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PHOSPHORUS LOADS TO SURFACE WATERS: A SIMPLE MODEL TO ACCOUNT FOR SPATIAL PATTERN OF LAND USE[1]

P. A. SORANNO,[2] S. L. HUBLER,[3] S. R. CARPENTER
Center for Limnology, University of Wisconsin, Madison, Wisconsin 53706 USA

R. C. LATHROP
Bureau of Research, Wisconsin Department of Natural Resources, Madison, Wisconsin 53716 USA

Abstract. Modeling nonpoint-source phosphorus (P) loading from land to surface waters can be both complex and data intensive. Our goal was to develop a simple model that would account for spatial pattern in topography and land use using geographic information system (GIS) databases. We estimated areas of the watershed that strongly contributed to P loading by approximating overland flow, and modeled annual P loading by fitting three parameters to data obtained by stream monitoring. We calibrated the model using P loading data from two years of contrasting annual precipitation for Lake Mendota, a Wisconsin eutrophic lake in a watershed dominated by agriculture and urban lands. Land-use scenarios were developed to estimate annual P loading from pre-settlement and future land uses. As much as half of the Lake Mendota watershed did not contribute significantly to annual P loading. The greatest contribution to loading came from a heterogeneous riparian corridor that varied in width from 0.1 km to ≈6 km depending on topography and runoff conditions. We estimate that loading from pre-settlement land use was one-sixth of the loading from present land use. A future scenario, representing an 80% increase in existing urban land (from 9 to 16% of total watershed area, which would be reached in 30 yr with current land-use trends), showed only modest increases in annual P loading but possible significant effects on water quality. If the watershed were to become entirely urbanized, P loading to the lake would double and potential effects on water quality would be severe. Changes in P loading were strongest with conversions of undisturbed vegetated lands, especially riparian areas, to either urban or agricultural uses. Variability in total annual rainfall leads to variability in the riparian area that affects P loading, with implications for policies intended to control nonpoint nutrient inputs.

Key words: export coefficients; flux coefficients; Lake Mendota; land use; nonpoint source; phosphorus loading; riparian; water quality modeling; watershed; Wisconsin.

INTRODUCTION

Lakes are strongly linked to their watersheds through the transport of materials carried by surface runoff. Transport of sediments and nutrients has increased in many watersheds due to agriculture, urban development, and mining and forestry practices (U.S. Environmental Protection Agency 1990, Kitchell and Sanford 1992, Duda 1993). These fluxes contribute to the cultural eutrophication of surface waters, generally through increased loading of phosphorus (P) (Schindler 1977). Due to land-use practices and rapid land-use changes in many parts of the world, nonpoint P loading is a serious threat to water quality (National Research Council 1992, Duda 1993).

The impact of watershed characteristics on surface water quality depends on regional geology, soil P content and erodibility (Hobbie and Likens 1973, Dillon and Kirchner 1975), watershed size, shape, topography, and land use (Omernik 1976, Osborne and Wiley 1988, Hunsaker et al. 1992), and precipitation (Sharpley et al. 1981). Phosphorus flux often increases with land disturbance, soil erosion, and proportion of impervious surfaces (Arnell 1982, Byron and Goldman 1989). However, at the scale of entire watersheds, relationships between land use and water quality are surprisingly variable (Omernik 1976, Hunsaker et al. 1992, Osborne and Kovacic 1993). Although the proportional areas of different land-use types in a watershed explain some of the variance in water quality parameters, much remains unexplained (Dillon and Kirchner 1975, Reckhow et al. 1980, Osborne and Wiley 1988, Hunsaker et al. 1992).

Phosphorus loading models

Early models examining the impact of land use on surface water quality were based on the P-export coefficient approach (Reckhow et al. 1980) using equations such as:

$$L = \sum_{i=1}^{m} c_i A_i \qquad (1)$$

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[2] Present address: Trout Lake Station, Center for Limnology, University of Wisconsin–Madison, 10810 County Highway N, Boulder Junction, Wisconsin 54512 USA.

[3] Present address: Biospherical Instruments Incorporated, San Diego, California 92110 USA.

where L is total P loading from land (in kilograms per year), m is number of land-use types, c_i is the P-export coefficient for land use i (in kilograms per hectare per year), and A_i is area of land use i (in hectares). Export coefficients (c_i) are estimated by monitoring runoff from either plots or watersheds of the appropriate land use and dividing by the entire watershed area drained (Dillon and Kirchner 1975, Rast and Lee 1978, Reckhow et al. 1980, Clesceri et al. 1986). Eq. 1 assumes that P export for each land type increases linearly with area. However, this assumption is often inappropriate (Prairie and Kalff 1986). Hydrologists have shown that sediment delivery ratios ([sediment delivered]/[gross sediment eroded]) decrease with watershed area (Walling 1983). Because P often reaches surface waters attached to sediment particles, sediment delivery should be considered by P input models (Novotny and Chesters 1989). Nutrients are deposited and transformed as they are transported from source areas to streams and lakes (Peterjohn and Correll 1984, Vaithiyanathan and Correll 1992). These processes may attenuate P fluxes as watershed size increases. Attenuation may vary among watersheds depending on topography, soil type, and weather. Consequently, estimating export coefficients is difficult and uncertain (Reckhow et al. 1980, Beaulac and Reckhow 1982, Frink 1991). Despite the uncertainties associated with export coefficients, they are intuitively simple and easy to apply, and they are used in many water quality models (Reckhow et al. 1980, Reckhow and Chapra 1983).

The most complex water quality models begin with hydrologic analyses that predict surface and sometimes subsurface flow. These models include data-intensive distributed parameter models and physically based hydrologic models (Grayson et al. 1992, DeVantier and Feldman 1993). While they break down complexity to elemental components that appear tractable at small enough spatial scales, these models are hard to apply because data are almost always insufficient, and the models are often restricted to small spatial units (DeVantier and Feldman 1993). Some of these models have large numbers of free parameters so it is easy to obtain a good fit to calibration data but, because the models are overparameterized, prediction errors may be large (Grayson et al. 1992).

Overview of an alternative approach

We suggest an approach to estimating P loading from land that accounts for flow distance, unlike export coefficients, but does so by fitting only one additional parameter, unlike the parameter-rich models:

$$L = \sum_{i=1}^{m} \sum_{p=1}^{n} f_i A_{p,i} T_i^p \tag{2}$$

where L is total P loading from land (in kilograms per year); m is the total number of land-use types; n is the total number of pixels (equal-sized grid cells in the GIS) in the contributing area; p is the distance of each pixel to water in the path of surface overland flow (in number of pixels); f_i is the P-flux coefficient for land use i (in kilograms per hectare per year); $A_{p,i}$ is the area of land use i at distance p from open water (in hectares); and T is the transmission coefficient ($0 < T < 1$, dimensionless), which represents the proportion of P that is transported to the next pixel in the path of surface overland flow. The additional parameter not found in export-coefficient models, T, accounts for the amount of P that is attenuated between pixels; and p, the distance of each land use from water along the flow path, determines how much P eventually reaches surface water. The attenuation term could take various forms. We chose T_i^p because it implies that a fixed proportion of P is attenuated for every unit of travel along the flow path. In contrast, traditional export coefficients assume that 100% of P eroding from a land unit will reach surface water. Export coefficients and P-flux coefficients have the same units (kilograms per hectare per year), but they differ in that flux coefficients represent P production and transport to the *next pixel* along the flow path rather than the production and transport that reaches surface waters. P-flux coefficients and traditional export coefficients should have the same value when applied to a given area of riparian or urban land where we assume all P is transported to surface waters.

