

Landscape Planning for Agricultural Nonpoint Source Pollution Reduction III: Assessing Phosphorus and Sediment Reduction Potential

Matthew W. Diebel · Jeffrey T. Maxted ·
Dale M. Robertson · Seungbong Han ·
M. Jake Vander Zanden

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Abstract Riparian buffers have the potential to improve stream water quality in agricultural landscapes. This potential may vary in response to landscape characteristics such as soils, topography, land use, and human activities, including legacies of historical land management. We built a predictive model to estimate the sediment and phosphorus load reduction that should be achievable following the implementation of riparian buffers; then we estimated load reduction potential for a set of 1598 watersheds (average 54 km²) in Wisconsin. Our results indicate that land cover is generally the most important driver of constituent loads in Wisconsin streams, but its influence varies among pollutants and according to the scale at which it is measured. Physiographic (drainage density) variation also influenced sediment and phosphorus loads. The effect of historical land use on present-day channel erosion and variation in soil texture are the most important sources of phosphorus and sediment that riparian buffers cannot attenuate. However, in most watersheds, a large proportion (approximately 70%) of these pollutants can be eliminated from streams with buffers. Cumulative frequency distributions of load reduction potential indicate that targeting pollution reduction in the highest 10% of Wisconsin watersheds would reduce total

phosphorus and sediment loads in the entire state by approximately 20%. These results support our approach of geographically targeting nonpoint source pollution reduction at multiple scales, including the watershed scale.

Keywords Riparian buffers · Nonpoint source pollution · Land cover · Phosphorus · Sediment · Prioritization

Introduction

Excessive amounts of sediment and nutrients (phosphorus and nitrogen) are delivered from agricultural land to streams (Carpenter and others 1998). These nonpoint source pollutants degrade aquatic ecosystems and impair water use by humans (USEPA 2000a). While land management practices, such as reduced fertilizer application and buffer strips, can help mitigate these problems, there is a need to target these practices to the places where they will be most effective. Approaches have been developed for targeting management within small watersheds (e.g., Tomer and others 2003), but not among watersheds at broader geographic scales (state or province).

Targeting pollution reduction on a landscape scale is worthwhile because watershed-unit-area pollutant export varies by up to several orders of magnitude among streams across the United States (Omernik 1977) and within smaller regions, such as states (Corsi and others 1997) and ecoregions (Dodds and Oakes 2004). These findings indicate that nonpoint source pollution control in watersheds with extremely high export rates may most efficiently reduce pollution on a regional scale. However, it is unclear whether available management practices are capable of dealing with the worst pollution problems (Wolf 1995).

M. W. Diebel (✉) · J. T. Maxted · M. J. Vander Zanden
Center for Limnology, University of Wisconsin,
680 N. Park Street, Madison, WI 53706, USA
e-mail: mwdiebel@wisc.edu

D. M. Robertson
US Geological Survey, Wisconsin Water Science Center,
8505 Research Way, Middleton, WI 53562, USA

S. Han
Statistics Department, University of Wisconsin,
1300 University Avenue, Madison, WI 53706, USA

The installation of riparian buffers is a common practice for restoring streams that have been degraded by agriculture. Their implementation is advocated and funded by state and federal conservation programs, including those supported by the federal Farm Bill. However, while buffers have generally been shown to be effective at reducing sediment and nutrient loading to streams (summarized in Wenger 1999), their effectiveness depends on landscape setting (Baker and others 2001). Local physical characteristics such as soils, topography, land use, and human activities, including legacy effects of historical conditions, may render a riparian buffer less effective in some settings than in others (Lowrance and others 1997). Furthermore, in a given setting, this potential effectiveness may differ among water quality constituents (Lowrance and others 1997).

In this article, we present an approach for evaluating the water quality restoration potential of riparian buffers. Specifically, we develop a simple model to estimate the expected load reduction of phosphorus and sediment in a stream, following intensive implementation of riparian buffers in that stream's watershed. Using a combination of geographical datasets and statistical methods, model parameters are estimated for a set of 1598 small watersheds that were developed for a nonpoint source pollution control program in Wisconsin (Maxted and others in review; UW-CALS 2005). Estimates of load reduction potential are then used to rank watersheds in a geographical prioritization scheme. The primary component of the model is an estimate of current constituent loads, which were estimated with regression models of landscape characteristics on measured loads. In developing these regressions, the relative influences of land cover and physiographic variables (and their interaction) on constituent loads were explored. The influence of the scale at which land cover is assessed was also examined, and used to infer constituent-specific transport and retention processes. The broad objective of this article is to offer a framework for prioritizing conservation efforts in agricultural landscapes. Efficient allocation of restoration funds should take into account both the magnitude of the problem and the degree to which available conservation tools can address it.

Methods

Model Structure

Restoration potential was defined as the expected reduction in the average annual watershed-unit-area load of sediment or phosphorus in a stream, following implementation of riparian buffers in that stream's watershed. Phosphorus and sediment were the focus because their effects on freshwater

ecosystems are typically stronger than those of nitrogen (Allan 1995) and because their sources and transport are governed by similar factors (Peterjohn and Correll 1984). The models were based on loads rather than concentrations because loads better integrate the often skewed distributions of pollutant concentrations in streams (e.g., Corsi and others 2005).

A simple model was constructed to estimate sediment and phosphorus loads in streams and to estimate the load reduction achievable through the implementation of riparian buffers. The model is:

$$R_c = L_c - U_c \quad (1)$$

where R is the load reduction potential for constituent c (suspended sediment or phosphorus), L is the current annual load of c , and U is the sum of unbufferable (defined below) sources of c . This model simulates load reduction under the scenario of full implementation of riparian buffers in the watershed. The model is based on the following assumptions: (1) all transport of dissolved and particulate phosphorus is in surface runoff (i.e., no groundwater transport), (2) use of a variable width buffer network, where width at any given location is proportional to the drainage area of that point (Tomer and others 2003), and (3) that this buffer network is capable of trapping all upland-derived nonpoint sources of phosphorus and sediment except that portion which is unbufferable as determined by soil texture (see details below). No assumption was made regarding buffer vegetative composition (i.e., trees, shrubs, grasses). Rather, it was assumed that this decision would be made based on local site conditions and management goals other than the improvement of water quality (see discussion). These assumptions create a best-case scenario for buffer effectiveness, but wherein spatial bias of potential inefficiencies is minimized. In other words, with realistic (i.e., imperfect) buffer implementation, actual load reduction may be lower than the potential values estimated herein, but the magnitude of the difference is likely to be similar everywhere.

Model parameters were estimated for a set of 1598 watersheds in Wisconsin as part of a nonpoint source pollution control program called the Wisconsin Buffer Initiative (WBI) (UW-CALS 2005; Maxted and others in review; Diebel and others in review). These watersheds were delineated to create a set of candidate landscape units in a geographical prioritization scheme (Maxted and others in review). One of the management goals of this process was to select watersheds where implementation of riparian buffers would produce the largest reductions in sediment and phosphorus loads (i.e., maximize R_c). An alternative approach is to select watersheds based on the proportion of current load that can be eliminated (i.e., maximize R_c/L_c)

(see discussion). The WBI provides a case study for implementation of the general approach, which should be applicable in any agricultural region. Because some of the model parameters may be more or less important in different regions, and different datasets or analytical techniques may be available for calculating these parameters, the following should be seen as an example of the general approach.

