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Biotic versus hydrologic control over seasonal nitrate leaching in a floodplain forest

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Abstract. Strong seasonal increases in aquatic (stream, ground and hyporheic water) nitrate have been observed in a variety of ecosystems. In most cases, changes in hydrological and vegetative activity occur contemporaneously, making it difficult to determine whether soil leaching is being driven by increases in the availability of leachable N or is simply due to flushing of N that has accumulated over longer periods. Three studies were conducted to better determine controls on soil nitrate leaching in a near-pristine temperate floodplain ecosystem receiving large N inputs via N-fixation by red alder: 1) an artificial rainfall experiment was conducted to estimate N-leaching potential during the summer, when plant uptake is high and new inputs of organic matter are low; 2) soil solution, groundwater and surface water were sampled during a major autumn storm to document exchanges at the seasonal transition, when plant uptake is low and inputs of senescent organic matter are high; and 3) monthly samples of soil and aquatic nitrogen were collected in 1997 and 1998 to document seasonal patterns of N exchanges. Collectively, these studies demonstrate the importance of hydrologic factors in controlling N flux. Nitrate was rapidly leached from soils during actual and simulated rainstorms. Two pathways of nitrate leaching were identified. Localized flooding and direct leaching of streamside soils into surface waters contributed to high solute concentrations in peak flows. Nitrate that leached into interstitial waters was subject to various factors that could delay or reduce its delivery to surface waters. Greater residence time may increase the influence of this component of stormflow on ecosystem productivity. While soil nitrate pools were rapidly depleted during rainstorms, accumulation of soil nitrate occurred over summer dry periods. Large differences in soil and aquatic nitrate concentrations between two years with contrasting rainfall highlight the potential for inter-annual hydrologic variability to affect ecosystem nutrient cycling.

Introduction

Temporal patterns in nitrogen (N) leaching from riparian soils into surface and subsurface (hyporheic/ground) waters are of interest because they constitute fluxes of a nutrient that is often limiting to terrestrial and aquatic production, and because of concerns over negative effects of increased atmospheric N deposition on terrestrial and aquatic ecosystems (Cole 1992; Fenn et al. 1998). Accumulation of labile N in excess of physical and biological retention capacity often leads to greatly increased nitrate leaching, which tends to occur at seasonal boundaries. Both fall and spring stream water nitrate pulses are observed (Likens et al. 1977; Weaver and Forcella

1979; Peterson and Rolfe (1982, 1985); Hill 1986; Foster et al. 1989; Brooks et al. 1998), with fall increases associated with increased precipitation and spring pulses reported mostly from areas with snowmelt-dominated hydrologies. These pulses have been attributed to different causes, including increased decomposition following soil thawing (Peterson and Rolfe 1985), flushing of nitrate deposited into snowpack (Groffman et al. 1993; Brooks et al. 1998), and increased inputs of labile organic matter from autumn litterfall combined with reduced plant uptake (Likens et al. 1977; Weaver and Forcella 1979). In almost all cases, contemporaneous seasonal changes in biotic activity and hydrology make it difficult to determine whether nitrate leaching is being driven by seasonal increases in nitrate availability or is simply due to flushing of N that has accumulated over longer periods of time.

Increases in the size of the nitrate pool can occur when mineral N release proceeds at a greater rate than consumption through uptake, adsorption and immobilization or loss via leaching and denitrification. This may be a chronic limitation of consumption by some other resource, or may be temporarily due to a lagged response to changes in resource availability. Even when there is an accumulation of soil nitrate, leaching can only occur if there is sufficient movement of water to provide transport. In many areas, seasonal transitions are accompanied by changes in precipitation or storage of precipitation in snowpack. The timing of rainfall and snowmelt relative to release of mineral N determines the magnitude of the leaching response. Two sets of circumstances could result in increased nitrate outputs. In the first, leaching is limited by the availability of nitrate, which is released from new organic inputs in the fall when vegetative uptake is reduced. Alternatively, soil nitrate could accumulate over longer time periods, and leaching be limited by the availability of transport. These two scenarios differ fundamentally in their underlying biogeochemistry and in the predicted response to environmental variation, such as might accrue from interannual variation or long-term changes in precipitation.

Three studies were undertaken to better determine controls on terrestrial-aquatic nitrate transfers in a pristine temperate floodplain ecosystem. First, monthly samples of soil and aquatic inorganic N were collected during the summer and fall of 1997 and 1998 to document seasonal patterns. Very different rainfall patterns during these two years permitted comparison of seasonal changes in soil and aquatic nitrate and ammonium pools under contrasting hydrological conditions. Two additional studies were conducted to evaluate mechanisms controlling these patterns. An artificial rainfall experiment was conducted to estimate the leaching potential during the summer, when vegetative uptake potential is high and new organic inputs are low. Second, soil solution and subsurface and surface water were intensively sampled during a major autumn storm to document N exchanges when vegetative uptake is low and N inputs from decaying vegetative matter are high.

While nitrate dominates leaching in forests with excess N (Cole 1992; Fenn et al. 1998), dissolved organic matter is a more important vector for N and phosphorus loss in N-limited old-growth forests in the Pacific Northwest (Sollins et al. 1980; Triska et al. 1984). In order to better understand differences in the movement of nitrate and organic solutes, dissolved organic carbon (DOC) concentrations were also measured in storm samples. Differences in nitrate and DOC concentrations between surface water, subsurface water and soil solution were used to make inferences about probable flowpaths of solutes in rainwater.

Methods

Site description

This research was conducted in the Queets River watershed (watershed area 1157 km^2), on the western edge of Washington State's Olympic Peninsula, USA. The study site and almost all the upstream watershed lie within Olympic National Park, and are largely free of human impacts. The area has dry summers, but very high precipitation ($\sim$ 3 m/yr), falling mostly as rain during fall and winter. Rapid channel migration has resulted in a complex network of side and abandoned river channels separating forest patches at different stages of soil and vegetative development. The coarse sediments that form river bars and terraces are hydrologically highly conductive, creating an extensive hyporheic zone in the floodplain. The study site is on a 1-km long alluvial reach, approximately 25 km east of the Pacific Ocean. The floodplain is 1 km wide at this point, and is subject to frequent fluvial disturbance. The more-developed soils on older terraces are classified as Entisols, belonging to the Huel series, and consist of moderately well-drained loamy fine sand with a weakly developed A and one or more C horizons. Soils of younger terraces are composed of recent fluvial deposits.

