Chapter 16
HISTORICAL AND PRESENT IMPACTS OF LIVESTOCK GRAZING ON FISH AND WILDLIFE RESOURCES IN WESTERN RIPARIAN HABITATS

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INTRODUCTION

What are riparian habitats? How abundant are they? Why should society, and those living in the West, be concerned about their management? How do they function? What is their importance to fish and wildlife? What is their present condition and how can they be better managed? Is domestic livestock grazing contributing to their degradation and, if so, how? Knowledge accumulated on these habitats throughout the West since the turn of the century will provide insight into these questions.

A riparian habitat ". . . is one which occurs in or adjacent to drainageways and/or their floodplains and which is further characterized by species and/or life-forms different from that of the immediately surrounding non-riparian climax" (Lowe 1964:62). In desert areas the transition from verdant floodplain vegetation with an underlying water table to the dry upland supporting creosote bush (*Larrea divaricata*) or big sage (*Artemisia tridentata*) may be less than a meter. Even at elevations where spruce (*Picea* spp.) or grasses dominate the uplands, the transition from the deciduous woodland trees and shrubs to upland species is usually quite distinct.

Soil moisture availability in the floodplain and water table depth are important elements that shape the quantity and quality of the vegetation over the riparian floodplain. The 1-3-year flood event marks a rather distinct boundary between the upland and floodplain vegetation. Upland vegetation is limited from encroaching onto the floodplain by the high water table and prolonged saturated soils from overbank watering during floods. Conversely, the lack of available soil moisture prevents floodplain vegetation from invading the upland.

Studies in hydrology, soils, and geology amply demonstrate that when considered as ecological units, riparian habitats cannot be separated from their watersheds (Hays 1975, Gregory et al. 1991). Therefore, proper management of riparian resources will only be achieved when best management decisions are made for entire watersheds. Management may be applied to individual components of the watershed, but the proper functioning of these ecosystems should always be the ultimate goal. Each species contributes in its own way to the functioning of a healthy ecosystem; the role of some are key or cornerstone to proper functioning of an ecosystem, while others play a more subtle part. Management should always strive to maintain all native species if only for biodiversity and knowledge.

Although there are very important riparian habitats for wildlife along intermittent and ephemeral streams, this chapter will emphasize perennial streams. Intermittent streams have seasonal surface water flow (e.g., from rainfall, snow melt, or water from springs), but there is subterranean flow throughout the year. Ephemeral streams only show surface water as a direct response to heavy rainfall. Though many ephemeral streams are dry most of the time, they have some subsurface flow or seepage that supports a flora different from that of the uplands. There are some drainages with deep alluvium or on bedrock supporting vegetation that may be slightly more robust than the same vegetation (conspecific) in the uplands—these are not riparian habitats.

Because riparian habitats are essentially the upland drainage corridors, they literally are green ribbons anastomosing higher to lower elevations. Though present in all landscapes, their areal or land extent is minuscule when examined as a percent of the total landscape. In California there are 138,800 ha or <0.5% (Sands and Howe 1977). For Arizona, Strong and Bock (1990) estimate 0.5%. On 70,400,000 ha of Bureau of Land Management (BLM) lands there are 40,000 ha of riparian conditions, or <1% (United States Department of the Interior [USDI] 1994). Of the 57,600,000 ha of United States Forest Service (USFS) lands in the West there are 880,000 ha of riparian habitat, or 1%. In the arid Southwest their areal extent is estimated at < 1% (Ohmart and Anderson 1986). At higher elevations with broad, wet meadows they may be ≤ 2% of the landscape, but the mean percent of riparian habitat in the 11 western states may be ≤ 1%.

The importance of riparian areas far exceeds their availability. They are vital to human survival and maintenance of health in the West. They provide drinking water and agricultural waters, which are essential if humans are to persist in the arid West. Functioning riparian systems trap sediments and biodegrade toxic compounds to improve water quality and quantity. The most productive farm lands are on alluvial soils along floodplains. Wilson (1979:82) termed them the "aorta of an ecosystem" because of their importance to the perpetuation of water, fish, wildlife, rangeland, and forest resources. Swanson et al. (1988) termed them one of the most dynamic portions of the landscape.

Gregory et al. (1991) provide an ecosystem perspective modeling spatial and temporal patterns of hydrologic and geomorphic processes, terrestrial plant succession, and aquatic ecosystems in riparian zones. Within the last 3 years numerous articles have appeared on riparian function, structure, energy flow, and landscapes, as well in an excellent and informative book by Malanson (1993).

Streams in riparian habitats may support or provide salmon (*Oncorhynchus* spp.) spawning areas at more northern latitudes, or habitats for the Colorado River squawfish (*Ptychocheilus lucius*) and desert mountain-sucker (*Pantosteus clarki*) in the arid Southwest. Birds reach higher breeding densities in these habitats than any other habitat in the contiguous United States (Carothers et al. 1974), and 60-70% of the total species of western birds are dependent on this habitat to survive and breed. Another 15% use riparian habitat some time in their annual cycle. Knopf (1985) reported 82% of the bird species in northern Colorado were found in riparian areas. Neotropical migrant birds, species that breed in the United States and winter in Mexico, Central America, and South America, make up 60-70% of western riparian breeding birds (Bock et al. 1993). In the Great Basin of southeastern Oregon, of 363 terrestrial species 288 (80%) are directly dependent on them or use them more than other habitats (Thomas et al. 1979). In addition, numerous species of small mammals, amphibians, and reptiles are totally dependent on them. Cross (1985) reported greater species richness and numbers of small mammals in riparian versus upland habitats in Oregon. Large mammals such as deer (*Odocoileus hemonius*, *O. virginianus*), elk (*Cervus elaphus*), desert bighorn sheep (*Ovis canadensis mexicana*), and others use them for water, thermal and hiding cover, as travel corridors, and as a forage source (Thomas et al. 1979, Seegmiller and Ohmart 1981, Krausman et al. 1985).

WHAT IS A PROPERLY FUNCTIONING RIPARIAN SYSTEM?

A recent document (BLM 1993) provides guidelines for assessing proper functioning conditions (PFC) of riparian systems. The PFC are:

> . . . when adequate vegetation, landform, or large woody debris is present to dissipate stream energy associated with high waterflows, thereby reducing erosion and improving water quality; filter sediment, capture bedload, and aid floodplain development; improve flood-water retention and ground-water recharge; develop root masses that stabilize streambanks against cutting action; develop diverse ponding and channel characteristics to provide the habitat and the water depth, duration, and temperature necessary for fish production, waterfowl breeding, and other uses; and support greater biodiversity. **The functioning condition of riparian-wetland areas is a result of interaction among geology, soil, water, and vegetation** (BLM 1993:4).

Another riparian evaluation assessment has a 3-level assessment with the latter being indepth and quantitative (USFS 1992). The final segment covers management implications and interpretations. The review of either of these evaluations provides a good understanding of PFC of riparian habitats.

Photographs of Parley's Fork (Fig. 1) and Red Butte Creek (Fig. 2) in central Utah provide a visual image of streams in PFC. Red Butte Creek has a steeper gradient than the theoretical stream discussed below.

If one assumes an idealized stream running over relatively thick alluvial soils (4.6-6.1 m above bedrock), a moderate or low gradient of 0.4-0.6%, and a floodplain width of 60-90 m on each side of a stream, the channel and vegetation can be generally described. The stream channel itself should be shaped either as a pipe or a U with the vertical legs of the U bent at their tops to provide slightly or well-formed overhanging banks. Overhanging banks exist when a dense root mass provides a physical barrier to the effects of stream velocity and turbulence, creating banks with high surface roughness and high stability (Smith 1976). The width of the stream channel and its depth are a function of the normal stream capacity.

To determine a stream's health, one should evaluate a minimum of 4 km to get a sense of: (1) gradient, (2) geology, (3) types of fluvial material being transported by the stream, (4) condition of the banks and channel, (5) bank material, (6) sinuosity, (7) herbaceous species present, (8) health and condition of the floodplain vegetation, and (9) the distribution of age classes of the major trees and shrubs along the floodplain. Channel geomorphology will probably change over 4 km, so note these changes and why they are occurring.

A very important criterion of PFC is, can the stream access its floodplain in the 1-3-year flood event? The width of the active floodplain, which determines the area available to dissipate stream energy and recharge groundwater, can be judged from the distance the more mesophyllic (i.e., water-loving) vegetation grows from the edge of the stream. Can the stream carry its sediment load? If the stream has more sediment than it can carry, there will be lateral and midchannel bar formation and a decrease in pool depth and numbers as sediments are deposited in pools. Is there an appropriate stream width/depth ratio? In general, a stream in PFC should have a low width/depth ratio, being deep and narrow, except in riffle areas where the stream is changing elevation.

The first indication of a degrading or improving state is usually seen in the condition of the banks. If degrading, they will commonly widen as they erode, changing channel shape from

FIGURE 1. Parley's Fork, Red Butte Canyon, Wasatch Mountains, Utah. Stream depth is approx. 0.8 m, exposed width is 0.2 to 0.3 m. Note the lush herbaceous ground cover, overhanging banks, and the vegetation overhanging the stream. Trees and understory are dense. Photograph by R. D. Ohmart, May 1994.

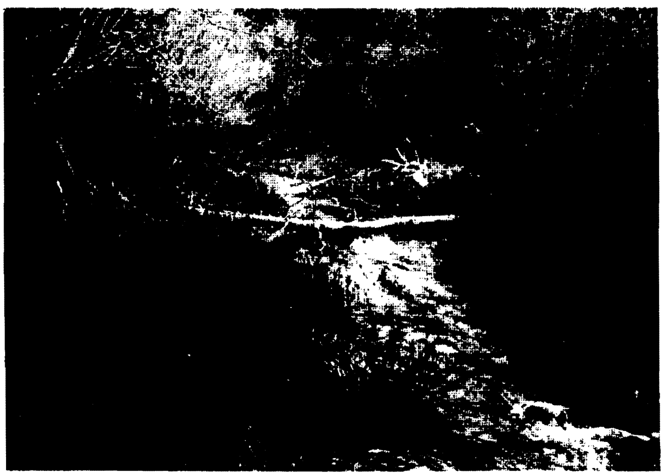

FIGURE 2. Red Butte Canyon, Wasatch Mountains, Utah. Riffle area with dense *Equisetum* spp. Lush herbaceous ground cover and dense shrub-tree understory are typical along the creek. Photograph by R. D. Ohmart, May 1994.

narrow and deep (low width/depth ratio) to shallow and wide (higher width/depth ratio). Road Creek (Fig. 3) shows a point bar, characteristic of sediment overload, on the inside of the bend, and in flood stage the stream is eroding the outer bank. Bear Creek (Fig. 4), a tributary to Road Creek, shows even worse bank instability and erosion. Road Creek is functioning but at risk, while Bear Creek is not in PFC. The above characteristics apply to floodplains with high rock content. Streams on floodplains with low or no rock content will initially downcut; after which, channel widening establishes a new floodplain at a lower level. In either case, the stream begins to lose its ability to access its floodplain and recharge the water table.

The stream channel may be highly meandering or relatively straight as it makes it way through the floodplain, depending on landform steepness and other variables. In a healthy riparian system, streambanks will be stoutly tied together with roots of trees, shrubs, grasses, and sedges. Many of these species are rhizomatous and/or stoloniferous, providing even greater soil stability. A short distance from the stream, woody vegetation in the form of trees, shrubs, or both begins covering the floodplain. At lower elevations, willow (*Salix* spp.) shrubs and/or trees are common along with Fremont cottonwood (*Populus fremontii*) or narrow-leaf cottonwood (*P. angustifolia*). Other tree species such as ash (*Fraxinus* spp.), box elder (*Acer negundo*), sycamore (*Platanus wrightii*), quaking aspen (*Populus tremuloides*), and big-toothed maple (*Acer grandidentatum*), may be common at moderate to higher elevations. At highest elevations and more northern latitudes, shrubby willows may dominate the floodplain with the stream flowing around individual root masses. All of these vegetation elements are vital to fish and wildlife. The aboveground vegetation forms living strata for wildlife and underground roots stabilize soils and reduce sediment transport.

Numerous researchers have emphasized the importance of the combination of the woody roots of trees, shrubs, sedges, and rushes in providing bank stability during flood events (Platts 1981*a*, Beschta and Platts 1986, Elmore and Beschta 1987, Clifton 1989, Elmore 1992). Elmore (1992:443) wrote:

> Riparian vegetation can withstand high velocities of water and still maintain the positive factors of the bank-building processes. The grasses, forbs, rushes, shrubs, and trees produce a variety of fibrous and woody roots that bind and hold soils in place. The woody roots provide physical protection against the hydraulic forces of high flows and allow the fibrous roots to bind the finer particles. This diversity of plant species is much more effective in promoting bank stability than is any single species alone.

Beschta and Platts (1986) similarly reported the importance of the woody and fibrous mix of roots that created high bank stability during flood stage in small streams. Platts et al. (1985) reported that along Big Creek in Utah where there was good bank structure, that in abnormally large floods well-vegetated banks were trapping sediment and actually building better channel banks. These observations corroborate Smith's (1976) finding that there is an inverse relationship between erosion and the percentage of vegetation roots in streambanks. He observed that as the percent of roots increased, bank erosion decreased. Streambanks containing a 5-cm root mix resisted erosion 20,000 times better than nonrooted streambanks.

Well-developed or mature sedge communities may approach the soil-holding capacity of a woody-fibrous root mix in resisting erosion. Manning et al. (1989) measured root length density (total root length) in a *Carex nebraskensis* community on the Sheldon Antelope Refuge in Nevada and found that in 16 cm^3 there were 15 m of root. This root density extended downward for 10 cm before it began to decline, and root depth was measured to 40 cm. The upper roots contained very little soil and these root length densities exceed any measured for any plant community type. This type of root density and depth combined with the tough fleshy leaves overlaying the roots in flood stage creates a formidable barrier to erosion.

FIGURE 3. Road Creek near Challis, Idaho. Large point bar formed on the inside stream bend. New materials are being eroded away on the outside bend in high flows. Photograph by R. D. Ohmart, May 1994.

FIGURE 4. Bear Creek near Challis, Idaho. The stream is entrenched and undercutting it's bank. In the absence of woody-rooted species the fibrous roots of grasses cannot stabilize the bank. The separation of the stream from its floodplain and a lowered water table has allowed sagebrush to invade. Photograph by R. D. Ohmart, May 1994.

The nonwoody vegetation covering the floodplain is referred to as the herbaceous groundcover and serves 3 very important functions during a flood. One is to be smashed over the floodplain soil and repel erosive forces. As the water subsides, the herbaceous vegetation lifts to provide roughness that slows the water and suspended sediments are trapped on the floodplain. Stems of shrubs and trees act similarly. The herbaceous groundcover is also important in creating a boundary layer over the soil to prevent solar heating and moisture from being swept away by wind. The under, mid, and overstory act similarly at a larger scale.

Stromberg et al. (1993) quantified the responses of a riparian floodplain following a 10-year flood event of 368 m^3/sec along the Hassayampa River, central Arizona, in March 1991. Pole-sized cottonwoods suffered 6% mortality in 1991 on the high floodplain, while those lower and closer to the channel had 40% mortality. The 150-200-m-wide floodplain received good overbank watering and a mean of 8 cm of new sediment. Densely vegetated areas received up to 0.5 m of new soil deposits. An abundance of new seedlings of cottonwoods and willows followed the flood along overflow channels and main channel sediment bars.

Natural floods play a vital role in the functioning and health of riparian systems. Normal 1-3-year floods in functioning systems define the stream channel characteristics and are key in maintaining the health and annual productivity of riparian systems. Usually 1 heavy annual flood occurs in late spring or early summer during snow melt. Some systems may experience late summer floods as well. Annual floods in functioning systems irrigate most, if not all, the floodplain and bring in alluvial soils and organic material for soil enrichment. Floodwaters saturate the overbank soils, hastening detrital decomposition releasing new nutrients. The flooding of the overbanks saturates these soils and this water eventually works its way back to the stream. This slow irrigation leaches surface and subsurface salts to the stream and out of the system.

Stream level at normal flow is at or slightly below the level of the floodplain and establishes the level of the groundwater table. In general, a mound of water parallels the edge of the stream as water is forced from the stream by hydrostatic forces between the water and stream channel interface. Along the outside edge of this mound the water table slopes gently downward following topography. The high water table irrigates the roots of the grasses and sedges keeping them constantly inundated. Some trees and shrubs cannot tolerate constant root inundation and grow farther from the stream where their roots are established in the capillary fringe of the water table.

