence its channel-forming ability, sediment regime and bedload, nutrient retention capacity, and vegetation communities (Pinay et al. 1991).

Management Objectives

Clear management objectives are critical to sustaining riparian systems. For example, annual heavy grazing can prevent natural regeneration and damage the overstory. As trees mature and die, the system gradually shifts from one dominated by trees to one dominated by grasses or shrubs. To maintain suitable grazing for wildlife and domestic livestock while also sustaining the riparian system, we need to know a given riparian system's capability of handling disturbance and be able to manage within those capabilities.

Conclusions

The process of returning declining riparian systems to sustainable states involves learning to identify causes and consequences of stress, identifying the natural range of variability, and using this knowledge to guide management. The present decline in western woody riparian ecosystems results from interactions among a variety of predisposing, inciting, and contributing factors that have increased stress for more than a century, reducing tree vigor, growth, and regeneration. In addition, increased mortality and decreased competitive ability have opened the way to invasion by exotic species. Insect and disease activity also has increased.

We can move declining systems back towards sustainability by using many of the same treatments (grazing, thinning, etc.) that initiated decline (Uresk and Boldt 1986, Elmore 1992). To accomplish this reversal, we must:

1. Understand the process of change. How does a riparian system move away from sustainability toward decline, or vice versa?
2. Understand the basic silvics of riparian species and their interactions with other species.
3. Identify the natural range of variability within these systems. What are the biological carrying capacities of these systems, given the variability of western climates? It is unlikely that we can return to a state within the natural range of variability where there is flow alteration (e.g., dams, irrigation withdrawals) or invasion of exotic species.
4. Identify the functions of system components, especially those that influence the system such as insects and diseases. What conditions within riparian zones cause them to move from sustainable levels to declining levels, or vice versa, and how do these differ from upland conditions?
5. Develop and test management prescriptions designed to return degraded riparian systems to a sustainable state or to keep them sustainable. These prescriptions should be developed from both an individual riparian zone and a landscape perspective, with the former addressing basic site interactions and the latter dealing with interactions within a watershed. These prescriptions also should address sustainable structures for fauna that utilize these areas.
6. Develop monitoring techniques that will measure system health and sustainability. Identify quantifiable methods of utilizing vegetation or other riparian components as bioindicators of system health.
7. Develop and implement a monitoring system that can measure the success or failure of management actions.
8. Develop an adaptive management approach to utilize information gained through monitoring to improve management activities.

Literature Cited

Anderson, M.T. 1985. Riparian management of coastal Pacific ecosystems. P. 364–369 *in* Riparian ecosystems and their management: Reconciling conflicting uses, Johnson, R.R., et al. (tech. coords.). First North Am. Ripar. Conf. USDA For. Serv. Gen. Tech. Rep. RM-120. 523 p.

Antrobius, W., A. Tagestad, C. Brumbaugh, and S. Kresl. 1991. Integrated pest management and forest health for Sullys Hill National Game Preserve. USDA For. Serv., North Dakota For. Serv., and U.S. Fish and Wildl. Serv., Bottineau, ND; Devils Lake, ND. Rep. No. 91-11. 20 p.

Bacon, C., and S. McConnell. 1989. Systems for conifer establishment and growth in riparian areas. *In* Silvicultural management of riparian areas for multiple resources, a coastal Oregon productivity enhancement program workshop. Gleneden Beach, Oregon. 106 p.

Barro, S.C., P.M. Wohlgemuth, and A.G. Campbell. 1989. Post fire interactions between riparian vegetation and channel morphology and the implications for stream channel rehabilitation choices. P. 51–53 *in* Proc. of the California riparian systems conf., Abell, D.L. (tech. coord.). USDA For. Serv. Gen. Tech. Rep. PSW-110. 544 p.

Binford, M.W., and M.J. Buchenau. 1993. Riparian greenways and water resources. P. 69–104 *in* Ecology of Greenways: Design and function of linear conservation areas, Smith, D.S., and P.C. Hellmund (eds.). Univ. of Minnesota Press, Minneapolis, MN. 222 p.

