Nutrient Cycling at the Land-Water Interface: the Importance of the Riparian Zone[1]

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Abstract.—Riparian ecosystems are sites of important biogeochemical processes that affect the composition and structure of the streamside biota as well as aquatic systems. Microbial activity, coupled with the slow diffusion of oxygen in waterlogged riparian soils, causes anaerobic conditions and reduction in reduction-oxidation (redox) potential. Redox potentials provide a useful measurement as to the intensity of anaerobic conditions and the degree of chemical transformation in riparian zones. Under anaerobic conditions, biogeochemical cycles differ greatly from the aerobic conditions of surrounding uplands. Different plant species are adapted to survive in varying levels of reduced waterlogged conditions such that different plant communities occupy sites of different redox potential. For example, Kentucky bluegrass *Poa pratensis*-timothy *Phleum pratense* communities occurred in areas of high redox potential (well drained soils) (570 mV), beaked sedge *Carex rostrata* in moderate redox potential (-48 mV), and tall mannagrass *Glyceria grandis* (-164 mV) in waterlogged, anaerobic soils. Recent research has shown that low redox potentials result in denitrification and that phosphorus immobilization processes occur within intact riparian ecosystems, thereby greatly influencing water quality. The effects of grazing or other anthropogenic disturbances can alter these biogeochemical cycles resulting in drastic alterations in riparian vegetation composition and productivity, aquatic ecosystems, and water quality. Given these important linkages to resource productivity, the effect of land use activities on biogeochemical cycles of riparian/stream ecosystems should be considered.

Riparian ecosystems perform a variety of functions essential to a healthy instream community and high water quality (Swanson et al. 1982). Brinson et al. (1981) estimated that over 70% of riparian communities have been altered and less than 2% of the land area in the USA consists of intact natural riparian communities. Major losses of riparian bottom land forest have occurred as a result of lumbering, drainage for agriculture (Korte and Fredrickson 1977; McGill 1979) or channelization (Funk and Robinson 1974; Barclay 1978; Triska et al. 1982). Historical activities such as debris removal, beaver *Castor canadensis* trapping, and grazing reduced the biological integrity of riparian zones before extensive research began (Harman et al. 1986; Naiman et al. 1986). These changes have greatly altered riparian ecosystem structure and function thereby greatly changing biotic interrelationships between riparian and aquatic ecosystems.

Livestock are believed to be attracted to riparian areas by the availability of water, shade, thermal cover, and quality and quantity of forage (Kauffman and Krueger 1984). Although riparian meadows occupy only 1-2% of the interior northwestern USA, they account for 81% of the forage removed by livestock (Roath and Krueger 1982). In reviewing livestock-riparian relationships, Kauffman (1988), separated the impacts of excessive grazing in the riparian zone into four components: (1) compaction of soils which increases runoff and decreases water availability to plants, (2) herbage removal which lowers plant vigor and changes competitive interactions among species, (3) physical damage to vegetation by rubbing, trampling, and browsing, and (4) changes in fluvial processes which may lower water tables and/or cause a decline in invasion sites for woody species.

Dahm et al. (1987) have concluded that human influences (beaver trapping, channelization, and grazing) have acted to uncouple riparian and stream ecosystem interactions. This has important implications as riparian zones are physical, biological, and chemical links between terrestrial and aquatic environments. Important, yet little studied ecological processes of riparian zones are the biogeochemical cycles that influence both riparian and aquatic systems. Riparian areas are unique in that they contain large areas of anaerobic (saturated) soil. In saturated soils, measurement of reduction-oxidation (redox) potential provides a useful indicator of the intensity of anaerobic conditions and therefore the degree of chemical transformation and the biogeochemical pathway operating in the riparian zone.

Nutrient inputs resulting from the biogeochemical processes of riparian zones not only affect the associated aquatic community, but also the downstream industrial, recreational, and agricultural water users as well (Coats et al. 1976; Rhodes et al. 1985). Pristine or intact riparian zones have been found to function as important sites for denitrification (Coats et al. 1976; Lowrance et al 1984; Jacobs and Gilliam 1985). Additionally, riparian zones are important phosphorus sinks where phosphorus adsorbed to clay particles is deposited during high flow or trapped in *runoff from surrounding uplands allowing time for plant* uptake or microbial use (Yarbro 1979; Cooper and Gilliam 1987; Cooper et al. 1987). Given the critical role biogeochemical cycles play in riparian ecosystem structure and function, management strategies will need to be designed to insure these ecosystems are functioning in a dynamic equilibrium.