The river is oligotrophic, and a previous study (Fevold 1998) established that aquatic productivity is often N-limited. The terrestrial environment, in contrast, is highly productive and N-rich (Bormann et al. 1994). Recently deposited river bars are colonized – usually within 10 years – by red alder (*Alnus rubra*), a vigorous N-fixing tree species (Binkley et al. 1994) which dominates the canopy for the following 50–70 years. The mature forest is dominated by Sitka spruce *(Picea sitchensis)* (Franklin and Dyrness 1973). Preliminary data collected from this site documented strong seasonal variation in hyporheic nitrate beneath red alder stands. Although organic N made up approximately 30% percent of the total N in hyporheic water N in the summer, the autumn N peak consisted almost entirely of nitrate.

Experimental design

In the three studies described here, simultaneous measurements of soil and aquatic inorganic N made at several points in time were used to infer terrestrial-aquatic transfers (Table 1). Nitrate and ammonium were measured in surface water (monthly sampling, storm sampling), subsurface water (all studies) and soil (monthly sampling, storm sampling). In addition, DOC was measured during the autumn storm sampling to help distinguish differences in the movement of organic solutes and inorganic N. Monthly sampling was conducted during two periods: September 1997 through January 1998, and August 1998 through December 1998 in order to document changes in inorganic N accompanying the transition from summer to autumn. Shorter-term measurements to document within-season patterns were conducted in summer (artificial rainfall experiment; September 1–5, 1998) and autumn (storm sampling November 12–17, 1998),

Homogenous ″patches″, determined on the basis of similarity in vegetation age and soil development, were selected as sampling sites. Ages for alder patches were determined by measuring growth rings from cores obtained from ten stand-dominant trees at each site. A total of 7 sites were used (Figure 1). The river is the source of all subsurface water beneath the 3-year willow-alder, 10-year alder, 24-year alder and 33-year alder sites (Clinton et al. 2002). Based on limited hydraulic head data, the 29-year alder, 63-year alder and spruce sites receive a combination of groundwater from a gently sloping terrace and hyporheic inputs, with proportions dependent on river stage and rainfall. The 33-year alder site was only used for the artificial rainfall experiment. The other sites were used for monthly sampling and autumn storm sampling with the following exceptions: the alder-willow site could not be sampled due to inundation during part of the autumn storm; the 24-year alder site was used only for storm sampling.

Sample collection and processing

Surface water samples were collected from the two sites that directly border on stream channels: the 29-year alder site on Pebble Creek, and the 10-year alder site near the main channel of the Queets River. Surface water samples were collected by grab sampling 15 cm beneath the water surface, and were collected in triplicate to reduce sampling variability. Subsurface water was sampled from two (one at artificial rainfall site) PVC wells installed in each site. Wells were installed to a depth of 30–50 cm below the summer low flow water table surface, which was between 2.8 to 4.3 m below the soil surface at all sites. Subsurface samples were collected using a battery-driven peristaltic pump and polyethylene tubing.

Soil inorganic N was measured in water extracts of soil samples obtained during regular monthly sampling. Soil solution samples were obtained from tension lysimeters during the storm sampling and the artificial rainfall experiment. Soil samples were collected from 0–15 cm depth with a 5 cm diameter steel tube, and were extracted in the laboratory for two hours in deionized water with agitation. Soil extracts were filtered through Whatman No. 42 filters (2.5 μm pore size). Deionized water was used for the extractant to provide a more realistic estimate of the inorganic nutrients likely to be leached by rainwater than by extractants (e.g., KCl) that displace ions from exchange sites. Three lysimeters (Soilmoisture Equipment Corp. Model 1900) were installed through augered holes to 60 cm depth. The lysimeter holes were augered at a 30° angle from vertical to minimize soil disturbance, and were allowed to equilibrate for three months prior to first sampling. Soil solution samples were collected at a tension of 0.02 MPa.

All field-collected water samples were filtered through Whatman GFF filters (0.45 μm pore size) into acid-washed polyethylene bottles. Water samples were

Table 1. Overview of studies: storm sampling sites are the same as monthly sampling site minus 3-year willow-alder site and with addition of 24-year alder site

	Sites	Sampling interval	Sampling period	Measurements				
				Rainfall	Discharge	Soil	Subsurface water	Surface water
monthly sampling	3-year willow-alder, 10-year alder, 29-year alder, 63-year alder, mature spruce	monthly	Sep–Jan 1997, Aug–Dec 1998	tipping bucket rain gauge, WRCC long-term data		inorganic N (water extract)	inorganic N	inorganic N
artificial rainfall experiment	33-year alder	2–6 hour	Sep 1–5, 1998	flowmeter			inorganic N, electrical conductivity	
storm sampling	10-year alder, 24-year alder, 29-year alder, 63-year alder, mature spruce	12–24 hour	Nov. 12–17, 1998	tipping bucket rain gauge	USGS gauging station	DOC, inorganic N (lysimetry)	DOC and inorganic N	DOC and inorganic N

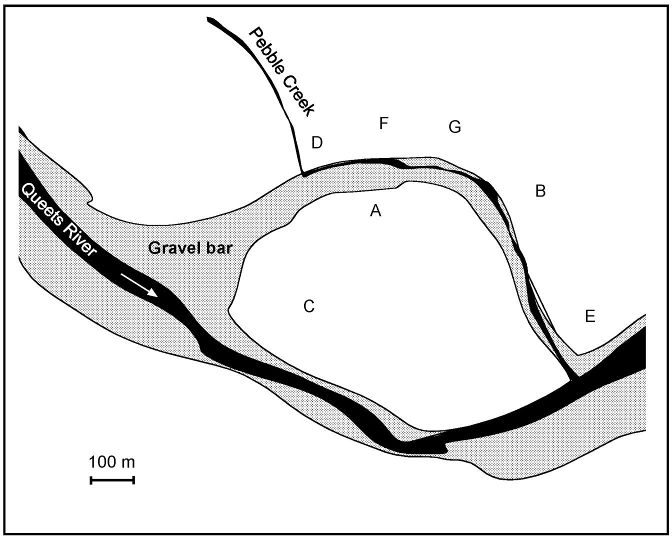

Figure 1. Study area. Study site locations are indicated by letters: A 3-year willow-alder, B 10-year alder, C 24-year alder, D 29-year alder, E 33-year alder, F 63-year alder, G mature spruce.

chilled on ice within two hours of collection and were either analyzed or frozen within 48 hours. All samples were analyzed colorimetrically for nitrate and ammonium (Mulvaney 1996) on a Spectronic 21 spectrophotometer. DOC was measured with a high-temperature combustion Shimadzu total organic carbon analyzer (Qian and Mopper 1996).

