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Research article

Upstream-to-downstream changes in nutrient export risk

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Abstract

Nutrient export coefficients are estimates of the mass of nitrogen (N) or phosphorus (P) normalized by area and time (e.g., kg/ha/yr). They have been estimated most often for watersheds ranging in size from 10^2 to 10^4 hectares, and have been recommended as measurements to inform management decisions. At this scale, watersheds are often nested upstream and downstream components of larger drainage basins, suggesting nutrient export coefficients will change from one subwatershed to the next. Nutrient export can be modeled as risk where lack of monitoring data prevents empirical estimation. We modeled N and P export risk for subwatersheds of larger drainage basins, and examined spatial changes in risk from upstream to downstream watersheds. Spatial (subwatershed) changes in N and P risk were a function of in-stream decay, subwatershed land-cover composition, and subwatershed streamlength. Risk tended to increase in a downstream direction under low rates of in-stream decay, whereas high rates of in-stream decay often reduced risk to zero (0) toward downstream subwatersheds. On average, increases in the modeled rate of in-stream decay reduced risk by 0.44 for N and 0.39 for P. Interactions between in-stream decay, land-cover composition and streamlength produced dramatic changes in risk across subwatersheds in some cases. Comparison of the null cases of no in-stream decay and homogeneously forested subwatersheds with extant conditions indicated that complete forest cover produced greater reductions in nutrient export risk than a high in-stream decay rate, especially for P. High rates of in-stream decay and complete forest cover produced approximately equivalent reductions in N export risk for downstream subwatersheds.

Introduction

Numerous surveys have established a strong relationship between land cover and nutrient (nitrogen [N] and phosphorus [P]) export (Omernik 1977; Reckhow et al. 1980; Beaulac and Reckhow 1982; Frink 1991; Fisher et al. 1998). The relationship between nutrient export and land cover is often expressed as a coefficient that represents the mass of N or P exported per unit area per unit time (Reckhow et al. 1980; Beaulac and Reckhow 1982; Frink 1991). Nutrient export coefficients are straightforward measurements that are often proposed as tools for watershed management (Beaulac and Reckhow 1982; Rast and Lee 1983; Frink 1991).

The relationship between land cover and nutrient export can be expressed appropriately as risk because many other physical and anthropogenic factors influence nutrient export, but are often more difficult to estimate (Wickham et al. 2000). For example, Hartigan et al. (1983) showed that P export from an agricultural watershed practicing conservation tillage was

12 to 15 times greater than P export from a forested watershed depending on annual precipitation depths. However, under non-conservation tillage, P export from an agricultural watershed increased to 72 to 114 times greater than the forested watershed over the same range of precipitation. The interaction of agricultural practices and inter-annual variability in precipitation resulted in dramatic differences in nutrient export between forested and agricultural watersheds. Some physical and anthropogenic factors that create variability in the land cover-nutrient export relationship include inter-annual changes in precipitation (Lucey and Goolsby 1993), cropping practices (Renard et al. 1997), timing of fertilizer application relative to precipitation events (Beaulac and Reckhow 1982), geology (Dillon and Kirchner 1975), and density of impervious surfaces (Arnold and Gibbons 1996).

Nutrient export estimates are often based on watersheds on the order of 10^2 to 10^4 hectares (e.g., Hartigan et al. (1983) and Clesceri et al. (1986), Jordan et al. (1997), Fisher et al. (1998)). Watersheds in this size range are usually nested upstream or downstream components of larger drainage basins, transporting nutrients across subwatershed boundaries. For example, a watershed map for the state of Maryland, delineates about 20 watersheds within one of the smaller eight-digit hydrologic accounting units developed by the US Geological Survey (Seaber et al. 1987).

The risk of encountering a given load of nutrients in a subwatershed depends on the magnitude of export from upstream neighbors. The purpose of this paper is to show how nutrient (N, P) export risk can be moved across subwatersheds and how that movement leads to spatial changes in risk. Bartell et al. (1992) and Suter (1993), Richards and Johnson (1998) discuss potential applications of risk assessment to landscape studies. Existing ecological risk assessments have focused on identifying spatial variation in risk across the landscape (Graham et al. 1991; Wickham et al. 2002; Wickham and Wade 2002). Movement of nutrients across subwatersheds is a complimentary approach that shows how ecological processes can produce spatial change in risk (Risser et al. 1984; Reynolds and Wu 1999).

Methods

Study area

The nutrient export risk models were run on four drainage basins in the state of Maryland, on the east coast of the USA (Figure 1). We chose these drainage basins for their variation in land-cover composition, availability of discharges estimates (water.usgs.gov), and because they represented four of the six major subwatersheds of the Chesapeake Bay (Linker et al. 1996). Discharge estimates (Table 1) were used to move risk across subwatersheds identified in Figure 1.

We used the land-cover data from the MultiResolution Land Characteristics (MRLC) consortium (Loveland and Shaw 1996) National Land Cover Data (NLCD) (Vogelmann et al. 2001) to estimate the area of the land-cover classes. Watershed boundaries were acquired from the Maryland Department of Natural Resources (MDDNR). Geographical Information System (GIS) software was used to tabulate proportions of forest, agriculture, and urban land cover by watershed. Areas mapped as wetlands were treated as forest because wetlands also reduce nutrient loads to surrounding waters (Preston and Bedford 1988).

Nutrient export risk

We estimated risk by compiling empirical distributions of nutrient export coefficients by land-cover class from existing literature, fitting empirical data to theoretical distributions, and using repeated trial simulation to estimate the frequency of equaling or exceeding a specified nutrient export threshold (Wickham et al. 2000). Simulated values were restricted to be within the minimum and maximum values of the observed ranges to provide conservative estimates of risk. We used the frequency of equaling or exceeding a specified threshold divided by the total number of trials as an estimate of risk. We ran 10,000 trials for each watershed.

