Biogeochemistry **70:** 117–134, 2004.

Nutrient mobilization and processing in Sonoran desert riparian soils following artificial re-wetting

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Received 14 April 2003; accepted in revised form 29 October 2003

Key words: Desert stream, Flash flood, Nutrient sources, Riparian zone

Abstract. Research in river-floodplain systems has emphasized the importance of nutrient delivery by floodwaters, but the mechanisms by which floods make nutrients available are rarely evaluated. Using a laboratory re-wetting experiment, we evaluated the alternative hypotheses that increased nutrient concentrations in riparian groundwater during flash floods are due to (H1) elevated nutrient concentrations in surface floodwaters entering the riparian zone or (H2) re-mobilization of nutrients from riparian soils. We sampled soils from the riparian zone of a 400 m reach of Sycamore Creek, AZ. Two sub-samples from each soil were re-wetted with a solution that mimicked the chemistry of floodwaters, with one sub-sample simultaneously treated with a biocide. We also measured structural characteristics of soils (texture, organic matter, moisture, and extractable nutrients) to investigate relationships between these characteristics and response to re-wetting. Riparian soils exhibited considerable variation in physical and chemical structure. Soil organic matter, moisture, and texture co-varied among samples. Re-wetting increased concentrations of nitrate and ammonium, and decreased SRP, relative to initial concentrations. Live soils were significantly lower in NO_3^- and SRP than biocide-treated samples. Extractable DIN pools were the best predictors of mobilization, and soil organic matter was strongly correlated with nitrate losses, probably via its relationship with microbial uptake. Nutrient mobilization and processing also varied considerably with depth, lateral position, and among plots. We estimate that 70–80% of N in riparian groundwater during flash floods is re-mobilized from riparian soils, but are unable to reject the hypothesis that flood inputs may be important sources of nutrients to riparian soils over longer time scales.

Introduction

The Flood Pulse Concept hypothesizes that floods increase the availability of resources in riparian soils, as well as provide habitat essential for many aquatic organisms (Junk et al. 1989). As a result, disruption of connectivity between the channel and floodplain by flow modification or flood-control levees can influence the ecological dynamics river-floodplain systems (Poff et al. 1997). While the FPC was originally conceived to describe dynamics of lowland tropical rivers, ecologists have used the flood pulse as an organizing concept for studying and managing large temperate and arid rivers as well (Ward and Stanford 1995). In general, these studies have focused on flow variability in terms of disturbance and connectivity. The importance of materials delivered to floodplains by elevated waters is often assumed, but the processes influencing nutrient availability in floodplain soils during inundation are poorly understood (Baldwin and Mitchell 2000).

Table 1. Chemistry of surface water and riparian ground water during baseflow and flood conditions (mean ± SE). Data are from Marti et al. (2000).

Solute	Surface baseflow	Surface flood	Riparian baseflow	Riparian flood
NH_4^+ (μg N L^{-1})	6 ± 2	98 ± 34	33 ± 5	84 ± 31
NO_3^- (μg N L^{-1})	28 ± 6	985 ± 161	8 ± 2	110 ± 27
SRP (μg P L^{-1})	19 ± 2	201 ± 61	18 ± 1	16 ± 2
DO (mg L^{-1})	>8	>8	1.6 ± 0.1	2.0 ± 0.1
Conductivity (μS cm^{-1})	503 ± 13	349 ± 46	784 ± 30	1354 ± 84

Flash floods are important events that shape ecological processes in arid streams (Fisher et al. 1982, Grimm 1987). While high flow events often scour algae and redistribute sediments in the active channel, riparian biota are less frequently disturbed, owing to lower flow velocities at the channel periphery and to the greater structural strength of riparian vegetation and sediments (Naiman et al. 1999). Flash floods are also spatially limited in arid lands; transmission losses to alluvial deposits are often high (Graf 1988). This dissipation of floods not only decreases disturbance intensity downstream, but also delivers water and its associated particulate and dissolved constituents to riparian soils. This increase in water availability influences the survival and growth of riparian vegetation in arid lands (e.g. Stromberg 2001); but the importance of the dissolved nutrients entering the riparian zone during these floods is less well known.

Surface waters during flash floods in arid catchments show elevated concentrations of many nutrients compared to baseflow conditions (Fisher and Minckley 1978, Fisher and Grimm 1985). Similarly, concentrations of DIN (esp. NO_3^-) and SRP increase by approximately ten-fold and three-fold, respectively, in riparian groundwater during floods (Table 1). Concentrations in riparian zones are much lower than surface floodwaters (Marti et al. 2000), suggesting that while floods are associated with increased solute concentrations in the riparian zone, nitrogen and phosphorus are processed rapidly.

Biogeochemical processes in soils are strongly influenced by their physical structure. In particular, soil texture influences hydraulic conductivity and therefore the residence time of riparian groundwater. In saturated environments, finer sediments typically have lower flow velocities, greater organic matter content, and lower redox potential. These characteristics influence rates of N transformations (Chestnut and McDowell 2000, Pinay et al. 2000) and sorption dynamics of P (e.g., Fabre et al. 1996). Moreover, heterogeneity of structure and process rates is high in riparian soils (Tabacchi et al. 2000).