Hydrologic measurements

Rainfall was measured on-site with a tipping bucket rain gauge (Texas Electronics Model 525) attached to a datalogger. Correlation analysis of 1997–1999 rain gauge data with long-term data obtained from a NOAA weather station on the Clearwater River (Western Regional Climate Center, Daily precipitation and temperature data 1930–1999) was used to estimate long-term rainfall for the Queets. River discharge data were obtained from a USGS gauging station located approximately 12 km downstream of the study site (United States Geologic Service, Stream flow data 1997–1998).

Artificial rainfall experiment

Nine spray heads, each with a 1.5 m circular or semi-circular application radius, were arrayed across a 10-m^2 plot in the 33-year alder site for the artificial rainfall experiment. The experimental plot was irrigated with river water at a rate of 2.5 cm/hr for ten hours the first day and for six hours on the second and third days of the experiment. This falls with the range of rainfall encountered during fall and

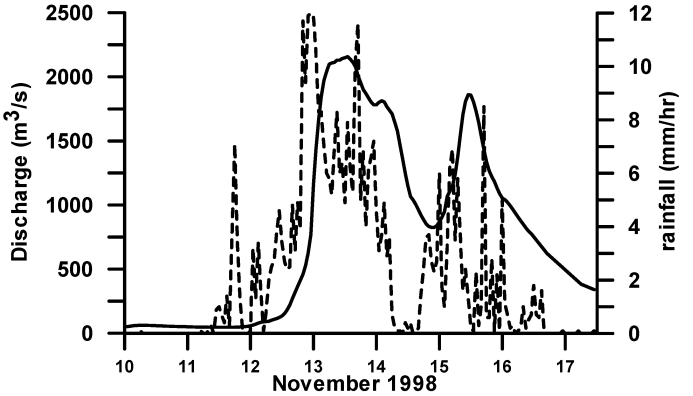

Figure 2. Queets River discharge — and rainfall - - - during November 12–17, 1998 storm.

winter storms; the Queets averages 10.3 days per year of precipitation in excess of 20 cm. The application rate was measured through a flow meter located at the pump inlet. Irrigation water was labeled with a sodium chloride tracer (60 mg NaCl/L H_20) sufficient to raise the electrical conductivity of irrigation water from a background conductivity of 64 to $\sim$ 120 μS/cm. Conductivity was periodically measured in spray water and periodically adjusted to ensure even tracer concentration in irrigation water. Conditions were dry; only 12.4 cm of precipitation were recorded during the preceding three months. A plastic tarp was placed over the well and surrounding area that had been disturbed during well installation to minimize the influence of the well installation on infiltration and solute chemistry. Subsurface samples were collected and conductivity measured from the well at two to four hour intervals during irrigation, and at six to eight hour intervals at night and on the fourth and fifth days of the experiment.

Intensive storm sampling

Five sites were intensively sampled during the first major storm in the autumn of 1998. The storm began on November 11, and was sustained on November 12–13, with accumulated precipitation of 28 cm. Rainfall on the following three days increased the total storm precipitation to 41 cm (Figure 2). Queets River discharge rose from 47 m^3/sec on November 11 to 2,158 m^3/sec on November 13, declined to 824 m^3/sec on the 14th and then peaked again at 1,860 m^3/sec on the 15th. Surface water, subsurface water and soil solution was collected every 12 hours, when possible, during the first four days of sampling. However, high river levels prevented access to some of the sites for periods up to 18 hours. A final set of lysimeter samples, collected seven days after the onset of the storm, contained samples collected over a two-day period.

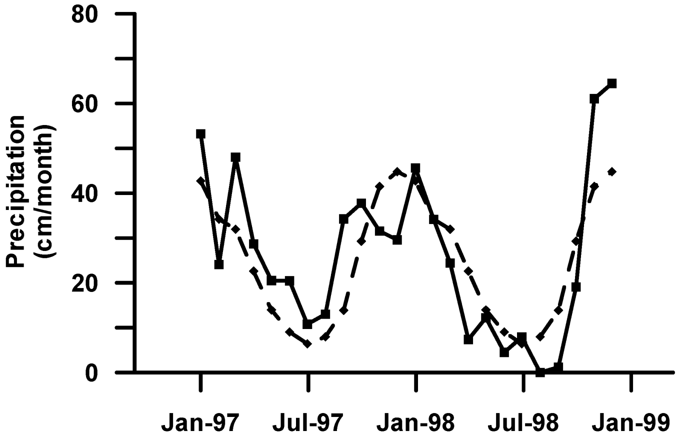

Figure 3. 1997–98 monthly rainfall –■– and long-term monthly average rainfall –•– for Queets River valley. Long-term averages are based on a 69-year record.

