Non-Point-Source Impacts on Stream Nutrient Concentrations Along a Forest to Urban Gradient

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ABSTRACT / We conducted statistical analyses of a 10-year record of stream nutrient and sediment concentrations for 17 streams in the greater Seattle region to determine the impact of urban non-point-source pollutants on stream water quality. These catchments are dominated by either urban (22–87%) or forest (6–73%) land cover, with no major nutrient point sources. Stream water phosphorus concentrations were moderately strongly (r^2 = 0.58) correlated with catchment land-cover type, whereas nitrogen concentrations were weakly (r^2 = 0.19) and nonsignificantly (at α < 0.05) correlated with land cover. The most urban streams had, on average, 95% higher total phosphorus (TP) and 122% higher soluble reactive phosphorus (SRP) and 71% higher turbidity than the most forested streams. Nitrate (NO_3), ammonium (NH_4), and total suspended solids (TSS) concentrations did not vary significantly with land cover. These results suggest that urbanization markedly increased stream phosphorus concentrations and modestly increased nitrogen concentrations. However, nutrient concentrations in Seattle region urban streams are significantly less than those previously reported for agricultural area streams.

The Clean Water Act (CWA) and similar laws in other economically developed countries have had tremendous success in dealing with classic point-source pollutants (i.e., discharges from municipal wastewater treatment plants, industry, food processing, etc.). Currently, non-point-source (NPS) nutrients are the main cause of lake, stream, and coastal area eutrophication in the United States (Carpenter and others 1998; NRC 2001). Eutrophication of surface waters can cause increases in phytoplankton biomass and declines in water clarity, increased incidences of nuisance and toxic algal blooms, taste and odor problems, and increased treatment costs for drinking water facilities (Welch 1992). In the most recent Environmental Protection Agency (EPA) assessment, approximately 45% of lake surface area and 35% of river reaches in the United States were found to violate the CWA's "fishable and swimmable" standard, with most of this impairment due to nutrient-induced eutrophication (US EPA 1996; US EPA 2000). NPS nutrients are difficult to quantify due to their diffuse nature and the fact that many small sources might contribute NPS pollutants. The first step in controlling the impact of NPS nutrients on water quality is developing a quantitative understanding of their sources. It is also vital to fully understand how various land uses impact NPS nutrient loading, so that changes in water quality due to modified land use can be predicted and realistically modeled.

During the past two decades, several field studies have examined the impact of watershed land-use activities on stream nutrient and sediment concentrations (Omernik 1976; Osborne and Wiley 1988; Wahl and others 1997; Tufford and others 1998; Tufford and others 2003). Urban land use, through alterations of physical and hydrologic features of watersheds and the production of additional anthropogenic nutrient sources (i.e., lawn fertilizers, pet waste, septic tank effluent, accelerated erosion) are thought to be an important cause of lake and stream water quality degradation (US EPA 1990; Carpenter and others 1998; Tufford and others 1998). In humid regions, urbanization is often accompanied by loss of forest cover

(Wickham and others 2002); however, in arid regions, trees can be more common in urban than outlying areas (McPherson 1998). The loss of forested cover with urbanization in humid areas minimizes recycling and uptake of nutrients due to reduced microbial and vegetative processes that immobilize nutrients in the forest canopy, forest litter, forest soils, and organic material (Wahl and others 1997; Abelho 2001). Moreover, replacement of forest areas with impervious surfaces is associated with long-term changes in the quantity and composition of suspended solids inputs, which, in turn, act as an important transport mechanism for nutrients in streams through flocculation, adsorption, and colloidal action (Stone and Droppo 1994; Mulliss and others 1996).

The relative space–time distributions of the various nutrient forms in stream water are primarily regulated by the source and biological availability of the nutrients, the depth and residence time in groundwater, and oxygenation rates of the streams (Coats and Goldman 2001). Natural watershed phosphorus transport is regulated by physical and chemical processes like watershed weathering, erosion and sediment transport, and chemical binding with clay and calcium minerals (Wetzel 2001). Nitrogen is naturally supplied to lakes and streams via photochemical- and lightening-produced nitrate in rainwater and especially microbially mediated nitrogen fixation by plants in the watershed (Wetzel 2001); however, anthropogenic activities can greatly increase atmospheric nitrogen transport (Paerl 1997). Nitrate is particularly vulnerable to autotrophic uptake, but dissolved organic nitrogen—especially from coniferous forest zones—is characterized by a lower biological availability that also varies over the annual cycle, depending mostly on the nitrogen source (Coats and Goldman 2001). Urban streams typically have larger peak flows during precipitation events and lower base flows during seasonal dry periods (Booth 1991; Paul and Meyer 2001), both of which can impact sediment and associated nutrient (especially phosphorus) transport.

Currently, there is a lack of studies quantifying urban NPS impacts on stream water quality. Several of the most widely known and extensively cited studies of urban impacts on stream water quality (Osborne and Wiley 1988; Meybeck 1998; USGS 1999; Paul and Meyer 2001) include streams or rivers that received between 10% and in some cases over 50% of their average annual flow from wastewater treatment plant effluent. Other studies that examined urbanization impacts on stream water quality actually included only a few sites with a substantial portion of urban area in their catchments. For example, Omernik's (1976) classic investigation of land-use im-

pacts on stream nutrients summarized data from 473 streams, of which only 11 were classified as mostly urban. However, most of these ''urban'' streams also had substantial agricultural land use within their catchments. Tufford and others (1998) studied eight streams, of which only one was substantially urban. Because of this, it is difficult to make general statements about urban NPS nutrient impacts on stream water quality using the currently available published literature.

The objective of this study was to quantify the impact of urban (relative to forest) land cover on stream nutrient and sediment concentrations in the humid northwest region of North America. This objective was pursued by comparing average stream constituent concentrations for a long-term database of stream water quality in the Seattle region in the state of Washington, USA to satellite derived measures of land cover (i.e., urban, forest, agricultural, etc.) for 17 streams. The catchments of these streams ranged from being strongly dominated by urban to being strongly dominated by forest land cover. We also assessed the relationship between water quality and physical (surface area, slope, geologic composition of soils) characteristics of these catchments. In addition, we examined how the constituent concentrations of these streams varied seasonally. The stream water parameters assessed were total phosphorus (TP), soluble reactive phosphorus (SRP–P), total nitrogen (TN), nitrate (NO_3–N), ammonium (NH_4–N), turbidity, and total suspended solids (TSS). Because major point-source nutrients were not present in any of the streams we sampled, the present study makes it possible to directly estimate the impact of urban NPS pollutants on stream nutrient and sediment concentrations.

