IMPACTS OF HISTORIC FIRE, STAND AGE, AND NITROGEN SATURATION STATUS ON SOIL NITROGEN CYCLING IN MID-APPALACHIAN FORESTED WATERSHEDS

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ABSTRACT.—Our research objective was to determine if past land disturbance due to fire, stand age, or reported nitrogen saturation status have any influence on soil nitrogen cycling in forested watersheds. Seven forested watersheds in northeastern West Virginia were chosen for the study: a burned watershed, and logged watershed (past land disturbance); an old-growth watershed, and two mature 60-90 year old watersheds (stand age); and a nitrogen saturated watershed, and nitrogen limited watershed (N saturation status). Monthly net N mineralization and nitrification rates were measured along 3 transects on each watershed in June, July, and August 1998 by employing buried bags in 0-10 cm of mineral soil. The old-growth watershed and two nearby mature watersheds had similar net N mineralization and nitrification rates. The burned watershed had lower net N mineralization rates than a nearby logged watershed, indicating the importance of past land disturbance. Watershed 4 on the Fernow Experimental Forest, WV, which has been cited as the best example of a nitrogen saturated watershed in the northeastern United States, had significantly higher net N mineralization and nitrification rates than nearly adjacent Fernow watershed 10. The observed difference in nitrate-N export in streamflow between Fernow 4 (5.12 kg ha^{-1} yr^{-1}) and Fernow 10 (1.22 kg ha^{-1} yr^{-1}) could be due to significantly higher soil nitrate production rates.

Northern temperate forests long have been considered nitrogen limited systems based on the relative absence of nitrogen leaching from forested watersheds. However, long-term monitoring of experimental forested watersheds throughout the eastern United States has shown significant increases in nitrate export over the past several decades (Johnson and Lindberg 1992, Stoddard 1994, Peterjohn et al. 1996). The theory of nitrogen saturation has been formulated to explain increasing nitrate leaching. A nitrogen saturated ecosystem is defined as one in which chronic atmospheric nitrogen deposition results in nitrogen supplies that exceed plant and microbe demands, and thus, nitrogen is available for export in streamflow (Agren and Bosatta 1988, Aber et al. 1989, Aber et al. 1998, Aber et al. 2003). However, not all forested watersheds in the eastern United States show nitrogen saturation symptoms. In fact, substantial variation (500%) in nitrate exports has been reported for forested watersheds within the mid Appalachian region (DeWalle and Pionke 1996).

Variability in soil nitrogen cycling has been identified as an important factor in explaining these significant nitrate export differences (Williard et al. 1996). Many factors can cause differences in soil nitrogen cycling and subsequent nitrate exports, including differences in wet and dry atmospheric nitrogen deposition (Dise and Wright 1995, Aber et. al. 2003), soil carbon:nitrogen ratios (Johnson and Lindberg 1992, Lovett et al. 2002), nitrogen saturation status (Peterjohn et al. 1996, Gilliam et al. 2001), forest species composition (Fenn et al. 1998, Lovett et al. 2002), stand age (Vitousek and Reiners 1975, Goodale and Aber 2001), and past land disturbance history including agriculture and severe fire (Raison 1979, Compton and Boone 2000, Goodale and Aber 2001).

Our research objective was to examine if past land use disturbance (severe fire), stand age (old-growth), and nitrogen saturation status influenced soil N mineralization and nitrification rates in 7 forested watersheds in northeastern West Virginia. We assessed each factor by comparing "treated" watersheds (burned, old-growth, and nitrogen saturated) to nearby "control" (logged, mature, and nitrogen limited) watersheds.

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Table 1.—Physical and vegetative characteristics of the seven study watersheds in West Virginia.