Methods

The data presented in this paper are from ongoing studies of the Catherine Creek riparian ecosystems. This research area is located in the foothills of the Wallowa Mountains of northeastern Oregon. Catherine Creek is a third order tributary of the Grande Ronde River, average discharge is 3.4m³/sec (USGS 1981) and peak flows usually occur from April to early June. Elevation of the study area is approximately 1030 m, and mean precipitation is 60 cm, the majority of which occurs as snow during the winter months.

Research in the Catherine Creek riparian zone has been conducted since 1978. Kauffman (1982) subjectively identified a total of 256 stands of vegetation representing 60 discrete plant communities. Livestock impacts on plant communities and streambanks were reported by Kauffman et al (1983a,b).

In four riparian plant communities we measured redox potential using platinum probes that were constructed using the methods of Mueller et al. (1985). Salt bridges (Veneman and Pickering 1983) were used to facilitate measurements when the upper soil layers were dry. Millivolts were read using a Beckman pH meter.

Importance of Riparian Areas

Streamside vegetation is a major source of energy and nutrients for instream communities. Cummins and Spengler (1978) estimated that 90% of the organic matter supporting headwater (first order) stream communities was of terrestrial origin. In these small streams of forested ecosystems, up to 99% of the energy input may be from allochthonous sources, and 1% is derived from instream (autochthonous) sources (Cummins 1974).

Riparian vegetation reduces the velocity and erosive energy of overbank flow (Li and Shen 1973; Petryk and Bosmajian 1975). Reduction of water velocity allows the riparian zone to function as a site of sediment deposition, trapping sediments that would otherwise degrade fish habitat and water quality (Meehan et al. 1977; Cooper et al. 1987). Cooper et al. (1987) found that riparian woodlands bordering agricultural fields in North Carolina trapped 84-90% of the sediment transported from these fields. This same riparian zone was found by Cooper and Gilliam (1987) to retain 50% of the phosphorus leaving agricultural fields. Yarbro (1979) found the sediments of a North Carolina stream trapped 1.7 kg/hectare of phosphorus.

Overhanging riparian vegetation moderates the thermal regime of aquatic communities by providing cooler instream temperatures (Meehan et al. 1977; Swanson et al. 1982). Removal of riparian vegetation can result in increases of water temperatures and decreases in levels of dissolved oxygen and carbon dioxide (Brown 1983). Warm water is conducive to algal and bacterial growth including bacterial species pathogenic to fish (Brett 1956). Less oxygen available for aerobic life processes will reduce populations of heterotrophs, increasing the amount of nutrients exported from the watershed.

Unlike upland areas, soils of riparian zones can be saturated throughout the year because of their low position on the landscape and close proximity to water. Even during periods of summer base flow, the deeper soil horizons remain saturated. Under saturated conditions the demand for oxygen exceeds the supply and anaerobic conditions develop in the soil profile. Anaerobiosis is not uniform throughout a submerged riparian soil profile (Patrick and DeLaune 1972; Ponnamperuma 1972). A thin oxidized layer at the soil-water interface results from the diffusion of oxygen from water or the atmosphere into the soil. The depth of this aerobic layer ranges from a few millimeters to a few centimeters and is controlled by the oxygen demand

of the soil microorganisms (Gambrell and Patrick 1978). This aerobic layer is of tremendous significance with respect to nutrient cycling. This is a small refuge of aerobic bacteria. These bacteria are responsible for nitrification processes (conversion of ammonium to nitrate). This form of nitrogen is soluble and negatively charged and therefore subject to leaching or denitrification in the anaerobic layer. Redox potentials can provide a useful measurement of the intensity of anaerobic conditions and therefore the degree of chemical transformation in riparian zones.