Results and discussion

Monthly sampling

1997 was an unusually wet year and 1998 was very dry. In 1997, spring, summer and fall rainfall were all > 30% above the long-term average (Figure 3). Beginning in November 1997, monthly rainfall fell below the long-term average. By September 1998, Queets River discharge was approximately one-third of the long-term average. During these two years there were strong contrasts in soil and aquatic N. In 1998, summer soil nitrate (Figure 4) was much higher at all sites than in 1997. The differences were greatest in the 29- and 63-year alder sites, where September nitrate was 30 and 40 times higher in 1998 than in 1997. The reverse was true of surface and subsurface water nutrients. Aquatic nitrate was lower in Summer 1998 than Summer 1997 at all but the two youngest sites (Figure 5). The most likely explanation for this pattern is that soil nitrate was flushed out in Summer 1997. This interpretation also explains the absence of a 1997 autumn peak in aquatic nitrate; surface and subsurface nitrate concentrations were highest in September and changed little through the fall and winter. The absence of a Fall 1997 peak in surface water nitrate was confirmed in data collected from the Hoh River, a watershed approximately 25 km to the north of the Queets. There, 1997 was one of only two years in a 14-year record of weekly stream water samples in which a fall peak in nitrate was not detected (Edmonds et al. 1998). Soil nitrate concentrations did increase in Fall 1997, as would be expected if inputs from senescing vegetation were controlling leaching patterns, but these inputs were not of sufficient magnitude to be detected in aquatic samples.

Although some nitrate removal has been measured from water moving through hyporheic sediments in the study area (Ritzenthaler 1998), the influence of denitri-

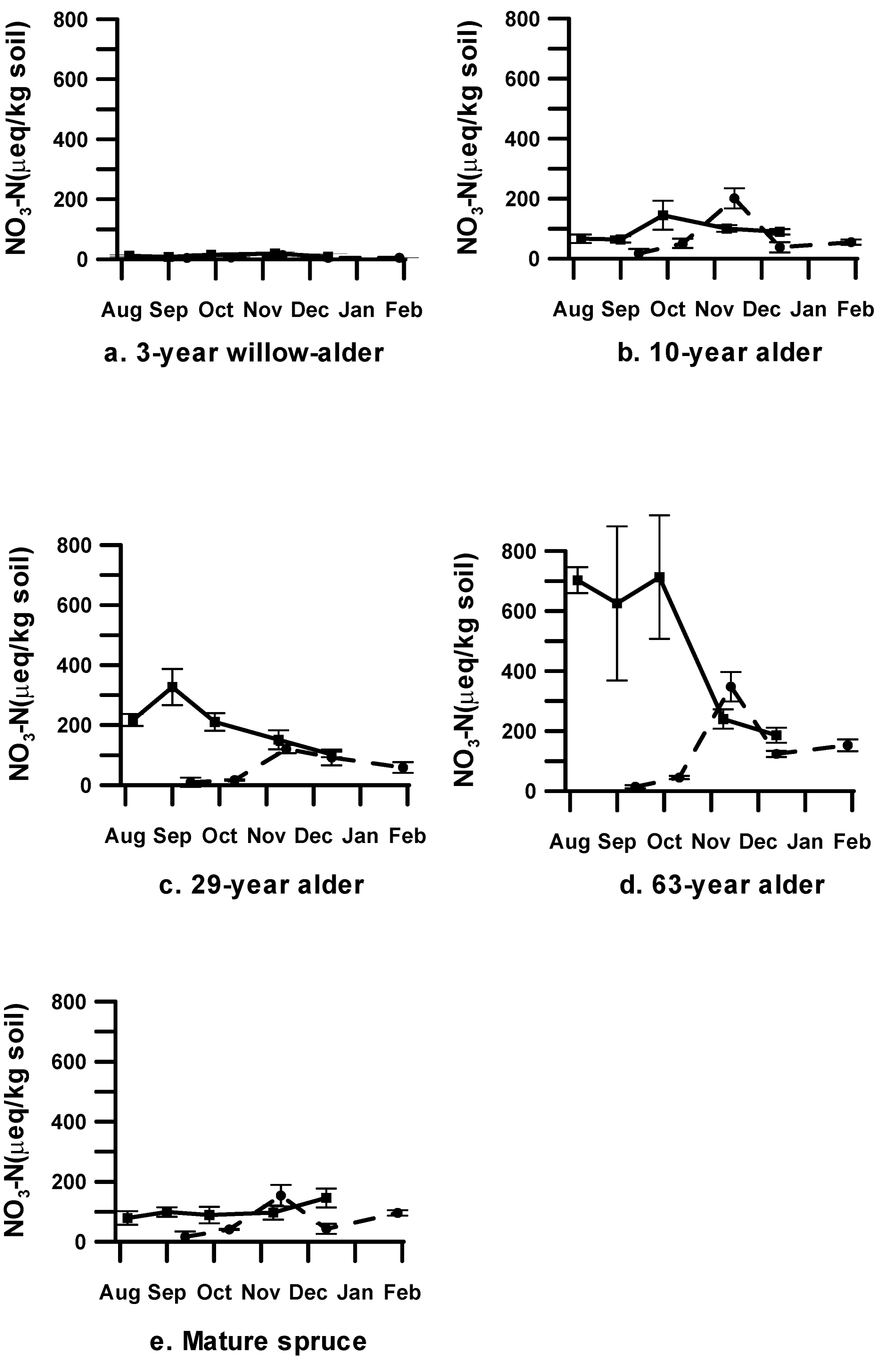

Figure 4. 1997–98 monthly sampling results: NO_3^-N concentrations in water extracts from soil samples (0–15 cm depth) during 1997 –■– and 1998 –●–. Bars indicate standard errors.