Methods

This study utilized a long-term database of stream nutrient and suspended material concentrations for 17 streams located in the Lake Washington and Lake Sammamish watershed of the Seattle region of Washington State, USA (Figure 1), to assess the statistical association between land-cover and stream water characteristics. This stream water quality assessment included every sizable stream within the overall watershed. The streams assessed in this study were from an overall stream sampling program that included approximately twice as many sampling sites. The streams included in this assessment were independent of each other, and almost all of the sampling sites not included in this study were upstream duplicates of included sites. These streams were sampled slightly up-
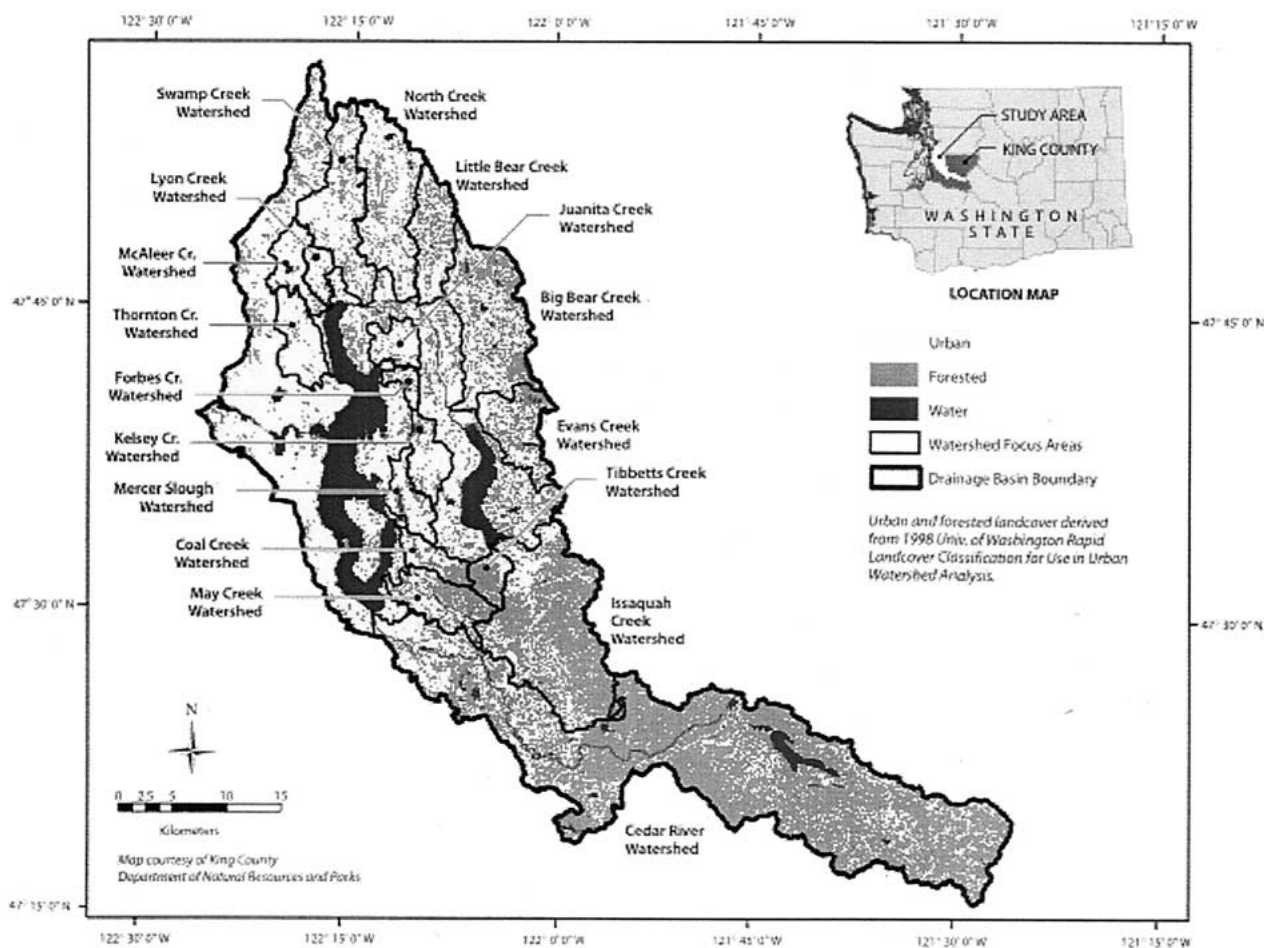
Figure 1. A map of the 17 study catchments and their land-cover characteristics.

stream of the point where they entered Lake Washington or Lake Sammamish, or the Sammamish River, which joins these lakes.

This watershed is composed primarily of forest and urban land cover with very little agricultural area. The most common trees are Douglas fir (*Pseudotsuga menziesii*), western red cedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and red alder (*Alnus rubra*). Glacial material (till, outwash, and drift) comprised 86.1%, volcanic material (volcanoclastic and andesite) comprised 6.4%, alluvium comprised 3.9%, and continental and marine sedimentary material comprised 2.5% of the geologic material in the overall watershed. The most common soil series in this region according to the USDA–NRCS classifications are Alderwood, Beausite–Alderwood, Everett, Alderwood–Kitsap–Indianola, Barneston–Klaus–Skykomish, Kaleeta–Melakwa, Tokul–Blethan–Ogarty, Tokul, and Elwell–Philippa (Goldin 1992). These soil types are comprised of glacial till, glacial outwash, and alluvium, or combinations of volcanic ash and glacial till, glacial outwash, colluvium, and/or alluvium. Unfortunately, a complete Geographic Information System (GIS) data layer for soil types in this area does not exist. Wastewater treatment plant (WWTP) or industrial effluent was not discharged into any of the streams assessed in this study.