N and P thresholds (kg/ha/yr) for estimating risk were identified using the allowable N and P loads (per major subwatershed) under the Chesapeake Bay nutrient reduction goals (Linker et al. 1996). N and P risk thresholds (Table 4) were derived by dividing the allowable total load for the major subwatersheds reported in Linker et al. (1996) by their area. The areas of the major subwatersheds were estimated using U.S. Geological Survey (USGS) eight-digit hydrologic unit maps (Seaber et al. 1987). Assignment of USGS

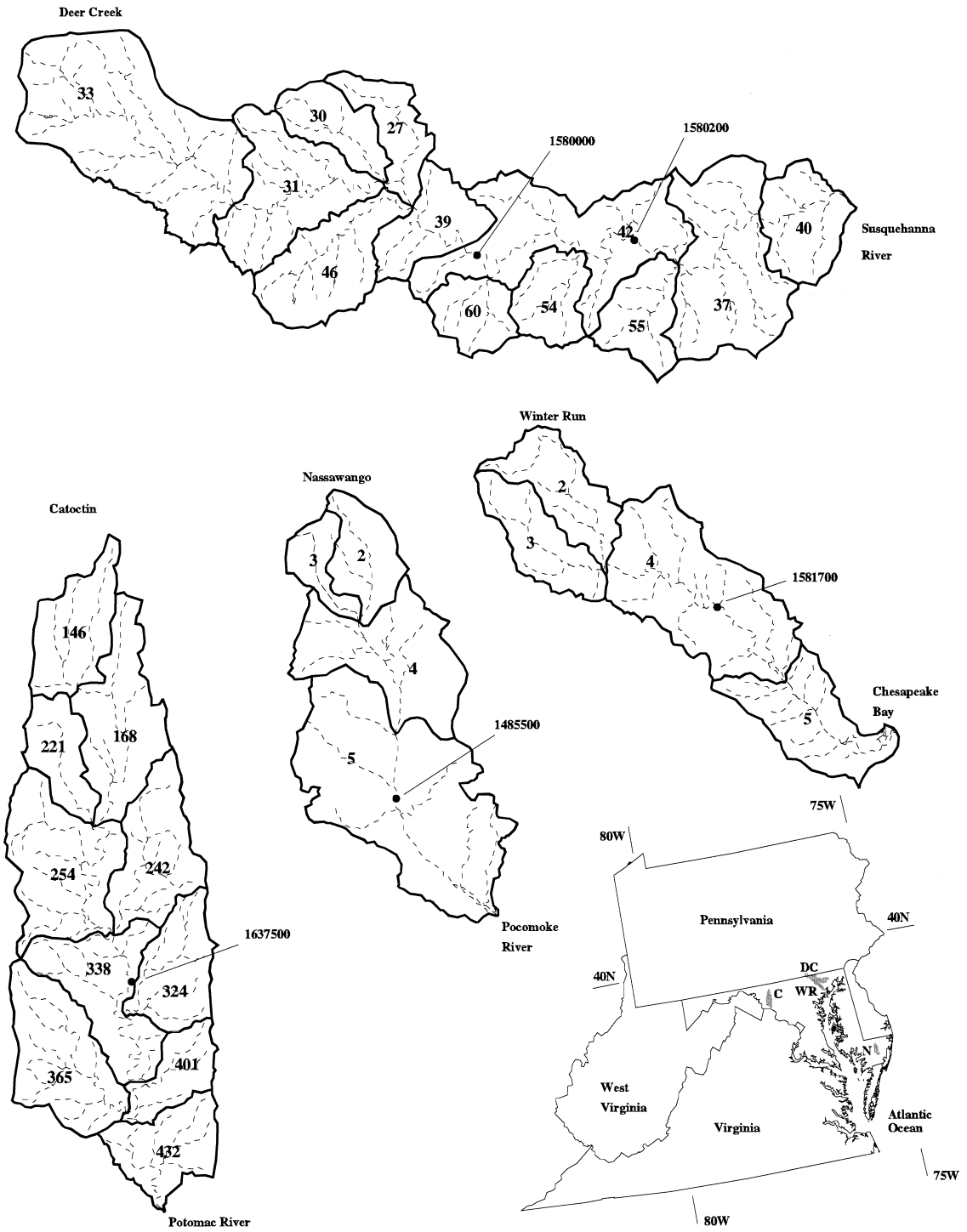

Figure 1. Location map. Bold lines are subwatershed boundaries and dashed lines are streams. Solid circles and associated numbers are the locations of gaging stations in the watersheds. Gaging station numbers match those listed in Table 1. The numbers in the subwatersheds are numerical labels. Locations of Deer Creek, Catoctin, Winter Run, and Nassawango watersheds are abbreviated DC, C, WR, and N, respectively, on the inset map. The watersheds drain into the water bodies identified at their termini.

Table 1. Discharges from monitored streams in study area.

Station Number	Watershed	$\bar{x}$	Q_{50}	Q_{50w}	Q_{50sp}	Q_{50su}	Q_{50f}
1485500	Nassawango	72	40	75	66	18	16
1580200	Deer Creek	192	147	167	185	130	104
1580000	Deer Creek	126	91	105	131	76	62
1637500	Catoctin	78	38	63	89	18	12
1581700	Winter Run	53	38	44	76	30	24

Note: Q_{50} values are annual, winter (w), spring (sp), summer (su), and fall (f) medians. $\bar{x}$ is the annual average. Discharge is in cubic feet per second (cfs). Station numbers match those in Figure 1. Data are from water.usgs.gov.

eight-digit hydrologic units to the Chesapeake Bay major subwatersheds was accomplished by visual comparison to the map in Donigian et al. (1994, Appendix E, page 62).

