The Status of Riparian Habitats in Pacific Northwest Forests*

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Introduction

It has been conservatively estimated that there are 74,356 km of running waters in the Pacific Northwest (Oregon, Washington, Idaho) that can or do support fisheries (Brinson et al. 1981). On public lands in Oregon, the Bureau of Land Management (1980) estimates that 71,135 ha are dominated by riparian communities. These ecosystems are among the most productive wildlife habitats in North America and the majority of wildlife species in the Northwest will utilize these during all or part of their life cycles (Thomas et al. 1979, Oakley et al. 1985). Optimally, riparian ecosystems are characterized by high degrees of species, structural, and spatial diversity. Critical ecological, physical, and biological linkages between riparian, aquatic, and upland ecosystems are intact.

Attempting to define the current ecological condition of riparian zones in relation to a pristine condition would be an exceedingly difficult task. There has been no single comprehensive inventory of riparian ecosystems in the United States to determine the amount of land originally covered by riparian ecosystems and the proportion of that area presently supporting natural riparian communities (Brinson et al. 1981). Historical analysis of human influences on streams indicates that through activities such as beaver trapping, debris removal, timbering, farming, and ranching, much of the biological integrity of streams and rivers was lost long before extensive research began (Harmon et al. 1986, Naiman et al. 1986). In short, it is difficult to compare the current status of riparian ecosystems with that of undisturbed systems because few of the latter exist. However, there are sufficient data to conclude that, in general, riparian ecosystems are not receiving adequate protection or management. Judging by the concern and attention presently focused on riparian zone management, it is probably safe to say that the overall status of riparian zones in both multiple- and single-use forests is far below what is acceptable to the concerned American public.

Different land-use activities will differentially affect community structure and, hence, the status of the riparian habitat. Severe impacts (e.g., those from dams, diversions, agriculture conversion, stream channelization, road construction, etc.) have permanently altered millions of acres of wetland habitat. Brinson et al. (1981) estimate that 23 million acres of the original floodplain forest has been converted to urban and cultivated agricultural land uses in the United States. Klopatek et al. (1979) estimate that northern floodplain forests have decreased 69 percent in area from their potential, and Hirsh and Segelquist (1979) estimate that 70 to 90 percent of all natural riparian areas have been subjected to extensive alteration. Little is known about the extent and status of mountain riparian ecosystems, which are affected primarily by impacts associated with other natural resource uses (i.e., timber harvest, recreation, livestock grazing, etc.). Recent federal and state surveys have found that 50 percent of all fish habitats on public and private lands in western Oregon have been altered since 1960 (Kadera 1987).

Undisturbed riparian ecosystems are typically characterized as those having a high spatial heterogeneity of plant communities of varying age classes or seral stages. Each seral

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stage is utilized by different species of wildlife; however, the greatest use has been described for those late-successional arboreal riparian communities dominated by a mix of coniferous and deciduous species. In intensively managed forests with short rotation schedules and no riparian zone protection, these communities may never develop and hence the biotic potential of the riparian ecosystem is never fully attained. If productive riparian/stream ecosystems are desired for a forest, then alterations in land use such as the establishment of buffer zones and extended rotation schedules may be necessary.

Overgrazing by livestock can result in declines in the productivity of early and mid-seral communities and retard succession to hardwood-conifer dominated riparian communities. Minimization of this disturbance is possible and may be necessary for the preservation of high-quality riparian zones or the improvement of degraded ecosystems.

The public outcry about the impoverished status of many riparian ecosystems in the Pacific Northwest should motivate resource personnel and scientists to develop and implement management alternatives to improve the current situation. This includes management to improve degraded riparian zones and activities geared to protect or prevent the decline of high-quality riparian zones.

In this paper I will review the ecological components relevant to the conceptualization of an optimal status for riparian ecosystems. These include the concepts of diversity, succession, and juxtaposition of riparian communities. In addition, the causal factors affecting the current and future status of mountain riparian ecosystems are discussed.

Concepts of Diversity that Aid in the Definition of Riparian Habitat Status

Most biologists, scientists, and land managers have some conceptual idea of optimum riparian wildlife habitats. Typically, they are seen as areas that provide an exceptionally high number of niches as a result of high species and structural diversity.

When relating the value of riparian ecosystems to wildlife communities, concepts of diversity can be helpful descriptors. Components of diversity are species richness, or the number of species, and heterogeneity, or the relative abundance of each species (Krebs 1978). Many riparian plant communities have high species diversities, as well as high structural diversities. Species diversity refers to the number and distribution of species in an area, and structural diversity is the number and distribution of vegetation layers. Diversity can be further broken down into alpha diversity (within a community), beta diversity (between communities), and gamma diversity (between ecosystems).