Watershed	Area (km^2)	Aspect	Geology[2]	Dominant Overstory Vegetation
Fernow 4	0.39	NNW	CCP	*Quercus rubra* L., *Liriodendron tulipifera* L.
Fernow 10	0.15	N	CCP	*Q. rubra* L., *L. tulipifera* L., *Acer rubrum* L.
Shays Run	6.03	S	PVA	*Tsuga canadensis* L.
Otter Run	0.48	E	PVA	*Prunus serotina* L., *A. rubrum* L.
W. Three Spring	0.12	E	MCG	*A. saccharum* Marsh., *Tilia americana* L.
Karly Run[1]	*0.28*	*SE*	*MCG*	A. saccharum *Marsh.,* T. americana *L.*
Freeland Run	1.28	NNW	MCG	*A. saccharum* Marsh., *Betula lenta* L.

[1]A previously unnamed stream.
[2]CCP represents Catskill/Chemung/Pocono shale and sandstone; PVA represents Pottsville/Allegheny sandstone; MCG represents Mauch Chunk/Greenbrier shale/limestone (Reger 1923, Reger 1931).

Table 2.—Four-week mean (June through August 1998) net N mineralization and net nitrification rates in the upper 10-cm of mineral soil for 7 study watersheds in West Virginia.

Watershed	Watershed Type	Mean Stream Nitrate-N (mg L^{-1})	Net N Mineralization (mg $kg^{-1}4wk^{-1}$)[2]	Net Nitrification (mg $kg^{-1}4wk^{-1}$)
Fernow 4	N-saturated	0.751	17.20 a	13.02 a
Fernow 10	N-limited	0.216	5.26 b	-0.17 b
Shays Run	Burned	0.043	4.00 a	5.77 a
Otter Run	Logged	0.111	11.76 a	8.00 a
W. Three Spring	Old-growth	0.753	14.05 a	11.11 a
Karly Run[1]	Mature	0.547	19.82 a	8.02 a
Freeland Run	Mature	0.500	14.62 a	8.53 a

[1]A previously unnamed stream.
[2]Net N mineralization and net nitrification means with different letters are significantly different among basins at α=0.05 using Tukey's HSD mean separation procedure.

There has been little research conducted on the long-term impacts of severe fire on forest soil nitrogen cycling (Gagnon 1965, Hornbeck and Lawrence 1996). Severe fires may potentially affect long-term forest nitrogen cycling by volatilizing significant amounts of soil nitrogen (Raison 1979). Thus, we hypothesized that a historically burned watershed would have significantly lower N mineralization and nitrification rates than a nearby historically logged watershed. Stand age can affect nitrogen leaching, because young aggrading forests normally have greater nutrient demands than mature or old-growth forests, and old growth forests have relatively low productivity resulting in greater soil N accumulation compared to C accumulation (Vitousek and Reiners 1975, Goodale and Aber 2001). We predicted that an old-growth (> 150 years old) watershed would have significantly greater N mineralization and nitrification rates than nearby mature (60 – 90 years old) watersheds. We further hypothesized that a nitrogen saturated watershed, characterized by increasing nitrogen export over an extended period of time, will have greater N mineralization and nitrification greater rates than a nitrogen limited watershed.

Study Areas

The study watersheds were located in Tucker and Randolph counties within the Monongahela National Forest and each was 100 percent forested with no major disturbances in the past 60 years. The watersheds are relatively small and contain first or second order streams (table 1). Predominant overstory species across the seven watersheds were sugar maple (*Acer saccharum* Marsh.) and northern red oak (*Quercus rubra* L.) (table 1).

The watersheds were classified according to their disturbance history (burned, logged, and old growth) and nitrogen saturation status (table 2). Burned watersheds experienced severe fires 60 to 90 years ago caused by the abundant slash left behind after the clearcutting of mid-Appalachian forests in the early 1900's. Current vegetation resulted from natural recovery. Historical fire records and maps from the Monongahela National Forest, WV were examined to locate these severely burned areas. The logged watersheds were cut 60 to 90 years ago, and did not experience any severe fires, based on fire records. Stand age inventories for the Monongahela National Forest, WV were used to identify old-growth watersheds. Old-growth stands were defined as any stand not harvested within the past 150 years. The old-growth watershed was located adjacent to the Otter Creek Wilderness. The watersheds within each watershed type comparison (burned vs. logged, old-growth vs. mature, and nitrogen saturated vs. nitrogen limited) contained the same underlying bedrock geology (table 1). All of the study watersheds were located on previously unglaciated terrain.

