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Wetland effects on lake water quality in the Minneapolis/St. Paul metropolitan area

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Abstract

A method developed to evaluate the cumulative effect of wetland mosaics on water quality was applied to **33** lake watersheds in the seven-county region surrounding Minneapolis-St. Paul, Minnesota. A geographic information system (GIS) was used to record and measure landscape variables derived from aerial photos. Twenty-seven watershed land-use and land-cover variables were reduced to eight principal components which described **85%** of the variance among watersheds. Relationships between lake water quality variables and the first six principal components plus an index of lake mixis were analyzed through stepwise multiple regression analysis. **A** combination of three landscape components (wetland/watershed area, agriculture/wetlands, and forest/soils components) explained 49% of the variance in a trophic state index, even though most of the lakes examined were already highly eutrophic, and thus were influenced by internal loading. The regression equations explained a range of 14 to 76% of the variation in individual water quality variables. Forested land-use was associated with lower lake trophic state, chloride, and lead. High lake trophic state was associated with agricultural land-use and with wetland distance from the lake of interest. The extent of wetlands was associated with low total lead and high color in lakes downstream. Wet meadows or herbaceous, seasonally-flooded wetlands contributed more to lake water color than did cattail marshes.

1. Introduction

The rapid loss of wetland area in the United States by draining and filling (Tiner 1984) has made it imperative to assess not only the effect of individual wetland loss on lake water quality, but also the cumulative effect of changes in the wetland mosaic on lake water quality. In the past, permit regulation for wetland drainage and filling has been based strictly on the nondegradation clause in Section 404 of the Clean Water Act, and permits have been considered on a case-by-case basis. However, environmental impact statements prepared under mandate of the National Environmental Policy Act of 1969 must also address the issue of cumulative impacts (Council of Environmental Quality; **38** CFR 1500.6). Cumulative impacts have been defined as 'the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions. . .' (40 CFR 1508.7).

Management of lake water-quality within a de-

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of inputs (dissolved, fine particulates, or coarse particulates; Oberts 1981; Richardson and Nichols 1985), and the composition of wetland sediments (Richardson 1985).

Not only the extent, but also the distribution of landscape sources and sinks relative to surface waters is important. In a previous study of stream watersheds in the seven county Twin Cities metropolitan area, Johnston *et al.* (1990) found that streams with wetlands in close proximity to the outlet had lower annual concentrations of inorganic suspended solids, fecal coliform counts, nitrate, flow-weighted ammonium, flow-weighted total phosphorus, and a lower proportion of phosphorus in dissolved form than those streams associated with wetlands farther upstream. Other researchers have demonstrated that land-use within the buffer zone surrounding surface waters can have a greater influence on water-quality than does land-use for the watershed as a whole (Osborne and Wiley 1988).

Multivariate analyses have been used in the past to explain differences in biological communities or water chemistry among lakes as a function of multiple lake morphometry and watershed variables (Zimmerman *et al.* 1983; Paloheimo and Zimmerman 1983). In this study, we apply an empirical multivariate approach developed to evaluate the cumulative effect of wetland mosaics on surface water quality to 33 lake watersheds in the seven-county region surrounding Minneapolis-St. Paul, Minnesota (Johnston *et al.* 1990). Twenty-seven variables were derived to describe the extent and distribution of wetland and land-use types, soil characteristics, topography, and lake morphometry. Principal components analysis was used to derive a smaller set of composite landscape variables that a) explained the variation among watersheds and b) were independent of one another. These principal components were then used as independent variables in a multiple regression analysis of lake water-quality variables.

Partial correlation analysis has also been used to partition variability in lake water quality among related watershed variables (*e.g.* Gorham *et al.* 1986). In the current study, partial correlation analysis was used to explore relationships between water quality and selected watershed variables while holding constant other watershed variables that were highly correlated with principal components used in the original multiple regressions. In this way, the importance of individual watershed variables, which may have been obscured by the dominating explanatory variables, could be examined. In general, the statistical analyses were used in an exploratory manner to identify possible casual relationships between landscape and water quality variables.

1.1. Study region

The low relief (180 m) and interrupted drainage of the Minneapolis/St. Paul metropolitan area have made it an area of numerous lakes and wetlands. There are 942 lakes ranging in area from 4 to 5,791 ha which constitute 6.7% of the total land area in the 7,330 km^2 metro region (McBride 1976). The majority of these lakes are small (< 40 ha), shallow (<5 m), and eutrophic (Metropolitan Council 1981). Although only about half of the pre-settlement wetland area remains (Anderson and Craig 1984), wetlands still constitute about 7.6% of the region (Owens and Meyer 1978).

2. Methods

2.1. Study site selection

Thirty three lake watersheds covering 589 km^2 were selected as study sites (Table 1, Fig. 1). Lakes were defined as areas of open water greater than 8 ha in size and $\geq$ 2 meters deep (Cowardin *et al.* 1979). We restricted our study sites to those lakes with total watersheds greater than 300 ha in size.