All three communities in Table 1 have a high alpha species diversity. This is evidenced by the high plant species richness for the communities (73-85), as well as by the high values of species diversity. In contrast, the dry meadow has a low and the black cottonwood (*Populus trichocarpa*) community a high alpha structural diversity (1 versus 5 vegetation layers). Foliage height diversity is often correlated with bird species diversity (MacArthur 1964; Balda 1975), and this is congruent with the high avian species diversity, density, and nesting use in the cottonwood community.

In riparian zones, hydrologic fluctuations, in concert with geomorphic cycles, biotic interactions, and, sometimes, human activities, result in an ecosystem characterized by a mosaic of vegetation habitats with many different seral stages of each habitat. These are the primary causal factors for the high beta diversity (between habitats) in riparian zones. The juxtaposition of arboreal hardwood, shrub, herbaceous, and aquatic communities results in a high species richness in both plant and animal communities relative to the more homogeneous uplands (Davis 1977). The high beta diversity in the Wallowa Mountain riparian ecosystem of Table 1 results in high avian species richness, foraging guilds, and nesting species of the riparian zone. Eighty-one species of birds utilized this 3-km strip of riparian vegetation, and 34 species nested in this ecosystem.

From a landscape ecology perspective, gamma diversity or between-ecosystems diversity is a useful concept in describing the value of riparian ecosystems for wildlife species in the context of entire watersheds. In desert and shrubland ecosystems, the diversity of the watershed (gamma diversity) is greatly enhanced by the structure of the riparian

Table 1. Avian and vegetation parameters for three plant communities in the Catherine Creek riparian zone, Wallowa Mountains, Oregon. Data are from Kauffman et al. (1982) and Kauffman et al. (1985)

	Community		
	Kentucky bluegrass meadow	Douglas hawthorne shrub	Black cottonwood forest
Avian parameters[1]			
Density (number/ha)	21.3	29.5	34.8
Species richness (S)[2]	22	21	28
Foraging guild richness (F)	8	8	9
Nesting species richness (N)	3	14	23
Species diversity (H')[3]	2.1-2.2	2.3-2.4	2.3-2.9
Equitability (J')[4]	.69-.78	.83-.84	.73-.96
Ecosystem totals[5]		S = 81, F = 15, N = 34	
Vegetation parameters[6]			
Species diversity (H')	1.9-3.1	2.9-3.4	2.8-3.0
Equitability (J')	.58-.81	.76-.85	.75-.85
Species richness (S)	78	85	73
Vegetation layers	1	2-3	5
Biomass of herb layer (kg/ha)	2463-4173	1462-2498	938-2688

[1]Avian parameters were data collected during the nesting brooding season.
[2]Species richness = total number of species encountered during the census period.
[3]Species diversity (H') = Σ pi $\log_e$ pi (Shannon-Weaver Information Formula).
[4]Equitability (J') = H'/$\log_e$S.
[5]Total avian population parameters from census taken during May to October.
[6]Vegetation parameters were collected late summer during peak expression of species richness and biomass.

areas as these are the only arboreal-dominated communities present. In mid-elevation coniferous forests, the presence of riparian hardwood, shrubland, and meadow habitats will enhance the diversity of the ecosystem and, hence, the wildlife community. Even in subalpine forests, where arboreal riparian communities do not exist, the gamma diversity is enhanced by the presence of willow (*Salix* spp.), huckleberry (*Vaccinium* spp.), and herbaceous plant-dominated communities that are of a different physiognomy and composition than the coniferous uplands. In other words, these habitats provide a resource or niche unique to the riparian zone.

What are Optimum Riparian Conditions?

Optimum riparian conditions are relative to the inherent site potential of any given stream corridor. The unique combinations of hydrologic, geomorphic, biotic, and human-caused processes may combine to cause the inherent productive capacity in one riparian zone to far exceed that of another. In general, a high status riparian/stream ecosystem supports a high diversity of wildlife because of high resource productivity, complex vegetation structure, high edge-to-area ratios, the presence of water, and a unique microclimate.