An important difference between Eqs. 1 and 2 is that Eq. 1 applies to the entire watershed area, whereas Eq. 2 is applied to only the "contributing area," that area that is determined to contribute strongly to loading. The contributing area may be equal to or less than the entire watershed area.

Urban lands attenuate sediment-bound nutrients much less than agricultural lands since the proportion of impervious surfaces is high and in many cases, water flow is efficiently channelled to surface waters through storm sewers. This is the case in the Lake Mendota watershed. Therefore, we let $T = 1$ for all urban lands regardless of distance to water. Thus, P-flux and P-export coefficients for urban land should be similar. We assumed that agricultural land, wetlands, and forests attenuated P, so T was allowed to vary between 0 and 1.

Like other P-export models, Eq. 2 assumes that land use, topography, and total runoff are the major determinants of P flux. Other sources of P input to lakes, such as atmospheric deposition, are not included in the model and must be treated separately. Because total phosphorus is transported to surface waters predominantly in sediment-bound form, the model applies only to P transport in overland flow (Vaithiyanathan and Correll 1992). While some P is transported in groundwater, this flux can be small compared to surface transport, especially in agricultural (Peterjohn and Correll 1984) and urban areas. We would not recommend Eq. 2 for nitrogen loading, because transport in dissolved fractions and especially through groundwater is very important (Peterjohn and Correll 1984).

Our model attempts to take advantage of both the spatial data structure of Geographic Information Systems (GIS) and the intuitively simple export-coefficient concept. Calibration data are total annual P loadings to the lake from all subwatersheds for at least 1 yr. Input data for the model are spatially referenced databases of watershed land use, topography, and hydrography. The watershed may be broken down into any number of equal-sized grid cells to use this model, although parameters and uncertainties will depend on grid-cell size (Vieux and Needham 1993). We used pre-existing U.S. Geologic Survey (USGS) GIS databases readily available to the public that were provided by the Wisconsin Department of Natural Resources (WDNR) Geographic Services Section. We tested the approach using watershed and loading data for the Lake Mendota watershed, a north-temperate culturally eutrophic lake in southern Wisconsin, USA. Parameters were estimated for the Lake Mendota watershed to calculate P loading under pre-settlement, current, and projected future land-use scenarios.

METHODS

Study site

Lake Mendota (area 4000 ha, mean depth 12.7 m) is a eutrophic lake near Madison in Dane County, Wisconsin (43°6′ N, 89°24′ W) (Kitchell 1992). This rich agricultural region has well-drained silt loam soils in the uplands, and poorly drained silts with organic material underlain by alluvial deposits in the lowlands (Cline 1965). During our study, ≈86% of the land was agricultural, 9% urban, 4% wetlands, and 1% forest. Five major streams and two main storm sewers drain the watershed (Fig. 1). The Yahara River subwatershed is 96% agricultural with low relief. Before entering the lake, the river is joined by Token Creek, a stream of higher base flow than the Yahara River (Lathrop 1992) draining a subwatershed of land use similar to that of the Yahara River. Pheasant Branch Creek enters from the west and drains a steeper terrain than the other subwatersheds. The two storm sewers, Spring Harbor and Willow Creek, drain urban areas and flow directly into the lake.

Prior to European settlement (1830s), the watershed vegetation was dominated by oak savannah and prairie (Curtis 1959). By ≈1870, the current proportion of agricultural land had been achieved in Dane County (Lathrop 1992). This conversion culturally eutrophied Lake Mendota (Kitchell and Sanford 1992). Currently, land in the watershed is being converted from agricultural to urban uses, especially around the Madison metropolitan area and surrounding smaller communities (Dane County Regional Planning Commission 1992).

Watershed data

Land use/cover, hydrography, and watershed-boundary data were obtained from the WDNR Geographic Information System Database (copyrighted, WDNR

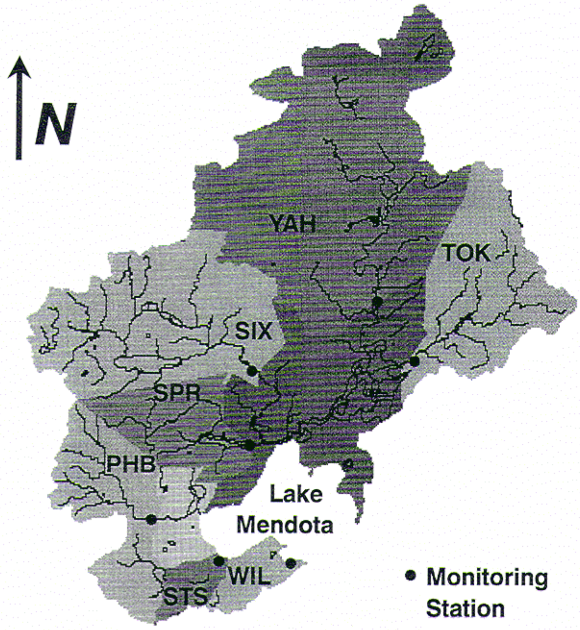

FIG. 1. Lake Mendota (Wisconsin) watershed and subwatersheds. The labelled subwatersheds were monitored in 1976 and 1977 at the monitoring sites shown. Subwatershed labels are: STS = Spring Harbor storm sewer, WIL = Willow Creek storm sewer, PHB = Pheasant Branch Creek, SPR = Spring Creek, SIX = Sixmile Creek, YAH = Yahara River, TOK = Token Creek. Note that unmonitored areas are not labelled.

Geographic Services Section, Madison, Wisconsin). The land use/cover data were derived from 1:250 000 Land Use and Land Cover (LULC) data (USGS) by the WDNR Geographic Services Section. We combined 24 surface feature categories of land use/cover into six classes: (1) Agriculture (row and non-row crops, pastures, and feedlots); (2) Urban (residential, commercial and services, industrial, mixed urban or built-up land, transitional areas, and industrial and commercial complexes); (3) Road (highway, communications, and utilities); (4) Forest (all forest types); (5) Wetland (forested and non-forested wetlands); and (6) Lake (small lakes and ponds excluding Lake Mendota). The original source of the hydrography data was 1:100 000 Digital Line Graph (DLG) data (USGS), where only hydrographic features defined by the USGS as perennial are present. We converted the data from vector to raster format at 100 × 100 m pixel size to match the topographic data. Since this resolution is coarse relative to many streams and rivers, we only considered the presence/absence of any perennial stream within each pixel. Because we did not model in-stream attenuation of nutrients, we assumed that if surface runoff (of P) reached a stream, all of the P was transported to the monitoring station. The digital elevation model (DEM) dataset was created by converting USGS 1-Degree DEM to raster format with elevations resampled at 100-m intervals (Lynn Usery, University of Wisconsin Geography Department, Madison, Wisconsin). The scaling issue was most critical for the DEM since these data were used to estimate overland flow. We used DEM data at the finest resolution available to us, and converted all other data to this grid size. Since DEM data had a distinct "stair-step" appearance, we smoothed them by interpolating between elevations. The algorithm first searched for groups of pixels with similar elevation and then interpolated quadratically using elevations of the end points (outside the flat region) and the middle point in the flat area. For example, a series of pixels with the following elevations:

9 1 1 1 9

would be transformed into:

9 4 1 4 9.