Studies in the area of biogeochemical interactions in riparian areas demonstrate that these habitats are the physical, biological, and chemical links between upland and aquatic environments (Dahm et al. 1987). They serve as phosphorus sinks where the ions are absorbed to clay particles to become trapped as sediment by the floodplain vegetation. The phosphorus ions are then available for plant or bacterial uptake (Cooper et al. 1987). Floodplains in PFC systems have also been reported as important areas for denitrification for maintenance of high water quality (Jacobs and Gilliam 1985). Green and Kauffman (1989) examined oxidation-reduction potentials in riparian zones and demonstrated the importance of nutrient cycling, especially at the land-water interface. They stress the patterning and diversity of vegetation from the stream's edge along this aquatic cline or gradient and how each plant community contributes to high water quality. In an undisturbed watershed in the Sierra Nevada, Rhodes et al. (1985) reported that over 99% of the incoming nitrate-nitrogen was converted to nitrous oxide or elemental nitrogen. Decoupling of the stream and its banks immediately begins degrading water quality. Streams in PFC produce high quality and quantity water and outside influences that alter the soil-water interface seriously impair the functional integrity of the system.

Occasionally a healthy stream will be subjected to a storm event with heavy loss of trees and shrubs. While an unusually heavy flood event may appear to be destructive, it serves to rejuvenate the system. Older or weak, senescent plant communities may be eroded away and banks lost; new sediment beds will be deposited in their place. As riparian plants evolved with floods, they are highly adapted to pioneering into newly deposited soils left as a flood recedes. Many tree and shrub species are rhizomatous and sucker when the roots are hit or abraded by rocks, and most have wind- and water-carried seeds that ripen and are dispersed prior to and during natural floods. These species usually depend upon the presence of new wet sandbars as nursery sites. Many riparian trees and shrubs can also propagate vegetatively; so if young plants are uprooted in 1 area and buried downstream, they root and begin sending up suckers.

STREAM VELOCITY AND EROSIVE FORCE IN FLOOD STAGE

To appreciate the value of riparian vegetation in spreading and slowing bank overflows and reducing flood damage consider the relationship between stream velocity, resistance to flow, and stream gradient. The equation (Chow 1959) for determining water velocity shows an inverse relationship between stream velocity and resistance to flow provided by riparian vegetation. Thus, if the resistance of flow is doubled (increase vegetation), stream velocity is cut in half. The floodwaters are slowed and spread laterally over the floodplain as the vegetation resists flow.

The erosive force or working power of the stream is proportional to the third power of velocity. Therefore, if water velocity over the floodplain is reduced by a factor of 5 the erosive power of water is reduced by 125. These physical relationships highlight the importance of the vegetated streambanks and the ability of the floodplain trees and shrubs to bend and sway but not break in flood events.

If 2 streams (1 with grassed or smooth rock floodplain, the other with dense willows) experienced a flood of about 142 m^3/sec, which is not unusual for a western small-order stream, water velocity in the channel and the floodplain of the rocked or grassed stream would be about 11.3 km/hr, while that over the floodplain of shrubs, trees, and willows would be about 2.3 km/hr. The vegetated floodplain reduced velocity approximately 5 times, which means the erosive power of the flood was reduced by 125.

merged vegetation. These features allow adults to hide and be blocked from the view of other fish. Trout studies in Arizona consistently showed good bank condition with high standing crop (Clarkson and Wilson 1991). A model predicting trout biomass in Wyoming showed that annual flow regime had the greatest influence on trout biomass (Binns and Eiserman 1979). If low base flows in late summer were adequate to keep adult habitats submerged, trout biomass remained high, but if base flows dropped to levels where shorelines and overhanging banks were exposed, trout biomass declined.

Winter habitat is characterized by deep water with low current velocity and protective cover. The latter consists of deep pools with large boulders and root wads (Bjornn 1971). Deep beaver (*Castor canadensis*) ponds provide excellent winter habitat. Behnke (1992) points out the importance of this habitat to winter survival, but it may be overlooked when a river is evaluated for trout habitat.

Trout biologists have suggested a list of conditions necessary for optimum trout habitats (Armour 1978, Bowers et al. 1979, Oregon-Washington Interagency Wildlife Committee 1979, Reiser and Bjornn 1979). A stream should have: (1) a minimum of 60% shade between 1000 and 1600 hours, (2) inorganic sediment should not exceed a covering of 15% of the gravel/rubble substrate, (3) a minimum of 80% of the streambank should be in stable condition, (4) a minimum of 50% of the streambanks should be overhanging, and (5) a minimum of 50% canopy cover for the entire stream.

Streamside vegetation is very important for optimum trout habitat. The fibrous roots of the herbaceous vegetation and the woody roots of trees and shrubs combine to stabilize banks. The trees and shrubs shade the stream reducing water temperatures. The herbaceous groundcover also insulates the soil in winter months, reducing freeze-thaw cycles, which makes them less vulnerable to erosion from high velocity runoff and ice floes (Bohn 1989).

Riparian vegetation also contributes detritus (i.e., leaves, stems, and other woody materials) to the stream. This energy source is extremely important to a trout fishery because > 99% of the stream energy production comes from this source (Bormann and Likens 1969, Likens and Bormann 1974).

Wildlife

In general, wildlife is more responsive to riparian vegetation components and less to physical factors, but some small mammals may be restricted to certain soil types (Harris 1971, Anderson and Ohmart 1984). Riparian habitats satisfy a wide array of wildlife values (e.g., high densities, high species richness, large number of breeding pairs, species that are rare or generally uncommon, many endangered species, large numbers of overwintering wildlife) (Carothers et al. 1974, Carothers and Johnson 1975, Gains 1977, Johnson et al. 1977, Stamp 1978, Knopf 1985).

Specific vegetation components required to fulfill wildlife needs have not been widely studied, even for birds. During 15 years of ecological studies along the Colorado River in western Arizona, Anderson and Ohmart (1984) systematically examined what were the most important vegetation components required to satisfy the ecological needs of groups such as birds and mammals (Rice et al. 1984). In some instances these components were examined at the species level with the objective of designing and revegetating areas that would contain high wildlife values.

Statistical methods were used to group hundreds of quantified communities throughout the Southwest (ignoring plant species composition) to examine common groups. Six structural groups emerged (Anderson and Ohmart 1986) (Fig. 5). This grouping allows comparison of wildlife values of similar structural types among themselves and between themselves based solely on structure or, when desired, plant species included. It also allowed tracking of changes in wildlife values as young communities changed through time to maturity. Ultimately, these analyses allowed the testing of numerous vegetation variables and which were most strongly correlated with highest wildlife values. For example, foliage volume or dense foliage at any layer always supported more species of birds and greater densities than sparse foliage volumes.

There are indirect data on the importance of foliage volume and the willow community in locations other than the Southwest. Duff (1979) reported a 350% increase in raptors and passerine birds on Big Creek in Utah with the inclusion of willows and increased foliage volume in the midstory. Taylor (1986), on the Blitzen River in southeastern Oregon, reported increases in avian species richness and densities 11-13 times higher in low willow foliage versus high willow foliage understory habitats. Similar avian responses have been reported for Sheep Creek in northwestern Colorado (Schulz and Leininger 1991).

Undoubtedly an important component of willows is their rich and diverse insect fauna. Southwood (1961) reported that the Salicaceae supports one of the richest and most diverse insect faunas found among tree families. This rich food abundance must be very attractive to insectivorous fishes, amphibians, reptiles, birds, and small mammals. Further, arthropod abundance has been demonstrated to be a better predictor of densities of insectivorous birds than either foliage volume or foliage height diversity (Brush and Stiles 1986). Rotenberry (1985) has also suggested that birds may respond more to plant taxa than structure, based on resources provided by the vegetation.

In the absence of definitive wildlife-vegetation data at higher elevations and at northern latitudes, desert riparian habitat data can be used in formulating meaningful wildlife management decisions in riparian habitats in the West. A riparian evaluation guide (USDA 1992), by the Intermountain Region of the USFS, supports this as they stress the importance of vertical and horizontal diversity of riparian forests to support greater animal species richness. Cottonwood and/or willow communities have high wildlife values in desert elevations (Carothers et al. 1974, Hubbard 1977, Johnson 1978, Duff 1979, Taylor 1986), at midelevations (Balda 1975, Knopf and Cannon 1982, Wright et al. 1983, Knopf 1985, McEneaney 1988, Thomas 1989), and in quaking aspen at higher elevations (Winternitz 1980, Winternitz and Cahn 1983).

The exotic saltcedar (*Tamarix chinensis*) that began dominating lowland riparian habitats in the twentieth century is now

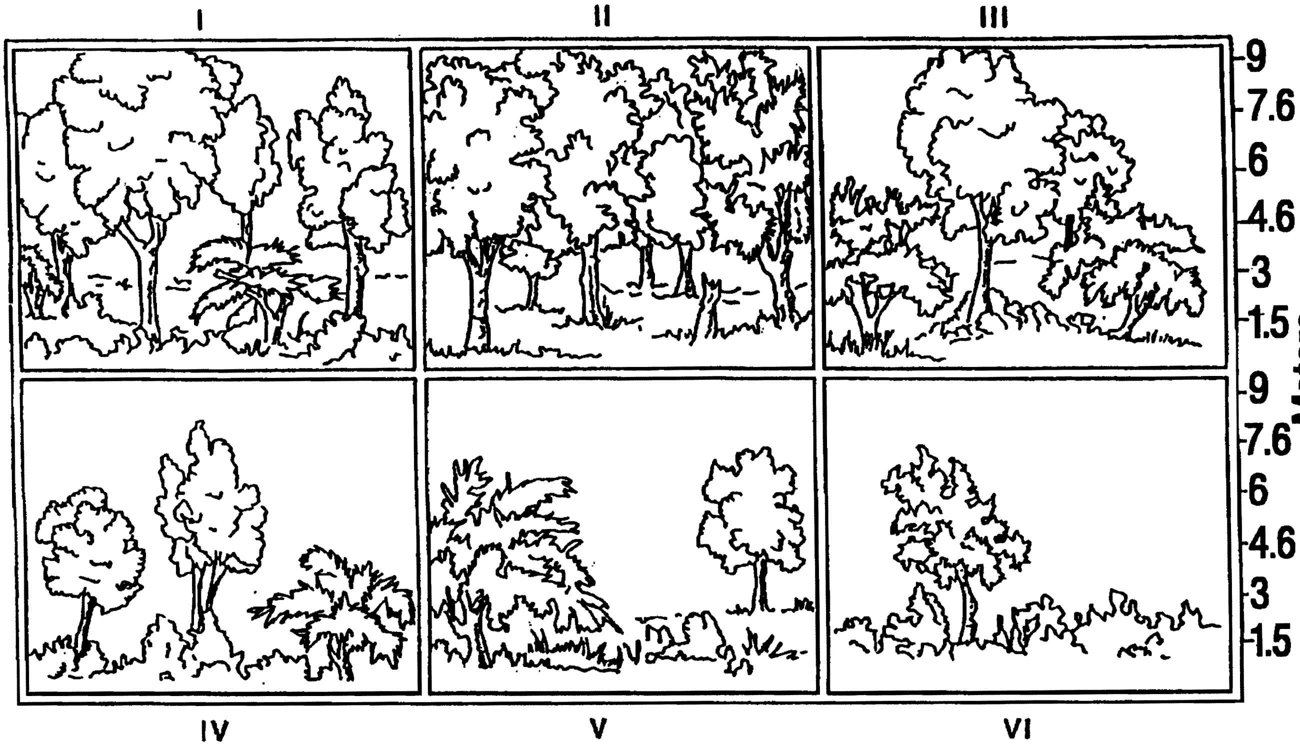

FIGURE 5. Vegetation structural types based on foliage profiles and volumes (Anderson and Ohmart 1984). The paucity of understory vegetation in type II is because of heavy shading by the dense midstory and canopy vegetation. As the forest matures, individuals and groups of trees die, allowing light penetration and an understory to develop.

becoming more common at higher elevations. It generally does not have high wildlife values, but it provides better wildlife habitat than bare soil. Only in exceptional instances have wildlife values in saltcedar begun to approach those of native plant communities (Engel-Wilson and Ohmart 1978, Brown and Trosset 1989). Russian olive (*Elaeagnus angustifolia*), another naturalized exotic, is becoming more abundant in riparian areas in the Intermountain West. Limited information indicates that its wildlife values are not equal that of native riparian trees (Knopf and Olson 1984).

Vegetation components most important to wildlife, in order of importance, are tree species and their densities, foliage height diversity, foliage volume, patchiness, and shrub species and their densities (Ohmart et al. 1988).

Individual Tree Species and Their Densities. The cottonwood-willow component is consistently more important to individual avian species than any of the other vegetation variables (Rice et al. 1984). Avian densities and species richness values in riparian forests are extremely high. Carothers and Johnson (1975) reported 1,059 breeding pairs/40 ha in cottonwood-willow forests on the Verde River in central Arizona; the highest reported values of any habitat in the continental United States.

Foliage Height Diversity. A community with a high foliage height diversity value is tall and structurally complex with high foliage volumes at the herbaceous under, mid, and overstory. This plant community attribute is also important to some arboreal rodents (Anderson and Ohmart 1984) and reptiles (Vitt and Ohmart 1978). Along the lower Colorado River, Ohmart and Anderson (1982) reported 19 breeding bird species associated with the dense canopy layer, 10 with the midstory, and 11 species with the understory. Of all of the possible tree associated communities, the dense mature cottonwood-willow forest has both the important tree species element and the vertical foliage profile, thus providing the 2 most important components in avian habitat selection.

Relative Foliage Volume/m³. The density of the vegetation in the overstory, midstory, understory, and herbaceous layers is extremely important to satisfying the habitat requirements of small mammals, reptiles, amphibians, and breeding birds. Many of the latter are neotropical migrants who tend to be habitat specialists (i.e., foliage gleaners) and as foliage volumes increase in any layer new wildlife species should be added and densities of existing species increased.

Plant Community Patchiness. Patchiness is the unevenness in mixes of different tree species or trees and shrubs horizontally throughout a relatively homogeneous plant community. For example, in the Intermountain West a mix of willow species, when mature, has different heights, providing patchiness. In a cottonwood-willow forest, cottonwoods will generally attain a taller stature at maturity than willows, providing a patchiness component throughout the canopy and midstory layers. In honey mesquite (*Prosopis glandulosa*) communities, quail bush (*Atriplex lentiformis*), wolfberry (*Lycium* spp.), or some other shrub mix provides horizontal patchiness through the community. Patchiness should not be confused with natural edges or ecotones, which are where 2 communities meet. This is considered an intercommunity value, whereas patchiness is an intracommunity value to wildlife.

It has been suggested that patchiness in plant communities provides extra niches or opportunities for bird species to occupy these areas, which would increase avian densities and species richness (Wiens 1989). Comparisons of saltcedar monocultures that have little or no patchiness to very patchy cottonwood-willow habitats, show there are significant differences in avian densities and species richness values (Ohmart and Anderson 1982).

Shrub and Shrub Species Components. Shrubs in desert riparian habitats have received little attention in their importance to wildlife, though willows at more northern latitudes have been reported as important foraging habitats, breeding areas, and as thermal and escape cover for birds (Wright et al. 1983, Krueger and Anderson 1985, McEneaney 1988, Chadde 1989). In desert riparian habitats wolfberry and quail bush are extremely important in the riparian shrub component. Both provide escape cover, but quail bush appears more important as it provides winter thermal cover because it is evergreen, unlike most riparian trees and shrubs. Quail bush's year-round green foliage also supplies a high insect population for foliage-gleaning forms of wildlife; the fruits are consumed as well. A mature quail bush is commonly 2.5 m tall and frequently covers >9.3 m^2. The moist, decomposing leaf litter under these shrubs is replete with detritivorous insects that are heavily fed upon by quail, thrashers, towhees, and small mammals. Along the Colorado River in western Arizona, Ohmart and Anderson (1982) reported that moderate densities of quail bush mixed with exotic saltcedar significantly increased avian species and densities year round.

Patch size or forest community extent is undoubtedly an important wildlife component in broad alluvial floodplains. Most have been modified, fragmented, or so degraded that there has been little opportunity to document the relative importance of this variable to wildlife. Intuitively, a 40-ha riparian forest would fulfill the habitat needs of more species and support greater densities of wildlife per unit area than a 10-ha patch.

