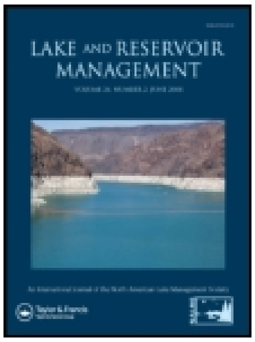

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Drivers of change for lakewater clarity

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Abstract

Baker, L.A., J.E. Schussler and S.A. Snyder. 2008. Drivers of change of lake clarity. Lake Reserv. Manage. 24:30–40.

Lakes in the Upper Midwest have undergone extensive lakeshore development over the past 30 years, raising concerns about eutrophication. We examined 11 case study lakes in Minnesota that had undergone substantial shoreline development over the past 30 years to evaluate drivers of change in clarity. Relationships between current Secchi disk transparency (SDT) and the density of permanent equivalent houses (PEHs) and between change in SDT and change in density of PEHs were not statistically significant. For lakes with large watershed area-to-lake area (WSA:LA) ratios, modeled worst-case scenarios for impacts of shoreline housing show that phosphorus (P) inputs may not be sufficient to reduce SDT. For sensitive lakes, improved P management policies may counteract increased shoreline development, at least in part. For lakes with large WSA:LA ratios, activity outside the shoreline area, particularly agricultural activity, is probably more important than shoreline development in affecting SDT. Although policies considered "lake management" operate at fairly small scales, drivers of change in SDT operate at various temporal and spatial scales, from household to global.

Key words: eutrophication, drivers, phosphorus, Secchi disk, shoreline development

Rapid development of housing has occurred on the shorelines of many lakes in the Upper Midwest. The number of lake homes in many lake-rich counties in Minnesota and Wisconsin has more than doubled since 1970 (Marcouiller *et al.* 1996, Kelly and Stinchfield 1998). People are concerned that this development may cause lake eutrophication and declining water clarity based on 2 premises: (1) lake algae abundance increases with increased phosphorus (P) inputs; and (2) shoreline development increases P inputs to lakes enough to cause increased algae abundance. The first premise is supported for Minnesota lakes by 3 lines of evidence: (1) most Minnesota lakes have nitrogen:phosphorus (N:P) ratios >25:1, indicating P limitation; (2) strong relationships exist between lake P and summertime chlorophyll *a*; and (3) lake clarity, as reflected by Secchi disk transparency (SDT), is directly related to algae abundance (Heiskary and Wilson 2005).

The second premise is more questionable. We do have reason to believe that P inputs from the shoreline increase following development. For example, Graczyk *et al.* (2003) showed that P loadings from lakeshore lawns may be several times higher than P exports from nearby woodland. And when septic systems on shorelines become clogged, P-rich sewage can flow directly to lakes. Additionally, Robertson *et al.* (1998) showed that sewage passing through leach fields in sandy soils can reach underlying groundwater and form plumes that migrate tens of meters down gradient. This means that contaminated groundwater could reach lakes. Nevertheless, shoreline development does not necessarily increase P loading enough to cause increased algae abundance and declines in SDT.

Furthermore, shoreline housing development does not occur in a vacuum. Other social, economic and regulatory changes often occur simultaneously with shoreline development, and these also affect P inputs to lakes. For example, during the past 30 years, about 80% of the dairy farms in Minnesota have ceased operation, while manure regulations have gradually become stricter for the ones that remain. Both trends have probably led to decreased P inputs from dairy farms to streams and lakes. Some areas that once had septic systems are now sewered, reducing the risk of P contamination of local lakes, and the amount of P in wastewater has declined due to a state-wide ban on P-containing detergents. Agricultural

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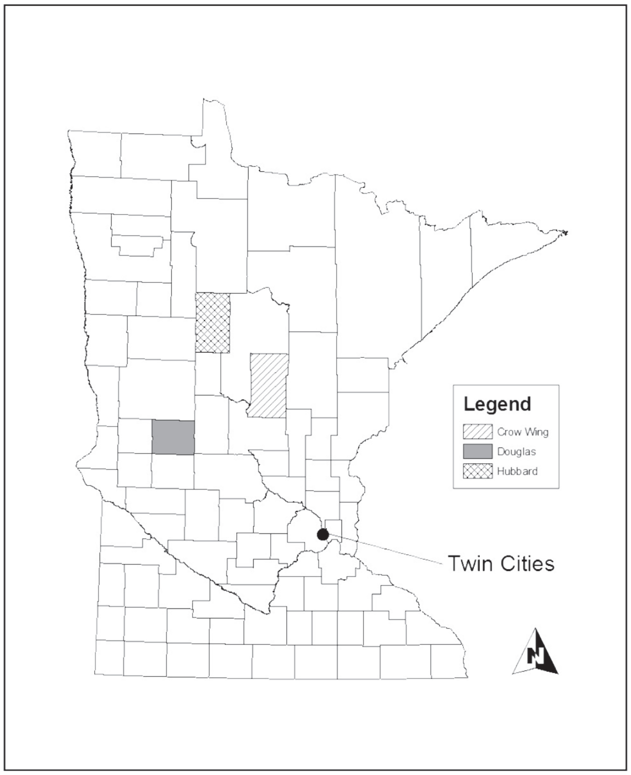

Figure 1.-Location of case study counties.

practices have become more efficient, reducing the amount of fertilizer P needed to achieve a given yield.

This study examined historical changes in watersheds and lake clarity in 11 recreational lakes in Minnesota that have undergone shoreline development over the past 30 years. Our goals were to: (1) determine which of several individual metrics of lakeshore development could serve as good predictors of lake water clarity; (2) identify major drivers of watershed P balances; and (3) determine the qualitative effect of each of these drivers on watershed P balances. Our results support a view of countervailing drivers affecting watershed P balances, and that shoreline development, *per se*, is generally not a good predictor of increasing P loading or declining clarity.

