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Research Article

Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region

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Abstract

There has been an increasing interest in evaluating the relative condition or health of water resources at regional and national scales. Of particular interest is an ability to identify those areas where surface and ground waters have the greatest potential for high levels of nutrient and sediment loadings. High levels of nutrient and sediment loadings can have adverse effects on both humans and aquatic ecosystems. We analyzed the ability of landscape metrics generated from readily available, spatial data to predict nutrient and sediment yield to streams in the Mid-Atlantic Region in the United States. We used landscape metric coverages generated from a previous assessment of the entire Mid-Atlantic Region, and a set of stream sample data from the U.S. Geological Survey. Landscape metrics consistently explained high amounts of variation in nitrogen yields to streams (65 to 86% of the total variation). They also explained 73 and 79% of the variability in dissolved phosphorus and suspended sediment. Although there were differences in the nitrogen, phosphorus, and sediment models, the amount of agriculture, riparian forests, and atmospheric nitrate deposition (nitrogen only) consistently explained a high proportion of the variation in these models. Differences in the models also suggest potential differences in landscape-stream relationships between ecoregions or biophysical settings. The results of the study suggest that readily available, spatial data can be used to assess potential nutrient and sediment loadings to streams, but that it will be important to develop and test landscape models in different biophysical settings.

Introduction

Scientists and environmental managers alike are concerned about broad-scale changes in land use and landscape pattern and their cumulative impact on hydrological and ecological processes that affect stream, wetland, and estuary conditions (Hunsaker and Levine 1995; O'Neill et al. 1997). Of particular concern is the degree to which landscape conditions at watershed scales influences nitrogen, phosphorus, and sediment loadings to surface waters (Hunsaker and Levine 1995; Ator and Ferrari 1997; EPA 1998). High levels of nutrients and sediment in water can pose significant human health and ecological risks (Ator and Ferrari 1997).

A number of studies have shown strong relationships of water quality, water quantity, and run-off to landscape characteristics. A decrease in natural vegetation indicates a potential for future water quality problems (Likens et al. 1977; Franklin 1992; Walker et al. 1993; Hunsaker and Levine, 1995; Smith et al. 1997). Other studies have shown that the land uses within a watershed can account for a relatively high percentage of the variability in stream and estuary water quality (Omernik et al. 1981; Hunsaker et al. 1992; Charbonneau and Kondolf 1993; Roth et al. 1996; Herlihy et al. 1998; Behrendt et al. 1999; de Wit and Behrendt 1999). Empirical studies have established the significant causal relationship between watershed characteristics and nutrient and sediment loads (Yates and Sheridan 1983). Changes in landscape conditions in the riparian zone and in areas surrounding water quality sample sites may have a greater influence on water quality than broader scale, watershed conditions (Lowrance et al. 1984; Peterjohn and Correll 1984), although the importance of near-site, landscape conditions may vary, depending on the biophysical setting (Clarke et al. 1991). Wetlands also play an important role in reducing nutrient loads to surface waters (Weller et al. 1996). Agriculture on slopes of greater than 3% increases the risk of soil erosion (Wischmeier and Smith 1978), and this can lead to increases in nutrient and sediment loadings to surface waters. High amounts of impervious surface and roads on watersheds also may result in high loadings of nutrients and sediment to streams (Burns 1972; Harden 1992; Arnold and Gibbons 1996), and atmospheric deposition can be a significant source of nitrogen in surface waters (Stensland et al. 1986).

Although there have been numerous attempts to monitor and model nutrient and sediment loadings to streams and estuaries (Yates and Sheridan 1983; Weller et al. 1996; Ator and Ferrari 1997), no comprehensive approach exists to evaluate potential loadings to streams based on landscape composition and pattern across regional scales. Moreover, there is a need to identify those surface waters at greatest risk to high levels of nutrient and sediment loads so that actions can be taken to reduce the risk (EPA 1988).

A new set of land cover databases being developed by the Multi-resolution Land Characteristics Consortium (MRLC, Vogelmann et al. 1998) and development of landscape pattern metrics from spatial data (Jones et al. 1997; O'Neill et al. 1997), offers an unprecedented opportunity to assess landscape conditions as they relate to important hydrological and ecological processes affecting stream conditions. In 1996, a regional-scale land cover database was developed for the five-state area of the United States Mid-Atlantic Region, and this database, along with other regional landscape coverages (e.g., topography, soils, road networks, stream networks, and human population density), was used to assess landscape conditions across the entire region (Jones et al. 1997; Wickham et al. 1999). The assessment used a set of landscape metrics to evaluate the spatial patterns of human-induced stresses and the spatial arrangement of forest, forest-edge, and riparian habitats as they influence forest habitat suitability and stream conditions. Advances in computer technology and geographic information systems (GIS) have made it possible to calculate landscape metrics over large areas (e.g., regions) at relatively fine scales (e.g., down to 30 m).

