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Long Term (1974-2001) Volunteer Monitoring of Water Clarity Trends in Michigan Lakes and Their Relation to Ecoregion and Land Use/Cover

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Abstract

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The approximately 11,000 inland lakes in Michigan are valued ecosystems yet are susceptible to degradation due to anthropogenic stresses. Few, if any long-term monitoring programs have been implemented in the inland lakes of Michigan by state agencies. However, Michigan has a lake-volunteer sampling program, the Cooperative Lakes Monitoring Program (CLMP). Our first objective was to assess statewide water quality trends from the early 1970s to present using these volunteer data. For this analysis, we used 71 inland lakes that were distributed across the state that had volunteer-collected Secchi depth (SD). Water clarity in most of these lakes has either increased or stayed the same since the 1970s. Thirty-one percent of the lakes significantly increased in water clarity, 63% had no significant trend and 6% significantly decreased in water clarity. Our second objective was to examine the relationship between lake water clarity and ecoregion and land use/cover. For this objective, we analyzed 54 lakes from the CLMP program during a time period from 1974-1983 for which we had land use data using t-tests, regressions and analysis of covariance. The mean SD was significantly lower for the southern ecoregion than the northern ecoregion, but we detected few significant relationships between land use/cover and water clarity across lakes. Volunteer monitoring programs provide an invaluable contribution to water quality information and can assist in setting priorities for statewide lake monitoring and management.

Key Words: Volunteer monitoring, Secchi, Michigan lakes, Land use/cover, Ecoregion, Water clarity.

The approximately 11,000 inland lakes in the state of Michigan are valued ecosystems yet are susceptible to lake degradation due to human-induced stresses such as point and nonpoint source pollution, exotic species invasions, water draw-downs, and shoreline erosion (NRC, 1992). The total sum of anthropogenic stressors can increase or decrease over time, and water quality may also be expected to change over time. However, most data on water quality trends in Michigan have focused on the Great Lakes. Less information is available on how Michigan's inland lakes have changed over time, or their response to anthropogenic stresses. In this study, we explore using volunteer collected data to examine trends in water clarity and to relate water clarity to ecoregion and land use/cover.

Resources at state agencies are often limited and agencies and staff cannot create long-term sampling programs for the large numbers of lakes. Programs that take advantage of citizen volunteers are relatively inexpensive and consequently can be maintained for long periods of time for a potentially large number of lakes. These volunteer programs can generate volumes of useful data that can serve a variety of purposes. For example, in 2000, the Florida Department of Environmental Protection estimated that over the past five years, of all the

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individuals and agencies, only the Department had provided more data on lakes than had Florida's LAKEWATCH program (Canfield *et al.* 2002). In addition, data from Illinois's volunteer monitoring program have resulted in a number of lakes being identified for restoration or protection activities (Sefton *et al.* 1983).

Citizen volunteer programs have been used in many states to collect lake data, and several studies have confirmed their validity and accuracy. For example, the Missouri Department of Natural Resources began a citizen lake-monitoring program in 1992, and part of the program included an evaluation of the reliability of volunteer-collected samples. During the 1992-1994 seasons, samples at 19 lakes were collected both by citizen volunteers and personnel from the University of Missouri on approximately the same dates (Obrecht *et al.* 1998). No statistical differences were found between the volunteer data and the University-collected data of total phosphorus, total nitrogen, chlorophyll and Secchi disk depth (SD) (Obrecht *et al.* 1998). In another study, the quality of the volunteer-collected data through the Watershed Watch Program was tested by staff from University of Rhode Island Cooperative Extension (Herron *et al.* 1994). The SD collected by volunteers at 21 lakes was as representative of lake water clarity as were the Extension staff's measurements (Herron *et al.* 1994). Finally, at Florida's LAKEWATCH program, SD samples taken by volunteers at 125 lakes were comparable to those taken by professionals and the mean values were strongly correlated ($r > 0.99$) (Canfield *et al.* 2002). One potential source of error with the volunteer's measurements may be change in volunteers over time. Cruikshank (1988) compared the variability over 6 weeks, between two SD volunteers, one with eight months experience and one with six years. Of the 20 measurements, there was no significant difference between the SD means for the two observers. These studies show the usefulness of citizen monitoring programs.

A citizen self-help program to monitor inland lake water clarity began in Michigan in 1974 [current name: Cooperative Lakes Monitoring Program (CLMP)]. The CLMP works with lake property owners, who measure water clarity levels using SD. In 1992, this program partnered with the Michigan Lake and Stream Associations, Inc. (ML&SA), and is now a cooperative effort that includes the Michigan Department of Environmental Quality and Michigan State University's Department of Fisheries and Wildlife. Although this program has been in operation for almost 30 years, the data have not been analyzed to assess trends in water clarity over time for all of the lakes with available data.

Monitoring water quality by measuring water clarity is a critical tool in managing a state's lake resources. Secchi disk depth is an inexpensive and fairly simple tool to use for measuring water clarity. Although SD is a rough measure of both water clarity and water quality, when used appropriately, the SD can be effectively used to monitor trends in lake water clarity through time (Heiskary and Lindbloom 1993; Terrell *et al.* 2000).

We address three main questions in this study:

1. How has the water clarity of Michigan's inland lakes changed from 1974 to present?

2. Is lake water clarity related to ecoregions? And, have lakes in different ecoregions exhibited different trends in water clarity through time?

3. Is lake water clarity related to watershed or riparian land use/cover around lakes? And, have lakes with different land use/cover exhibited different trends in water clarity through time?

To answer the above questions, we compiled CLMP data from 1974-2001 for 71 lakes that were widely distributed across Michigan's lower peninsula and across most of the ecoregions of the state. For each analysis, we analyzed a subset of the lakes based on availability of data. We examined both riparian and watershed land use/cover in this study because previous studies have shown that sometimes one measure predicts water quality better than the other, but not always (Omernik *et al.* 1981; Osborne and Wiley 1988).