Effects of Redox Potential on Nutrients

Redox potential is a measure of electron availability in chemical and biological systems (Mitsch and Gosselink 1986). Redox potential can provide a relative measure of the availability of certain nutrients and chemical conditions in saturated soils (Table 1). The reduced form of many elements, such as iron, manganese, and sulfide, can be toxic to plants and occur under conditions of low redox potential. Chemical reactions such as denitrification also occur under low redox potential and have a direct effect on water quality. Low redox potentials (200 to -400 mV) are associated with reduced, submerged soils. Well oxidized soils have redox potentials of about 300 to 800 mV (Ponnamperuma 1972). Turner and Patrick (1968) found that when soil oxygen was at a level of 4%, redox potential began to decrease sharply. Much of this drop in redox potential results from increasing populations of anaerobic bacteria which use inorganic compounds such as nitrate-nitrogen as terminal electron acceptors (Brock et al. 1984). This reduction and use of nitrate-nitrogen as a terminal electron acceptor is termed denitrification. Denitrification plays a *major role in water quality by removing nitrate-nitrogen from the riparian system and converting it to gaseous* forms.

Reduction of nutrients occurs in a predictable pattern as redox decreases (Table 1). This process begins within a few hours to a few days after waterlogged conditions occur depending upon soil temperature and organic matter content (Glinski and Stepniewski 1985). Warm spring temperatures, high amounts of organic matter, water saturation, and rapid microbial growth will result in rapid declines in redox potential (Meek and Stolzy 1978). After waterlogged conditions occur in the spring or winter, oxygen is rapidly depleted by aerobic microorganisms. Oxygen is undetectable at a redox potential below the range of 320 to 340 mV (Gambrell and Patrick 1978).

Redox potential decreases as obligate and facultative anaerobic microorganisms begin to utilize nitrate-nitrogen as a terminal electron acceptor during the process of respiration. Denitrification occurs at about 230 mV (Gambrell and Patrick 1978). Denitrification in riparian zones can be important for maintenance of high water quality. Rhodes

Table 1.—Oxidized and reduced forms of several elements and approximate redox potentials for transformation. (From Mitsch and Goselink 1986.)

Element	Oxidized form		Reduced form		Redox potential (mV)
Nitrogen	NO_3-	Nitrate	N_2O, N_2, NH_4		220
Manganese	Mn^{4+}	Manganic	Mn^{2+}	Manganous	200
Iron	Fe^{3+}	Ferric	Fe^{2+}	Ferrous	120
Sulfur	SO_4^{2-}	Sulfate	S^{2-}	Sulfide	-75 to -150
Carbon	CO_2	Carbon dioxide	CH_4	Methane	-250 to -350

et al. (1985) found that in an undisturbed watershed of the Sierra Nevada over 99% of the incoming nitrate-nitrogen was converted to nitrous oxide (N_2O) or elemental nitrogen (N_2).

Reduction of manganese from manganic (Mn^{4+}) to manganous (Mn^{2+}) form occurs at a redox potential of approximately 200 mV (Gambrell and Patrick 1978). Reduced Mn is mobile in the soil profile and can be toxic to plants at high concentrations (Armstrong 1982). When redox decreases to around 120 mV, ferric iron (Fe^{3+}) is reduced to ferrous iron (Fe^{2+}) (Gambrell and Patrick 1978). The reduction of Fe^{3+} to Fe^{2+} is most discernable since the brownish-red color of Fe^{3+} hydroxides change to the grey or bluish-grey of Fe^{2+} hydroxides. In soils that are seasonally waterlogged, a mottled pattern of Fe^{3+} and Fe^{2+} is apparent. A soil horizon with this mottled pattern or solid bluish-grey matrix is referred to as a gleyed horizon (Buol et al. 1980). The red iron deposits often seen at the outflows of springs or seeps are an indication of reducing conditions in the soil profile behind the spring. Reduced iron (ferrous) is water soluble and can move with ground water until it reaches an aerobic environment where it is oxidized to ferric iron and precipitates out of solution.

Sulfur compounds are the next electron acceptors after iron as redox potential decreases. Sulfates are reduced to sulfides at a redox potential of about -150 mV (Gambrell and Patrick 1978). This is typified by a "rotten egg" smell characteristic of some marshes and riparian communities. Conversion of carbon dioxide (CO_2) or methyl groups to methane occurs when CO_2 and methyl groups are used as electron acceptors by certain bacteria. This requires a low redox potential (-250 to 350 mV) and the absence of all other terminal acceptors (Mitsch and Gosselink 1986). This appears to be the final redox system in soils (Ponnamperuma et al. 1966). Ferrous, manganous and sulfide ions can be toxic to plants at high concentrations (Armstrong 1982). However, low levels of these nutrients reduce the competitive ability of plant species that are not adapted for growth in soils of low redox potential. This is reflected in the distribution of plant communities over landscapes influenced by shallow water tables. For example, Pierce (1953) found in Wisconsin, sedge *Carex* spp. communities occupied sites of lower redox potential (-318 mV) than cattail *Typha* spp. communities (-216mV). In the same study aspen *Populus tremuloides* communities occupied sites of higher redox potential (-107mV) than black spruce *Picea mariana* communities (-262mV).