The median point and values outside the range do not change. We used the interpolation algorithm on the rows and columns separately in all four directions and averaged results pixel by pixel.

Contributing area

The contributing area is defined as the area of the watershed that potentially contributes to P loading via overland water flow, which we assumed followed the path of steepest descent. If a given pixel was lower or equal in elevation to all of its neighbors, then it was assumed not to contribute to surface runoff. Water flow paths were estimated using a hexagonal grid, which approximates distance more accurately than a four-neighbor scheme by allowing measurement in six directions instead of four. To convert the 100 × 100 m DEM data to a hexagonal format, we shifted every other row by one-half of a pixel. This maintained all four directional differences while making half of the diagonals a distance one and the other half a distance two. Thus, on average, diagonal movement had distance 1.5 instead of the 2.0 for a four-neighbor "city block" scheme.

Within the contributing area, we define the effective land area as the area that actually contributes to runoff (and P loading) based on the model calibration. By definition, the effective land area will always be equal to or less than the contributing area. Effective land area may vary depending on rainfall and runoff. The contributing area, on the other hand, is fixed for a given watershed.

Coefficients

Total P and water discharge from the five major streams and two storm sewers draining into Lake Mendota watershed were monitored in 1976 and 1977 (Lathrop 1979). Water samples for total P were collected during both runoff events and baseflow conditions. Annual nonpoint P loadings were calculated by direct integration using P concentrations and continuous stream discharge data for 1976 and 1977. The two years contrasted in weather conditions: 1976 had high precipitation during the spring months followed by a summer drought, and 1977 had average spring precipitation with high summer rainfall (Lathrop 1979). Total runoff for 1976 was much higher than in 1977. Based on an 18-yr time series of annual stream discharge for the Pheasant Branch Creek subwatershed, runoff was above average in 1976 (<1 SD above the mean) and below average in 1977 (1 SD below the mean) (USGS Water Resources Data, Wisconsin).

Three parameters for Eq. 2 (T for all non-urban lands and P-flux coefficients for agricultural and urban lands) were estimated by minimizing the root mean squared errors (RMSE) between calibrated and observed data. First, T was fit by direct search, then the P-flux coefficients were calculated for each T value, and the combination of parameters was chosen that minimized RMSE. Three different sets of model parameters (P-flux coefficients and T) were estimated: for 1976, 1977, and both years combined. To compare our method to the P-export method, we fit the data to Eq. 1 to estimate agricultural and urban export coefficients and calculate the RMSE for all three calibration datasets.

Forest and wetlands combined covered <5% of the total watershed area in 1976–1977, and we assumed their relative contribution of P was small compared to urban and agricultural land. Consequently, our estimate of T was determined primarily by agricultural lands. We set forest P-flux coefficients to 0.10 $kg \cdot ha^{-1} \cdot yr^{-1}$, an average P-export coefficient from the literature (Reckhow et al. 1980). For wetlands, we used a value appropriate for the soil and geology of the region (0.05 $kg\ P \cdot ha^{-1} \cdot yr^{-1}$).

To estimate average P-export and runoff coefficients for the subwatersheds as they are commonly defined in the literature, we divided the annual total P loads and runoff from each monitoring station by the respective subwatershed areas for both 1976 and 1977. We also compiled a set of traditional export coefficients from the literature that best represent the low, high, and "most likely" value for both urban and agricultural land in the Lake Mendota region. These values were based on regional geology, soil type, and crop or urban type (Reckhow et al. 1980, Frink 1991). When possible, we chose coefficients derived from small field plots (< 1 ha) so they would be more comparable to P-flux coefficients. The "most likely" export coefficient for agriculture was estimated by calculating an average of export coefficients weighted by the proportional area of crop types found in Dane County (Lathrop 1992).

Phosphorus loading scenarios

We estimated nonpoint P loading to Lake Mendota using three different land-use scenarios: pre-settlement, baseline (land use in 1976–1977), and urban × 1.8. The pre-settlement scenario was simulated by converting all urban and agricultural pixels of the baseline land-use map to natural vegetation. We assumed that

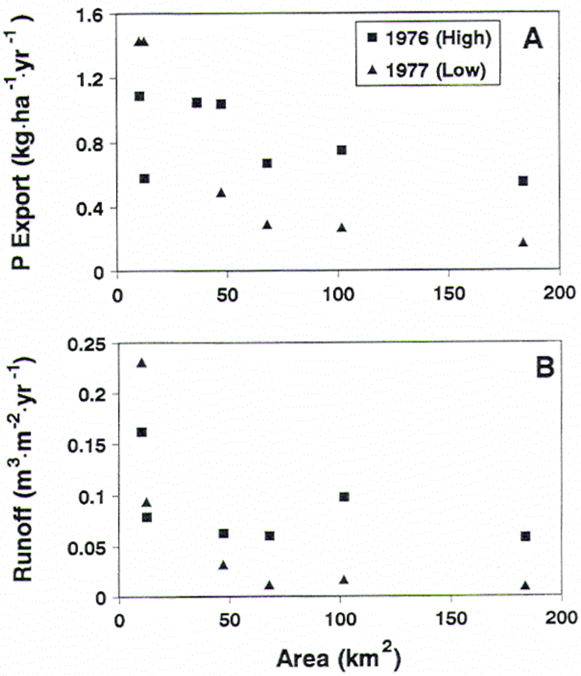

FIG. 2. Phosphorus export coefficients calculated for each subwatershed using Eq. 1 for 1976 and 1977 vs. subwatershed area (A), and runoff coefficients for 1976 and 1977 vs. subwatershed area (B).

average P-export coefficients for native prairie (0.3 kg·ha^{-1}·yr^{-1}) were appropriate for pre-settlement natural vegetation (Wojner 1976, Timmons and Holt 1977).

The urbanization scenario was based on a 30-yr projection that urban lands will increase by 80% (Dane County Regional Planning Commission 1992). We converted pixels to urban land use in areas where urban growth is expected to occur: the cities of Madison and Middleton around the southwestern shores of Lake Mendota, and the smaller communities throughout the watershed. Since the Lake Mendota watershed is predominantly agricultural, this scenario increased the percentage of urban land in the watershed from 9 to 16%. We also considered a scenario in which the watershed was entirely urban. To estimate P loading for each scenario, model runs were performed using each of the three sets of model parameters, which bracket a wide range of precipitation: 1976, 1977, and the two years combined.