Boldt, C.E., D.M. Uresk, and K.E. Severson. 1978. Riparian woodlands in jeopardy on the northern high plains. P. 187–189 *in* Proc. for National symp. on strategies for the protection and management of floodplain wetlands and other riparian ecosystems, Johnson, R.R., and McCormi, J.F. (tech. coords.). USDA For. Serv. GTR W0-12. 410 p.

Boldt, C.E., D.W. Uresk, and K.E. Severson. 1979. Riparian woodland enclaves in the prairie draws of the northern high plains: A look at the problems, a search for solutions. P. 31–32 *in* Riparian and wetland habitats of the Great Plains. Proc. of the 31st annual meet. Great Plains Agric. Counc. publ. no. 91. 88 p.

Borell, A.E. 1962. Russian-olive: For wildlife and other conservation uses. USDA Leaflet No. 517, Washington, DC. 8 p.

Bradley, C.E., and D.G. Smith. 1986. Plains cottonwood recruitment and survival on a prairie meandering river floodplain, Milk River, Southern Alberta and Northern Montana. Can. J. Bot. 64:1433–1442.

Brickler, J.S. 1993. Identification of a slime flux pathogen in *Populus fremontii* and *Salix goodingii* in Central Arizona. Prelim. rep. submitted to Center for Environmental Studies. Arizona State Univ., Tempe. 3 p.

Brown, E.B., C.H. Lowe, and J.F. Hausler. 1977 Southwestern riparian communities: Their biotic importance and management in Arizona. P. 210–211 *in* Importance, preservation, and management of the riparian habitat, Johnson, R.R., and Jones, D.A. (tech. coords.). USDA For. Serv. Gen. Tech. Rep. RMF & RES–43. 217 p.

BUSCH, D.E., AND S.D. SMITH. 1992. Fire in a riparian shrub community: Postburn water relations in the *Tamarix-Salix* association along the lower Colorado River. USDA For. Serv. Gen. Tech. Rep. INT-289. 232 p.

BURNS, R.M., AND B.H. HONKALA (TECH. COORDS.). 1990. Silvics of North America: Volume 2. Hardwoods. Agric. Handb. No. 654, USDA For. Serv., Washington, DC. 877 p.

BUTLER, J., AND H. GOETZ. 1984. Influence of livestock on the composition and structure of green ash communities in the northern Great Plains. P. 44–49 *in* Proc. of Wooded draws: Characteristics and values for the northern Great Plains, Noble, D.L., and Winokur, R.P. (eds.). Great Plains Agric. Counc., Rapid City, SD. Publ. No. 111. 52 p.

CAROTHERS, S.W., AND R.R. JOHNSON. 1975. Water management practices and their effect on non-game birds in range habitats. P. 210-222 *in* Proc. of the symp. on Management of forest and range habitat for nongame birds, Smith, D.R. (tech. coord.). USDA For. Serv. Gen. Tech. Rep. WO-1. 343 p.

CHANEY, E., W. ELMORE, AND W.S. PLATTS. 1993. Managing change: Livestock grazing on western riparian areas. U.S.EPA, Northwest Resour. Inf. Cent., Eagle, ID. 31 p.

CLEMENTS, C. 1991. Beavers and riparian ecosystems. Rangelands 13:277–279.

CLEVERLY, J.R., S.D. SMITH, A. SALA, AND J.R. DEVITT. 1997. Invasive capacity of *Tamarix ramosissima* in a Mojave Desert floodplain: The role of drought. Oecologia 111:12–18.

CROUCH, G.L. 1979. Changes in the vegetation complex of a cottonwood ecosystem on the South Platte River. P. 19–22 *in* Riparian and wetland habitats of the Great Plains, Proc. of the 31st annual meet. Great Plains Agric. Counc. Publ. No. 91, Fort Collins, CO. 88 p.

DAHMS, C.W., AND B.W. GEILS (TECH. EDS.). 1997. An assessment of forest ecosystem health in the Southwest. USDA For. Serv. Gen. Tech. Rep. RM-GTR-295. 97 p.