Under lentic situations, redox potential decreases with increasing depth from the soil surface (Gambrell and Patrick 1978; Howes et al. 1981; DeLaune et al. 1983; Bertness and Ellison 1987). However, we have found different situations in riparian zones. In a *Carex-Poa pratensis*-mixed forb riparian community along Catherine Creek, the redox potential did not uniformly decrease with increasing depth (Table 2). The redox potential data (Table 2) were collected 6 July 1988. At this sampling date, water table depth was 35 cm. Reduced conditions existed at the 5-and 10-cm depth (112 and -116 mV, respectively). However, below the water table redox potential increased to 374 mV. The higher redox potentials found in the lowermost sampling depth may be a function of subsurface flow of oxygenated water.

Ecological inferences on vegetation-environmental dynamics can be made from comparisons of these data (Tables 1 and 2). At all of our sampled depths, reduced forms of nitrogen, manganese, and iron were present. Redox readings at the 5- to 10-cm depth indicated that Fe^{3+}

Table 2.—Mean (SE) soil redox potential in a *Carex-Poa pratensis*-mixed forb community along Catherine Creek, Oregon, 6 July 1988.

Depth (cm)	Redox potential (mV)
5	112 (25)
10	-116 (18)
30	386 (32)
60	374 (7)

was reduced to Fe^{2+}. This site is restricted to those plant species that have adaptations to facilitate survival in an environment where reduced iron is present. Anthropogenic activities that change water relations of this community could alter redox potential and ultimately change vegetation composition.

The effects of human-induced disturbance on redox potential, pH, organic matter decomposition, and therefore, the rates of nutrient cycling are only beginning to be investigated in riparian ecosystems. Redox potential and subsequent effects on soil chemistry strongly influence the patterning of vegetation communities. Many riparian species are intimately adapted to survive periods of low soil redox potential. These adaptations may include aerenchyma, thickened cuticles, and oxygen loss at the root tip (radial oxygen loss) (Armstrong 1972, 1982). Different types riparian plant communities will display differences in redox potential. For example, we sampled redox composition along a gradient of decreasing soil saturation during the period of vegetation anthesis (July 1988). Redox potential at the 10-cm depth was -164 mV for emergent, tall mannagrass *Glyceria grandis* stands, -48 mV in wet meadow beaked sedge-dominated *Carex rostrata* stands, -165 mV in Nebraska sedge dominated stand *Carex nebraskensis*, and 570 mV in Kentucky bluegrass *Poa pratensis*-timothy *Phleum pratense* communities. Tall mannagrass appears to be well adapted for survival in a highly reduced environment having aerenchyma and hollow stems which probably allow for greater oxygen diffusion to the roots. Radial oxygen loss creates aerobic conditions in the environment immediately surrounding the plant roots (Teal and Kanwischer 1966; Armstrong 1970). This would result in oxidation of the reduced forms of iron, manganese, and organic soil toxins before they diffuse to plant tissues. Evidence of this can be seen through observation of a thin reddish coating (oxidized iron) on the root surface of many wetland species with roots in waterlogged soils.

Beaked sedge and Nebraska sedge appear to be adapted to lower redox potentials and have a thickened epidermis and, in the case of Nebraska sedge, a glaucous coating on the leaf blades. These are adaptations that decrease transpirational water loss by plants and, hence, decrease uptake of reduced soil toxins in the soil water. Kentucky bluegrass and timothy are found at the drier end of the gradient and do not appear to have any adaptations for growing in soils with lower redox potentials (waterlogged) for extended periods during the growing season. In summary, the distinctive patterning of vegetation communities in riparian zones is only one indication of ecosystem diversity. Each vegetation pattern indicates a unique combination of biogeochemical conditions which in turn influence the rhizosphere, vegetation, terrestrial wildlife communities, and the aquatic system.