For example in a mature stand of willows 1,000 m long and 800 m wide, the outer perimeter (40-60 m) of the stand serves as a buffer area deterring the entry of nest parasites and predators to the core or central portion of the habitat. The core area provides optimum conditions for willow thicket specialists to live and reproduce be they birds, small mammals, amphibians, or reptiles. If this community is fragmented or broken in half by a road or some other interference the core habitat is significantly reduced in size since nest parasites and predators now begin working these new perimeters. This model applies to any expansive plant community type be it a deciduous forest or wetland. Habitat fragmentation can also be highly detrimental to wildlife along streams with narrow bands of vegetation. If the vegetation is destroyed at right angles to the stream so that wildlife populations will not travel across these open areas then gene exchange and dispersal will be stopped until the vegetation regrows to provide cover for movement of individuals.

Relative to migratory wildlife, riparian habitats: (1) provide protective cover and rest areas; (2) supply a rich and abundant insect resource for replenishing fat stores; and (3) serve as wintering habitat for some species. Southwestern riparian habitat importance was summed up by Laymon (1984:595) as "... an essential link for long-distance migrants from the north and are an important wintering ground for many species." Stevens et al. (1977) reported that riparian study plots supported up to 10.6 times as many migrants as paired upland sites. These habitats probably reach their zenith of importance in the Southwest as resting and refueling sites in as they are surrounded by an arid and depauperate upland environment. Terrill and Ohmart (1984) reported that some wood warblers, in an attempt to overwinter as close as possible to the breeding grounds, do so in these habitats as long as winters are mild and insect resources are abundant. Their importance as fueling and stopover sites is explained by studies examining fat reserves, body mass changes, and duration of stay (Cherry 1982, Moore and Kerlinger 1987).

BEAVERS: A KEYSTONE SPECIES IN SMALL-ORDER STREAMS

It is difficult to fathom that before European settlement the beaver population in North America was somewhere between 60,000,000 and 400,000,000 and extended from the Arctic tundra to the deserts of northern Mexico (Seton 1929). Their ability to influence small-order streams is very significant (Naiman et al. 1986, 1988), and yet science is still far from understanding their full role in riparian ecology. Their importance in larger streams as controlling agents may be more significant than currently presumed (see Dobyns 1981).

Naiman et al. (1986, 1988) reported that when beavers remain unexploited they can dramatically alter ecosystem structure and stream dynamics, especially in second- to fifth-order streams. Alteration may be as much as 20-40% by: (1) modifying channel geomorphology and hydrology; (2) retaining sediment and organic matter; (3) creating and maintaining wetlands; (4) modifying nutrient cycling and decomposition dynamics; (5) modifying the plant species composition and physiognomy of plants; (6) influencing the timing, rate, and volume of water and sediment movement downstream; and (7) through the creating of pools and backwaters generating totally new fish and wildlife habitats which results in significant increases in biodiversity. Allred (1980) working in Idaho documented increases in habitat types by beavers and their value to many wetland plants and animals. They may selectively harvest trees to open and modify riparian forest composition and age classes to increase patchiness (Jenkins 1979, 1980). The efficiency of sediment trapping by beaver dams has been reported by Smith (1980), who measured as much as a 90% reduction below dams. Not surprisingly, these habitat alterations are persistent over the riparian landscape for centuries (Rudemann and Schoonmaker 1938, Ives 1942, Neff 1957).

Impounded waters behind dams provide habitat for fish and waterfowl, while emergent and lush vegetation around the pond is favored forage for browsing mammals. Medin and Clary (1991) compared small mammal populations around a willow-dominated beaver pond and an adjacent nonwillow riparian habitat in east-central Idaho. Relative density of small mammals was 3.06 times higher and standing crop biomass was 2.71 times higher in the willow-dominated habitat around the beaver pond. Species richness and diversity were similar between the habitats, but voles (*Microtus* spp.) and shrews (*Sorex* spp.) were more abundant around the beaver pond (Medin and Clary 1991).

To observe a stream in PFC supporting a beaver population is an educational experience, especially with a stream gradient of about 3%. On Rough and Tumbling Creek, Pike National Forest, Colorado, beavers had totally negated the gradient and each dam was a living classroom of hydric to xeric succession. Behind new dams one could see early stages of sediment deposition, older dams showed trapping of sediment by rushes and sedges, others showed stabilization of soils by the woody roots of willows and the forming of backwaters, and ultimately willows and quaking aspen with little surface water. As an interesting exercise, the reader may want to reread the PFC definition (BLM 1993) at the beginning of **What is a Properly Functioning Riparian System** with the beaver in mind.

Some streams may not have the capacity to support beavers for more than a few years (W. Elmore, BLM, pers. commun.). These are streams where shrubby willows dominate the floodplain and the tall deciduous tree element is highly restricted or absent. This may be a food limitation for beavers and after a few years the reduced food supply forces the animals to move to new areas. The importance of beavers in western streams is poorly understood and why they are transient in the above streams needs examination.

HISTORICAL RAMIFICATIONS OF WESTERN LIVESTOCK GRAZING

To give the reader a feel for the evolution and impacts of domestic livestock grazing on western rangeland and riparian habitats, a brief history of this industry's activities seems appropriate. Platts (1979) also gives a thumbnail historical sketch West-wide, and Young and Evans (1989) discuss historical events in Nevada. The West was open and grazing uncontrolled prior to the establishment of the USFS and the various national forests around the turn of the century and the BLM around 1946. A calf could be purchased for $5 and sold a few months later at $65 with the grass and land being free. Arizona (then a territory since statehood was granted in 1912) is used as a model of the consequences of open range and unabated livestock use throughout the 11 western states.

Domestic livestock have grazed portions of the southwestern United States since about 1700. Early Americans did not possess domestic livestock but obtained access to them when the Spaniards brought cattle, horses, sheep, and goats. Simpson (1952) reports that around 1675 there were approximately 200,000 cattle and 2,000,000 sheep on the Central Plateau of Mexico. Within 50 years these numbers would increase to 1,000,000 cattle and 8,000,000 sheep. Ranches were established near the southern fringe of the Sonoran Desert by 1610 (Ewing 1934). By 1694 cattle were grazing the grasslands on the Bavispe River (northern Sonora, Mexico) and headwaters of the San Pedro River (southern Arizona) as reported by Bolton (1948).

Cattle spread rapidly into New Mexico, Arizona, and southern California as each new mission was established. As Father Kino traveled and explored the Pimería Alta, he gave livestock as gifts (Bolton 1948), and in 1701 he made 1,400 animals available to Baja California. The mission in Tucson, San Xavier del Bac, received 700 head in 1702, and by 1703 another 3,500 head were available from Kino's home base in Dolores, Sonora, Mexico. Domestic livestock were extremely important to the new settlers in that they provided a reliable supply of meat, milk, wool, and leather in a harsh and unpredictable environment. By 1750 individual herds of 4,000-5,000 animals were not uncommon (Pfefferkorn 1949).

Many of these cattle became feral as Apaches raided the haciendas and ranches. Bancroft (1883) reported the Apaches preferred horse meat to cattle so raids on ranches were thought to be more for horses and mules, with the cattle being liberated as ranch hands were either killed or abandoned the area. Herds of wild cattle were frequently reported in journals from 1846-1854 (Clarke 1852, Cox 1925, Powell 1931, Durivage 1937, Evans 1945).

The 200-year dominance of the Spanish was essentially terminated at the end of the eighteenth century. Domestic livestock had an important influence on the Indians, and this continued with the Mexicans. Land grants were made along major rivers where water and feed for domestic livestock were most reliable and abundant.

Cooke in 1846 (Bieber 1938), camped near Agua Prieta Creek, wrote that wild cattle were so numerous that the spring had the appearance of a stockyard. Many wild cattle were slain by the officers and an estimated 5,000 watered at the spring (Bieber 1938).

From 1700 to 1850 numbers of domestic livestock grazing in the Southwest were significant and increasing, but stocking rates were much less than those that would be reached between 1850 and 1900. In 1870 there were only 5,000 head of cattle reported in the Arizona Territory (U.S. Bureau of Census 1872:III, 75). Over the next decade this industry grew to provide beef to Army posts, Indian reservations, and growing pioneer settlements. Most cattle brought into Arizona were driven from Texas and Sonora, Mexico. Two drivers brought in over 15,000 head in 4 herds in 1872 (Wagoner 1952). By 1880, 2,500 head were reported east of the San Pedro River in southeastern Arizona; the San Pedro Valley contained 10,000-12,000 head of sheep and about 8,000 cattle. There were 20,000 cattle south of the Gila River and the Arizona Territory contained about 35,000 cattle (Wagoner 1952, U.S. Bureau of Census 1883:III 141-42).

By 1883-1884 in Arizona ". . . every running stream and permanent spring were settled upon, ranch houses built, and adjacent ranges stocked" (Report of the Governor 1896:21). By 1885, there were 435,000 head reported and half were not censused. "This number is being rapidly increased, and within another year it is expected that ranges with living springs and streams will be fully stocked" (Report of the Governor 1885:8).

The 1880s were not a time of tranquility on these open rangelands. Battle lines were drawn and those who controlled water access dictated who grazed the range. Many ranchers recognized the problems of overgrazing but could do little about it. More cattle came while established ranches continued to build their herds. By 1890 it was estimated that >1,000,000 head grazed the territorial ranges of Arizona, and possibly as many as 1,500,000 (U.S. Bureau of Census 1895:I, 29). Peterson (1950) reported >2,500,000 head in Montana, >2,000,000 in New Mexico, and 1,250,000 for Utah and Wyoming in 1890. It is estimated that about 19,000,000 cattle and sheep were grazing the arid West in the late 1880's (General Accounting Office [GAO] 1988). Wilkenson (1992) estimated 26,000,000 cattle and 20,000,000 sheep in the western United States at the end of the century.

Drought struck the cattle industry in Arizona and adjacent states from 1891 to 1893. In 1891 the Governor (Report of the Governor 1896:22) estimated 1,500,000 head of domestic livestock in Arizona. Poor summer rains, coupled with reduced winter moisture, intensified overgrazing to the point that it would be extreme before the drought was over. Cattle died on poorer ranges in the hot dry months of May and June of 1892. Below-normal rains in July and August compounded the problem throughout Arizona. By late spring of 1893 the Governor (Report of the Governor 1896:22) reported the losses as "staggering." Land (1934) stated "Dead cattle lay everywhere. You could actually throw a rock from one carcass to another." J. W. Toumey, Chief Botanist in charge of Grass and Forage Plant Investigations for the Arizona Experiment Station in Tucson in 1891, wrote regarding the southeastern Arizona grasslands, "There are valleys over which one can ride for several miles without finding mature grasses sufficient for herbarium specimens without searching under bushes or in other similar places" (Bahre 1991:113). Livestock mortality estimates were placed at 50-75% (Report of the Governor 1896:22). Wagoner (1952:120-21) supported these mortality estimates for Pima and Cochise counties. Even if mortality rates were only 30% on better rangelands, the ecological destruction of watersheds and riparian habitats in Arizona were easily predictable. Vast areas of rangeland were left barren and unprotected from erosion by wind and rain (Hastings and Turner 1965, Dobyns 1981). In heavy storm events topsoil eroded into the now highly weakened and poorly vegetated riparian habitats. Mature riparian forests were scoured out, leaving more soils vulnerable to erosion from the next storm. With heavily reduced or no groundcover on Arizona watersheds, even small storm events resulted in high surface runoff and heavy soil erosion. Even if flood conditions were conducive to seedling establishment, "continued overuse of riparian bottoms eliminates essentially all reproduction as soon as it becomes established" (Davis 1977:60).

Development of the grazing industry in the other western states does not differ dramatically in timing and consequences to both watersheds and riparian habitats (McArdle et al. 1936, Anderson and Harris 1973, Adams 1975, Behnke 1978, Meehan and Platts 1978, GAO 1988, Chaney et al. 1990). Apparently ranges in northeastern Nevada suffered similarly and during winter of 1889-1890 there was a 95% loss of cattle (Young and Evans 1989). The legend was that one could walk for 161 km along the Mary River, a fork of the Humboldt River, and step from carcass to carcass and never touch the ground (Young and Evans 1989).

Not only did cattle starve, but the resources suffered as well as depicted for Chaco Canyon, New Mexico (Chapman 1933). The first white settlers in 1849 reported the streambed was approximately 3 m wide and 0.6 m deep. In 1924, the stream had entrenched to 10 m and the arroyo was about 100 m wide. The ecological balance had been so ". . . disturbed by overgrazing, erosion has moved a thousand fold more soil in 50 years than in the preceding ten centuries" (Chapman 1933:75). In 1924, Bryan (in Chapman 1933) listed 21 important streams in Arizona, Colorado, New Mexico, and Utah. All streams had floodplains supporting forests of cottonwoods and willows and at that time only supported scattered sage (*Artemisia* spp.), greasewood (*Sarcobatus vermiculatus*), or mesquite.

Deterioration of western riparian systems began with severe overgrazing in the late nineteenth century, and extensive field surveys in the 1980s demonstrate that much of them are in the worst condition in the history of this nation (Chaney et al. 1990). Drought may not have intensified overgrazing as abruptly in the other western states, but the ecological consequences of overgrazing to riparian habitats were similar, throughout western rangelands.

Overgrazing of public lands continued virtually unabated into the twentieth century. Range conditions similar to those in Arizona were reported by Esplin et al. (1928) on lands in Utah, by Keck (1972) in the Great Basin, and by McArdle et al. (1936) when they provided descriptions of unclaimed public lands (now BLM lands). They reported that approximately 84% of these lands had lost more than half of their forage value and forage was depleted on an average of 67% throughout the West.

Overgrazing of National Forest lands "became so critical" (Platts 1981*a*), that the Taylor Grazing Act was passed in 1934 to protect the remaining unclaimed public land and stabilize the livestock industry. Though this action established allotments and adjudicated numbers on these unclaimed public lands, it was, at best, token service to overgrazing on public lands in the 11 western states. On USFS land when permitted livestock numbers were assigned to permittees they were frequently too high. Most USFS lands were not fenced until the 1930s (Bahre 1991). Bahre (1977:27) quotes old timers stating, "The cattle went where the feed was when there was open range, whereas today with fences and supplemental feeding, the cattle stay in pastures for longer than the grass can feed them, ruining the land." The only importance given to riparian habitats during this period was their value in providing extra forage and water for livestock. Up until the late 1960s, riparian habitats were viewed as sacrifice areas. The more valuable grazing allotments contained ≥ 1 perennial streams within their boundaries.

LIVESTOCK IMPACTS TO RIPARIAN HABITATS

This chapter deals with the effects of domestic livestock grazing on riparian habitats, but the reader should be aware that other human activities, both past and present, have destroyed and heavily degraded riparian habitats as well. For example, the virtual elimination of beavers by trapping undoubtedly had a large impact on riparian habitats throughout the West. These animals have major influences on small-order streams and their removal must have radically altered water retention and sediment trapping capabilities of streams. Naiman et al. (1986, 1988), Ehleringer et al. (1992), Elmore and Kauffman (1994) and many others share similar conclusions.

In more recent times, western water management has destroyed and degraded untold thousands of hectares of riparian habitats along major perennial rivers (Stevens et al. 1977, Ohmart et al. 1988). Reservoirs inundate many thousands of hectares of riparian habitat and regulated flows below dams have heavily degraded riparian habitats by stopping or highly altering natural floods. Without natural floods the life cycle of cottonwoods and willows is broken. These moderate-lived tree species persist for many years but eventually disappear because no seedlings are produced as replacements. Along heavily managed streams, channelization and riprapping of banks follow dams and reservoirs. Vegetation is stripped from the banks to place large boulders or riprap, and channels are deepened by dredging. This further decouples the stream from its floodplain and lowers the water table, drying up old oxbows and marsh areas. With the threat of floods eliminated, farming expands on the alluvial floodplain allowing rapid conversion of native habitats to cotton, alfalfa, and other farm crops. Evaporation from reservoirs and leached salts in return irrigation flows to the river increases downstream soil and water salinities, providing optimum conditions for the rapid invasion of saltcedar. Other water management projects such as cutting riparian trees to salvage or save water was undertaken along many perennial streams in the West. Activities such as logging, mining, groundwater pumping, construction of roads, woodcutting, offroad vehicle use, and uncontrolled recreation have also degraded riparian habitats (Busby 1979, Noh 1979, Swan 1979). In general, water management and groundwater pumping has had its greatest impacts to western riparian habitats at lower elevations along most perennial rivers, and domestic livestock grazing has manifested itself ubiquitously at all elevations in the West.

General Considerations

All evidence indicates that virtually all riparian habitats received unmanaged grazing throughout the 11 western states as the livestock industry developed (Elmore and Kauffman 1994). Even Grand Gulch in southeastern Utah, with its vertical sandstone walls of 61 to 122 m, eventually had trails built so that livestock could access the forage (Blackburn 1993). Few western streams with significant forage availability escaped domestic livestock grazing.