Simulated nutrient export was calculated after Reckhow et al. (1980):

$$N, P = \sum_{i=1}^{n} c_i A_i \qquad (1)$$

where c_i is a nutrient export coefficient for land-cover class *i*, and A_i is the area of land-cover class *i*. Multiplying c_i by the areal extent of its associated land-cover class and summing over all *n* land-cover classes in the watershed yields an estimate annual export from the watershed. Units are typically kilograms per year (kg/yr).

We used the Reckhow et al. (1980) data for forest, mixed agriculture, and urban to develop our risk model, and discarded the data on row crops, non-row crops, and grazed and pastured watersheds. We used the data for mixed agriculture because they were from watersheds that were 10^2 to 10^4 hectares and included both crop and pasture agricultural uses. The data for row crops, in contrast, were often from watersheds that were much less than one hectare, and the data for grazed and pastured watersheds were less than 12 hectares except for one observation.

We selected the data compiled by Reckhow et al. (1980) because it was from watersheds with homogeneous land-cover composition (except for five observations for mixed agriculture). Many nutrient export studies are dependent on the location of stream gauges, and the associated watersheds often include a mix of different land-cover classes (e.g., Dickerhoff Delwiche and Haith (1983) and Lowrance et al. (1985), Clesceri et al. (1986), Jordan et al. (1997), Fisher et al. (1998)). Inclusion of watersheds whose land cover is not homogeneous would have made it more difficult to isolate the effects of different land-cover classes on nutrient export risk.

Evaluation of nutrient export risk model performance

We evaluated the performance of the nutrient export risk model by comparing output from the N and P risk models to observed values of total nitrogen (TN) and total phosphorus (TP) reported in Jones et al. (2001). There were 44 sites for TN and 61 sites for TP, distributed from Pennsylvania south to Virginia. Adequacy and reliability were used to evaluate model performance (Gardner and Urban (in press)). Adequacy and reliability measure different aspects of the degree of congruency between the two lines (Figure 2). Adequacy is the ratio of the range of modeled output (**M**) to the range of observed output (**O**), using only the range of modeled output that is with the range of observed output. Adequacy is one (1) when the line representing M spans the entire range of the line representing O. Reliability is the ratio of the range of modeled output that is within the range of observed output to the entire range of modeled output. Reliability is one (1) if the range of modeled output does not extend beyond the range of observed output, i.e., the model does not produce unobserved predictions.

For each watershed, we retained the 5th (Q_{05}), 25th (Q_{25}), 50th (Q_{50}), 75th (Q_{75}), 95th (Q_{95}) percentiles, and minimum and maximum modeled values for comparison with the observed output, and measured adequacy and reliability using three different modeled data ranges: the inter-quartile range (ICR, Q_{25} to Q_{75}), Q_{05} and Q_{95}, and the minimum and maximum modeled values. We also estimated goodness-of-fit using bivariate regression of the 50th percentile (Q_{50}) versus observed.