Seasonal Fluxes in Redox Potential

Given differences in water table depth, redox potential, and vegetation composition, it is clear that biogeochemical cycles are unique and complex along the spatial gradient from emergent vegetation stands to dry meadows. Just as there are spatial gradients of redox potential, so are there temporal gradients of redox potential within any given plant community. In northeastern Oregon, we have found that redox potential is low at the onset of the growing season when soils are saturated (Table 3). Gradually, redox potential increased as the soil temperatures increased and soil moisture decreased. A depression in redox potential at the 10-cm depth occurred on both May 23 and June 6 sampling dates. This was due to heavy spring rains that resaturated the soil horizon.

Redox values at the onset of the growing season are well within the range for conversion of oxidized iron to reduced iron. Throughout the sampling period, reduced forms of manganese and iron would be expected to be present in the soil profile (Table 3). Redox potential indicates that denitrification processes are occurring well into early June.

This is a very stressful ecosystem in that plants survive in an ecosystem characterized by dramatic seasonal changes. Plants must begin growth in a highly reduced environment and cope with high, potentially toxic, concentrations of reduced elements in an anaerobic environment as well as cold soil temperatures. By the end of the growing season, soil environments are quite different. Soils are dry, well oxidized, and may be quite warm.

Nitrogen in Riparian Areas

Despite the fact that riparian zones differ from other wetlands by the presence of running water, it is likely that nitrogen cycling pathways described for wetlands are similar in the riparian zones of northeastern Oregon (Figure 1). Nitrogen has been found to be a limiting nutrient to vegetation productivity in salt marshes (Valiela and Teal 1974), freshwater tidal marshes (Simpson et al. 1978), and freshwater marshes (Klopatek 1978; Grace 1988). Degrees of nitrogen limitation have not been quantified in riparian zones of the intermountain western USA.

Biological fixation of atmospheric nitrogen (N_2) is only known to occur within a small group of free living blue-green algae, and bacteria, symbiotic bacteria, and actinomycetes (Sprent 1987). Nitrogen fixation has been quantified in salt marshes (Kaplan et al. 1979) and in freshwater lakes (Viner and Horne 1971). However, Golterman and

Figure 1.—Nitrogen cycling pathways in undisturbed riparian zones of northeastern Oregon, as indicated by redox potential (Eh).
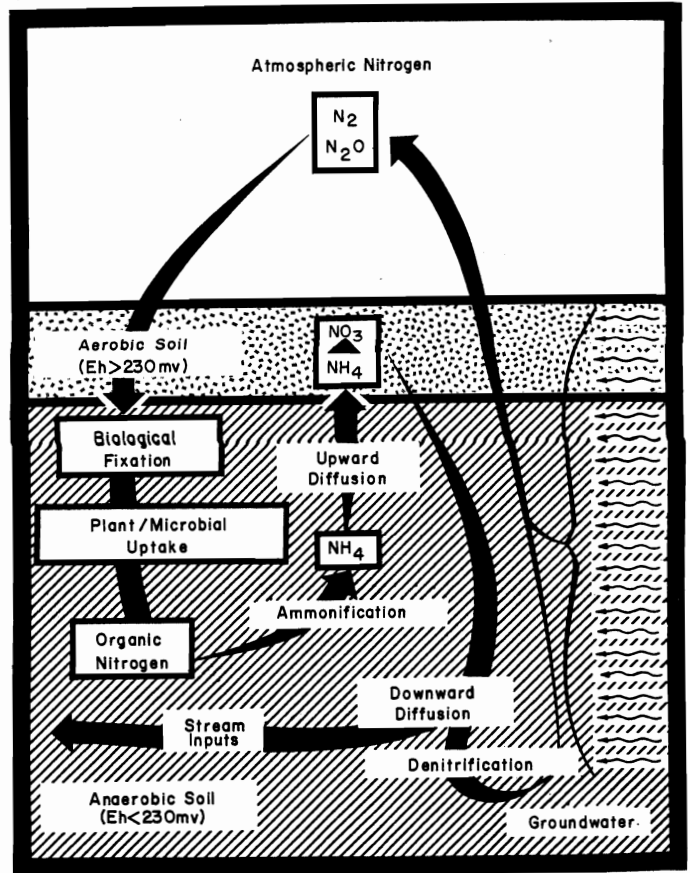
Kouwe (1980) feel it is unlikely that nitrogen fixation contributes significantly to the nitrogen cycle of freshwater lakes. Ammonium-nitrogen (NH_4-N), the product of nitrogen fixation, is immobilized into a variety of compounds in the biomass of nitrogen fixers or their hosts. Upon organic matter decomposition, nitrogen is released in this form. Under anaerobic conditions ammonium may become locally abundant and plants can take up nitrogen in this form (Bidwell 1979; Mitsch and Gosselink 1986). When anaerobic soil layers are present, ammonium can diffuse into an aerobic horizon where it is rapidly converted to nitrate-nitrogen (NO_3-N) by the bacterially mediated proc-

Table 3.—Mean (SE) redox potential and soil temperatures at two depths and water table depths during spring and summer in a *Carex-Poa pratensis*-mixed forb community along Catherine Creek, Oregon, 1988.