Current Loads

Annual loads for the 1598 WBI watersheds were predicted with linear regression relations between measured annual loads for suspended sediment, total phosphorus, and total nitrogen (described below) and specific landscape characteristics. Although reduction potential was not estimated for nitrogen, regression analyses were included for this constituent because its relationships with landscape characteristics were different than those of phosphorus and sediment, providing an interesting contrast.

Water-Quality Data

Water-quality data for this analysis were assembled from data collected by the United States Geological Survey (USGS) and the Wisconsin Department of Natural Resources (WDNR). USGS data were retrieved from the National Water Information System (NWIS). WDNR data were obtained from the United States Environmental Protection Agency (USEPA) legacy and modernized Storage and Retrieval (STORET) database or directly from the WDNR. For each constituent, included stream sites had at least two years of record and a total of at least 25 concentration samples collected during 1970–2002. Included sites also had at least five years of complete daily streamflow records from a gage at the site or a nearby site. A nearby gage is defined as a gage on the same stream or a nearby stream with a drainage-area ratio (water-quality station area divided by gaged area) between about 0.5 and 1.5.

Annual loads (calculated by summing daily loads) were estimated by a regression approach by use of the Fluxmaster program (Schwarz and others 2006). In this study, estimated daily loads (L) were computed on the basis of relations between constituent load (in kgs) and three variables: streamflow (Q , in m^3/d), time of the year (T , in radians), and $DECTIME$ (years in decimal format, used to adjust for temporal trends at specific sites). The general form of the model is:

$$\ln(L) = a + b[\ln(Q) - c] + d[\sin(T)] + e[\cos(T)] + f[DECTIME] \quad (2)$$

Values for the regression coefficients (a , b , c , d , e , and f) were computed for each site by the use of multiple-regression

analyses between daily loads (daily average streamflows multiplied by instantaneously measured concentrations, in mg/L) and Q , T , and $DECTIME$. Because a natural logarithmic transformation was used in Eq. 2, daily loads were adjusted to account for a retransformation bias by use of the minimum variance unbiased estimate procedure (Cohn and others 1989). Average loads were computed for each site, and then normalized to drainage area to produce an annual yield ($\text{kg}/\text{km}^2/\text{year}$). To maintain site independence, when the watersheds of two or more sites overlapped, only the site with the longest and/or most recent period of record was used. A total of 116 sites were used, of which 39 had nitrogen data, 61 had phosphorus data, and 72 had suspended sediment data.

Landscape Characteristics

Watersheds for each stream site were delineated from 1:24,000-scale hydrography (WDNR 2003) and a 30m digital elevation model (DEM) (USGS 1999) using ArcHydro (Maidment 2002). Then, using the Analytical Tools Interface for Landscape Assessment (Ebert and Wade 2004) extension in ArcView (ESRI, Redlands, CA), landscape characteristics were calculated at several scales that were expected to influence water quality. WiscLand (WDNR 1998) level 1 (coarse classification) land cover data were used to calculate percentages of each land cover type in the watershed, in 30 and 90 m wide riparian zones along all 1:24,000-scale streams, in the same zones along 2nd order and larger streams (hereafter, “large stream riparian zone”), and for agricultural land cover, on land with slopes greater than 3, 6, and 9%. Average watershed soil erosivity (k factor) was calculated from the U.S. Department of Agriculture (USDA) STATSGO soil layer for Wisconsin (USDA 2005). Drainage density was calculated as the total length of streams in the watershed divided by the watershed area. Mean land slope in each watershed was calculated from a slope grid built from the 30 m DEM. Annual rates of atmospheric nitrate deposition were estimated by kriging recent (1980–1996) average rates from National Atmospheric Deposition Program (2006) monitoring stations in and around Wisconsin. Summary statistics for all landscape characteristics are given in Table 1.

Statistical Methods

Determining the direct effects of specific landscape characteristics on stream water quality is complicated by the interrelatedness of many of the characteristics (King and others 2005). This lack of independence can lead to many different models having similar predictive abilities, thereby concealing the true drivers of water quality. This analysis does not attempt to find the single best model for all interpretive and predictive purposes. Instead, separate

Table 1 Candidate variables tested in regression models for sediment, phosphorus, and nitrogen

Basin characteristics					
Mean soil erosivity (k factor)	0.26 ± 0.08				
Watershed area (km ²)	616 ± 784				
Drainage density (km/km ²)	0.93 ± 0.36				
Mean land slope (%)	3.0 ± 2.2				
Atmospheric nitrate deposition (kg/ha/year)	11.4 ± 2.1				
Scale/Land cover category (%)	Forest	Wetland	Urban	Grassland	Cropland
Watershed	28.2 ± 25.3	9.5 ± 8.3	3.1 ± 9.5	11.3 ± 6.7	45.8 ± 28.5
Riparian zone (90 m)	25.6 ± 21.3	20.6 ± 16.4	2.2 ± 6.6	11.0 ± 7.4	39.0 ± 26.2
Riparian zone (30 m)	23.9 ± 20.1	27.4 ± 20.6	1.9 ± 5.8	10.1 ± 7.1	35.4 ± 25.3
Large stream riparian zone (90 m)	24.5 ± 21.1	25.7 ± 17.8	1.7 ± 3.5	9.8 ± 6.5	35.0 ± 25.7
Large stream riparian zone (30 m)	22.4 ± 19.4	34.3 ± 22.6	1.4 ± 3.0	8.4 ± 6.2	30.6 ± 25.1
Land slope >3%	NA	NA	NA	4.2 ± 3.8	12.0 ± 12.7
Land slope >6%	NA	NA	NA	1.6 ± 2.2	3.5 ± 5.5
Land slope >9%	NA	NA	NA	0.6 ± 1.1	1.1 ± 2.2

Reported values are means ± 1 standard deviation for all model-building sites

approaches, all based on regression, are used to address three objectives:

1. Cropland and grassland are expected to be positively correlated with all three constituents (Much of the grassland in Wisconsin is pasture, but is indistinguishable from undisturbed grassland in land cover datasets). However, the relative influence of these land-cover classes on different constituents, and the scale at which they exert their strongest influence, is uncertain. Therefore, simple regressions were performed on cropland and grassland at three spatial scales (watershed, 90 m riparian, and 30 m riparian) for each constituent. The predictive ability of multiple regression models composed of variables selected from watershed-scale and riparian-scale candidate sets were also compared.
2. While many large-scale studies have examined the influence of land cover on stream water quality, few have addressed the influence of physiographic variables. This is surprising, considering the emphasis given to such variables as slope and soil type in small-scale models such as those using the Universal Soil Loss Equation (Wischmeier and Smith 1978). Therefore the amount of additional predictive power gained by adding physiographic variables to land-cover-only models at watershed and riparian scales was evaluated.
3. For predictive purposes, a best subsets model selection with all candidate variables was used to identify the best (highest coefficient of determination [r^2] among models with all variance inflation factors < 2) models for each constituent whose structures (variable

coefficient signs) were consistent with the present knowledge of landscape processes. These models were used to predict current annual loads of phosphorus (L_P) and sediment (L_S) for the WBI watersheds.