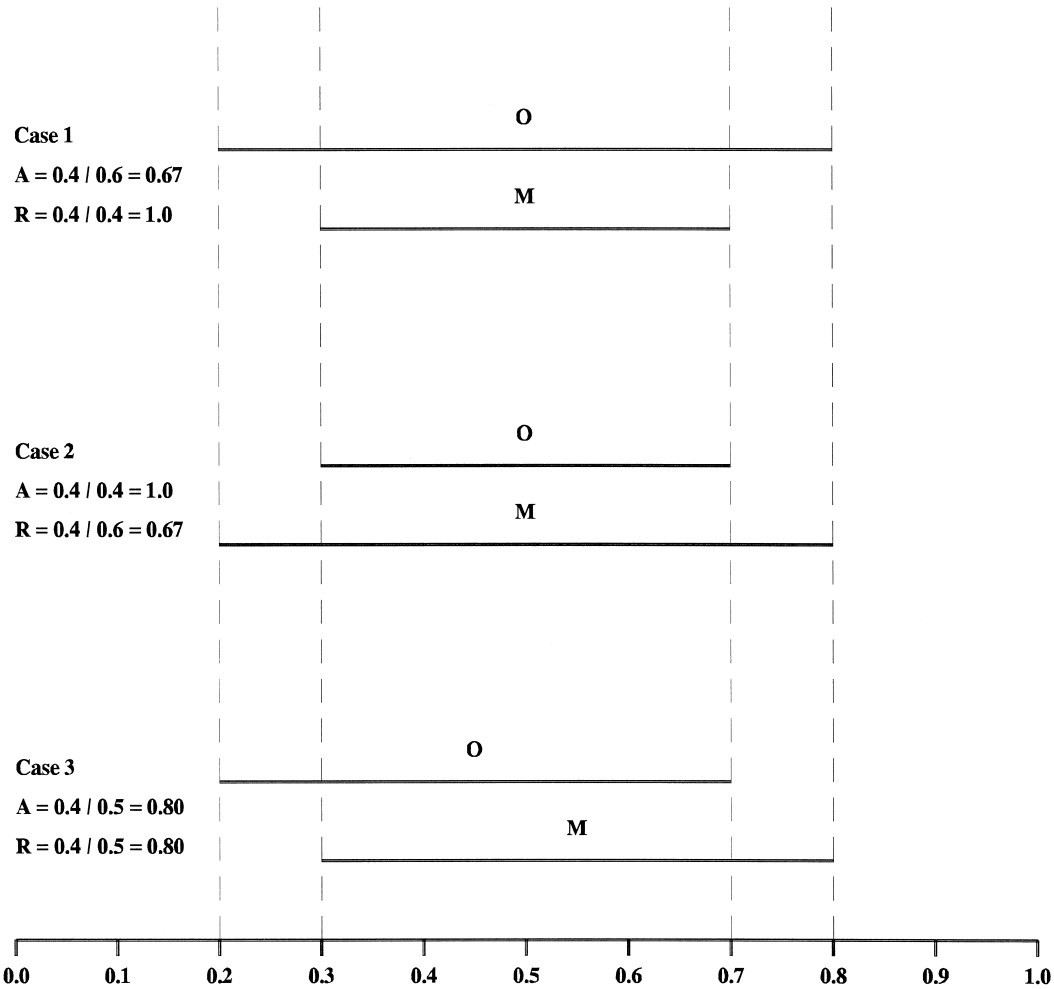

Figure 2. Three hypothetical examples of adequacy and reliability. **O** represents the observed data range and **M** represents the modeled data range. In case 1, the modeled results does not reproduce the entire range of observed data (adequacy < 1; reliability = 1). In case 2, the model produces results that are not observed (adequacy = 1; reliability < 1). In case 3, modeled and observed data ranges are offset (adequacy < 1; reliability < 1).

Propagating risk

Applying Equation 1 to the numbered subwatersheds in Figure 1 does not account for upstream contributions. To account for upstream contributions, the nutrient load was passed across subwatershed boundaries by keeping a running tally of loads accumulated from upstream subwatersheds in the estimation of loads for downstream subwatersheds. To include the nutrient export for an upstream subwatershed in the risk estimate for a downstream subwatershed, the output of Equation 1 for the upstream subwatershed was passed along the main stream channel of the downstream subwatershed, with some of the estimated load from upstream subwatershed removed due to instream processes. In other words, the accumulated load for any downstream subwatershed was that watershed's estimate (from Equation 1) plus any fraction from upstream subwatersheds not lost as the nutrient load passed through the main channel of the downstream subwatershed. Inclusions of fractional loads from upstream subwatersheds can be expressed as a modification of Equation 1 for downstream subwatersheds:

$$N, P = \sum_{i=1}^{n} c_i A_i + fA' \qquad (2)$$

where, fA' is the cumulative output from Equation 1 for one or more upstream subwatersheds that has not

Table 2. Proportion of N load delivered to a downstream point.[1]

Stream velocity			In-stream loss coefficients			
meters/sec.	feet/sec.	Travel time[2]	0.7595[3]	0.4768[4]	0.3842[4]	0.2981[4]
0.6096	2.00	0.285	0.805	0.873	0.896	0.919
0.4572	1.50	0.380	0.749	0.834	0.864	0.892
0.3048	1.00	0.569	0.648	0.762	0.803	0.844
0.1524	0.50	1.139	0.421	0.589	0.646	0.712
0.1000	0.33	1.736	0.268	0.437	0.513	0.595
0.0751	0.25	2.311	0.172	0.332	0.411	0.502

1. Proportion of in-stream nutrient load (L) delivered to a downstream point for a 15,000 meter stream reach, estimated as $L = e^{(-d \times T)}$, where d is in-stream loss coefficient and T is travel time (see Smith et al. (1997)).
2. Travel time (T) is per day (day^{-1}). T is measured as stream length divided by stream velocity, and converted to days.
3. In-stream loss coefficient for streams with discharge volumes $\leqslant$ 200 ft^3/s (Preston and Brakebill 1999).
4. In-stream loss coefficient for streams with discharge volumes $\leqslant$ 1,000 ft^3/s (Smith et al. 1997).

been lost along the stream channel, and c_iA_i is the contribution from the downstream subwatershed. For example, c_iA_i represents Deer Creek subwatershed 31 and fA' represents subwatershed 33 (see Figure 1).

Loss of N and P as they move downstream is a function of processes such as denitrification, uptake by aquatic biota, and sedimentation (Hill 1981; Burns 1998). Smith et al. (1997) have modeled these processes as a function of stream discharge and travel time. The amount of N and P delivered to a point from upstream is an exponential function of travel time multiplied by a loss coefficient. Travel time is estimated as the quotient of stream length divided by stream velocity, and converted to days. Velocity is estimated as a function of discharge (Leopold and Maddock 1953; DeWald et al. 1985). As discharge drops, velocity also declines (Leopold and Maddock 1953), travel time increases, and proportionally more of the discharge volume is in contact with the stream channel, resulting in greater losses of N and P (Table 2).

Accumulation of N and P from upstream watersheds was modeled using both high and low rates of loss (Smith et al. 1997; Preston and Brakebill 1999). Ranges for in-stream loss were framed as matched pairs of low velocity with high loss coefficients and high velocity with low loss coefficients (Table 3). High and low stream velocities were based on empirical velocity-discharge relationships in DeWald et al. (1985) that were representative of the discharges listed in Table 1. The higher stream velocity used for Deer Creek reflects the greater discharge volume in this watershed compared to the others.

Table 3. Matched pairs of stream velocity and loss coefficients.

	Loss Coefficient	
Stream Velocity[1]	N	P
0.0751 m/s	0.7595	0.3497
0.4572 m/s	0.2981	0.1885
0.6096 m/s	0.2981	0.1885

[1]The high stream velocity of 0.4572 m/s was used for the Catoctin, Nassawango, and Winter Run watersheds, and 0.6096 m/s was used for Deer Creek. A higher stream velocity for Deer Creek was chosen to reflect its higher discharge (see Table 1).