Methods

Field Methods

Monthly net N mineralization and nitrification rates were measured in June, July, and August 1998 along three 16-m transects on each watershed using buried bags in 0-10 cm of mineral soil (Eno 1960). Protocols for the buried bag method were adopted from Hart et al. (1994). The three transects were located at the watershed mouth (near the stream sampling point) at a low elevation, near the middle of the watershed at a mid-slope elevation, and in the headwaters at a ridge top elevation. Nine sampling points were spaced at 2-m intervals along each 16-m sampling transect. The organic horizon was removed at each point and set aside. Then a 10-cm diameter by 10-cm deep mineral soil core was taken and gently extruded into a quart-sized zipper-lock plastic bag. Bags were closed, placed back in the resulting hole, and covered with the organic horizon material. After a four-week incubation period, the buried bags were collected and composited by threes to yield nine composite samples per watershed for post-incubation soil ammonium and nitrate measurement.

Another soil sampling transect was located adjacent to the buried bags to measure pre-incubation levels of soil ammonium and nitrate. At each transect, nine mineral soil cores were collected and composited in the field by threes and put in coolers on ice for transport to The Pennsylvania State University.

Monthly net N mineralization and net nitrification rates were computed according to the following equations:

Net N mineralization = [(NH4-N + NO3-N)t+1 - (NH4-N + NO3-N)t]
Net nitrification = [(NO3-N)t+1 - (NO3-N)t]

where t represents the pre-incubation levels and t+1 represents the post-incubation levels. NH_4-N and NO_3-N were measured in mg per kg of soil. Net nitrification and N mineralization rates were expressed as mg of N per kg of dry soil per 4-week period (mg kg^{-1} $4wk^{-1}$). Negative net mineralization values were possible and represented net immobilization of inorganic nitrogen by the soil microbes during the incubation period.

At each sampling transect, three additional soil cores were taken for gravimetric determination of soil moisture. One Taylor maximum-minimum thermometer was buried 5-cm in mineral soil to measure maximum and minimum soil temperatures during the incubation period. Soil temperature at the time of the site visits also was noted at the start and end of each incubation period.

In August 1998, sampling transects for standard soil chemistry were established adjacent to (within 3 meters) the 3 buried bag transects on each watershed. Nine sampling points were spaced at 2-m intervals along each 16-m sampling transect. The entire organic horizon and the top 10 cm of mineral soil were collected at each point. Organic and mineral soil samples were transported to The Pennsylvania State University, Agricultural Analytical Laboratory for standard soil tests [exchangeable Ca, Mg, K, and P, effective cation exchange capacity (CEC), acidity, and pH]. Subsamples of organic and mineral soil were analyzed for total C and N at the Northeastern Research Station's Timber and Watershed Laboratory in Parsons, WV.

Stream water grab samples were collected during baseflow from the 7 watersheds during August 1997 and March 1998. They were transported on ice to the Water Analysis Laboratory at The Pennsylvania State University for dissolved nitrate, ammonium, and organic N analysis.

Laboratory Measurements

Prior to analysis, organic and mineral soil samples were dried at 70 °C for up to 12 hours and then passed through a 2-mm sieve. Soil nitrate was extracted with an ammonium sulfate and boric acid solution and analyzed with an ion-specific electrode (Griffin 1995). Soil ammonium was extracted with a 0.5 N potassium chloride solution and determined with a Technicon Autoanalyzer (Keeny and Nelson 1982). Results were expressed as mg of NO_3-N and NH_4-N per kg of dry soil. Exchangeable Ca, Mg, and K and labile, mineralized P were determined with the Mehlich 3 method (Wolfe and Beegle 1995). Exchangeable acidity was extracted and measured by the SMP buffer method (Sims and Eckert 1995). Soil pH was measured in a soil water slurry (5 g : 5 ml) using an electronic pH meter (Sims and Eckert 1995). Effective CEC was calculated by summing the exchangeable Ca, Mg, K, and acidity (Ross 1995).