Land-cover patterns for the study area were acquired from the Center for Water and Watershed Studies, University of Washington (see Hill and others 2003). Land cover was classified from a Landsat satellite image obtained in 1998 for the Puget Sound lowlands of northwestern Washington State. The image resolution was 30 m; thus, each pixel represented 900 m^2. The land-cover classes were chosen to reflect categories that can be readily distinguished in the satellite data and to represent important differences in watershed characteristics, especially for urban and urbanizing areas. The final classified image contained the following seven land-cover categories: forest, grass/

Table 1. Characteristics of the land-cover categories according to Hill et al's (2003) classification scheme

	Forest	Grass/shrub/crop	Water	Bare soil	Urban forest	Urban grassy	Urban paved
Trees	70%				25%	<25%	
Paved	<20%	<20%	<20%	<20%	20–60%	<60%	60%
Grass		50%				25%	
Water			80%				
Bare earth				75%			

shrub/crop, water, bare soil, urban forest, urban grassy, and urban paved (Table 1). Urban forests differ from natural forests in that they have much less area covered by trees and more area covered by paved surfaces. For complete descriptions of this classification process, see Hill and others (2003). The stream layer network and watershed subbasin delineation were obtained from the Washington Department of Ecology. These layers were developed using a combination of a digital elevation model and the Washington State Department of Natural Resources stream network map. The geological substrate classification was based on a 1:100,000 scale geologic map produced by the Washington State Department of Natural Resources, Division of Geology and Earth Resources. All of these GIS layers were obtained online from the University of Washington library (http://wagda.lib.washington.edu/data/washdata.html). The land-cover patterns within specified distances of the stream channel were determined using ARC/INFO software. We created three buffer distances (100 m, 220 m, and 340 m), and the extracted land-cover data were compared with data from a 1992 Landsat satellite image to test whether the 1992 and 1998 images gave similar descriptions of catchment land cover.

The long-term database of stream water constituents covers the period from 1972 to the present, but because the greater Seattle area has urbanized during this time, we only used data collected from the beginning of 1990 until the end of 1999. The basic design of the field-sampling program included monthly grab samples collected on a fixed schedule for each stream irrespective of weather. These samples represent ambient conditions in the streams and will be referred to as "baseline" samples. The database used for these analyses had on average 112 ± 9 (± 1 SD) grab samples per stream. Samples for analyses of dissolved nutrients ($SRP–P$, $NO_3–N$, $NH_4–N$) were filtered and frozen within 24 h of collection. The stream water organic nitrogen (Org-N), NO_3, and NH_4 were used to calculated TN accordingly: $TN = Org-N + NO_3 + NH_4$. The database used for these analyses was 98–100% complete for TP, SRP, NO_3, turbidity, and TSS, and 68–70% complete for TN and NH_4.

All nutrient and sediment analyses were conducted at the King County Environmental Laboratory according to methods described in Standard Methods (APHA 1998). All samples were analyzed within their respective hold times, and quality assurance/quality control procedures included the use of blanks and laboratory duplicates and matrix spikes on 5% of all samples. Soluble reactive phosphorus was determined according to the automated ascorbic acid method (SM4500-PF), total phosphorus (TP) was determined according to the automated ascorbic acid method after manual persulfate digestion (SM4500-P-B, E), nitrate + nitrite nitrogen was determined according to the automated cadmium reduction method ($SM4500-NO_3–N-F$), ammonium nitrogen was determined according to the automated phenate method ($SM4500-NH_3–N-H$), organic nitrogen was determined according to the block digestion and flow injection method ($SM4500-N_{ORG}$-D), total suspended solids was analyzed according to SM2540-D, and turbidity was determined with a Hach turbidity meter using the nephrolometric method. Turbidity concentrations were expressed as nephelometric turbidity units (NTU).

The geometric mean concentration was used to describe the constituents summarized in this study. This approach followed the actual distribution of the data and allowed us to examine the central tendency for each constituent in each stream without being unduly influenced by a few extremely high values (Limpert and others 2001). These geometric mean concentrations were very strongly correlated ($r^2 = 0.967 \pm 0.026$; ± 1 SD) with the corresponding median values for these data. We did not present flow-weighted mean concentration estimates because flow data were not consistently collected in most of the study streams during the period assessed. Using these data, we conducted a series of regression analyses, where regression equations $\alpha < 0.05$ were considered statistically significant. Correlations were classified as weak when the regression coefficient (r^2) was <0.3, moderate when r^2 values were >0.3 but <0.6, and strong when r^2 values were >0.6. Our statistical analyses were performed using SPSS® version 10 for Mac.

Table 2. Catchment characteristics for each of the 17 streams assessed in this study

	Watershed area (km²)	Average flow (m³/s)	Average slope (degrees)	Glacial	Volcanic	Alluvium	Sedimentary	Forest	Grass/shrub/crop	Water	Bare soil	Urban forest	Urban grassy	Urban paved
Big Bear	91	2.20	3.3	95.8%	0.0%	4.2%	0.0%	45.6%	7.1%	0.8%	0.3%	33.2%	10.9%	2.1%
Cedar	500	18.68	5.1	82.0%	1.7%	2.1%	14.2%	69.3%	4.1%	1.7%	0.3%	16.8%	6.2%	1.7%
Coal	17	0.36	8.2	43.5%	42.1%	4.9%	3.1%	52.1%	4.7%	0.1%	0.4%	19.3%	21.0%	2.4%
Evans	40	0.33	4.1	90.7%	0.0%	9.3%	0.1%	45.6%	8.3%	0.1%	1.0%	30.9%	11.7%	2.4%
Forbes	9	0.10	3.3	100.0%	0.0%	0.0%	0.0%	18.1%	6.0%	0.2%	0.5%	40.0%	25.8%	9.3%
Issaquah	145	3.99	10.0	66.9%	18.4%	5.6%	4.9%	73.3%	4.1%	0.1%	0.5%	15.2%	5.2%	1.6%
Juanita	17	0.32	4.0	99.8%	0.0%	0.0%	0.0%	12.6%	5.5%	0.0%	0.2%	40.7%	33.2%	7.7%
Kelsey	21	0.21	3.1	99.8%	0.0%	0.2%	0.0%	23.0%	6.2%	0.2%	0.6%	39.9%	17.6%	12.6%
Little Bear	39	0.59	3.3	97.1%	0.0%	2.9%	0.0%	39.2%	7.7%	0.0%	1.7%	31.0%	13.4%	7.0%
Lyon	11	0.17	5.0	95.2%	0.0%	4.8%	0.0%	18.6%	5.0%	0.2%	0.1%	46.1%	27.1%	2.9%
May	36	0.62	8.1	65.9%	29.6%	2.1%	1.5%	55.9%	7.1%	0.4%	0.5%	19.1%	15.0%	2.1%
McAleer	21	0.41	4.0	95.0%	0.0%	4.4%	0.0%	13.5%	6.8%	2.9%	0.3%	37.3%	28.1%	11.0%
Mercer	21	0.64	5.1	92.1%	0.0%	0.0%	0.7%	20.4%	6.0%	0.1%	1.6%	26.6%	28.2%	17.1%
North	74	1.75	2.7	96.5%	0.0%	3.5%	0.0%	27.0%	9.0%	0.9%	1.4%	30.4%	23.8%	7.3%
Swamp	63	1.10	3.1	96.0%	0.0%	4.0%	0.0%	27.9%	8.1%	0.8%	1.2%	28.5%	23.7%	9.8%
Thornton	29	0.32	3.5	96.8%	0.0%	3.2%	0.0%	6.5%	6.2%	0.1%	0.3%	38.0%	37.5%	11.6%
Tibbets	14	0.08	11.0	49.6%	17.5%	15.5%	17.3%	72.7%	4.5%	0.0%	0.6%	8.8%	8.0%	5.4%
Mean	35[a]	0.58[a]	5.1	86.1%	6.4%	3.9%	2.5%	36.5%	6.3%	0.5%	0.7%	29.5%	19.8%	6.7%
Standard dev.			2.6	18.1%	12.7%	3.8%	5.2%	22.1%	1.5%	0.8%	0.5%	10.6%	9.7%	4.7%