Comparison of the relative importance of land cover and in-stream decay

In-stream decay and subwatershed land-cover composition were dominant variables in the nutrient export risk models. The effect of each variable was estimated by setting it to zero (while keeping the other constant), and comparing that output with output where the variable was not set to zero (e.g., Gardner et al. (1987) and Wickham et al. (1999)). We examined the effect of in-stream decay by comparing results for the high rate of in-stream decay with results where in-stream decay was set to zero. Likewise, land cover was set to a conceptual zero or null state by assuming all subwatersheds were 100% forest (i.e., agriculture and urban do not produce greater nutrient export than forest), and compared with results using the actual land-cover proportions. In-stream decay was set to zero for the comparison of 100% forest to actual land-cover composition. The relative importance of land-cover composition versus in-stream decay was exam-

Table 4. Risk Thresholds.

Major Sub-Watersheds of the Chesapeake Bay	Allowable Total Loads (Linker et al. 1996) (kg/yr × 10^6)		Area (ha)	Risk Thresholds for N and P Derived from Allowable Total Loads (kg/ha/yr)		Associated Watersheds in this Study
	N	P		N	P	
Susquehanna	54.6	1.98	6,852,284	7.97	0.289	Deer Creek
Potomac	27.2	2.03	3,900,533	6.97	0.520	Catoctin
Western Shore	8.3	0.39	864,736	9.60	0.451	Winter Run
Eastern Shore	10.1	0.73	1,090,503	9.26	0.669	Nassawango

Table 5. Nutrient Export Model Performance.

Comparison Nitrogen	Adequacy	Reliability
Obs. vs ICR	0.51	1.00
Obs. vs $Q_{05} - Q_{95}$	0.79	1.00
Obs. vs. min./max.	0.98	0.97
Phosphorus		
Obs. vs ICR	0.67	0.97
Obs. vs $Q_{05} - Q_{95}$	0.83	0.81
Obs. vs. min./max.	0.93	0.73

Nutrient	R-square	Model
Nitrogen	0.55	$O = -1.60 + 2.0(Q_{50})$
Phosphorus	0.43	$O = 0.35 + 1.1(Q_{50})$

ined by comparing the output from the two null comparisons.

Results

Nutrient risk model performance

Adequacy ranged from 0.51 to 0.98 for N, and 0.67 to 0.93 for P as the amount of modeled data included in the comparison increased from the inter-quartile range (ICR) to the minimum and maximum (Table 5). As expected, adequacy improved when the observed data was compared to a larger range of the modeled data. Reliability changed from 1 to 0.975 for N, and 0.972 to 0.73 for P as the modeled range increased from the ICR to the minimum and maximum. Modeled values outside the range of observed values increased as the range of modeled data used increased, especially for P. Comparing to the hypothetical cases in Figure 2, model performance for N matches case 1 most closely and model performance for P matches case 3 most closely.

Goodness-of-fit between the observed and the median value (Q_{50}) from the risk models was about 0.55 for N and 0.43 for P (Table 5). Our goodness-of-fit values are about equivalent to those found by Jones et al. (2001) between the observed values and land-cover proportions. The data we used to estimate risk were similar to the modeled results for the Chesapeake Bay (Donigian et al. (1994), see also Table 1 in Wickham et al. (2002)).

Upstream-to-downstream changes in risk

Estimated risk was dependent on the threshold (Figures 3a–3d and 4a–4d). The dramatic difference in P risk between Nassawango and Winter Run is partly due to the nearly 50% difference in the P risk threshold between the two drainage basins (see Table 4). Risk is inversely related to the rarity of the event (O'Neill et al. 1982), and hence more common events (lower thresholds) yield higher estimates of risk.

Upstream-to-downstream changes in land-cover composition had a noticeable impact on risk. For all rates of in-stream decay, Nassawango showed downstream increases in risk for N and P because of an approximate 20 percent decline in forest from subwatershed 4 to subwatershed 5 (Table 6). For Catoctin, a downstream decrease in forest cover resulted in a downstream increase in risk.

Except for Nassawango, high rates of in-stream decay resulted in upstream-to-downstream declines in nutrient export risk. High rates of decay superseded the risk posed by non-forest land cover. Nutrient export risk often declined toward zero under high in-stream decay.

Results for high in-stream decay show dramatic declines in two cases for P export risk. P export risk dropped from about 0.75 to 0.25 between Deer Creek subwatersheds 39 and 42, and P risk dropped from about 0.25 to nearly zero between Catoctin subwatersheds 401 and 432. In both cases, the sharp drop in risk was the result of the combined effect of high decay and long streamlengths. Deer Creek subwatershed 42 and Catoctin subwatershed 432 had streamlengths that were 2 to 4 times greater than the upstream subwatersheds draining into them (see Table 6). The long streamlength combined with the longer travel times associated with high decay resulted in a dramatic decline in risk.

N and P export risk estimates were substantially lower when in-stream decay was included in the model. On average, the difference between low and no decay was 0.12 for N and 0.03 for P, and the difference in risk estimates between high and no decay was 0.56 for N and 0.42 for P. Comparison of the null cases of no in-stream decay and 100% forest indicated that land-cover composition had a greater influence on nutrient export risk than the rate of in-stream decay (Figure 5). On average, the null case for land cover (actual land-cover composition and no in-stream decay versus 100% forest and no in-stream decay) resulted in a 4% greater reduction in risk for

