chapter five

Restoration effects on N cycling pools and processes

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5.1 Introduction

Over the past several years, there has been an acceleration of restoration efforts to mitigate the consequences (i.e., ground and surface water chemical pollutants, erosion, etc.) of degraded ecosystems and enhance structural and functional components of watershed ecosystems that regulate biogeochemical cycling and associated aquatic components. Biogeochemical processes are complex because they operate at a variety of spatial and temporal scales (e.g., near instantaneous soil chemical reactions vs. bedrock weathering). Large-scale assessments of the integrity of these fine-scale processes would be a daunting

task; however, our knowledge of ecosystem processes at larger scales (e.g., stands, watersheds, basins) suggests emergent (and measurable) properties that integrate these finer-scale processes. For example, nutrient budgets are key indicators of watershed health and have proven useful for evaluating response and recovery to a variety of disturbances and management activities (Swank and Vose 1997; Swank et al. 2001). Since water moves nutrients through the system, hydrologic, aquatic, and biogeochemical processes are tightly linked. Nutrient budgets are comprised of inputs (e.g., atmospheric deposition, fixation), internal pools and cycling (e.g., soil, litter, and vegetation pools and cycling rates), and outputs (e.g., stream chemistry, seepage or groundwater chemistry, denitrification) (Figure 5.1). The nutrient cycling components shown in Figure 5.1 serve as a focal point for identifying ecosystem nutrient pools or processes requiring restoration to enhance biogeochemical processes and aquatic systems. However, pool sizes and cycling rates can vary considerably among ecosystems (Table 5.1). This variation makes general assessments of restoration effects on biogeochemical cycling difficult because similar types of ecosystem degradation (e.g., insect-related mortality) may not have similar impacts on
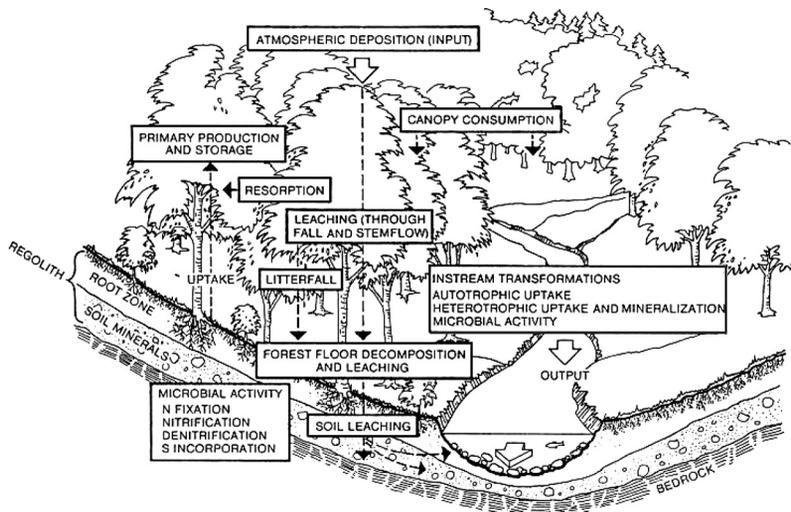

Figure 5.1 Components of ecosystem biogeochemical cycling in a watershed context.

Table 5.1 Comparison of Pool Sizes and Cycling Rates Among Three Contrasting Forest Ecosystem Types in North America (adapted from Swank 1986)

	Oak/Hickory (North Carolina)			Northern Hardwood (New Hampshire)			Western Conifer (Oregon)		
	N	Ca	K	N	Ca	K	N	Ca	K
Pool Size (kg ha^{-1})									
Vegetation	995	830	400	532	484	218	560	750	360
Forest Floor	140	130	20	1,256	372	66	740	570	90
Mineral Soil	6,800	2,500	124,000	4,890	9,600	—	4,500	—	—
Cycling Rate (kg ha^{-1} year^{-1})									
Litterfall	33	44	18	54	41	18	21	41	9
Woody Increment	13	23	13	9	8	6	−2	−4	−1

nutrient pools and processes among ecosystems. Similarly, this variation also implies that restoration treatments applied in one ecosystem may not have similar results in another. Despite these complexities, it is important to develop and apply restoration practices that restore biogeochemical cycling processes in degraded ecosystems, and to develop methods, metrics, and approaches for assessing the recovery of biogeochemical cycling pools and processes among contrasting ecosystem types. This chapter focuses primarily on developing metrics and approaches for assessing the recovery of biogeochemical cycling pools and processes.