For question one, we expected to see some lakes with decreasing lake water clarity across the state, especially in urbanized areas, due to increasing housing development pressures and intensity of residential land use/cover around many lakes in Michigan (Walsh *et al.* 2003). However, we also expect to see some lakes with an increase in lake water clarity due to zebra mussel invasions, better regulated shoreline development, storm water management plans, or federal water protection laws such as the Clean Water Act. For question two, we expected ecoregion to be an important variable in predicting lake water clarity. Ecoregion delineations are a way to organize the landscape into distinct units for management, comparisons, or conservation (Bailey 1983; Omernik 1987). The ecosystem classification system of ecoregions categorizes the landscape into regions based on biotic and aboitic factors, including geology, soil, climate, vegetation, and animals (Albert, 1995). Other studies, for example, Heiskary *et al.* (1987), have found significant relationships between lake water quality and ecoregions. We expect that lakes in different ecoregions will have significantly different water clarity. Finally, for question three, we expected water clarity to be correlated to watershed and riparian land use/cover. Elevated levels of nutrients and sediments running off from human-dominated land uses (such as agriculture and urban development) often lead to excess phytoplankton growth, and thus decreasing water clarity in lakes (Davies-Colley *et al.* 1993; Carpenter *et al.* 1998). However, natural land covers also have been shown to influence water clarity. Generally, both forested land and wetlands are known to have filtering properties that help remove nutrients before they enter water bodies, thus potentially increasing water clarity in lakes (Lowrance *et al.* 1984; Detenbeck *et al.* 1993; Jansson *et al.* 1994). For this question, we focus on explaining differences in lake water clarity using both human and natural land use/cover.

Methods

Data collection and lake sampling

Although there was not complete control over the volunteers' sampling procedures, the MDEQ has developed standard procedures for SD measurements and volunteers are trained by MDEQ or ML&SA staff. Volunteers collected SD readings weekly or every other week from mid-May through mid-September, although not all volunteers sampled this often. The SD measurements were taken in the deepest basin of the lake, and were recorded in feet to the nearest half-foot (0.15 meter). When lakes had multiple sampling stations, only measurements from the deepest basin were used for analysis. On the rare occasion that lakes had more than one SD reading for the week, either the first date or the date closest to the day of week that other samples were taken was used.

For these analyses, 71 lakes were selected from the over 200 lakes participating in the CLMP program. Criteria for selecting these lakes were as follows: 1) SD data must have been available for a minimum of nine years between 1974 and 2001; one of the nine years must have included the most recent year available, either 2000 or 2001. Nine years was chosen because it was the best cut-off point in the availability of the CLMP data representing the largest number of lakes. We wanted to use the largest dataset possible, but also capture as many years of monitoring as possible because more years of data are required to detect more subtle shifts in water quality of 10 to 20% (Heiskary et al, 1994). The location and characteristics of the 71 selected lakes are listed in Table 1. The selected lakes are distributed across the entire lower peninsula of Michigan (Fig. 1).

We collected land use and ecoregion data on each of the above lakes. For ecoregion delineations, we use Albert's (1995) ecoregion delineations at the section and subsection level (Fig. 1). Albert uses essentially the same criteria as Omernik's (1987) ecoregions (vegetation, physiography, climate and bedrock geology), but omits land use/cover in the delineation. In our analyses, we sought to examine the effects of land use/cover and ecoregion separately. Because Omernik's ecoregions includes land use/cover, the two analyses would be confounded.

The land use/cover information was obtained from Michigan Resource Information System (MIRIS) data (MDNR, 1999). Land use/cover information for MIRIS was obtained from aerial photos and a compilation of data from regional planning commissions. The land use/cover data were classified using level 1 classes in the Anderson Classification scheme (Anderson *et al.* 1976), which includes: urban, agriculture, non-forested vegetation (*i.e.*, grasses and shrubs), forest, water, and wetlands. Because the urban category is primarily made up of residential land use/cover near lakes, we used the term 'residential' for this land use/cover. The minimum resolution of the MIRIS land use/cover data is approximately 1 ha. We calculated land use/cover around each lake by performing two buffer analyses. The first buffer size calculated the land use/cover in the 100 m buffer zone around each lake, which is assumed to characterize the riparian land use/cover, and the second calculated 500 m around each lake, which is assume to more represent the 'watershed' land use/cover.

Data Analysis

To capture the summer stratified season, only SD from July,

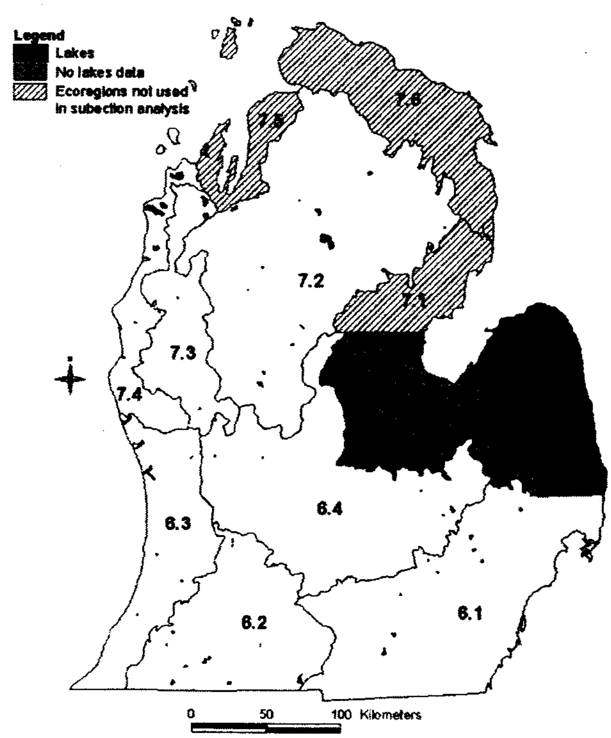

Figure 1.-The location of the 71 lakes and the ecoregion subsection delineations. No SD data were available for any lakes in the upper peninsula or for ecoregions 6.5 and 6.6. Ecoregions 7.1, 7.5 and 7.6 were dropped in the ecoregion subsection analyses due to low sample size. The ecoregion delineation is based on Albert (1995). Only lakes in Michigan's lower peninsula were used in these analyses.

Table 1.-Characteristics of the 71 CLMP lakes.

All
by surface area. Average Secchi Depth (Ave. SD) is the mean SD from 1996-2001. Ecoregion is based on Albert's (1995) ecoregion classification system. Blank values are unavailable data.