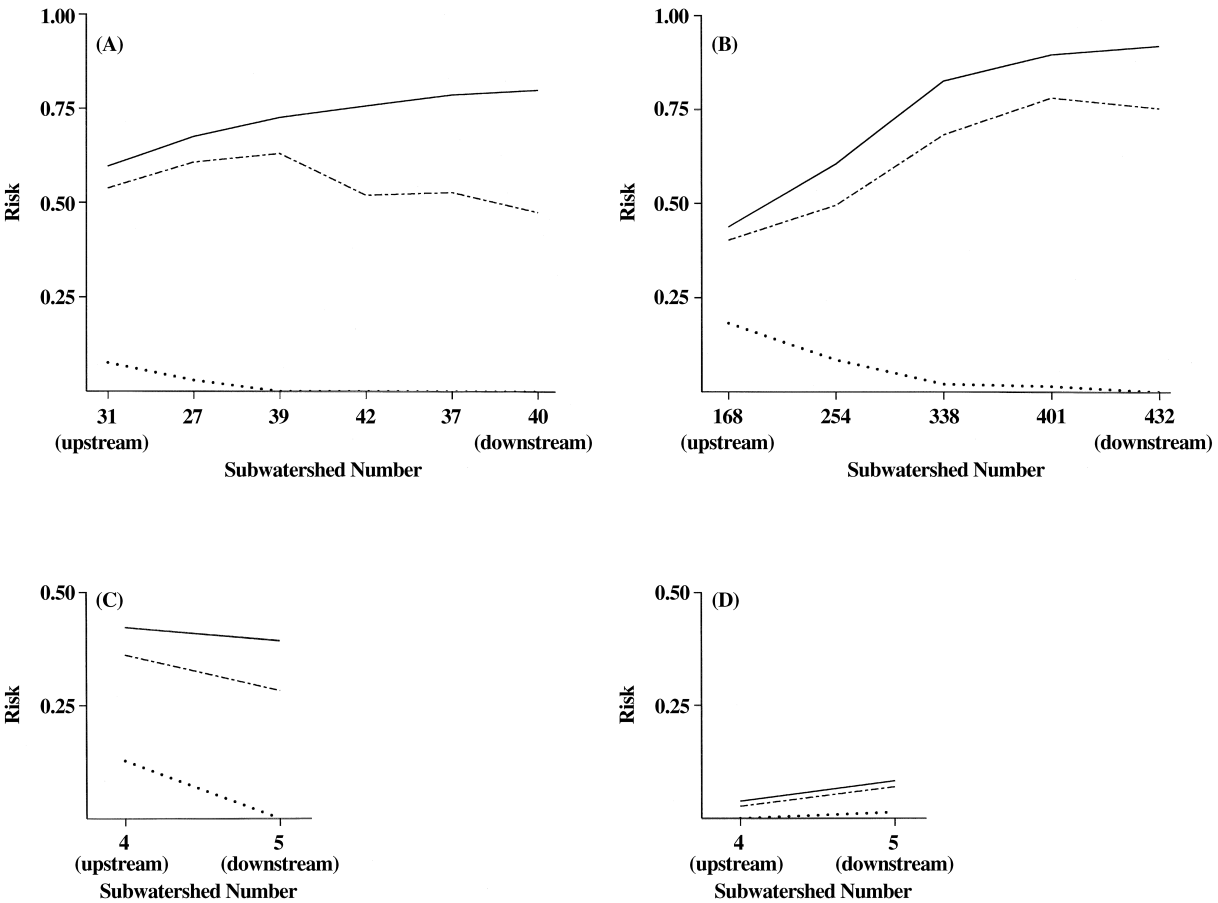

Figure 3. N export risk estimates for Deer Creek (A), Catoctin (B), Winter Run (C), and Nassawango (D). Y-axes are risks of exceeding N export thresholds reported in Table 4. X-axes are the subwatershed numbers in Figure 1, organized in an upstream-to-downstream order. Solid lines are risks with no in-stream decay. Dashed lines are risks with low in-stream decay, and dotted lines are risks with high in-stream decay.

N and a 30% greater reduction in risk for P than the null case for in-stream decay (actual land-cover composition and no in-stream decay versus actual land cover composition and high in-stream decay).

Discussion

The evaluation of nutrient export risk model performance was dependent on the data used to represent the observed (**O**) and modeled (**M**) sets. The observed data we used were based on five-year averages (Jones et al. 2001). Treating each year as a different observation would have likely increased the range of observed values, since average annual loads often vary significantly from year to year (see, for example, Table 3 in Fisher et al. (1998)). Likewise, the variance in simulated values could have been increased by retaining values outside the range of the Reckhow et al. (1980) data that we used to drive the risk models. It is plausible that making these changes to both the observed and simulated values would have yielded model performance results matching hypothetical case 2 in Figure 2, i.e., optimizing adequacy at the expense of reliability. Optimizing adequacy at the expense of reliability may be appropriate for estimating nutrient export risk. Inter-annual variability in nutrient export and lack of a well developed monitoring network suggest that actual nutrient export variability may be greater than what has been measured.

Many factors influenced risk in this study, including land-cover composition, subwatershed area, location of subwatershed boundaries, streamlength, stream velocity, in-stream decay rate, and the risk threshold. Changing the rate of in-stream decay often produced diverging downstream trends in risk that