Date	Redox potential (mV) depth (cm)		Temperature °C depth (cm)		Watertable depth (cm)
	5	10	5	10	
April 14	-40 (9)	-24 (49)	10.4	10.4	11
April 28	-1 (32)	-30 (27)	10.5	10.2	11
May 12	114 (15)	118 (27)	12.6	11.2	12
May 23	218 (9)	-167 (8)	13.3	12.4	8
June 6	7 (48)	-119 (8)	11.7	11.2	8
June 20	311 (22)	295 (1)	15.4	14.7	21
July 6	374 (8)	386 (32)	13.6	12.7	35

ess of nitrification. Nitrate-nitrogen is very labile, and *because of its negative charge, is mobile in the soil profile* (Hausenbuiller 1978); the majority of nitrogen entering the riparian zone via ground water is in this form (Jacobs and Gilliam 1985).

If nitrate-nitrogen is not immobilized by organisms, it can be leached into groundwater or denitrified in anaerobic layers (Patrick and DeLaune 1972; Ponnamperuma 1972; Gambrell and Patrick 1978). In riparian zones, nitrate-nitrogen present in subsurface flows can be denitrified in those areas of the soil profile where redox potential is less than 230 mV (Table 1). Nitrate-nitrogen entering these anaerobic areas is converted into the gaseous forms of N, nitrous oxide (N_2O) or dinitrogen (N_2), the relative proportions of which are dependent on pH (Van Cleemput 1975). Nitrate-nitrogen that is not denitrified or biologically incorporated will be discharged to the stream.

Nitrate-nitrogen can greatly influence the water quality of streams and lakes where it is a limiting nutrient. Increased loading of nitrate-nitrogen in lakes or streams can cause algal blooms (Harned 1982). When algae that have accumulated during a bloom die and decay, the decomposition processes greatly depresses levels of dissolved oxygen in the water resulting in death of aerobic organisms or changes in species composition (Ewel 1976). Algal growth caused by elevated nitrate levels will also decrease the clarity in lakes and rivers thereby decreasing aesthetic value (Coats et al. 1976). Nitrate-nitrogen can also be toxic to humans; for example the World Health Organization recommends less than 11.3 mg N/L in drinking water.

Recent research has shown that riparian zones are important sites for denitrification. Rhodes et al. (1985) found that in an undisturbed watershed of the Sierra Nevada, California, over 99% of the incoming nitrate-nitrogen was denitrified. Vegetation uptake of nitrate-nitrogen was found to be insignificant in their study. In North Carolina, Jacobs and Gilliam (1985) found that 10 to 55 kg/hectare/yr of nitrate-nitrogen was denitrified in the riparian zone of streams bordering agricultural fields. Gambrel et al. (1975) and Gilliam et al. (1978) found significant losses of NO_3-N in the riparian zone of North Carolina streams. In Georgia, Lowrance et al. (1984) found nitrate-nitrogen concentration in subsurface water leaving agricultural fields decreased 66% as it passed through the riparian zone. It was concluded that the riparian forest also acts as a sink for calcium, magnesium, potassium sulfate, and phosphorus.

Phosphorus Riparian Areas

In numerous wetland ecosystems, phosphorus has been found to be a limiting nutrient. It was reported to be limiting in northern bogs (Heilman 1968), southern deepwater swamps (Mitsch et al. 1979), and freshwater marshes (Klopatek 1978). Phosphorus was not considered to be a limiting factor in areas of rice cultivation (Patrick and Mahapatra 1968; Gambrell and Patrick 1978), and salt marshes (Whitney et al. 1981; Grace 1988). Riparian areas have been reported to act as a sink for phosphorus (Cooper and Gilliam 1987), but no criteria has been established to ascertain whether phosphorus is a limiting nutrient in riparian zones.