[a]Geometric mean.

Note. The slope and geological characterizations were for the entire watershed, whereas the land-cover classifications were generated using a 340-m buffer width. This stream channel buffer comprised 71 ± 12% (±1 SD) of the entire catchment area and provided nearly identical land-cover classifications as 100-m and 200-m stream buffers. Stream flow data were obtained from USGS or King County stream flow gauges. Continuous flow data for the last 20 years was available for some streams, whereas others only had 3–5 years of continuous data.

Table 3. Geometric mean nutrient concentrations for baseline flow conditions ($n \approx 112$) for each of the 17 streams

Concentration	Total phosphorus (µg/L)	Soluble reactive P (µg/L)	Total nitrogen (µg/L)	Nitrate (µg/L)	Ammonium (µg/L)	Turbidity (NTUs)	Total suspended solids (mg/L)
Big Bear	71.1	31.0	990	582	40.5	3.5	8.5
Cedar	16.2	6.2	321	228	18.9	1.1	2.7
Coal	38.0	18.3	845	605	24.2	2.6	3.9
Evans	61.0	27.5	680	311	22.7	2.9	5.4
Forbes	87.0	49.9	936	584	34.4	3.4	4.6
Issaquah	30.4	11.4	1155	951	30.2	1.9	4.8
Juanita	58.4	28.0	1599	1326	29.9	3.0	5.0
Kelsey	72.4	40.6	1245	906	31.7	2.1	3.3
Little Bear	62.8	29.0	1218	855	29.7	3.2	6.6
Lyon	57.0	25.7	1444	1121	28.2	2.6	4.6
May	44.8	21.8	1423	1227	22.3	2.1	3.9
McAleer	65.9	28.6	1596	1255	32.6	3.4	9.6
Mercer	83.2	48.6	901	610	27.9	3.3	4.8
North	77.0	40.0	1275	802	31.0	3.0	5.5
Swamp	54.9	27.1	1247	891	25.5	2.5	3.9
Thornton	70.5	34.6	1484	1156	32.7	2.7	5.6
Tibbets	38.0	13.2	1362	956	43.5	5.5	5.9

Results

Catchment Land-Cover and Physical Characteristics

Figure 1 and Table 2 depict and describe the area, mean annual flow, slope, geologic composition, and land-cover classifications for each of the 17 stream catchments summarized in this study. These catchments were small to moderate sized, gently sloping to steep in some places, dominated by glacial till and outwash soils with outcroppings of volcanic, alluvia, and sedimentary material, along an urban to forest dominated gradient. The most forested catchments tended to be in the east and southeast portion of the overall watershed.

Because the stream channel network used for these analyses was extremely detailed, on average 23 ± 7% of the entire catchment area was within the 100-m buffer width, 51 ± 12% was within the 200-m buffer width, and 71 ± 12% was within the 340-m buffer width. The land-cover estimates provided by the 1992 (using a 500-m buffer width) and 1998 (using a 340-m buffer width) satellite images were highly correlated for both urban ($r^2 = 0.89$) and forest ($r^2 = 0.93$) land cover. After assessing the correspondence between the 1992 and 1998 land-cover classifications, we compared the land-cover estimates provided by the 1998 classifications for each of the buffer distances (100, 220, and 340 m) calculated. The correlations between the percent coverage estimates for the forest and sum of urban land cover for the three buffer widths were very strong ($r^2 \geq 0.97$). Because the land-cover classifications for each buffer width were so highly correlated, the impact of buffer width was not considered in the remainder of this study, and all results presented were for the 340-m buffer width. The most forested/least urban catchments tended to have slightly larger surface areas ($r^2 \approx 0.29$), steeper slopes ($r^2 \approx 0.63$), and more volcanic geologic material ($r^2 \approx 0.65$). The urban catchments were dominated by glacial geologic material and the Alderwood and related soil series. Table 3 summarizes the geometric mean concentration for TP, SRP, TN, NO_3, NH_4, turbidity, and TSS for each of the streams sampled.