DEBANO, L.F., AND L.J. SCHMIDT. 1989. Interrelationship between watershed condition and health of riparian areas in the southwestern United States. P. 45–52 *in* Practical approaches to riparian resource management: An educational workshop, Debano, L.F., Gresswell, R.E., Kershner, L. (eds.). US Bur. of Land Manage., Billings, MT. 93 p.

DEBANO, L.F., AND L.J. SCHMIDT. 1990. Healthy riparian areas as related to watershed stability. P. 55–56 *in* Proc. of conf. XXI erosion control: Technology in transition. Internat. Erosion Control Assoc., Steamboat Springs, CO. 324 p.

DEBANO, L.F., D.G. NEARY, AND P.F. FFOLLIOTT. 1998. Fire's effects on ecosystems. Wiley, New York. 333 p.

DEBYLE, N.V. 1985. Management of aspen for esthetics and recreation, forage, water, and wildlife. P. 223–232 *in* Aspen: Ecology and management in the western United States, Debyle, N.V., and Winokur, R.P. (eds.). USDA For. Serv. Gen. Tech. Rep. RM-119. 283 p.

DEWAAL, L.C. (ED.). 1994. Ecology and management of invasive riverside plants. Wiley, New York. 217 p.

ELMORE, W. 1992. Riparian responses to grazing practices. P. 442–457 *in* Watershed management: Balancing sustainability and environmental change, Namian, R.J. (ed.). Springer-Verlag, New York. 542 p.

ELMORE, W., AND R.L. BESCHTA. 1987. Riparian areas: Perceptions in management. Rangelands 9:260–265.

FAIL, J.L., JR., B.L. HAINES, AND R.L. TODD. 1988. Riparian forest communities and their role in nutrient conservation in an agricultural watershed. Am. J. Altern. Agric. 2:114–121.

FAIRWEATHER, M. 1993. Biological evaluation of *Melampsora* rust disease of Arizona Willow (*Salix arizonica*) on the Apache-Sitgreaves National Forest, Arizona. USDA For. Serv. SW Region. For. Pest Manage. Rep. 7 p.

FILIP, G.M., L.D. BRYANT, AND C.A. PARKS. 1989. Mass movement of river ice causes severe tree wounds along the Grande Ronde River in Northeast Oregon. Northwest Sci. 63(5):211–213.

FILIP, G.M., C.A. PARKS, AND G.L. STARR. 1992. Incidence of wound associated infection by *Cytospora* spp. in mountain alder, red-osier dogwood, and black hawthorn in Oregon. Northwest Sci. 66(3):194–198.

FROILAND, S.G. 1962. The Genus *Salix* (Willows) in the Black Hills of South Dakota. USDA For. Serv. Tech. Bull. No. 1269. 75 p.

FURNISS, R.L., AND V.M. CAROLIN. 1977. Western forest insects. USDA For. Serv. Misc. Publ. No. 1339. 654 p.

GIRARD, M.M., H. GOETZ, AND A.J. BJUGSTAD. 1987. Factors influencing woodlands of southwestern North Dakota. Prairie Nat. 19:189–198.

GLINSKI, R.L. 1977. Regeneration and distribution of sycamore and cottonwood trees along Sonoita Creek, Santa Cruz County, Arizona. P. 116–123 *in* Importance, preservation, and management of riparian habitat: A symp., Johnson, R.R., and D.A. Jones (eds.). USDA For. Serv. Gen. Tech. Rep. RM-43. 217 p.

GREEN, D.M., AND J.B. KAUFFMAN. 1989. Nutrient cycling at the land–water interface: The importance of the riparian zone. P. 61–68 *in* Practical approaches to riparian resource management: an educational workshop, Debano, L.F., et al. (eds.). US Bureau of Land Manage., Billings, MT. 193 p.

GRISSINO-MAYER, H.D. 1996. A 2129-year reconstruction of precipitation for northwestern New Mexico, USA. P. 191–204 *in* Tree rings, environment and humanity, Dean, J.S., et al. (eds.). Radiocarbon, Dep. of Geosciences, Univ. of AZ, Tucson. 889 p.