Using landscape metric data generated from Jones et al. (1997), and nutrient (nitrogen and phosphorus) and sediment data provided by the U.S. Geological Survey (Langland et al. 1998), we tested the hypothesis that landscape metrics are strongly related to nutrient and sediment loadings on a set of watersheds in the Chesapeake Bay Basin.

Methods

Data from this study were compiled from two independent sources. The nutrient and sediment yield data were acquired from the US Geological Survey (USGS) and the landscape metrics were derived from the data used by Jones et al. (1997).

The USGS calculated annual nutrient and suspended sediment loads and yields for 148 stations on non-tidal streams within the Chesapeake Bay Basin (Langland et al. 1998). The Chesapeake Bay Basin contains more than 150 000 stream miles in the District of Columbia and parts of New York, Pennsylvania, Maryland, Virginia, West Virginia, and Delaware. The Basin is comprised of six major river systems: the Susquehanna, Patuxent, Potomac, Rappahannock, York, and James rivers (Langland et al. 1998).

The USGS annual load estimates that we used were based on the USGS water year which is October 1 through September 30. The inputs for the USGS model were measured concentration of analyte, measured discharge, and time. USGS models and their results are described in detail by Langland et al. (1998). The USGS annual loads were based on two different sampling regimes: flow-driven and fixed-interval sampling. In flow-driven (or total stream flow) sampling regimes, samples were collected on the basis of stream flow conditions. Fixed-interval samples were collected on a regular schedule, usually monthly or quarterly (Langland et al. 1998). Loads were reported by the USGS in tons per year, which were then converted to metric units and normalized by watershed area to produce a yield estimate (kg/ha/yr). Our analysis was conducted with the load data normalized by watershed area only and we combined load values from flow-driven and fixed interval sampling.

Watershed support areas were delineated using Arc/Info GIS software (ESRI 1996) for each of the water quality monitoring locations so that our support areas consisted of only that part of the watershed actually contributing to the water quality monitoring point. For the landscape metrics, we acquired digital coverages of landscape metrics generated by Jones et al. (1997) and then calculated landscape metrics for each of the watershed support areas. The source of the land cover map from which many of the landscape metrics were derived was the MRLC program (Vogelmann et al. 1998). The MRLC data were derived from Landsat Thematic Mapper (TM) data and had a resolution of 30 m and 15 land cover classes (Vogelmann et al. 1998). However, before calculating the landscape metrics, we used an Arc/Info reclassification routine to aggregate the 15 land cover classes into six classes: urban, agriculture, wetland, forest, barren, and water (Table 1). The aggregation of land cover classes improves the accuracy of the individual land cover classes (Zhu et al. 1999). The landscape metrics and their associated range of values are listed in Table 2 and correlation coefficients between individual landscape metrics are given in Table 3.

Figure 1 shows the delineated watersheds used in this analysis within the Mid-Atlantic region. Although it can not be detected in Figure 1, several of the watersheds were actually nested within larger watersheds (see results). We eliminated watersheds with others nested within them as keeping them in the regression analysis would have violated the independence assumption necessary for regression analysis. The points on the figure represent the location of the water quality data collection. We did not generate watersheds for eleven of the water quality monitoring points because areas delineated would have been massive and would have exceeded our data computing resources. These locations on the figure are represented by a point only and were not used in the analysis.

For each site, annual nutrient and suspended sediment yields were averaged for 1989 to 1994 to coincide with the approximate dates of the satellite imagery used to generate the land cover map. Average annual loads were calculated for the following constituents where data were available: total nitrogen, total nitrate, ammonia, total phosphorus, dissolved phosphorus, and suspended sediment. In addition to nested watersheds, we deleted watersheds that had no data between 1989 and 1994, watersheds from the analysis that did not overlap our study area by at least 50%, and watersheds whose boundaries could not be delineated (primarily in coastal zones where topographic relief was lacking). As a result, a total of 78 watersheds were used in the analysis, ranging in size from 84 to 1 733 414 ha with an average size of 90 000 ha.

Table 1. Reclassification of the MRLC land cover map (Vogelmann et al. 1998) into six general classes. The reclassified land cover map was used to generated landscape metrics (see Methods).