The degree of biological vs. physiochemical regulation of pool sizes and flux rates is element dependent. For example, nitrogen (N) cycling is especially dependent on microbially driven biological processes such as mineralization, nitrification, and denitrification. Most of the stored N pool is contained in soil organic matter and vegetation, and inputs are almost exclusively driven by atmospheric contributions. In contrast, calcium (Ca) cycling is especially dependent on soil and bedrock weathering for inputs and often most ecosystem Ca is stored in the mineral soil. These differences are especially relevant when evaluating restoration alternatives and evaluating the rate and magnitude of potential responses.
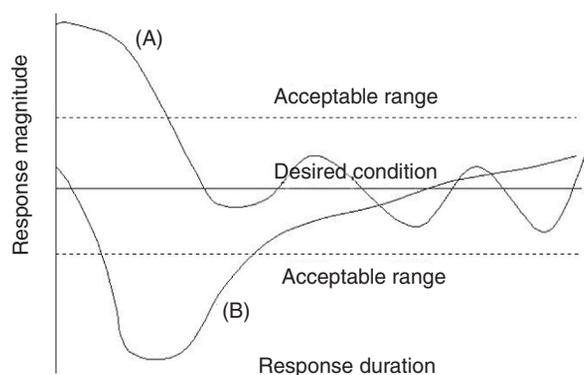
The complexity and interdependency of nutrient cycles require focusing on "key indicators" that are responsive to restoration activities. In some cases, the choice of the "key indicator" is obvious, especially if the overall goal of restoration is to alter the pool size or cycling rate of a particular element or combination of elements. For example, examining restoration success to offset or mitigate the effects of elevated sulfur (S) deposition (e.g., liming) would focus primarily on the S cycle, with additional measurements of base cations and acidity as codependent variables. In contrast, restoration goals may not be directly focused on biogeochemical cycling *per se*. In this case, examining nutrient cycling processes may provide an integrated estimate of ecosystem-level responses or indicate undesirable (or desirable) consequences to other "nontargeted" ecosystem components. For example, prescribed burning has been advanced as a restoration tool to improve the structure and diversity of forest ecosystems. Because fire alters nutrient pools and cycling rates, many of the functional aspects of biogeochemical cycling are either directly or indirectly affected by restoration burning. The degree of departure (in terms of both magnitude and duration) of biogeochemical pools or processes from baseline or desired conditions may serve as a tool for evaluating restoration success or comparing restoration alternatives. Because biogeochemical cycling processes are complex and involve multiple elements, several factors need to be considered when choosing an appropriate metric(s) for evaluating biogeochemical responses to restoration. First, the indicator (or response variable) needs to be responsive to the restoration activity. For example, if restoration involves manipulation of forest floor and/or surface soils, then pools (e.g., forest floor biomass and nutrient content, available nutrients, soil solution chemistry) or processes (e.g., litter decomposition, mineralization, microbial activity) tightly linked with those ecosystem components should be monitored. Second, due to the inherent variability in biogeochemical cycling processes at the watershed scale, response variables need to be selected such that their response magnitude exceeds the natural variability of the system. For example, seasonal and interannual variation in stream chemistry can result in nearly twofold differences in NO_3^-, Ca, and K concentrations (Swank et al. 2001). Restoration practices that cause only minor changes in pools or processes may not result in measurable responses (+ or −) in terrestrial or aquatic biogeochemical cycling. For example, in ecosystems where baseline stream chemistry is inherently high and variable, integrated responses reflected by changes in stream nutrient concentration may be too subtle to detect (Likens et al. 1970).

Data from long-term watershed studies provide some insight into elements that meet the criteria outlined above (Swank and Vose 1997). Analyses of nutrient budgets suggest that terrestrial ecosystems are inherently conservative (i.e., outputs< inputs) for nutrients that are in limiting supply and whose availability is largely controlled by biological processes (Swank 1986). For example, when examining N budgets across a range of ecosystem types in North America, Henderson et al. (1978) found comparable rates of nitrogen retention, despite large differences in N inputs. The most significant pathway for N loss from terrestrial ecosystems is solution loss in groundwater or streamwater. Undisturbed ecosystems that receive low amounts of N inputs generally lose very little inorganic N via stream or groundwater export. This high degree of N conservation is indicative of tightly linked N pools and cycling processes such as storage, uptake, immobilization/mobilization, N fixation, and microbial processes. When some (or many) of these pools and processes are altered due to disturbance or restoration, measurable changes in N pools, processes, and outputs are likely. For example, in the southern Appalachian mountains of Western North Carolina, stream NO_3 is a strong indicator of ecosystem response to disturbances such as insect defoliation (Swank 1981), clearcutting (Swank et al. 2001), or hardwood forest conversion to grass cover or pines (Swank and Vose 1997).

In this chapter, we present a conceptual framework for evaluating biogeochemical responses to restoration and synthesize data from several restoration studies to illustrate this conceptual framework. Our chapter focuses specifically on N cycling pools and processes for two reasons. First, as explained above, N pools and processes are sensitive to disturbance (and restoration) and may serve as "key indicators" of ecosystem responses to restoration. Second, restoring N cycling pools and processes is a primary goal of many restoration efforts because N commonly limits terrestrial ecosystem productivity, and N (e.g., NO_3^-) losses to groundwater and surface water can have important consequences for human health and for the ecological integrity of aquatic ecosystems. The studies contrast the potential uses of resistance/resilience analogy. In the first set of studies, the primary goal of restoration was to restore N cycling processes in degraded riparian zone ecosystems. In the others, the primary goal was to alter vegetation structure and composition using a variety of prescribed burning techniques. Here, we use the resistance/resilience analogy to evaluate restoration impacts on other "nontarget" ecosystem components.

5.2 Conceptual framework for evaluating ecosystem responses to restoration

The resistance and resilience model of ecosystem response to disturbance (Holling 1973) is a useful conceptual framework for evaluating the restoration success of enhancing biogeochemical cycles and aquatic systems (Waide 1988; Carpenter et al. 2001). In the original development of resistance/resilience theory, resilience was defined as the amount of change a system can undergo before moving into a different state space and a different set of controls (Holling 1973). Resistance has been defined as the amount of external pressure needed to bring about a given amount of disturbance in the system (Carpenter et al. 2001). In further revisions of resistance/resilience theory, organic matter pool sizes and turnover rates were key indices of disturbance responses (Webster et al. 1975; Waide 1988), and it has been further modified to evaluate ecosystem responses to restoration burning (Vose 2000). The resistance–resilience model can be used as a conceptual construct for evaluating spatial and temporal aspects of current ecosystem condition, desired ecosystem condition, and ecosystem response to restoration (Figure 5.2). Here, we define the *y*-axis as the response magnitude (i.e., percent change from initial condition) and the *x*-axis as the response duration. The line labeled (a) represents an ecosystem where the initial condition
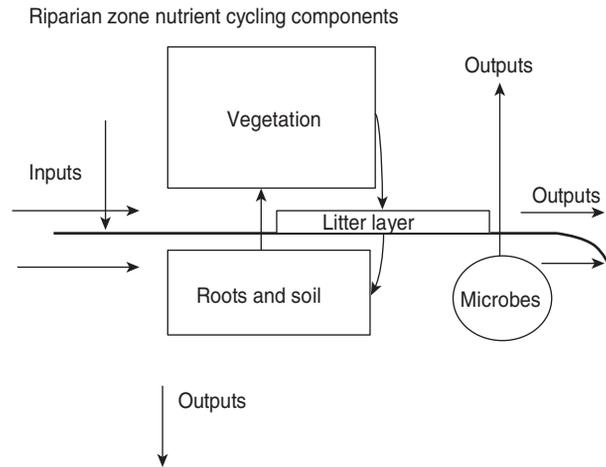
Figure 5.2 Conceptual model of approach to evaluating ecosystem responses to restoration. The approach focuses on determining the magnitude (*y*-axis) and duration of response (*x*-axis) with respect to desired conditions and acceptable ranges. Line (a) represents the hypothetical response of a parameter outside the range of acceptable conditions and the response to restoration. Line (b) represents the hypothetical response of a nontargeted parameter that may respond negatively to the restoration of other ecosystem components.