Because of the fertility of floodplain soils, disturbance, rather than resource availability, is viewed as the major organizer of riparian forests (Hupp and Osterkamp 1996, Bendix and Hupp 2000). In a broad survey of European and North American floodplains, Spink et al. (1998) found that few sites responded to fertilization of N and P. However, some evidence exists that riparian trees are limited by N in arid lands, and that they are able to use pulses of nutrients associated with floods. Adair and Binkley (2002) found that cottonwood seedlings were simultaneously limited by water and nutrient availability. Schade et al (2002) found that foliar %N of riparian willows was correlated with local increases in water table elevation during floods, although the best predictor of foliar N was N mineralization rates in soils. The mechanism(s) by which floods make N available to trees in arid lands is unclear.

One possible source for observed increases in nutrient concentrations in riparian groundwater during floods is floodwaters (H1). Under baseflow conditions, removal of N (via denitrification and/or plant uptake) is commonly observed at stream-riparian interfaces (Hill 2000), and some evidence suggests that similar processes occur in arid streams (Schade et al. 2002); however, movement of floodwaters into the riparian zone may occur too rapidly for processes at the stream interface to remove nutrients during high-flow events. An alternative explanation is that elevated concentrations are due to the remobilization of materials from riparian soils (H2). Conductivity of riparian groundwater doubles during floods, while surface floodwater conductivity is reduced by half, suggesting that even during floods when water moves rapidly into the riparian zone, physical release (i.e., dissolution) can occur quickly (Marti et al. 2000). Re-wetting of riparian soils can also activate microbial processes, including mineralization and nitrification (Mitchell and Baldwin 1999).

We tested these alternative hypotheses using an artificial re-wetting experiment. In this experiment, we exposed riparian soils to water with and without a biocide. We predicted that if flood delivery of nutrients (H1) is important, then (a) re-wetting should not result in increased nutrient concentrations and (b) processing rates should be insufficient to remove nutrients from the incoming floodwaters. If re-mobilization of nutrients (H2) is important, then (a) re-wetting soils should result in an increase in nutrient concentrations and (b) processing rates should be sufficient to remove floodwater-derived nutrients. We also sought to determine the riparian soil characteristics that influenced the magnitude of re-mobilization and processing, and to describe the spatial pattern of those soil characteristics.

Methods

Study site

Sycamore Creek, located 32 km northeast of Phoenix, AZ, is a tributary of the Verde River, and is located in the Tonto National Forest. The catchment is at the southwestern edge of the Central Highlands Province, and is bounded on the east by the Mazatzal mountains (Montgomery and Harshbarger 1989). The stream drains a 505-km^2 catchment that ranges in elevation from 427 to 2164 m. Precipitation is bimodally distributed, with peaks in late summer and winter. Precipitation is also unevenly distributed in space, with higher elevations (>1000 m) receiving an average of 51 cm per year, and lower elevations receiving an average of 39 cm (Fisher et al. 1982).

Soil samples for this experiment were collected from a 400 m reach located at 610 m in elevation. At this site the active channel is ~20 m wide, and consists mostly of coarse sand and gravel. Riparian terraces extend an additional 10–30 m on each side. Gooding's willow (*Salix goodingii*), mesquite (*Prosopis velutina*), and ash (*Fraxinus velutina*) are the dominant trees at this site. The shrub seep-willow (*Baccharis salicifolia*) is also common on sand bars and terrace banks. Understory vegetation is abundant and dominated by annual grasses. Adjacent uplands at this site are derived from Precambrian granites and support Sonoran desert scrub vegetation.

Soil sampling and analyses

Soils were collected on 8 July , 2002, during a prolonged drought. Surface water had been absent since early April and the water table was more than 1 m below the active channel surface at the time of sampling (J.B. Heffernan, personal observation). We established six pairs of 1-m^2 plots. Within each pair, one plot was located on the bank between the active channel and the riparian terrace, and the second on the riparian terrace, 3–5 m laterally from the active channel. Plot pairs were separated by ~150 m longitudinally, with three pairs on each side of the stream. Within each plot, we took three replicate cores ($\text{area} = 23.75\,\text{cm}^2$) at each of two depths (0–10 and 30–40 cm; total $n = 72$). Soils were sealed in plastic bags and stored in coolers until return to the lab. Samples were weighed, and sieved into coarse (>2 mm) and fine (<2 mm) fractions. Sub-samples from the fine fraction were used in all subsequent analyses, and we corrected all values for relative abundance of coarse and fine particles.

For each sample we determined bulk density, percent moisture, percent organic matter, percent silt + clay, and extractable NO_3^- and NH_4^+. Moisture and AFDM were determined gravimetrically by drying soils at 60 °C for 48 hours then ashing them at 550 °C for 4 h. We extracted NO_3^- and NH_4^+ by shaking 10 g of soil for one hour in 50 ml of 0.5 M K_2SO_4, then drained soils through pre-ashed and pre-rinsed Whatman 42 paper filters. Samples were frozen until analysis. We determined silt and clay content using the density hydrometer method, using settling times of 40 s and 7 h to distinguish sand from silt and silt from clay, respectively.