The primary objective was to relate lake water quality to wetlands and other watershed characteristics, therefore study sites were selected based on the availability of: (1) epilimnetic water quality data, and (2) aerial photographs of the monitored watershed taken concurrent with water quality data collection. We selected dimictic lakes as much as possible, using mixing ratio (mean depth/square

Table 1. Watershed size and lake morphometry for 33 lakes in the Twin Cities metropolitan area, MN (from McBride 1976).

lake	lake no.	code	year of aerial photo	lake surface area	contributing watershed area[a]	dilution ratio lake vol/ contributing area (m)	mean depth (m)	mixing ratio mean depth/ √surface area
Bald Eagle	1	BA	1987	4.5	29	0.36	2.4	1.2
Bass	2	BS	1980	0.7	12	0.17	2.7	3.3
Bryant	3, 4	BR	1966, 1984	0.8	11	0.36	5.2	5.8
Chub	5	CH	1980	1.0	5	0.27	1.2	1.2
Coon	6	CO	1980	6.3	19	0.97	3.0	1.2
Crystal	7, 8	CR	1966, 1980	0.4	5	0.63	2.7	2.5
Cynthia	9	CY	1980	0.9	41	0.04	2.1	2.4
Diamond	10	DI	1980	1.7	2	1.43	1.5	1.2
Dutch	11	DU	1980	0.7	6	0.48	4.4	5.3
Eagle	12	EA	1980	1.4	7	0.93	3.6	2.6
Fish	13	FI	1980	0.8	7	0.69	5.4	5.7
Forest	14, 15	FO	1957, 1980	0.3	3	3.53	2.7	1.3
George	16	GE	1980	2.0	6	1.08	2.7	1.8
Golden	17	GO	1980	0.2	28	0.02	2.4	5.4
Independence	18	IN	1980	3.3	31	0.66	6.1	3.3
Johanna	19	JO	1980	0.8	8	0.62	6.1	6.6
Lotus	20	LO	1980	0.9	5	1.45	6.4	6.3
Marion	21	MA	1980	2.2	19	0.48	4.6	3.3
Medicine	22	ME	1980	4.4	42	0.46	5.2	2.7
Minnewashta	23	MI	1980	2.9	9	1.65	4.6	2.6
Orchard	24	OR	1980	1.0	5	0.45	2.4	2.5
Otter	25	OT	1980	1.3	3	0.83	1.8	1.6
Owasso	26	OW	1980	1.8	12	0.33	2.7	2.3
Parley	27	PA	1984	1.2	48	0.07	1.8	1.3
Pierson	28, 29	PI	1957, 1980	1.2	4	2.38	6.1	5.2
Riley	30, 31	RI	1980, 1984	1.2	19	0.48	7.6	7.0
Sarah	32	SA	1980	2.3	15	0.94	6.1	4.0
Schutz	33	SC	1984	0.4	4	0.80	5.0	6.6
Spring	34	SP	1980	3.1	41	0.33	5.6	3.4
Waconia	35	WA	1980	13.7	29	2.17	4.8	1.3
White Bear	36	WH	1980, 1987	9.9	26	2.26	6.1	2.0
Wolsfeld	37	WO	1980	0.2	6	0.07	3.0	8.0
Zumbra	38	ZU	1984	0.9	1	3.15	4.3	4.5

lake area

root of surface area) as a predictor of mixis (Osgood 1988). The water column of a dimictic lake is thoroughly mixed during spring and fall overturn periods, but remains vertically stratified during the growing season. Contributions from internal nutrient generation, which could obscure relationships between lake water quality and watershed characteristics, should be less in these lakes than in polymictic lakes, which are continually mixed. Lakes were excluded from study if there were (1) known point sources of pollution in the watershed (*e.g.* sewage treatment plants), (2) artificial water additions or withdrawals from the lake (McBride 1976), or (3) lake water-quality management programs (*e.g.* hypolimnetic aeration) in effect. Of the 33 lakes selected, six had both water quality data and aerial photographs available for two separate periods, so that a total of 39 site-years were identified.

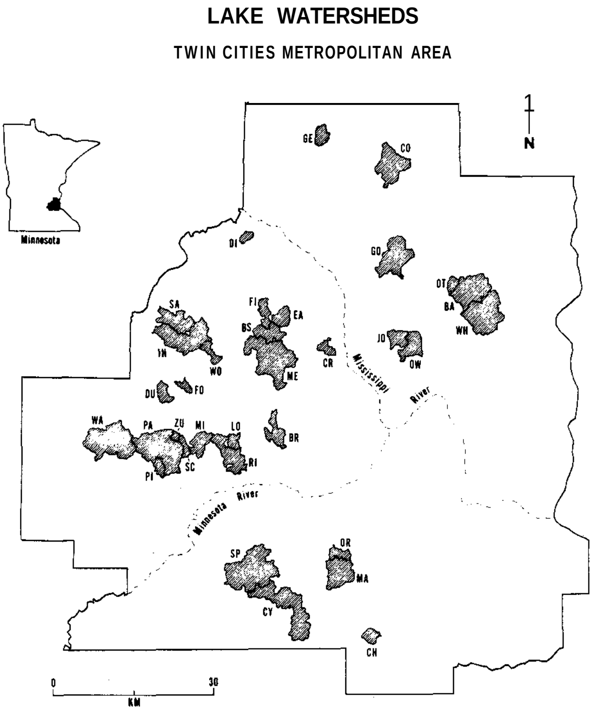

Fig. 1. Location of 33 selected lake watersheds in the Minneapolis-St. Paul metropolitan area. Lakes corresponding to two-letter codes are listed in Table 1.