GROENEVELD, D.P., AND T.E. GRIEPENTROG. 1985. Interdependence of groundwater, riparian vegetation, and streambank stability: A case study. P. 44–48 *in* Riparian ecosystems and their management: Reconciling conflicting uses, Johnson, R.R., et al. (tech. coords.). USDA For. Serv. Gen.Tech. Rep. RM-GTR-120. 523 p.

HADLEY, R.F., AND W.W. EMMETT. 1998. Channel changes downstream from a dam. J. Am. Water Resour. Assoc. 34:629–637.

HANSEN, P.L., G.R. HOFFMAN, AND A.J. BJUGSTED. 1984. The vegetation of Theodore Roosevelt National Park, North Dakota: A habitat classification. USDA For. Serv. Gen. Tech. Rep. RM–113. 35 p.

HIBBERT, A.R. 1979. Managing vegetation to increase flow in the Colorado River basin. USDA For. Serv. Gen. Tech. Rep. RM-66. 27 p.

HORTON, J.S. 1977. The development and perpetuation of the permanent *Tamarix* type in the phreatophyte zone of the Southwest. P. 124–127 *in* Importance, preservation, and management of the riparian habitat, Johnson, R.R., and D.A. Jones (tech. coords.). USDA For. Serv. Gen. Tech. Rep. RMF & RES–43. 217 p.

HOWE, W.H., AND F.L. KNOPF. 1991. On the imminent decline of Rio Grande cottonwoods in central New Mexico. Southw. Natur. 36:218–224.

HUBBARD, J.P. 1977. Importance of riparian ecosystems: Biotic considerations. P. 14–18 *in* Importance, preservation, and management of riparian habitat: A symposium, Johnson, R.R., and D.A. Jones (eds.). USDA For. Serv. Gen. Tech. Rep. RM-43. 217 p.

JACKSON, J.K., AND V.H. RESH. 1989. Activities and ecological role of adult aquatic insects in the riparian zones of streams. P. 342–345 *in* Proc. of the California riparian syst. conf., Abell, D.L. (tech. coord.). USDA For. Serv. Gen. Tech. Rep. PSW-110. 544 p.

JOHNSON, R.R., AND S.W. CAROTHERS. 1982. Riparian habitats and recreation: Interrelationships and impacts in the Southwest and Rocky Mountain Region. RMF&RES, Ft. Collins, CO. Eisenhower Consort. Bull. No. 12. 31 p.

JOHNSON, R.R., AND L.T. HAIGHT. 1984. Riparian problems and initiatives in the American Southwest: A regional perspective. P. 404–410 *in* California riparian syst. conf.: Ecology, conservation, and productive management, Warner, R.E., and K.M. Hendrix (eds.). Univ. of California Press, Berkeley. 1035 p.

JOHNSON, W.C. 1992. Dams and riparian forests: Case study from the upper Missouri River. Rivers 3:229–242.

JOHNSON, W.C., R.L. BURGESS, AND W.C. KEAMMERER. 1976. Forest overstory vegetation and environment on the Missouri River floodplain in North Dakota. Ecol. Monogr. 46:59–84.

JOHNSON, W.T., AND H.H. LYON. 1988. Insects that feed on trees and shrubs. Ed. 2. Comstock Publ. Assoc., Ithaca, NY. 556 p.

JONES, J.R., N.V. DEBYLE, AND D.M. BOWERS. 1985. Insects and other invertebrates. P. 107–114 *in* Aspen: Ecology and management in the western United States, Debyle, N.V., and R.P. Winokur (eds.). USDA For. Serv. Gen. Tech. Rep. RM–119. 283 p.

KARR, J.R. 1996. Disturbance and riparian corridors. P. 43–46 *in* At the waters edge: The science of riparian forestry—conf. proc., Laursen, S.B. (ed.). Minnesota Ext. Serv., Univ. of Minn., St. Paul. 160 p.