Flood simulation experiments

In order to determine the relative importance of flood-derived and re-mobilized nutrients, we artificially re-wetted soils in the lab. The re-wetting experiment consisted of two treatments: Live, to which we added a solution intended to simulate Sycamore Creek floodwaters, and Dead, to which we added the simulated floodwater with $HgCl_2$ (Table 2). Mercuric chloride is a highly effective biocide that does not influence physical or chemical structure of soils and sediments at the concentrations used in this study (0.30 $\mu g\ HgCl_2/g$ sediment; Tuominen et al. 1994, Trevors 1996).

Table 2. Concentrations of solutes in experimental floodwater. Target concentrations are based on Fisher and Minckley (1978) and Fisher and Grimm (1985). Mercuric chloride was added to only one of each pair of subsamples.

Solute	Target concentration (ppm)	Blank concentration (±SE) (ppm)	Added as
Ammonium	0.10	0.123 (0.014)	$(NH_4)_2SO_4$
Nitrate	1.00	1.05 (0.07)	KNO_3
Orthophosphate	0.20	0.209 (0.002)	$NaH_2PO_4 \cdot H_2O$
DOC	1	–	Dextrose
Sodium	15	–	NaCl and $NaHCO_3$
Chloride	10	–	NaCl
Potassium	9	–	K_2SO_4 and KNO_3
Sulfate	8	–	K_2SO_4
Bicarbonate	5	–	$NaHCO_3$
Mercury	450	–	$HgCl_2$

Floodwater chemistry in arid streams can vary considerably (Fisher and Grimm 1985, Jacobson et al. 2000). The solute concentrations we chose were similar to mean values from several studies, and were well within the range of concentrations that occur in Sycamore Creek floodwaters. One exception is dissolved organic carbon, which averages 10 mg/l in Sycamore Creek floodwaters. We added 1 mg DOC/l as dextrose, which is considerably more labile than most DOC compounds carried by flash floods (J. Edmonds, personal communication). While we sacrificed realism in terms of DOC quality, greater realism would have been sacrificed by not adding any DOC, which may be a significant energy source for microbes in organic-poor soils.

Because bulk density of soils was highly variable, we used a constant volume rather than a constant mass of soil in the re-wetting experiments. We added 15 ml of floodwater solution to ~30 (25–70 g) ml of sediment, shook vigorously for 10 s to ensure complete mixing. In cases where 15 ml was insufficient to fully saturate samples, we added additional floodwater in 5 ml increments until soils were fully saturated. Soils were incubated at room temperature (25 °C) for 4 h, then re-shaken and gravity drained through Whatman 42 Paper filters. Because yield of draining pore-waters was relatively low (0.8–6 ml), we immediately diluted samples to approximately 10 ml. Weights of soil samples and volumes of floodwater added, pore-water drained, and dilluent were determined gravimetrically for each sample. Prior to the experiment, we reserved two aliquots of the floodwater solution for chemical analysis. Two additional water samples were not exposed to sediments, but were otherwise treated the same as other samples, and were used as blanks. Samples were then refrigerated until analysis within 24 h for ammonium and nitrate, and 48 h for SRP and chloride. Differences between blanks and the floodwater solution were within analytical error for nitrate and SRP. Ammonium samples showed small (relative to variation among samples) but consistent contamination for which we corrected.

Chemical analyses

We analyzed porewaters for NO_3^-, NH_4^+, and SRP on a Latchat QC8000 Flow Injection Analyzer with an auto-dilutor. We measured nitrate using the cadmium reduction method (Wood et al. 1967); ammonium using the phenolhypochlorite method (Solorzano 1969); and SRP using the molybdate blue method (Murphy and Riley 1962). K_2SO_4-extractable nutrients were measured on a Bran and Leubbe TRAACS 800 Segmented Flow Analyzer using the same chemical analyses. External source quality controls were usually within 10% of expected values.

Calculations

Our experimental design allows us to use a mass balance approach to determine abiotic and biotic fluxes of nutrients to and from riparian soils. We determined remobilization of material from sediments as the difference in concentration between initial floodwater and the post-incubation solution of the biocide treatment. Because masses of soils and volumes of floodwater solution added differed slightly (<10%) between paired sub-samples, we calculated biotic processing as the difference in concentration between the live treatment and an expected concentration (i.e. in the absence of biological processing), which was based on per mass remobilization from the biocide treatment.

We calculated release/sorption of nutrients from sediments on a unit mass basis:

$$R = \frac{V_B(C_B - C_I)}{M_B} \tag{1}$$

where R is re-mobilization (µg N or P/g sediment), C_I is the initial concentration of the nutrient in the floodwater solution (from Table 2; mg/l), C_B is the concentration in the biocide treatment at the end of the incubation (mg/l), V_B is the volume of water added to the biocide-treated sample (ml), and M_B is the mass of the biocide-treated sample (g). C_B was calculated as the product of measured concentration and the dilution coefficient for each sample. Although values for R were both positive and negative (i.e., net de-sorption and adsorption were observed), we will refer all abiotic processes as *mobilization*.

We used the biocide treatment to predict mobilization in the live samples, because soil mass and solution volume differed between live and dead sub-samples:

$$C_E = \frac{R \cdot M_L}{V_L} + C_I \tag{2}$$

where C_E is the expected concentration (mg/l), M_L is the mass of soil in the live sample (g), and V_L is the volume of water added to the live treatment (ml).