For all regressions, the response variables (loads) were $\log_{10}$ transformed to improve normality. Several predictor variables were also transformed (square root, $\log_{10}$, inverse) to improve their normality. To evaluate the relative influence of individual variables in the multiple regressions, partial correlation coefficients were computed as the correlation between each variable and the residuals of a regression with the other variable(s). All regressions were performed in Minitab 13.3.

Unbufferable Sources

Unbufferable sources (U_c) are defined as any component of a constituent load that cannot be attenuated using buffers between agricultural land and streams. These include meander-belt erosion, fine-textured soils, point-source discharges, and urban stormwater. The magnitude of all of these sources varies among regions in Wisconsin, making them useful for targeting (Hyman and Leibowitz 2000). Watersheds that have a high proportion of unbufferable phosphorus or sediment sources have less water quality restoration potential using riparian buffers.

Meander-Belt Erosion

The annual sediment load of a stream is derived from both upland and channel sources. Channel sources may be divided

into two components: that which originates from devegetated banks and that which originates from meander-belt migration. Sediment losses from devegetated banks can be reduced using buffers (Zaimes and others 2004). In contrast, meander-belt migration is a natural process whereby streams erode sediments from their floodplains as they adjust their course. The amount of sediment eroded through this process is determined, in part, by the position of the stream channel in relation to the floodplain. Incised streams and those with aggraded floodplains are more prone to severe sediment losses because of the increased erosive power of confined high flows (Trimble 1997a).

In some areas of Wisconsin, particularly the Driftless Area ecoregion (southwestern Wisconsin), the combined effects of poor past land management, steep slopes, and erodible soils created severe floodplain aggradation. This sediment source is, for the most part, not reducible using buffers (Trimble 1993). At Coon Creek, a typical agricultural Driftless Area stream, meander-belt erosion produces approximately 25% of the annual sediment load (J.C. Knox, UW-Madison Geography Department, written communication, 2002). Through a similar geomorphic process in a northern Wisconsin Lake Superior tributary stream, a combination of bluff erosion and channel down-cutting contributes as much as 90% of the annual load (Fitzpatrick and others 1999). In contrast, the banks of streams in low gradient, forested basins can contribute less than 5% of the load (Walling and Kane 1984).

Despite the scarcity of quantitative estimates of this process, meander-belt erosion was recognized as an important and highly variable process. It was therefore estimated as follows: for the Lake Superior watersheds that cross the topographic break associated with a post-glaciation Lake Superior shoreline, a constant value of 90% of the estimated sediment load was assigned. Based on the pronounced meander-belt erosion common in the Driftless Area (Trimble 1993), it was assumed that the combination of agricultural land cover and the physiographic patterns of this area (characterized by high drainage density) were its primary drivers. A linear regression was then fit where the intercept was equal to 5% (expected lower limit for meander-belt sediment erosion), the slope variable was assumed to be equal to (watershed drainage density $\times$ percent agricultural land cover in the 30 m riparian zone) ($D \times A$), and the slope coefficient was fit to the values of these variables in the Coon Creek watershed. Therefore, meander-belt sediment erosion as a proportion of the total load (C_S) was computed for each WBI watershed using the resulting equation: $C_S = 0.002 \times (D \times A) + 0.05$.

Because floodplain soils are often derived from erosion of upland agricultural areas, they may contain significant amounts of phosphorus. Therefore, entrainment of these sediments through erosion will also deliver phosphorus to

the stream. Though the phosphorus content of these soils surely varies considerably, a study in an agricultural part of Minnesota (Sekely and others 2002) found that in a river where approximately 37% of the sediment load was derived from stream banks, 9% of the total phosphorus load was delivered along with this meander-belt sediment, or about one-fourth as much. Therefore, we approximated the proportion of the total phosphorus load coming from meander-belt phosphorus erosion (C_P) as 25% of the meander-belt sediment erosion.

Fine-Textured Soils

Riparian buffers are capable of trapping a large portion of sediment derived from upland sources (Wenger 1999). This is particularly true where soils are mainly composed of silt and sand, since most particles of these sizes settle out and are retained in buffers through slowing of runoff (Owens and others 2007). However, in areas with fine-textured soils, such as clay, a larger portion of this sediment remains suspended in runoff as it moves through the buffer and is delivered to the stream (Syversen and Borch 2005). Phosphorus attached to these soil particles is also more likely to pass through the buffer. Therefore, regional variation in soil texture influences the potential for buffers to reduce sediment and phosphorus delivery to streams.

Based on field-scale experiments in Wisconsin, nearly all the soil particles that pass through grass buffers in runoff are in the clay size class (J.M. Norman, University of Wisconsin-Madison Soils Dept., written communication, 2005). Norman found that approximately 75% of clay particles in a typical cultivated soil are aggregated into larger conglomerates and are therefore not transported through buffers in runoff. The average proportion of clay in soils was estimated by first calculating the mean of the minimum and maximum clay content (attribute names: clayl and clayh) of STATSGO soil units (USDA 2005), and then calculating the area-weighted average of these values for sections of each soil unit underlying each watershed. The proportion of sediment that will pass through a buffer (S_S) was estimated as one-fourth of the average proportion of clay in the watershed. Because phosphorus preferentially binds with clay particles, the amount of sediment-bound phosphorus that will pass through a buffer (S_P) was estimated as half the proportion of clay in the watershed, or 2 times that for sediment. Therefore, in a basin with 100% clay soils, 25% of the sediment load and 50% of the phosphorus load will pass through the buffer (i.e., is unbufferable).

Dissolved phosphorus in surface runoff is generally poorly retained by riparian buffers (Lowrance and others 1997). Retention rates reported in the literature range from negative values (net phosphorus loss) to 73% (summarized

in Wenger 1999). However, because most phosphorus is attached to soil particles, overall phosphorus retention rates by riparian buffers are often high. The model assumes that 25% of the total phosphorus in runoff is in dissolved form (average annual value from runoff monitoring at six Wisconsin farms in 2004, J.M. Norman, University of Wisconsin-Madison Soils Dept., written communication, 2005), that riparian buffers are capable of retaining 35% of this phosphorus (by facilitating infiltration of runoff and subsequent phosphorus uptake by plants), and that these proportions do not vary geographically. Therefore, in all watersheds, the proportion of the total phosphorus load that is unbufferable because it is dissolved (DP) = $0.25 \times (1 - 0.35) = 0.1625$.

Point-Source Discharges

In some streams, part of the nutrient and sediment load comes from point sources in the basin, such as sanitary sewage treatment and industrial facilities. Since these discharges deliver pollutants directly to streams through pipes, buffers are not capable of mitigating against their effects. The annual load of phosphorus and total suspended solids was calculated for all point sources in the WDNR Pollutant Discharge Elimination System database using records from 2004. Individual point-source loads were summed within each WBI watershed to produce a total point-source load. Total point-source loads were then divided by watershed area to give unit-area point source loads ($\text{kg}/\text{km}^2/\text{year}$) of phosphorus (P_P) and of suspended solids (P_S).