Animal species richness is presumed to be enhanced when the overstory, midstory, and understory layers are present and equally distributed throughout the riparian area (Short 1985). The classic example of high wildlife is that of riparian arboreal communities dominated by deciduous hardwoods, sometimes codominant with conifers (Table 1; Glinski 1977, Brady et al. 1985, Kauffman et al. 1985). In eastern Oregon and Washington, these are usually dominated by black cottonwood, mountain alder (*Alnus incana*), and quaking aspen (*Populus tremuloides*). West of the Cascade divide, they may be dominated by black cottonwood, red alder (*A. rubra*), big-leaf maple (*Acer macrophyllum*), Oregon ash (*Fraxinus latifolia*), and conifers. In the Wallowa mountain riparian zone (Table 1), these communities provided nesting habitat for 23 bird species. Communities with lower structural diversity had fewer nesting species. Similar results have been found in cottonwood habitats in other regions of the U.S. (Stauffer and Best 1980, Meents et al. 1984). Meents et al. found that avian habitat specialists were primarily those that were restricted to arboreal cottonwood/willow communities. In an Arizona riparian cottonwood stand, Carothers and Johnson (1975) recorded the highest population of noncolonial nesting birds ever reported in North America.

In describing "optimum riparian habitat," we must recognize that what is optimum nesting habitat for a mallard (*Anas platyrhynchos*) is totally unacceptable for a killdeer (*Charadrius vociferus*) (Oregon-Washington Interagency Wildlife Council 1978). Even though cottonwood communities support the greatest number of species, there are obligate species dependent on other communities as well (Kauffman 1982). Therefore, a riparian zone with only a single but structurally diverse community type may not have the wildlife productivity of one with a high beta diversity (i.e., one with a high degree of horizontal patchiness). In the Wallowa mountain riparian zone, judged a productive habitat for many wildlife species, the high beta diversity was reflected in the description of 258 vegetation stands representing 60 plant communities (Kauffman et al. 1985).

Problems with the Assessment of Riparian Ecosystem Status

There have been few successful attempts to develop methods to quantify the value of different riparian communities to terrestrial wildlife. Short (1985) and Garcia (1985) have developed algorithms for assessment of the value of riparian zones to wildlife resources. Like all resource value ratings, these are based on the subjective judgment of the assessor. Part of the problem in identifying the status of a given riparian ecosystem is identification of site potential. Classifications of Northwest riparian ecosystems are only available in limited areas (Ganskopp 1978, Kauffman et al. 1985, Youngblood et al. 1986, Kovalchik 1987). In addition, few studies have been conducted on the identification of successional pathways or condition classes of riparian plant communities. A major problem in classification and delineation of riparian communities arises from the difficulties of applying classification theory developed in upland communities. The processes and patterns of succession, stability, and site potential are different in riparian ecosystems.

The Current and Future Status of Riparian Ecosystems

Changes in riparian ecosystems occur when they are utilized by man. These alterations are perceived as ecological stresses that change the pattern of energy flow and the movements of materials to and from riparian ecosystems (Brinson et al. 1981). As a result, wildlife habitats are altered. Significant human-caused impacts on riparian ecosystems began approximately 450 years ago in the southwestern U.S. (Davis 1977) and in the early 1800's, with the arrival of trappers, in the Pacific Northwest. In the last 100 years the rate of alteration has increased significantly due to ever-increasing human pressures (Davis 1977). Hirsh and Segelquist (1979) estimate that 70 to 90 percent of the riparian areas in the United States have been altered to some degree. In comparison with estimates of the loss of natural vegetation in uplands, riparian lands have been categorized as among the most severely altered ecosystems in the United States (Brinson et al. 1981).

Severe impacts are those that result in a permanent change in the structure and function of riparian ecosystems. These include activities oriented towards stabilizing the dynamic nature of fluvial processes (i.e., the construction of dams and impoundments, channelization, etc.) and permanent land-clearing activities (farming, mining, etc.). These typically decrease many fundamental ecosystem properties such as species diversity, rates of primary productivity, nutrient cycling, and animal productivity (Brinson et al. 1981). Mountain riparian ecosystems probably have not changed as distinctly as those in lowland floodplain areas. Even though there have been some agricultural activities, construction of dams and roads, mining, and so on, most of the vegetation along mountain streams has been maintained by near-normal ecological processes (Horton 1972). Impacts on mountain riparian ecosystems of the Pacific Northwest have largely come from activities associated with logging and road construction, livestock grazing, recreation, and beaver trapping. The amount and extent of long-term decline in the biotic potential of riparian communities due to these activities has not been quantified and will vary among riparian ecosystems. Some will be more resilient than others. The ecological factors that must be considered are those associated with alterations in the biotic, fluvial, and geomorphic processes, environmental conditions, and the extent to which propagules are available for reinvasion.