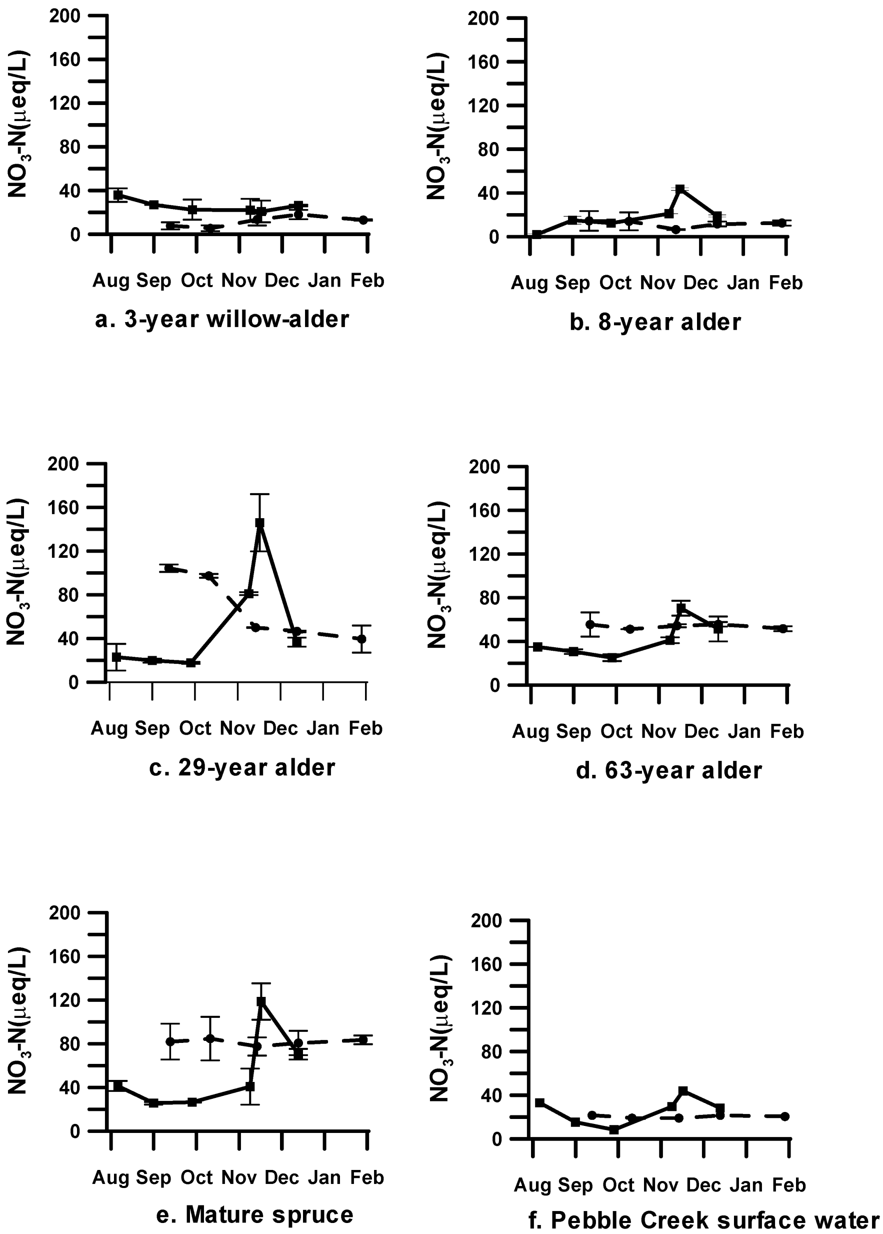

Figure 5. 1997–98 monthly sampling results: NO_3^-N concentrations in (a–e) subsurface water under riparian sites and (f) surface water in Pebble Creek during 1997 –■– and 1998 –•–. Bars indicate standard errors.

fication on nitrate fluxes is minimized by coarse sediments. Except for localized areas of low hydraulic conductivity, ground/hyporheic water remains well-oxygenated throughout the year (Clinton et al. 2002). Silt and clay in all soils are below concentrations found necessary to support denitrification in other floodplain soils (Pinay et al. 2000).

Soil nitrate increases in response to soil drying are frequently reported from agricultural systems receiving N-fertilization (Giambiagi et al. 1993; Stout et al. 2000). There was ample visual evidence that red alders were drought-affected, including premature leaf abscission (by up to one month) in younger patches and in stands located further from the river. Although 1998 conditions may represent an extreme case, summer drought is an important influence on vegetative activity in the Pacific Northwest (Weaver and Forcella 1979), with implications for N cycling. Young soils on river bars are especially vulnerable to drought as they are usually coarse textured and have little organic matter and as a result have little water-holding capacity.

Apparent increases in soil nitrate with stand age in alder patches (Figure 4) are consistent with continued high rates of N-fixation in older alder stands. Despite high energetic costs, N-fixation was either unaffected or stimulated by high N availability in a number of studies of N-fixation red alder (reviewed in Binkley). Continued high rates of N-fixation were also evidenced in highly nodulated roots of older alder trees uncovered during soil pit excavation for a related study at the same site (Bechtold et al. submitted). In that study, rapid accumulation of soil C and N during the first 20–25 years of patch development plateaued at about the same time as soil nitrate concentrations increased, suggesting nitrate leaching was associated with exhaustion of a limited N-retention capacity.

Artificial rainfall experiment

Irrigation stimulated large increases in subsurface water nitrate concentrations (Figure 6). Elevated nitrate concentrations were detected in the first pulse of irrigation water to reach subsurface water, ten hours after the initiation of irrigation. Nitrate concentrations rose from a background concentration of 2 to 276 μeq NO_3^-N/L by hour 12 of the first day. This nitrate concentration was substantially higher than measured in autumn subsurface samples in either 1997 or 1998. Nitrate achieved similarly high concentrations on days two and three of the experiment, indicating a substantial pool of readily leachable nitrate was present in the soil. Nitrate concentrations were well correlated with electrical conductivity ($R^2 = 0.70$, $p < 0.01$, from a linear regression) suggesting that increased subsurface concentrations were dominated by simple leaching of an existing nitrate pool. Initial ammonium concentrations were 2.4 μeq/L and never exceeded 3.8 μeq/L (Figure 6). Ammonium concentrations were not statistically correlated with conductivity, which partly reflects error associated with measurements made near the lower detection limit for ammonium. Concentrations of nitrate and ammonium in irrigation water were very low (1.0 and < 1.0 μeq/L, respectively), and were not a significant source of N.