Use of the term "unmanaged livestock grazing" refers to the practice of releasing livestock into an area without any planned riparian growing season rest or measures designed to protect the health of the vegetation along the stream or its floodplain. Unmanaged grazing always results in excessive utilization in riparian areas, impairment of plant species vigor, and physical damage to the channel and banks.

Unmanaged grazing of riparian systems has been and continues to be practiced. Today even though most allotments have management plans, all were designed to meet phenological growth requirements of upland vegetation. Watersheds may benefit from these grazing approaches, but riparian habitats are degraded under these plans and will continue to be until management changes are made.

When livestock are put into an allotment or large pasture, they go where they wish or, in many instances, riders drive the animals to wet meadows or other riparian areas where forage and water are abundant. In cow-calf operations the veteran cows know the allotment and where they want to be. Cattle, like most animals, have home ranges, favorite foraging areas that usually include some or all of a riparian habitat, and centers of activity (Martin 1979).

Riparian habitats provide the 4 basic requisites essential to wildlife or domestic livestock: food, water, cover, and space. The attractants of lush vegetation, water, and shade are such that cattle will spend 5-30 times longer in riparian habitats than adjacent uplands, based on areal extent (Skovlin 1984). Cattle congregate in the floodplain in the hotter, drier summer months, imposing heavy use during the heart of the growing season, and in many instances throughout the growing season. Platts and Nelson (1985) reported nearly 100% herbage removal in riparian habitats in the semiarid big sagebrush zone. "Because cattle prefer stream side environments, deterioration of riparian habitats has been significant and much of the deterioration continues" (Platts 1979:48). If grazing use is year-round or even extends into the cooler months, some livestock may disperse into the uplands, but enough will remain in the riparian area to disallow seed development or stored energy reserves for winter. Reduction in livestock numbers is not a management approach to eliminate degradation to riparian habitats.

This was demonstrated in Nevada on Mahogany Creek, where herd size was reduced in efforts to improve trout habitat. Dahlem (1979:34) concluded that

> Based on photographic evidence and data availability, one fact is apparent. The reduction in livestock grazing but continued annual use, had little beneficial effect on riparian habitat along Mahogany Creek. Only after complete removal of livestock use by fencing was significant riparian habitat improvement accomplished along Mahogany Creek.

Gus Hormay related to Olson and Armour (1979:69):

> Vegetation in certain areas, such as meadows and drainage ways, are invariably closely utilized under any stocking rate or system of grazing. Such use may be detrimental to wildlife, esthetic or recreational or other values. Where this is the case, about the only way to preserve values is to fence the area off from grazing. Reducing livestock or adjusting the grazing season usually will not solve such a problem.

The presence of cows (wt ±400 kg each) and/or bulls (wt ±800 kg each) concentrated along streams, foraging along streambanks, and constantly crossing the stream, either season long or year-round, causes extensive physical damage to banks and the channel. That, combined with vegetation removal by each animal (about 350 kg of air dry-forage monthly) for >100 years over most western rangelands has had a devastating effect on riparian systems. Busby (1979) contends things are better in the uplands and he may be correct, but all observations indicate that riparian habitats are highly degraded and generally continue in that state. The Oregon-Washington Interagency Wildlife Council (1979), based on numerous studies in the 11 western states, identified domestic livestock grazing as a major factor in causing serious reductions in wildlife habitat productivity. It was suggested by Bowler (1976) that overgrazing is the largest environmental problem in the United States. Szaro (1989) after extensively surveying riparian conditions in the Southwest reported that livestock may be the major disturbance factor in western riparian habitats. The GAO (1988:11), after talking with agency officials (USFS and BLM) and examining many studies, stated, "Poorly managed livestock grazing is the major cause of degraded riparian habitats on federal range lands." Mosconi and Hutto (1982) working in Montana, suggested that domestic livestock grazing is the major cause of riparian habitat disturbance in the West. Chaney et al. (1990) stated that "extensive field observations in the 1980s suggest riparian areas throughout much of the West are in the worst condition in history." Carothers (1977:3) wrote that "... the most insidious threat to the riparian habitat today is domestic livestock grazing."

A GAO (1988) report was very negative on unmanaged livestock grazing and the condition of riparian habitats in the West. It also dealt with restoration of some riparian areas and how these restored areas were highly beneficial to the permittees by providing advantages other than more forage production. These managed areas showed high soil stability and improved range conditions. I have thoroughly reviewed the GAO (1988) document and from my many years of assessing riparian habitats on public lands, interacting with a multitude of USFS and BLM personnel, and working with permittees, I can only say that, it is the most candid and valid assessment of conditions and problems facing riparian restoration. In most instances management knows the problem and generally how to solve it. However, resistance or total opposition by the permittee (and, sometimes agency personnel) and the cost of making changes, severely slows or stops any progress toward better riparian habitat management. Meehan (1991:9) working with salmonid fishes comments on domestic livestock grazing and stream improvement, "Persuasion has been difficult, and change has occurred slowly."

The importance of livestock forage production in riparian habitats is demonstrated in northeast Oregon where 1 ha of moist meadow soils has the potential grazing capacity of 10-15 ha of forested range (Reid and Pickford 1946). These wet meadows represent ≤ 2% of the range and produce approximately 20% of the forage (Roath and Krueger 1982). They further report that because of the way livestock concentrate, the steepness of terrain, and poor water distribution away from the stream, in reality the 2% wet meadow is producing 81% of the practically usable forage in the Blue Mountain grazing allotment.

Most plants in the floodplain are highly palatable to livestock. Sedges in the genus *Carex* maintain a relatively constant level of crude protein throughout the growing season and until the first killing frost (Kauffman and Krueger 1984). Many sedges in riparian habitats in the Pacific Northwest have higher protein and caloric content than key upland forage species (McLean et al. 1963, Skovlin 1967, Paulsen 1969). Not all allotments in the West have the sedge component or broad wet meadows, but the relative value of the riparian forage (plus water availability) to the drier uplands is about the same throughout the West.

Reasons for Management Change

"There is a general acceptance by managers today that most riparian areas are in an unacceptable condition and that approaches to restoration in the past have had limited success" (Elmore and Kauffman 1994:219). The above statement is very true but instead of "most riparian areas" my experiences are that almost all riparian areas are in unacceptable condition. To avoid greater problems that ultimately may exclude grazing in riparian habitats, agencies and permittees should immediately begin to undertake livestock management in riparian habitats.

Important riparian issues loom on the horizon, such as the continued listing of endangered species, more species being considered for listing, water quality, and recreation. If neotropical migrant birds are unquestionably found to be declining in the 11 western states because of domestic livestock grazing, this will elevate the significance of riparian habitat condition to a new level. The affluent and well-educated cadre of birding enthusiasts that pursue this hobby will exert tremendous political pressure on elected officials and federal agencies for immediate legislation to protect riparian habitats. Spring will not be totally silent (Carson 1962), but 60 to 70% of the songbird species breeding in riparian habitats in the western North American are neotropical migrants (Bock et al. 1993).

The willow flycatcher (*Empidonax traillii extimus*), a neotropical migrant, has recently been listed as endangered (U.S. Fish and Wildlife Service 1995). Approximately 718 km or more of streams may be included as critical habitat for this species in the Southwest. Listing packages are in preparation for several other birds (all neotropical migrants) that will only exacerbate user problems in that grazing decisions then must pass Section 7 Consultation under the Endangered Species Act. The gravity of these listings and the rapidity of their occurrence is seen on BLM lands where 10 years ago there were 75 wildlife species federally listed as threatened or endangered. Today, there are 216 and 1,000 more species being readied for listing (Horning 1994). These numbers do not include plants directly affected by livestock grazing. Western livestock growers perceived the Reagan and Bush Administrations as allies, but in reality these elected officials harmed the industry by not enacting slight management alterations over their 12 years that could have avoided drastic management changes today. As it is, permittees may lose use of pastures or possibly entire allotments as new species are federally listed and critical habitats delineated.

What financial burden is being placed on the taxpaying citizens of this country in attempts to recover some of these species that are now endangered from domestic livestock grazing? An indepth cost analysis has not been attempted, but there are some data for mineral extraction activities (Losos et al. 1995). A few examples provide insight into this question. In 1989, BLM, in trying to recover 5 bird species, averaged > $700,000 per species

(McClure et al. 1991). Two million dollars have been expended over the past 20 years to recover the Gila trout (*Oncorhynchus gilae*), and another million will be expended by the year 2000 (U.S. Fish and Wildlife Service 1993*b*). The species is in greater peril of extinction today than when recovery efforts started because the team wants to avoid controversial issues such as domestic livestock grazing. The U.S. Fish and Wildlife Service (1994) plans to spend $15.5 million over the next 12 years to recover the desert tortoise (*Gopherus agassizii*). Horning (1994) estimates that the BLM total recovery cost for the Lahoutan cutthroat trout (*Oncorhynchus clarki henshawi*) will exceed $14 million, with fencing costs being estimated at $3,000/km. The U.S. Fish and Wildlife Service Office in Phoenix, Arizona, will spend $1.5-2 million to fence Apache trout (*O. apache*) habitat, while cattle continue to overgraze and degrade other streams containing the fish (Horning 1994). These costs have prompted the National Wildlife Federation to press for federal policies that include a thorough cost-benefit analysis to find the most cost-effective approach; graze or not graze riparian habitats (Horning 1994).

Not only are there economic costs but the ecological costs (see Fleischner 1994) from the disruption of ecosystems and the alteration of riparian community structure may well be of greater economic cost than attempting to recover threatened or endangered species. Riparian systems show moderate resistance to grazing and are resilient once livestock are excluded. Systems may return to a semblance of PFC, but can they ever be recovered to their original functioning condition since being so heavily degraded for so long? They are vital to westerners for cheap high-quality and quantity water and to fish and wildlife.

Other driving forces are where sediments are being carried into trout or salmon streams containing listed species (Anderson et al. 1993). Platts (1991) examined the effects of livestock grazing on salmonid fishes and of the 21 studies that he examined all but one had stream and riparian habitats degraded from domestic livestock grazing. All showed habitat improvement when grazing was prohibited. The exception was herded sheep grazing on a well-managed sheep allotment.

There will also be increased restrictions under the Clean Water Act on nonpoint pollution programs with legislation being encouraged by groups such as Mothers for Clean Water. These are but a few of the compelling reasons that managed grazing of riparian habitats is critical if permittees are to continue using them on public lands.

Phases of Pristine Riparian Habitat Degradation with Unmanaged Livestock

Riparian habitat degradation is broken into 3 phases in the hope that it will be easier for the reader to visualize and understand the temporal, physical, and biological changes that occur in each phase. With riparian degradation in 3 phases, along with the knowledge of what biotic and abiotic components are most important to fish and wildlife, it becomes clear when and why certain animal groups began to be stressed by habitat degradation. It is also impressive how long many of these species have managed to persist in spite of this stressor and its duration.

Phase I. Degradation is estimated at 1-10 years. In general, streambanks and channel morphology, herbaceous and understory vegetation, and water quality are changed. The herbaceous groundcover species mix, if not eliminated after a few years, also changes from highly palatable, better soil-holding species to less or even nonpalatable, shallow-rooted annuals and perennials. These changes come about from physical changes of the banks and channel, elimination of herbaceous and understory vegetation, increased erosion from normal and heavy flood events, channel entrenchment, and lowering of the water table.

The concentration of livestock in riparian areas on a yearlong basis or even total growing season use exacerbates the process of bank degradation and stream siltation. As the stream channel deteriorates by widening, more water from each flood event is carried in the channel with greater velocities and erosive force, further widening the channel through in-stream erosion. Channel widening often triggers channel straightening and channel incision, resulting in a dropping water table.

Phase II. Phase II occurs over 100-125 years and as it begins there is a full complement of tree species with high densities, a mature foliage profile, and high foliage volumes at the midstory and canopy layers. In willow-dominated systems without the taller tree element, Phase II may only take 50 or so years with willows being eliminated or becoming highly scattered. Willows managing to persist have a highly modified hourglass physiognomy. Most recruitment of young trees and shrubs ceases and as the youngest trees that escaped the initial grazing mature there are no replacement forests.

A common statement is, "I've lived on this creek all my life and it has always looked the same." In general that statement is true, but after the initial riparian degradation in Phase I, like the aging process, the changes go unnoticed by casual observation. No one living today observed Phase I (but see San Pedro River wildlife consequences) but it did not go unobserved by ranchers (see Bahre 1991). People do not notice themselves aging on a daily basis, but photographs at 5-year intervals show definite changes.

Through the past 100 years deciduous riparian forests, once continuous, have been slowly fragmented leaving small forest islands that have since been subfragmented as individual trees die. Some trees have died of old age, others in blow downs, many have been washed out in more violent floods after watershed and phase I degradation, beavers (where they persist) have girdled and killed many, and others have been left to die with roots perched above declining water tables as a stream downcuts. The slow loss of individual trees through time has progressed to the point today that foliage volumes in the remaining canopy and midstory layers are very low. The decline of the cottonwood-willow gallery forest in Arizona has been so rapid that funds were allocated to quantify the total amount and riparian community types for the state. There are 106,714 ha of floodplain along Arizona's 8,097 km of perennial streams (Valencia et al. 1993). Of the total floodplain, 4.2% or 4,482 ha are remaining cottonwood-willow association. The Arizona Nature Conservancy (1987) reported this community type as the rarest forest type in North America.

In the Zuni Mountains of western New Mexico in the Cibola National Forest streams prior to 1850 were described by hydrologists as, "... narrower, deeper and less entrenched. Floodprone areas were broad and densely occupied with hydric and mesic vegetation" (Jackson 1994:4). The author cites extensive clearcutting and extreme overgrazing as being major contributions to the reduction of the original riparian vegetation by 70-90%. Riparian habitat losses are ≥90% along the East Fork of the Gila River in the Gila Wilderness where cattle grazing is the primary stressor (Ohmart In Press).

Phase III. Phase III is the death and collapse of riparian forests in the West and is estimated to take about 50 years. Some streams are in late Phase II, while others are in early Phase III. Upper Black Canyon in the Gila National Forest, New Mexico, is a mid-Phase III. The stream, in the Aldo Leopold Wilderness Area, was once the habitat of the endangered Gila trout. The banks are laid back to predominantly cobbles, the fines having been washed away (Fig. 6), and the stream is entrenched. The stream has become so degraded that it is only marginally suitable for any type of trout.

There are no stands of young cottonwoods represented along the 11 km of Upper Black Canyon that I hiked. There are a few scattered trees (approx. 10-15 years old) but a few scattered trees do not make a forest. Remnant skeletons of mature cottonwood communities are evident along parts of the trail (Fig. 7). There is an occasional line of remaining cottonwoods with an understory of conifers (Fig. 7). However, most cottonwoods are dead and down and the few remaining alive are frequently girdled by beavers (Fig. 8). The gnawed rings are usually 10-15-cm deep and the beavers have begun consuming junipers (*Juniperus* spp.). Willows have been extirpated along Upper Black Canyon except a few decadent hourglass-shaped individuals on the deeded property just outside the wilderness area.

Collapse of decadent quaking aspen communities in phase III may be of shorter duration than 50 years. There is evidence along streams in Idaho that a number of them once supporting willow-aspen mix now only support willows. Once aspens disappear they may or may not pioneer rapidly into the floodplain even with grazing management.

FIGURE 7. Same stream and location as Figure 6. Stream entrenched and banks heavily eroded. The cottonwood forest has begun to collapse. To the back left is a small grove of cottonwoods mixed with conifers. Photograph by R. D. Ohmart, May 1992.

The above is exemplified on a small unnamed stream on the San Felipe Allotment (BLM) near Challis, Idaho, where a 1.5-ha cattle exclosure was constructed about 1988. The only remaining aspens or evidence thereof along this stream are in the exclosure (Fig. 9). The contrast between the grass and sedge-stabilized banks in the exclosure (Fig. 10) is striking against the raw, eroding outside banks. Vegetation in the elk exclosure did not differ from that within the cattle exclosure. Elk pellet groups were inside the cattle exclosure and light utilization of willows was evident but there were no raw or trampled streambanks.