of the pool or process is outside the boundary of acceptable conditions and restoration treatments are used to change the pool/process into an acceptable range. The line labeled (b) represents potential responses of nontarget ecosystem pools/processes. Here, the initial condition may be within acceptable limits; however, restoring other ecosystem components may directly (or indirectly) alter nontargeted ecosystem components. In application, the historical and contemporary disturbance legacy of an ecosystem (or watershed) determines its degree of departure from desired conditions, influences the rate of attainment of desired ecosystem conditions or response duration, and influences the magnitude of response to the restoration treatment. We emphasize that the desired conditions (i.e., processes or pools) are defined as existing within a boundary of acceptable conditions reflective of the range of natural variability or some other (e.g., land manager defined) acceptable range of variability. This point is especially important in evaluating biogeochemical responses to restoration because nutrient cycling processes are not constant, but are more likely to persist in a dynamic equilibrium. In contrast, severely degraded ecosystems are characterized as having initial pools and processes that are farthest from desired conditions, consistently remain outside the boundary of acceptable conditions, require substantial restoration efforts to move toward desired conditions, and take the longest time to attain desired conditions.

5.3   Restoration of N cycling processes in degraded riparian zones

Riparian zones are the key linkage between terrestrial and aquatic ecosystems. While several aspects of riparian zone structure and function remain unknown, the importance of riparian zones for reducing nutrient inputs to aquatic systems is well documented (Lowrance et al. 1984; Gilliam et al. 1997). For example, several studies have demonstrated substantial (e.g., 60 to 100% of inputs) reductions in N concentration in subsurface waters by both grass and forest riparian zones (Daniels and Gilliam 1996; Cooper et al. 1987; Osborne and Kovacic 1993). Riparian zone nutrient cycling processes that reduce nutrient fluxes to aquatic systems include physical entrapment of organic matter and sediment, uptake by riparian vegetation and microbes, and microbial processes such as denitrification (Figure 5.3). Factors that influence nutrient uptake efficiency include the type and

Riparian zone nutrient cycling components

Figure 5.3 Key nutrient cycling pools and processes in riparian zones; inputs come from uplands and outputs are to the stream.

density of riparian vegetation, soil surface conditions (i.e., the presence or absence of a litter layer; the presence or absence of compacted soil), and soil physical characteristics (e.g., infiltration rates) (Osborne and Kovacic 1993).

In many ecosystems, the structure and function of riparian zone ecosystems have been severely degraded. The most dramatic changes occur where the natural landscape has been converted to alternative uses, such as urban and suburban development, livestock grazing, hayfields, or row crop agriculture. More subtle changes include altered disturbance regimes (e.g., changes in flooding regimes, timber harvesting), changes in species composition (e.g., introduction of exotic species, loss of native species), or lower-intensity disturbances such as concentrated recreation use or prescribed fire. Natural resources managers, farmers, and local citizen-based groups are becomingly increasingly aware of the importance of riparian zones for protecting water quality. In the U.S., many regions have set goals (i.e., linear stream miles of riparian zones to be restored) for the restoration of degraded riparian zones and long-term protection. In this example, we quantify the short-term recovery of N cycling processes in restored riparian zones in agricultural (pasture and hayfields) watersheds and use the resistance/resilience analogy to evaluate responses.

5.3.1   Study sites

Two study sites were selected for the study of N cycling recovery following restoration. The Killian Farm (KF) site was located on Cartoogechaye Creek, a tributary of the Little Tennessee River in Macon County, North Carolina. Livestock grazing for several years heavily impacted the site and cattle were allowed creek access along the entire restoration treatment area (approximately 400 m). Unstable streambanks, heavily compacted areas, trails leading into and within the riparian area, and sparse vegetative cover characterized the site. A 100 m section upstream from (but adjacent to) the degraded area served as the control. The control area was fenced and had a lush understory and fully developed overstory along the riparian zone. The topography of area adjacent to the creek was gently sloping (~5%) toward the riparian area for approximately 30 m, and steeply sloping (20%) thereafter.

The Rabun Farm (RF) site was located on Sutton Branch, a small tributary of Betty's Creek in Rabun County, Georgia. The site was heavily impacted by cattle, resulting in significant compaction and bare areas in the riparian zone over about 75% of the stream (approximately 300 m) reach. The other 25% (approximately 100 m) had a thick vegetative

cover of alder and rose and had no evidence of current or historical cattle use. Since this area was unused by cattle, it served as a control. The streambank was generally stable, with only a few areas showing signs of excessive erosion. The topography of area adjacent to the branch was gently sloping (~5%) toward the riparian area for approximately 20 m, and steeply sloping (25%) thereafter.

5.3.2 Restoration treatments

Restoration treatments were initiated in late fall 1997 at the KF site, and early spring 1998 at the RF site. Cattle were excluded from the area during restoration treatments and fencing was installed to permanently exclude cattle from the riparian zone. Since cattle were active at both sites, the distance from the stream edge to the fence defined the buffer width. At the KF site, the distance from the stream edge to the fence line ranged from 7 to 28 m, and averaged 11 m. At the RF site, the distance from the stream edge to the fence line ranged from 2 to 9 m, and averaged 6 m. Tree seedlings of a variety of riparian zones species and species with potential wildlife benefits (e.g., *Cornus florida*, *Salix* spp., and *Platanus occidentalis*) were planted (~10 m spacing) at all sites. Riparian areas outside the fenced area continued to be used by livestock and served as unrestored sample areas for some parameters.

To illustrate the effects of riparian zone restoration on N cycling pools and processes, comparisons were made using several approaches. Comparisons were made between restored areas and nearby controls, between restored areas and undisturbed forested riparian zones, and between restored and unrestored areas.