Urban Stormwater

Much of the runoff from urban areas is carried directly to receiving waters by stormwater conveyances, such as pipes and concrete channels. A recent study in the Chicago area found that buffers of natural vegetation did little to mitigate against the effects of urban areas, presumably because of hydrologic alterations that bypass riparian areas (Fitzpatrick and others 2005). Urban land cover was not selected in the regression equations used to predict current loads (see results). Thus, the proportion of those loads attributable to urban sources (and therefore, in this analysis, unbufferable) is uncertain. However, urban land has been tied to significant sediment and phosphorus loading in other studies (Osborne and Wiley 1988; Jones and others 2001; Dodds and Oakes 2004; Robertson and others 2006). Therefore, as an index of its impact, the proportion of the current load of phosphorus and sediment that is unbufferable because it is derived from urban sources (U) was estimated as the proportion of urban land in the watershed.

Load Reduction Potential

The load reduction potential achievable with riparian buffers in each WBI watershed was calculated by expanding equation 1 to the forms:

$$R_S = L_S \times (1 - C_S - S_S - U) - P_S \quad (3)$$

$$R_P = L_P \times (1 - C_P - S_P - D_P - U) - P_P \quad (4)$$

Geographical and frequency distributions of load reduction potential are used to show that the water quality restoration potential of riparian buffers is highly variable and spatially structured. A sensitivity analysis is then used to test the relative influence of the model parameters on the estimates of load reduction potential. The results of this analysis, along with qualitative assessments of parameter uncertainties, are used to make recommendations for further research.

Results

Current Loads

Annual load estimates for the model-building stream sites were highly variable (Fig. 1). The distribution of suspended sediment loads was the most skewed and had the highest coefficient of variation (1.91), followed by phosphorus (1.25), and nitrogen (0.78).

The percentages of cropland and grassland at all spatial scales were positively correlated with loads of all constituents, although none of the regressions were significant for grassland at the watershed scale and the relations with sediment were weak. r^2 's for both land cover classes and all constituents were highest with the land cover measured in the 30 m riparian zone, followed by the 90 m riparian zone, followed by the whole watershed (Fig. 2), although this trend was weak for cropland and nitrogen. Cropland was more strongly associated with loads of nutrients than was grassland. Conversely, riparian grassland was more strongly associated with sediment than was cropland. Agricultural land cover was best able to predict (highest r^2) nitrogen, followed by phosphorus, followed by sediment.

When additional land cover variables were considered, wetland (in addition to cropland) became important, at both watershed and riparian scales (Table 2). As with the agricultural-variable-only models, the ability to predict phosphorus and sediment was notably better with riparian-scale variables than with watershed-scale variables, whereas this pattern was not seen as strongly for nitrogen. Drainage density was the only physiographic variable

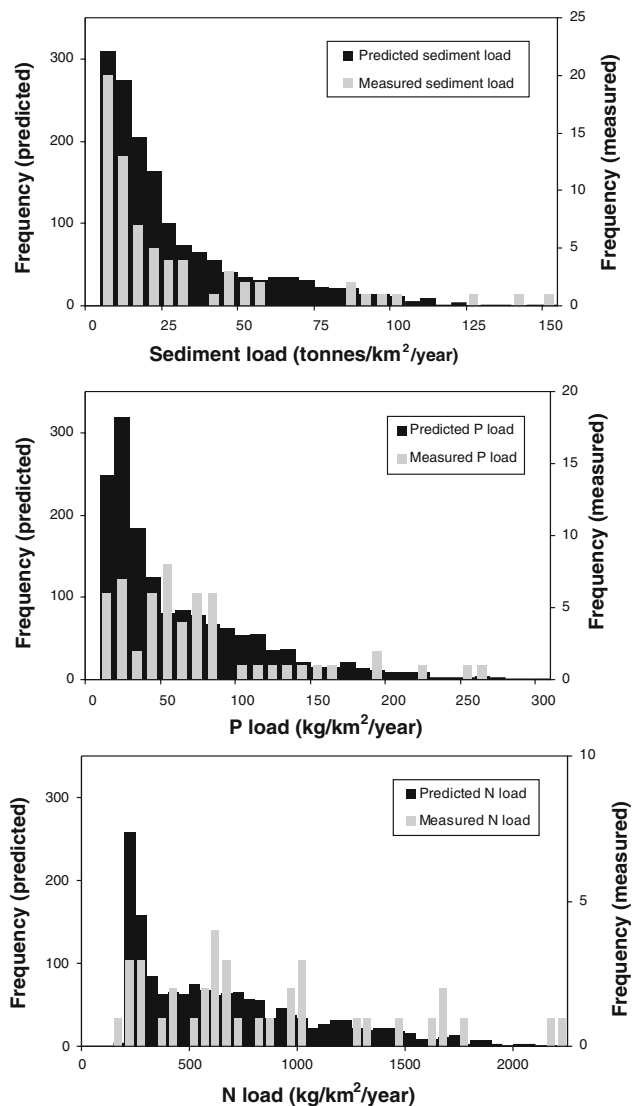


Fig. 1 Predicted phosphorus and sediment loads exhibit lognormal frequency distributions among watersheds. The measured loads used to build the predictive models have similar distributions

which added explanatory power to land-cover-only models, and it did so primarily for phosphorus. When all variables were considered, the best models had significant variables from a mix of scales and types. In general, nitrogen was more accurately predicted by simpler models than was phosphorus which in turn was more accurately predicted than sediment.

Frequency distributions of predicted current phosphorus and sediment loads are log-normally distributed, and span two orders of magnitude (Fig. 1). In general, the range in predicted loads is similar to those measured, indicating that the models are not predicting loads that are significantly different from the calibration data set. Because some of the variables that influence these predictions (e.g., topography) vary over larger spatial scales than the size of the mapped

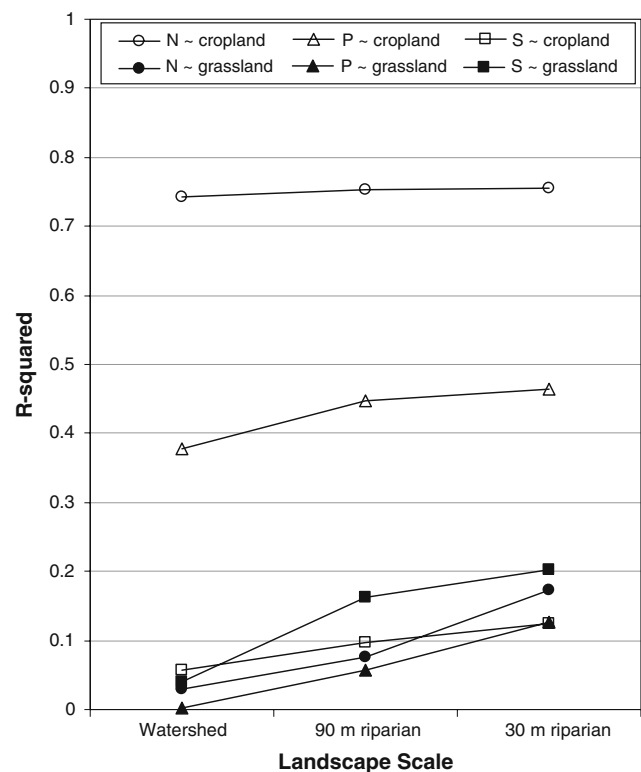


Fig. 2 Coefficients of determination (r^2) for total phosphorus (P), total nitrogen (N), and suspended sediment (S) regressed on percentages of cropland and grassland measured at different spatial scales

watersheds, there are broad geographical patterns in predicted loads (Figs. 3, 4, and 5). Moran's I detected significant spatial autocorrelation in predicted loads of all constituents among the mapped watersheds (nitrogen = 0.30, phosphorus = 0.20, sediment = 0.17, all $p < 0.00001$). In particular, the Driftless Area in southwestern Wisconsin, with its combination of high drainage density and riparian agriculture, has higher phosphorus and sediment loading than other parts of the state. In contrast, the distribution of high nitrogen loads maps on the highly agricultural areas of Wisconsin.