Original MRLC classification	Classification used in this study
Class 1: Water	Water
Class 2: Low intensity developed	Urban
Class 3: High intensity developed	Urban
Class 4: Hay/pasture/grass	Agriculture
Class 5: Row crops	Agriculture
Class 6: Probable row crops	Agriculture
Class 7: Conifer (evergreen) forest	Forest
Class 8: Mixed forest	Forest
Class 9: Deciduous forest	Forest
Class 10: Woody wetlands	Wetlands
Class 11: Emergent wetlands	Wetlands
Class 12: Barren; Quarry areas (excluding spectrally dark coal mines)	Barren
Class 13: Barren; Coal mines	Barren
Class 14: Barren; Beach areas	Barren
Class 15: Barren; Transitional (including clear cut areas)	Barren

We first examined the individual relationships between nutrient and sediment yields (dependent variables) and landscape metrics (independent variables) using individual scatter plots and regression analyses (SAS 1990). Our initial analyses produced regression results with unequal error variances and non-normally distributed error terms. To compensate for these departures, we log-transformed the yield values using a natural log transformation. It should be noted that the geographic coverage was not the same for all of the analyses performed in this study. These differences are illustrated along with the different models in Figures 2 and 3.

Results

Landscape metrics consistently explained a high percentage of the total variation in nitrogen, phosphorus,

Table 2. Description and range of values for landscape metrics used in this analysis. Calculation methods and details of each indicator can be found in Jones et al. (1997). Metrics were calculated for each watershed support area.

Metric	Explanation	Minimum	Maximum	Mean
Riparian agriculture (ripa)	Percent of watershed with agricultural land cover adjacent to stream edge. One pixel (30 × 30 m) wide.	0.04	58.5	18.5
Riparian forest (ripf)	Percent of watershed with forest land cover adjacent to stream edge. One pixel wide.	3.2	99.8	63.4
Forest fragmentation (ffrg)	Forest fragmentation index for watershed. Of all pairs of adjacent pixels in the watershed that contain at least one forest pixel, the percentage for which the other pixel is not forest.	0.6	79.5	17.4
Road density (rd)	Road density for watershed expressed as an average number of kilometers of roads per square kilometer of watershed.	81.7	494.1	179.1
Forest land cover (flc)	Percent of watershed with forest land cover.	1.3	99.6	58.4
Agricultural land cover (alc)	Percent of watershed with agricultural land cover (pasture/crops).	0.38	96.9	35.7
Agricultural land cover on steep slopes (ags3)	Percent of watershed with agriculture occurring on slopes greater than 3 percent.	0	52	12.5
Nitrate deposition (nd)	Estimated average annual wet deposition of nitrate (kg/ha × 100).	1249	2173	1758
Potential soil loss (poso)	Proportion of watershed with the potential for soil losses greater than 2240 kg/ha/yr.	0	67.8	33.2
Roads near streams (rxs)	Proportion of total stream length having roads within 30 m.	0	1.47	0.6
Slope gradient (sg)	Average percent slope gradient for watershed.	0	19.4	7.5
Slope gradient range (sgr)	Percent slope gradient range (maximum minus minimum) for watershed.	0	143	58.8
Slope gradient variance (sgv)	Percent slope gradient variance for watershed.	0	229	66
Urban land cover (purb)	Percent of watershed with urban land cover.	0.01	36.9	3.5
Wetland land cover (pwetl)	Percent of watershed with wetland land cover.	0	19.9	1.1
Barren land cover (pbar)	Percent of watershed with barren land cover. This includes quarry areas, coal mines, and transitional areas such as clear cut areas.	0	8.8	0.9

and sediment yield, but especially metrics of the amount of agriculture, riparian forests, atmospheric nitrate deposition, and roads in the watershed (Table 4, Figures 2 and 3). The greatest amount of variation explained by a landscape metric model was for total nitrate in streams ($r^2 = 0.86$, Table 4). This model included several significant landscape metrics, with the amount of agriculture in the watershed explaining 50% of the variation (positive association) and atmospheric nitrate deposition explaining 27% of the total variation (positive association, Table 4). The remaining landscape metrics in the model that explained relatively small, but significant portions of the total variation were related to roads, watershed slope, and the percentage of the watershed with urban land cover (Table 4).

Together, the proportion of total stream miles in the watershed with forest (riparian forest, negative association) and nitrate deposition (positive association),

Table 3. Correlation matrix of landscape metrics used in the stepwise regression analysis. Abbreviations for landscape metrics are given in Table 2. Underlined correlation numbers indicate values of 0.80 or greater.