KAUFFMAN, J.B. 1988. The status of riparian habitats in Pacific Northwest forests. P. 45–55 *in* Streamside management: Riparian wildlife and forestry interactions, Raedeke, K.J. (ed.). Inst. of For. Resour., Univ. of Washington, Seattle. 277 p.

KAUFFMAN, J.B., AND W.C. KREUGER. 1984. Livestock impacts on riparian ecosystems and streamside management implications...a review. J. Range Manage. 37:430–438.

KAUFFMAN, J.B., R.L. BESCHTA, N. OTTING, AND D. LYTJEN. 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22:12–24.

KAUFMANN, J.B., W.C. KRUEGER, AND M. VAVRA. 1983. Effects of late season cattle grazing on riparian plant communities. J. Range Manage. 36:685–690.

KAY, C.E. 1993. Aspen seedlings in recently burned areas of Grand Teton and Yellowstone National Parks. Northwest Sci. 67:94–104.

KAY, C.E., AND S. CHADDE. 1991. Reduction of willow seed by ungulate browsing in Yellowstone National Park. P. 92–99 *in* Symp. on Ecology and management of riparian shrub communities, Clary, W.B. (comp.). USDA For. Serv. Gen. Tech. Rep. INT–289. 232 p.

Kindschy, R.R., 1985. Response of red willow to beaver use in southeastern Oregon. J. Wildl. Manage. 36(6):26–28.

Kovalchik, B.L. 1987. Riparian zone associations: Deschutes, Ochoco, Fremont, and Winema National Forests. USDA For. Serv. Rep. R-6-ECOL-TP-279-87. 171 p.

Kreuper, D.J. 1993. Effects of land use practices on western riparian ecosystems. P. 321–328 *in* Status and management of neotropical migratory birds, Finch, D.M., and Stangel, P.W. (eds.). USDA For. Serv. Gen. Tech. Rep. RM-229. 422 p.

Leopold, L.B. 1951. Vegetation of southwestern watersheds in the nineteenth century. Geogr. Rev. 41:295–316.

Leopold, L.B. 1959. Climatology and the problems of western grasslands. P. 281–287 *in* Grasslands, Proc. of a symp, presented at the New York meeting of the Am. Assoc. for the Advance. of Sci., Sprague, H.B. (ed.). Am. Assoc. for the Advance. of Sci., Washington, DC. 406 p.

Lins, H.F. 1997. Regional streamflow regimes and hydroclimatology of the United States. Water Resour. Res. 33:1655–1667.

Manion, P.D. 1981. Decline diseases of biotic and abiotic origins. P. 324–339 *in* Tree disease concepts, Manion, P.D. (ed.). Prentice-Hall Inc., Englewood Cliffs, NJ. 399 p.

Manion, P.D., and D. Lachance. 1992. Forest decline concepts: An overview. P. 181–190 *in* Forest decline concepts, Manion, P.D. and D. Lachance (eds.). The Am. Phytopathol. Soc. Press, St. Paul, MN. 249 p.

Manning, R.E., 1979. Impacts of recreation on riparian soils and vegetation. Water Resour. Bull. 15:30–43.

Manning, M.E., S.R. Swanson, T. Svejcar, and J. Trent. 1989. Rooting characteristics of four intermountain meadow community types. J. Range Manage. 42:309–311.

Masters, L.S., J.W. Burkhardt, and R. Tausch. 1991. The geomorphic process: Effects of base level lowering on riparian meadow communities. Rangelands 13:280–284.

Medina, A.L. 1996. Native aquatic plants and ecological conditions of southwestern wetlands and riparian areas. P. 329–335 *in* Desired future conditions for southwestern riparian ecosystems: bringing interests and concerns together, Shaw, D.W., and D.M. Finch (tech. coords.). USDA For. Serv. Gen. Tech. Rep. RM-272. 359 p.

Minckley, W.L., and D.E. Brown. 1982. Interior and Californian riparian deciduous forests and woodlands. Desert Plants 4:250–254.

Minckley, W.L., and J.N. Rinne. 1985. Large woody debris in hot desert streams: An historical review. Desert Plants 7:142–153.