5.3.3 Overland flow sampling

Total suspended solids (TSS) entering the streams in overland flow were sampled on a storm-event basis. TSS included the combined contributions of sediment and organic matter. We did not measure the N content of the overland flow samples; however, both sediment and organic matter contain N and we can infer that changes in N amount in overland flow are proportional to changes in TSS. Collectors were installed in locations where overland flow was obvious within 1 week after restoration treatments were completed. Overland flow collectors were installed at five locations in the degraded riparian zones at the KF and two locations in the controls. Four collectors were installed in degraded areas at the RF site and two in the controls. Collectors were checked after all storms, but typically only collected overland flow when storms exceeded 1.25 cm during the first month following restoration treatments, and 5.0 cm thereafter. When overland flow was present in the carboys, they were removed from the overland flow collector and vigorously agitated to resuspend solids, and then quickly subsampled with a 1,000 ml nalgene bottle.

5.3.4 Soil solution sampling

Porous cup lysimeters (30 cm depth) were used to sample soil solution NO_3^- concentration. Lysimeters were installed at 3-m spacing from the stream edge to just inside the unrestored area along transects (nine transects at RF, eight transects at KF) equally spaced throughout the riparian zone. After an equilibration period of 3 to 4 weeks, lysimeters were sampled weekly for approximately 2 years at each site. Samples were composited and analyzed monthly.

5.3.5 Nitrogen trace gas sampling

Trace gas fluxes were determined using a flow-through chamber mass balance technique. Details of the sampling and analytical procedures are presented in Walker et al. (2002); hence, only a summary is presented here. Nitric oxide (NO) and NH_3 fluxes were
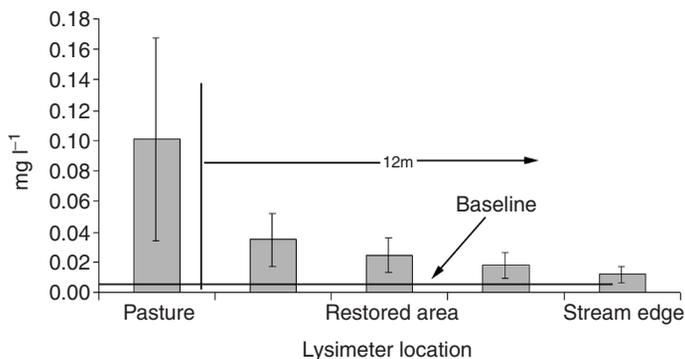
measured approximately every 3 months between July 1998 and August 2000, with sampling intervals roughly corresponding to seasonal midpoints. Nitrous oxide (N_2O) measurements began in June 1999. Fluxes were measured in two locations within both the restored and unrestored riparian zone during each measurement period. Chambers were placed in the same general area from season to season. Reported fluxes are for daytime values only. Nighttime values were generally depressed, largely due to the exponential temperature dependence of emissions.

5.3.6   *Soils and vegetation characterization*

Bulk density (g cm^{-3}), infiltration rates (cm h^{-1}), and plant biomass (g m^{-2}) were measured at both the KF and RF sites. Bulk density was measured at 20 random locations within the restored riparian zone ($n = 20$ at KF and $n = 20$ at RF) and ten random locations in the controls. Bulk density samples were taken during the winter (soils were not frozen) in 1997, 1998, and 1999 at KF and in 1998 and 1999 at RF. Infiltration rates were measured with an infiltrometer at random locations within the restored riparian zone ($n = 16$ at KF and $n = 20$ at RF) and five random locations in the controls. The peak standing plant biomass (midsummer) was measured on 20 randomly located 0.84 m^{-2} clip plots within the restored riparian zone and 20 locations in the degraded riparian zone. Both woody and herbaceous vegetation were sampled. Clipped vegetation was dried at 70°C to a constant weight. Samples were taken in 1998 and 1999 at KF and in 1999 and 2000 at RF.

5.3.7   *Nitrogen cycling responses to riparian zone restoration*

Several components of the riparian N cycle were improved by restoration. The "initial condition" of soil solution NO_3^- in the degraded riparian zone was approximately tenfold higher than the baseline value of 0.01 mg l^{-1} (i.e., typical for soil solution NO_3^- concentrations in undisturbed riparian zones in the southern Appalachians) (Yeakley et al. 2003). Restoration resulted in a considerable reduction in soil solution NO_3^-, with values in near-stream areas comparable to baseline levels (Figure 5.4). We attribute this decrease in soil solution NO_3^- to a combination of reduced inputs and increased storage in the recovering vegetation, litter layer, and recovering microbial populations and transformation rates. For example, vegetation biomass increased two- to fourfold on the restored riparian area compared to the degraded riparian zone and was comparable to baseline levels from control areas (Table 5.2). Increased vegetation biomass, in combination with changes in bulk density and infiltration, also appears to have played a role in decreasing the movement of
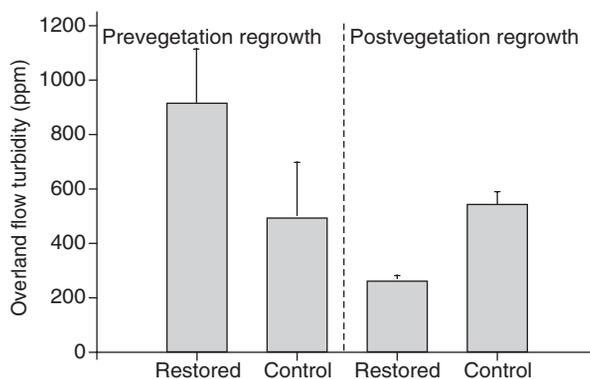

Figure 5.4   Soil solution NO_3^- concentrations (30 cm soil depth) along a transect from stream edge to unrestored pasture two years after restoration.

Table 5.2 Mean Vegetation Biomass (g m^{-2}) 2 Years after Restoration Compared to a Nearby Degraded Riparian Zone

Site	Restored (g m^{-2})	Degraded (g m^{-2})
RF	394 (133)	154 (95)
KF	399 (135)	91 (51)

Data in parentheses are standard errors ($n = 20$).