Land Cover and Stream Water Constituent Relations

Total phosphorus and SRP concentrations and turbidity were moderately or strongly correlated with percent urban and forested land cover ($r^2 = 0.33–0.62$; Figure 2 and Table 4). TN was weakly ($r^2 \approx 0.21$) and only marginally correlated ($P \approx 0.06$) with land cover (Table 4), whereas NO_3, NH_4, Org-N, and TSS concentrations were not significantly correlated with urban or forested land cover (Figure 2 and Table 4). Because forested and urban land covers were very strongly correlated, both had very similar (but mirror image) associations with stream constituent concentrations (Table 4). We used the subclassifications for urban land cover (i.e., urban forest, urban grassy, and urban
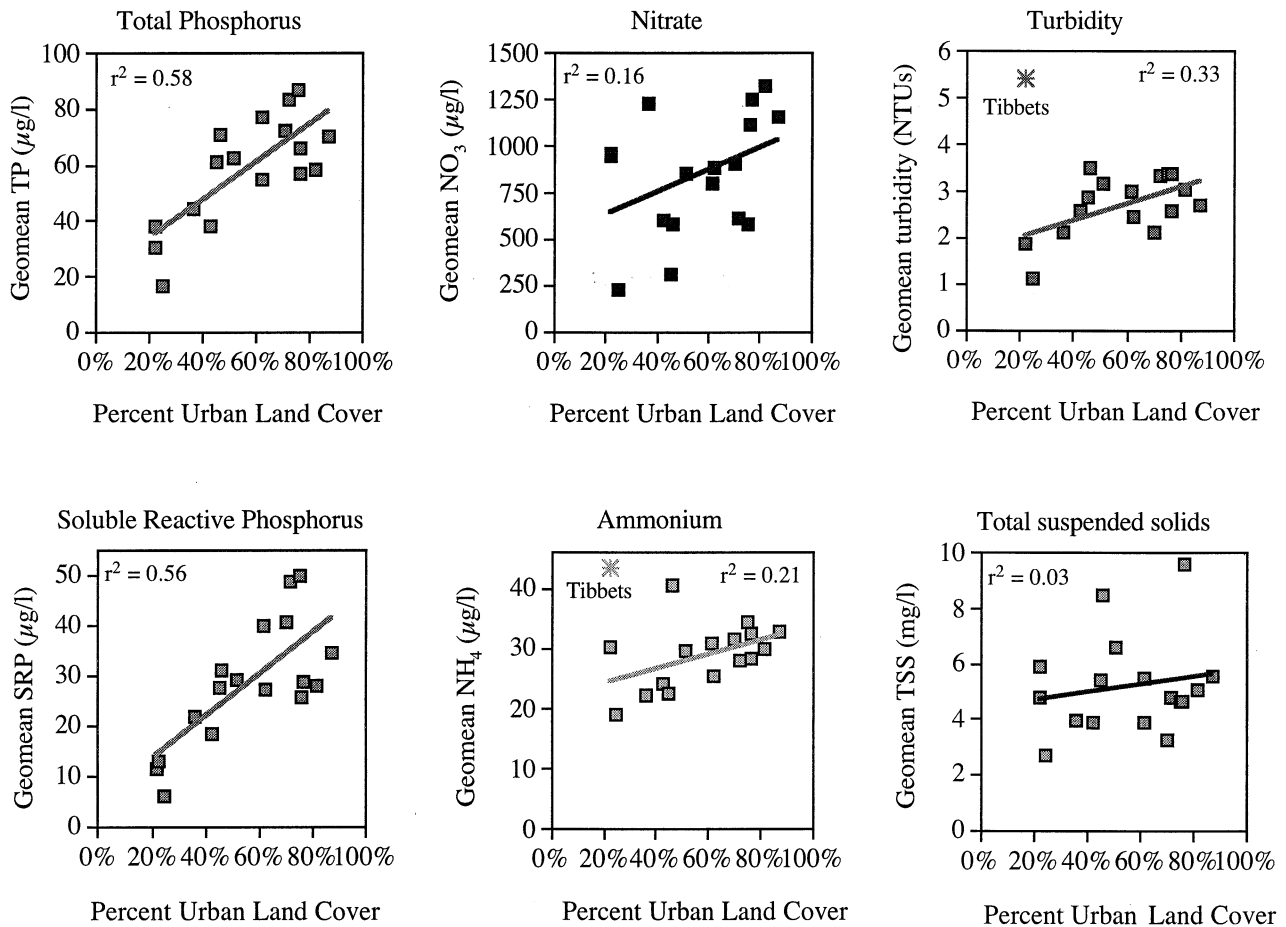
Figure 2. Bivariate plots of percent urban land cover versus geometric mean constituent concentration for the 17 streams. Tibbets Creek data were excluded when calculating the regression coefficients for NH_4–N and turbidity.

paved) to generate multiple regression models. This resulted in somewhat improved fits and reduced prediction error relative to the single variable sum of urban models for TP and SRP concentrations (Table 4). We used the Akaike Information Criterion (Akaike 1970) to test whether these multiple regression models actually performed better than the single regression models, and according to this criterion, the two variable regression models did perform better. The coefficients obtained for the multiple regression models suggest that paved urban areas generated two to three times more phosphorus than did forested urban areas.

To derive an estimate of the relative impact of urban (relative to forest) land cover on stream nutrient and sediment concentrations, we compared concentrations in the most urbanized streams to those in the most forested streams. Because urban and forested covers were the two dominant land types in our study area, we used a simple ratio of average concentrations in the four most urban streams divided by average concen-

trations in the four most forested streams to derive an urbanization enrichment index. These calculations showed the most urban streams had, on average, 122% higher SRP concentrations, 95% higher TP concentrations, and 44% higher TN concentrations during baseline conditions. Turbidity was 71% higher in the most urban catchments; however, this result is equivocal because it was obtained by excluding data from Tibbets Creek, which was both heavily forested and had the highest turbidity (Figure 2). NO_3, NH_4, and TSS concentrations were not statistically higher in the most urban catchments (Table 4).

Catchment Area, Slope, Geology, and Land Cover

The watershed basin area ($\log_{10}$ transformed data) was generally weakly negatively correlated ($r^2 \approx 0.20$) with constituent concentrations; however, these weak correlations were driven by only one stream, which drained the largest and most pristine catchment (i.e., Cedar River). Without the Cedar River observation, r^2

Table 4. Regression equations for percent land cover versus stream water geometric mean constituent concentration