2.2. Landscape analyses

Landscape analyses used to characterize individual watersheds were described in detail by Detenbeck *et al.* (1991) and Johnston *et al.* (1988; 1990). Several variables were used to characterize lake and drainage basin morphometry (Tables 1 and 2). Drainage basins were delineated and measured for each of the lakes using 7½ minute U.S.G.S. topographic maps. **A** dilution ratio was calculated for each lake by dividing lake volume by contributing watershed area (Schindler 1971). Lake volume, depth, and surface area were obtained from a U.S.G.S. database maintained for metropolitan area lakes (McBride 1976).

Existing aerial photographs were used to map the lake drainage basins for each site-year (Table **2).** Because water quality and runoff is potentially affected by all land within a watershed, both upland and wetland cover types were mapped for each date of photography. Photo enlargements were used whenever possible. Stereoscopic magnification (3x) was required to interpret the 1957 air photos because they were only available as 9" x 9" contact prints.

Mapping of the five drainage basins larger than 4,000 ha (Cynthia, Medicine, Parley, Spring, and Waconia) was done by U.S. Public Land Survey quarter-quarter sections (~40 acres each). The quarter-quarter section boundaries correspond with land use boundaries (*e.g.*, roads, fields, fence lines, woodlot boundaries), so they are easily delineated on aerial photos. **A** clear mylar overlay gridded into 40 acre (16 ha) cells prepared for each air photo scale was used as a guide for locating the actual quarter-quarter boundaries on the photos. In this way, the ground location of each area mapped was exactly the same from year to year, despite air photo scale differences. The forty acre resolution was too coarse for small watersheds, so a 10 acre (4 ha) grid size was used to map the remaining 28 watersheds smaller than 4,000 ha (34 site-years).

Each cell was classified by the land use which constituted the majority of its area: agriculture, forest, urban/residential, lake, or wetland. If the major cover type was wetland, it was further classified into one of seven wetland categories based on U.S. Fish and Wildlife Service criteria (Cowardin *et al.* 1979): open water permanent, open water semi-permanent, herbaceous semi-permanent, herbaceous seasonally-flooded or saturated, herbaceous temporarily-flooded, woody semi-permanent, and woody seasonally-flooded/saturated. Within the Twin Cities metropolitan area, seasonally-flooded wetlands generally correspond to weedy or cultivated depressions flooded only briefly following periods of heavy run-off, or to weedy or scrub/shrub wetlands within the floodplain of a stream. Saturated wetlands correspond to wet meadows dominated by grasses or sedges. Semi-permanent open water wetlands correspond to shallow-deep marshes typically dominated by cattails (*Typha* spp.) and bulrush (*Scirpus* spp.). National Wetlands Inventory maps were used for 1980, and additional wetland mapping was done under the supervision of air photo interpreters having 5 or more years experience with the National Wetlands Inventory. The land-use code for each grid cell was recorded on 1:24,000 maps and then digitized.

Table 2. Average, standard error (s.e.), and range of landscape and lake morphometry variables for 33 lake watersheds in the Twin Cities metropolitan area, MN (% W = watershed, % LF = % 200 m lake fringe area, n = 38 watershed-years).

variable	code	units	average	s.e.	min	max
Watershed area	WAREA	km²	17	2.4	2	50
Contributing watershed area	CAREA	km²	15	2.1	1	48
Dilution ratio = Lake volume/Contr'g area	DILRATIO	m	1.03	0.15	0.02	3.53
Lake mixing ratio = Avg area	MIXRATIO		3.6	0.32	1.2	8.0
Wetland acreage	WTLND	% W	14	1.5	0	41
Agricultural acreage	AGR	% W	38	2.9	3	71
Forested acreage	FOR	% W	12	1.2	2	36
Urban/residential acreage	URB	% W	19	3.6	0	83
Other lakes acreage	OLAKES	W	1.2	0.35	0.0	9.4
Herbaceous semi-permanent wetlands	HERBSP	% W	3.2	0.41	0.0	7.6
Average watershed-lake elevation	ELEVDIFF	m	15.9	1.03	2.1	28.0
Average K-factor (erodibility)	KFCTR		0.29	0.010	0.18	0.38
Urban lake fringe	URBLKFR	LF	42	4.9	0	97
Wetland lake fringe	WTLDLKFR	LF	18	2.0	0	50
Herbaceous semi-permanent wetland lake fr.	HRBSPLFR	LF	8	1.5	0	32
Average wetland distance upstream	WTLDDIST	km	2.2	0.31	0.0	6.7
Average soil phosphorus	SOILP	Index (1–3)	2.69	0.066	1.50	3.00
Lake rank downstream (headwater = 0)	NDWNSTRM	No. of lakes	1.3	0.20	0.0	5.0
Average watershed slope	AVGSLP	%	0.76	0.060	0.17	2.38
Average soil pH	SOILPH	Estimated pH	5.98	0.048	5.80	6.73
Stream fringe area	STRFRG	% W	11.0	2.30	0.0	24.1
Herbaceous wetlands	HRBSSWD	% W	6.0	0.79	0.0	18.8
Woody wetlands	WDYSSWD	W	3.3	0.73	0.0	20.6
Agricultural lake fringe	AGRLKFR	% LF	26	3.3	0	86
Forested lake fringe	FORLKFR	% LF	16	2.4	0	71
Herbaceous seasonally-flooded or saturated lake fringe wetlands	HRBSSLF	LF	6.5	0.98	0.0	24.1
Woody seasonally-flooded or saturated lake fringe wetlands	WDYSSLF	% LF	2.3	0.64	0.0	16.1
Average wetland size	AVWTLDSZ	km²	0.218	0.0037	0.000	1.03
Average size of stream wetlands	STWTLDSZ	km²	0.37	0.0083	0.00	2.13