Figure 5.5 Effectiveness of vegetation regrowth in restored riparian zones for reducing overland flow turbidity.

nutrients (and sediment) through the riparian zone in overland flow (Figure 5.5). We observed a general decline in the amount of TSS in overland flow over time (Figure 5.6); however, since baseline values for TSS in overland flow in undisturbed riparian zones in the southern Appalachians are extremely low due to high infiltration rates (e.g., 25 cm h^{-1}) (Swift 1986), restoration treatments have not resulted in a full recovery of soil surface/vegetation conditions that promote rapid infiltration and TSS trapping. For example, the infiltration rate on our study sites (mean of the KF and RF sites) was only 8.5 cm h^{-1} 1 year after restoration, well below the values observed in undisturbed ecosystems. It should be noted that the size and severity of disturbed upland area, as well as the width of the riparian zone, have a significant influence on the capacity of the riparian zone to reduce TSS. Hence, small "fully restored" riparian zones with large, highly disturbed upland areas may still not have the functional capacity to trap and store all of the nutrients or sediment in overland flow. In our study, restoration did result in values comparable to the control (Figure 5.5), indicating that recovery was achieved within the constraints of the upland-riparian complex. In general, microbial processes regulating N trace gas emissions also increased after restoration (Walker et al. 2002). In this case, the desired condition is high levels of trace gas N flux to the atmosphere to prevent transport of subsurface N into surface water or groundwater. This contrasts with previous desired conditions, where we have used baseline values (which are typically quite low) (Davidson and Swank 1986) from undisturbed riparian zones. Here, NO fluxes were five- to sixfold greater in the restored riparian zone the first year following restoration (Figure 5.7), and available soil N was more efficiently converted to N trace gases (i.e., flux per unit soil [N]) in the restored riparian zone relative to the degraded areas (Walker et al. 2002). This pattern is likely due to the recovery of microbial communities following changes in soil physical characteristics (i.e., decreased bulk density [−5% from pre-restored conditions] and increased infiltration [+73% from pre-restored conditions]) that promote aeration.
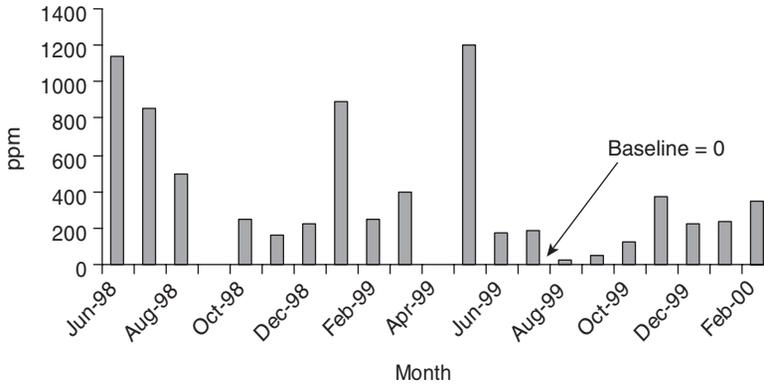
Figure 5.6 Temporal patterns in overland flow turbidity (TSS in ppm) immediately after restoration (June 1998) and 1.5 years later (February 2000).
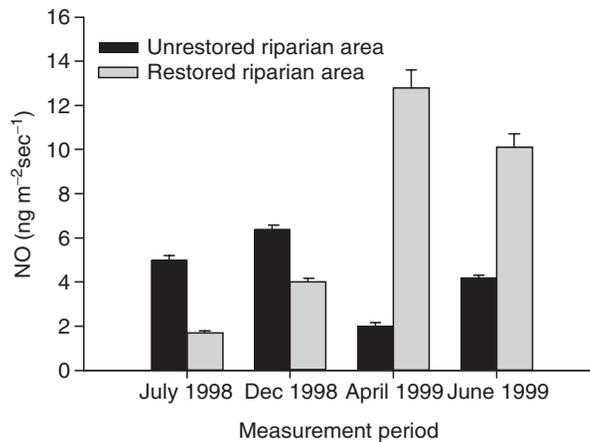

Figure 5.7 Temporal patterns in NO flux immediately after riparian zone restoration (July 1988) and 1 year later.

Evaluation of restoration of N cycling processes using the resistance/resilience analogy suggests a rapid and efficient recovery of many key N cycling pools, processes, and resultant fluxes (Figure 5.8). For example, soil solution NO_3^-, which was elevated tenfold as a result of riparian zone degradation and increased inputs from animal wastes, decreased to near-baseline conditions by the second year of recovery. Nitrogen trace gas fluxes and microbial N processing efficiency increased substantially in the first year of recovery, suggesting a rapid recovery of microbial populations and activity to desired conditions (i.e., significant and efficient conversion of subsurface N to gaseous forms). Both soil solution NO_3^- and N trace gas fluxes exhibited considerable resilience and low resistance (i.e., the return to desired conditions occurred rapidly and with minimal restoration intensity). In contrast, TSS (and the N it contains) in overland flow did not achieve desired conditions during the period of study. In part, this may be related to an incomplete recovery of infiltration rates and vegetation regrowth. This pattern of responses indicates that physical changes in soil surface characteristics may be more resistant and less resilient than N cycling-related biological responses. However, because the processing and storage capacity for TSS in overland flow is related to riparian zone width and the amount of material transported from upland areas in TSS (which is related to size, amount of
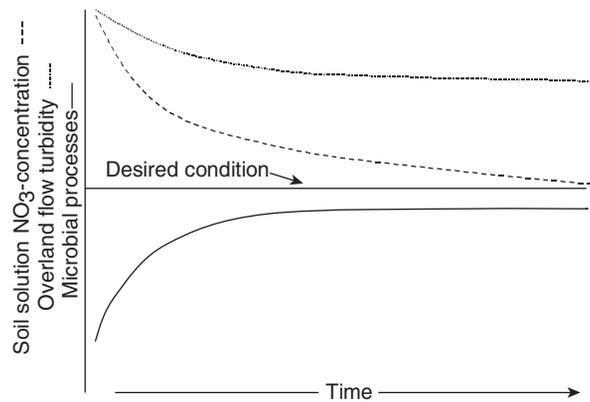
Figure 5.8 General patterns of changes in nutrient cycling pools and processes 1 to 1.5 years following riparian zone restoration.
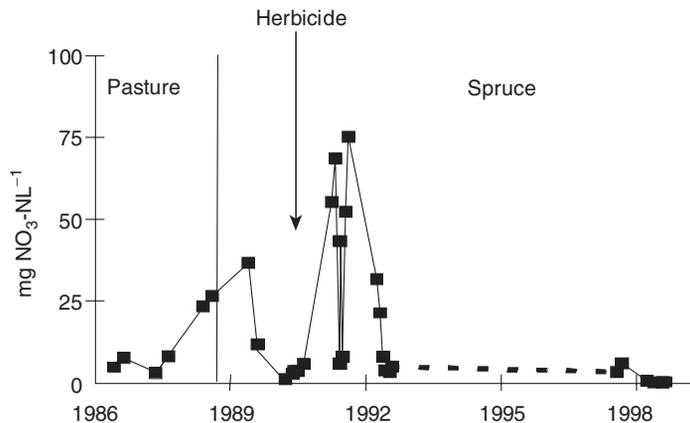