Although not used in microbial respiration processes as is iron, manganese, or sulfur, phosphorus is affected by soil redox potential. Availability of phosphorus increases in anaerobic soil compared to aerobic (well drained) soils

(Shaprio 1958; Khalid et al. 1976). Patrick and Wyatt (1964) *found that extractable phosphorus increased by a factor of* three as redox potential was reduced from 200 to -200 mV. DeLaune et al. (1981) determined that the concentration of dissolved phosphorus was highest under low redox (-200 mV) and low pH (5.0). Ponnamperuma (1972) attributed the increase of soluble phosphorus to three factors: (1) hydrolysis of ferric and aluminum phosphates, (2) release of phosphorus adsorbed to clays and hydrous compounds by anion exchange processes, and (3) reduction of ferric iron compounds containing phosphorus to more soluble ferrous compounds. High levels of phosphorus can cause algal blooms with the same results as elevated nitrate concentrations discussed previously.

Riparian zones principally influence phosphorus concentration through their function as sites of deposition for phosphorus enriched clay particles or by absorption of phosphorus from overlying water by soil or sediments. Stream banks and sediments have been reported to absorb significant quantities of phosphorus (Brinson 1977; McCallister and Logan 1978; Yarbro 1979). Cooper and Gilliam (1987) found deposition of sediments to be more important than sediment absorption as a mechanism of phosphorus removal from streams in North Carolina. Phosphorus is also immobilized in the riparian zone by vegetation uptake. Yarbro (1979) estimated that annual growth increment of trees accounted for 0.6 to 1.2 kg phosphorus/hectare in a North Carolina floodplain forest.

Anthropogenic Effects on Biogeochemical Cycles

When redox potential is lower than 230 mV, denitrification will occur (Table 1). This process may be very important for the maintenance of aquatic ecosystems. When nitrate-nitrogen occurs at abnormally high concentrations, a reduction in stream water quality will take place as a result of algal blooms, turbidity, and, ultimately, oxygen depletion (Harned 1982). Because it is negatively charged, nitrate-nitrogen ions are very labile in soils and ground water (Hausenbuiller 1978).

Many undisturbed, montane, low gradient riparian areas of the western USA are characterized by high water tables throughout most or all of the year. In wet meadows, the anaerobic layers will be close to the soil surface throughout the year. In the meadows of Catherine Creek, we have found that organic matter contents are usually highest at the soil surface and decrease with soil depth. Therefore, optimal environments for denitrifying microorganisms exist and include a rich supply of organic matter as a source of energy and anaerobic conditions for much of the growing season. Hussey et al. (1985) found that significant numbers of denitrifying bacteria existed in the upper 5 to 15 cm of riparian soils. Rates of denitrification were lower at the 30-cm depth than at 5 cm. Soils close to the edge of the stream contained more organic matter, moisture, and higher rates of denitrification than other sites.

Conditions allowing for high rates of denitrification will disappear when riparian zones are destroyed as a result of erosional downcutting of the stream channel (Figure 2). Erosional downcutting increases the soil depth at which anaerobic conditions occur. It is probable that organic matter available for use as an energy source will be less at these deeper depths. Additionally, as the riparian zone decreases through channelization, drainage, removal of beaver dams, or downcutting, the total area in which denitrification can occur decreases. Hydraulic residence times would also be decreased. Hydraulic residence time in