	ags3	flc	nd	rd	rxs	alc	Purb	ffls	Pwetl	poso	Pbar	ripa	ripf	sg	sgr	sgv
ags3	1.00															
flc	−0.58	1.00														
nd	0.12	0.06	1.00													
rd	0.35	−0.67	0.11	1.00												
rxs	0.45	−0.13	0.25	0.29	1.00											
alc	0.66	−0.96	−0.00	0.50	0.16	1.00										
Purb	−0.07	−0.29	−0.11	0.76	0.07	0.03	1.00									
ffls	0.42	−0.87	−0.01	0.66	0.03	0.81	0.32	1.00								
Pwetl	−0.21	−0.19	−0.23	0.05	−0.31	0.08	0.07	0.12	1.00							
poso	0.78	−0.51	0.01	0.25	0.51	0.64	−0.14	0.40	−0.38	1.00						
Pbar	−0.32	0.31	−0.02	0.03	−0.12	−0.41	0.11	−0.20	−0.01	−0.40	1.00					
ripa	0.60	−0.83	0.15	0.38	0.20	0.88	−0.01	0.69	0.11	0.52	−0.46	1.00				
ripf	−0.61	0.91	−0.02	−0.53	−0.24	−0.94	−0.11	−0.76	−0.01	−0.62	0.37	−0.88	1.00			
sg	−0.24	0.76	0.22	−0.58	0.31	−0.65	−0.37	−0.69	−0.40	0.01	0.03	−0.46	0.55	1.00		
sgr	−0.25	0.65	0.18	−0.57	0.15	−0.54	−0.35	−0.65	−0.40	−0.05	−0.01	−0.38	0.46	0.80	1.00	
sgv	−0.36	0.73	0.14	−0.62	0.17	−0.62	−0.36	−0.69	−0.34	−0.05	−0.08	−0.46	0.54	0.92	0.85	1.00

explained 83% of the total variation of total nitrogen in streams (59 and 24%, respectively, Table 4, Figure 2).

Sixty-five percent of the variability in total ammonia was explained by a landscape model that included road density (50% of the variation) and riparian forests (15% of the variation, Table 4, Figure 2).

Landscape models for total and dissolved phosphorus differed considerably in the amount of variation that they explained; the landscape model for dissolved phosphorus explained 73% of the variation while the landscape model for total phosphorus explained only 45% of the variability (Table 4, Figure 3). Relatively high unexplained variance in the total phosphorus model may result from a larger number of samples representing a larger geographic area (Figure 3) and number of biophysical settings (e.g., different soil types and parent rock materials) than the dissolved phosphorus samples. However, riparian forest was by far the most important landscape variable in each model (63 and 41% of the variation in dissolved and total phosphorus, respectively, Table 4).

A landscape model consisting of riparian forests and the proportion of the watershed surface with wetland and urban land cover explained 79% of the total variation in sediment yield (Table 4, Figure 3), with the former metric explaining 47% of the variation and the latter two explaining 24% and 8.2%, respectively (Table 4). Sediment samples used in this model were available only for portions of the Piedmont and Coastal Ecoregions (Figure 3).

In one case a landscape variable in the model had a coefficient sign that was counter-intuitive to the hypothesized relationship (Table 4) – riparian agriculture had a negative association with total nitrate. Riparian agriculture explained a relatively small proportion of the total variability in the model, and when compared to the nutrient parameter, revealed the hypothesized relationship (e.g., a positive association between riparian agriculture and nitrate yield). Although variance inflation factors for these variables were generally well below the threshold of multicollinearity suggested by Neter et al. (1996), collinearity is probably still a factor in coefficient sign confusion. Many of the landscape metrics related to forest and agricultural land cover were highly correlated (Table 3), and the implications of these correlations to the results of the regression analyses are addressed in the Discussion section.

Discussion

Results of this study suggest that the types of landscape metrics used in the Mid-Atlantic Landscape Indicator Atlas (Jones et al. 1997) can be used to help explain potential loadings of nutrients and sediment to streams and to evaluate the 'health' of watersheds