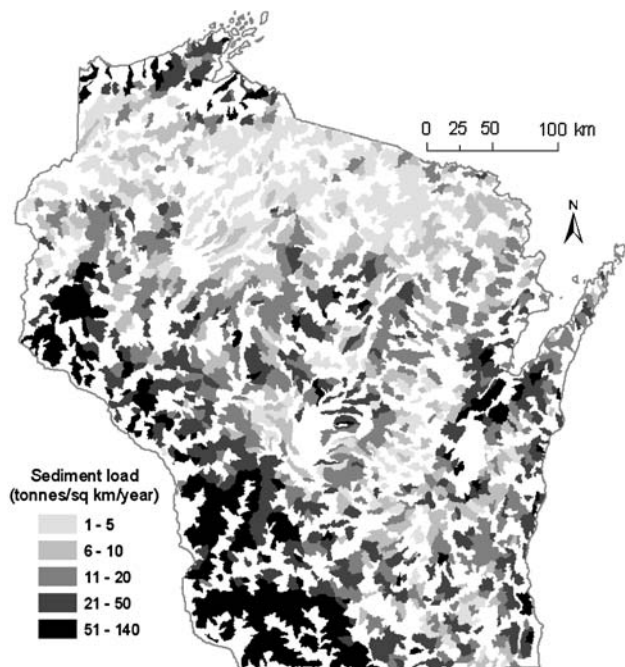
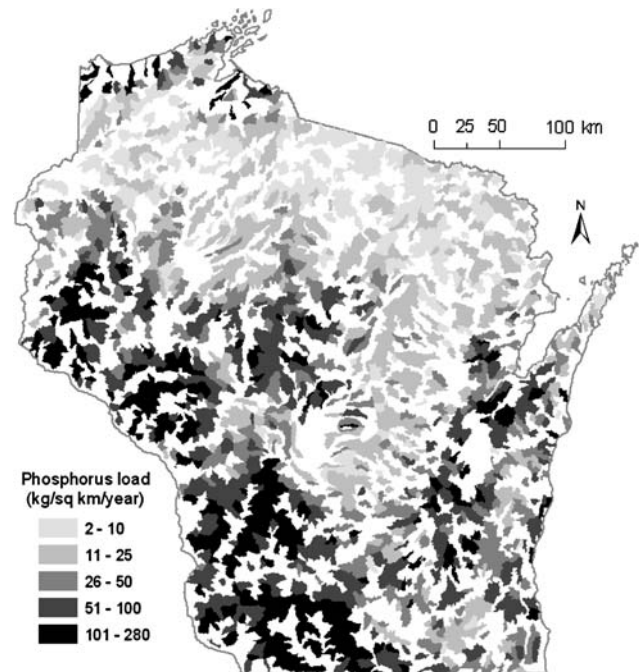
Unbufferable Sources

On average, about 26% of phosphorus and 18% of sediment is estimated to be unbufferable. For sediment, meander-belt erosion is much more important (average 15%) than the other sources, which were typically negligible, while for phosphorus, fine-textured soils and meander-belt erosion were similarly important (both average 4%). The frequency distributions of values for each unbufferable source are all log-normal or negative exponential, however, meaning that the averages are not very informative. A graphical sensitivity analysis illustrates that

Table 2 Comparison of the best regression models from different combinations of variable categories: land cover at watershed and riparian scales and with or without physiographic variables

Variable categories	Nitrogen	Phosphorus	Sediment
Watershed land cover	0.82 Cropland (0.75) Wetland (−0.46)	0.53 Cropland (0.44) Wetland (−0.44)	0.46 Wetland (−0.68)
Watershed land cover & physiographic	0.82 Cropland (0.75) Wetland (−0.46)	0.68 Drainage density (0.65) Cropland (0.54)	0.55 Wetland (−0.52) Drainage density (0.37)
Riparian land cover	0.83 Cropland in 90 m LSRZ (0.83) Grassland in 30 m LSRZ (0.44)	0.62 Wetland in 90 m RZ (−0.47) Cropland in 90 m RZ (0.40)	0.66 Wetland in 30 m LSRZ (−0.81)
Riparian land cover & physiographic	0.83 Cropland in 90 m LSRZ (0.83) Grassland in 30 m LSRZ (0.44)	0.68 Drainage density (0.59) Cropland in 90 m RZ (0.53)	0.66 Wetland in 30 m LSRZ (−0.81)
All variables	0.83 Cropland in 90 m LSRZ (0.83) Grassland in 30 m LSRZ (0.44)	0.80 Drainage density (0.76) Forest in watershed (−0.72) Grassland in 30 m RZ (0.54)	0.72 Wetland in 30 m LSRZ (−0.43) Grassland in 90 m RZ (0.28) Drainage density (0.26) Forest in watershed (−0.25)

Candidate variables are described in Table 1 (RZ, riparian zone of all streams; LSRZ, riparian zone of large streams (≥ 2 nd order)). Bolded numbers are whole-model coefficients of determination (r^2); variables listed were significant ($p < 0.05$); numbers in parentheses are partial correlation coefficients

**Fig. 3** Map of predicted sediment loads in Wisconsin watersheds**Fig. 4** Map of predicted phosphorus loads in Wisconsin watersheds

each source has a different pattern of influence when watersheds are ranked according to load reduction potential (Fig. 6).

Load Reduction Potential

According to the load reduction model (eq. 1), variation in sediment and phosphorus load reduction potential in small

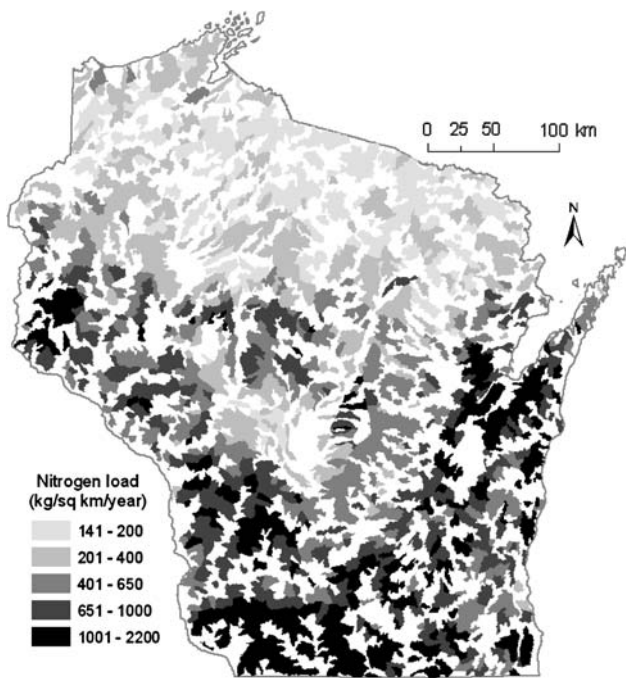


Fig. 5 Map of predicted nitrogen loads in Wisconsin watersheds

Wisconsin watersheds is primarily driven by variation in current loads (Fig. 7). That is, much of the current phosphorus and sediment can be eliminated from streams using riparian buffers. Also, the most important unbufferable source of sediment — meander-belt erosion — tends to covary with estimates of current loads. Load reduction potential for phosphorus and for sediment is positively correlated ($r = 0.62$) among watersheds. In general, watersheds that had higher relative phosphorus reduction potential than sediment reduction potential had high predicted meander-belt erosion. Conversely, watersheds with high soil clay content and lower drainage density had relatively higher sediment reduction potential. However, the influences of unbufferable contributions were generally minor compared to the variation in current loads. A major exception to this is the large meander-belt-erosion-derived sediment source in certain Lake Superior tributaries (see methods).