Figure 2.—Nitrogen cycling pathways in degraded riparian zones of northeastern Oregon, as indicated by redox potential (Eh).
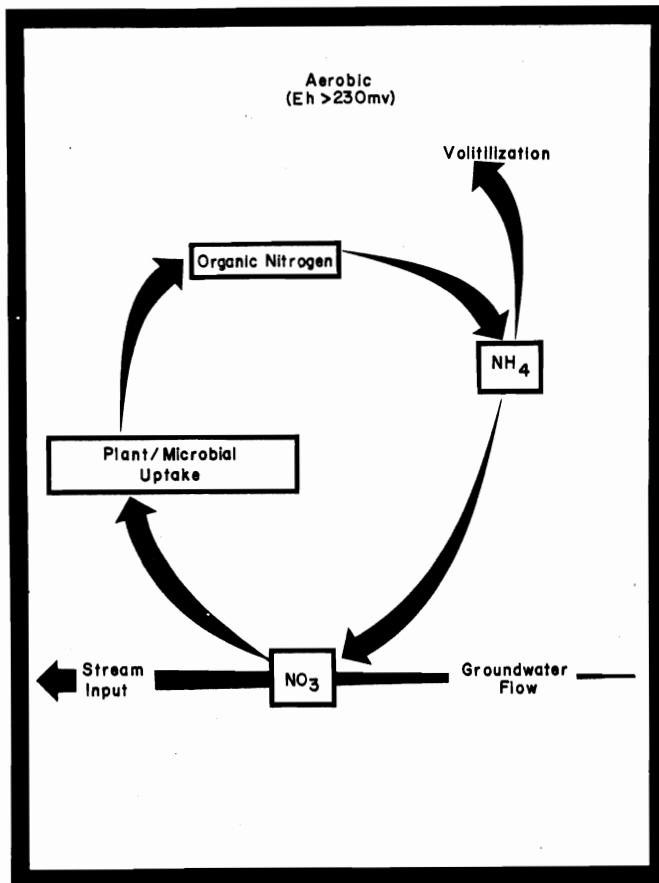
riparian ecosystems is that period in which ground water is present in the riparian zone as it moves from uplands to the stream channel. We hypothesize that decreasing hydraulic residence times through channelization or erosional downcutting, as a result of poor management practices, reduces the potential for biological interaction (i.e., nutrient uptake and chemical transformation) between riparian vegetation, microorganisms, and ground water. Alteration in both channelized and overgrazed riparian systems can be characterized as changes from an equilibrium of lateral inputs and outputs to an unidirectional movement and leaching into the aquatic system (Brinson et al. 1981). For example, channelized coastal streams in North Carolina were found to have higher concentrations of nitrate-nitrogen than in unchannelized streams (Kuenzler et al. 1977). The source sink relationship of nitrate-nitrogen in an intact riparian zone can be overridden during storm events when increased subsurface flows from precipitation decrease hydraulic residence time in the riparian zone (Coats et al. 1976; Melgin 1985).

Numerous examples exist in central Oregon and other regions of the semi-arid western USA where stream flow originating from intact riparian zones is perennial, but an abused or degraded section of the same stream is ephemeral (Elmore and Beschta 1987). This reflects a decrease in hydraulic residence time in the degraded section that results in decreases in total riparian area, lower rates of biological interaction with surface flows, lower quantities of nutrient uptake by vegetation, lower quantities of nutrients stored in the riparian ecosystem, and greater quantities exported out of the watershed.

The decline in surface biomass of vegetation in degraded riparian zones or exclusion of the stream from its riparian zone by channelization or erosional downcutting will result in reduced amounts of sediment trapping in the riparian zone. Many nutrients are also deposited in the riparian zone during high flows and are lost to downstream systems in degraded situations. Yarbro (1979) found that sediments in the riparian zone of a North Carolina stream trapped 1.7 kg/hectare of phosphorus. Cooper and Gilliam (1987) found that 50% of the phosphorus leaving upland agricultural fields in North Carolina passed by the riparian zone they studied.

Conclusion

The condition of the riparian zone is a reflection of many complex and biogeochemical processes with linkages to both aquatic and terrestrial communities. Biogeochemical processes greatly influence water quality, the aquatic ecosystem, and the pattern and productivity of the riparian vegetation. In our study, preliminary data show that Kentucky bluegrass-timothy communities which are not adapted to anaerobic conditions occupied areas of higher redox potential than tall mannagrass communities which are adapted to anaerobic conditions. Currently, little quantitative data exist on the effects of grazing, logging, channelization, recreation, or other anthropogenic activities on biogeochemical cycles in western riparian ecosystems. We do know that isolating the stream from its riparian zone, either by channel downcutting or by channelization, seriously impairs the functional integrity of the riparian zone. Source-sink relationships between riparian and stream ecosystems are severed. This ultimately will result in declines in those resource values associated with water quality, livestock grazing, fisheries, and wildlife diversity and productivity. Research should focus on quantifying the landscape-level relationships between the terrestrial, riparian, and aquatic ecosystems.

Developing land management systems that allow ecosystem processes to function as close to a natural equilibrium as possible will insure a stable output of resource values for all interested users. Without an increased understanding of how riparian ecosystems function and the development of management strategies that reflect this knowledge, we can expect further declines in the quality and productivity of our riparian ecosystems.

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