One possible explanation for the nitrate pulse is that the sodium chloride tracer displaced large amounts of inorganic N from exchange sites. However, displacement of sufficient ammonium from cation exchange sites to account for the mea-

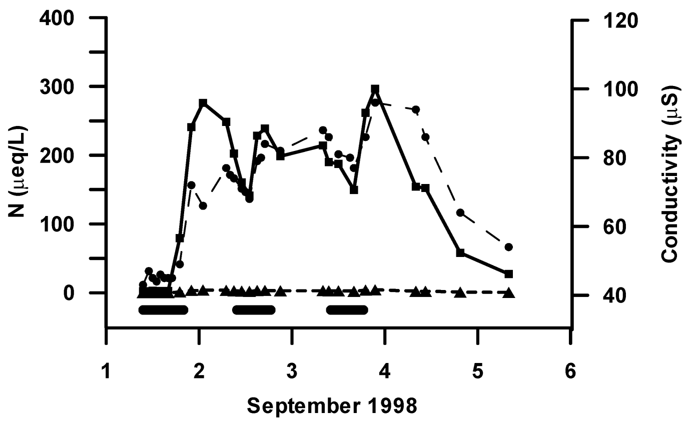

Figure 6. Artificial rainfall experiment results: NO_3^-N, –■– NH_4^-N - - - and electrical conductivity –●– in subsurface water. Irrigation period — is shown at bottom of figure.

sured nitrate pulse, even with extremely rapid nitrification, should have resulted in measurable ammonium increases in subsurface water, and ammonium concentrations remained extremely low in soil solution and subsurface water throughout the experiment. Unweathered, temperate soils do not usually have significant anion exchange capacity (Brady and Weil 1999). Field and laboratory studies using a NaCl tracer demonstrated no chloride ion adsorption and no influence of NaCl tracer on nitrate adsorption in similar soils in red alder stands even at 2.5 times the concentration used here (Johnson et al. 1986).

Electrical conductivity, nitrate and ammonium all declined to near-background levels two days after cessation of irrigation. This indicates that there are substantial subsurface flows beneath the study site, and that the high well concentrations were caused by continuous inputs of nitrate to subsurface water, and were not due to a pooling of leached nitrate in subsurface water. The lag in response of subsurface nitrate and conductivity relative to irrigation period indicates a travel time of about 10 hours for the 3.5 m distance from the soil surface to the saturated zone in dry conditions during the first day of the experiment, and five to eight hours during the two following days of irrigation at above field capacity soil moisture. A percolation rate of 30–60 cm/hr is rapid, but not unexpected in well-drained soils (Brady and Weil 1999).

Intensive storm sampling

Intensive sampling of the first major autumn storm of 1998 demonstrated rapid depletion of soil nitrate. Two distinct pathways of terrestrial-aquatic transfer were inferred. Peak flows were dominated by high DOC concentrations, which appeared to derive from direct transfers from soils to surface water. This was followed by a slowly attenuating period of nitrate movement through subsurface pathways into surface waters.

Soil response to the rainstorm at all alder sites was similar (Figures 7a–7d). Soil solution nitrate concentrations were initially high (300–600 μeq NO_3^-N/L), but were quickly depleted. Concentrations decreased to below 100 μeq NO_3^-N/L by the third day of the storm at all but the 24-year alder site. In contrast, soil solution nitrate from the conifer site was initially low and did not change (Figure 7e). Increases in subsurface nitrate at the 10-, 24- and 29-year alder sites directly follow soil nitrate depletion, indicating efficient terrestrial-aquatic transfer. Subsurface nitrate increases were first detected in samples collected on November 12 (the second day of the storm) and peaked on November 14 or 15. Subsurface nitrate remained substantially elevated above background concentrations through November 17 (105, 145,145 post-storm μeq NO_3^-N/L versus 41, 76, 81 pre-storm μeq NO_3^-N/L for the 10-, 24-, and 29-year alder sites, respectively). Less direct coupling of soil nitrate depletion and increases in subsurface nitrate were observed in samples from the 63-year alder and the mature spruce sites. Although a large decrease in soil nitrate was measured at the 63-year alder site, only small increases in subsurface nitrate were detected (Figure 7d). At the mature spruce site, subsurface nitrate increased even though soil nitrate leaching was negligible. These sites sit on a high, slightly sloping terrace along the floodplain edge, and have deep soils (> 1 m to river cobbles). Landscape position suggests they receive groundwater from the upland forest, in contrast to the other sites where most water is hyporheic.

In contrast to nitrate, little DOC was leached to subsurface water (Figure 7a–7e). Substantial amounts of DOC were measured in soil solution at all sites, and soil solution DOC levels remained high throughout the period sampled. Subsurface DOC concentrations were much lower, and never increased above 1 mg/L. This indicates that C was efficiently being retained by the soil, either through adsorption or rapid metabolism. The one exception to this pattern was the 10-year alder site (Figure 7a), where DOC increased to 2.6 mg/L. However, this increase may reflect intrusion of surface water from a nearby overflow channel rather than soil inputs.

Surface water nitrate concentrations peaked early in the storm (68 and 37 μeq NO_3^-N/L for Pebble Creek and the main channel Queets River, respectively, Figure 7f), but remained substantially above pre-storm concentrations for the duration of the sampling period. Compared with subsurface water, surface water DOC was high and nitrate was low (Figure 7f). The highest concentrations of nitrate and DOC were measured near the top of the rising limb of the first peak in the hydrograph. An increase in surface water DOC, but not nitrate, was observed in Pebble Creek near the second peak in discharge on November 15.

Two distinct pathways of soil leaching can be inferred from these data, each distinguished by routing, timing, and solute concentrations. First, most surface water solutes in peak flows were derived from lateral transfer and localized flooding of streamside soils into surface waters. This is indicated by nitrate and DOC, which achieved their maximum concentrations in surface waters long before subsurface solute concentrations peaked. Absolute concentrations of DOC were much higher in surface than subsurface water, and the proportion of DOC to nitrate was much higher in surface water. Direct translation of soil DOC to surface waters is the most likely explanation for the observed surface water increases. These results are con-