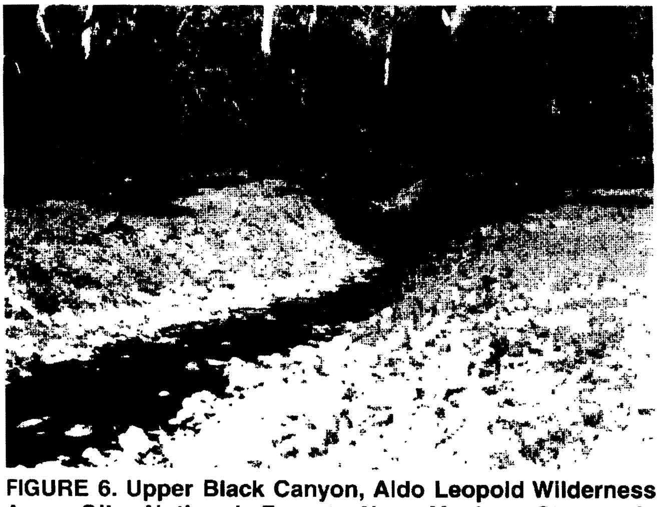

FIGURE 6. Upper Black Canyon, Aldo Leopold Wilderness Area, Gila National Forest, New Mexico. Stream is entrenched at least 1 m, and heavy cobble now forms most banks and the stream channel. Conifers mixed with scattered cottonwoods along the primary floodplain. Photograph by R. D. Ohmart, June 1992.

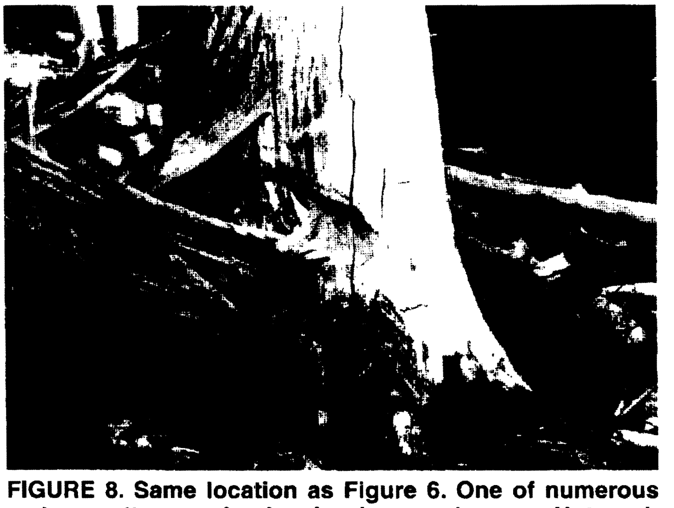

FIGURE 8. Same location as Figure 6. One of numerous mature cottonwoods showing beaver damage. Note collapsed cottonwood forest around the general area. Photograph by R. D. Ohmart, June 1992.

FIGURE 9. Cattle exclosure on small, unnamed stream on the San Felipe Allotment near Challis, Idaho. The only mature aspen on the stream are within the exclosure as are the only young trees (two in background and black stems in the photograph). Photograph by R. D. Ohmart, May 1994.

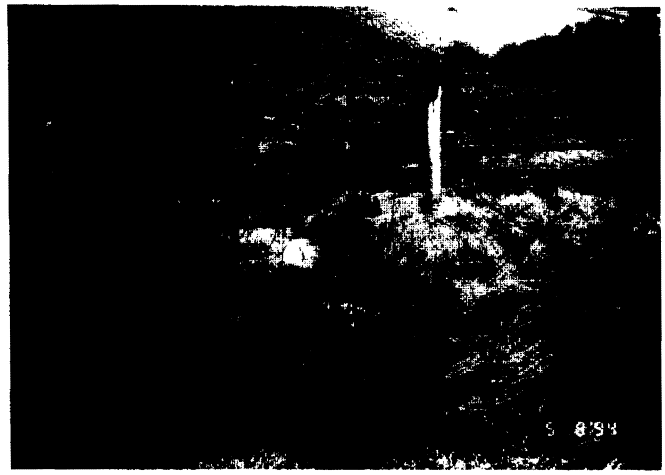

FIGURE 10. Fence line contrast with grass and sedge-covered banks within the exclosure contrasted with raw trampled banks outside. Aspen sapling in background of Figure 9 is that on left side. Photograph by R. D. Ohmart, May 1994.

Consequences to Fish and Wildlife

Fish. Most species are sensitive to changes in channel morphology and water quality and quantity, so the immediate physical and biological degradation of streambanks and channels affected this group early in Phase I and continues to do so in most streams today. The above changes in the stream are detrimental to trout populations (Armour 1977, Behnke and Raleigh 1978, Meehan and Platts 1978, Platts 1979). Armour (1979:39) stated ". . . we are concerned about overgrazing adversely impacting thousands of miles of streams associated with federally administered rangeland in the West." Bakke (1977) reported that loss of trout and salmon habitat from overgrazing has been a frustrating problem in Oregon. Behnke and Zarn (1976) identified livestock grazing as the greatest threat to the integrity of trout stream habitat in the West. The physical and biological degradation by domestic livestock grazing of most western streams has prompted fisheries biologists to advocate the abolition of livestock grazing for full stream recovery (Behnke 1979, Dahlem 1979).

Storch (1979:56), working in eastern Oregon, summarizes the problem,

> Uncontrolled livestock grazing has seriously affected the water quality of streams throughout the country. Indiscriminate use of streams by livestock results in breaking down the streambanks, eating and trampling shrubs that shade the streams and/or provide habitat for wildlife, and disturbing stream bottoms. The effects of such use has been erosion of stream banks, higher water temperatures, increased sedimentation, soil compaction, and reduction of the quantity and quality of forage.

The continued deterioration of fisheries habitats on western public rangeland from uncontrolled domestic livestock grazing has prompted the American Fisheries Society to publish a position statement (Armour et al. 1994). The paper has been in preparation a number of years and states, "Overgrazing of riparian and stream ecosystems by domestic livestock has damaged thousands of linear miles in the ecosystems" (Armour et al. 1994:9). Previous position statements and this one point out ". . . overgrazing by domestic livestock was one of the principal factors contributing the damage and loss of riparian and stream ecosystems in the West" (Armour et al. 1994:10).

Hansen (1993:334) observed riparian habitat degradation and stated, "It only takes a few weeks of unauthorized use or overgrazing to set back years of progress in improvements of riparian-wetland systems." Duff (1979) witnessed an area rested for 4 years degrade rapidly after the reintroduction of cattle; overhanging banks were quickly eliminated and after 6 weeks of midsummer grazing the banks fractured and eroded into the stream. Kauffman et al. (1983:683) examined the erosion component in northeastern Oregon and stated, ". . . erosion related to livestock grazing was enough to create significantly greater annual streambank losses when compared to an ungrazed area." Degradation time is rapid when compared to the slowness of the reversal process of 50 years if Wickiup Creek in Oregon is a general indicator of healing time (Clifton 1989). Gregory and Ashkenas (1990), working in Oregon, estimate that with proper management recovery of fish habitat, riparian areas, and water quality may require 25-200 years depending on existing conditions, stream type, and availability of fine sediment for bank rebuilding.

Clarkson and Wilson (1991) examined differences between unmanaged grazing, light, and no grazing during a 4-year study from 243 sampling stations among 75 reaches of 21 high-elevation trout streams in east-central Arizona. The focus of this study was the federally endangered Apache trout. In the data analysis, the amount of ungulate damage to streambanks consistently explained the greatest amount of variation in standing crop of fishes. Clarkson and Wilson (1991) concluded that better livestock management is necessary if the fishery potential of these streams is to be realized.

Banks along some streams may not recover in a lifetime once degraded. In general, these are small-order or headwater streams that carry little, if any sediment load. North Fork Cottonwood Creek may be an example in that there has been no significant change in channel width since livestock exclusion for 24 years (Kondolf 1993). Sediment load is reported to be low, but continual trespass by domestic livestock and exclosure size (0.5 ha or 135 × 35 m) confounds understanding channel response since exclusion (Kondolf 1993).

Numerous studies have examined bank and channel healing after livestock were excluded. Portions of Big Creek in Utah were excluded for 4 years and bank widths of the season-long (May-Oct) grazed area were 173% or almost twice as wide as the rested area (Duff 1979). Sedges and grasses responded rapidly after exclusion, increasing 63% (Duff 1983). Streambanks were initially bare or sparsely covered and within 4 years were described as luxuriant, grassy, and overhanging.

As the protective herbaceous groundcover over the floodplain is heavily grazed and weakened, the inevitable degradation process described earlier begins. The once relatively stable sinuous stream begins to straighten as it erodes its banks. Once incised to a stable point the lowered stream must widen the incised channel to a point that a new a new floodplain can be formed inside the old one. The straightened stream will then begin to reestablish its meander pattern.

Initial vegetation removal generally begins by livestock consuming grasses, sedges, and rushes along the stream and over the floodplain. As this forage resource is depleted, livestock begin browsing young trees and shrubs. If flood events are such that new tree or shrub seedlings germinate they are quickly consumed (Davis 1977), marking the end of tree and shrub recruitment to the riparian community. As stream width increases, large trees near the stream may also be undercut and fall.

Unmanaged grazing extirpates palatable native species and creates opportunities for the establishment and expansion of exotic species that may be undesirable and unpalatable. Cottam and Evans (1945) reported the presence of 10 native grass species in a canyon protected from grazing since the late 1800s (Red Butte), whereas these species were absent in a severely grazed canyon (Emigration) in Utah. Palatable grasses were 5 times greater in Red Butte than Emigration Canyon. Ruderals (unpalatable annuals and perennials, some being exotic such as cheat grass [*Bromus tectorum*]), were 7 times more abundant in Emigration Canyon. Young and Evans (1989) tie deteriorated range condition to the establishment and spread of exotic and noxious weeds in Nevada. Duff (1979) reported that in an exclosure on Big Creek in Utah, the more mesophyllic vegetation along the stream was moving outward from the stream as groundwater reserves increased, while in the grazed portion upland vegetation (i.e., sagebrush) continued invading the floodplain.

Dense shrubs along the stream (e.g., willows) provide shade for the stream, detritus for insect food, and stabilize banks. On Trout Creek in Montana, Marcuson (1977) reported shrub production to be 13 times greater in an ungrazed area as compared to a heavily grazed site. Prior to exclusion of livestock on Big Creek, in Utah, willows were so severely grazed that they were hedged back to basal stems. After exclusion of livestock willows responded slowly, but after 4 years they were 0.5 m tall, and in bend areas mean stem densities were $0.2/1.4\ m^2$ (Duff 1979). In northern Colorado seasonal grazing practices significantly altered shape, size, volume, and quantities of live and dead willow stems (Knopf and Cannon 1982). Martin (1979) listed livestock tree preference in Arizona as willow, velvet ash (*Fraxinus pennsylvanica* ssp. *velutina*), Arizona alder (*Alnus oblongifolia*), netleaf hackberry (*Celtis reticulata*), and Arizona sycamore, with even the least palatable young trees suffering damage in July. Storch (1979) reported on Camp Creek in eastern Oregon that shrub canopy was < 20% before exclusion of livestock, but 4 years after exclusion it was providing up to 75% shade to the stream. Livestock may remove ≥2 years of willow growth in a summer grazing period (Chaney et al. 1993).

Willows are an extremely important component of riparian areas and probably were one of the first woody elements to decline in the West. A historical literature review covering 1812-1880 reported extensive willow stands throughout western rangelands, but "... by the early 1900's, many of these stands were severely damaged or eliminated through cattle overuse" (Kovalchik and Elmore 1992:111). Though willows can withstand heavy browsing and not die, they cease seed production which alters their population dynamics and demography for many generations (Verkaar 1987). Kay and Chadde (1992) studied seed production in 3 willow species in Yellowstone National Park subjected to elk browsing, and willows in exclusions produced a range of 109,000-583,000 seeds/m^2. Browsed willows outside exclosures did not even produce catkins, much less seed. In southeastern Utah there were few, if any, bank or coyote willow (*Salix exigua*) in 5 heavily grazed canyons draining the east side of Cedar Mesa, while on the west side in Grand Gulch (cattle excluded for 20 years), bank willow shoots equaled or exceeded 30/m^2 (Figs. 11 and 12).

As shrubs are overgrazed year after year much of the dense shade component is eliminated. Combined with channel widening, water temperatures increase and oxygen tension levels decline. As willows disappear, the woody roots for stabilizing banks are reduced and may even be lost. Large shifts in water temperature affect fish populations and aquatic insects (Rhodes and Hubert 1991). Platts (1979:41) states: "Streamside vegetation protects streambanks by reducing erosive energy, by helping deposits build the streambanks, and by keeping the streambank from being damaged by ice, log debris or animal trampling." Streams in the Intermountain and Pacific Northwest are frequently icebound in winter. As the ice breaks up in spring it causes shifting dams, which forces the water over the floodplain. If riparian vegetation is not sufficient to protect the banks they can become heavily eroded (Platts 1991).

Comparative water temperatures inside and outside livestock exclosures demonstrate the value of riparian vegetation in depressing water temperatures. After 1 year of livestock exclusion, Van Velson (1979) reported water temperatures were reduced from 24C to 22C in Nebraska. Storch (1979) reported that on Camp Creek in eastern Oregon, mean daily water fluc-

FIGURE 11. Road Canyon in southeastern Utah that drains the east side of Cedar Mesa and has the same physical characteristics as Grand Gulch (Fig. 12). Virtually uncontrolled livestock grazing occurred until December 1993. The stream is entrenched with 2-4 m cut banks with cobble or bedrock bottoms typical along the drainage. Any surface stream flow undercuts banks to further eliminate the remaining riparian trees. No external disturbance occurred except for domestic livestock. Photo by J. Feller, March 1992.

FIGURE 12. Grand Gulch in southeastern Utah that drains the west side of Cedar Mesa and has the same physical characteristics as Road Canyon (Fig. 11). Livestock grazing has been eliminated for 20 years. The dominant woody vegetation along streamside is willow. Some cottonwoods are in the background. Photograph by J. Feller, June 1992.

tuations outside an exclosure were 27C compared to 13C inside the exclosure. Maximum temperatures outside and downstream from the exclosure averaged 11C higher than inside the exclosure. Mean daily water fluctuations were 15C outside the exclosure and 7C inside the exclosure.

Water quality is degraded by sedimentation. Behnke and Raleigh (1978) reported that overgrazing can cause accelerated sedimentation and silt degradation of spawning and insect production areas. Winegar (1977) working on Camp Creek reported sediment loads reduced by 48-79% as it flowed through a 5.6 km exclosure. Accelerated erosion (that caused by grazing) was examined under 3 different grazing levels in Utah to detect sediment transport levels (Croft et al. 1943). They intensively sampled 3 canyons in the Wasatch Mountains and ranked them as to grazing use: lightest (City Creek), moderate (Red Butte), and heavy (Emigration Canyon). Heaviest soil losses were where grazing was heaviest and highly reduced where grazing was lightest. They strongly suggest ". . . grazing management is as much a problem of soil management as of forage management" (Croft et al. 1943:16).

Phillips et al. (1975) reported fine sediments killing fish embryos. Platts (1979) reports that fine sediments cause embryos to receive less oxygen and allow toxic metabolic wastes to accumulate. These sediments also fill spaces in gravel beds, which reduces the protective cover and forces young fish to surface waters where they are more vulnerable to severe winter temperatures and predation. Platts (1978:42) reported that "... fish forced to remain in turbid waters may have trouble feeding, using oxygen, and reproducing."

Livestock grazing may also cause chemical and bacterial changes in a stream, but changes may not be manifested immediately. Johnson et al. (1978) did not find any chemical differences between an excluded reach and a grazed reach during the grazing season. However, following the grazing season a significant increase was noted in total dissolved solids, indicating livestock waste entering the stream, possibly from rain showers. In the stream reach where cattle were grazed, there was a significant increase of fecal coliform and fecal streptococci until about 9 days after grazing ceased. Numerous workers have attributed high fecal coli counts in streams to livestock grazing (Kunkle 1970, Darling and Coltharp 1973, Skinner et al. 1974).

Chemical and bacterial changes may, in concert, with physical changes negatively affect fish populations. This was the case in 2 springs in Pahranagat Valley in Nevada (Taylor et al. 1989). Ammonia and nitrate levels became so high that nitrifying bacteria consumed oxygen to levels that fish died. Bacterium populations of *Pseudomonas aeruginosa* and *Aeromonas hydrophila* also increased (Taylor et al. 1989). One fish, the White River springfish (*Crenichthys baileyi baileyi*), was federally endangered and livestock were removed allowing the fish population to recover. Livestock were not removed at Brownie Spring, which supports Pahranagat dace (*Rhinichythys osculus*), and that population has not recovered (Taylor et al. 1989).