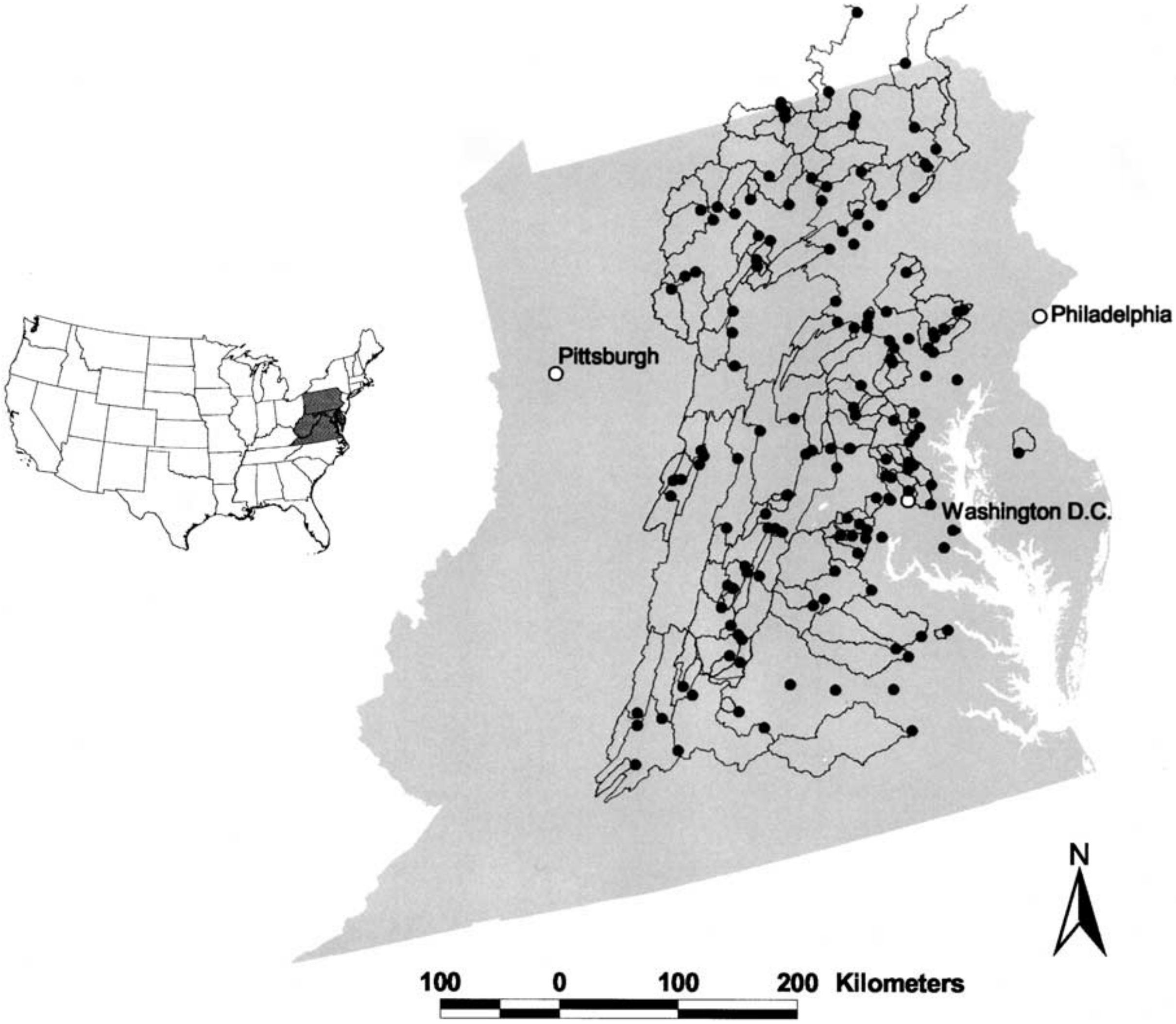

Figure 1. Locations of watersheds used in this study and the general location of the Mid-Atlantic Region in the United States. Points on the map represent water quality sample locations and polygons represent watershed locations.

relative to water quality (after Rapport et al. 1998). The amount of forest along streams (riparian forest) and agriculture consistently explained a high percentage of variation in nutrient and sediment yield. The linkage between intact riparian areas and high water quality is well established, especially in the eastern United States (Karr and Schlosser 1978; Yates and Sheridan 1983; Lowrance et al. 1984; Cooper et al. 1987). Riparian habitat functions as a 'sponge', greatly reducing nutrient and sediment runoff into streams (Peterjohn and Correll 1984; Cooper et al. 1987). However, atmospheric nitrate also consistently explained a relatively high percentage of nitrogen in streams. This suggests that for at least portions of the eastern United States one must understand patterns of atmospheric nitrogen deposition in order to determine potential loadings to streams – landscape metrics generated from surface characteristics will under-predict the loadings.

Differences in the landscape models for the different forms of nitrogen and phosphorus in surface water may result partly from differences in the chemical activity and source of the different forms of these nutrients (Behrendt et al. 1999; de Wit and Behrendt 1999), or because samples for each represent different geographies and different biophysical settings. The amount of total nitrate in water is a function of the amount of fertilizer, waste disposal, and sewage