We then used expected mobilization to calculate rates of biotic processing as follows:

$$B = \frac{V_L(C_E - C_L)}{M_L} \tag{3}$$

where B is biotic processing (μg N or P/g sediment), and C_L is the concentration in the live treatment following the incubation (mg/l). Since several mechanisms (e.g., mineralization, uptake, denitrification, etc.) could be responsible for differences between observed and expected concentrations, we refer to this term as *processing*, with decreases in concentration interpreted as positive. We used values of mobilization (R) and processing (B) in subsequent statistical analyses.

In order to determine the relative importance of flood-delivered and re-mobilized nutrients to a given soil core, we calculated inputs of each nutrient from floodwaters and re-mobilization:

$$I_F = p \cdot C_I \cdot V_C \tag{4}$$

where I_F is flood input (μg N or P), C_I is floodwater concentration (mg/l), V_C is core volume ($\sim 230\,\text{cm}^3$), and p is porosity of riparian soils (0.3: Jones et al. 1995). We calculated inputs via re-mobilization as follows:

$$I_R = R \cdot M_C \tag{5}$$

where I_R is re-mobilization input, and M_C is the mass of the soil core. We used these inputs to determine the relative importance of flood-delivered nutrients:

$$\%\text{FDN} = \frac{I_F}{I_F + I_R} \tag{6}$$

where %FDN is the percentage of groundwater nutrients delivered by floodwaters.

Statistical analyses

Differences between initial floodwater and post-incubation concentrations were evaluated for each nutrient species using a one-sample *t*-test (H_0: Δ concentration/g sed = 0) for both the biocide and live treatment. Differences in nutrient concentrations between the live and biocide treatments were determined by paired *t*-test using weight-corrected change in concentration. Un-transformed values were used in *t*-tests. We used log-transformed values in subsequent statistical analyses when residuals showed deviation from normality as assessed using probability plots. We used forward stepwise multiple regression analysis to determine relationships between soil organic matter, percent moisture and K_2SO_4-extractable nutrients and mobilization and biotic processing of nutrients. We also performed these analyses omitting extractable nutrients as independent variables.

Spatial patterns in both state variables and process rates were assessed using a three-way ANOVA, with plot, lateral position, and depth as factors. Our *a priori* assumption that different physical, chemical, and biological processes might operate on sediments in these locations is reflected in our sampling regime; in effect, we view variation in these processes as a natural experiment, and have analyzed data accordingly. We used Bonferonni's adjustment to determine significant differences among groups. All statistical analyses were carried out using Systat (v.10.0) statistical software.

Results

Soil characteristics

Riparian soil texture varied from sandy gravels to loam, with silt and clay abundance highly correlated ($r^2 = 0.76$; $p < 0.001$). Silt + clay content was significantly correlated with soil organic matter ($r^2 = 0.53$) and moisture ($r^2 = 0.67$). Organic matter and moisture also showed significant covariation ($r^2 = 0.44$). Shallow bank soils had higher silt and clay, organic matter, and moisture contents than other soils (Figure 1).

Extractable nutrient pools were correlated with organic matter content and soil moisture. Multiple-regression analyses showed that extractable ammonium was positively associated with OM, and negatively associated with soil moisture (multiple-$r^2 = 0.69$, $p < 0.001$; Figure 2(a and b)). The same variables were significantly correlated with nitrate concentration, but explained less of the observed variance (multiple-$r^2 = 0.33$, $p < 0.001$; Figure 2(c and d)). Extractable ammonium was higher in bank soils than in terrace soils ($F_{1,48} = 63.6$; $p < 0.001$), and higher in shallow soils than in deep soils ($F_{1,48} = 12.6$; $p < 0.001$). The major source of variance in extractable nitrate was variation in lateral patterns among plots ($F_{5,48} = 9.95$; $p < 0.001$), but plot ($F_{5,48} = 5.64$; $p < 0.001$) and depth ($F_{1,48} = 7.62$; $p = 0.01$) were also significant sources of variance.

Re-mobilization and processing

Exposure of floodwater to biocide-treated riparian soils resulted in significant increases in NH_4^+ ($p < 0.001$) and NO_3^- ($p < 0.001$), raising concentrations by 670 and 135%, respectively; and a significant decrease in SRP ($p < 0.05$; Figure 3). Live treatments were significantly lower than biocide-treated riparian soils in NO_3^- ($p < 0.001$) and SRP ($p < 0.001$), but biocide treatment did not influence concentrations of NH_4^+ ($p = 0.947$; Figure 3). Over the course of the 4 h incubation, nitrate concentrations in the live treatment decreased by 40%, and SRP by 25%, on average, compared to expected values. The mean decrease in nitrate was approximately equal to initial flood inputs (92%), although this percentage varied considerably among samples.

Mobilization of inorganic N was strongly controlled by extractable pools (NH_4^+: $r^2 = 0.79$, $p < 0.001$; NO_3^-: $r^2 = 0.91$; $p < 0.001$; Figure 4), although the slope of the relationship differed greatly between nitrate and ammonium. The slope of the nitrate regression approaches unity, indicating that nearly all nitrate was released into the water column. In contrast, only ~10% of the ammonium in the extractable pool was re-mobilized from soils. Spatial patterns for R_{NH_4} and R_{NO_3} closely matched patterns in extractable pools. R_{SRP} varied as a function of depth ($F_{1,48} = 16.85$, $p < 0.001$), with deep soils acting as abiotic sinks of SRP and shallow soils not statistically different from zero.