Model sensitivity

Plots of RMSE vs. T assessed the sensitivity of model results to changes in T. To examine the sensitivity of P-flux coefficients to individual subwatersheds, we used jackknifed data sets where one subwatershed was left out at a time (Sokal and Rohlf 1981). Agricultural and urban P-flux coefficients were recalculated for each iteration, using T from the complete data set for each calibration run.

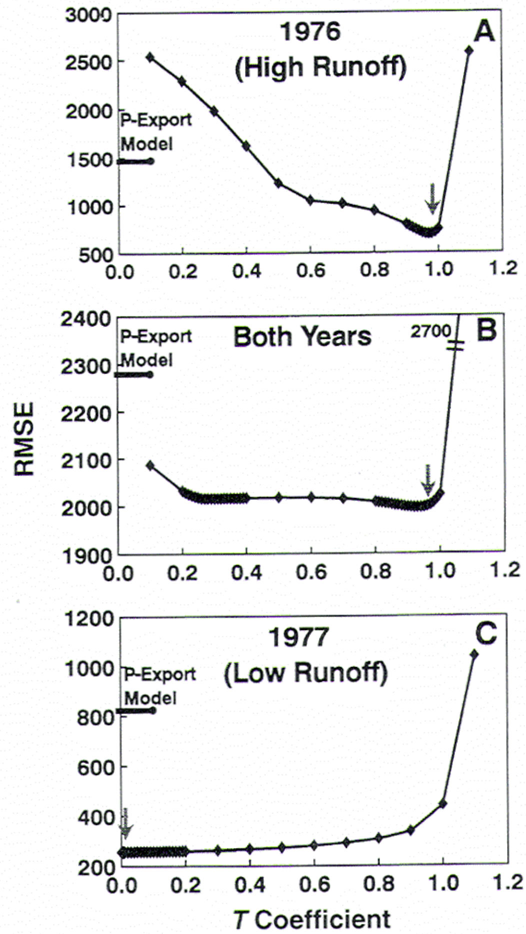

FIG. 3. Root mean squared errors (RMSE) for calibration fits of P-flux model (Eq. 2; ◆———◆) and P-export model (Eq. 1; ———) to stream monitoring data for 1976 (A), for the two years combined (B), and for 1977 (C). Arrows indicate the transmission coefficient (T) where the RMSE is minimum.

RESULTS

Phosphorus export coefficients and runoff coefficients showed that both P export and runoff per unit area decreased with increasing subwatershed area, although the trend seems more pronounced for P export (Fig. 2). However, differences in land use between subwatersheds confound these trends. Land use in the two smallest subwatersheds, Willow Creek and Spring Harbor storm sewers, is mostly urban (81% and 67%, respectively), but in the other subwatersheds, urban lands comprise only 2–6% of the total area. Our modeling effort was, in part, an attempt to account for the combined effects of area and land use.

The transmission coefficient (T) that minimized the RMSE was 0.97 for 1976, 0.93 for the two years combined, and 0.0002 for 1977 (Fig. 3). Except for 1976, optimal T values are poorly defined by the data, but values for 1976 and for the two years combined are clearly not close to 0, and the value for 1977 is clearly

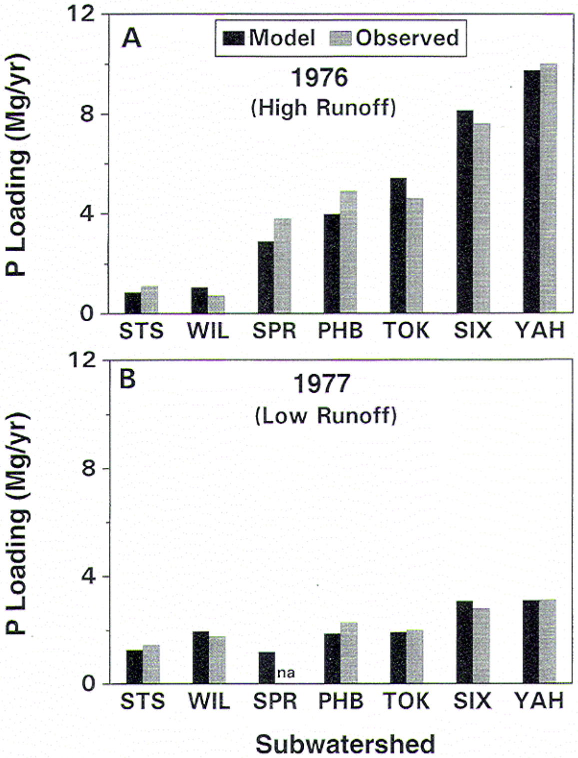

FIG. 4. Subwatershed P loads for model calibration and observed data for 1976 (A) and 1977 (B).

not close to 1, indicating real differences in T. Goodness of fit, as measured by RMSE, was better for the P-flux model (Eq. 2) than for the P-export model (Fig. 3). The P-flux model fits more closely because it accounts for the attenuation evident in the calibration data (Fig. 2). An F-test of the two models comparing the extra sum of squares due to the extra parameter in Eq. 2 (Draper and Smith 1981, Bates and Watts 1988) shows that Eq. 2 represents an improvement over Eq. 1 for the 1976 and 1977 model fits ($P < 0.05$ for 1976; $P < 0.01$ for 1977). However, the improvement was not significant for the calibration fit to both years of data combined ($P > 0.10$).

Model calibrations of the P-flux method were, on average, within 16% of the observed values in 1976, and within 9% in 1977 (Fig. 4). Total P loading was more than twice as high in 1976 than in 1977, and variation among subwatersheds was large during the high-runoff year.

Contrasts in effective land area (the area that contributes to loading) between high- and low-runoff years are shown in Fig. 5D–I. During low-runoff years, only areas adjacent to surface waters and urban areas (both shown in light grey) contributed significantly to loading. The largest contrast is between the urbanized scenario and all others (Fig. 5F, I). Even in relatively dry years (1977), urban areas will contribute to loading, whereas in agricultural or vegetated areas, much of the P produced in the contributing area is attenuated before reaching water. Differences between high- and low-runoff years were larger than the differences between land-use scenarios.

The transmission coefficient determines the effective land area, i.e., that amount of the source area that significantly contributes to loading in a given year. As T decreases, the effective land area decreases, so areas farther from streams do not contribute P (Table 1). The distance of the edge of the effective land area to the stream channels varied from 0.1 km to almost 6 km depending on runoff conditions and local topography. In the low-runoff year, only 30% of the contributing area transported a significant amount of P to surface waters, but during the high-runoff year, 87% of the contributing area was a significant source of P. This contrast is important, since land use within the effective land area may be quite different from land use within the entire watershed (Fig. 6). For the baseline scenario, only 9% of the total watershed is urban land. However, 53% of the effective land area was urban during 1977, the low-runoff year (Fig. 6B). These trends are amplified in the urbanization scenario (Fig. 6C). The difference in the effective land areas between high- and low-runoff years in the pre-settlement scenario is less important (Fig. 6A).