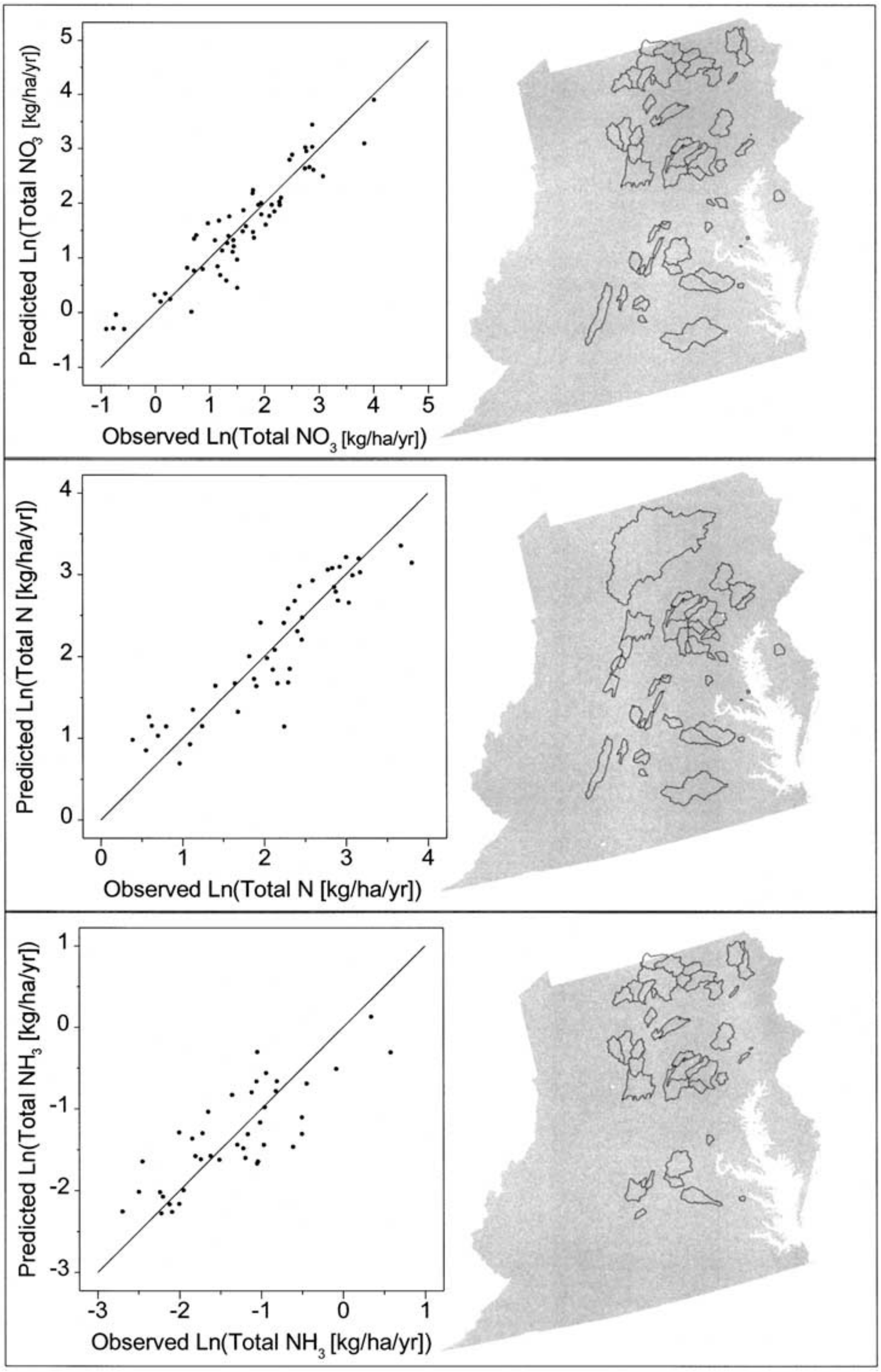

Figure 2. Predicted to observed values for total nitrate, total nitrogen, and total ammonia and watersheds from which the models were generated. Models were based on multiple linear regression relating landscape metrics (independent variables) to water quality variables (dependent variables). The lines shown in the graphs represent a one-to-one relationship.