Discussion

Influence of Landscape Characteristics on Water Quality

Land cover variables were the best predictors of annual loads of phosphorus, nitrogen, and sediment in Wisconsin streams. For nutrients, this finding is in accordance with numerous previous studies, at scales ranging from moderate-sized watersheds (Osborne and Wiley 1988), to multi-state regions (Jones and others 2001; Robertson and Saad

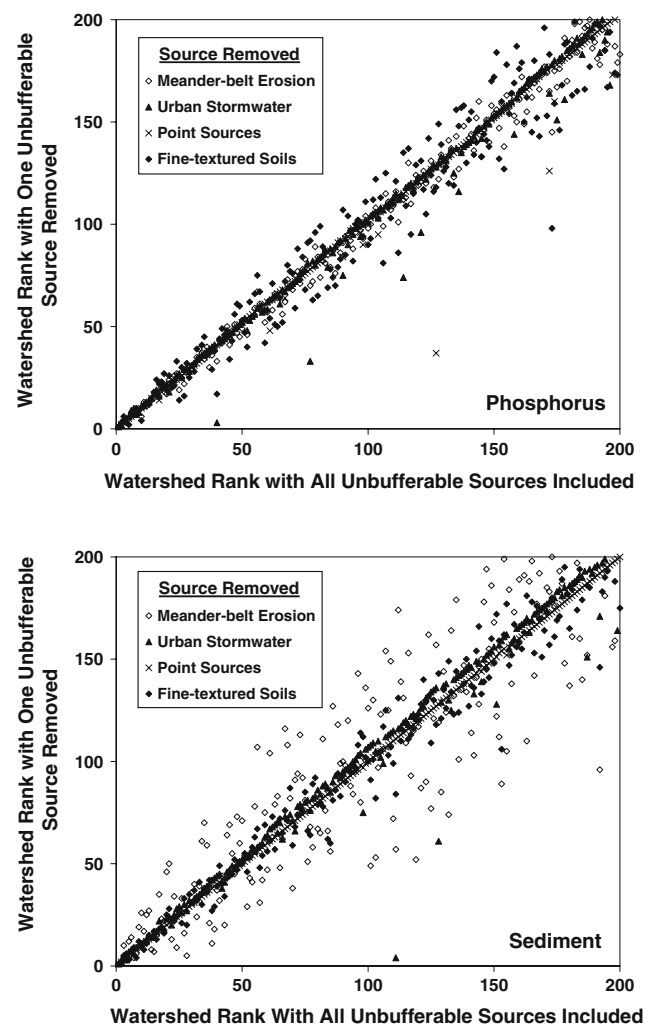


Fig. 6 Sensitivity of the top 200 watershed ranks to unbufferable sources (rank of 1 means that the watershed has the highest reduction potential). Removing one unbufferable source at a time allows its influence on rank order to be seen. For sediment, meander-belt erosion is consistently important, while phosphorus reduction potential is only moderately affected by unbufferable sources

2003; Strayer and others 2003; King and others 2005), to the entire United States (Omernik 1976). However, previous sediment studies in Wisconsin have found soil type (and underlying geology) to be the primary driver of variability in loads (Robertson and Saad 2003; Robertson and others 2006).

While cropland has frequently been associated with increases in pollutant levels, grassland has not. Commonly used land cover datasets such as the National Land Cover Dataset (Homer and others 2004) and the one used in this study, WISCLAND (WDNR 1998), do not differentiate between grazed and ungrazed grassland. This is not a trivial issue. Natural grassland, particularly in riparian areas, has very low rates of nutrient and sediment loss and can serve as a buffer against, and a sink for, upslope

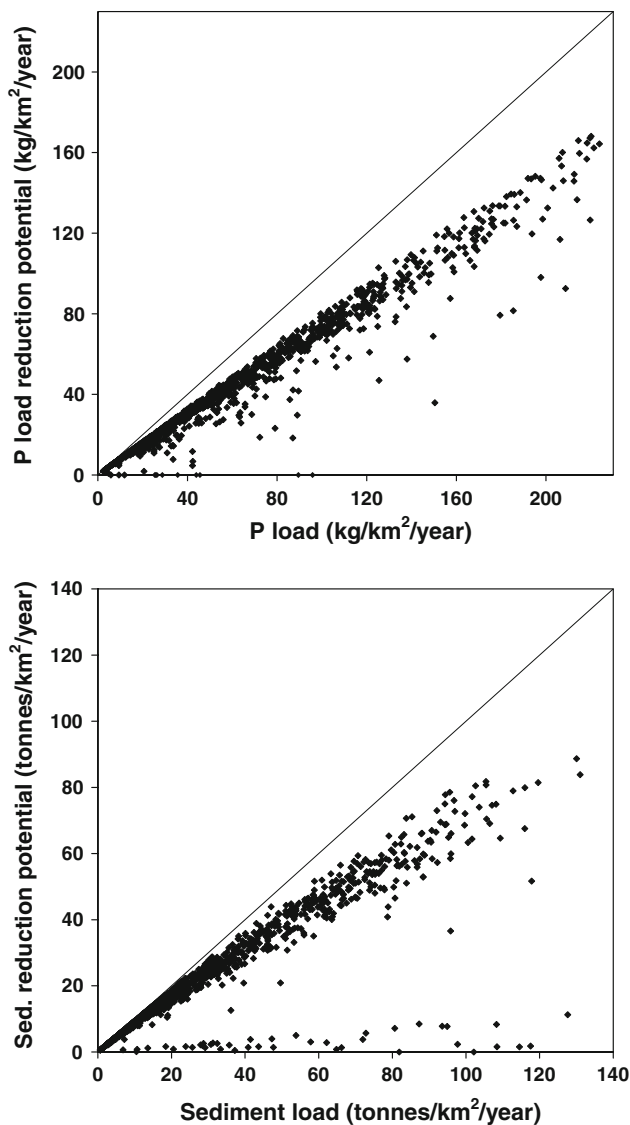


Fig. 7 Phosphorus and sediment load reduction potential compared to estimated load for 1598 small watersheds in Wisconsin. Load reduction potential is always lower than load; watersheds that fall far below the 1:1 line have high proportions of unbufferable sources

pollutant sources (Schultz and others 1995). On the other hand, grazed pastures in riparian areas can be major sources of sediment (Trimble and Mendel 1995) and nutrients (del Rosario and others 2002). In Wisconsin, both natural grassland (often enrolled in the USDA's Conservation Reserve Program) and pasture are common, and are often interspersed at a small spatial grain. The positive correlation between riparian grassland and all constituent loads (Fig. 2) suggests that most grassland is grazed. Furthermore, the increase in explanatory power associated with grassland at more proximally riparian scales supports the notion that grazing-derived constituents are most efficiently delivered when animals are close to or have direct access to streams (Davies-Colley and others 2004).