Methods

Selection of case-study lakes

We selected case study lakes typical of highly developed recreational lakes in Minnesota. We first identified 2 regions that had undergone rapid development using the U.S. Forest Service's Changing Midwest Assessment study (Potts *et al.* 2004) and information on demographic and housing development trends (Hammer *et al.* 2004). One region encompassed 28 lakes in the Northern Lakes and Forest Ecoregion (Province 212) and the other encompassed 16 lakes in the Northern Central Hardwoods Ecoregion (Province 222). We then screened to remove lakes that had maximum depths <10 m and with surface areas >80 ha and <810 ha. The depth criterion roughly excludes nonstratified lakes, and the surface area criterion was intended to include most important recreational lakes while excluding several extremely large lakes that would have been difficult to analyze. Candidate lakes were then screened for adequate Secchi disk transparency (SDT) data, our metric for "clarity." Criteria included: (1) SDT data for most of the period 1985–2000; (2) reasonably even distribution of data across the study time period; and (3) multiple observations within most years. Finally, we interviewed local water planners who recommended inclusion or exclusion of lakes based on various other factors, such as unusual hydrologic conditions. The screening process resulted in 11 case study lakes in 3 counties (Hubbard, Crow Wing and Douglas; Fig. 1 and Table 1).

Data sources

SDT database

To evaluate long-term changes in SDT, historical SDT data were compiled for the period June–September. Because this approximately represents "summer" in Minnesota, we used simple linear regression with no seasonal adjustments to analyze trends. Data from the Minnesota Pollution Control Agency (MPCA) were available for all case study lakes. The Alexandria Lakes Area Sanitary District (ALASD) also had a SDT database for 3 lakes (Le Homme Dieu, Victoria, and Darling). For these lakes, the ALASD and MPCA databases were merged for regression analysis. For Lobster Lake, 2 sampling locations had very similar averages, so data from the 2 sites were merged. For Gilbert Lake, we used the "east" site, which was in the main basin.

Regression analysis was used to determine trends in SDT, using year as the independent variable and average summertime SDT as the dependent variable. For all but 2 lakes, predicted (modeled) changes in SDT values were computed for the period 1985–2000 using coefficients from the regression model. For 2 lakes with very sparse data prior to 1990, we modeled changes for 1990–2000.

Shoreline housing data

Shoreline housing counts for 1969 were obtained from an unpublished DNR database; data for 1982 were obtained from Kelly and Stinchfield (1998), and data for the present (~2003) were obtained from GIS databases in county tax assessors' offices. For all 3 periods, houses were classified as "recreational" or "permanent." We collected housing data

Table 1.-Characteristics of the case study lakes.

County and Lake Name	Lake area (LA), ha	Watershed area (WSA), ha	WSA:LA	Ave. depth, m	HRT, years[a].	SDI[b]
Crow Wing County						
Big Trout	567	3,502	6	19	11.8	1.7
Gilbert	146	2,301	16	7	1.7	2.8
Hubert	526	1,601	3	13	14.3	1.4
Sibley	162	13,906	86	6	0.3	2.8
Douglas County						
Darling	405	47,519	117	9	0.3	1.6
Le Homme Dieu	769	13,706	18	13	5.2	1.6
Victoria	162	6,603	41	9	1.6	2.5
Lobster	486	14,106	29	10	2.4	3.7
Hubbard County						
Belle Taine	587	29,112	50	8	0.7	4.3
Fish Hook	668	63,026	94	11	0.5	1.3
Long	810	6,402	8	20	9.9	3.0

Sources: Minnesota Pollution Control Agency (MPCA) and the Minnesota Department of Natural Resources (MDNR).

[a] HRT = hydraulic residence time = lake volume ÷ outflow

[b] Shoreline development index = shoreline length/$(2^*(\pi^*\text{area}))^{0.5}$, from Wetzel (1983).

over the 3 time periods to illustrate the long-term pattern of change, although only the data from 1982 and 2003 were used in statistical analysis. We developed a metric "permanent effective houses" (PEH) to integrate the impact of permanent and seasonal housing. PEH is the sum of permanent homes plus an adjustment factor multiplied by the number of seasonal homes:

$$PEH = W_pN_p + W_sN_s \qquad (1)$$

Where W_p;W_s = weighting factors for permanent (p) and seasonal homes (s) = number of days used per year/365; and N_p;N_s = number of permanent and seasonal homes.

We assumed that permanent homes are used throughout the year ($W_p = 1.0$). We used an estimate of 100 user days per year for seasonal homes based on several studies of lake home use (Marcouiller *et al.* 1996, Stewart and Stynes 2006), yielding $W_s = 0.28$.

Other data

Land cover data for each watershed were developed for 1980 and 2000 using classified satellite imagery data for the Upper Midwest. The classification was broken into 6 Anderson level I classes (Anderson *et al.* 1976). County-level data were obtained at decadal intervals from 1970 to 2000 for agricultural production (Minnesota Agricultural Statistics), economic activity (SETA 2005), and population (U.S. Census). Finally, we reviewed major regulations at the national, state, county, and local levels that had an impact on the case study lakes. We also visited each lake and interviewed county water planners, agricultural extension agents, sewage treatment plant administrators, and others to evaluate how regulations were being implemented.