P-flux coefficients from the model were compared to P-export coefficients fit to Eq. 1 and to literature values. P-export coefficients from the literature for agricultural and urban lands (Fig. 7) represent the likely range for the Lake Mendota watershed based on soil type, land-use practices, and weather conditions (Reckhow et al. 1980, Frink 1991). Both P-flux and export coefficients for agricultural and urban lands were within the range of plausible P-export coefficients, although P-export coefficients estimated using Eq. 1 were on the low end of the range. Urban coefficients estimated from the two methods were almost identical, and well within the published range, since urban lands were not subject to attenuation in our model. Coefficients from our model indicate that on an areal basis, agricultural P flux is generally larger than urban P flux in the Lake Mendota watershed, contradicting results from the P-export co-

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FIG. 5. Land-use data for the model for pre-settlement (A), baseline (B), and urbanization (C) land-use scenarios; and the effective land-area maps for the high-runoff model calibration set for the three scenarios (D, E, F), and for the low-runoff model calibration set for the three scenarios (G, H, I). The effective land area (the area that contributes to P loading) is shown as the black and grey areas in D–I. Tan areas are regions that are classified as non-contributing. In the light grey areas, all P produced on each 1-ha piece of land is transported to water. The monochromatic scale represents the proportion of a pixel's P that reaches water.

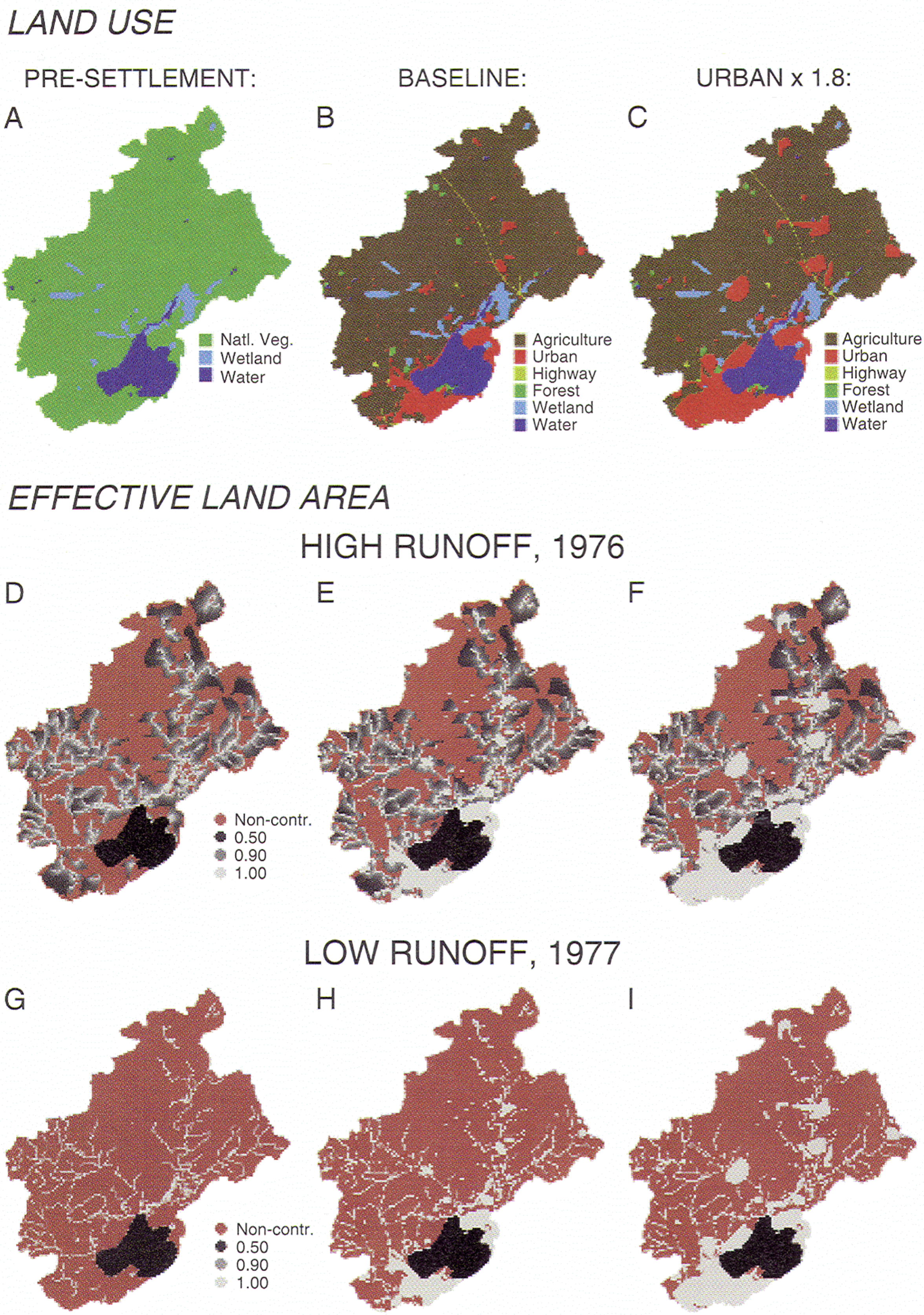
LAND USE
PRE-SETTLEMENT:
BASELINE:
URBAN x 1.8:
A
B
C
Natl. Veg.
Wetland
Water
Agriculture
Urban
Highway
Forest
Wetland
Water
Agriculture
Urban
Highway
Forest
Wetland
Water
EFFECTIVE LAND AREA
HIGH RUNOFF, 1976
D
E
F
Non-contr.
0.50
0.90
1.00
LOW RUNOFF, 1977
G
H
I
Non-contr.
0.50
0.90
1.00

TABLE 1. The cumulative effective land area (presented as % of total contributing area) within given distances from surface water for four values of the transmission coefficient (T), in years with low, average, and high runoff.

Distance from surface water (m)	Transmission coefficient, T			
	1.00	0.97 (High yr)	0.93 (Avg. yr)	0.0002 (Low yr)
0–100	29.9	29.9	29.9	29.9
0–200	40.4	40.1	39.7	29.9
0–1,000	84.7	78.3	71.1	29.9
0–2,000	96.3	85.8	75.2	29.9
0–5,800	100.0	87.3	75.7	29.9

efficient method. There is little change in estimated P-flux coefficients with systematic removal of single sub-watersheds (Fig. 7).

The range of P loading estimated by the model is much lower than the range estimated from published P-export coefficients (Fig. 8). The most striking difference among land-use scenarios is the six-fold increase in loading that occurred from pre-settlement to baseline land use for all three runoff conditions. This increase in loading is due to disturbance of vegetated land by both agriculture and urbanization. The urbanization scenario led to a modest increase in annual P loading in the high-runoff year, and a larger increase in the low-runoff year. However, the urban area only increased from 9 to 16% of the watershed (Fig. 6B,C). If the entire watershed were urbanized, P loading to the lake would be double its present value (see Table 2).