The relative increases in explanatory power associated with increasing riparian proximity of land cover were different among constituents. In particular, riparian variables were most important in predicting variation in sediment, followed by phosphorus, followed by nitrogen. These results support current conceptions of the sources, transport, and retention of these constituents. The connectivity of sediment sources to flow paths largely determines whether eroded particles are terrestrially re-deposited or are delivered to receiving water bodies. The sediment delivery ratio (Ferro and Porto 2000) describes the proportion of eroded sediment that actually reaches a continuous stream network, and is dependent on flow distance and velocity. In contrast, nitrogen (particularly nitrate) is primarily transported in groundwater flow due to its water solubility (Jordan and others 1997). Although the distance of groundwater nitrogen transport may be related to opportunities for uptake or removal, retention rates may be more strongly determined by localized subsurface flow paths (Baker and others 2001). Phosphorus is transported in both dissolved and particulate form, hence its intermediate dependency on the riparian proximity of land cover.

Among the physiographic variables that were assessed, drainage density was the only one that added significant explanatory power to land-cover models. Its influence probably has two causes: (1) a correlation with land slope, which is a variable in the Universal Soil Loss Equation (Wischmeier and Smith 1978), and (2) an indicator of the degree of interaction between the land and the stream network. High drainage density is a feature of mature, fluvially-created landscapes, which are characterized by short flow path lengths and efficient sediment transport. Soil erosivity, despite being positively correlated with loads of all constituents ($p < 0.01$ for all), was absent from all of the multiple regression models, presumably because it was positively correlated with the more influential cropland variable. Similarly, the potential effect of atmospheric nitrate deposition on nitrogen loads was hidden by its strong correlation with cropland ($r = 0.76$).

Most analyses of the influence of riparian land cover on streams assume that riparian land anywhere along the stream network is comparably important (King and others 2005 is a notable exception). This approach relies on consistency in the mapping of streams, and on consistency in riparian zone function across a range of stream size. In the models developed in this study, wetland at all scales was an important negative correlate of sediment. However, its influence was strongly spatially structured. Wetland in a narrow riparian band (30m) around large streams (2nd order and larger) was a stronger predictor of sediment loads than wetland at successively more extensive scales (Fig. 8). This phenomenon may indicate a relatively larger role of wetlands as sediment sinks when they are located downstream of sediment

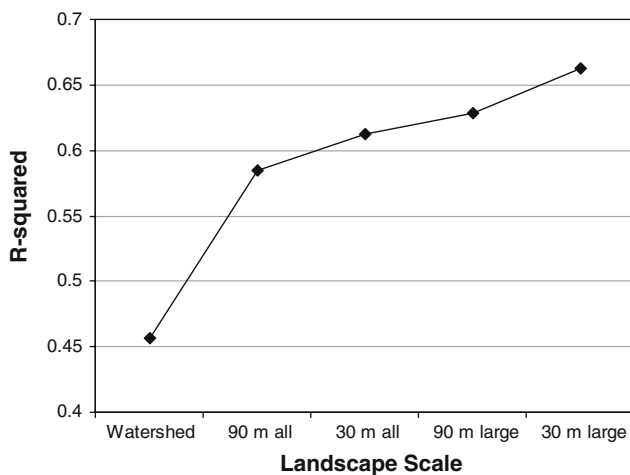


Fig. 8 Coefficients of determination (r^2) for regressions of wetland land cover at different spatial scales on sediment loads (90 and 30 m indicates the width of the riparian zone in which land cover was evaluated; all indicates that the riparian zone was delineated around all stream lines in the WDNR 1:24k hydrography; large indicates that 1st order streams were deleted prior to riparian zone delineation)

sources. It may also indicate inconsistencies in representing the headward extents of first order streams on stream maps that were originally drawn from air photos (such as the 1:24,000 scale maps used in this study; WDNR 2003). Consistent stream mapping is particularly difficult in areas with high drainage density (Hansen 2001).

Frequency and Geographical Distributions of Predicted Loads

The log-normal distributions of predicted phosphorus and sediment loads (Fig. 1) indicate that high loads at downstream sites, such as the Mississippi River (Turner and Rabelais 1991), are disproportionately driven by processes taking place in a limited number of areas (Nowak and others 2006). While these areas are to some extent spatially aggregated (Figs. 3, 4, and 5), neighboring (in the same larger river basin) WBI watersheds often have very different predicted loads, which reinforces the need to target management at this spatial scale (Maxted and others in review). Important nutrient and sediment source areas within the watersheds identified as having high loads are probably log-normally distributed as well (Bennett and others 2005). Thus, effective targeting of management action will require assessment of smaller-scale patterns of nutrient and sediment loss (e.g., UW-CALS 2005).

The geographical distributions of predicted loads are reflective of underlying land cover and physiographic patterns. In Wisconsin, the areas with the highest phosphorus and sediment loads are not simply those with the most

agricultural land cover. In addition, they are characterized by terrain that both facilitate nutrient and sediment transport and constrain land use patterns to locations — riparian areas — where their impacts are likely to be especially strong. In contrast, nitrogen loads are largely independent of underlying physiography, except inasmuch as it controls the locations where agriculture occurs.

Unbufferable Sources

In prioritizing buffer implementation, accounting for unbufferable sources is only important if (1) their magnitude varies geographically, and (2) that variance is at least semi-independent of load estimates. Our source estimates generally fit both criteria, although meander-belt erosion covaries strongly with both phosphorus and sediment load estimates. The contributions of point sources and urban stormwater, while not important on average, significantly detract from the water quality restoration potential of riparian buffers in certain watersheds. This is not to imply that urban and other point-source-dominated watersheds are not worthy of any management, but rather that strategies other than riparian buffers are necessary to effectively address water quality impairment.

Estimates of unbufferable source amounts in this study vary in their reliability. For example, point source totals are based on measured discharges, although there may be dischargers that are not included in the database we used. On the other hand, the estimates of meander-belt-erosion-derived phosphorus and sediment are based on sparse quantitative information and are therefore more speculative. It is clear, however, that estimates of sediment load reduction potential are sensitive to this parameter (Fig. 6). Further research should study how the extent of meander-belt erosion varies in different landscapes.

In some agricultural landscapes, tile drainage may represent a potentially important unbufferable source of nutrients (Osborne and Kovacic 1993). In most settings, the installation of artificial drainage systems decreases sediment loss by reducing surface runoff, and increases soluble nitrogen loss by accelerating leaching and thus reducing opportunities for plant uptake and denitrification (Skaggs and others 1994). Phosphorus loss can either be increased or decreased by drainage, depending on a number of factors, including soil texture, organic content, and irrigation practices (Sims and others 1998). Tile drainage was not estimated in this study because it is not prevalent in Wisconsin and because there is insufficient information on its geographic distribution. However, in landscapes where drainage is common and can be mapped, its effect on the potential performance of riparian buffers should be taken into account.