Biologists, naturalists, outdoor enthusiasts, foresters, livestock operators, and land managers have long recognized the highly productive nature of riparian zones. With respect to wildlife, these are considered the most productive habitats in the Pacific Northwest (Thomas et al. 1979, Oakley et al. 1985). Lewke (1975) reported that the number of birds in riparian communities is two to four times that in equal areas of non-riparian communities. Cross (1985) found that riparian zones support a more diverse small-mammal fauna than the adjacent forest uplands.

Depending on management, activities such as timber harvest, grazing, and recreation may cause a change in vegetation (composition and structure), stream morphology, and water quality. As a result, the productivity of the terrestrial vertebrate component may decline. Reducing the size of a community type progressively eliminates species requiring large areas of the particular type and favors expansion of species associated with new habitat. Many avian species (as well as other forms of wildlife) that utilize riparian zones are ubiquitous, but some species have narrow ecological requirements. A high fidelity to a particular plant community may limit the capacity for persistance in an area if the favored community is rare or requires a long period of time to develop after disturbance (Meents et al. 1984). Cottonwood and other deciduous arboreal communities are classic examples of this phenomenon in riparian zones. Decreases in the quality and quantity of riparian ecosystems have contributed to the widespread elimination of several avian species (Goldwasser et al. 1980, Gaines and Laymon 1981, Serena 1982, Gray and Graves 1984).

Livestock-Riparian Relationships in Northwest Forests

Although riparian meadows cover only 1 to 2 percent of forested range in the interior Northwest, they have been estimated to account for 81 percent of the total herbaceous vegetation removed by livestock (Roath and Krueger 1982). The main attributes believed to attract livestock to riparian areas are similar to those that attract wildlife (i.e. availability of water, shade, thermal cover, and quality and variety of forage) (Kauffman and Krueger 1984).

If grazing in riparian zones is excessive, the vegetation, and hence the wildlife habitat, can be degraded, altered, or eliminated (Oakley et al. 1985). Impacts to riparian vegetation induced by livestock can basically be separated into four areas: compaction of soils, which increases runoff and decreases water availability by plants; herbage removal, which lowers plant vigor and changes competitive interactions among species; physical damage to vegetation by rubbing, trampling, and browsing; and changes in the fluvial processes, which may lower water tables and/or cause a decline in invasion sites for woody vegetation (Glinski 1977, Severson and Boldt 1978, Kauffman and Krueger 1984, Brady et al. 1985).

Reproduction of tree populations is affected by browsing on young plants (Glinski 1977, Dahlem 1978, Kauffman et al. 1983). Without population recruitment of young trees, riparian forests will develop unstable age structures and may eventually be eliminated. This would greatly affect the long-term suitability for the wildlife species that utilize these communities. Coarse woody debris inputs will be decreased, and the hydrologic properties on the stream will therefore be altered. This, in turn, would further affect the composition and structure of riparian vegetation.

Management systems designed to improve the status of riparian habitats are now being developed. With proper management, it may be possible to graze livestock and preserve or improve desired wildlife habitats in riparian zones (Platts 1982, Kauffman et al. 1983, Bryant 1985). There are numerous solutions to the problem for those willing to recognize that a problem exists and to accept responsibility for solving it by implementing the solution (Davis 1985). Control of the timing and intensity of livestock use are key factors to good management. However, the response of riparian communities to grazing can vary, resulting in variation in management needs. For example, Marlow and Pogacnik (1985) report that if utilization is 20 percent or less, little degradation and possibly improvement in riparian habitats will occur. Bryant (1985) indicates that riparian productivity will increase if forage utilization is less than 70 percent.

Timber-Riparian Relationships in Northwest Forests

Riparian zones are often highly productive timber sites. Anderson (1985) estimated that there are 2 billion board feet of harvestable timber within 100 feet of streams in the Siskiyou National Forest, Oregon. Timber management activities and their associated road systems may affect the status of riparian wildlife habitats by decreasing the structural diversity, increasing (or decreasing) downed woody debris, altering the microclimate, and changing the water column. The more dramatic regeneration cuts (i.e., clear-cutting, seedtree, and shelterwood) will have a greater impact than single tree or group selection cuts (Thomas et al. 1979, Oakley et al. 1985). Prior to the 1970's, riparian areas were clear-cut and roaded with little regard to the environment, particularly along nonfish-bearing streams (Anderson 1985). In these riparian zones and in those areas where buffer strips have failed or were not left, the habitat suitability for those wildlife species that require mature communities with a high degree of structure has greatly declined.