Parameter	Model	n	r^2	RMS	F-Test	P-Value
Urban land cover versus constituent concentration						
Total phosphorus	TP = 19.9 + 0.682*%Urban	17	0.58	12.9	20.57	0.0004
Soluble reactive P	SRP = 4.7 + 0.422*%Urban	17	0.56	8.4	18.90	0.0006
Total nitrogen	TN = 743 + 7.44*%Urban	17	0.22	314	4.16	0.0593
Nitrate	NO_3 = 512 + 5.95*%Urban	17	0.16	306	2.80	0.1152
Ammonium	NH_4 = 21.0 + 0.119*%Urban	16	0.21	4.9	3.65	0.0767
Organic nitrogen	Org-N = 250 + 0.156*%Urban	17	0.12	85	1.99	0.1785
Turbidity	Turbidity = 1.65 + 0.0182*%Urban	16	0.32	0.56	6.60	0.0223
Total suspended solids	TSS = 4.86 + 0.00139*%Urban	17	0.03	1.8	0.47	0.5047
Forest land cover versus constituent concentration						
Total phosphorus	TP = 83.3 − 0.687*%Forest	17	0.62	12.3	24.17	0.0002
Soluble reactive P	SRP = 43.7 − 0.422*%Forest	17	0.59	8.1	21.26	0.0003
Total nitrogen	TN = 1422 − 7.18*%Forest	17	0.21	315	4.05	0.0625
Nitrate	NO_3 = 1048 − 5.55*%Forest	17	0.14	308	2.53	0.1325
Ammonium	NH_4 = 32.9 − 0.118*%Forest	16	0.21	4.9	3.68	0.0757
Organic nitrogen	Org-N = 399 − 0.169*%Forest	17	0.14	83	2.53	0.1326
Turbidity	Turbidity = 3.36 − 0.0190*%Forest	16	0.36	0.54	7.80	0.0144
Total suspended solids	TSS = 5.81 − 0.00163*%Forest	17	0.04	1.8	0.66	0.4281
Multivariate land cover models						
Total phosphorus	TP = 17.1 + 0.95*%Urb.Forest + 1.95*%Urb.Paved	17	0.68	11.6	15.18	0.0003
Soluble reactive P	SRP = 4.0 + 0.49*%Urb.Forest + 1.47*%Urb.Paved	17	0.69	7.3	15.48	0.0003

Note. Tibbets Creek data were not used when calculating the regression equations for ammonium and TSS. RMS is residual mean square.

values between catchment area and constituent concentration were generally <0.05. Catchment slope was negatively correlated with TP and SRP ($r^2 \approx 0.44$) concentrations; however, the slope was not significantly correlated with the other constituents assessed. TP and SRP concentrations were also positively correlated with the proportion glacial geologic material ($r^2 \approx 0.45$) and negatively correlated with the proportion volcanic ($r^2 \approx 0.28$), alluvial ($r^2 \approx 0.21$), and sedimentary ($r^2 \approx 0.48$) material. The other constituents assessed had, for the most part, very weak statistical associations with catchment geologic composition.

Seasonal Variation in Constituent Concentrations

The database was broken into four seasons: winter (December 20–March 19), spring (March 20–June 19), summer (June 20–September 19), fall (September 20–December 19). We then calculated constituent geometric averages for each stream during each time period (Figure 3). TP was the least seasonally variable, whereas SRP concentrations had a minimum during the winter and spring at approximately 17% below the annual mean and a peak during the summer at 29% above the annual mean. NO_3 concentrations were quite seasonal, with concentrations peaking during the winter at 42% above the annual mean and approximately 14% below the annual mean during the other

seasons. NH_4 concentrations were 25% above the annual mean during the fall and 25% below the annual mean during the summer. TSS and turbidity concentrations were very seasonal and showed nearly identical trends, with high concentrations during the winter and low concentrations during the summer. The seasonal minimum and maximum values were observed during opposite seasons for SRP and NO_3.

Discussion

We conducted statistical analyses of a 10-year record of stream nutrient and sediment concentrations for 17 streams in the greater Seattle region. The results showed that stream water nutrient (and especially phosphorus) concentrations were higher in the streams draining urban catchments. In addition, we developed regression equations showing that a conversion of 10% of catchment area from forest to urban land cover will result in an increase in average stream water concentrations of 7 μg/L for TP, 4 μg/L for SRP, 73 μg/L for TN, 57 μg/L for NO_3, 1 μg/L for NH_4, 0.2 NTU for turbidity, and 0.021 mg/L for TSS.

Land cover, watershed area, slope, and geologic composition were correlated in the study catchments, which complicates the interpretation of these results somewhat. The five most forested catchments had
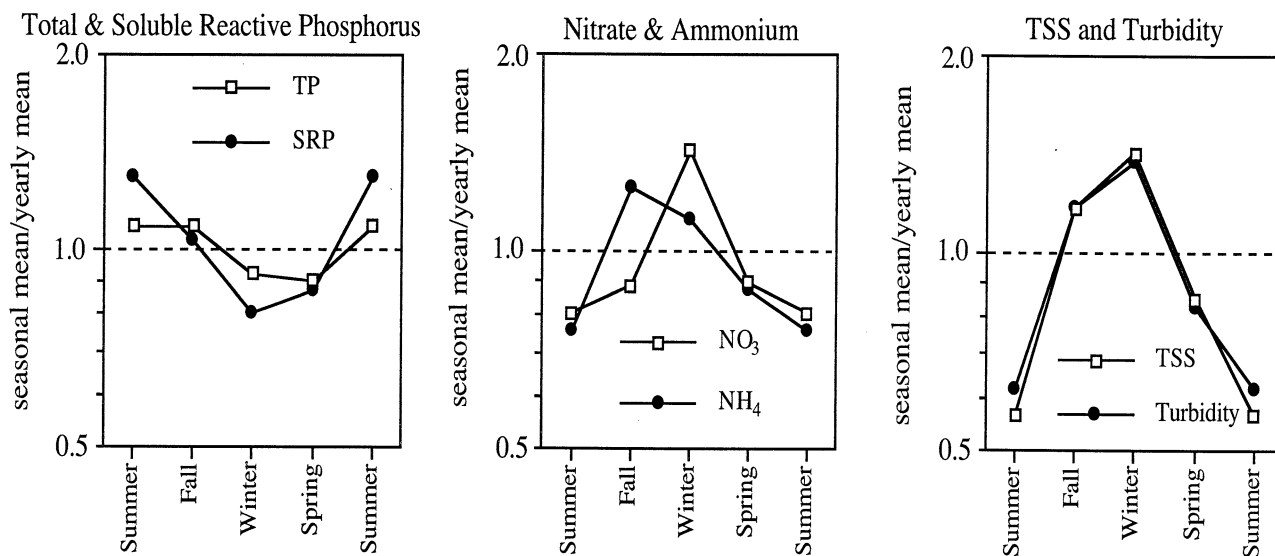
Total & Soluble Reactive Phosphorus

Nitrate & Ammonium

TSS and Turbidity

Figure 3. Seasonal variation for each of the major constituents considered in this study. The values presented are the average of season means divided the annual mean for each stream. The *y* ordinate on these plots is logarithmic.