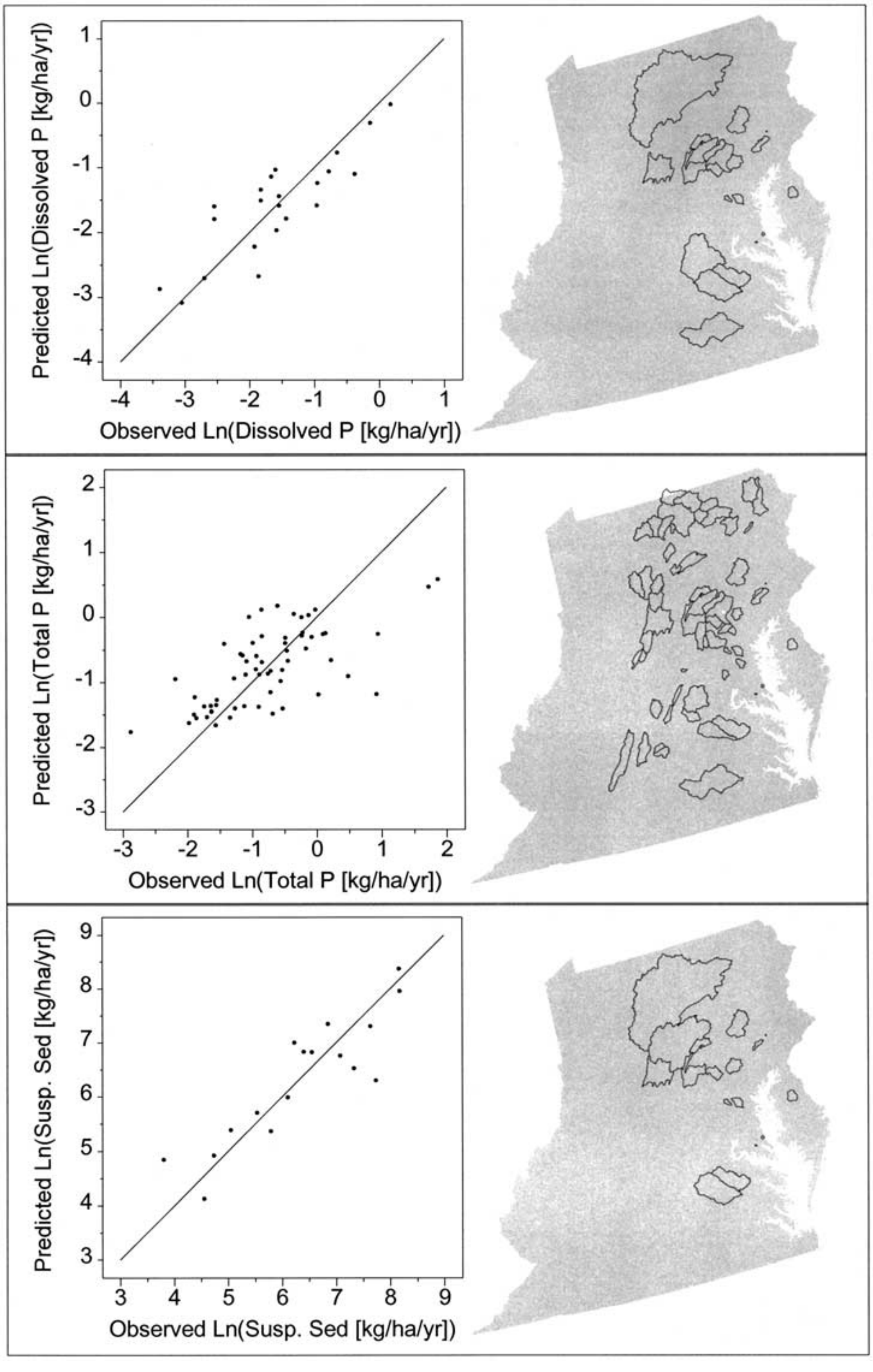

Figure 3. Predicted to observed values for dissolved phosphorus, total phosphorus, and suspended sediment and watersheds from which the models were generated. Models were based on multiple linear regression relating landscape metrics (independent variables) to water quality variables (dependent variables). The lines shown in the graphs represent a one-to-one relationship.

Table 4. Results of stepwise regression analysis relating nutrient and suspended sediment constituents (dependent variable) to landscape metrics (independent variables). Landscape metrics retained in the model were significant at $p < 0.05$.

Constituent (kg/ha/yr)	Number of sites	Metric	Variation explained by metric (%)	Total variation explained (%)
Total nitrogen as N	45	ripf	59	
		nd	24	83
Equation: log (total nitrogen) = 0.767 + 0.002 nd − 0.025 ripf				
Total nitrate as N	58	alc	50	
		nd	27	
		rxs	4.2	
		purb	2.0	
		sgv	1.7	
		ripa	1.6	86
Equation: log (total yield) = −3.408 + 0.002 nd + 0.812 rxs + 0.047 alc + 0.040 purb − 0.023 ripa + 0.004 sgv				
Total Ammonia as N	41	rd	50	
		ripf	15	65
Equation: log (total ammonia) = −1.302 + 0.006 rd − 0.015 ripf				
Dissolved phosphorus as P	22	ripf	63	
		pbar	10	73
Equation: log (dissolved phosphorus) = 0.061 − 0.421 pbar − 0.025 ripf				
Total phosphorus as P	61	ripf	41	
		purb	3.8	45
Equation: log (total phosphorus) = 0.840 + 0.025 purb − 0.026 ripf				
Suspended sediment	17	ripf	47	
		pwetl	24	
		purb	8.2	79
Equation: log (suspended sediment) = 8.472 + 0.079 purb − 0.116 pwetl − 0.038 ripf				

at the watershed scale (Hem 1985). This conclusion is supported by the fact that nitrate is readily transported in water and remains stable over a wide range of conditions (Hem 1985). Conversely, ammonia is readily absorbed by mineral surfaces and, therefore, its concentration in surface water may be more closely associated with near-site sources of pollution than watershed level conditions. Our results support these conclusions; 86% of the variability in total nitrate concentration in surface waters was explained by watershed-scale, landscape conditions (including atmospheric nitrate deposition), whereas a lower percentage of the variation in total ammonia concentrations was explained by landscape conditions (65% of the variation). Moreover, road density, which is a strong indicator of development in the Mid-Atlantic region, explained by far the greatest amount of variation in ammonia yield (50% of the total variation); this may reflect a higher concentration of sewage and waste disposal associated with urban and residential areas.