We examined how the lake might respond to changes in P loading using an empirical model that estimates spring total P (TP) from annual P loading, lake mean depth, flushing rate, and P retention (Dillon and Rigler 1974) (Table 2). P retention was calculated according to Dillon and Rigler (1974), using estimated annual budgets for Lake Mendota (Lathrop 1979). We estimated average summer biovolume of blue-green algae from spring TP using a regression fit to a 15-yr time series (Lathrop and Carpenter 1992; Table 2). Average pre-settlement spring TP concentrations were about five times lower than those of the average baseline condition. Only minor increases in spring TP (2%) followed an 80% increase in urban land during the high-runoff condition, but substantial increases in spring TP occurred during the low-runoff condition (30%). Biovolumes of blue-green algae exceeded nuisance levels (2 μL/L, Lathrop and Carpenter 1992) for average and high runoff conditions with baseline land use, and under all runoff conditions with increased urbanization.

Discussion

Our model relates land use, topography, and annual precipitation to P loading of surface waters. The model could be calibrated at scales ranging from field plots operating over hours or days, to regions of thousands of kilometres over decades. We chose a scale relevant

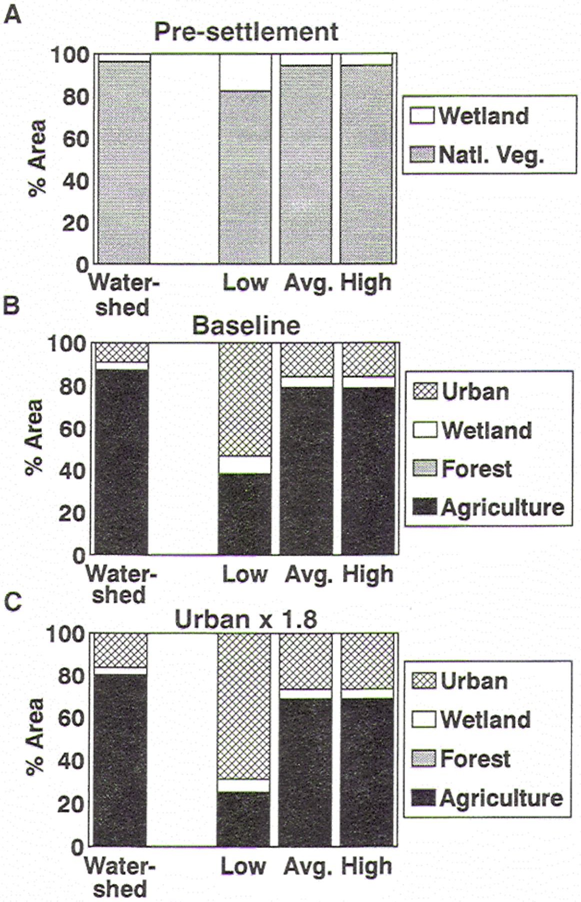

FIG. 6. The relative percentage of each land-use type in the total watershed area (Watershed) and in the effective land area from the three model runs for low, average, and high runoff conditions for pre-settlement (A), baseline (B), and urbanization (C) scenarios.

to the management of entire watersheds from year to year. The spatial grain of our analysis was dictated by pre-existing GIS databases commonly available to researchers and managers (USGS). While many analyses will adopt scales similar to those used here, the scale-dependency of loading models remains an unresolved issue in the literature.

The P-flux vs. P-export approach

The P-flux approach appears to offer advantages over the P-export coefficient approach. The P-flux model fits our data more closely than the P-export model, and estimates loading with less uncertainty than published export coefficients. Since P-flux coefficients account for attenuation of nutrients as they are transported over land, more accurate comparisons can be made among land-use types that vary in attenuation rates. Also, the model can determine which areas of the watershed contribute more strongly to P loading. All of the above factors should help direct management strategies aimed at protecting surface-water quality in the face of changing land use and will be examined in more detail below.

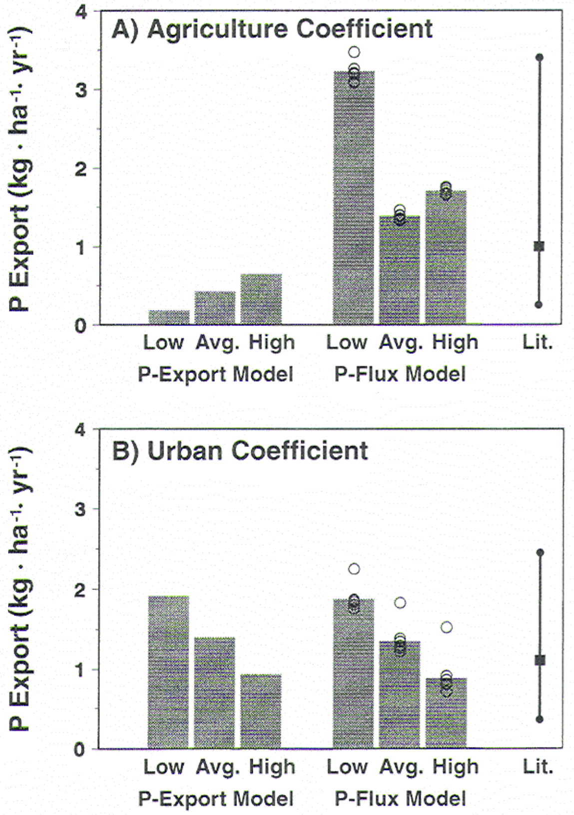

FIG. 7. P-flux coefficients estimated from the P-flux model (Eq. 2; see text) and P-export coefficients calculated from the P-export model (Eq. 1) for low, average, and high runoff conditions for agricultural (A) and urban (B) lands. ○ = jackknifed parameter estimates. Also shown (vertical lines) are the ranges of P-export coefficients compiled for the Lake Mendota watershed from the literature (Lit.), representing most likely (■), and low/high (●) values for agricultural (A) and urban land uses (B).

P-flux coefficients can be viewed as P-export coefficients unbiased by size of watersheds and should be similar to P-export coefficients derived from small plot studies. This distinction is important for agricultural lands, where attenuation of nutrients transported over land can be large. Consequently, P-flux coefficients for agricultural lands estimated from the model are much larger than coefficients estimated from Eq. 1, but are well within the range of plausible literature values. On the other hand, attenuation does not seem to be an important factor for urban lands. Estimated urban P-flux and P-export coefficients are close to literature values, corroborating our assumption that little attenuation occurs in urban areas.