Lake Name	County	Surface Area (ha)	Max Depth (m)	Ave. Depth (m)	Ave. SD (m)	Eco-region
Arbutus	Grand Traverse	153	13.4	3.7	5.1	7.2
Avalon	Montmorency	156	22.6	10.6	7.0	7.6
Baldwin	Montcalm	25	10.7	1.8	3.4	6.4
Barlow	Barry	73	18.6		3.1	6.2
Bear	Manistee	758	6.1	3.8	2.6	7.4
Beaver	Alpena	280	23.5	8.5	3.6	7.6
Big Platte	Benzie	1025	27.4	8.2	3.7	7.4
Bills	Newaygo	81	27.4	5.7	3.1	6.4
Blue	Mason	27	18.3		8.0	7.3
Blue	Mecosta	93	15.2		4.2	7.2
Byram	Genesee	54	18.3		3.5	6.4
Camp	Kent	55	15.2		4.1	6.4
Christiana	Cass	72	12.2	6.3	2.2	6.2
Clear	Jackson	52			2.9	6.1
Clear	St. Joseph	261	9.4	3.6	3.5	6.2
Coldwater	Branch	640	28.0	5.6	2.5	6.2
Coon	Livingston	39			2.2	6.1
Corey	St. Joseph & Cass	242	24.4	7.7	3.1	6.2
Crockery	Ottawa	42	16.5	7.5	1.8	6.3
Crooked	Clare	107	22.3	4.9	3.2	7.2
Crystal	Benzie	3994	48.8	17.5	6.0	7.4
Cub	Kalkaska	23	7.0	2.9	5.9	7.2
Devils	Lenawee	531	19.2	4.3	2.7	6.1
Dewey	Cass	91	15.2		1.8	6.2
Donnell	Cass	100	19.2	7.6	2.8	6.2
Duck	Grand Traverse	787	27.4	7.3	3.3	7.3
Eagle	Allegan & Van Buren	83	17.1	6.5	4.0	6.3
Emerald	Newaygo	31	15.0		2.9	7.3
Fenton	Genesee	351	27.4	6.2	4.3	6.4
Ford	Mason	74	22.9	10.8	4.3	7.4
George	Clare	52	7.6	2.8	3.2	7.2
Glen	Leelanau	1969	39.6	21.8	5.6	7.4
Hackert	Mason	49	15.8	2.0	3.8	7.4
Harper	Lake	34	18.0	5.5	4.2	7.3
Higgins	Roscommon	4122	41.1	15.8	7.6	7.2
Horsehead	Mecosta	179	12.8		3.3	7.2
Hutchins	Allegan	154	10.4	3.2	2.5	6.3
Indiana	Cass	33	21.0		4.0	6.2
Juno	Cass	88	11.3		1.8	6.2
Klinger	St. Joseph	338	21.9	6.4	3.5	6.2
Lake of the Woods	Van Buren	122	9.1	4.5	3.1	6.2
Lakeville	Oakland	174	20.1	3.0	3.4	6.1
Leelanau- North	Leelanau	1194	36.9	12.4	3.8	7.5
Leisure	Shiawassee	94			4.2	6.4
Little Glen	Leelanau	565	4.0	1.8	2.3	7.4
Little Paw Paw	Berrien	41	9.1		1.9	6.3
Long	Branch	50	13.7		1.5	6.2
Long	Grand Traverse	1178	24.4	7.9	7.0	7.3
Long	Iosco	197	18.9	5.0	3.1	7.1
Margrethe	Crawford	778	19.8	4.7	3.9	7.2
Mecosta	Mecosta	126	11.3	3.2	3.7	7.2
Painter	Cass	42	8.2		1.6	6.2
Payne	Barry	46	13.1	4.9	2.9	6.2
Pleasant	St. Joseph	104	16.2	5.9	3.9	6.2

Table 1. (Continued)-Characteristics of the 71 CLMP lakes.

Lake Name	County	Surface Area (ha)	Max Depth (m)	Ave. Depth (m)	Ave. SD (m)	Eco-region
Round	Mecosta	64	13.7	4.8	3.41	7.2
Sapphire	Missaukee	100	2.4	1.2	2.1	7.2
School Section	Mecosta	49	10.1	3.1	4.1	7.2
Sherwood	Oakland	99	6.1		2.1	6.1
Shingle	Clare	107	22.9		3.6	7.2
Spider	Grand Traverse	180	9.8	2.8	4.6	7.2
Stone Ledge	Wexford	34	6.1		2.9	7.2
Sylvan	Newaygo	41	19.2		2.5	7.3
Taylor	Oakland	15	18.3		5.0	6.1
Twin Lakes- North	Cass	26	16.5	5.2	3.8	6.2
Van Etten	Iosco	570	10.1	4.6	1.2	7.1
Vaughn	Alcona	45	19.8	6.5	3.3	7.2
Vineyard	Jackson	219	12.8	4.2	2.9	6.1
Walled	Oakland	261			4.0	6.1
West Twin	Montmorency	528	9.1	2.2	3.4	7.2
White	Oakland	210	9.8	3.3	5.0	6.1
Zukey	Livingston	60	10.7		2.3	6.1

August, and September were used in all analyses. Heiskary *et al.* (1987) and Stadelmann *et al.* (2001) found mid-July to mid-September was the best time to measure SD because lakes behave similarly and in-lake variability is minimized. Kloiber *et al.* (2000) also found that SD transparency variability is relatively small during late summer (July 15-September 15) and varied only about 20% from the mean.

Because the objective of our analyses was to examine annual changes in water clarity, we averaged all summer SD to calculate one SD value per summer. We included data for all years where there was a minimum of three samples over the three month summer period, with at least one sample per month, but allowing one missing month of sampling. Stadelmann *et al.* (2001) found that two measurements during the summer period could estimate the summer SD mean clarity with a relative error of 30%. Sixty-eight of 1,183 (5.8%) lake years of CLMP data had a skipped month of sampling, and the majority missed September. Only three lake years had the minimum of three samples and the average number of summer samples for all lakes, per lake year was 11. Prior to analysis, the data were converted from feet to meters, then an annual mean SD was determined by averaging the summer data points for each lake.

The data were normally distributed, and there was no seasonality because of the annual averaging. We used linear regressions and t-tests to analyze the data. Other studies have used regressions to examine water quality trends in annual data such as these (Byron and Goldman 1989; Francis *et al.* 1994; Schindler *et al.* 1996). For all analyses, a P-value of 0.1 or less was considered significant. The objective of these analyses was to examine general trends; therefore the 0.1 level was used to increase our ability to detect real trends (*i.e.*, reduce the chance of Type II error) by accepting a higher chance of finding a trend that was spurious (*i.e.*, Type I error). Even if the stricter 0.05 level was used, our basic conclusions remain the same since the majority of the results are still significant at the 0.05 level.

To examine state-wide trends, we calculated a state-wide SD average of 31 lakes in approximately five year intervals (1974-1980, 1981-1985, 1986-1990, 1991-1995, 1996-2001). To avoid biasing the trend with the larger number of lakes sampled in more recent years, only the lakes (n=31) with SD from each time interval were used. Only two lakes consistently had data every year, and only nine started sampling in 1974. Therefore the approximate 5-year intervals allowed us to capture more of the lakes in a state-wide analysis. For each lake, the average of the annual means for each time period was calculated. Then the average across all lakes for each time period was calculated. We then regressed the means against time. Similarly, to quantify the presence of water clarity time trends for the 71 individual lakes, we regressed the annual SD means against time for each of the lakes individually. We chose to use simple regression to examine trends in our datasets. For datasets with relatively small sample sizes and relatively normal distributions, parametric statistics (eg. regression) are preferred over distribution-free methods such as Kendall's tau (Reckhow *et al.* 1993).