Muller-Dumboise, D. 1992. A natural dieback theory, cohort senescence as an alternative to the decline disease theory. P. 26–37 *in* Forest decline concepts, Manion, P.D., and Lachance, D. (eds.). The Am. Phytopathol. Soc. Press, St. Paul, MN. 249 p.

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. Oconnor, P.L. Olson, and E.A. Steel. 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest coastal region. P. 127-188 *in* Watershed management: Balancing sustainability and environmental change, Naiman, R.J. (ed.). Univ. of Washington, Center for Streamside Studies, Seattle. 542 p.

Neary, D.G. 1995. Effects of fire on water resources—a review. Hydrol. Wat. Resour. Ariz. Southw. 25:45–53

Neary, D.G., and J.N. Rinne. 1998. Baseflow trends in the upper Verde River relative to fish habitat requirements. Hydrol. Wat. Resour. Ariz. Southw. 27:57-64.

Neary, D.G., and A.L. Medina. 1996. Geomorphic response of a montane riparian habitat to interactions of ungulates, vegetation, and hydrology. P. 143–147 *in* Desired future conditions for southwestern riparian ecosystems: Bringing interests and concerns together, Shaw, D.W., and Finch, D.M. (tech. coords.). USDA For. Serv. Gen. Tech. Rep. RM-272. 359 p.

Newton, M. 1989. Community response of riparian vegetation to forest management *in* Silvicultural management of riparian areas for multiple resources, a Coastal Oregon productivity enhancement program workshop. Gleneden Beach, Oregon. 106 p.

Oliver, C.D., and T.M. Hinckley. 1987. Species, stand structures, and silvicultural manipulation patterns for the streamside zone. P. 257–275 *in* Proc. streamside management: Forestry and fishery interactions. Inst. For. Res. Contrib. No. 57, Univ. of Washington, Seattle. 471 p.

Oliver, C.D., and B.C. Larsen. 1990. Forest stand dynamics. McGraw–Hill New York. 468 p.

Page, J.M. 1981. Drought-accelerated parasitism of conifers in the mountain ranges of northern California. Environ. Conserv. 8:217–226.

Peterson, E.E. 1979. Where the coyotes howl and the winds blow free. P. 1–8 *in* Riparian and wetland habitats of the Great Plains. Proc. of the 31st annu. meet., For. Comm., Great Plains Agric. Council, Great Plains Agric. Publ. No. 91, Colorado State Univ., Fort Collins, Colorado. 88 p.

Pinay, G., A. Fabre, P. Vervier, and F. Gazelle. 1991. Control of C,N,P distribution in soils of riparian forests. Landsc. Ecol. 6:121–132.

Powers, R.F. 1991. Soil as a controlling factor in forest health and sustainability. P. 6–8 *in* Proc. of the 1991 annual California forest pest counc, meet., Riose, J., and D. Adams (eds.). Calif. Pest Action Counc., Sacramento. 63 p.

Risser, P.G. 1992. Impacts on ecosystems on global environmental changes in Pacific Northwest Watersheds. P. 12–24 *in* Watershed management: Balancing sustainability and environmental change, Naiman, R.J. (ed.). Univ. of Washington, Center for Streamside Studies, Seattle. 542 p.

Rood, S.B., and J.M. Mahoney. 1993. River damming and riparian cottonwoods: Management opportunities and problems. P. 134–143 *in* Riparian management: Common threads and shared interest—a western regional conference on river management strategies, Tellman, B., et al. (tech coords.). USDA For. Serv. Gen Tech. Rep. RM-226. 419 p.

Rood, S.B., J.M. Mahoney, D.E. Reid, and L. Zilm. 1995. Instream flows and the decline of riparian cottonwoods along the St. Mary River, Alberta. Can. J. Bot. 73:1250–1260.

Rosgen, D. 1996. Applied river morphology. Wildland Hydrology, Pagosa Springs, CO. 352 p.