For all calibration runs of the model, agricultural flux coefficients are greater than urban flux coefficients. This result contrasts with most studies that present both agriculture and urban export coefficients (Rast and Lee 1978). Past studies, at the scale of the watershed, have

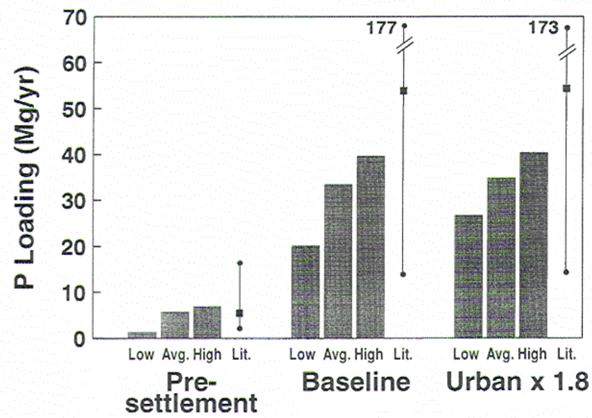

FIG. 8. Total P loading calculated for pre-settlement, baseline, and urbanization scenarios from the three model runs for low, average, and high runoff conditions; and literature estimates of P loading for each scenario using low, most likely, and high export coefficients for the Lake Mendota watershed (Lit).

not estimated the attenuation of P flow with distance from surface waters. Neglect of attenuation could lead to large errors in estimating export coefficients and may explain part of the enormous variance in agricultural export coefficients in the literature.

The additional parameter in the P-flux approach, the T coefficient, provides a measure of attenuation. The actual value of T determines the rate of attenuation of nutrients for each 100 m of travel within the contributing area. T is close to 1 for two of three model fits,

TABLE 2. Total P loading, spring total phosphorus (TP), and average blue-green algae (BGA) biovolume during summer months for four land-use scenarios during low, average, and high runoff conditions. Note that BGA biovolume does not include other algal taxa.

Scenario	P loading (kg/yr)	Spring TP* (mg/L)	BGA biovolume† (μL/L)
Pre-settlement			
Low	1352	0.004	0.00
Avg	5778	0.019	0.00
High	6958	0.023	0.15
Baseline			
Low	20 200	0.066	1.67
Avg	33 530	0.097	2.77
High	39 660	0.129	3.90
Urban × 1.8			
Low	26 710	0.087	2.41
Avg	34 789	0.108	3.16
High	40 380	0.131	3.97
All urban			
Low	99 450	0.323	10.78
Avg	71 620	0.233	7.59
High	46 550	0.151	4.68

* Based on model that estimates spring TP from TP loading (Dillon and Rigler 1974).

† Obtained from regression equation of spring TP and summer average of BGA in Lake Mendota from 1976–1990 (R^2 = 0.31, P = 0.03) (Lathrop and Carpenter 1992).

and close to 0 for the other model fit. A major difference between the P-flux and P-export approaches is that the P-flux approach applies only to the contributing area of the watershed, which is ≈50% of the total area in the Lake Mendota watershed. In contrast, the P-export method applies to the entire watershed area. When T is close to 1, then, attenuation *within* the contributing area may be relatively small. Perhaps in some watersheds, Eq. 1 (P-export model) is actually a special case of Eq. 2, where the total watershed area equals the contributing area and $T = 1$. This does not appear to be the case for the Lake Mendota watershed.

Contributing area

We attempted to estimate which parts of the watershed contributed most strongly to loading based on topography and distance to water. About half of the Lake Mendota watershed could potentially contribute to surface loading. This low percentage was due to the flatness of much of the watershed, and also the low stream density especially in the Yahara subwatershed. Similar results were found for three physiographic regions in North Texas, where contributing areas ranged from 39 to 89% of total watershed area (Levine et al. 1993). Refinements of our estimates may be possible as digital elevation and loading data become available at finer resolutions (DeVantier and Feldman 1993). At finer spatial and temporal scales, it may be necessary to include factors such as antecedent soil conditions, rain intensity and duration, and groundwater recharge (Black 1991).

An isolated intense storm event on 23–24 February 1977 allowed a check of our contributing-area estimate. At the time, the ground was frozen with little snow cover, so much of the rain should have been converted to surface runoff. We calculated the volume of rain that fell on each subwatershed, and determined the volume of runoff (excluding baseflow) that resulted. The runoff volumes above baseflow at the two largest and predominantly agricultural subwatersheds, Yahara and Sixmile Creek, were 14.1% and 21.7%, respectively, of the total volume of water that fell on each subwatershed as rain. In the two smaller subwatersheds (Spring Harbor and Willow Creek), which are dominated by urban lands, flow volume above baseflow was 54.1% and 47.8% of the total rain volume. These results are consistent with our assumption that large agricultural watersheds are more likely to attenuate water and nutrients than urbanized watersheds.

The actual area of land that contributes to storm runoff, and thus sediment transport, can be relatively small and dynamic (Walling 1983, Hibbert and Troendle 1988, Eshleman et al. 1993). This "variable source area concept" (Troendle 1985, Black 1991) has usually been applied at the event scale for relatively small watersheds. Others have suggested, however, that this concept be applied to large watersheds to explain other processes such as nutrient cycling (Naiman et al. 1992). Our results suggest that it is appropriate to examine P loading from nonpoint sources in a similar context.

In our model, the transmission coefficient is linked to the width of this contributing area, and is highly variable between years of contrasting runoff conditions. Since T attempts to account for the attenuation of sediment-bound nutrients from source areas to watershed outlets, it is similar to the sediment delivery ratio (Walling 1983). The wide range of T estimated by the model suggests strong weather effects on this function, which seems reasonable considering the wide range of sediment delivery ratios that have been found in previous studies. Piest et al. (1975) have shown sediment delivery ratios to vary from 1 to 520% for individual storm events for a single watershed measured over 10 years, and from 1 to 72% for annual estimates. It is reasonable to expect similar variability in T.

Riparian areas

The attenuation of sediment and nutrients by runoff is influenced not only by distance to water, but also by topography, riparian land use, disturbance, and the presence of impervious surfaces (Lowrance et al. 1984, Peterjohn and Correll 1984, Hibbert and Troendle 1988, Byron and Goldman 1989). Some studies have found that including riparian terms does not improve regressions of water chemistry parameters against land-use indices (Omernik et al. 1981). Others have found that regressions improved when riparian land was considered, or weighted more heavily than other lands (Osborne and Wiley 1988, Levine and Jones 1990). Field studies have shown that riparian vegetation can reduce nutrient flow to surface waters (Schlosser and Karr 1981, Peterjohn and Correll 1984, Osborne and Kovacic 1993). Our results suggest that the area of the watershed that contributes most of the loading is much less than the total watershed area, dynamic in width, and strongly dependent on precipitation.

How might land use in the agricultural riparian areas affect P loading? Calculations so far assume that the agricultural land within 100 m of open water (3485 ha) contributes directly to loading ($T = 1$). We estimate the potential effect of forest buffers two ways. First, if all of the riparian agricultural area was forested, annual P loading would be reduced by 15% during the low-runoff year, and by 55% during the high-runoff year. A second estimate assumes that forest buffers are added to riparian agricultural lands and remove 80% of overland P flux to water (Peterjohn and Correll 1984). Annual P loading would be reduced by 44% during the low-runoff year, and by 71% during the high-runoff year. Under both estimation methods, reductions are substantial and imply large water quality benefits.