Differences in the landscape metrics among the nitrogen models may reflect differences in the geographic distribution of the samples. For example, atmospheric nitrate deposition is more important in the total nitrate model than it is in the total nitrogen model. Total nitrate samples were located predominately in the central and northern parts of the region where atmospheric nitrate deposition is greatest, whereas total nitrogen samples were located predominately in the central and southern parts of the region where atmospheric nitrate deposition is lower (Jones et al. 1997). Herlihy (1998) found that landscape models for nitrogen concentration in streams differed among ecoregions. However, despite the large number of ecoregions represented in the total nitrogen and nitrate samples, the landscape models still explained a high proportion of the total variance in stream nitrogen yield.

Differences between the importance of riparian forest and agriculture in the total nitrogen and nitrate models may reflect the size and scale of agricultural communities in the geographic areas that the models represent. Many of the total nitrogen samples were taken in areas with large-block agriculture; in these areas the presence of forest along streams may be critical in filtering nitrogen from water before it enters into a stream (Peterjohn and Correll 1984). Conversely, several of the nitrate samples were taken in forested watersheds of central and northern Pennsylvania where agriculture tends to be in smaller blocks interspersed among forests; in these areas an impact might be seen when the amount of agriculture on the watershed reaches a certain amount or threshold. Correlations between the amount of forest, agriculture, and forest near streams might also produce these results. Further research is needed to determine the causes for differences in the models.

The percentage of stream miles with forest at the watershed scale explained large amounts of the variation in dissolved phosphorus in streams (63%), but a much lower percentage in the total phosphorus model (41%). These two measures of phosphorus are functionally similar; high concentrations in water result from fertilizer applications in agricultural areas and animal and human waste (Hem 1985; Behrendt et al. 1999; de Wit and Behrendt 1999). Therefore, the differences between the two models probably reflect differences in sample locations and the biophysical settings that they represent, and sample size; samples of dissolved phosphorus were limited to the central part of the region (22 samples) whereas samples of total phosphorus were geographically dispersed (61 samples). These results also suggest that there may be some important differences among ecoregions (Omernik et al. 1981; Omernik 1987). Dissolved phosphorus samples and associated watershed support areas were limited to a few ecoregions whereas total phosphorus samples represented several ecoregions. Unfortunately, the number of existing samples did not allow us to assess differences between ecoregions. Moreover, point sources were not included in the analysis and studies have shown these to be significant sources of phosphorus (Behrendt et al. 1999). Weller et al. (1996) showed that wetlands and forested riparian areas reduce phosphorus loadings into surface waters, although their study was limited in geographic extent.

Samples of suspended sediment were limited almost entirely to the central and eastern part of the region and were not likely representative of mountainous areas within the region. However, the results appear to be consistent with the findings of Yates and Sheridan (1983) for streams in the Coastal Plain region of the southeastern United States; vegetated floodplains and wetlands explain a relatively high proportion of variability in suspended sediment in streams. Analysis of landscape-stream interactions in other ecoregions, especially those with high relief, are needed to assess potential sediment loadings to streams across the Mid-Atlantic region.

In addition to differences in sample locations, other factors may have contributed to differences among the landscape models and the amount of unexplained variance. Our analysis did not include metrics of point-source density or distribution, although point-source density and distribution may be correlated to road density and population. We will develop and evaluate metrics of point source distribution in our next set of analyses. Error in the land cover map may also account for some of the unexplained variance, although we attempted to reduce this type of error by combining land cover classes (Zhu et al. 2000). Differences in the management of agricultural, urban, and residential areas also may have introduced variation into the models. The data used in this study were not of fine enough resolution to assess the impact of management and policies on nutrient and sediment loads to streams. Additionally, because many of the forest and agricultural metrics were highly correlated, it is very difficult to explicitly distinguish the effects of these individual variables without further research. Finally, contributions of nitrogen from ground water were not included in our models and these can be important sources of nitrogen in streams (Ator and Ferrari 1997).

The results of this study suggest that regional-scale landscape data, including land cover databases being developed by the MRLC nationally, may be useful in assessing potential loadings to streams and watershed health with regards to water quality. However, because of the potential for differences in landscape-water quality relationships in different biophysical settings (Smith et al. 1997), it will be necessary to develop and test landscape models within and among different regions of the United States; the results of this study may only apply to certain biophysical settings of the eastern United States.

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