Desert fishes were undoubtedly heavily impacted by overgrazing since 1880 (Hastings 1959, Miller 1961, Minckley 1973), but water management activities and introduction of exotic fishes have been more devastating and expedient in eliminating populations (Miller 1961, Minckley 1973). Many species were extirpated before the impacts of domestic livestock were known or fully understood. Hastings and Turner (1965:64, 65, 69, 74) show early photographs (circa 1890) of springs and streams that supported native fishes and these clearly show a highly degraded condition. Possibly many of these springs and small streams supporting native fishes were highly degraded earlier in that Cooke in 1846 described a stream on Agua Prieta Creek with the appearance of a stockyard (Bieber 1938).

Studies of fish populations from streams in PFC demonstrate superior fish habitat conditions. Deeper and narrower streams increase cover, movement areas for trout, and provide a combination of pool types (Raleigh 1982). Fisheries biologists report that lower stream width-depth ratios provide better fish habitat (Behnke and Zarn 1976, Platts 1981*a,b*). Differences in trout standing crop for ungrazed portions of Sheep Creek in Colorado were twice that in the grazed portion (Stuber 1985). In Montana, Gunderson (1968) reported a 30% increase in brown trout (*Salmo trutta*) in an ungrazed stream reach; and Marcuson (1977) reported a brown trout population 3.4 times greater than a grazed reach in Montana. On the Little Deschutes River in Oregon, Lorz (1974) reported trout populations 3.5 times greater in ungrazed versus grazed stream reaches. Similarly, in Rock Creek in Montana, Marcuson (1977) reported brown trout biomass 3.4 times higher in ungrazed stream reaches. Kimball and Savage (1977) reported a 4.25 increase after livestock exclusion for 4 years in Diamond Creek in Utah. Van Velson (1979) reported 88% of a fish population were rough fish while an area was grazed, and after 8 years rest only 1% of the population was rough fish. In Washington, significant reductions in biomass for coho salmon (*Oncorhynchus kisutch*), cutthroat trout, and other salmonids were reported in heavily grazed areas versus ungrazed areas (Chapman and Knudsen 1980).

The validity of some of the above fishery standing crop values have been questioned by Platts (1982). He questions sample size, statistical reliability, lack of controls, and other facets of some of the studies. Some terrestrial studies could be subjected to the same concerns. The inclusion or exclusion of those studies does not change the overall picture of uncontrolled livestock grazing on the degradation of western riparian habitats and their effects on native fishes.

Fifty years of livestock exclusion on Wickiup Creek in the Blue Mountains of central Oregon shows the reversal that occurred when a riparian system was relieved of grazing (Clifton 1989). A 1933 photograph prior to livestock exclusion shows the meadow barren of vegetation, exposed soils, channel banks devoid of vegetation, and banks about 1.3 m high (Clifton 1989). The channel was trapezoidal in shape with outsloping or widened banks. Ten years after exclosure the meadow showed vegetation, the channel had aggraded about 0.6 m, and the channel banks were vegetated. Fifty years after livestock exclusion, the channel had undergone a 94% reduction in cross section and was described as having ". . . thickly vegetated overhanging banks [that] obscure a narrow and deep channel" (Clifton 1989:128). Similar vegetation responses were reported for Sheep Creek on the Roosevelt Forest in north-central Colorado at 2,500 m (Schulz and Leininger 1990). They reported that after 30 years of cattle exclusion there was twice the litter in the protected site, and 4 times more bare ground in the grazed area. Willow canopy was 8.5 times greater in the protected site while Kentucky bluegrass (*Poa pratensis*) was 4 times greater in the grazed site. Fowl bluegrass (*P. palustris*) was 6 times greater in the protected site. Caged plots within the grazed area only produced a peak standing crop of 1,217 kg/ha, while 2,410 kg/ha were produced in the exclosure. I refer the reader to the 1939 repeat photograph of the stream (Schulz and Leininger 1990:296). Twenty years of livestock exclusion in Grand Gulch in southeastern Utah has transformed an entrenched, intermittent stream running on bedrock or heavy cobble (Blackburn 1993) to one aggraded with well-defined banks, there are indications that it may become perennial, and it now supports dense willow-cottonwood communities (Fig. 12).

Wildlife. Structural damage to streambanks along with their denudation and that of the floodplain in Phase I began impacting amphibians, some reptiles, and ground-nesting birds. Data presented by Szaro et al. (1985) from an exclosure in a high-elevation riparian community of alder and willow in New Mexico, demonstrates the importance of the floodplain understory for the wandering gartersnake (*Thamnophis elegans elegans*). In the exclosure (≤10 years protection) both vegetative groundcover and debris accumulated to a level to provide habitat for this snake. Gartersnake density was significantly higher in the ungrazed versus the grazed site (capture rate 5:1). Loss of the herbaceous groundcover and the understory in Phase I probably occurred shortly after bank and channel degradation, and has continued for so long that I suspect many populations of these species were locally extirpated, and if not, significantly reduced in density as was the wandering gartersnake.

The dramatic decline in the herbaceous groundcover and thinning of the understory in Phase I took its toll on all wildlife populations dependent on these layers. Moulton (1978) suggested that species richness in small mammals might increase with grazing because it would create new microhabitats with more diversity. This might be true in some localities, but Medin and Clary (1989) reported the reverse with higher small mammal species richness (11 species vs. 6) in Nevada on a site protected for 11 years compared to a grazed site. They reported a higher standing crop biomass (3.24), species richness (1.83), and species diversity (1.25) on the ungrazed site. Moulton (1978) also reported that grazing may have limited densities of the prairie vole (*Microtus ochrogaster*) which prefers dense groundcover, while improving habitat for mice in the genus *Peromyscus*. Schulz and Leininger (1991) reported trapping 28 small mammals in a grazed site and 41 in a site that had not been grazed for 30 years. The ubiquitous deer mouse (*Peromyscus maniculatus*) dominated the grazed site (15:1) and the western jumping mouse (*Zapus princeps*), preferring dense herbaceous groundcover, dominated the ungrazed site (22:1).

The avifauna inhabiting the understory should be dramatically affected if foliage volume is an important wildlife habitat component. Rucks (1978) stated that understory depletion displaced shrub-nesting species with more generalists that had no preference for nest placement. Taylor (1986) found a significant correlation between increased annual grazing frequency and decreases in bird abundance, shrub volume, and shrub height, as well as between bird abundance and shrub density and height. Numbers of species decreased as intensity of grazing increased and density values were 5-7 times higher on an area ungrazed since 1940 than on 2 areas grazed annually until 1980. His examination of 1930 photos, ". . . showed a tall deciduous upper canopy along the river . . ." and that ". . . cattle grazing

FIGURE 13. San Pedro River, Arizona, approx. 1985. The area was in private ownership with unmanaged livestock grazing. Raw and eroding banks on right, open and wide channel, and absence of herbaceous and understory vegetation are characteristic symptoms of unmanaged grazing. Photo courtesy of BLM Safford District Office, Arizona.

can eliminate or reduce the upper canopy by preventing the establishment of saplings . . ." (Taylor 1986:257).

A 64-km reach of the San Pedro River in southeastern Arizona provides unique insight as to what most perennial desert streams resembled about 1875-1885 (Figs. 13-15) as Phase I was completed. A rare glimpse of this river area before the livestock boom of the 1880s was provided by a pioneer rancher named H. C. Bayless. In 1901, D. A. Griffiths, Chief Botanist over Grass and Forage Plant Investigations for the Arizona Experiment Station in Tucson, sent a circular to a select group of pioneer ranchers in an effort to better understand the role of livestock and the condition of the range prior to and after the 1891-1893 drought (originals not seen *in* Bahre 1991, but also see Hendrickson and Minckley 1984). To a question on entrenchment of the river, Bayless wrote,

> Above twelve years ago the San Pedro Valley consisted of a narrow strip of subirrigated and very fertile lands. Beaver dams checked the flow of water and prevented the cutting of a channel. Trappers exterminated the beavers, and less grass on the hillsides permitted greater erosion, so that within four or five years a channel varying in depth from 3 to 20 feet was cut almost the whole length of the river (Bahre 1991:111).

Bayless' response to the question of whether the current situation was caused by overstocking, drought or both was:

> The present unproductive conditions are due entirely to overstocking. The laws of nature have not changed. Under similar conditions vegetation would flourish on our ranges today as it did fifteen years ago. We are still receiving our average amount of rainfall and sunshine necessary to plant growth. Droughts are not more frequent now than in the past, but mother earth has been stripped of all grass covering. The very roots have been trampled out by the hungry herds constantly wandering to and fro in search of enough food. The bare surface of the ground affords no resistance to the rain that falls upon it and the precious water rushes away in destructive volumes, bearing with it all the lighter and richer particles of the soil. That the sand and rocks left behind are able to support even the scantiest growth of plant life is a remarkable tribute to our marvelous climate. Vegetation does not thrive as it once did, not because of drought, but because the seed is gone, the roots are gone, and the soil is gone (Bahre 1991:112).

The once subirrigated farmland and marshy conditions disappeared on the river as it entrenched and water tables dropped. Somewhere about the turn of the century, cottonwood and willows became established, possibly when livestock numbers were extremely low after the drought and before numbers were reestablished once the range improved. A portion of the San Pedro River described by Bayless came under BLM control as a Riparian National Conservation Area and domestic livestock grazing was eliminated in January 1987. Streambanks, channel, and understory conditions at the time grazing ceased were

FIGURE 14. (Left) San Pedro River, June 1987. The area is now BLM National Riparian Conservation Area (NRCA) and livestock were removed 1 January 1987. This would be typical of southwestern streams about 1885 with a highly modified channel, trampled banks, no herbaceous groundcover, and understory depauperate. Photograph courtesy of BLM Safford District Office, Arizona. (Right) Repeat of (Left) 4 years (June 1991) after livestock were removed. Photopoint moved to left because developed understory disallowed the photo. Note bank vegetation beginning to narrow and deepen the channel. Photograph courtesy of BLM Safford District Office, Arizona.

FIGURE 15. (Left) San Pedro River, June 1987. Different location in the National Riparian Conservation Area but stream conditions are typical of unmanaged grazing along stream courses. Photograph courtesy of BLM Safford District Office, Arizona. (Right) Repeat of (Left), June 1991. Stream being narrowed and deepened with encroachment and sediment trapping of the vegetation. Photograph courtesy of BLM Safford District Office, Arizona.

essentially as I described at the end of Phase I. Within 4 years after livestock exclusion, the understory and bank vegetation had increased significantly (Krueper 1993).

Response of neotropical birds on the San Pedro River during the 4 years after exclusion ranged from virtually unchanged for those species foraging on volant insects to moderate increases of 2-6-fold for those gleaning foliage insects (Krueper 1993). Highly significant density increases were observed in foliage gleaning and understory thicket specialists such as the common yellowthroat (*Geothlypis trichas*) and song sparrow (*Melospiza melodia*) that showed a 25-fold and 61-fold increase, respectively (Table 1). In portions of Sheep Creek in northern Colorado that have been excluded from grazing for 30 years (Schulz and Leininger 1991) showed Wilson's warbler (*Wilsonia pusilla*) and Lincoln's sparrow (*Melospiza lincolnii*), thicket specialists, more common in the restored site. Finch (1986) reported these 2 species dominating healthy subalpine willow communities in southeastern Wyoming.

The ecological contributions that birds make to forest communities are poorly understood but studies over the last decade have focused more attention to the contributions that this group makes to forested ecosystems. Prior to these studies the lay public was highly emotional toward this group (Carson 1962) and birds were perceived more as jeweled forest songsters. Frugivorous forms have been documented as important dispersers of seeds away from the parent tree (Howe and Vande Kerckhove 1979, 1981; Pratt and Styles 1983, Masaki et al. 1994). Insectivorous birds in forests have often been assumed to simply be a small additive factor of mortality to phytophagous or plant-eating insects and Crawford and Jennings (1989) reported a great reduction in densities of spruce budworm by bird predation. The most impressive demonstration of phytophagous insect control

TABLE 1. Increase in bird numbers after removal of cattle from grazing for 5 years. Table adapted from Krueper (1993). NA = data not available.

	Years					
Species (densities are birds/40 ha)	1986	1987	1988	1989	1990	1991
Yellow-billed cuckoo (*Coccyzus americanus*)	6	10	8	6	13	NA
Western wood-pewee (*Contopus sordidulus*)	8	16	22	38	28	29
Brown-crested flycatcher (*Myiarchus tyrannulus*)	21	33	27	36	26	26
Bell's vireo (*Vireo bellii*)	7	11	7	12	15	16
Yellow warbler (*Dendroica petechia*)	29	84	99	227	131	176
Common yellowthroat (*Geothlypis trichas*)	7	24	39	115	110	149
Yellow-breasted chat (*Icteris virens*)	26	44	47	95	100	110
Summer tanager (*Piranga rubra*)	44	84	73	167	94	108
Song sparrow (*Melospiza melodia*)	0	11	14	38	36	61
Northern oriole (*Icterus galbula*)	28	35	28	34	21	32

by birds examined white oak (*Quercus alba*) growth over a 2-year period in a Missouri deciduous forest (Marquis and Whelan 1994). Study trees were compared in a natural setting (controls), caged that allowed insect passage and excluded birds, and trees sprayed with a pesticide. Controls lost 13% leaf area, sprayed trees 6%, and caged plants 25% at the end of the first season (24, 9, and 34%, respectively, the second season). Differences in above ground biomass production (growth) were reduced by one-third in caged trees from sprayed trees with controls having intermediate values. Bird populations significantly controlled insect populations in these studies.

The importance of riparian habitats as nesting and refueling sites for migrating wildlife is frequently mentioned in the literature, but few studies have examined this subject in any depth. Stevens et al. (1977) summarized the literature and from their own data reported that riparian plots contained up to 10.6 times as many migrants per hectare as paired upland sites. More recent and refined studies, many by biologists studying the plight of neotropical migrants, are beginning to provide enough information to indicate the importance of these habitats to migratory wildlife.

Livestock grazing is not the central issue, but, combined with water management, it has contributed heavily to the decline in quality stopover and wintering habitat. Southwestern riparian habitats are an important stopover and wintering area since they are surrounded by arid uplands. As they are degraded and reduced in size their availability and suitability for migrants becomes more limited. Laymon (1984) suspects that riparian forest fragmentation and tiny forest size may now be limiting avian densities nesting to the north. Stanley et al. (1991) contends that they are extremely important areas for migrating birds since they remain green and productive during late summer post-breeding dispersal and in fall migration when there is little upland productivity. The extensive and multiple kilometers of riparian forest along the Sacramento River in California are now only a few trees wide and highly fragmented into patches (Tompson 1980). The same holds true on the lower Colorado River (Ohmart et al. 1988).

Evidence suggests that passage migrants select stopover sites and length of stay based on the intrinsic suitability of the habitat (Moore and Simons 1992). Therefore as riparian habitats continue to be destroyed, fragmented, and degraded in foliage volume and insect productivity, migrant passage or survival in passage could be highly limited as riparian forest size and productivity decline.

A few studies are beginning to indicate the importance of these riparian sites as refueling areas for passage migrants. For example, in the white-crowned sparrow (*Zonotrichia leucophrys*; Cherry 1982) and in wood warblers (Moore and Kerlinger 1987), leaner birds stayed longer and stored larger amounts of fat than those birds with good fat stores. Without quality habitats en route many of the birds in poor fat condition might not finish the migration without rebuilding sufficient fat reserves (Winker et al. 1992).

An interesting data set comes from a 2-ha remnant riparian area in California surrounded by urban and agricultural development along Coyote Creek upstream from where the creek enters the San Francisco Bay. This area was mist netted from 1987-1991 to examine migrant use and body mass changes. Of the 4 species examined in spring and fall migration few (6-18% depending on season and species) stayed on the site >1 day. Most (52-79%) gained or maintained body mass, suggesting the stopover was for refueling. Otahal (ms) calculated flight distance from stored fat for the most extreme specimen, a 10 g yellow warbler (*Dendroica petechia*) which gained 5 g of fat, could then potentially fly 2,848 km on the added fat stores. The continuing loss and degradation of riparian forests may have extensive effects on migratory and overwintering wildlife.

Many western riparian habitats are beginning to approach the threshold where cover and/or foliage availability for insect production for habitat specialists is barely sufficient to sustain populations. Not only vegetation density and distribution, but forest island size may also be a determining factor for some species. Almost all of the most important terrestrial wildlife habitat elements described earlier are essentially gone or highly degraded. The cottonwood-willow community and their tree densities at low and moderate elevations are rapidly disappearing (The Arizona Nature Conservancy 1987). Extensive stands of dense willow have been fragmented and, in some instances, eliminated (Elmore and Kauffman 1994). The foliage profile is now a skeleton of what it was 50-100 years ago and foliage volumes are sparse at all layers. Also, through time the intracommunity plant patchiness element has slowly disappeared. Knopf and Cannon (1982) suggest that in northern Colorado horizontal and vertical structure of the shrub willow community has been eliminated for birds by seasonal grazing over the past 75-100 years. The shrub component may persist along some second terraces, but reduced densities have left a sparse shrub element or it has been converted to agriculture.