Relationships between biotic processing and soil characteristics varied among nutrients. Soil characteristics (moisture, organic matter) were poor predictors of

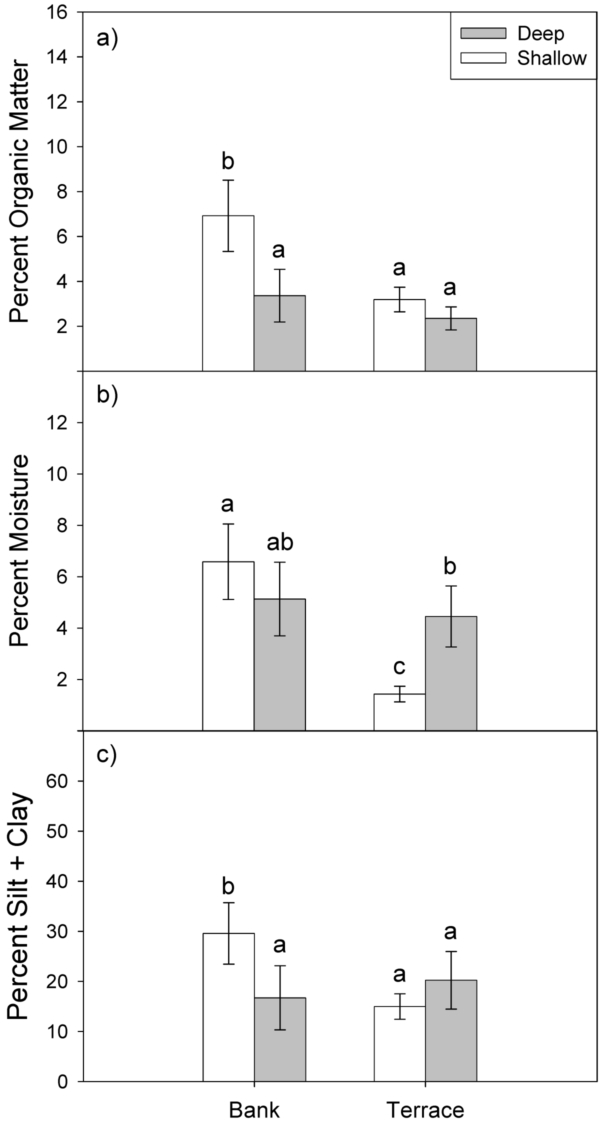

Figure 1. Spatial pattern of (a) organic matter, (b) moisture, and (c) silt + clay content of Sycamore Creek riparian soils. Data presented are untransformed, but statistical analyses were carried out on log-transformed data. Error bars are ±2 S.E. Letters indicate significant differences among groups.

$B_{\mathrm{NH_4}}$ (multiple-$r^2 = 0.19$, $p < 0.001$). In contrast, multiple-regression using organic matter, $R_{\mathrm{NO_3}}$, and moisture explained a large portion of the variance in $B_{\mathrm{NO_3}}$ (multiple-$r^2 = 0.65$, $p < 0.001$; Figure 5). $\mathrm{B_{SRP}}$ was best predicted by R_{SRP}, with soil organic matter also significantly correlated (multiple-$r^2 = 0.82$).

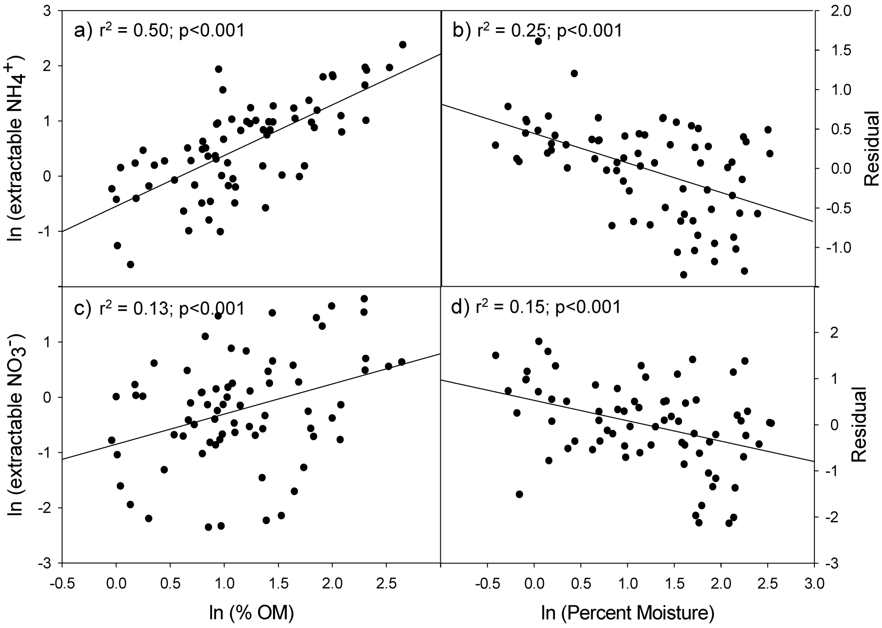

Figure 2. Relationships between soil organic matter and moisture and extractable N pools. (a) and (c) depict relationships between organic matter and extractable ammonium and nitrate, respectively. (b) and (d) depict relationships between soil moisture and residuals from (a) and (c), respectively.