Dried sieved organic and mineral soil was prepared for C:N determination by placing approximately 6 mg of soil in a 5 mm x 9 mm tin capsule along with a small amount of vanadium pentoxide. Total C and N percentages in the soil were determined by combustion using an organic elemental analyzer (Carlo Erba NA 1500 CNS Analyzer, Valencia, CA) (Baccanti et al. 1993).

Moisture content of the soil for the net N mineralization and nitrification study was determined on a dry weight basis (Gardner 1965). Wet soil was weighed to the nearest 0.01 mg, dried in an oven at 105 °C for 24 hours, and re-weighed to determine the percent soil moisture.

Dissolved (<0.45 ¼m filtered) stream nitrate, ammonium, and organic N (difference between total dissolved and nitrate plus ammonium) were analyzed at the Water Analysis Laboratory, The Pennsylvania State University via cadmium reduction, automated phenate, and persulfate oxidation methods, respectively (American Public Health Association 1995).

Statistical Methods

Prior to performing analysis of variance procedures (ANOVA), data sets were analyzed to determine if error terms were normally distributed and homogeneous. Normal probability plots and Shapiro-Wilk test statistics (SAS Institute Inc. 1985) confirmed that all the data sets were distributed normally, and Bartlett's tests (SAS Institute Inc. 1985) showed error variances for the various ANOVA models to be relatively homogeneous; consequently, no data transformations were required.

An ANOVA model for the net N mineralization and nitrification data set was performed with watershed type, month, and elevation effects. Another ANOVA model was developed with soil chemistry data to determine if soil chemistry parameters were significantly different among watershed types. Tukey's HSD mean separation procedure was added to each of the ANOVA models (SAS Institute, Inc. 1985). Maximum R^2 improvement stepwise regression models were developed to predict net N mineralization and net nitrification rates from soil chemistry parameters, soil moisture, and soil temperature (SAS Institute, Inc. 1985).

Results and Discussion

N Saturation Effects

Comparisons of net N mineralization and net nitrification among different watershed types yielded only one statistically significant result (table 2). Fernow 4, which has been cited as the best example of a nitrogen saturated watershed in the northeastern United States (Peterjohn et al. 1996, Fenn et al. 1998), had significantly higher net mineralization and nitrification rates than Fernow 10. Christ et al. (2002) also found significantly greater soil net nitrification potentials on Fernow 4 than Fernow 10. The two watersheds are nearly adjacent, have similar management histories, and both serve as experimental controls in the Fernow Experimental Forest, WV. The observed difference in nitrate-N export in streamflow between Fernow 4 (5.12 kg ha^{-1} yr^{-1}) and Fernow 10 (1.22 kg ha^{-1} yr^{-1}) could be due to the significantly greater soil nitrate production rates (table 2). Mean soil chemistry differences may be causing these differences in nitrate production. Mean mineralized phosphorus concentrations were over five times greater on Fernow 4 compared to Fernow 10, indicating microbial populations on Fernow 4 are actively decomposing organic matter and rapidly cycling nutrients. Phosphorus has been cited as a limiting nutrient of N-fixing bacteria

(Vitousek and Howarth 1991). N-fixation on Fernow 10 may be limited by low phosphorus levels, resulting in smaller ammonium pools for nitrifying bacteria. Measured ammonium pools were significantly lower on Fernow 10 (13.62 mg kg^{-1}) compared to Fernow 4 (16.47 mg kg^{-1}), providing support to this hypothesis. The significantly lower calcium concentrations on Fernow 10 (0.344 meq $100g^{-1}$) compared to Fernow 4 (1.267 meq $100g^{-1}$) also could be limiting the activity of N-fixing bacteria and ultimately reducing soil nitrification rates (Alexander 1977).