The present-day state-wide SD average was calculated from the 1996-2001 SD means from all 71 lakes. For the ecoregion analysis, we calculated SD means for the state's ecoregion sections and subsections from the same time span as the land use analyses (see below). For the subsection analysis, ecoregions 7.1, 7.5, and 7.6 were dropped due to inadequate sample size and we ran an analysis of variance on the subsection data, using Fisher's Least-Significant-Difference test. For the ecoregion section analysis, we used a t-test.

To examine the effects of land use/cover, we only examined annual SD means within a ten-year span of the land use/cover data (1974-1983). For each lake that had SD during this time period, we calculated the average of the annual SD means. We plotted these means against the percent land use/cover within both a 100 and 500 m buffer around each lake. The land use/cover categories selected for analyses were: residential, agricultural, residential combined with agricultural, forested, wetlands, and forested combined with wetlands. Finally, using the results from the individual lake trend analysis, we analyzed lakes with increasing or decreasing clarity trends by land use/cover to determine if trends varied by ecoregions.

To examine the effects of land use and ecoregion together to see if they interacted in any way, we used analysis of covariance (ANCOVA) with ecoregion section as the independent factor and land use type percentage as the covariate. An ANCOVA adjusts or removes the variability in the dependent variable (SD) due to the covariate (land use). We assumed homogeneity of slopes at a $P > 0.1$ for the factor × covariance interaction terms.

Results

The present-day state-wide trophic status calculated from the 1996-2001 SD means from all 71 lakes shows the majority of the lakes (52%) are mesotrophic, 28% are oligotrophic, and 20% are eutrophic (as defined by Forsberg and Ryding 1980) (Table 1). For the individual lake trend analysis, we found 26 lakes with significant ($P = 0.1$) trends in water clarity (Table 2). Of the individual lake trends, 22 (31%) are increasing in clarity, 4 (6%) are decreasing in clarity, and 45 (63%) have no trend. For the state-wide SD trend, the 31 lakes showed

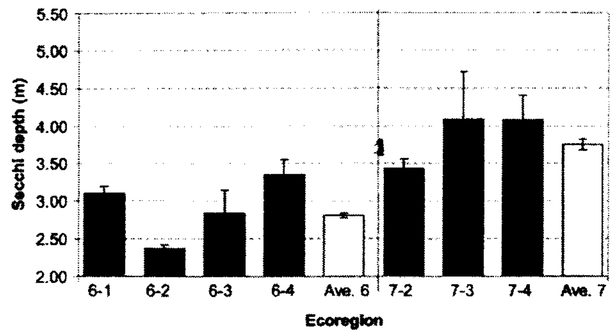

Figure 3.-The 1974-1983 average SD for the ecoregion sections and subsections. Ecoregions 7.1, 7.5 and 7.6 were dropped in the ecoregion subsection analyses due to low sample size. The error bars are standard errors.

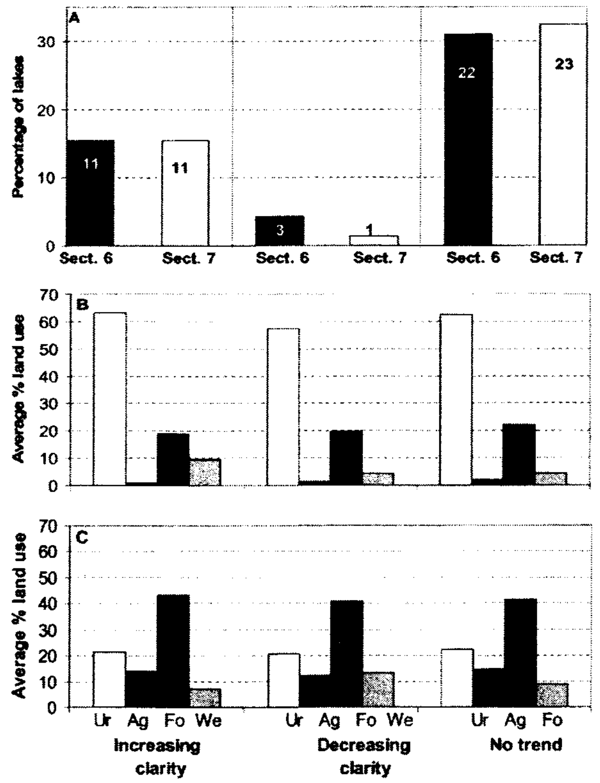

Figure 4.-(A) Lakes with significant ($P = 0.1$) increasing, decreasing, or no significant clarity trend by ecoregion section. The number indicates the number of lakes in each category. (B) Lakes with significant ($P = 0.1$) increasing, decreasing, or no significant clarity trend by average percent land use/cover in the 100 m buffer. Only lakes with SD from 1974-1983 were used. (C) Lakes with significant ($P = 0.1$) increasing, decreasing, or no significant clarity trend by average percent land use/cover in the 500 m buffer. Only lakes with SD from 1974-1983 were used.

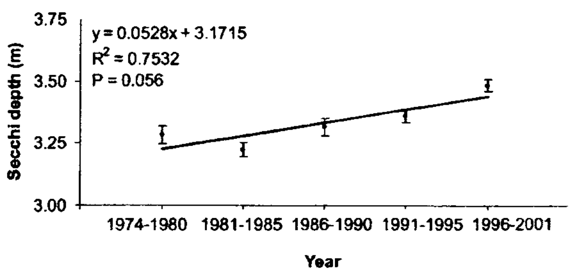

Figure 2.-The state-wide SD clarity trend for the 31 lakes that were sampled in each time period. The error bars are standard errors. 17 of the 31 lakes are from ecoregion 6 and 14 of the lakes are from ecoregion 7.

Table 2.-Lakes with significant time trends in SD.
The lakes with significant (P = 0.1) time trends, the direction of the trend, increasing through time (+) or decreasing through time (-). The P-value and R^2 from the regression of SD versus time are also presented.