Digital data of soil K-factor (erodibility), surface soil phosphorus, surface soil pH, average slope, and mean elevation at a 40 acre (16 ha) resolution were obtained for the region from the Minnesota State Planning Agency. Values for soil K-factor and available surface soil phosphorus in these data coverages do not represent measurements made within each pixel, but have been derived by state-agency soil scientists through interpretation of soil type and geomorphic units (see Minnesota Soil Atlas 1973; 1980). Available soil phosphorus is expressed as an index value from 1 (low) to 3 (high). The maximum elevation difference (ELEVDIFF) was calculated as the difference between minimum and maximum elevation values associated with pixels within each watershed.

Three different PC-based geographic information systems were used to enter, measure, and analyze the wetland and land-use data. ARC/INFO was used for digitizing, EPPL7 (Minnesota State Planning Agency 1987) was used to read the digital soil and topographic files, and an ERDAS GIS was used for digitizing and data analysis. Files were exchanged between ARC and ERDAS using the ARC/INFO grid conversion programs, and between EPPL7 and ERDAS using a program written by Anderson and Scheer (1987).

Land use, watershed boundaries, and streams were digitized from U.S.G.S. 1:24,000 topographic maps. Land use and wetland types were measured using the ERDAS GIS, and expressed as both a percentage of the lake fringe area (a 200 m band or 4 ha pixel width surrounding each lake) and total watershed area (Table 2). The GIS also was used to extract individual watersheds from the regional data files, and to compute average soil (pH, available soil P) and topographic variables (maximum elevation difference: Table **2)** for each watershed. Streams flowing into each lake were divided into **1** km increments from the inlet to the headwaters using a map wheel. The position of wetlands in the drainage network was then determined by intersecting the stream segment file with the wetland file, and computing an average distance upstream (weighted by wetland area) for all wetlands within a watershed. The average size of individual wetlands was computed for each watershed as a whole and for stream-associated wetlands.

2.3. Water quality data

Recent water-quality data for metropolitan area lakes were retrieved from the computerized STORET database **(U.S.** Environmental Protection Agency). As part of the routine surveys of lakes managed for fish and wildlife, limited data had been collected prior to **1970;** these data were obtained from a database recently automated by the Minnesota Dept. of Natural Resources (Dave Pederson, personal communication). Most of the water quality data used in this study were collected as part of an ongoing study conducted by the Minneapolis-St. Paul Metropolitan Council (Metropolitan Council **1981; 1982; 1984).**

Water quality data included nutrients (total and dissolved phosphorus, total Kjeldahl nitrogen, ammonium, nitrate + nitrite), dissolved oxygen, chloride, hardness, total alkalinity, turbidity, pH, specific conductivity, color, total and volatile suspended solids, Secchi depth, total and phaeophytin-corrected chlorophyll ***a*,** fecal coliform, metals (total lead, zinc, cadmium) and arsenic. Water quality variables reported by the Metropolitan Council, Barr Engineering, MN Pollution Control Agency (MPCA) and the MN DNR were analyzed according to Standard Methods (APHA **1936** through **1980)** or according to EPA-approved procedures **(U.S.** EPA **1979).** Most samples were analyzed by the MN Dept. of Transportation (metals), MN Dept. of Health (MPCA samples), or by the Metropolitan Waste Control Commission (Metropolitan Council samples). Methods used by these laboratories are listed in Table **3** for comparison. Detection limits for samples analyzed by the MWCC were **0.01** mg $P \cdot L^{-1}$ for total or dissolved P, **0.02** mg $N \cdot L^{-1}$ for Kjeldahl nitrogen, 0.05 mg $N \cdot L^{-1}$ for nitrate, and **0.02** mg $N \cdot L^{-1}$ for NH, (Metropolitan Council **1981).**