Even many wildlife refuges have been subjected to intensive domestic livestock grazing. The 73,200-ha Malheur National Wildlife Refuge in southeastern Oregon was grazed by 40,000 AUMs in the 1930s, by the 1950s this had been increased to over 100,000 AUMs, and in the late 1960s the mean for 9 years was 118,000 AUMs (Taylor 1986). Refuge personnel also used herbicides and grubbing to remove willows to increase livestock forage. Predictably, willow flycatcher and yellow warbler numbers plummeted, but as cattle numbers were reduced in the 1970s the willow element began to recover. Breeding bird surveys showed 7 yellow warblers in 1972 and no willow flycatchers. By 1982 yellow warblers had increased to 56 and willow flycatchers numbered 30. Bird data from transects on the refuge showed similar trends and vegetation data from these transects showed a negative correlation between shrub volume and frequency of cattle use on an annual basis.

Unless grazing management changes are made soon it is predictable that many more species, especially neotropical birds, will be placed on the endangered species list. Horning (1994) reported that of the 76 federally listed plant and animal species on BLM lands where livestock grazing was a significant factor in their decline, 61 species were riparian dependent or associated with riparian habitats.

BEAVERS AND CATTLE

In the western United States it was probably best for the extended longevity of riparian systems that beavers were virtually eliminated prior to the introduction of extensive numbers of domestic livestock to western rangelands. Beavers and unmanaged livestock grazing in stream systems are extremely damaging and together expedite the collapse of riparian forests. Both can be in direct competition for food (i.e., woody and herbaceous) depending on the condition of the riparian habitat (USDA 1992). Livestock crush dams in their efforts to consume the lush forage of sedges, rushes, and willows. They also consume suckering new growth of young trees cut by beavers for dam repair and food, and imbalance the beaver-stream equilibrium that has evolved over the years. As the young trees cut by beavers attempt to put up new shoots these are consumed until the energy reserves are depleted and the tree dies. Repeated growing-season grazing weakens the woody and fibrous-rooted species until they are either washed out in large storm events or die. In the absence of woody and fibrous roots the alluvial soils are then vulnerable to further erosion in each storm event. As the stream widens and downcuts, the water table is lowered, leaving wetland species not eliminated by grazing with soil moisture levels too low to survive. In the damaged system, trampling destroys new dam efforts and with time the elimination of young trees for dams and food begins stressing the beaver population. Beavers are then forced to consume the cambium of the larger deciduous trees expediting the collapse of the mature riparian forest (Figs. 16 and 17). Apparently, once a stream degrades to this level beavers (and livestock) must be removed to expedite the recovery of the stream (L. Meyers, USFS, pers. commun.).

Livestock were not involved in the below example, but it shows how important beaver can be in modifying hydrological and floristic processes. Red Butte Creek in Utah had beavers trapped from the canyon, but they were reintroduced in 1928 (Bates 1963). The Army, then in possession of the canyon, had the beaver removed in 1982, fearing water contamination at Fort Douglas.

Where beavers were active along the stream, the vegetation cover was affected approximately 91 m on either side (Bates 1963). Sediment deposition behind the dams ranged from 0.6-2.4 m deep. Earlier, Scheffer (1938) had reported that 2 dams in the canyon had trapped 4,468 m^3 of silt. In 1983, a year after the beavers were removed, a large storm eroded huge quantities of sediment and incised the stream again, creating a large delta in the reservoir at the mouth of the canyon. In the absence of beavers, 55 plant species have either been extirpated or are highly restricted in their distribution (Ehleringer et al. 1992). Personnel from the USFS claim that flood damage to the canyon would not have been as severe or prolonged had the beaver been active during the floods (Ehleringer et al. 1992).

Beneficial Effects of Livestock Grazing on Riparian Habitat

There is no advantage or benefit to riparian habitats in PFC to be grazed by any large ungulate, be it livestock or elk (Houston 1982, Chadde 1989, Arizona Game and Fish Department 1993). Several years ago when public hearings were held on the transfer of the San Pedro River, Arizona, from private holdings to the BLM, I testified that cattle could be used to economically reduce the fuel load of tall sacaton grasses (*Sporobolus* spp.) growing adjacent to and within cottonwood-willow habitats. Removal of this material would prevent fires that are highly detrimental to these forests. Krueper (1993:323), working on the San Pedro River, stated "Grazing within the riparian zone may be used to reduce dense annual growth . . ." to prevent fires. In this very limited situation, cattle may be useful to help reduce fuel loads and prevent wildfires that are especially detrimental to cottonwood trees.

FIGURE 16. Main Diamond Creek, Gila National Forest, New Mexico. Beaver lodge with sticks of dam visible to back left. The animals are living in the water table and consuming mature cottonwoods around them because there are no young trees for food. Photograph by R. D. Ohmart, May 1994.

FIGURE 17. Same location as Figure 16. Note extent of tree felling and eventual demise of the cottonwoods. Foreground shows cow pies, tracking, and absence of herbaceous groundcover from unmanaged grazing. Photograph by R. D. Ohmart, May 1994.

It has been suggested that cattle might serve to open dense willow thickets and help wildlife in high altitude riparian habitats in southeastern Wyoming (Krueger and Anderson 1985). Willow densities in the study streams were 2,007/ha and 897/ha, the latter stream having a record of overgrazing. Some increased bird densities were recorded in the tunneled willows, but attracted species were habitat generalists and not the specialists that prefer dense willow thickets. Small mammal habitats might be improved by the opening of willows and creating more grass and sedge areas (Krueger and Anderson 1985). Grazing could also be used to create low density willow habitat, but the authors state, "Enough riparian habitat has been overgrazed to create plenty of low density shrub-willow habitat" (Krueger and Anderson 1985:303).

Unquestionably, grazing can be used to enhance habitat for some avian species (Burgess et al. 1965, Kirch and Higgins 1976, Ryder 1980, Crouch 1982, Schulz and Leininger 1991, Clary and Medin 1992). Most of the species added are habitat generalists whose numbers are common in the uplands. Those species thought to be declining and possibly being eliminated are foliage volume or thicket specialists. For example, on a grazed versus ungrazed stream in Colorado, American robin (*Turdus migratorius*) numbers were 30 and 15, Lincoln's sparrows 4 and 13, mountain chickadees (*Parus gambeli*) 8 and 17, and Wilson's warblers 0 and 9, respectively. The ubiquitous American robin increased in heavily grazed riparian habitats, while those species requiring dense habitats to either forage, nest, or for cover declined (Schulz and Leininger 1991). Bock et al. (1993) in reviewing the literature reported that of 43 neotropical migrants, 8 responded favorably to grazing, 17 were negatively affected, and 18 were unresponsive or showed mixed responses. There is no problem with managed grazing or using grazing as a tool to increase biodiversity, but examples of managed grazing are so few and unmanaged grazing so common in riparian habitats that biodiversity is rapidly being lost (Horning 1994).

It's also been advocated that cattle can serve as a tool to modify floodplain terraces to improve groundcover (A. Savory, Holistic Resources Management, pers. commun.). I can only visualize this being true on rivers where the natural flooding process has been stopped by dams. Reduced instream flows below dams with a concomitant decline in the water table converts the higher second terraces to upland habitats. Cattle might be useful in converting decadent shrub communities into grass communities. Otherwise, I fully agree with Chaney et al. (1993:14):

> Some people tout that livestock trampling as a 'tool' to lay back steep or undercut streambanks. The channel of a stream low in sediment could take decades to recover from being 'laid back.'

STATUS OF RIPARIAN AREAS ON PUBLIC LANDS

An early status report on BLM lands (Trout Unlimited 1979) reported that 77% of 30,577 km of streams were in unsatisfactory condition. A 1990 status report on riparian habitats from the USFS in the western national forests estimated that 93,339 km of riparian areas within grazing allotments did not meet and were not moving toward meeting forest plan objectives (GAO 1992). The BLM reported that on 0.5 million ha of riparian-wetland and 78,856 km of riparian streams in 10 BLM state offices, only 7% of the riparian areas were meeting objectives, 8% were not meeting them, and riparian response was unknown in 85% of the areas. The 85% unknown concerns me in that if it is not known what condition they are in then the probability is good that they are not functioning properly. As an example, a BLM report in 1989 by the Gunnison Basin Resource Area Office in Colorado stated,

> . . . that 60 to 100 percent of the riparian areas were being overgrazed. Overgrazing damaged the riparian areas to the extent that forage production was below normal; plant species composition was undesirable; stream channels and stream banks were unstable, causing erosion; soils were compacted, reducing water infiltration; vegetation cover was reduced, resulting in excessive silt from heavy runoffs; groundwater reservoirs were not able to recharge and out of bank heavy runoffs were not slowed down and dispersed (Office of Inspector General 1990).

The most recent (USDI 1994) data estimates that for BLM-managed lands, of the approximately 400,000 ha of riparian habitats 20% are nonfunctioning and 46% are functioning at risk, which means they are threatened by domestic livestock grazing. About 34% are in proper functioning condition. Of the riparian habitats on USFS lands, 63% are meeting objectives while 27% are not. These data very closely agree with data on the Uncompahgre-Gunnison National Forest in Colorado where of 5,885 km of perennial streams, 65% are meeting objectives and 35% are not (R. L. Storch, Forest Supervisor, USFS, pers. commun.). My observations are that of most of the forests in Southwest Region 3 (i.e., Arizona and New Mexico), the number of streams in PFC would be more the opposite. Further, no one in the Regional or Forest Supervisor Offices (Region 3) was aware how the National numbers were obtained, or if they ever were for Region 3. Apparently, Region 3 was not included in the 1994 data set. For example, when the GAO (1988) did its survey they looked indepth at 5 locations and reported that on the Tonto National Forest (Region 3) that 80-90% of the riparian areas were in unsatisfactory condition.

Along important streams for fish, wildlife, or scenic values, which serve as riparian pastures, one would think that the agencies and permittees would have showcased managed-grazing examples. The East Fork of the Gila River, Gila Wilderness Area, Gila National Forest, New Mexico, once a cold-water fishery, is now a warmwater fishery with eroding and caving banks, high sediment loads, and virtually no woody vegetation along the stream. Though a riparian pasture, it is questionable when it will receive better management. The Comb Wash Canyons, Comb Wash Allotment, San Juan Resource Area, BLM in southeastern Utah are entrenched to bedrock or large cobble, support very little riparian vegetation, and provide some of the most scenic riparian habitats in the West. These riparian pastures have only recently received management protection (Rampton 1993).

POSITIVE RESULTS FROM COOPERATING PERMITTEES

Some ranchers have altered their riparian grazing approaches (voluntarily, persuaded, and/or through endangered species restrictions) that has resulted in improved riparian habitats. This has provided increased forage production from these habitats (GAO 1988). The GAO (1988) interviewed many of these permittees and reported savings in reduced feed costs, availability of permanent water supplies where streams had been intermittent, better utilization of upland forage by livestock where they previously had not grazed, and generally better livestock health and calving rates.

A few ranchers have taken the initiative to improve riparian habitats voluntarily. Date Creek in Arizona is a case in point (Fig. 18). After 24 years of dormant-season-only grazing this small stream looks (in 1991) totally incongruous compared to most streams in Arizona. New banks have been formed from trapped sediment and are now matted with grasses, sedges, and rushes. A young and healthy age mix of willows and cottonwoods dominated the floodplain, and in many stream reaches the luxuriant vegetative growth has to be separated to find the 15-20-cm wide banks that encase the 30-40-cm deep riverlet. I visited the stream in July 1991 after heavy March storms had created highly erosive floods throughout Arizona. I expected to see extensive scouring and the possible loss of the 1.6 km-long managed area, but this had not occurred. Figure 18 shows the contrast where there is no management versus the fence line where there is only dormant season use. The obvious difference in standing crop in the fore and background of the photo clearly shows the advantage of increased forage production for both wildlife and livestock. The managed area supports several species of songbirds, nesting zone-tailed hawks (*Buteo albonotatus*), javelina (*Tayassu tajacu*), and mule deer, while only a few songbirds were observed in the few mature trees upstream.

FIGURE 18. Date Creek near Wickenburg, Arizona. A fence separates the unmanaged allotment from the managed downstream area. A major difference in herbaceous groundcover for soil protection and forage availability is obvious in the managed area. Photograph by R. D. Ohmart, June 1992.

Much (approx. 90%) of the floodplain or first terrace of Date Creek was lost in extremely heavy storm events in January and February 1993. A century of January rainfall weather records was broken in 1993, which indicates the magnitude of the event. It should also be kept in mind that the watershed and stream have been degraded for over 100 years and possibly the degradation might have been negligible had they been in good ecological health. The positive side is that management improvements that had accrued over the past 24 years protected the integrity of much of the riparian community. A line of 15-24-year-old willows and cottonwoods had developed along the outer edge of the first terrace and these trees withstood (Fig. 19) and dissipated the erosive force of the flood, keeping most of the scouring in and along the first terrace. In a few places where

FIGURE 19. Date Creek, Wickenburg, Arizona, livestock managed area with trees lining the primary floodplain. Grasses, sedges, and rushes reestablishing streambanks and beginning to stabilize the sandy soils after the 1993 flood event. Photograph by R. D. Ohmart, June 1994.

FIGURE 20. Date Creek managed area showing rapid invasion of knot grass (*Paspalum distichum*) and cattail to begin soil stability, sediment trapping, and stream containment. Photograph by R. D. Ohmart, June 1994.

the tree line did not exist, heavy erosion cut new channels, damaging the second terrace. Though the flood removed much of the first terrace, small clumps of sedges (*Carex* spp., *Cypreus* spp.), rushes (*Juncus* spp.), and cattails (*Typha* spp.) persisted and these are rapidly pioneering into bare areas (Fig. 20).

The allotment upstream that has year-round grazing and no management was devastated by the flood. Channel changes were rampant, mature trees were uprooted near the stream and the second terrace of velvet mesquite (*Prosopis velutina*) (prior to degradation and entrenchment this area was flooded in higher than normal flood events) was heavily eroded (Fig. 21). I repeated the photopoint taken in 1992 (Fig. 18) at the water gap between the 2 properties and though much of the primary floodplain is gone from the managed area, the young trees remain in the background.

I know of a few permittees in Arizona that have, on their own, made financial and personal sacrifices to improve riparian habitats on their deeded lands and leased or public lands. I hope there are more in Arizona and throughout the West who are quietly going about their work and doing similarly. These permittees are to be commended and supported for they will be a standard to those who have not made this commitment. As public concern and litigation rapidly increase over the degraded condition of western riparian habitats, the fate of the livestock industry on public lands may rest on these dedicated permittees. They show how natural resources protection and domestic livestock are compatible with proper grazing management. It can be done but it will take more tax dollars and sacrifices on the permittees' part. But, as riparian systems begin functioning properly, the annual harvestable forage for both wildlife and livestock from these flood-irrigated pasturelands will far surpass a decade of forage production from a degraded system.

GRAZING SYSTEMS AND MANAGEMENT APPROACHES

The best way to manage riparian habitats is not to graze them. If they are to be grazed, the manager must learn how to use the forage resource while maintaining stream PFC. This will be a management challenge, because livestock concentrate in and are highly attracted to riparian areas, these habitats are usually scattered throughout the allotment, and each riparian system has its own set of vulnerable biotic and abiotic components (Elmore and Kauffman 1994). Kinch (1989) reports that the management of domestic livestock grazing in riparian-wetland areas is one of the most difficult and complex issues facing western rangeland management.

Numerous grazing approaches or systems have been developed over the years in an attempt to help deteriorated rangelands and increase forage production. Few, if any of these approaches, consider the condition or grazing impacts on riparian communities (Platts 1981*a*, 1989). Even the most recent treatise on classifying, inventorying, and monitoring rangeland (National Research Council [NRC] 1994) only devotes about 5 sentences to riparian habitats. Present approaches concentrate on forage removal in the uplands, and by the time that grazing level has been achieved most riparian habitats have been heavily overgrazed. For example, Krueger and Bonham (1986) report that cattle are so attracted to riparian areas in the summer that 90-95% of the adjacent uplands receive little or no use. Meyers (1989) examined 34 grazing systems in Montana and 25 (74%) showed no improvement in riparian areas over 10-20 years, while most showed improvement on the watershed. Clary and Webster (1989) in discussions with managers and after reviewing the literature reported there is not a single grazing management approach that has produced consistent improve-

FIGURE 21. Date Creek, before heavy entrenchment this stream once flooded the velvet mesquites on the second terrace to the left of the photograph above the cut bank. Note the absence of rock which would have prevented severe entrenchment. Riparian trees are gone and the sparse understory is dominated by the unpalatable seep-willow (*Baccharis salicifolia*). Photograph by R. D. Ohmart, June 1994.