Lake Name	County	Yrs. of Data	Yrs. of Data Span	Slope	P-value	R^2
Bear	Manistee	24	77-01	- 0.0221	0.044	0.172
Coon	Livingston	18	74-01	- 0.0128	0.094	0.165
Long	Branch	25	77-01	- 0.0149	0.091	0.119
Sherwood	Oakland	21	80-00	- 0.0818	0.001	0.440
Baldwin	Montcalm	23	77-01	0.0531	< 0.001	0.613
Big Platte	Benzie	23	77-01	0.0808	0.001	0.415
Blue	Mason	14	88-01	0.338	0.006	0.480
Blue	Mecosta	19	81-01	0.0647	< 0.001	0.543
Cub	Kalkaska	9	93-01	0.1716	0.015	0.592
Dewey	Cass	25	74-00	0.0289	0.022	0.207
Horsehead	Mecosta	21	81-01	0.0539	0.002	0.419
Klinger	St. Joseph	17	82-01	0.0945	< 0.001	0.729
Lake Leelanau- North	Leelanau	25	77-01	0.0559	0.001	0.396
Lake of the Woods	Van Buren	21	81-01	0.0366	0.087	0.146
Lakeville	Oakland	19	76-01	0.0237	0.091	0.159
Little Paw Paw	Berrien	10	92-01	0.0559	0.026	0.482
Long	Grand Traverse	15	79-01	0.0332	0.104	0.190
Long	Iosco	28	74-01	0.0275	0.003	0.296
Mecosta	Mecosta	18	81-01	0.0378	0.032	0.257
Payne	Barry	11	90-00	0.0564	0.040	0.390
Pleasant	St. Joseph	22	78-01	0.0999	< 0.001	0.823
School Section	Mecosta	11	90-00	0.1771	0.001	0.729
Twin Lakes- North	Cass	10	92-01	0.1265	0.024	0.491
Vaughn	Alcona	10	75-01	0.0607	0.034	0.447
Vineyard	Jackson	21	77-01	0.0265	0.039	0.206
Zukey	Livingston	13	80-01	0.0315	0.017	0.416

a significant (P = 0.056) increasing clarity trend (Fig. 2). Although 31 is a fraction of the total lakes in Michigan, and these lakes were not randomly selected, these lakes appear to be representative of the state, because they were fairly evenly distributed between ecoregion section six and seven, 55% and 45%, respectively, and there was at least one lake in each subsection.

The 1974-1983 SD mean for ecoregion section six (southern Michigan) was 2.8 m, and for ecoregion section seven (northern Michigan), 3.7 m (Fig. 3). The t-test showed a significant difference between these means (P = 0.007). For the ecoregion subsection means, Fisher's Least-Significant-Difference test showed a significant difference (P < 0.1) between the means of 6.2 and each of 7.2, 7.3, and 7.4 (Fig. 3). Thus, most of the differences between ecoregions are captured at the section level.

Using results from the individual lake trend analysis, lakes with significant (P = 0.1) increasing, decreasing, or no clarity trends were grouped by ecoregion section six or seven (Fig. 4a). For both ecoregions, the majority of the lakes had no trend, but for the significant trends, both ecoregions had more lakes with increasing water clarity than decreasing (11 versus 3 for ecoregion 6 and 11 versus 1 for ecoregion 7). To examine whether land use/cover could explain any patterns in the water clarity trends, we examined land use around lakes in each of the above categories (increasing trend, decreasing trend or no trend). We found no pattern with land use and the trends in water clarity across these lakes (Fig. 4 b,c).

We also examined land use/cover to see if it could explain other patterns in the SD data by examining differences in land use around lakes by ecoregion. Land use/cover around the lakes was significantly (P = 0.1) different between ecoregion section six and seven for agricultural and forested land use/cover (Table 3). Agricultural land use/cover was higher in ecoregion six than seven, and forested land use/cover was higher in ecoregion seven than six.

To see if land use/cover explained patterns in SD across lakes, we regressed land use/cover for each lake against its mean SD (Fig. 5 and 6). We found only two significant relationships. Residential land use/cover in the 100 m buffer showed a positive trend with SD (P = 0.07) (Fig. 5a) and wetlands in the 500 m buffer showed a negative trend (P = 0.004) (Fig. 6). Residential land use/cover in the 500 m buffer, the combined residential and agricultural land use/cover for both