In calculating water quality values for a given sample date, data were averaged among samples taken at different depths within the epilimnion, or for the entire water column during periods of complete mixis. Epilimnetic or water column averages for each sampling date were then combined to calculate growing season averages for the period of stratification (mid-May to early September) for each year of record. For Diamond Lake, which exhibited constant mixing over the summer months, averages were calculated over the typical stratification period for dimictic lakes. When available, averages for three consecutive years were combined to minimize the influence of year-to-year climatic variability. To reduce the potential problem of autocorrelation in subsequent regression analyses, data points representing the same lake were each weighted by 0.5. A Wilks-Shapiro test was used to test the normality of distributions for water chemistry variables (STATISTIX **1987).** Log10 (log) or arc sine (arcsin) transformations were performed on water quality variables where necessary to stabilize the variance (Snedecor and Cochran **1980).** Principal components were approximately normally distributed according to Wilks-Shapiro statistic and thus were not transformed before inclusion in stepwise linear regressions.

2.4 Statistical analyses

To reduce the large number of watershed characteristics to a smaller number of variables, we performed a principal components analysis (PCA) without rotation on selected watershed variables (Norusis **1988).** The original watershed variables were not expressed in common units and thus had a wide range of variances (Table **2).** Therefore, PCA was performed on standardized variables, ***i.e.***

Table 3. Comparison of analytical methods used.

storet code	variable	agency	methods	years methods used	ref[a]
00410	Alkalinity, total	MPCA	Potentiometric endpoint, pH 4.5	1953–74	1
			Automated Colorimetric Modified	1974–1976	1
			Titration Brom Cresol Green End Point	1976–1988	1
		MDNR	Colorimetric endpoint (methyl orange)		1
00680	Carbon, total organic	MPCA	Dohrman DC-80 TOC Analyzer	1987–88	2
		MC	Dohrman Analyzer	1980–87	2
33211	Chlorophyll ***a***	MPCA	Spectrophotometric	1976–88	3
		MC	Spectrophotometric	1980–87	1
31505	Coliform, fecal	MPCA	Multiple tube technique	1953–1984	1
31625	Coliform, fecal	MC	Membrane filter technique	1980–87	3
00080	Color	MPCA	Visual comparison with standards	1953–88	1
00095	Conductance, specific	MPCA	Conductivity meter	1967–88	1
00300	Dissolved oxygen	MPCA, MDNR	Winkler. mod'd Azide method	1953–88	1, 2
00900	Hardness. total	MPCA	EDTA method	1953–72	1
		MPCA	Selective ion electrode	1972–75	4
		MPCA	Summation (Ca + Mg)	1975–88	2
00610	Nitrogen, ammonia	MPCA	Distillation, colorimetric Nesslerization	1960–79	1
		MPCA	Alkaline oxidation, diazotization (aut'd)	1979–88	5
		MC	Automated colorimetric phenate method	1980–87	2
00630	Nitrate + nitrite	MPCA	Cadmium reduction, colorimetric	1974–88	1
		MC	Hydrazine reduction, colorimetric	1980–87	2
00605	Organic N	MPCA	Acid digestion, Colorimetric Nesslerization	1960–79	1
		MPCA	TKN minus NH3-N	1979–88	1
00625	Total Kjeldahl N	MPCA	(Ammonia-N + Organic-N)	1953–79	1
		MPCA	Block digest, AAII Salicylate	1979–88	1
		MC	Manual digestion, aut'd colorimetric phenate	1980–87	2
00403	pH	MPCA	Electrode	1953–88	1
70507	Orthophosphorus	MPCA	Colorimetric, Ascorbic acid	1977–88	1
00665	Total phosphorus	MPCA	Persulfate digestion, manual colorimetric	1974–79	1
			Block digestor, aut'd colorimetric	1979–88	1
		MC	Persulfate digestion, automated colorimetric	1980–87	2
00530	Total suspended solids	MPCA, MC, MDNR	Gravimetric, dried at 105 C	1953–88	1
00535	Volatile susp'd solids	MPCA, MDNR	Gravimetric, ignition at 600 C	1953–77	1
		MPCA, MC, MDNR	Gravimetric, ignition at 550 C	1977–88	1
00076	Turbidity	MPCA	Helige turbidimeter	1953–69	1
		MPCA, MDNR	Hach turbidimeter	1970–88	1
	Metals	MC	Low level: flameless AA spectrophotometric	1980–87	2

1 = APHA (1980 and earlier editions); 2 = US EPA (1979); 3 = USEPA (1973); 4 = Orion Research (1973); 5 = Richards and Kletsch (1964).

with the correlation matrix used in place of the covariance matrix (Johnson and Wichern 1982). Because mixing ratio is an important predictor of internal nutrient regeneration and was not strongly correlated with any of the landscape principal components we used, we excluded it from subsequent PCAs and included it as a separate independent variable in regression analyses.

Many of the variables were linearly related because they were expressed as percentages of a fixed total:

% agricultural + % urban/residential +
% forested + % lakes + % wetlands = **100**

or because they represented the sum of a subset of variables:

% wetlands = % herbaceous wetlands + % woody wetlands.