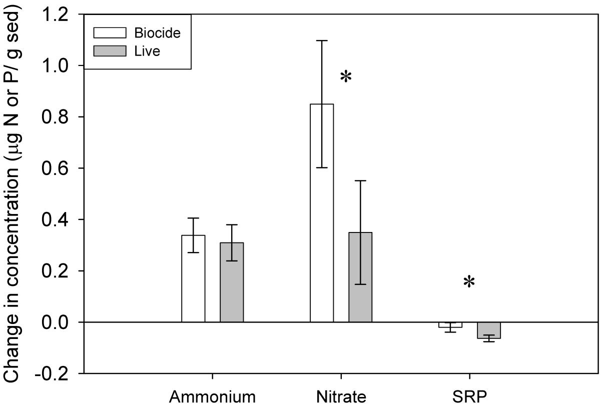

Figure 3. Effects of exposure to sterilized and live soils on water chemistry. Net changes in concentration of all three solutes were significantly different from zero for both biocide and live treatments. Asterisks indicate that live samples were significantly different from biocide-treated samples. Error bars are ±2 S.E.

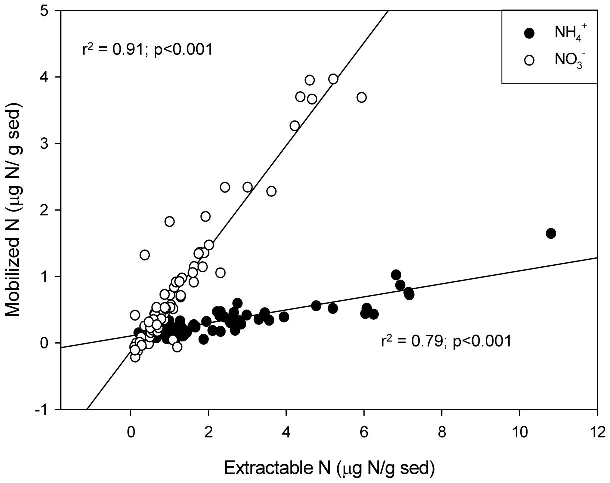

Figure 4. Relationship between extractable pool size and mobilization of NO_3^- ($y = 0.77x - 0.13$) and NH_4^+ ($y = 0.11x + 0.05$) from riparian soils.

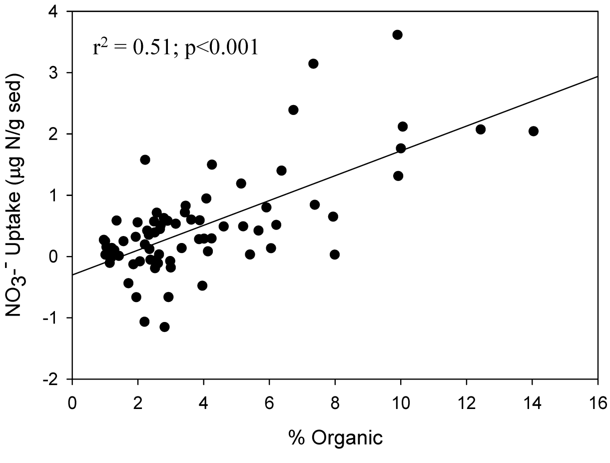

Figure 5. Relationship between soil organic matter and nitrate processing in Sycamore Creek riparian soils.

B_{NO_3} and B_{NH_4} varied among sampling locations. Although B_{NH4} was not statistically different from zero when all samples were considered, B_{NH_4} did vary significantly with depth ($F_{1,48} = 26.66$, $p < 0.001$; Figure 6(a)). Shallow soils were net sources of ammonium (i.e. live soils had higher ammonium than biocide-