Annual phosphorus loading

Phosphorus loads from land increase with increasing disturbance, soil erosion, fertilizer applications, and proportion of impervious surfaces (Arnell 1982, Byron and Goldman 1989). The dominant land uses in the Lake Mendota watershed, agriculture and urban land, increase some or all of these factors. The change in land use from pre-settlement to baseline conditions increased P loading by ≈6 times (Fig. 8). This increase is corroborated by paleolimnological evidence of a massive increase in secondary production of zooplankton and large changes in sedimentary pigments in the lake through the 1800s as the watershed was first farmed (Hurley et al. 1992, Kitchell and Sanford 1992). Total annual P loading responded only modestly to an 80% increase in urban land use for several reasons. First, after increasing by 80%, urban land still only made up 16% of the total watershed area. Thus, agricultural areas still exerted the strongest influence on total loading. Second, although the P-flux coefficient for urban areas is lower than that for agriculture, all urban areas contribute to loading regardless of location in the watershed. This is not the case for agriculture, where P originating in non-riparian areas is attenuated before reaching surface waters. These two factors offset one another, so that although urban production of P per area is less than for agricultural lands, none of it is attenuated, resulting in little net change in total annual P loading as agricultural land is urbanized. In addition, much of the agricultural land that was converted to urban use was already in the "contributing area." Larger effects may occur when previously non-contributing land is urbanized.

These results suggest that the sequence in which land-use changes occur is important. Transitions from natural vegetation to land disturbed by either agriculture or urbanization had the strongest effects on total loading, especially when the changes occur in large proportions of the watershed or affect riparian areas. Similar conclusions were reached in a study of land-use changes within the Chesapeake Bay watershed of Maryland (Houlahan et al. 1992).

Although we have not modeled the transient effects of land conversion (e.g., construction site erosion), we can estimate what these may be. Based on sediment loss from construction sites of 227×10^3 $kg \cdot ha^{-1} \cdot yr^{-1}$ (Dane County Regional Planning Commission, *personal communication*), and a P concentration (based on soil tests of available P) of 60 mg/kg sediment (average of all soils in Dane County; Combs and Bullington 1992), we estimate the loss of P from construction sites to be ≈13.5 $kg \cdot ha^{-1} \cdot yr^{-1}$ (not including attenuation). At recent rates of development for the Lake Mendota watershed, construction sites exported ≈1780 kg P/yr. This loading will probably occur in brief, intense events that could be locally significant in space and time, and merit further research.

Climatic variability is often ignored as a driver of P loading (Correll et al. 1992). We found a larger difference in P loading between the high- and low-runoff years than between the baseline and urbanization scenarios. This finding is consistent with studies showing large year-to-year variation in runoff and loading due to weather differences (Menzel et al. 1978, Correll et al. 1992). Agricultural lands are very sensitive to the timing and magnitude of rainfall in determining runoff and loading (Sharpley et al. 1981, Sharpley and Smith 1990, Vaithiyanathan and Correll 1992). In urban lands, this rainfall-loading relationship is decoupled since there is little attenuation of P flow. This decoupling explains why the difference between high and low rainfall years is smaller in urbanized watersheds (urbanization scenario) than in less-urbanized watersheds (baseline condition).

While agricultural and urban areas may be quantitatively similar in the magnitude of P loading, they are qualitatively different in their linkage to surface waters and response to precipitation. Urban areas can be tightly linked to surface waters through storm sewers. Thus, urban lands may export P during small runoff events that are attenuated in agricultural lands. Urban lands export a larger fraction of dissolved P compared to agricultural lands, where the majority of nutrients are transported in particulate form (Frink 1991). Dissolved nutrients are readily available for algal uptake, unlike particulate nutrients, which may sink out of the water column (Browman et al. 1979, Sonzogni et al. 1982).

Implications for lake water quality

In many culturally eutrophic lakes, loading from nonpoint sources represents the major external source of nutrients that support high levels of primary production. This primary production is most commonly dominated by blue-green algae, which are usually P limited and can form noxious blooms (Trimbee and Prepas 1987, Paerl 1988, Lathrop and Carpenter 1992). Long-term monitoring of Lake Mendota has shown great variability in water clarity, a result of variable densities of bloom-forming nuisance blue-green algae (Lathrop and Carpenter 1992). Nuisance levels of blue-green algae are caused by the interaction of (1) events that cannot be predicted or controlled, such as storms causing runoff and lake mixing; and (2) events that are partially controllable, such as sizes of fish populations (leading to presence or absence of large zooplankton), and land-use policies that affect external nutrient loading (Paerl 1988, Carpenter 1992, Lathrop and Carpenter 1992, Rudstam et al. 1993). Years with unusually low runoff and P loading have unusually high water clarity, especially if herbivorous zooplankton are also abundant (Lathrop 1992). The management goal for sustaining water quality is to reduce blooms of blue-green algae, and reducing external P load is one obvious approach to doing this.

Blue-green algal biovolumes >2 μL/L represent poor water quality and bloom conditions (Lathrop and Carpenter 1992). The biovolumes of blue-green algae estimated from the baseline land-use scenario (Table 1) represent the water quality of Lake Mendota of the past several decades, in which blooms were highly variable among years (Lathrop and Carpenter 1992). In contrast, for the urbanization scenario during all runoff conditions, blue-green algal biovolumes were always >2 μL/L. This effect was amplified when the entire watershed was urbanized, in which case summer average biovolumes of blue-green algae were well above critical levels for all runoff conditions. With an increasing proportion of urban lands, P loading will become both elevated and less variable in time. As a result, the probability of blue-green algal blooms will increase.

Management and policy applications

The watershed is the appropriate scale for management of land resources that impact water quality (National Research Council 1992). Piecemeal decision making at local or small scales is known to be risky since contributions from any given hectare of land are minor (Sidle and Sharpley 1991) and effects of seemingly large changes in land use on total annual loading can be difficult to detect. As in wetland management, where it is necessary to examine wetlands' cumulative impacts on water quality as components of landscapes (Preston and Bedford 1988, Johnston et al. 1990), one must consider the location of land management in relation to overland flow and receiving waterbodies.

Failure to consider the spatial distribution of land uses and climatic variability may contribute to failures of management plans to improve water quality in many lakes (Persson et al. 1983, Johengen et al. 1989, Meals 1993). Land use and its spatial pattern are critical for evaluating nonpoint sources of nutrients (Schlosser and Karr 1981). While many studies show that riparian areas strongly impact P loading, our results indicate that the critical width of the riparian region is dynamic and closely linked to total precipitation. Some policies directed at the control of nonpoint-source nutrients recognize critical areas near surface water that are important to total nutrient loading (e.g., Maryland's Critical Areas Act, Houlahan et al. 1992). However, the greatest uncertainties may lie in the definition of the "critical" width of this riparian zone, and the recognition that it may vary from year to year.

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