The presence of linearly related variables will produce a singular matrix (and computer overflow errors) or an ill-conditioned matrix with associated round-off errors (Draper and Smith **1981).** To resolve this problem, we chose a subset of **27** watershed variables to eliminate redundant combinations and gave preference to variables of known importance to water quality (Table **2).** For example, when summary variables (*e.g.*, % wetlands) were included, we eliminated one or more components of that summary variable (*e.g.*, % herbaceous temporarily-flooded wetlands).

If the water quality data matrix had been complete with no missing values, it would have been appropriate to apply canonical correlation analysis to the whole data set to identify relationships between multivariate landscape components and multivariate water quality components. Instead, because selected water quality data were missing for some site-years, a second PCA was performed on a smaller complete matrix that included eight water quality variables (dissolved oxygen, Secchi depth, log total phosphorus, log total nitrogen, log organic nitrogen, log chlorophyll *a*, log (nitrate + nitrite), log ammonium) and **28** site-years. The water-quality principal components were used as dependent variables in subsequent regressions. However, regressions also were performed using the original water quality variables because a greater number of cases were available for analysis.

To relate watershed variables with water quality variables, we performed stepwise multiple regression analyses with principal components **1** to **6** and lake mixing ratio (MIXRATIO) as the independent variables and a water quality parameter or water quality principal component as the dependent variable (Norusis **1988).** Principal components are appropriate variables to use in multiple regression because they are uncorrelated so that problems of multicollinearity of independent variables are reduced (Tatsuoka **1971).** Thus, estimates of regression coefficients do not depend on the order in which independent variables are included in regression analyses.

Independent variables were selected for inclusion in regression equations based on the magnitude of partial correlations with the dependent variable. Selected variables were then included ($p < .05$) or rejected ($p > .10$) on the basis of F-tests (Norusis **1988).** To reduce the probability that regression results are spurious, we restricted the final number of variables included in equations at the *end* of stepwise variable selection so as to produce a case:parameter ratio of **5:1** or greater. Regression results were checked for the presence of influential outliers using Cook's distance (Norusis **1988).** Finally, the influence of each original watershed variable that was highly correlated ($p < .01$) with principal components included in regression equations was tested through an analysis of partial correlations (Norusis **1988).** Landscape variables were transformed to approximate normal distributions before performing partial correlation analyses.

3. Results

3.1. Watershed, lake and wetland characteristics

Our study watersheds contained slightly more wetlands and less developed land (average: **39%** agricultural, **19%** urban, **14%** wetland, **12%** forest) than the Twin Cities metropolitan area as a whole, *i.e.*, **27%** urban land, **8%** wetlands (Oberts and Jouseau **1979).** Spring Lake watershed had the most agricultural area **(71%** agriculture) and Crystal Lake watershed was most highly developed (*e.g.* **83%** urban/residential), while Chub Lake **(36%** forest) and Golden Lake watersheds **(41%** wetlands) were the least developed watersheds. Total watershed area ranged from **2** to 50 km^2, with an average of **17** km^2 (Table **2).** Because lakes had been selected in a stratified fashion to cover the range of lake sizes found in the Twin Cities metropolitan area, lakes in our subsample were larger on average (**0.2**– 15.0 km^2) than Twin Cities metropolitan area

Table 4. Variance explained by first eight principal components of landscape or lake morphometry variables for 33 lake watersheds in Minneapolis-St. Paul metropolitan area (n = 38 site-year observations).

principal component	eigen-value	percent variance explained	cumulative percent variance
1 Wetlands/watershed area	7.0	26	26
2	4.0	15	41
3	3.7	14	54
4 Herbaceous wetland type	2.4	9	63
5 Forest/soils	2.0	7	70
6 Dilution ratio	1.6	6	76
7 Other lakes	1.3	5	81
8 Watershed relief	1.1	4	85

lakes as a whole (Metropolitan Council 1981). Thus, 6% of our 33 lakes are less than 0.25 km^2 in size, as compared to 71% of Twin Cities metropolitan area lakes.

Half of the lakes we selected (16 of 33 lakes) had a mixing ratio between 3 and 9, and, hence they are lakes transitional between summer polymictic and dimictic lakes (Osgood 1988). Mixing ratios for the remaining lakes were less than 3, which according to Osgood's criteria were polymictic. However, when temperature profiles were checked for each lake, only one lake (Diamond Lake) exhibited frequent mixing.

Wetlands constituted between 0 and 41% of the 33 watersheds studied. The majority of wetlands (63%) were herbaceous, with woody wetlands and open water wetlands constituting 25% and 12% of total wetland area, respectively. The predominant wetland water regime was rated (66% of total wetland area), followed by semi-permanently and permanently flooded. Temporarily-flooded wetlands constituted a negligible proportion of the total (<1%).

3.2. Principal components analysis of watershed variables

To reduce the number of variables considered, we extracted only the first eight principal components which had eigenvalues greater than 1.0. These eight principal components (PCs) explained 85% of the variance of the 27 original landscape variables (Table 4). The first six of these PCs, explaining 76.3% of the variance, were used in subsequent regression analyses. PC7 and PC8, which explained only a small fraction of the variance, were omitted to maximize the case:variable ratio.