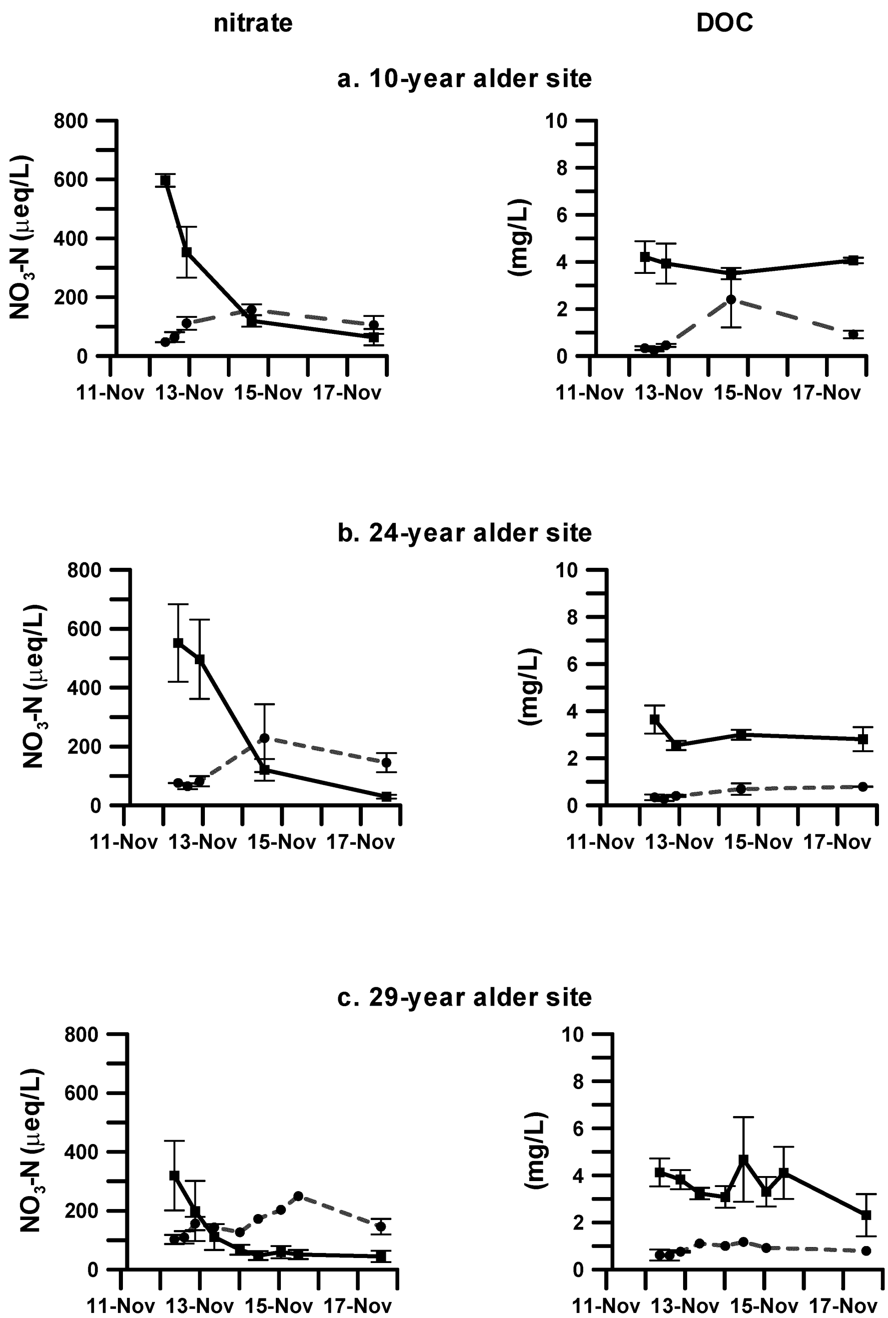

Figure 7. November 11–17 storm sampling results: a–e. NO_3^-N and DOC in soil solution –■– and subsurface water –●– in riparian sites; f. NO_3^-N and DOC in surface water in the Queets River –■– and Pebble Creek –●–. Bars indicate standard errors.