Severe Fire Effects

Within the PVA geology category, Shays Run, a historically burned watershed, had lower net mineralization rates than logged watershed Otter Run (table 2). Even though the difference was not statistically significant, it was large enough to warrant discussion. Fire can cause significant N losses from forest ecosystems via volatilization of N in the litter layer and upper mineral horizons (Grier 1975, Raison 1979). Repeated small N losses due to low intensity fires or substantial one-time N losses due to an intense burn can deplete long-term soil N pools (Gagnon 1965, Carreira et al. 1994, Hornbeck and Lawrence 1996). Soil N pools on Shays Run apparently have been restored since the devastating fire on Canaan Mountain in the early 1920s, which completely consumed a thick organic horizon and consumed the top portion of the mineral soil (L. White, Monongahela National Forest - personal comm.); soil N pools (0.266%) on Shays Run were similar to Otter Run (0.301%). Given that all soil chemistry parameters were relatively similar for Shays Run and Otter Run, it is difficult to determine why the burned watershed had lower net mineralization rates. Perhaps the severe fire has had a long-term effect on the ammonifying bacteria populations.

Stand Age Effects

Net N mineralization and nitrification rates in West Three Spring, an old-growth watershed, were on the same order of magnitude as the other two logged watersheds with the same underlying geology (table 2). The old-growth watershed had the second-highest net nitrification rates among the seven watersheds sampled. This was a surprising result considering classical succession theory that predicts nitrification will be progressively inhibited through succession (Robertson and Vitousek 1981, Robertson 1982). However, recent research has shown that significant gross soil nitrification can occur in late successional forest ecosystems, and significant nitrate may become available if microbial immobilization is decoupled through a disturbance such as the buried bag technique (Stark and Hart 1999).

Predicting Net N Mineralization and Nitrification

Soil N, mineral P, moisture, and C:N ratios together explained 46 percent of the variation in net nitrification across all watersheds. Soil moisture helps determine nitrification rates by controlling rates of microbial activity (Stanford and Epstein 1974). Mineralized phosphorus levels are positively correlated with net nitrification rates, since they are both the result of an active microbial population. Competition for nitrogen between heterotrophic bacteria and nitrifying bacteria plays a major role in determining soil nitrate production, and the amount of labile C directly affects this competition (Riha et al. 1986). Heterotrophic bacteria are thought to be the most successful short-term competitors for N (Schimel and Firestone 1989), and C serves as the energy source for bacteria. Consequently, higher carbon levels result in greater N demand. Thus, soils with more available C have more heterotrophic demand for ammonium and less ammonium is available to be nitrified (Riha et al. 1986).

The prediction of net N mineralization was much weaker than net nitrification. Soil N and mineral P levels together explained only 19 percent of the variation in net N mineralization rates. C:N ratios were not significant in explaining net N mineralization rates.

Conclusions

The best example of a nitrogen saturated watershed in the northeastern United States, Fernow 4, exhibited significantly greater net N mineralization and nitrification rates than a nearby watershed with similar management history, Fernow 10. These results demonstrate that soil nitrogen dynamics are important in controlling nitrogen exports from forested watersheds. A historically burned watershed had lower net N mineralization rates than a nearby logged watershed that did not experience severe fire, indicating that past land disturbance can impact forest soil nitrogen cycling. An old-growth watershed and two mature (60-90 year old) watersheds had similar net N mineralization and nitrification rates, supporting the theory that soil nitrate production can be significant in late successional forested watersheds. Changes in factors that affect soil nitrogen cycling, such as soil C:N ratios, soil fertility, and soil moisture can significantly affect nitrate leaching from forested watersheds.

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