Three of the first eight PCs (PC1, PC2, and PC4) were strongly related to wetland variables (Table 5). The first principal component (PC1) was correlated with 18 of the 27 landscape variables (p .05), but was most strongly correlated with variables associated with wetland size and extent, and to a lesser extent with total and contributing watershed area. The second PC represented the contrast between wetland extent and agricultural land-use. The graph of watershed locations in principal component space defined by PC1 and PC2 (Fig. 2) demonstrates the tendency of the larger watersheds with high PC1 values (*e.g.* Parley, Cynthia, Spring, and Coon) to have a wider variation in agricultural land-use relative to wetland extent (**AGR** = 19–71%, WTLD = 6–38%), as compared to the smaller watersheds at the left end of the PC1 axis (*e.g.* Zumbra, Crystal, and Lotus Lakes) which show little variation (**AGR** = 3–35%, WTLD = 0–2.3%).

PC4 represented the dominance of herbaceous semi-permanently-flooded versus herbaceous wetlands (*e.g.* wet meadows) in different watersheds. Watersheds with a high positive value for PC4 had predominantly herbaceous semi-permanently flooded wetlands

Table 5. Significant correlations (p < .05, **p < .01**) between 27 landscape or lake morphometry variables and 8 principal components (pc) for 33 lake watersheds in Minneapolis-St. Paul metropolitan area. Fringe areas defined as 200 m width band surrounding lake or stream. (n = 38 site-year observations).

	pc1[a]	pc2[b]	pc3[c]	pc4[d]	pc5[e]	pc6[f]	pc7[g]	pc8[h]
Watershed area	**0.72**	0.34	−0.36					
Contributing watershed area	**0.75**	0.39	−0.32					
Dilution ratio	−0.37					**−0.71**		
Average wetland size	**0.86**							
Stream wetland size	**0.83**							
Wetland distance upstream	**0.83**	0.37		−0.32				
% Wetland	**0.75**	**−0.54**						
% Wetland lake fringe	0.32	−0.39	**0.55**	**0.53**				
% Herbaceous semi-permanent wetlands	**0.61**			**0.60**				
% Herb. semi-perm. wtld lake fringe	0.33		**0.44**	**0.70**				
% Herb. seasonally-flooded/saturated wtlds	**0.47**	**−0.56**		**−0.51**				
% Herb. seas-fl/sat'd wetland lake fringe		**−0.46**	**0.53**				0.36	0.40
% Woody seas-fl/sat'd wetlands	**0.58**	**−0.54**						
% Woody seas-fl/sat'd wetland lake fringe	0.34	**−0.50**				0.37		
% Agricultural	**0.42**	**0.53**	**0.57**					
% Agricultural lake fringe		**0.62**					**−0.43**	
% Forested	−0.33		**0.50**		**0.68**			
% Forested lake fringe			**0.52**		**0.52**	0.32		−0.36
% Urban/residential	**−0.43**		**−0.72**					
Average slope	**−0.53**		0.40		0.37			
Average elevation difference		**0.73**				0.32		**0.47**
K-factor		0.39	**0.59**		−0.33			
Soil phosphorus	**−0.42**		0.32		**−0.46**			
Soil pH				**0.51**	**0.60**			
# Lakes upstream	**0.56**	0.40				0.32	**0.42**	
% Other lakes	**0.41**	**0.54**					**0.44**	−0.32
% Stream fringe area	**0.51**		0.37	**−0.54**				

[a]Wetlands/watershed area; [b]Agriculture/wetlands; [c]Agriculture/urban-residential; [d]Herbaceous wetland type; [e]Forest/soils; [f]Dilution ratio; [g]Other lakes; [h]Watershed relief.

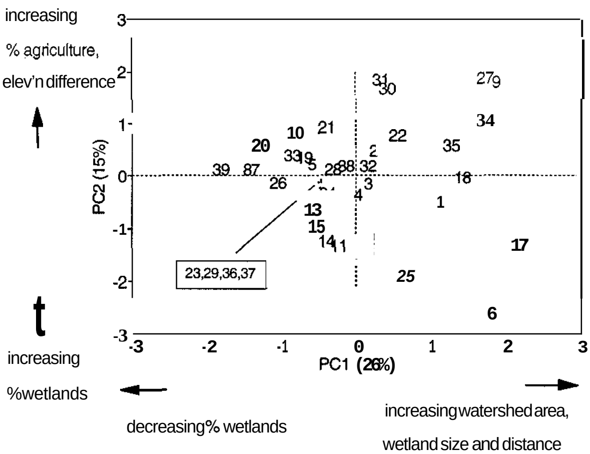

Fig. 2. Thirty-three lake watersheds (38 site-years) selected from within the Minneapolis-St. Paul metropolitan area graphed in landscape principal component space, PC1 (wetlands/watershed area) versus PC2 (agricultural/wetlands). See Table 1 for explanation of codes.