steeper slopes (mean 10.5% versus 3.7% for the other 13 catchments; *t*-test *P*-value = 0.0042), less glacial geologic material (mean 62% versus 96% for the other catchments; *P*-value = 0.0067), and more volcanic geologic material (mean 22% versus 0.0% for the other catchments; *P*-value = 0.0314). The most forested catchments also had a higher percent sedimentary geologic material (mean 8% versus 0.1% for the other catchments), but this difference was only marginally significant (*P*-value = 0.0623). The relationship between catchment geologic composition and stream water phosphorus concentrations is counterintuitive because the Alderwood soil series, which is by far the most common in this region and especially in the most urban catchments, has been described as having low fertility and is generally considered poor for agricultural purposes (http://www.kingcd.org/soil_ald.htm). Furthermore, the most prevalent soil series in the most forested catchments other than the Alderwood (i.e., Barneston, Klaus, Skykomish, Kaleeta, Melakwa, Tokul, Blethan, Ogarty, Elwell and Philippa), all have high volcanic ash content, which is often associated with high soil fertility and phosphorus content. Based on the fact that the catchments with the highest stream water phosphorus concentrations are dominated by the nutrient-poor Alderwood soil series and the streams with the lowest phosphorus concentrations had soils rich in volcanic ash, we infer, that the impact of land cover (i.e. urban versus forest) on stream water phosphorus content is more important than the impact of soil type and geologic material in these catch-

ments. At first glance, the negative association between stream water phosphorus content and catchment slope might also seem counterintuitive because TP transport is often associated with erosion and erosion is positively associated with slope (Nearing 1997; Liu and others 2000; see, however, D'Arcy and Carignan 1997). The negative association between catchment slope and stream water phosphorus content is probably an artifact of the strong association between catchment land cover and slope; that is, steep catchments might be dominated by forest land cover because areas with steep slopes are unsuitable for urban and suburban development.

Most previous studies of urbanization impacts on stream nutrient concentrations either compared gradients from urban to agricultural land cover or included urban streams that also received WWTP effluent (Osborne and Wiley 1988; Meybeck 1998; USGS 1999; Paul and Meyer 2001). This is problematic because both agricultural and urban land uses can enrich streams with nutrients (Soranno and others 1996; Carpenter and others 1998) and WWTP effluent can greatly exaggerate the real impact of urban land cover on NPS nutrients. Osborne and Wiley (1988) is a widely cited work on urban impacts on stream water quality; however, this study has two confounding issues. First, the urban streams in the watershed they studied received all of that region's WWTP effluent; therefore, their results cannot be used to assess urban NPS impacts on stream nutrients because much or most of the nutrient load in these streams was due to WWTP
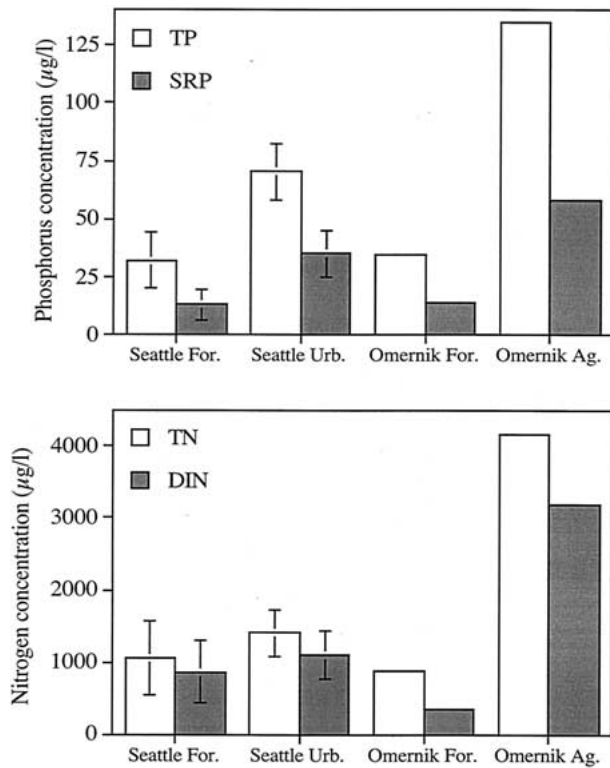
Figure 4. A comparison of average (±1 SD) total and dissolved phosphorus and nitrogen concentrations for our four most forested and urban streams and Omernik's (1976) "mostly forest" ($n = 170$) and agricultural ($n = 91$) land-use categories. Omernik did not present variability estimates for his data. Dissolved inorganic nitrogen (DIN) equals nitrate plus ammonium concentrations. In the Seattle area streams, TP and SRP concentrations were higher in urban than forest streams ($P \approx 0.0004$), TN concentrations were marginally higher in urban streams ($P \approx 0.06$), and DIN concentrations were not significantly related to land cover (Table 4).

effluent. Osborne and Wiley (1988) also reported extremely high SRP concentrations (usually >1000 μg/L), which are similar to WWTP effluent that has not undergone tertiary treatment (Tchobanoglous 1991), suggesting that very little dilution of the WWTP effluent occurred or a systematic error in their reported SRP concentrations. Of the eight streams Tufford and others (1998) used to generate regressions between land-cover and stream nutrient concentration, only one had a substantial proportion of urban area in its catchment (i.e., Gills Creek at 47% urban). Thus, this study would have difficulty differentiating agricultural from urban impacts on stream water quality.

If we compare nutrient concentrations in the most forested Seattle area streams to those in the most urban streams, we can compare the relative "urban enrichment" to the "agricultural enrichment" ob-

served in Omernik's (1976) classic study. We used the average concentrations in the four most urban and forested streams to characterize Seattle area "urban" and "forest" stream nutrients (Figure 4). Omernik included six classifications in his study, including "mostly forest" and "agricultural." The average nutrient concentrations observed in the forested Seattle area streams were nearly identical to the averages reported by Omernik (1976) for his "mostly forested" streams for TP, SRP, and TN; however, the dissolved inorganic nitrogen (DIN; equals NO_3 + NH_4) concentrations observed in the Seattle area streams were substantially higher. This difference in forested stream DIN concentrations might be due to the fact that red alder (*Alnus rubra*), a nitrogen-fixing plant, is a very common component of the riparian and forest vegetation of the Pacific Northwest. The TP concentrations in Seattle area urban streams were, on average, 48% less than those observed in Omernik's (1976) "agricultural streams"; similarly, the SRP, TN, and DIN concentrations in Seattle urban streams were 39%, 66%, and 65%, respectively, lower than those observed in Omernik's agricultural streams. Overall, Omernik's agricultural streams had about twice as much phosphorus and three times as much nitrogen as observed in Seattle area urban streams, however, Seattle area forest streams had substantially higher nitrate concentrations than did Omernik's.