Figure 5.9 Effects of afforestation and herbicide treatment on groundwater N.

disturbance, and nutrient inputs from external sources such as animal waste and fertilizer), recovery in these restored riparian zones may never be sufficient to achieve conditions comparable to undisturbed forest ecosystems.

Results from a 2-year study provide only a snapshot of potential longer-term responses and very little information concerning year-to-year variability. For example, while results from the riparian zone restoration study suggest rapid recovery, we can make few conclusions about how ecosystem pools and processes will respond to factors such as extremes in abiotic variables, disturbance, or successional changes as the soils and vegetation continue to develop. As an example, results from longer-term afforestation studies in Denmark suggest that post-restoration vegetation dynamics are very important in regulating NO_3^- - N concentration in seepage water (Figure 5.9) (Callesen et al. 1999). Prior to afforestation, groundwater NO_3^- - N ranged from 10 to 25 mg l^{-1}, and then decreased substantially in the first few years after afforestation with Norway spruce (*Picea abies*). Herbicide treatment to release the spruce resulted in a significant spike (> 75 mg l^{-1}) in seepage water NO_3^- -N, followed by a reduction to levels around 5 mg l^{-1} in the fourth year and <1 mg l^{-1} by the tenth year.

5.4   Effects of stand restoration burning on stream nitrogen

The onset of fire suppression early in the 20th century significantly reduced the role of fire in shaping the structure and function of ecosystems in the southern Appalachians. A century of fire exclusion has resulted in a buildup of woody and fine fuels, increasing the risk of catastrophic wildfires, as well as altering ecosystems and wildlife habitat historically dependent on periodic wildfire for maintenance (Van Lear and Waldrop 1989). Prescribed fire has been used as a tool in many forest ecosystems to enhance overall stand health and productivity and reduce fuel loads (Sanders and Van Lear 1987; Van Lear and Waldrop 1989). While often satisfying the primary goal of reducing competition to commercially desirable tree species, prescribed burning also improves habitat for both avian and terrestrial wildlife (Cooper 1971). The continued (and, perhaps, expanded) use of fire in the southern Appalachians has generated considerable interest about the effects of prescribed fire on ecosystem integrity, particularly how it influences losses of key plant nutrients such as nitrogen. For example, total ecosystem nitrogen may decrease in forested systems following fire (Neary and Currier 1982; Rapp 1990) due to volatilization of nitrogen stored in coarse and fine fuels, and/or increased leaching of released NO_3^- from the system (Knoepp and Swank 1993). While the objective of the restoration burning did not focus on biogeochemical cycling, we propose that the magnitude (e.g., resistance) and duration (e.g., resilience) of NO_3^- response in stream water may be a useful metric for assessing ecosystem response to restoration burning. Here, we summarize and compare stream NO_3-N responses from four southern Appalachian watersheds receiving different burn prescriptions and use the resistance/resilience analogy to evaluate ecosystem responses.

5.4.1   Study sites

All four burned areas are on the Nantahala National Forest in western North Carolina, U.S. Three of the four sites are characterized as the xeric oak-pine forest type, in a generally degraded condition due to the combined effects of land management practices (e.g., high-grade logging), fire exclusion, and drought. Stands typically consisted of mixtures of pitch pine (*Pinus rigida*), Virginia pine (*Pinus virginiana*), shortleaf pine (*Pinus echinata*), scarlet oak (*Quercus coccinea*), chestnut oak (*Q. prinus*), and red maple (*Acer rubrum*), and have dense understories of the ericaceous shrub, mountain laurel (*Kalmia latifolia*). Much of the yellow pine overstory on these sites was dead because of recent drought-related southern pine beetle (*Dendroctonus frontalis*) infestations (Smith 1991).

The fourth site is within the Joyce Kilmer-Slickrock Wilderness Area and Memorial Forest, located approximately 60 km west of the other three sites. The area has an old-growth designation, and is one of the few remaining large uncut forested tracts in the southern Appalachians. This high-elevation site contains large areas of both mesic and xeric forest types.

5.4.2   Burning methods and sample collection and analysis

Due to the degraded condition of many xeric oak-pine stands in the southern Appalachians, the fell-and-burn site preparation prescription has commonly been used to increase commercial viability and enhance other attributes of ecosystem health (Vose et al. 1999). In 1990, a 5.25-ha watershed, designated Jacobs Branch (JB), was selected to examine ecosystem-level effects of the fell-and-burn treatment. Initial and long-term results have been published previously (Clinton et al. 1993; Elliott and Vose 1993; Knoepp and Swank 1993; Swift et al. 1993; Vose and Swank 1993; Elliott and Vose 1995; Clinton et al. 1996; Clinton and Vose 2000), and a complete burning characterization has been described by Swift et al. (1993). All woody vegetation on JB was felled in the summer of 1990. The downed material was allowed to cure for approximately 10 weeks before a headfire was

set at the base of the slope. The entire slope burned in less than 30 minutes. The fire was characterized as a high-intensity, low-severity fire that consumed most of the litter layer and a small portion of the humus layer (Vose and Swank 1993). Stream samples were collected weekly from a small stream at the base of the burned watershed beginning 6 months prior to burning and continued for more than 12 months after burning. Weekly samples were composited to obtain a monthly mean.