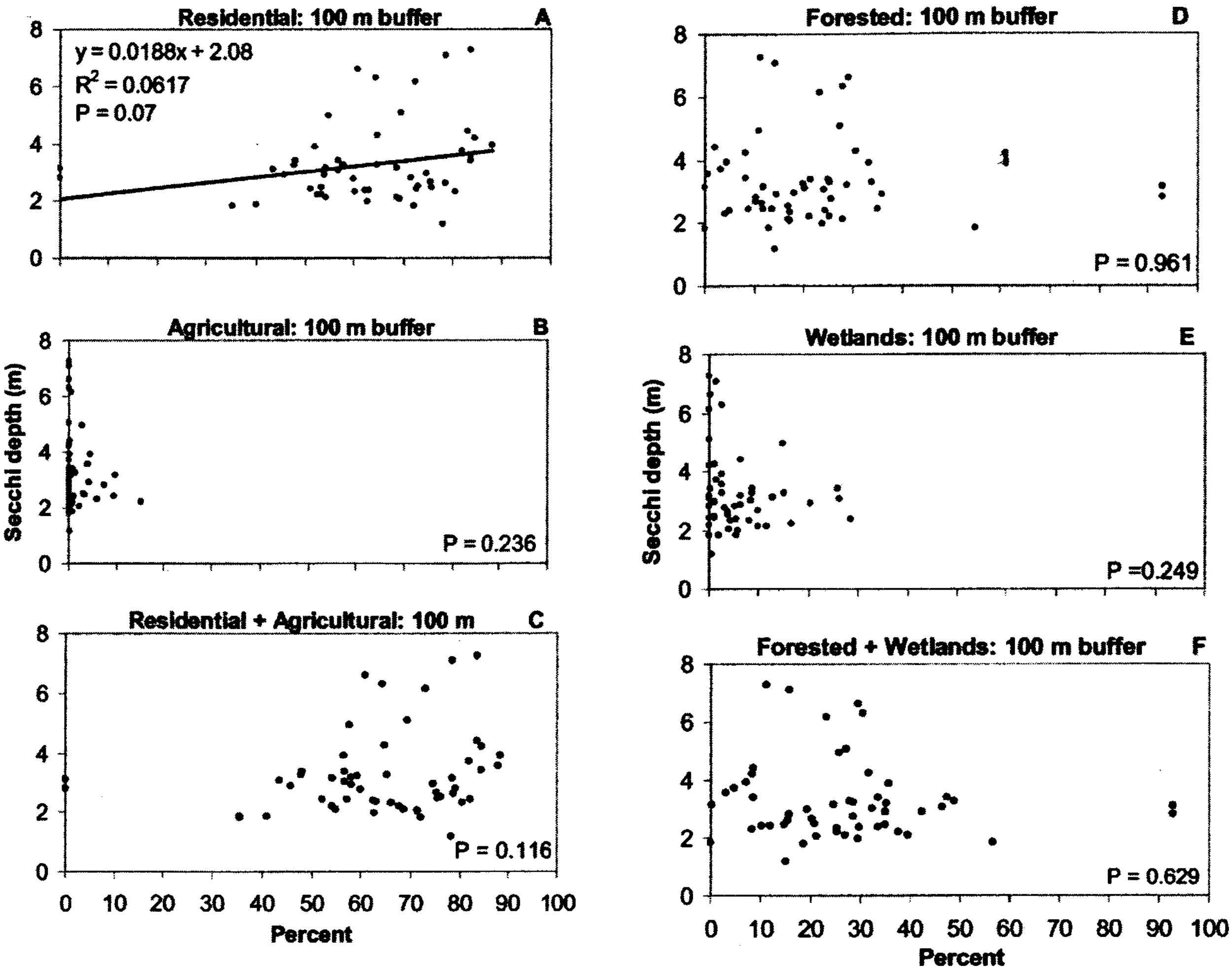

Figure 5.-Percent land use/cover in the 100 m buffer (riparian land use) versus SD for lakes with SD from 1974-1983. Each dot presents the annual SD average for each lake across the time period, versus (A) residential land use/cover; (B) agricultural land use/cover; (C) residential + agricultural land use/cover; (D) forested land use/cover cover; (E) wetlands; and (F) forested + wetlands.

Table 3.-Percent land use/cover in the 100 and 500 m buffers around lakes in each ecoregion section.
Compares the average land use/cover for ecoregion Section 6 (southern Michigan) versus section 7 (northern Michigan), for both buffer distances (100 m and 500 m), using a t-test. The P-value if for a t-test comparing the average percentages of land use/cover type between the two ecoregions. Significant differences (P = 0.1) are highlighted in bold.

Buffer Size (m)	Land use/cover Type	Ecoregion 6 (%)	Ecoregion 7 (%)	P-value
100	Residential	65	59	0.282
100	**Agricultural**	**3**	**< 1**	**0.003**
100	Residential + Agricultural	68	60	0.116
100	**Forested**	**16**	**26**	**0.055**
100	Wetlands	5	7	0.244
500	Residential	21	23	0.601
500	**Agricultural**	**19**	**9**	**0.023**
500	Residential + Agricultural	40	32	0.140
500	**Forested**	**37**	**47**	**0.091**
500	Wetlands	10	7	0.216

buffer sizes, and forested land use/cover in the 500 m buffer showed positive trend with SD, but none were significant. Agricultural land use/cover in both the 100 and 500 m buffers, forested in 100 m buffer, wetlands in the 100 m buffer, and the combined forested and wetlands land use/cover for both buffer sizes all showed negative trend with SD, but none were significant.

In an effort to separate the effects of ecoregion from land use/cover on lake water clarity, we first regressed SD means against the percent land use/cover, for both the 100 and 500 m buffers in each of the ecoregion sections. Of the significant trends (P = 0.1), three supported the previous positive relationship of SD with residential land use/cover: ecoregion six residential land use/cover in the 100 m buffer, ecoregion seven residential land use/cover in 500 m buffer, and ecoregion six residential + agricultural land use/cover in the 100 m buffer (Table 4). The most significant relationships (P = 0.03) were negative correlations with SD and wetlands in ecoregion seven's 100 and 500 m buffer (Table 4). We then conducted an ANCOVA analysis to examine the interaction between land use and ecoregion. For most cases, there was not a significant interaction term between land use and ecoregion, which means that the slopes of the regressions of land use versus SD are not significantly different in the different ecoregions (Table 5). The exceptions to this case were for wetland land cover in the 100 m buffer, agriculture in the 500 m buffer and forest in the 500 m buffer where the interaction terms were significant (P=0.029, P=0.056 and P=0.024 respectively). Ecoregion and % wetlands in the 100 m buffer are confounded so that the relationship between SD and % wetlands was different in each ecoregion (*i.e.,* the regression lines have different slopes). In general, there

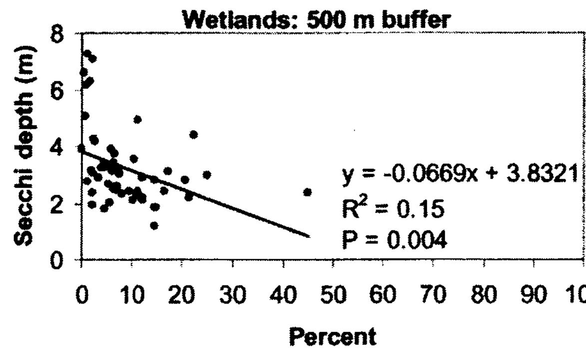

Figure 6.-Percent wetland land use/cover in the 500 m buffer (watershed) versus SD for lakes with SD from 1974-1983. Each dot presents the annual SD average for each lake across the time period.

Table 4.-Land use/cover versus SD by ecoregion.
Percent land use/cover in the 100 and 500 m buffer versus SD for lakes with SD from 1974-1983, separated into ecoregion six, versus ecoregion seven lakes. Lakes with SD from 1974-83 were chosen to match the land use/cover data (1978). Significant (P = 0.1) relationships are highlighted in bold.

Buffer Size (m)	Ecoregion	Land use/cover	Slope	P-value	R^2
100	**6**	**Residential**	**0.0196**	**0.07**	**0.113**
100	6	Agricultural	- 0.007	0.88	0.001
100	**6**	**Residential + Agricultural**	**0.018**	**0.09**	**0.102**
100	6	Forested	- 0.011	0.39	0.028
100	6	Wetlands	0.0194	0.47	0.020
100	7	Residential	0.0241	0.14	0.095
100	7	Agricultural	- 0.2096	0.60	0.210
100	7	Residential + Agricultural	0.238	0.14	0.092
100	7	Forested	- 0.0075	0.64	0.010
100	**7**	**Wetlands**	**- 0.0896**	**0.03**	**0.183**
500	6	Residential	0.0075	0.41	0.025
500	6	Agricultural	0.0071	0.48	0.019
500	6	Residential + Agricultural	0.0091	0.22	0.055
500	**6**	**Forested**	**- 0.0124**	**0.07**	**0.120**
500	6	Wetlands	- 0.0185	0.47	0.020
500	**7**	**Residential**	**0.0495**	**0.10**	**0.111**
500	7	Agricultural	- 0.0399	0.12	0.104
500	7	Residential + Agricultural	- 0.0051	0.84	0.002
500	7	Forested	0.0294	0.14	0.091
500	**7**	**Wetlands**	**- 0.0813**	**0.03**	**0.201**

Table 5.-Results from the ANCOVA analyses for land use and SD relationships.
Significant effects are highlighted in grey boxes. A non-significant interaction term means that the slopes of the regression lines between SD and each land use type for each ecoregion are not significantly different (*i.e.*, the lines are parallel). Whereas a significant ecoregion effect means that the mean SD's are different between the two ecoregion sections. For situations where the interaction term was not significant, the P values presented are for models with no interaction term in the final model.