Figure 4 also shows that in Seattle area urban streams, the phosphorus enrichment (∼110%), relative to forested streams, was substantially larger than the nitrogen enrichment (∼40%) for these same streams. The weak trends in stream water TN and nitrate concentrations with land cover were due to three streams (May, Tibbets, and Issaquah), where both the percent forest land-cover and stream water nitrogen concentrations were high. If we treat these sites as outliers and recalculate the regression equations provided in Table 4, the r^2 values for these associations increased from ∼0.18 to 0.66. However, it is extremely difficult to justify treating 20% of the observations in any dataset as outliers. Based on what is known about the typical riparian and forest vegetation in second-growth forests in the Puget Sound region, it is entirely plausible that some forest streams in this region naturally have moderately high nitrate concentrations.

The "urban enrichment" values presented in Figure 4 are confounded by the fact that even our least urban streams still contained approximately 26 ± 7% (± 1 SD) urban land cover in their catchments. An alternative means of calculating urban land-cover impacts on stream nutrient and sediment concentrations is to compare the concentrations predicted by the

regression equations presented in Table 4 for fully built-out catchments (i.e., 85% urban according to our land-cover data) to hypothetical pristine catchments (i.e., 0% urban). However, the results of this approach should be interpreted with caution because the predictions of these regression equations are being extended to conditions that were not observed when they were generated (i.e., urban land cover <20%). Given these caveats, this alternative means of estimating extreme urban enrichment values suggests that completely urban catchments would have 291% higher TP, 763% higher SRP, and 85% higher TN concentrations, as well as 94% higher turbidity compared to ''pristine'' streams. These results also predict 99% higher nitrate and 48% higher ammonium concentrations in fully built-out streams; however, these results were not statistically significantly.

In contrast to others studies that found that Org-N was the dominant fraction of TN [see Omernik's (1976) survey of eastern United State streams; Coats and Goldman's (2001) study of Lake Tahoe tributaries; and Edmonds and others (1995) study of Olympic National Park of Washington State streams], this study found Org-N only accounted for ~20% of TN in Seattle area streams. In Seattle area streams, NO_3–N was by far the dominant nitrogen fraction at approximately 60–80% of TN. This strong dominance of TN by NO_3 has only been previously reported for hardwood forests in the eastern United States and tropical forest catchments (see references in Coats and Goldman 2001).

In common with previous studies in the northwest (Edmonds and Blew 1997), we found NO_3 concentrations peaked during winter high-flow conditions. Investigators who have studied streams in colder climates have noted NO_3 pulses during early snowmelt in a number of Sierra Nevada catchments (Sickman and Melack 1998; Coats and Goldman, 2001). This phenomenon has previously been attributed to freeze–thaw cycles causing accumulations of NO_3 in the snow pack (see discussion in Coats and Goldman 2001). In contrast, Kendall and others (1995) used isotopic studies to show that these pulses actually originated from the soil in several New England and Rocky Mountain watersheds. Because we observed similar NO_3 pulses in catchments that do not accumulate snow and we observed pulses of NO_3 in the most forested catchments during rain events, these results lend support to the argument that this NO_3 has its origin in the soil. One potentially very important source for NO_3 in these watersheds is alders, which are a common tree in this region and have previously been shown to be important sources of inorganic nitrogen via symbiotic N_2 fixation (Goldman 1961; Hart and others 1997). Apparently, red alders produce ammonium

during the summer, which is nitrified and stored in the forest soils and then washed out during the winter when biological uptake of inorganic nitrogen is at an annual minimum and subsurface flows through forest soils increase seasonally (Van Miegroet and others 1992). Direct rainwater impacts on stream NO_3 concentrations cannot be used to explain the seasonal dynamic in stream NO_3 concentrations because rainwater NO_3 concentrations in this region at ~300 µg/L (Ebbert and others 1985) are far below those observed in these Seattle area streams during the winter.

The seasonal cycles showing SRP concentrations peaked during the summer (with a winter minimum) while nitrate concentrations peaked during the winter suggest very different watershed leaching properties for these nutrients. The high summer SRP concentrations might to be a simple function of the fact that groundwater typically has much higher SRP concentrations than does rainwater. According to Perkins and others (1997) ground water in this region has a median SRP content of 103 µg/L, whereas according to Ebbert and others (1985), rainwater has a median TP content of ~10 µg/L. During the summer, precipitation is relatively low in this climatic region and stream flows are dominated by groundwater contributions. During the winter, surface runoff and subsurface flow of recent rainwater dominates stream flows. According to a 45-year record of rainfall data collected at the SeaTac International Airport, low-lying areas in this watershed have, on average, 35 cm of rain during the winter months, 15 cm of rain during the spring months, 9 cm of rain during the summer months, and 39 cm of rain during the fall months.

Conclusions

These results show that the phosphorus content of wadable streams in the Seattle, Washington, area during baseline conditions was moderately strongly correlated with urban land cover, whereas stream water nitrogen concentrations were only weakly correlated with land cover. TSS concentrations were not correlated with land cover and turbidity was only weakly correlated. Stream water NO_3–N, turbidity, and TSS concentrations peaked during the winter and had a minima during the summer. SRP and TP peaked during the summer, and NH_4–N concentrations peaked during the fall. Phosphorus and nitrogen concentrations in Seattle area urban streams were, on average, only 50% and 30%, respectively, as high as what has previously been reported by agricultural streams nation in the United States. However, Seattle area forested streams had very high nitrate concentrations compared

to other forested streams in the United States. Our study also found a counterintuitive negative relation between stream water phosphorus content and catchment slope, which was a result of the strong association between catchment land cover and slope. Presumably this was because steep catchments were dominated by forest land cover because areas with steep slopes are unsuitable for urban and suburban development. More research is needed to determine the impact of urban and forest land cover, catchment slope, and stream flows on nutrient transport.

Acknowledgments

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