The Wine Spring Creek (WSC) burn was implemented in April 1995 to simulate a stand-replacement wildfire covering approximately 82 ha. The fire prescription was designed to examine the effects of wildfire on responses of vegetation, soil processes, and stream chemistry and turbidity (Major 1996; Elliott et al. 1999; Vose et al. 1999). The site was ignited by a helicopter and helitorch using strip headfires along the contour beginning at the lower and mid-slope positions. Crown fires were frequent at mid- and upper-slope positions due to the combination of heavy fuels, steep slopes, thick mountain laurel (*Kalmia latifolia*) understories, and firing techniques. Streams and roads provided natural firebreaks, and a backing fire was set along the top of the ridge. Automated time-increment proportional stream samplers (American Sigma, Inc.) were installed in 1993 at the base of the burned slope and upstream of the burned area, as well as in several streams in other unburned areas to be used as references, for the purpose of characterizing dissolved inorganic chemistry. In addition, grab samples (e.g., 250 ml collections at designated locations) were collected weekly at each stream sampler location.

The Joyce Kilmer-Slickrock Wilderness Area (JK) wildfire occurred in November 1999 and burned approximately 970 ha or about two thirds of the Little Santeetlah Creek Watershed within the JK site. The fire was an understory burn that traveled well into riparian zones of many small streams draining the watershed. During the time of the burn, stream samples were being collected for an unrelated study of stream water chemistry. On the JK site, weekly grab samples were taken beginning in August 1999 and continued after the burn for 6 months.

The Hickory Branch (HB) site was prescribed-burned (stand restoration burning) in March 1999. HB site vegetation was predominantly dry-site mixed-pine hardwood in the overstory and mountain laurel in the understory. Using a helitorch, the site was strip-burned along the contour beginning at the ridge and continuing to the base of the watershed. Crown fires frequently occurred on steep, south-facing slopes containing standing dead yellow pine snags and thick mountain laurel understory. Approximately 75% of the watershed burned, with little or no burning occurring in the more mesic areas around riparian zones. Stream water grab samples were taken weekly, beginning approximately 12 months before the burn at the watershed outlet on Pine Branch (hereinafter referred to as Lower Pine Branch) and at two locations upstream immediately above the confluence of Pine Branch (Upper Pine Branch) and Matheson Branch. Approximately 2 weeks before the burn, an automated flow-increment proportional sampler (American Sigma, Loveland, CO, U.S.A.) was installed at the watershed outlet (Lower Pine Branch), which collected samples for 2 months after the burn. Grab samples continued to be collected at all three collection sites.
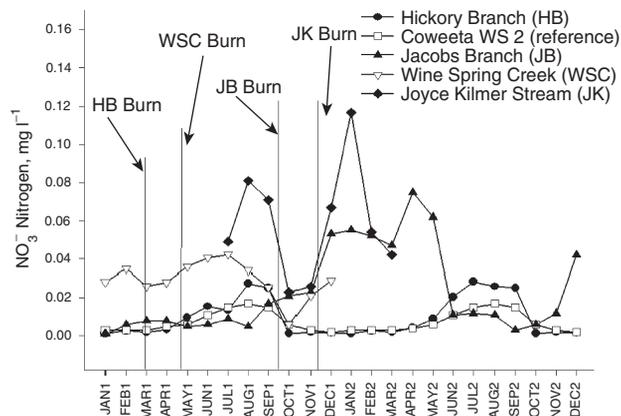
All stream water samples were analyzed at the Coweeta Hydrologic Lab using established protocols (Deal et al. 1996) for the determination of NO_3-N concentrations using a Dionex ion chromatograph (Dionex Inc.). To account for seasonal differences and provide long-term baseline values, long-term monthly mean stream NO_3 concentration for a south-facing mixed-hardwood control watershed at Coweeta (Watershed 2) was used as an undisturbed reference for stream NO_3 concentration.

5.4.3 *Effects of fire on stream NO_3^-*

Stream NO_3-N responses to burning varied among the four studies. Two of the studies exhibited increased stream NO_3^-. After the JB burn, NO_3-N in stream water increased

from <0.01 to 0.055 mg N l^{-1} after 3 months and up to a maximum of 0.075 mg l^{-1} after 7 months, and remained elevated until June of the following year (Figure 5.10). Similarly, stream water NO_3^- concentration following the JK wildfire showed a marked increase over preburn values. Compared to the values for the previous 2 months, stream NO_3 increased from a low of 0.02 to 0.12 mg N l^{-1} within 6 weeks of the burn, and then quickly returned to near preburn values (Figure 5.9). In contrast, on both the HB and WSC burns, there were no measurable fire effects on stream NO_3-N (Figure 5.9). For example, using pre- and postburn regressions of stream NO_3, Vose et al. (1999) found no difference in NO_3 concentrations between the streams directly below the WSC burned area and an unburned reference stream. Similarly, there were strong similarities in monthly mean stream water NO_3 concentrations between the HB burn site and an undisturbed reference watershed both in terms of the magnitude and the seasonality before and after the burn (Figure 5.10).