Modeling changes in watershed P inputs

We previously calculated whole-watershed P balances for our 11 case study watersheds for the present period (Schussler 2005, Schussler *et al.* 2007). Briefly, the watershed P balance is:

$$\text{Input P} = \text{deliberate P exports} + \text{P retention} + \text{P stream export} \qquad (2)$$

For our case study watersheds, inputs included food for humans and livestock, lawn and agricultural fertilizer, and atmospheric deposition. Deliberate exports included net outputs of animal products (meat and milk) and crops. Sewage can be imported to a watershed or exported from it via sewers. If sewage enters a watershed and is treated, the sewage sludge can then be exported from the watershed (by truck) or applied to cropland within the watershed. Retention refers to accumulation of P within the terrestrial watershed.

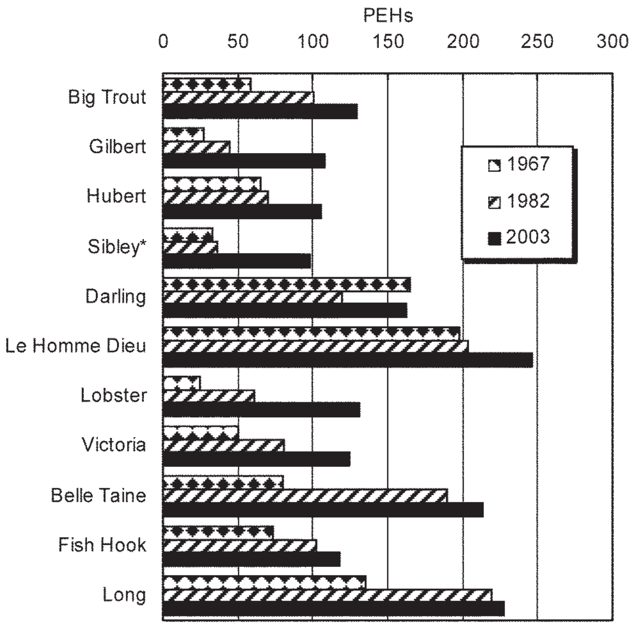

Figure 2.-Change in permanent equivalent houses (PEHs) on the shorelines of the case study lakes. For Sibley Lake (marked with an asterisk, *), there were a small number of homes (~6) on the Cass County side of the shoreline for which we did not have historical records.

The term retention is interchangeable with "accumulation" or "storage," and also has units of mass time^{-1}. Stream export transports P from the watershed to the lake.

We could not construct complete quantitative historical P balances for our case study watersheds due to lack of data, so we sought to determine the direction of change in major components of the terms in equation 2. The period of consideration is approximately 1980–2000. This approximation of endpoints was necessary because some types of data are not available at precisely these times.

Results

Relationship between shoreline development and SDT

Shoreline housing trend

Shoreline housing, as measured by PEHs, increased by an average of 144% from 1967–2000 and by 59% from 1982–2000 (Fig. 2). Most of the increase in PEHs was due to the construction of new homes, not transition from seasonal to permanent use. On average, the percentage of shoreline houses for permanent use increased only slightly, from 45% in 1967 to 50% in 2000. The lakes fell into 3 fairly distinct groups. The number of shoreline homes increased by >100% for Group 1 lakes, with half or more current houses as seasonal residences. This group included Big Trout, Hubert, Sibley, Lobster, Belle Taine, and Long. With the exception of Sibley Lake, these lakes have surface areas >400 ha and are located well outside city boundaries. All had fewer than 8 houses per km of shoreline in 1969. Group 2 lakes also had >100% growth in shoreline houses since 1969 and fewer than 8 houses/km in 1969, but the majority of houses were permanent residences. These lakes are located just outside cities (Victoria outside Alexandria; Gilbert outside Brainerd; Fish Hook outside Park Rapids), close enough that lakeshore residents could readily commute to these cities. Group 3 lakes (Darling and Le Homme Dieu) were developed earlier (>10 houses/km in 1969), had slower growth in shoreline housing growth since 1969, and most homes were permanent dwellings.

SDT trend

Based on our regression analysis, 8 of the 11 lakes exhibited either no significant change ($\alpha = 0.05$) or an actual increasing SDT trend (Table 2) over the study period. Predicted SDT changes were <1 m for all but 2 lakes. The exceptions were Belle Taine, which had a predicted SDT change of +2.5 m since 1990, and Big Trout, which had a predicted change of +1.5 m for the period 1985–2003. Three lakes, Hubert, Long and Sibley, had a significant downward trend in SDT. For Hubert and Long lakes, the predicted changes from 1985–2003 were >0.5 m.

Relationship between shoreline development and SDT

Lake development, as measured by several metrics of shoreline housing, does not appear to be closely correlated with clarity. There was no significant relationship (at $\alpha = 0.05$) between changes in SDT and changes in PEHs per km^2 lake surface from the 1980s to present, nor any relationship between current average SDT and current PEHs per km^2 (Fig. 3). There were also no significant relationships when change in SDT was related to change in PEHs per km of shoreline or when current SDT was related to current PEHs per km of shoreline. The lack of relationships between shoreline development metrics and SDT suggests that shoreline housing development, even when adjusted for seasonal occupancy, is a poor predictor of SDT.

Our metrics of housing development (PEHs normalized to lake area or shoreline) did not appear to be good predictors of SDT. Stedman and Hammer (2006) also observed a lack of relationship between shoreline development and SDT. We therefore shifted our focus from simple metrics of development to evaluating changes in the P balances of the case study watershed, based on the widely accepted premise that SDT is directly related to P inputs.

Table 2.-Regression equations for SDT trend, screened for June–September measurements only. For all lakes except Belle Taine and Sibley the model predictions are for 1985–2003. For these 2 lakes, the model period is 1990–2003.