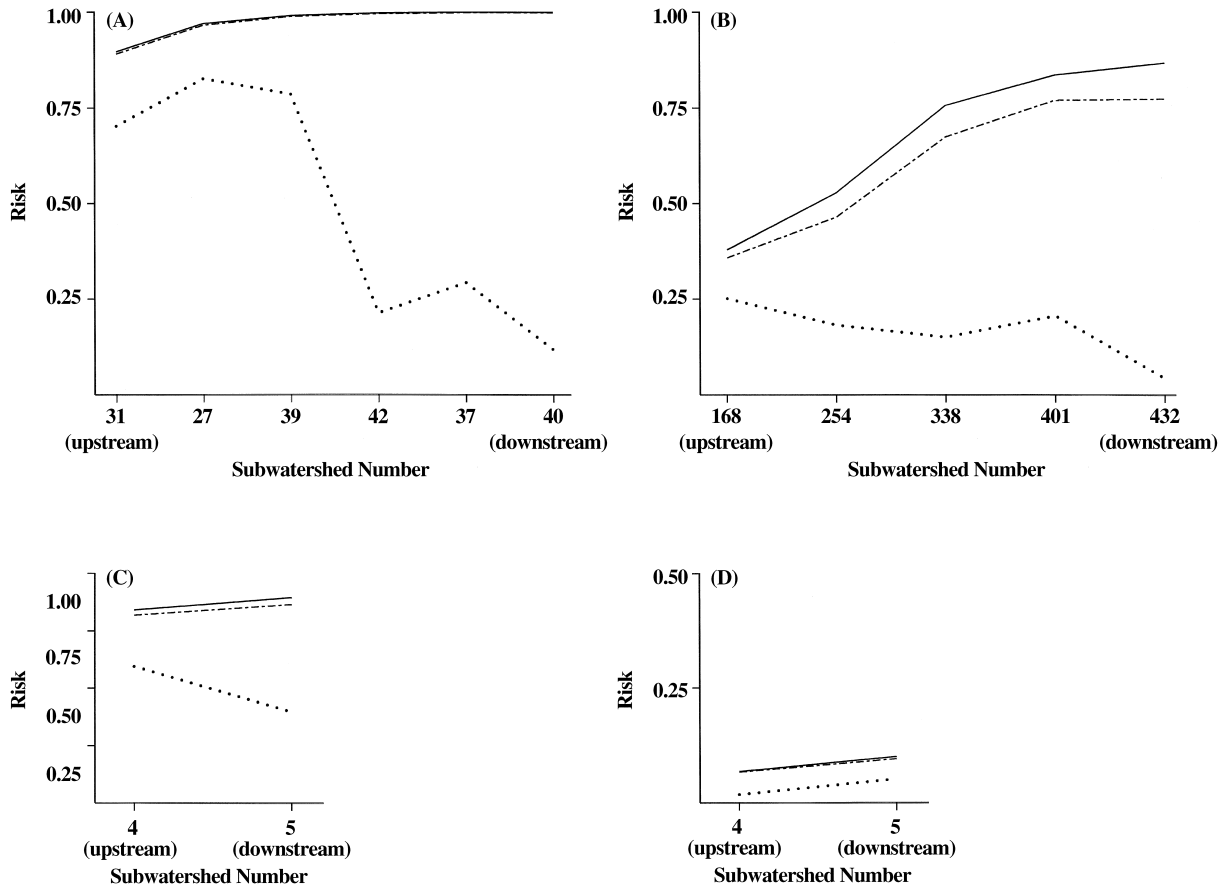

Figure 4. P export risk estimates for Deer Creek (A), Catoctin (B), Winter Run (C), and Nassawango (D). Y-axes are risks of exceeding P export thresholds reported in Table 4. X- axis and line symbols are the same as in Figure 3.

were modified by subwatershed land-cover composition and streamlength. One or more of these factors and interactions among them influenced changes in risk across subwatersheds.

Variance in in-stream decay rates both within and across studies (Smith et al. 1997; Preston and Brakebill 1999; Alexander et al. 2000) likely reflects natural variability in nutrient removal from streams (Alexander et al. 2001; Peterson et al. 2001) and natural variability in the relationship between stream discharge and stream velocity (DeWald et al. 1985). Decay coefficients are surrogates for the biotic and abiotic processes that remove nutrients from streams (Burns 1998). There are few studies of N and P transformation and most have concentrated on denitrification (e.g., Hill (1979, 1981) and Sjodin et al. (1997), Burns (1998)). We used matched pairs of low stream velocity with high in-stream decay and high stream velocity and low in-stream decay to capture this variability. Use of velocity and in-stream decay in our study highlights the impact of knowledge uncertainty (sensu Hession et al. (1996)) on estimating risk.

Despite the uncertainties we incorporated in modeling nutrient export risk, in-stream decay implies that restoration activities targeted to reduce nutrient export (e.g., reforestation) should start at downstream portions of the watershed and move upstream. If downstream portions are forested, that portion of the drainage basin will contribute little to the load exported while permitting in-stream processes to remove nutrients introduced upstream. Such a restoration strategy assumes management for average annual conditions with the goal of reducing nutrient input into larger streams, where nutrients tend to be conserved (Smith et al. 1997). All of the watersheds studied here, with the possible exception of Nassawango, represent places to test this restoration strategy, since each has low proportions of forest for downstream subwatersheds and each drains directly to a large stream (Sus-

Table 6. Land-cover proportions and streamlengths for downstream subwatersheds.

Deer Creek			Land-cover Proportions		
WS#	Area (ha)	Streamlength (m)	Forest	Agriculture	Urban
31	4431	14475	0.447	0.547	0.000
27	1906	2711	0.268	0.716	0.001
39	2357	6148	0.530	0.457	0.005
42	6281	24278	0.521	0.465	0.003
37	5833	8048	0.356	0.620	0.017
40	2615	7927	0.414	0.573	0.001
Catoctin					
168	3949	11909	0.615	0.381	0.001
254	4495	13969	0.461	0.514	0.021
338	3666	14190	0.218	0.766	0.014
401	1694	4303	0.224	0.759	0.016
432	2254	8141	0.293	0.692	0.012
Winter Run					
4	6689	18060	0.375	0.495	0.124
5	3427	9498	0.430	0.386	0.143
Nassawango					
4	4646	7524	0.824	0.130	0.005
5	9816	13185	0.651	0.294	0.004

quehanna, Potomac, Pocomoke) or the Chesapeake Bay.