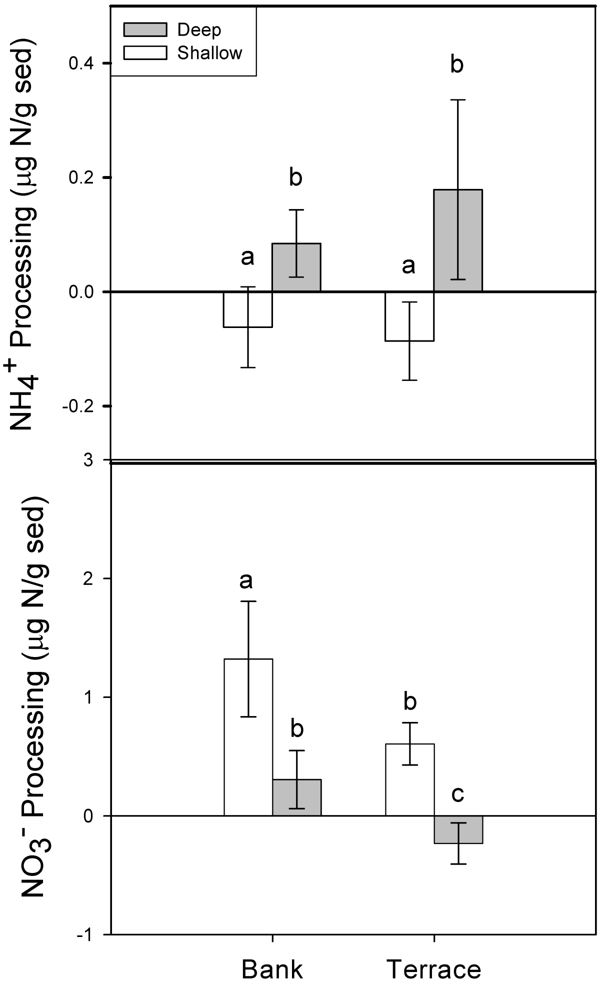

Figure 6. Spatial pattern of (a) nitrate processing and (b) ammonium processing in riparian soils. Positive processing indicates net decreases in concentration. Data presented are untransformed, but statistical analyses were carried out on log-transformed data. Error bars are ±2 S.E. Letters indicate significant differences among groups.

treated soils), while shallow soils were net sinks. Biotic processing resulted in large decreases in nitrate overall, but this processing also exhibited spatial variability (Figure 6(b)). Bank soils had higher rates of B_{NO_3} than did terrace soils ($F_{1,48} = 42.17$, $p < 0.001$) and shallow soils had higher rates than did deep soils ($F_{1,48} = 19.25$, $p < 0.001$). Deep terrace soils were actually net sources of nitrate. B_{SRP} varied with depth ($r^2 = 0.41$, $F_{1,48} = 15.63$, $p < 0.001$), with shallow soils acting as larger sinks of SRP than deep soils.

Importance of nutrient sources

Under the conditions prevailing during the preceding dry period and imposed by this experiment, we estimate that 25% (±5%; 95% C.I.) of DIN in riparian groundwater during flash flood is delivered by incoming floodwaters, with the remainder being re-mobilized from riparian soils. If flood delivery is compared to extractable DIN pools, which are presumably available to both plants and micro-organisms, then only 7% (±1%; 95% C.I.) of available DIN is flood-derived. In contrast, nearly all of SRP in riparian groundwater is flood-delivered, since most sediments were a sink, rather than a source, of P.

Discussion

Our results suggest that interactions between riparian groundwater and soils during inundation are element specific. Both NO_3^- and NH_4^+ were released from sediments into the interstitial water. Further, the resulting increases were large relative to concentrations of incoming floodwaters, in agreement with prediction 2(a). Over the course of the 4 h incubation, biotic processing of N was approximately equal to the flood-delivered nutrient flux, in agreement with prediction 2(b). We therefore accept hypothesis 2 (re-mobilization as a source of nutrients to riparian groundwater) with respect to N. However, since floodwaters probably enter the riparian zone over the course of seconds to minutes, rather than hours, some proportion of floodwater N undoubtedly does enter the riparian soil N cycle, and we cannot reject floodwater nutrients as a potential source to riparian groundwaters. In contrast to the behavior of NH_4^+ and NO_3^-, SRP was lost from the water column to sediments, in agreement with prediction 1(a). We therefore reject hypothesis 2 with respect to P, and infer that flood-delivery is the primary source of P in riparian groundwater during flash floods.

Re-mobilization and processing of nutrients by riparian soils varied greatly in space. In some cases, we can attribute that variability in process to variation in soil characteristics. For example, extractable and re-mobilized ammonium were correlated with organic matter, which varied most strongly with depth. Thus, extractable and re-mobilized ammonium also varied with depth. We observed similar patterns in B_{NO_3} (Figures 5 and 6b). In other cases, sampling position accounted for more of the observed variance than did measured soil characteristics (e.g., extractable NO_3^-, B_{NH_4}). This suggests that these variables respond to some unmeasured factor (e.g., soil C:N) that varied at the scale of our sampling regime (meters to hundreds of meters). Such unexplained spatial variation complicates any efforts to scale up the results of our experiment.

Fate of re-mobilized nutrients

The results of this study suggest that re-mobilization is an important source of N but not P to Sycamore Creek riparian groundwater during flash floods. However, we caution against the inference that floods are unimportant sources of N to longer-term riparian soil pools. The implications of repeated saturation, drainage, and desiccation of these soils for long-term accumulation or loss of nutrients depends on the subsequent fate of both flood-delivered and re-mobilized materials. Nutrients in riparian groundwater may be incorporated into living biomass or sorbed to soil sediments (i.e., stored internally and potentially recycled) or immediately lost to the system via gaseous or advective transport. The partitioning of N among these potential fates will determine whether floods act as long-term sinks or sources of material to riparian soils, and the nature of interactions with other fluvial subsystems.

While we did not measure specific processes, limited evidence suggests that re-incorporation into microbial biomass was the major vector of loss of N from the water column. Denitrification potentials of riparian bank soils in Sycamore Creek have been measured at 60–80 ng N g^{-1} h^{-1} (Holmes et al. 1996). We observed declines of 300 ng NO_3-N g^{-1} h^{-1} in shallow bank soils. Maximum denitrification rates could therefore account for only 20–25% of observed declines in nitrate. Given the high levels of nutrient availability and likely low microbial biomass prior to re-wetting, net microbial growth rates are likely to be high following re-wetting, and likely accounted for a large proportion of observed N declines.