Lake	Regression period	x intercept	Slope	r^2	p level	Predicted (modeled) Start (1985 or 1990)	Predicted (modeled) 01-Jun-03	Predicted change, m
Significant upward trend (positive slope; p < 0.05)								
Belle Taine	1990-2003	-12.43	5.21E-04	0.19	<0.01	4.8	7.2	2.5
Big Trout	1975-1996	-2.52	2.35E-04	0.20	<0.01	4.8	6.4	1.5
Lobster	1985-2003	-0.92	1.02E-04	0.03	<0.01	2.5	2.9	0.5
Victoria	1977-2003	1.012	1.05E-04	0.05	<0.01	2.2	2.9	0.7
Significant downward trend (negative slope; p < 0.01)								
Hubert	1974; 1987-2003	7.69	-1.09E-04	0.27	<0.01	4.3	3.6	-0.7
Long	1984-2003	6.72	-8.90E-05	0.09	<0.01	3.9	3.4	-0.6
Sibley	1987-2003	2.88	-3.05E-05	0.01	0.04	1.9	1.7	-0.1
No significant trend (p > 0.05)								
Darling	1974-2003	2.19	1.74E-05	0.004	0.42	--	--	--
Le Homme Dieu	1976-2003	1.58	4.50E-05	0.11	0.055	--	--	--
Fish Hook	1988-2003	3.80	-8.93E-06	0.00	0.84	--	--	--
Gilbert East	1988-2003	7.00	-3.87E-05	0.01	0.46	--	--	--

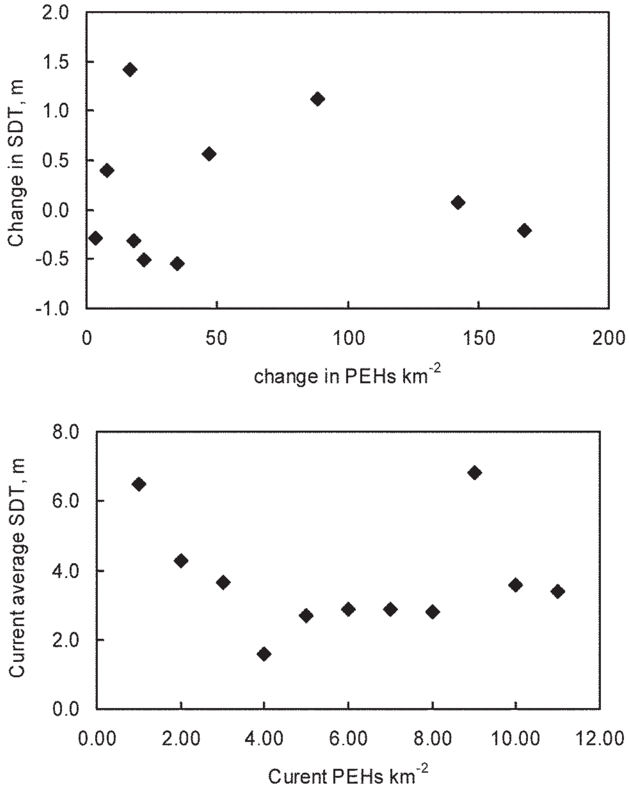

Figure 3.-Top. Change in PEHs per km² of lake area versus change in SDT, 1982 to present. The datum for Belle Taine is missing because there were no SDT data for the 1980s. Bottom. Current PEHs per km² lake versus current SDT.

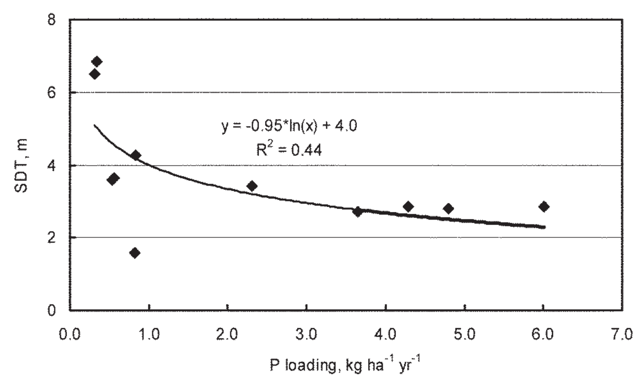

Figure 4.-Areal watershed P input SDT (present period, average for 2000–2003). The regression line is significant at the 0.02 level.

Relationship between SDT and phosphorus (P) inputs

Total areal P inputs to the case study watersheds were significantly related to the current SDT of lakes in the watersheds (Fig. 4). With one exception (Sibley), average SDT values >3 m occurred only when watershed P inputs were <3 kg ha^{-1} yr^{-1}. Since SDT values among lakes are directly related to watershed P input in the present, it is reasonable to infer that SDT for a given lake would change in response to changes in P input over time.

If watershed P inputs are statistically related to SDT, then why did shoreline housing increases not reduce SDT in our case study lakes? We evaluated 3 hypotheses. The first hypothesis is that the amount of lakeshore development,

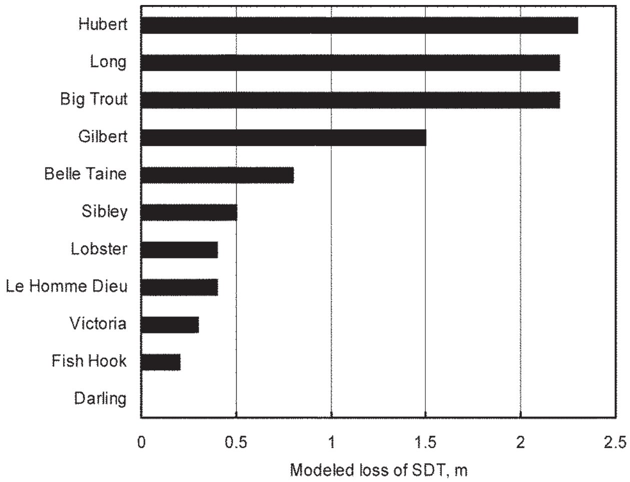

Figure 5.-Modeled loss of SDT for worst-case scenarios of shoreline development.