Land use percent	Buffer width	Ecoregion effect (P)	Land use effect (P)	Interaction (P)
Urban	100 m	0.003	0.022	No
Agriculture	100 m	0.022	0.805	No
Urban + Agriculture	100 m	0.003	0.026	No
Forest	100 m	0.007	0.409	No
Wetland	100 m	<0.001	0.155	0.029
Forest + Wetland	100 m	0.003	0.132	No
Urban	500 m	0.120	0.130	No
Agriculture	500 m	0.004	0.178	0.056
Urban + Agriculture	500 m	0.009	0.563	No
Forest	500 m	0.302	0.348	0.024
Wetland	500 m	0.026	0.010	No
Forest + Wetland	500 m	0.007	0.371	No

Note: The ecoregion effect indicates whether the effect of ecoregion is significant (P < 0.1). A land use effect indicates whether the effect of land use is significant at P < 0.1 (*i.e.*, the slope of the relationship between land use and SD differed from zero). The interaction term indicates whether the slope differed by ecoregion at P< 0.1.

were significant differences in SD between the ecoregions, however effects of land use on SD are not as strongly supported by these data.

Discussion

The majority of lakes in our dataset showed water clarity trends that have either increased or stayed the same since the 1970s. In addition, the statistically significant difference between ecoregion sections suggests that ecoregions may useful to guide management efforts, because they indicate regional differences in lake water quality. However, we were not able to detect a strong effect of land use/cover on water clarity in lakes across the state. We explore each of these major findings in more detail below.

Changes in the water clarity of Michigan's inland lakes from 1974 to present

It is apparent from the individual lake trends, as well as the state-wide analysis that in general, the clarity of many of Michigan's lakes in the lower peninsula has been either increasing or staying the same since 1974. The 31 lakes in the state-wide analysis, although a fraction of the total lakes in Michigan, were well distributed across the state and varied widely in land use/cover. We argue that our dataset is likely to be representative of many lakes in Michigan and possibly the U.S. upper Midwest region. Trend analyses in Minnesota have shown similar patterns in water clarity. Heiskary and Lindbloom (1993) studied volunteer-collected SD from 152 lakes with 8 or more years of data, ranging from the early 1970s to 1992. Twenty nine percent of lakes showed significant increase in clarity, 8% significant decrease, and 63% had no trend. In Florida, Terrell *et al.* (2000) analyzed volunteer and agency-collected SD from 127 lakes over 30 years and found no significant change in clarity. However, their analysis excluded 13 lakes with known management changes such as point source removal or artificial fertilization. The reasons for increasing water clarity in Michigan lakes may be many, including improved management practices around lakes to control polluted runoff, effects of laws and regulations such as the Clean Water Act, removal of phosphorus from soap products, changes from septic to sewer systems in residences surrounding lakes, improved urban storm water management, changes in fish or plant communities, or long-term rainfall patterns/ water level changes. However, it is beyond the scope of this paper to investigate all these possibilities.

One explanation for the increasing water clarity that we could explore with our data was the influence of zebra mussels (*Dreissena polymorpha*). We used the Sea Grant (2001) database of zebra mussel monitoring to determine if and when any zebra mussels were detected in our lake database. Because zebra mussels have been shown to filter water (Reeders *et al.* 1989), and thereby increase water clarity (Budd *et al.* 2001; Schloesser and Muth 1993), we examined whether zebra mussel presence in the CLMP lakes was related to changes in water clarity. Of the lakes with significant increases in clarity, 42% contained zebra mussels. Zebra mussels were also present in 4% of the lakes with significant decreases in clarity. The 1996-2001 mean SD for all lakes with zebra mussels was 4.0 meters, and for those without, 3.4 meters (Fig. 7). A t-test showed this difference was significant (P = 0.091). To determine whether there was a difference between these two groups of lakes before zebra mussel invasion, the average SD of the two groups of lakes were also compared for the time period 1974-1990 (prior to any zebra mussel invasion in the lakes). The mean SD for all lakes with zebra mussels was 3.7 meters, and for those without, 3.1 meters (Fig. 7). A t-test showed this difference was marginally significant (P = 0.11). These results suggest that the lakes

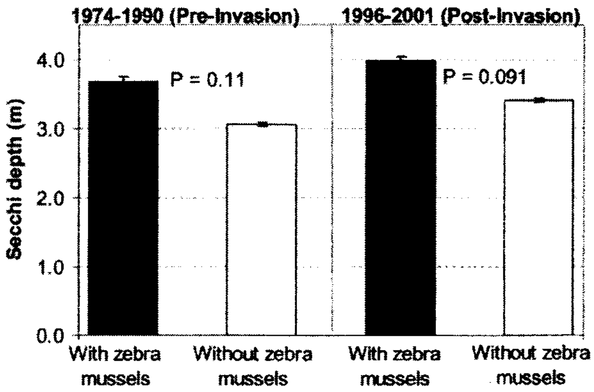

Figure 7.-SD for lakes with and without zebra mussels in 1996-2001 and 1974-1990. Lakes were classified using the Sea Grant (2001) database of zebra mussel monitoring. The error bars are standard errors and a t-test was used to compare zebra mussels versus non-zebra mussels lakes in each time period.

that have been invaded by zebra mussels may have been clearer to begin with, at least for our dataset. Additionally, if we examine water clarity in the zebra mussel lakes before and after invasion, there is no significant difference. Thus, it appears that zebra mussel invasions alone cannot explain our state-wide patterns of increasing water clarity.

The effect of ecoregion on water clarity

The significant differences in SD between ecoregion sections and subsections (Fig. 3) suggest that management strategies could consider taking into account a lake's ecoregion in the process of setting water quality goals or standards, as they illustrate regional differences in some measures of lake water quality. The SD difference is approximately one meter, with section six bordering on eutrophic and section seven at the top end of mesotrophic (as defined by Forsberg and Ryding, 1980). This result may be due to the physical properties inherent to the ecoregions. For example, the southern ecoregion is primarily composed of silt and clay loams, while the northern ecoregion is dominated by sands (Albert, 1995). Silty soils have greater erosion potential than sandy soils, and lakes could be more susceptible to sedimentation-induced clarity reductions in the southern ecoregion. In addition, differences in land use/cover between the ecoregions may help explain differences in water quality (see below). Other studies have found similar results. For example, Heiskary *et al.* (1987) found great differences in median epilimnetic total phosphorus concentrations in Minnesota's lakes when categorized by four ecoregions. Additionally, natural resource managers in Minnesota have created a model in which ecoregions are used to predict runoff, precipitation, evaporation, stream phosphorus concentration and atmospheric phosphorus deposition (Wilson and Walker, 1989).