The initial fuel condition and the type and season of burning appeared to be the primary factors determining the observed differences in the amount and duration of stream water NO_3^- responses (Table 5.2). Large differences were observed between fall burns and spring burns. The two sites that burned in the fall (JB and JK) showed large increases in NO_3^- concentration compared to the spring burns (WSC and HB) (Table 5.2). For example, the two sites that showed an NO_3^- response were burned in early and late fall when vegetation uptake and microbial immobilization were low. On the JB site, Knoepp and Swank (1993) observed significant increases in soil water NO_3^- concentration at 30 and 60 cm depths soon after the burn in September and a sharp increase again in June the next year. The initial postburn increase (Figure 5.10) was likely due to leaching of excess NO_3^- resulting from reduced uptake and increased mineralization (Knoepp and Swank 1993), and the later increase was probably related to increased decomposition of exposed surface organic layers and other fine organic material, and delayed increases in rates of nitrification often observed following fire (Knoepp and Swank 1993; White 1986). Because virtually no uptake of soil nutrients was taking place during the dormant season, and because of vegetation removal to within a few meters of the stream, excess NO_3-N was leached to the stream. To some extent, the same explanation can be used for observed increases in stream NO_3^- concentration following the JK burn. In late fall, no vegetation uptake was occurring to immobilize excess NO_3^-. Additionally, some of the riparian areas were burned, thus providing a potential direct input of NO_3^- to the stream.

Figure 5.10   Response of streamwater nitrate (NO_3-N) to a variety of burning prescriptions in the southern Appalachians, U.S. Coweeta WS 2 represents unburned reference conditions.

The streams that showed no measurable response (HB and WSC) drained watersheds that burned in early spring at the onset of leaf growth and nutrient uptake. Hence, vegetation that was not killed by the fire may have immobilized NO_3^- via uptake. In addition, Vose et al. (unpublished data) have demonstrated the effectiveness of riparian zones in reducing NO_3^- delivery to streams through microbial uptake. On the WSC site, the riparian zone served to buffer fire effects because it did not burn (Vose et al. 1999). Similarly, the riparian zone on the HB site, although burned in mid-March, may have immobilized most of the NO_3^- mobilized during the burn, since only a weak stream water NO_3^- response in the upper reaches of the watershed was observed on the HB site 2 weeks after burning.

Some of the variation in stream NO_3^- losses among burns may be related to differences in the initial condition (i.e., amount and quality) of forest floor fuels. For example, the fell and burn site (JB) had extremely high woody fuel loadings because all standing vegetation was felled prior to burning. Higher severity fires resulting from the heavy fuel loads have the potential to reduce the vigor of regrowing vegetation and reduce nutrient uptake (Elliott et al. 2002). For example, stream NO_3^- response on the JB site suggested low resistance (+750%) and resilience (30 week duration). This is likely due to high intensity and severity fires, which reduce the ability of vegetation regrowth to mediate the effects of burning. In contrast, the stand-replacement burns showed high resistance and resilience to disturbance due in part to lower fuels (hence, lower intensity and severity fires), an intact riparian buffer, and season of the burn. The JK wildfire was intermediate in resilience but less resistant to disturbance, possibly due to large accumulations of recalcitrant nitrogen in old-growth forest floor. Another possible explanation for the pulse of NO_3^- following the JK wildfire is low resistance to disturbance in old-growth forest ecosystems. There is some evidence to suggest that old-growth forests are less capable of sequestering nutrients mobilized by disturbance than actively growing, aggrading forest ecosystems. In addition, stand age, in the absence of fire, determines the accumulation of recalcitrant forms of N that are released during burning and become subject to volatilization and leaching from the system.

Applying the restoration response graph to the data (Figure 5.11) suggests that the "fell and burn" restoration treatment has the greatest potential for undesirable effects on stream NO_3^- because the magnitude (i.e., +750%) and duration (i.e., 30 weeks) of response exceed that observed in the old-growth wildfire. However, it should be noted that even the "worst-case scenario" of peak stream water NO_3^- concentration (i.e., 0.12 mg l^{-1} for the JK wildfire) is still quite low and within acceptable limits for surface water
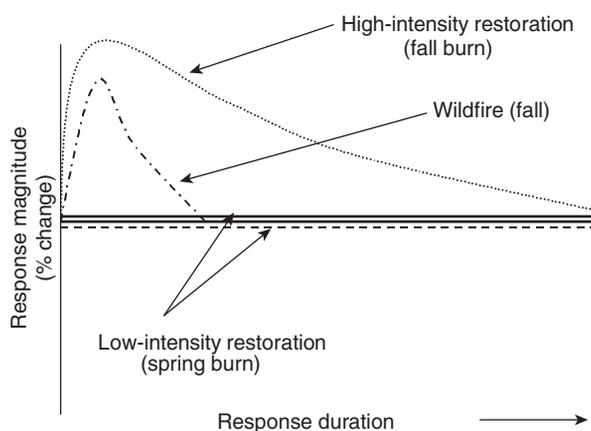

Figure 5.11   General patterns of changes in stream NO_3^- in response to different burning prescriptions.

quality. Using the resistance/resilience analogy, we conclude that while low baseline stream NO_3^- levels allow for a high-resolution assessment of response patterns across a range of fire types, using any of these prescribed fire restoration techniques to alter vegetation structure does not result in unacceptable effects on stream NO_3^-.

5.5 Summary

We have presented two different aspects of evaluating the effects of restoration on biogeochemical cycling and aquatic ecosystems. The first examined the direct responses of nutrient cycling pools and processes to restoration efforts aimed at improving nitrogen retention and processing in degraded riparian zones. The second, stream NO_3^- responses to restoration burning, examined responses to restoration efforts that do not target biogeochemical cycling *per se*, but directly or indirectly influence nutrient cycling pools and processes. We presented an approach to evaluate short- and long-term responses of restoration on biogeochemical cycling using the resistance and resilience analogy. Our adaptation of the resistance/resilience analogy to evaluate restoration responses focused on three key attributes. First, the complexity of ecosystem biogeochemical cycling requires the determination of key indicators of response. In our example, we focused on N cycling pools and processes because N is often tightly conserved, responds to changes in physical and biological conditions, limits productivity in many ecosystems, and is toxic in some forms and concentrations. Second, evaluation of responses (direct or nontarget) can be approached by assessing response magnitude and duration. Restoration effectiveness can be assessed in terms of how quickly pools and processes return to desired conditions and how long they stay with desired boundary conditions. Finally, variable responses should be expected since the impacts of degradation may impact pools and processes differently. For example, pools and processes that depend primarily on changes in biological parameters may respond quite differently than those dependent on changes in physical parameters.

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