Historically, streams contained massive accumulations of woody debris that created a complex aquatic environment (Harmon et al. 1986). These channel constrictions caused frequent floodplain flooding and developed a complex network of channels (Everest et al. 1985). Those processes associated with large organic woody debris influenced the site potential for establishment of many riparian species. Oakley et al. (1985) review studies that suggest that in small headwater watersheds, road building, clear-cutting, and other activities associated with timber harvesting may result in significant increases in annual water yield, decreases in summer flows, and increases in large major winter peak flows if more than 5 percent of the watershed is compacted. As fluvial processes have been demonstrated to directly influence the composition and successional pathways leading to arboreal hardwood dominance, these land-use activities may drastically alter the long-term ecological status of any given riparian zone.

In order to protect stream corridors, buffer strips are left following clear-cuts on public lands. Though not without problems, well-designed leave-strips have generally been successful in achieving management objectives for water quality and fish habitat (Oakley et al. 1985). Buffer strips decrease erosion (and pollution), retard runoff, trap sediments and nutrients, maintain suitable water temperatures for aquatic life, and provide vegetation and invertebrates as food sources for birds and other wildlife (Curtis and Ripley 1975).

Most riparian legislation has been directed toward protection of fish habitat. There have been few studies in forests that have determined the effects of buffer strips on wildlife communities. The National Forest Management Planning Act and its implementing regulations require special consideration be given to a minimum of 100 feet (about 30 m) on each side of perennial streams. Effective leave-strip width will vary by stream order, stream width, topography, vegetation, and management objectives. Erman et al. (1977) recommend buffer strips of at least 30 m to protect aquatic life from logging. Brinson et al. (1981) found that many vertebrates, especially riparian mammals, reptiles, and amphibians, concentrate their activities well within 60 m of water.

Stauffer and Best (1980) found that a 200-m-wide strip of riparian vegetation is needed to accommodate the breeding territories of most songbirds. However, this study was not in a forested ecosystem. In the Pacific Northwest, the optimum or minimum size of streamside leave strips that are self-maintaining and can provide habitat for both resident and transient wildlife remains to be determined (Cross 1985).

Even though there is no clear consensus on the minimum size riparian buffer strip needed to accommodate wildlife populations, such strips have been shown to preserve or maintain the high status of riparian habitats to some forms of wildlife. Cross (1985) found that buffer strips maintain populations of several types of small mammals, indicating a variety of habitat niches are also maintained. Stauffer and Best (1980) predicted 32 out of 41 breeding bird species would be eliminated when all woody vegetation was removed from riparian zones. Six species were predicted to be eliminated when woody vegetation was decreased to a narrow strip along streams.

There is an apparent economic cost to the timber industry in protecting riparian zones. Typically, timber harvest will remove 40 to 50 percent of the standing timber in riparian zones associated with nonfish bearing streams and up to 10 percent along streams that support fish (Anderson 1985). Anderson estimates that the annual sale of timber was decreased 13 percent to protect riparian areas and fish resources. No studies and little data could be found that documented or predicted the aesthetic, biological, or economic costs associated with the apparent decline in wildlife resources due to timber harvest activities.

Conclusion

Riparian ecosystems occupy a small percentage of the total area in forested regions of the Northwest. For example, in the Blue Mountains of eastern Oregon they occupy 1 to 2 percent of the land area. Even though they occupy small areas, they are the ecosystem most heavily utilized by a large number of wildlife species. Riparian zones are also disproportionately valued for other forest and range uses. They are among the most productive timber and forage-producing sites. People also utilize riparian zones heavily for recreational purposes.

Quantification on a systematic basis of the status of riparian ecosystems and the extent of their alteration has not been accomplished, but it has been estimated that 70 to 90 percent of all riparian zones have been altered in some way. In order to quantify the status of any given riparian ecosystem, successional patterns and the inherent biotic potential of the system must be described. Currently, these are generally abstractions, hypotheses, or qualitatively based descriptors.

The status of a riparian ecosystem is a value judgment, and the needs of society must therefore be considered in ascertaining it. Technology is available to maintain or improve the status of riparian ecosystems for wildlife in forested zones. The degree to which this technology is being utilized in land management will certainly affect the future ecological condition and productivity of riparian zones. Management activities should vary according to the unique features of the riparian zone, the desired land uses, and the targeted wildlife species. The management philosophy that will influence the future of riparian wildlife populations, water quality, timber production, or livestock production will ultimately reflect the importance of these resources to the citizens of the Pacific Northwest. Land managers, scientists, and biologists have much to offer by developing and implementing management techniques that insure the ecological integrity and sustainability of these valuable zones.

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