Relationships between ecoregion and land use and the effect on water clarity

The ANCOVA analysis on land use and ecoregion shows that there are only a few differences in the relationship between land use and water clarity between ecoregions. This result means that we can examine the relationship between land use and SD across ecoregions. The exception to this conclusion is for wetlands (100 m buffer), agriculture (500 m buffer) and forests (500 m buffer). Different ecoregions have different relationships of these three variables to SD in lakes. These results are important because it means that when developing models linking land use to water clarity (and possibly other measures of water quality), we may have to first factor in ecoregion, but only for some variables.

Perhaps our most surprising result was that residential land use/cover in the 100 m buffer showed a significant positive relationship with SD. Although this result is surprising, several possible explanations may explain it. First, it may be that residential land use/cover primarily affects shoreline water areas during the summer months when lake mixing is reduced. In a study of Higgins Lake, Michigan, Minnerick (2001) found that rapid lakeshore residential development of up to 246% between 1970-1990 had degraded water quality in the shallow shoreline areas, but had not yet affected the whole lake, or the deeper basins. Second, lakes with higher residential development might be more likely to have connections to municipal sewage systems, whereas lakes with lower residential development would have septic tanks, which have been shown to negatively impact lake water quality (Hayes *et al.* 1990). Third, clearer lakes may be favored for housing developments and residences. Interestingly, residential land use/cover is very similar around lakes regardless of ecoregion, suggesting development is occurring around lakes throughout the state of Michigan. As was shown by the ecoregion and land use/cover analysis, the higher density of residential use is within the 100 m buffer around the lake (Table 3), compared to the watershed as a whole, which is supported by other studies of residential development (Walsh *et al.* 2003; Schnaiberg *et al.* 2002). High-quality water is important to people, and degraded water bodies can affect property values. Several studies have illustrated the negative economic consequences of cultural eutrophication of lakes and other water bodies. For example, a study in Maine showed the detrimental impact of poor water quality on lakeside property values, as market prices dropped 10-20% with a one-meter reduction in clarity (Bouchard, 1995). Water quality had a significant influence on home values along the shores of Chesapeake Bay (Leggett and Bockstael, 2000), and home values along Lake Champaign, Vermont were lower compared to homes by a less polluted lake (Young, 1984). The trends in our data support the above research linking water quality and lakeshore development.

Agricultural land use/cover is greater in southern Michigan and forested land use/cover is greater in northern Michigan (Table 3). This pattern may explain the greater mean clarity of lakes in northern Michigan, because agricultural land use/cover is known to be an origin of nonpoint source pollution to water bodies (Sharpley *et al.* 1994; Carpenter *et al.* 1998). However, the relationships between SD and agricultural land use/cover or forest land use/cover across all lakes in the dataset for both buffer distances were not significant (Figure 5). It may be that the range of our data is too narrow. For example, in the 100 m buffer, agricultural land use/cover around lakes averages 2.7% for ecoregion six and 0.3% for ecoregion seven, and in the 500 m buffer, 18.8% for ecoregion six and 9.0% for ecoregion seven (Table 3). It also may be that the category of "agriculture" is too broad and needs to be further broken down into sub-categories of fallow land, row crops, feed lots, etc. before significant relationships are detected.

We also found some significant relationships between water clarity and 'natural' land cover types. For example, we found a significant negative relationship between water clarity and wetlands in the 500 m buffer around lakes. Because wetlands can export colored humic material to lakes (Wetzel, 2001), water clarity can be affected when wetlands occur near lakes and are hydrologically connected to them. Detenbeck *et al.* (1993) found that the color of lakes increased (and water clarity decreased) as the extent of wetlands and seasonally flooded wetlands increased in the lake's watershed. However, surprisingly, we also found forested land use/cover in ecoregion six's 500 m buffer to be negatively related to SD (Table 4). Rather than a direct mechanistic link to forest cover, we feel that this result is due to factors that are likely correlated to the presence of forest in the watershed, although we were not able to tease apart the possible causal factors with our data.

The fact that we found few strong relationships with SD and land use/cover, and that even significant regressions explained very small amounts of variation in SD may be because of possible data limitations of our study. First, our measure of water quality (Secchi disk depth) may be too 'coarse' of a measure of water quality to detect relationships to land use/cover. Second, it may be that the 500 m buffer is too poor of an approximation of a lake's true watershed and that land use/cover in the 'true' watershed is quite different from the 500 m buffer. Similarly, the 100 m buffer could also be a poor approximation of the true riparian zone. Despite these limitations, our approach uses a common approach to measure both riparian and watershed land use/cover and the results are important because they show that at the scale measured in this study, land use/cover effects on water clarity are difficult to detect.

Conclusions

Although it appears that water quality of Michigan's lakes as measured by water clarity is good, it is also important to remember that SD is only one component of water quality and we do not know how all human impacts and other measures of water quality have changed through time. It is interesting to compare Michigan's 303(d) list of impaired lakes (not meeting one or more designated uses) to the CLMP database. Of the 71 lakes we analyzed, eight are on the list, out of a total of 102 (USEPA, 2002). The impairments of these eight lakes are metals, fish consumption advisory, and pesticides (USEPA, 2002). None of these lakes are listed for phosphorus or turbidity, the water quality parameters that are measured by SD readings. State-wide, Michigan may be reducing factors that threaten lake water clarity, however we do not know the status of the other measures of water quality.

It is important to both rigorously analyze volunteer data such as the CLMP data to understand the water quality status and changes of Michigan's lakes, and to close the feedback loop by disseminating the information to the volunteers and the general public. One possibility of distribution could be to store the data on a website where it could easily be accessed by the public. When citizens are involved in taking care of their resource, it brings greater awareness, cooperation, and buy-in to management activities.

Using volunteers is not only an opportunity to educate the public about the resource, but it is also an efficient method of monitoring large numbers of lakes. With proper training and administrative support, volunteer programs can provide comprehensive monitoring of lakes and valuable data. Volunteer monitoring programs provide people-power for agencies that cannot spend the time or money to send staff to large numbers of lakes across the state to collect data. These data not only help us determine trends through time and relationships to such factors as land use and ecoregion, but they can also help managers set baseline conditions. In addition, these data can be used to identify specific lakes for more comprehensive state monitoring in the future.

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