as measured in PEHs, could not increase P inputs enough to increase algae abundance and reduce SDT, even with a worst-case scenario (developed below). The second hypothesis is that P loading to lakes per shoreline PEH has declined over time due to better management. This might be caused by changes in P input (*e.g.*, changes in lawn fertilization practice), changes in P retention (*e.g.*, saturation of septic system leach fields), or changes in deliberate P export (*e.g.*, sewered shoreline housing). If this were the case, the effect of improved P management might counterbalance the effect of increasing PEHs on the shoreline. The third hypothesis is that changes in the P dynamics of the watersheds outside the shoreline region affect P loadings to the lake.

To evaluate the first hypothesis, we modeled the potential (worst-case) impact of lakeshore development to determine the maximum potential decline in SDT that could result from shoreline development across the range of hydrological and morphological conditions of the case study lakes. To analyze the other 2 hypotheses, we analyzed the likely change in direction of P inputs, P outputs, and P retention in each of the case study watersheds from about 1980 to the present.

Sensitivity of SDT to shoreline development: worst-case scenario

Our worst-case scenario assumed 100% shoreline development, in accordance with Minnesota's Shoreline Zoning Rule. These rules stipulate one house per 100 ft of shoreline and a property area of 20,000 ft^2 for General Development (GD) lakes, and 150 ft of shoreline and a property area of 40,000 ft^2 for Recreational Development lakes. Because model results were virtually indistinguishable, we present only results from the GD scenario. To calculate P inputs, we assumed a P loading of 0.72 kg/capita-yr (Schussler *et al.* 2007), with 2.6 people per household (the average for the counties in which case study lakes were located). We further assumed that half of each property was fertilized, at a "medium fertility" rate of 0.5 lb P_2O_2/1000 ft^2 and that all P from human waste and lawn fertilizer entered the lake, a total of 2.8 kg/household per year.

Predicted reductions in SDT were calculated using the MINLEAP (Minnesota Lake Eutrophication Analysis Procedure) model (Wilson and Walker 1989) using the ecoregion-specific default assumptions to compute SDT. The default P loadings in MINLEAP are intended to represent "minimally impacted" lakes within each ecoregion (Wilson and Walker 1989). The model was rerun by adding the calculated shoreline P loading for our worst-case scenarios to the original P load and then computing the ratio of new P loading to background loading. This ratio was used to increase the stream P concentration value in the model.

Modeled scenarios (Fig. 5) show that only 4 lakes (Hubert, Long, Big Trout, and Gilbert) would lose more than 1 m SDT. We consider these lakes "sensitive" to shoreline P inputs. Lakes with modeled SDT declines >1 m had WSA:LA ratios ≤ 8 years and HRTs ≥ 10 years. Five others would lose <0.5 m SDT and are considered "insensitive". Insensitive lakes had WSA:LA ratios $\geq 18{:}1$ and HRTs ≤ 5 years. Complexity of shoreline, as measured by SDI, was not closely related to modeled loss of SDT.

Evaluation of trends in watershed P balances

We evaluated trends in P inputs, deliberate P exports, and P retention (see equation 2) for each of the case study watersheds during the period 1980–2000, to the extent possible. Historical data to perform this analysis are not complete, hence we could generally identify only the trend in direction (upward, downward, or none) in P flux terms (Table 3).

Changes in P inputs to watersheds

Change in crop fertilizer inputs. The average state-wide effective application rate of P fertilizer to corn (% planted acres fertilized × application rate per fertilized acre) decreased by 18% from 1980–2000. The P fertilization rates for soybeans and wheat remained unchanged (ERS 2005). Acreage planted in major crops (corn, soybeans and wheat) in the study counties has also changed since 1980, but the changes are smaller and not in a consistent direction. Assuming that changes in planted acreage and fertilization rates in the case study watersheds paralleled changes at the county and state level, we estimate that the reduction in crop P fertilization is equal to 15–32% of the current total watershed P inputs for 5 watersheds with extensive agricultural land (Darling, Victoria, Fish Hook, Lobster, and Le Homme Dieu; see

Table 3.-Summary of likely changes in components of the P balances for the case study lakes. Symbols: ○ = little or no change; ↑ = probable significant increase; ↓ = probable significant decrease; ? = probable significant change, but direction uncertain. SDT trends ($p < 0.05$ level; see Table 2) are in the rightmost column.

Lake	P inputs				P outputs			Retention, % and direction	SDT Trend
	Detergent	Crop fertilizer	Human food input	Livestock feed	Crop P output	Animal products	Sewage or sludge		
Big Trout	↓	○	○	○	○	○	○	63 ○	↑
Gilbert	↓	○	↑	○	○	○	○	89 ○	○
Hubert	↓	○	↑	○	○	○	○	59 ↓	↓
Sibley	↓	○	○	?	○	○	○	64 ↓	↓
Darling	○	↓	○	↓	↑	↓?	○	28 ↓	○
Le Homme Dieu	↓	↓	○	○	↑	↓?	○	10 ↓	○
Victoria	○	↓	○	↓	↑	↓?	○	18 ↓	↑
Lobster	○	↓	○	↓	↑	↓?	○	53 ↓	↑
Belle Taine	↓	○	○	○	○	○	○	70 ↑	↑
Long	○	○	○	?	○	○	○	70 ↓	↓
Fish Hook	○	↓	○	○	↑	○	○	31 ↑	○

trend arrows in Table 3). For the other watersheds, estimated reductions in fertilizer P inputs were <10 % of current watershed P input.