FIGURE 22. Repeat photograph of Figure 18 after an unusually heavy flood event in January-February 1993. The large 4-trunked tree (visible in 1992) in the center is now covered by the small tree to right of the big tree. The line of trees on either side of the primary floodplain kept the erosive waters contained and dispersed erosive energies. Photograph by R. D. Ohmart, June 1994.

ment of degraded riparian-wetland areas over western rangelands. Elmore and Kauffman (1994) support this conclusion.

The 2 grazing approaches most detrimental to riparian habitats are total growing-season grazing and year-long grazing (Elmore and Kauffman 1994). Cattle are so attracted to riparian habitats that any grazing approach that extends throughout the growing season will insure some cattle in riparian habitats the entire time, unless herded or excluded by fencing. Similar cattle behavior has been displayed in New Mexico (Goodman et al. 1989) and the western Dakotas (Severson and Boldt 1978).

Numerous ecological approaches have been developed for grazing riparian habitats or restoring them (Skovlin 1984, Kinch 1989, Kauffman et al. 1993, Elmore and Kauffman 1994). Elmore (1992) discusses a number of grazing systems and their shortcomings relative to riparian protection and recovery. Selecting a grazing approach in riparian habitats is difficult in that a multitude of variables are involved (Chaney et al. 1993, Elmore and Kauffman 1994). Many workers (Platts 1981*b*, Kinch 1989, Clary and Webster 1989) have examined riparian grazing approaches and no single method has been successful for improving degraded riparian areas. Elmore and Kauffman (1994) summarize grazing approaches based on their experiences and I briefly present what they consider the poorest and the best approaches. Continuous or season-long cattle grazing is the poorest, creating the greatest amount of degradation to the physical and biotic components of the riparian area. This is equal to holding sheep or cattle in riparian areas. Equally poor is short-duration and high-intensity cattle grazing in riparian areas. Winter use with cattle or sheep receives a moderate rating relative to riparian degradation. The best and obvious approach is total closure or rest to riparian areas from all classes of livestock. Two approaches right below exclusion are rest rotation with seasonal preference with sheep, or corridor fencing with either sheep or cattle. Rating high, as well, is fencing the riparian area for prescribed use.

Total closure may be inviting to many but it also has numerous implications. The most obvious is the cessation of domestic livestock grazing on public lands, which I would not support. Secondly, fence all riparian areas, but present fencing on public land is considered more than enough (Jacobs 1991) and costs to fence riparian areas would be overwhelming (current charge $3,000-3,500/1.6 km of fence). Platts (1991) estimated that it would cost $90 million just to fence all of the 24,135 km of fishable streams on BLM lands. These and associated problems such as wildlife entanglement, fences acting as traps and concentrating cattle along streams, operation and maintenance costs, and other problems immediately exclude this option. Herding by the permittee with riparian use constraints appears to be the most viable approach to this point. It is costly to the permittee but less so than the above alternatives.

Interestingly, good riparian healing with high numbers (600-800 head) for a short time (approx. 6 days) in late summer and early fall can occur (R. L. Storch, Forest Supervisor, Grand Mesa, Uncompahgre-Gunnison National Forest, Colorado, pers. commun.). R. L. Storch (pers. commun.) stated that willows and other woody vegetation responded very favorably to this management approach. Fencing of the riparian area includes wide portions of the uplands and cooler temperatures during these times better disperses livestock during the short grazing period.

Some years ago the USFS developed the Integrated Resource Management approach, and I attended one of their training workshops on its use. The team was composed of individuals representing all resource areas. I never observed one of these teams that was not all USFS employees. With range resources being such a dominant part of this agency, I suspect that other resources were not adequately represented or listened to. Further, most team members are low echelon personnel who are knowledgeable of how decisions will be made regardless of their input.

Elmore and Kauffman (1994) have suggested using an interdisciplinary management team (i.e., soils, fishery biologists, botanists, and others) to visit allotments and formulate management plans. They present 6 general recommendations for the team. I observed 1 of these teams, in which all recommendations were carried out except possibly the last and most important action, which was insuring strong compliance recommendations. Further, this team was composed of well-trained, mature, and knowledgeable veteran personnel.

If the time and knowledge of such teams are to be used then the permittee must have an incentive to see that the recommendations are followed. If the team recommends leaving an X-cm tall stubble height over the floodplain for soil protection and sediment trapping in floodstage, it should be the permittee's responsibility to monitor this and report to the agency when limits are being approached. Random measurements by agency personnel would confirm utilization levels and give a date when all livestock must be out of the pasture. If utilization levels were exceeded by the permittee then no utilization would be allowed in the pasture the following year nor would excess numbers be allowed in other parts of the allotment. Attention to detail by the permittee would increase under this approach and the user would share management responsibilities with agency personnel. Most agencies do not have adequate range personnel to properly do their present work load, much less shoulder the total responsibilities of insuring compliance with the recommendations of a management team.

There is strong merit in using a team approach (Elmore and Kauffman (1994). Platts (1979:39) does not exactly suggest a team approach, but alludes to it by saying ". . . no single discipline possesses the skills and knowledge for all problem solving . . ." in riparian management. With an observed team, each discipline had its concerns expressed and the recommendations formulated were more ecologically sound than they would have been if a single person had attempted to encompass all disciplines.

Possibly a set of teams could be established over ecological regions in the West. Each team would be composed of agency and nonagency specialists and would respond to BLM, USFS, or private land managers requesting recommendations on riparian restoration or best grazing management practices. There are excellent people in and out of the agencies whose sole interest is to see range resources improve so that greater conflicts are avoided in the future. Because the permittee would accompany the team, for balance, I recommend that a conservationist who has shown interest in the natural resources on the allotment be invited. Both could enter discussions but *only* the condition of the resource and the team would dictate the management practices to be followed. Time and travel demands on team members may be such that nonagency personnel might have to be employed on a full-time basis. I further suggest these teams report directly to the Secretary or Assistant Secretary of the Interior to remove them from as much agency-permittee politics as possible.

The most striking difference between riparian recovery with total rest and riparian recovery with grazing is the time factor involved. With total rest, most of the systems that I have observed show tremendous change within 8-10 years. These general numbers seem to hold in Colorado (Schulz and Leininger 1990), central Oregon (Clifton 1989), Nevada (Medin and Clary 1989), and numerous stream systems throughout the West (GAO 1988). Other workers have also reported that exclusion provides the most dramatic and rapid rate of recovery in riparian systems (Beschta et al. 1991, Elmore and Kauffman 1994).

With managed grazing riparian healing time is twice (16-20 years) and maybe 4 times (32-40+ years) longer than exclusion. The important question is, can the most degraded riparian areas hang onto their thread of existence for another 30-50 years? They are very weakened and degraded after >100 years of unmanaged grazing, and my experiences are that the agencies and permittees, if willing, could not move rapidly enough with improved management for it to really begin within the next 10-20 years. It might come about faster than this if the managing agencies took a more aggressive role in management as is being done on the Uncompahgre-Gunnison National Forest (R. L. Storch, pers. commun.).

Allotments with improved management approaches ready to be implemented within the next 3-5 years should be enacted. Otherwise, agencies should develop and implement plans to remove cattle from heavily degraded riparian areas for a minimum of 5 years. This suggestion is not new in that many biologists have suggested that there is no grazing plan that allows riparian restoration (Ames 1977, Davis 1977, Behnke 1979, Dahlem 1979, Kindschy 1978, Szaro 1980). If cattle are to be grazed in the floodplain again, riparian habitats should be closely monitored with rigid utilization standards to insure continuing improvement.

KEYS TO BETTER RIPARIAN MANAGEMENT

Most agencies managing public lands in the West have some type of monitoring of upland habitats but have used different methods to evaluate the ecological conditions of western rangelands (NRC 1994). Much of the data are at best trend information, and Box (1990) reported that even these data were lacking for 12% of National Forests lands and 26% of BLM lands.

Monitoring is the key to knowing and documenting riparian improvement. These data are virtually absent for riparian areas in western rangelands. Their collection is essential if management is to recognize changes for improved riparian conditions. A beginning that is cheap, simple, and quick is to establish repeat photographic points at a few representative areas along a stream. A photograph showing the channel, banks, and floodplain at 2-3 year intervals the same month each year along with brief notes on mean channel width and bank and floodplain condition (presence or absence of young trees, percent groundcover of the floodplain) within 2 photographic periods would immediately inform the agency of needed management changes. Funds must be made available to perform these duties.

Too many land managers, both in the USFS and BLM, still believe that presently used grazing systems to improve uplands will also help improve riparian habitats. Related to this is that upper management (typically District Rangers and above and Area Managers and above) have very little knowledge of what management practices are needed to improve riparian areas and of resource conditions on the ground. Too little time is spent in the field by these people to appreciate problems and solutions.

Riparian habitats, unlike uplands, respond very quickly to improved management (usually within 2 years) unless highly degraded. Furthermore, there should be no controversy over what is improvement, stability, or degradation as there is in the uplands. Easily quantifiable and visible objectives can be established allowing the range conservationists and permittee to easily judge if conditions are being met. The consensus of the advisory team, irrespective of the agency person or permittee, would be the sole criterion on meeting riparian objectives.

A second key is to tie specific riparian improvement objectives to land managers' annual performance ratings. These people should be accountable both professionally and monetarily to improved riparian health.

The last key would be to make permittees accountable for riparian health as well. If grazing fees were based on riparian health there would be greater attention and concern to PFC.

Riparian conditions meeting objectives AUM = \$3.00
Riparian conditions improving, but not meeting objectives AUM = \$5.00
Riparian conditions unsatisfactory and not improving AUM = \$15.00

Short courses could be conducted for agency personnel and permittees (attending together) showing the various management targets to be achieved. This approach, I think, would foster a very close, cooperative working relationship to improve riparian conditions between permittees and agency personnel.

SUMMARY

Riparian habitats in western rangelands have exceedingly high values for society, fish, and wildlife. Their resource values far exceed their approximate 0.1% of the land area they cover. They serve to trap and stabilize eroded sediments, detoxify compounds, act as phosphorus sinks for soil enrichment, and serve as denitrification areas to provide high water quality. When functioning they provide bank storage of water and extend the flow regime to perenniality or increase instream flow.

They are vital to fishes when properly functioning by providing uncontaminated cool water, high in dissolved oxygen and low in suspended sediment. These important water quality parameters are all related to proper stream channel shape, bank stability, transport of sediment load, and the relationship of the stream to access its banks. Important physical factors to native salmonids are water velocity, water temperature, amount of dissolved oxygen, pool volume (number, size, and depth), escape cover, and annual discharge and flow. Highest quality fisheries exist under these conditions along with water quality and quantity.

In riparian habitats, vegetative components, in general, fulfill the ecological needs of the greatest array of wildlife species. In their order of importance are tree species and their densities, foliage profile, foliage volumes in the profiles, horizontal patchiness, and shrub species and their densities. The three most important vegetative components are satisfied in the Southwest by mature Fremont or narrowleaf cottonwood intermixed with willow species, in the Pacific Northwest by black cottonwood (*Populus trichocapra*) with willows, and in the Intermountain region by quaking aspen and willow or pure willow habitats. The shrub species and densities element may be most important in the Southwest, but pure willow communities may function similarly. High insect production in willows lends them added importance to wildlife.

Wildlife values in riparian habitats, i.e., density, species richness, biodiversity, number of rare species, number of breeding pairs of birds, and biomass, are extremely high compared to adjacent uplands. Highest fisheries values exist were streams are properly functioning. As knowledge becomes more complete, they may provide some of the most important fish and wildlife habitats in the coterminous United States. Southwestern riparian habitats may be vital to migratory wildlife as migratory corridors and resting, refueling, and wintering habitats in that they provide linear oases when uplands are least productive.

Though knowledge is limited, beavers are unquestionably a keystone species in riparian areas of second- to fifth-order streams. They have the ability to alter habitats by 20-40% by changing channel geomorphology and hydrology; by retaining sediment and organic matter; by creating and maintaining wetlands; by modifying nutrient cycling and decomposition dynamics; by modifying species composition and dynamics of plants; by influencing the movement of water and materials transported downstream; and by creating totally new fish and wildlife habitat that significantly increases biodiversity.

The development of the livestock industry was examined with Arizona being used as an example. Though drought in the 1890's expedited ecological degradation in Arizona, other western states reported similar ecological problems as ranges were overstocked. Riparian habitats suffered the greatest ecological damage by being both highly overgrazed and then experiencing unprecedented flood damage from degraded watersheds.

Livestock are attracted to riparian areas because of lush forage, shade, and water, especially in hotter, arid months. Current management approaches only consider the health and condition of the uplands or watersheds, which does not give grazing relief to riparian habitats. As late as the 1960s riparian habitats were viewed as sacrifice areas. Though fish and wildlife values in riparian habitats are extremely high, there has been little progress in making livestock management changes.

There are many agents of riparian destruction and degradation other than overgrazing by domestic livestock. Most of these have been along major western streams, while unmanaged livestock degradation has been ubiquitous and at all elevations.

Three conceptual phases were used to facilitate and illustrate the consequences of unmanaged livestock degradation of fish and wildlife resources in pristine riparian habitats (minus beavers) from the inception of livestock grazing to present. Completion of the process in Phase I took from 3-10 years depending on livestock density. Though the process was complete the degraded condition of the resource continued and in many instances it has intensified to present. Degradation includes destruction of overhanging banks, overburdened sediment loads, stream channel changes, widening of the channel, virtual elimination of the herbaceous groundcover and the understory, and the cessation of tree and shrub reproduction along the floodplain. Heavier than usual flood events eroded and widened channel widths and depths, and divorced the stream from accessing its floodplain. Sediment loads have increased concomitantly.

Consequences of bank destruction, channel widening, and high sediment loads severely impact fish populations. The elimination of understory trees and shrubs promotes higher water temperatures, lower oxygen levels, and reduces detrital input (the main energy source for small-order streams), further stressing fish populations. Species of small mammals, birds, reptiles, and amphibians requiring dense herbaceous vegetation or dense understory vegetation have experienced density declines and possibly local extirpation as these changes occurred about 1890 or even earlier throughout riparian habitats in the West.

Phase II was initiated about this time and it will take 100-125 years to complete. Beginning Phase II there were extensive and generally continuous riparian forests consisting of cottonwoods and willows, pure willows, or willow-quaking aspen along perennial streams in the West. Tree species abounded in their habitats, the foliage profile was complete (minus the understory), foliage volumes were high in the canopy and midstory, and intracommunity patchiness was high. Phase II is a slow and subtle process of degradation, almost imperceptible without repeat photography or quantified botanical data. Essentially it is the slow process of the youngest trees that escaped being eaten when cattle began heavily grazing these habitats circa 1870 to the point they begin reaching decadence. During the intervening years, older age classes of trees have lived out their life span slowly thinning the forests. This process, in turn lessens foliage volumes at the mid and overstory layers. As these young trees mature and there is no recruitment of young age classes the patchiness element declines as well. There is tree loss through natural mortality, but there are other losses as well, i.e., blow downs, beavers girdling trees, firewood or building material use, trees left perched as water tables declined, and trees washed out with more severe floods from degraded watersheds. Continuous forests over time have been fragmented and thinned, leaving small islands of trees or thin stringers.

The cumulative effects of all human activities in eliminating or degrading riparian habitats is beginning to manifest itself through the reduction in riparian resources in the West. Tree species and their densities, foliage profiles, foliage volumes, patchiness levels, and remaining forest sizes are approaching threshold levels where many more species should show declines in numbers or be locally extirpated.

Phase III is the collapse of the riparian deciduous forest. Most riparian forests are in late Phase II, while some are in early or mid-Phase III. It will take about 50 years to complete this phase. Management options are the (1) immediate use of managed grazing or (2) eventually be forced to use external seed sources with possibly less-adapted ecotypes and/or (3) revegetation efforts which are extremely expensive as compared to grazing management changes.

The management challenge of the twenty-first century will be the continued use of western rangelands while simultaneously healing riparian habitats. Abolition of livestock grazing on public rangelands and fencing are ruled out because of social acceptance and cost. The most viable method at present is herding with stubble height constraints. Strong incentives to both the land manager and permittee to restore proper functioning condition of western streams are key to restoring riparian habitat for optimum social, fish, and wildlife resource values. An approach is suggested.

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