Load Reduction Potential

According to the load reduction potential model, riparian buffers are capable of reducing large percentages of the phosphorus and sediment that is currently being carried by Wisconsin streams. Even in watersheds with extremely high loads (top 10%), an average of about 70% of the sediment and phosphorus can be reduced through buffer implementation. These estimates are on the high end of ranges from other studies that used different methodologies. A meta-analysis that summarized field studies of riparian buffers found phosphorus reduction that averaged 50%, and ranged up to 80% (Gitau and others 2005). When riparian buffers and other conservation practices were simulated for two Minnesota watersheds using a process-based model, sediment reduction estimates ranged from 32 to 84% (Zimmerman and others 2003).

The estimates of load reduction potential in this study are best-case scenarios. In practice, there are several reasons why prompt and lasting changes may be difficult to achieve. First, buffer implementation may cause transient increases in sediment and nutrient loss as the stream channel adjusts to changes in vegetation type (Parkyn and others 2005). Second, net reductions in constituent loads may be delayed if reductions in upland-derived constituents are offset by increased erosion and resuspension of sediment and phosphorus stored in the channel. And third, riparian buffers can release stored sediment and phosphorus, particularly during overbank flows or when vegetation senesces (Dillaha 1989). However, these potential shortcomings can be moderated if expectations of buffer performance are seen in a long-term context and a commitment to buffer maintenance (e.g., harvesting vegetation to remove phosphorus) is made.

Vegetative composition of buffers can influence their ability to retain nutrients and sediment, and to perform a variety of other ecological functions. The relative merits of grass and forest buffers have been discussed elsewhere (Osborne and Kovacic 1993; Trimble 1997b; Montgomery 1997; Lyons and others 2000; Blann and others 2002). Taken together, these studies suggest that any choice of buffer vegetation type will result in tradeoffs among management goals (e.g., nitrogen versus sediment retention). Optimal buffer vegetative composition may vary among stream reaches, based on local geomorphic conditions (e.g., floodplain aggradation), additional management objectives (e.g., stream shading), and landowner preferences for access and harvesting. Rather than prescribing a particular composition, our model acknowledges the need for variability, and simply assumes that any chosen design will function as described in the methods. Recently, zoned, multi-species buffers have been advocated as a way to achieve multiple management objectives (Schultz and

others 2004), although their effectiveness has been assessed in a limited geographic setting.

Management agencies often seek to achieve high total and high percentage pollutant reductions. This analysis identifies watersheds with the potential to meet both of these objectives (Fig. 9; box B). Estimates of phosphorus load reduction potential were used to rank watersheds for buffer implementation (UW-CALS 2005). Figure 10 illustrates how additions from the top of this ranked list should produce the largest marginal benefits. For example, buffering the top 10% (by area) of these watersheds would reduce the sum of phosphorus exports from all watersheds by 20%. In contrast, buffering the bottom 10% would only reduce 1% of all the phosphorus.

The concept of load reduction potential provides an alternative to the reference condition concept, which is the water quality status expected in the absence of human influence (USEPA 2000b). The reference condition concept is useful for partitioning the influence of natural landscape features from the influence of human activities on water quality, especially in moderately degraded systems. However, in many cases, limiting factors (e.g., unbufferable sources) may make the reference condition unattainable using available management practices. The reference condition also places undue emphasis on small variations in natural background conditions whose ranges are dwarfed by the loads observed in some highly polluted streams. For example, the 90th percentile of our modeled phosphorus loads among 1598 small watersheds in Wisconsin is 134 kg/km²/year. This amount is over ten times the difference between the highest (18 kg/km²/year) and the lowest (6 kg/km²/year) predicted reference phosphorus load for the upper Midwest (Robertson and others 2006).

Load reduction potential may be most useful in a triage-type approach, where the management goal is to induce the

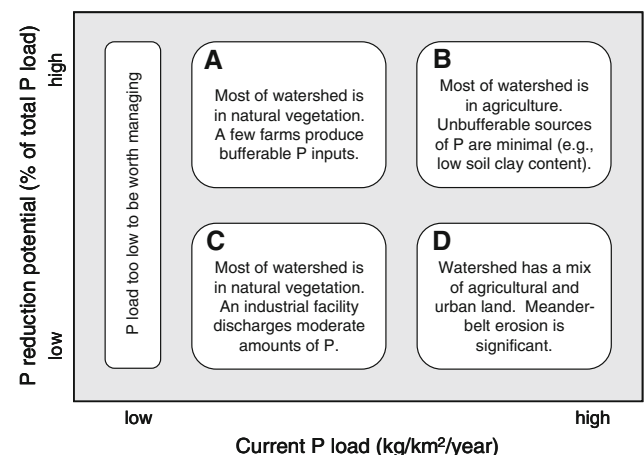


Fig. 9 Watershed scenarios that illustrate different types and degrees of water-quality restoration potential. Riparian buffer implementation in watershed B would produce the largest reduction in phosphorus

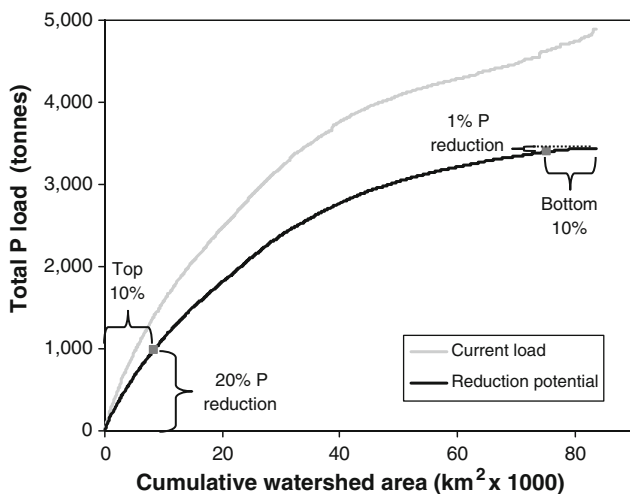


Fig. 10 Cumulative frequency distribution of modeled stream phosphorus loads across Wisconsin watersheds (gray line). The black line shows the expected increase in phosphorus load reduction in response to additional buffered area. For both curves, watersheds were added in descending order of phosphorus load reduction potential

largest improvements in water quality where there are the fewest impediments to improvement. This approach could be a cost effective way to target management for large-scale objectives, such as reducing phosphorus inputs to a downstream lake or impoundment. Alternatively, the reference condition may be most useful for managing specific streams that are ecologically or recreationally valuable, and where knowledge is sought on of the degree of progress toward a feasible endpoint.

Conclusions

Estimates of the water quality benefits that are potentially achievable using specific management practices (such as demonstrated for riparian buffers) can help target locations for implementation of those practices on heterogeneous landscapes. The models presented in this article synthesize diverse sources of information into a common framework that provides quantitative estimates of the stream water quality restoration potential of riparian buffers. Perhaps the most important finding of this study is that even in watersheds with extremely high yields of phosphorus and sediment, large proportions of these constituents can be retained on the landscape using a practical management practice. Furthermore, the skewed distribution of predicted pollutant loads across Wisconsin indicates that targeting management in watersheds with high water quality restoration potential should produce much greater benefits than traditional first-come, first-served approaches.

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