Change in human food inputs. For 9 of our case study lakes, human food comprised <15% of total watershed inputs, so changes in food input would have had little effect on changes in total watershed P inputs (no trend in Table 3). For the watersheds of Hubert and Gilbert lakes, human food was 29% and 50% of total P inputs, respectively. Even population growth on the shoreline of these 2 lakes represents an increase in P input from human food that is ~5% of current watershed P input.

Inputs of detergent P. Minnesota's P detergent ban enacted in 1977 reduced inputs of P to households. The ban caused wastewater P concentrations to decline by ~6 mg L^{-1} (Litke 1999). Using an estimate of 260 L $capita^{-1}$ day^{-1} for interior water use (Mayer *et al.* 1999), this corresponds to a reduction of 0.6 kg $capita^{-1}$ yr^{-1}, roughly the same as inputs from human food. If the ban had not been put in place, watershed inputs from detergent P would have be >10% of watershed P input to all lakes except Darling, Victoria, Lobster, Long, and Fish Hook.

Livestock food inputs. Eight of our case study watersheds had no importation of animal feed from outside the watershed. For these watersheds, we presumed that crops grown within the watershed provided sufficient food for livestock in 1980, as they do now. Currently, livestock feed imported into the watersheds of Lobster, Victoria, and Darling accounted for 56%, 51% and 17% of total watershed P inputs, respectively. Livestock production in Douglas County has declined substantially since 1980 (see following section). This, combined with improved feed rationing of remaining animals, has probably led to substantial reduction of P inputs in feed to these watersheds (Table 3).

Change in deliberate P exports

Exports of sewage and sludge. The watersheds of lakes Victoria and Darling are mostly sewered and export sewage to the Le Homme Dieu watershed. Sewage from the city of Pequot Lakes exports treated sewage effluent to a land application site outside the Sibley Lake watershed. Although the P fluxes associated with these exports are large, there has been no major change in these exports over the study period.

Change in export of crops and animal products. Deliberate export of crops and animal products currently accounts for <20% of total P inputs to the watersheds of the Crow Wing and Hubbard County lakes; so changes in animal production methods since 1980 have probably not had a major impact on P export from these watersheds. Agricultural products account for 38–58% of total inputs to the watersheds of the 4 Douglas County lakes (Darling, Lobster, Le Homme Dieu, Darling, and Victoria). Intensification of agriculture has probably increased P utilization efficiency for both crop and animal production. However, agricultural production in Douglas County has changed dramatically, with less production of animal products (milk production declined by 10% and numbers of beef cows and hogs declined by ~50%) and a simultaneous doubling of corn and soybean production. The direction of the change in export of agricultural products is almost certainly positive because it is inherently more efficient from a P-balance perspective to export crops directly than to grow crops, feed them to animals, and export animal products.

Change in watershed P retention

Watershed P retention may have changed in some of our case study watersheds, altering the overall P balance. At present 10–89% of input P is retained in the case study watersheds (Schussler *et al.* 2007). Watersheds with deliberate exports of sewage or agricultural products (Le Homme Dieu, Victoria, Darling, Fish Hook, and Lobster) had lowest P retention values (Table 3).

Agricultural systems. Several trends in crop and livestock management affect retention of P in watersheds. First, inadvertent export of P from feedlot runoff has probably been reduced over the past 20 years as the result of tightening regulations, improved enforcement of existing regulations, and consolidation of livestock operations. Second, cost-share programs have encouraged farmers to build manure containment facilities, fence stream banks, remove highly erodible land from production, shift to conservation tillage and implement other best management practices (BMPs) that would generally increase P retention.

Simultaneously, there has also been a trend toward more efficient utilization of P: more efficient transfer of P from fertilizer to crops, from crops (feed) to animals, and from animals (manure) to crops. The increase in efficiency tends to increase deliberate export of P and thereby decrease P retention. The P conversion efficiency of the agricultural systems (P in agricultural exports ÷ P in feed + fertilizer) varied widely among our case study watersheds. For case study watersheds in which agricultural P inputs were >50% of total P input, agricultural P efficiencies generally were >40%. The 2 exceptions were the watersheds of Sibley and Long lakes, which had very low agricultural P transfer efficiencies (25% and 15%, respectively). Unlike the other agriculturally intensive watersheds with low P retention and high P export, these 2 watersheds had >60% P retention and low deliberate export (~around 20%). Notably, these 2 lakes also had declining SDT trends, consistent with the hypothesis that P retention capacities in theses watersheds is becoming exhausted, resulting in increased P export to lakes.

For the other intensively farmed case study watersheds, agricultural P efficiencies were: Darling (82%), Fish Hook (115%), Le Homme Dieu (114%), Lobster (51%), and Victoria (91%). For these watersheds, P retention is probably declining due to increased agricultural efficiency. In Wisconsin, retention of P in agricultural soils declined from 41% of input P in 1980 to 16% of input P in 2000 as crop exports rose (Bundy 1998). Although there are no comparable studies in Minnesota, it is likely that a similar trend is occurring in the Douglas County watersheds.