Sands, A., and G. Howe. 1977. An overview of riparian forests in California: Their ecology and conservation. P. 98–101 in Importance, preservation, and management of the riparian habitat, Johnson, R.R., and D.A. Jones (tech. coords.). USDA For. Serv. Gen. Tech. Rep. RMF & RES–43. 217 p.

Sedell, J.R., P.A. Bisson, F.J. Swanso, and S.V. Gregory. 1988. What we know about large trees that fall into streams and rivers. P. 47–81 *in* From the forest to the sea: A story of fallen trees. USDA For. Serv. Gen. Tech. Rep. PNW-229. 153 p.

Severson, K.E., and C.E. Boldt. 1978. Problems associated with management of native woody plants in the Western Dakotas. P. 51–57 *in* Wyoming shrublands, Proc. of the sixth Wyoming shrub ecology workshop, Johnson, K.L. (ed.). Laramie, Wyoming section, Soc. for Range Manage. 71 p.

Shepperd, W.D. 1993. The effect of harvesting activities on soil compaction, root damage, and suckering in Colorado Aspen. West. J. Appl. For. 8:62–66.

Sinclair, W.A., H.H. Lyon, and W.T. Johnson. 1987. Diseases of trees and shrubs. Cornell Univ. Press, Ithaca, NY. 574 p.

Snyder, W.D., and G.C. Miller. 1992. Changes in riparian vegetation along the Colorado River and Rio Grande, Colorado. Great Basin Natur. 52:357–363.

Spies, T.A., J.F. Franklin, and T.B. Thomas. 1988. Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology 69:1689–1702.

Spurr, S.H., and B.V. Barnes. 1980. Forest ecology. Wiley, New York. 687 p.

Steinbrenner, E.C. 1951. Effect of grazing on floristic composition and soil properties of farm woodlands in southern Wisconsin. J. For. 49:906–910.

Stromberg, J.C. 1993a. Riparian mesquite forests: A review of their ecology, threats, and recovery potential. J. Arizona-Nevada Acad. Sci. 27:111–124.

Stromberg, J.C. 1993b. Fremont cottonwood–Gooding willow riparian forests: A review of their ecology, threats, and recovery potential. J. Arizona–Nevada Acad. Sci. 26:97–110.

Stromberg, J.C., J.A. Tress, S.D. Wilkens, and D.C. Sheldon. 1992. Response of velvet mesquite to groundwater decline. J. Arid Environ. 23:45–58.

Svejcar, T. 1997a. Riparian zones: 1) What they are and how do they work. Rangelands 19:4–7.

Svejcar, T. 1997b. Riparian zones: 2) History and human impacts. Rangelands 19:8–12.

Swanson, F.J. 1981. Fire and geomorphic processes. P. 401–420 *in* Fire regimes and ecosystem properties: Proc. of the conf., Mooney, H.A., et al. (tech. coords.). USDA For. Serv. Gen. Tech. Rep. WO-26. 594 p.

Sweeney, B.W. 1992. Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in Eastern North America, unpubl. rep. 50 p.

Trimble, S.W., and A.C. Mendel. 1995. The cow as a geomorphic agent—a critical review. Geomorphology 13:233–253.

Truett, J. 1996. Bison and elk in the American Southwest: In search of the pristine. Environ. Manage. 20:195–206.

Tyree, M.T., J.K. Kimberley, S.B. Rood, and S. Patino. 1994. Vulnerability to drought induced cavitation of riparian cottonwoods in Alberta: A possible factor in the decline of the ecosystem? Tree Physiol. 14:455–466.

Uresk, D.W., and C.E. Boldt. 1986. Effect of cultural treatments on regeneration of native woodlands on the northern Great Plains. Prairie Natur. 18:193–202.

USDA For. Serv. Public Affairs Office. 1992. Forest ecosystem health on the Deschutes National Forest. Chief's brief. pap. (2/92). NW Region, Deschutes National Forest, Public Affairs Office, Bend, OR.

Welsch, D.J. 1991. Riparian forest buffers: Function and design for enhancement and protection of water resources. USDA For. Serv. NE Area State and Priv. For. Rep. No. NA-PR-07-91. 20 p.