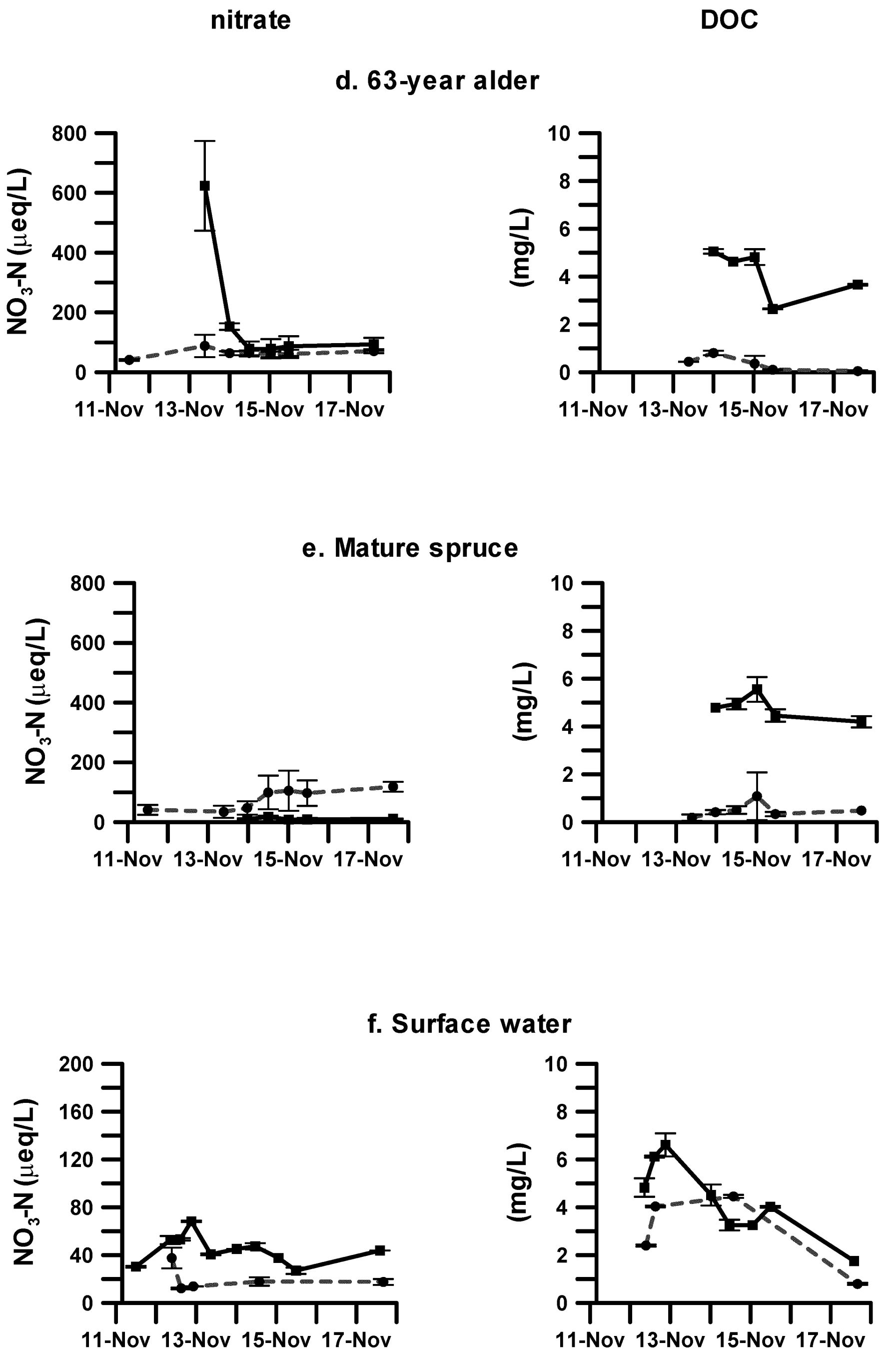

Figure 7. Continued.

sistent with the ″Flushing Hypothesis″ (Hornberger et al. 1994), which proposes interception and shallow routing of solutes through soil by a rising water table as the mechanism responsible for the commonly observed peak in solute concentrations on the rising limb of hydrographs. This flushing mechanism has been used to model DOC (Foster et al. 1989; Hornberger et al. 1994) and nitrate (Murdoch and Stoddard 1992; Creed and Band 1996) responses to snowmelt and storm flow pulses for several small catchments with shallow water tables.

The second pathway of leaching is from soils through subsurface waters to surface water in stream channels. Compared with the direct pathway described above, this water takes longer to reach surface channels, is enriched in nitrate and contains little DOC. The travel times of leached nitrate through subsurface pathways, as inferred from nitrate concentrations in subsurface waters (Figures 7a–e), are variable. Although subsurface nitrate concentrations were declining at most sites by the end of the sampling period, they were still several times higher than surface water concentrations. As storm flows subside, discharge is increasingly dominated by subsurface inputs. Adsorption or denitrification may reduce subsurface nitrate concentrations before it enters surface waters, but the pattern of declining DOC and steady or increasing nitrate in surface waters (Figure 7f) suggests an increased influence of subsurface solutes on surface water solute concentrations toward the end of the sampling period.

High nitrate concentrations in lysimeters sometimes occur as an artifact of soil disturbance during installation (Liator 1988). This does not appear to be a factor in explaining the high concentrations measured in this study. Organic C concentrations at the 60 cm depth where samples were collected were low (0.25–0.6% C) at all sites (Bechtold 2000). The angled insertion of the lysimeter tubes and the large fluxes of water moving through the soil during the storm would help minimize any effects of soil disturbance during installation. Lysimeters had been evacuated several times prior to first sampling, which should have removed accumulations of mineralized N.

Although results of the summer rainfall experiment are not directly comparable to those of the November storm, there did appear to be a much more sustained soil nitrate pool in the summer. Total water inputs during the artificial rainfall experiment and during the November storm were similar in magnitude (55 and 41 cm, respectively). The rapid decline in soil solution nitrate during the autumn storm suggests that the leachable N pool had already been substantially depleted by rainstorms earlier in the fall (total precipitation between September 1 and November 12 was 24 cm) or by increased biological demand early in the autumn. There was little evidence of rapid recharge of soil nitrate between November 14 and 17 when rainfall was much reduced. Nitrate concentrations increased at only one site during that period, the 63-year alder site. Ammonium concentrations in all samples were the detection limit and did not increase in any of the November 17 samples, as might have been caused by mineralization of organic matter.

Synthesis

The three studies collectively demonstrate the dominance of hydrologic processes in controlling the seasonal movement of nitrate from riparian soils to the aquatic ecosystem in an N-rich riparian forest. Soil nitrate was rapidly leached during actual and simulated rainstorms, and – in the autumn at least – quickly depleted the soil nitrate pool. The data support a two-phase conceptual model of N leaching. During peak flows, direct inputs of nitrate and DOC from saturated soils bordering stream channels were the main source of solutes to surface water. In areas further from stream channels, nitrate – but not DOC – leached into subsurface waters. Although absolute fluxes and surface water concentrations of nitrate are reduced from those occurring during peak flows, the nitrate draining through subsurface waters probably has a greater influence on local productivity as it is released slowly and over periods of lower nitrate concentrations, whereas high nitrate concentrations in peak flows are rapidly flushed from the system with little opportunity for absorption by biota.

While soil nitrate was rapidly depleted during rainstorms, accumulation of soil nitrate occurred over long dry periods with low aquatic nitrate. The greatest increases in soil nitrate occurred during an extended drought in the summer of 1998. This nitrate was loosely retained by the soil, and could readily have been mobilized had there been rain, as demonstrated by the artificial rainfall experiment. In contrast, in Summer 1997 when rainfall was high, little soil nitrate accumulated and aquatic nitrate concentrations were high. The lack of a summer buildup of soil nitrate was the apparent cause of unusually low Fall 1997 aquatic nitrate concentrations. Thus, interannual variability in summer rainfall can have far reaching effects on bio-available concentrations of terrestrial and aquatic N.

The low nitrate and high DOC measured in the mature spruce site are consistent with patterns observed in other mature conifer forests in the Pacific Northwest (Sollins et al. 1980; Binkley et al. 1982; Triska et al. 1984). In those studies, soil N was tightly retained. Nitrate production was low; N export was very low and dominated by organic N. In contrast, the rainfall driven nitrate pulses measured in the alder sites are similar to patterns observed in areas receiving anthropogenic N inputs and in other alder-influenced ecosystems (Cole 1992). A large proportion of riparian forest in the Pacific Northwest is dominated by alder. Despite large N inputs from the riparian forest, streams remain oligotrophic and for the most part N-limited. The most visible evidence of riparian nutrient subsidy of aquatic habitats is enhanced primary and secondary productivity in localized areas receiving upwelled subsurface water (Fevold 1998). Temporal and spatial variability of nutrient exchanges is an important aspect of the biogeochemistry of these ecosystems, and contributes to the high habitat heterogeneity that is characteristic of natural floodplains.

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