Septic systems. Most of the houses in the case study watersheds in Hubbard and Crow Wing County are on septic systems. For new septic systems, P is removed from sewage by adsorption onto soils underlying the leach fields, retaining P. Over time, the P adsorption capacity of sandy soils may become exhausted, allowing phosphate to migrate to groundwater (Robertson *et al.* 1998). Many homes surrounding our case study lakes are 20 or more years old, and we postulate that P is leaching through some of these septic leach fields, even if they are still properly functioning and legally compliant. This means that P retention is declining, and movement of P to lakes is increasing. We think this mechanism may be responsible for the declining SDT of Hubert Lake. This lake is nearly surrounded by houses, many built prior to 1980 on a shoreline with sandy soils. It has a hydrologic residence time >10 years, which means that the lake is more vulnerable to a given amount of P input than a lake with a shorter residence time. Also, no other sources of P can account for declining clarity. We have no reason to believe that other mechanisms (such as changing inputs of dissolved organic carbon or inorganic turbidity) would be responsible for the decline in SDT.

In addition to P breakthrough in properly functioning (compliant) systems, many noncompliant systems are or have been on the shorelines of our case study lakes. Septic surveys conducted from the late 1980s to the present on most of the case study lakes with shoreline septic systems reported nonconformance rates of 30–60%, with roughly half of these classified as "imminent health threat" or "polluting." Most of these systems have been brought into compliance, presumably increasing the retention of septic system P.

Summary of changes in P inputs, exports and retention

Most lakes had countervailing changes, with some tending to increase P export and others tending to reduce P export.

Lakes with declining SDT. Only 2 lakes (Hubert and Long) had observed SDT declines >0.5 m. Phosphorus inputs to these lakes have presumably increased. As shown earlier, Hubert is particularly susceptible to eutrophication from shoreline housing. The most likely cause of increased P input to Hubert is decreased retention of P entering septic systems. Long Lake has an inefficient agricultural system (little agricultural export relative to fertilizer P inputs) and high watershed P retention, indicating a potential for P saturation and increased stream P concentrations. Long Lake also has many older homes on septic systems along its shoreline. We speculate that one or both factors could be the cause of decreased SDT in Long Lake.

Lakes with increasing SDT. Four lakes, Belle Taine, Big Trout, Victoria, and Lobster, have had significant increases in SDT since 1980. Septic surveys on the shoreline of Belle Taine, Big Trout, and Lobster Lakes were followed by measures to bring septic systems into compliance. This measure undoubtedly increased watershed P retention. Lakes Lobster and Victoria have experienced ~50% growth in shoreline houses since the early 1980s, but this housing replaced agricultural fields, so it is not clear that this resulted in decreased P inputs. These lakes may also have experienced declining P inputs from remaining agriculture as the result of decreasing livestock production and increasing agricultural efficiency. Additionally, about 40% of the homes on the shoreline of Lake Victoria are sewered, so 40% of the P from human food is exported from the watershed.

Lakes with no significant change in SDT. Most of the case study lakes did not experience declines in SDT, even though most experienced large increases in shore land housing and a conversion from seasonal to year-round occupancy. For these lakes, countervailing drivers apparently result in little or no net change in lake clarity. For shorelines where housing has replaced forest, shoreline development increases inputs of human food and lawn fertilizer but is offset in part by the large decline in P input that resulted from detergent P ban in 1977. Septic surveys from the mid-1990s to present resulted in replacement of failing systems for several lakes. This measure increases P retention of the watershed.

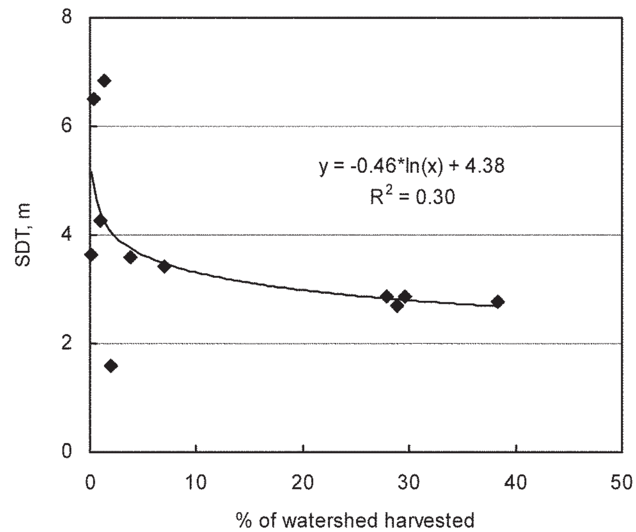

Figure 6.-Percentage of watershed in harvested cropland vs. SDT for the case study lakes.

Discussion

Shoreline Housing Development and SDT

Shoreline housing development does not necessarily lead to declining SDT for several reasons. First, for watersheds with large WSA:LA ratios and short HRTs, even the worst-case shoreline development scenarios may not add sufficient P to cause large increases in algae abundance or declines in SDT. Based on our modeling analysis (Fig. 5), a rough rule of thumb for sensitivity to shoreline development would be: lakes with WSA:LA ratios >20:1 and HRTs <5 years are not likely to undergo large declines in SDT due to shoreline development alone. We note that shoreline housing may affect other ecological characteristics of lakes, such as emergent macrophytes (Radomsky and Gorman 2001), woody habitat (Christensen *et al.* 1996) or periphyton.

For lakes that are more sensitive to eutrophication (WSA:LA ratios <5; HRT >10 years), P inputs per house probably vary enormously. The detergent P ban enacted in 1977 reduced P inputs to septic systems by roughly one-half, lengthening the time to reach P saturation in leach fields and reducing the amount of P entering lakes from failed septic systems. The enactment of septic inspection programs over the past 10–15 years has resulted in replacement of many failing septic systems, increasing the retention of sewage P, and reducing P inputs to lakes. However, we raise the question of long-term sustainability of P retention in septic leach fields. We postulate that the declining SDT for Lake Hubert is the result of P saturation of septic leach fields, which resulted in increased P inputs to the lake. Understanding the potential for P saturation of septic system soils and the potential impact on Minnesota's lake should be a research priority. In general, however, management efforts to reduce P inputs to lakes have a countervailing effect to increases in shoreline housing numbers.