Summary and conclusion

In-stream decay rate, land-cover composition, and streamlength were important factors for estimating upstream-to-downstream changes in N and P risk. N and P risk increased (with one exception) toward downstream subwatersheds under low in-stream decay. On average, the difference in risk between high and low in-stream decay was 0.44 for N and 0.39 for P. Changes in the proportion of forest across subwatersheds also influenced risk, and interacted with streamlength and in-stream decay to produce dramatic changes in risk for some subwatersheds. High in-stream decay reduced risk, on average, by 0.56 for N and 0.42 for P, when compared to the null case of no in-stream decay. The null case of 100% forest with no in-stream decay reduced risk to zero (0) in most cases.

Nutrient export coefficients have typically been developed for watersheds ranging in size from 10^2 to 10^4 hectares, and the continuing development of GIS and higher resolution digital elevation data have fostered the delineation of smaller watersheds in favor of what can be attained nationally (Seaber et al. 1987). One goal behind the creation of smaller watersheds is to provide units better suited to implementation of management options (U.S Department of Agriculture (USDA), National Resource Conservation Service (NRCS) 2001). At 10^2 to 10^4 hectares, watersheds are often upstream-to-downstream components of larger drainage systems, as illustrated by the watersheds used in this study. Propagation of risk to downstream subwatersheds, especially under lower rates of in-stream decay, suggests that these small watersheds are not independent management units where nutrient export issues are concerned.

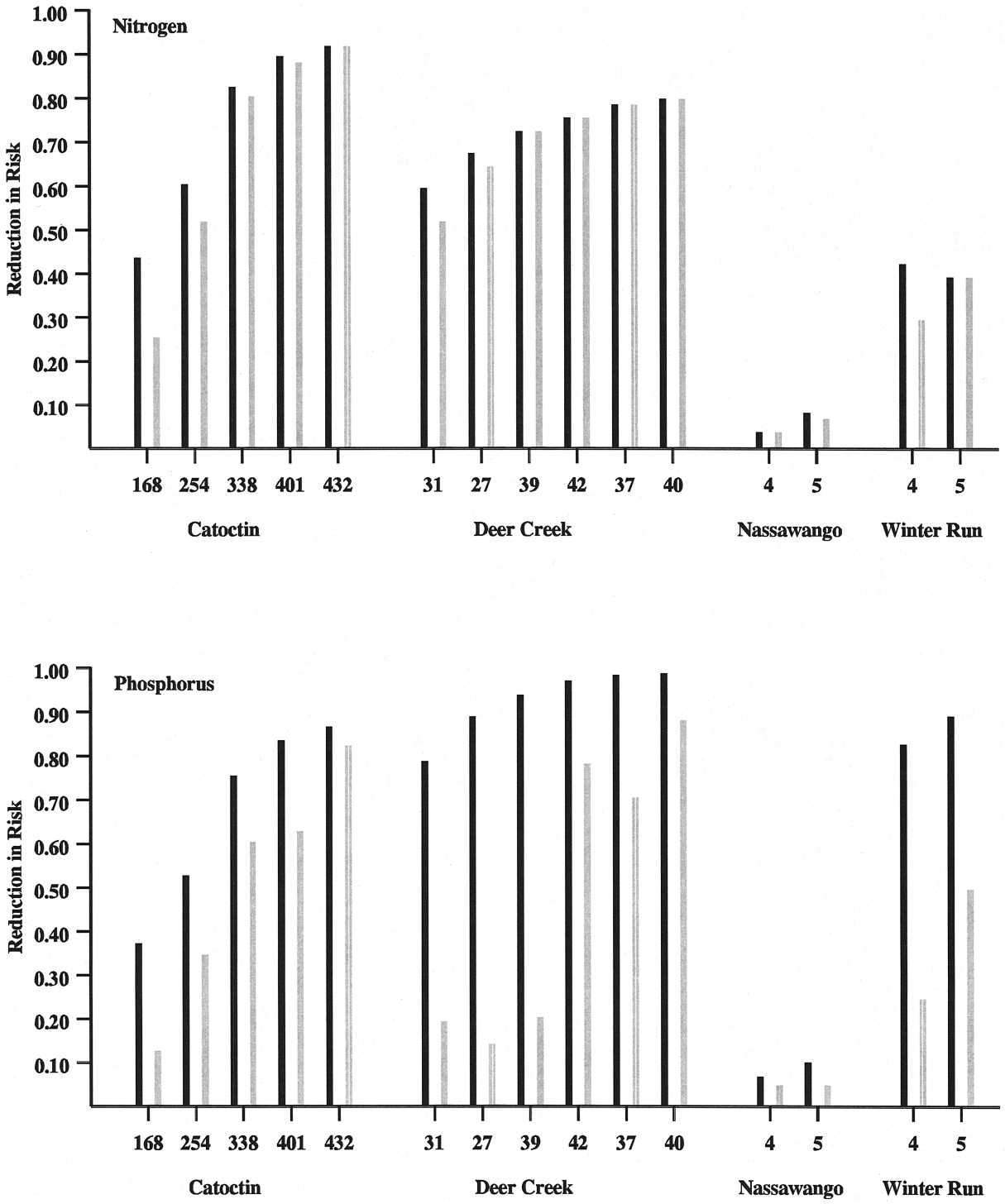

Figure 5. Comparison of the relative importance of in-stream decay and land-cover composition in reducing nutrient export risk. Black bars are the difference in risk between actual land-cover composition and no in-stream decay versus homogeneously forested subwatersheds and no in-stream decay. Gray bars are the difference in risk between no in-stream decay and actual land-cover composition versus high in-stream decay and actual land-cover composition.

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