While this study addressed some aspects of heterogeneity in riparian sediments, a potential shortcoming of this experiment is that flowpaths through riparian soils were not reproduced. During actual flood events, materials re-mobilized from one parcel of soil might be processed by another parcel, or exported via bank drainage. In addition, duration of saturation is likely to vary among soils as a function of their hydraulic conductivity. Depending on the arrangement of these soils relative to the direction of flow and the residence time of floodwaters in riparian soils, this could result in greater or lesser fluxes of dissolved nutrients to or from riparian soils.

Flood-drought cycles and sediment nutrient dynamics

The results of this study agree generally with several recent studies that indicate that re-mobilization is an important aspect of nutrient cycling in seasonally or episodically saturated sediments. Baker et al (2000) found that the region of seasonal saturation (ROSS) of a New Mexico stream was a major source of the organic carbon that fueled hyporheic metabolism in receiving systems. In ephemeral lakes in Australia, re-mobilization contributes to observed increases in N and P concentrations, although the relative importance of riverine inputs and re-mobilization depends on regulation regime (Scholz et al. 2002). Qiu and McComb (1996) found substantial release of ammonium from desiccated wetland sediments, while Mitchell and Baldwin (1999) observed no flush of N from dried reservoir sediments. Studies of P dynamics after re-wetting of dried sediments have also shown varied results, with sediments acting as sources (e.g., Qiu and McComb 1994, Mitchell and Baldwin 1998) or sinks (e.g., Tockner et al. 2002). Our findings are contrast with these studies in that abiotic processes were net sinks of P. However, our results are consistent with previous observations at Sycamore Creek, where SRP concentrations in the riparian zone remained low during floods despite elevated concentrations of P in surface waters (Table 1).

In a variety of soils and sediments, the importance of re-mobilization varies as a function of the length of the unsaturated interval and degree of desiccation (Birch 1960, West et al. 1992). In general, the observed pattern is for available nutrients to initially decline with decreased moisture, then increase as soils and sediments become desiccated (Baldwin and Mitchell 2000), although Mitchell and Baldwin (1998) observed decreases in P release as sediments became more desiccated. Microbial biomass typically declines linearly with decreases in sediment moisture (Vangestel et al. 1992, West et al. 1992), and mortality of microbes and decomposition of osmoregulatory molecules increases availability of nutrients and organic matter upon re-wetting (Fierer and Schimel 2003). We are unable to evaluate directly the magnitude of drought effects in this experiment, but our results suggest that they may be considerable. The negative correlation of soil moisture with re-mobilization of nitrate, ammonium, and SRP (Figure 4 (b and d)) in our experiment suggests that the length of time since the last flood may influence the types and amounts of nutrients re-mobilized from riparian soils.

As Tockner et al. (2000) argue, flow variability is not limited to dichotomous bankfull and baseflow conditions. More frequent ‘flow pulses’ have important influences on system connectivity. Such pulses may also result in suspension and re-mobilization of materials at a wide range of spatial and temporal scales, from diel water table fluctuations (Valett 1993) to seasonal expansion and contraction of the stream network (Stanley et al. 1997). In arid systems, where moist conditions are rare and desiccation rapid, such pulses may play an important role in biogeochemical cycles.

Conclusion

Delivery of dissolved N to riparian soils by floodwaters represents a small proportion of the DIN pools made available to plants and microbes by re-wetting and subsequent re-mobilization. Microbial processes stimulated by re-wetting do influence concentrations of nitrate over relatively brief time periods, but at rates that are probably not sufficient to remove incoming N. Inorganic phosphorus is rapidly sorbed to riparian sediments, so that whatever P is observed in riparian groundwater during floods is likely to be flood-derived. Thus, floods do deliver nutrients to the riparian zone in amounts that may be important over annual to decadal time scales; however, our data do not allow us to determine the importance of flood delivery to riparian soil N and P pools with any confidence

Predicted shifts in climate are expected to influence precipitation regimes worldwide (IPCC 2001). Climate models predict increased frequency of extreme hydrologic conditions, suggesting that flood/drought cycles of both regulated and unregulated streams and rivers are likely to increase in frequency and intensity. Evidence from this and other studies suggests that both inundation and desiccation influence nutrient dynamics of riparian soils. Thus expected changes in flood frequency and drought duration may be expected to influence not only availability of colonizing sites and water to riparian plants of arid lands, but also the biogeochemical processes that determine nutrient availability in riparian sediments.

Acknowledgements

We thank Darrel Jenerette, John Cetin, and Susan Ford for their assistance in the field; and David Lewis, Cathy Kochert, Tom Colella, and Jennifer Edmonds for their help with laboratory analyses. Conversations with Jill Welter and James Watts improved this study. Stuart Fisher and two anonymous reviewers provided suggestions that substantially improved this manuscript. Research was supported by a grant from NSF (DEB 0075650) to SGF. This material is based upon work supported under a National Science Foundation Graduate Research Fellowship to JBH.

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