Most P inputs for lakes with large WSA:LA ratios occur outside the immediate shoreline area. For our case study lakes, the percentage of agricultural land in the watershed is a better predictor of SDT than shoreline development (Fig. 6), though still marginally significant ($p = 0.08$). Even so, on balance it appears that inputs of P to streams from agricultural land have been reduced due to more efficient crop harvesting methods, improved animal feed rationing, better handling of manure P, and reduced erosion. This is consistent with statewide and nationwide assessments that show downward trends of P concentrations in streams and rivers (Smith *et al.* 1994, MPCA 2006).

P balances as long-term assessment tool

Tracking watershed P balances is an effective tool for long-term watershed management. Although we were limited in our analysis of historical P balances in this study, current technology (*e.g.*, geographic information systems in county planning offices) and modern databases (*e.g.*, digitized land use based on high resolution satellite imagery) make it pos-

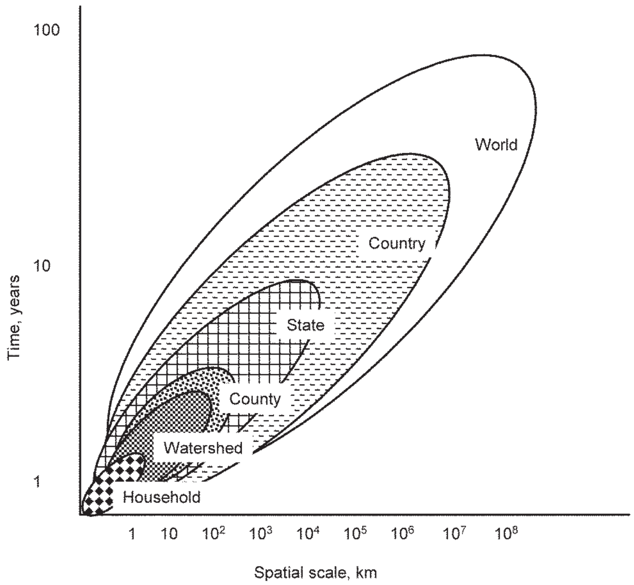

Figure 7.-Temporal and spatial scales of actions influencing lake clarity. Ranges indicated by bars are qualitative estimates.

sible to use watershed P balances as a practical tool for lake management at the present time (Schussler *et al.* 2007) and into the future. Developing P balances at regular intervals, such as 5 years, would be a valuable tool for evaluating the impacts of development, changes in farming practices and deliberate nutrient management practices within a watershed. Whole-watershed P balances could be one of several feedback loops developed to guide watershed managers in an adaptive management framework.

Long-term P balances would also greatly enhance interpretation of stream monitoring studies. Another valuable tool for augmenting interpretation of watershed P balance trends would be paleolimnological analysis. Paleo studies could provide a longer and more complete record of changes in lake trophic status, plus information on sediment accumulation rates (indicating changes in watershed erosion) and P accumulation (to corroborate inferences from historical analysis). Another major advancement would be direct measurement of watershed soil P accumulation, which could be done using sampling programs based in spatial statistics (Hope *et al.* 2005). Direct measurement of soil P storage would allow watershed managers to compute changes in watershed P retention independent of input-output calculations.

Temporal and spatial scales for drivers of change

Drivers of change in P balances of lake watersheds, and hence clarity, operate at various spatial and temporal scales (Fig. 7). At the smallest scale, individual households set personal, informal policies. Homeowners decide how to maintain their septic systems, whether to fertilize their lawns to the lakes' edge, and whether to grow shoreline vegetation buffers. At the next scale, watershed organizations and counties often conduct septic surveys and manage conservation programs. These programs can often reduce P inputs to lakes within a few years.

In Minnesota, minimum shoreline standards are set by the state but are usually enforced at the county level. The effects of implementation may take decades as a lakeshore becomes developed or redeveloped. Minnesota's detergent P ban, enacted in 1977, immediately reduced the amount of P in sewage, but the effects of the ban on lakeshores probably occurred more slowly, slowing the rate at which the soils of leach fields became saturated. The recently enacted state-wide restriction on the use of lawn P fertilizers will probably have the effect of slowly reducing P loading in lawn runoff over the period of a decade or more (Baker, unpublished data).

Watershed P fluxes are also affected by policies at the national and international level. The largest influences may be indirect effects of policies seemingly unrelated to lake management, like agricultural subsidy practices and international trade agreements. For example, the decline of small dairies in our case study watersheds is at least partly attributable to the decline in federal price supports in the 1980s.

Traditionally, lake management has been considered an activity that occurs primarily at a local (watershed) level, with exception of point source control, which is governed at the state level. Analysis of drivers of change in lake clarity show that policy and economic drivers of lake clarity occur at much broader scales and suggest that analyses of policy decisions regarding agriculture and rural development include analysis of potential impacts on lakes.

Conclusions

In this study we did not observe a statistical relationship between shoreline housing development and lake SDT. For lakes with high WSA:LA ratios and short HRTs, the potential P input from shoreline development may not be enough to have a major impact on SDT. For more sensitive lakes (low WSA:LA ratios and longer HRTs) the impact of more shoreline housing may be counterbalanced by policies that reduce the potential for P inputs to lakes. Although it is probably impossible to reconstruct historical changes in watershed P balances with a high degree of accuracy, we propose that prospective watershed P balances developed every 5–10 years could be a useful tool for guiding future watershed management plans.

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