(*i.e.*, cattail marshes). Eagle and Orchard Lakes, for example, had 6.1 and 6.0% coverage, respectively. In contrast Golden Lake, with a low PC4 score, had marshes covering only 1.0% of the watershed, and 18.8% coverage of herbaceous seasonally-flooded/saturated wetlands.

Two of the remaining PCs (PC3 and PC5) were related to land-use categories. PC3 represented the contrast between agricultural (+) and urban (−)

Table 6. Average, standard error, minimum, and maximum epilimnetic values of chemical and physical parameters for 38 lake water-quality observations within the Twin Cities metropolitan area, MN.

variable	abbrev	units	n	avg	std err	min	max
Total phosphorus	TP	mg P/L	38	0.08	0.017	0.01	0.60
Dissolved phosphorus	DP	mg P/L	25	0.04	0.014	0.01	0.36
Total nitrogen	TN	mg N/L	33	1.7	0.20	0.6	7.2
Organic nitrogen	ON	mg N/L	31	1.5	0.21	0.2	6.8
Nitrates + nitrites	NOx	mg N/L	34	0.06	0.012	0.01	0.33
Ammonium (NH4-N)	NH4	mg N/L	32	0.14	0.022	0.02	0.47
Un-ionized ammonia	NH3	mg N/L	15	0.015	0.0025	0.006	0.03I
Secchi depth	SECCHI	m	35	1.59	0.129	0.20	3.29
Chlorophyll a	CHLA		30	49	12.5	6	309
Chl. a, corr'd	CHLACORR		8	17	4.7	1	47
Color	COLOR	Pt-Co units	12	21	4.2	6	63
Turbidity	TURB	Hach FTU	8	3.7	0.64	1.1	5.8
Total suspended solids	TSS	mg/L	21	14	3.1	1	70
Volatile suspended solids	VSS	mg/L	20	12	3.0	1	66
Dissolved oxygen	DO	mg/L	34	8.6	0.25	4.2	11.5
Total hardness	HARD	mg/L CaCO,	20	134	8.4	68	234
Total alkalinity	TALK	mg/L CaCO,	33	118	4.3	71	162
Lab pH	PH	[H+1	33	8.3	0.15	4.7	9.5
Conductance at 25°C	COND	MHO/cm	32	304	12.5	153	446
Chloride	CL	mg/L	24	32	6.2	4	144
Fecal coliform	FCOL	Colonies/100 mL	23	4	0.8	1	16
Arsenic	AS	µg/L	18	7.7	0.79	2.0	14.0
Total lead	PB		18	2.4	0.23	0.8	4.7
Cadmium	CD		19	0.26	0.03	0.04	0.62
Total mercury	HG		18	0.17	0.018	0.05	0.30
Zinc	ZN		19	10.9	1.3	1.0	24.0
DERIVED PARAMETERS							
Volatile/total SS	VSS/TSS		20	0.80	0.041	0.40	1.00
TN:TP	TNTP		33	27.5	1.79	6.4	46.1
Dissolved/total P	DP/TP		25	0.46	0.032	0.17	0.76
Organic N/Total N	ON/TN		31	0.83	0.028	0.24	0.98
N	NH4/TN		31	0.11	0.024	0.01	0.70
NOx-N/Total N	NOx/TN		33	0.04	0.08	0.01	0.19

land-use, but also was strongly correlated with soil erodibility. Watersheds with high **PC3** values (Forest Lake **1957,** Dutch Lake) were relatively undeveloped **(AGR** = 29–47%, FOR = 15–30%), while Crystal Lake **(1966, 1980)** with the lowest **PC3** score had the highest percent urban/residential coverage **(83%)** of all watersheds. PC5 was positively related to forested area, forested lake fringe, and soil pH, and was weakly inversely related to surface soil phosphorus. Chub Lake, with **36%** forested land, was distinct from the other watersheds with a high **PC5** score.

The strong negative association between **PC6** and the dilution ratio, with only weak correlations with other landscape variables reflects the independence of lake morphometry from watershed characteristics. Chub and Bryant Lakes, with the highest values of **PC6,** have small to intermediate-size watersheds **(6–12** km²) and are relatively small **(0.8–10.0** km²) lakes. Waconia Lake, with the lowest value for **PC6** has a large watershed **(42** km²), and is the largest lake in our study **(140** km²).

In the following discussion, the first **six** PCs will be referred to as the area **PC** **(PC1),** the **PC (PC2),** the **PC (PC3),** the herbaceous wet-

Table 7. Significant correlations between lake water quality variables and water-quality principal components (WQPC) for 28 Minneapolis/St. Paul metropolitan area sites represented in reduced data matrix ($p < 0.05$, $\mathbf{p < 0.01}$).

variable	wqpc1[a]	wqpc2[b]	wqpc3[c]
Dissolved oxygen	**0.60**		−0.36
Secchi depth	**−0.87**		
Log total phosphorus	**0.90**		
Log ammonium-N		**0.93**	
Log total nitrogen	**0.96**		
Log organic nitrogen	**0.87**	−0.35	
Log chlorophyll *a*	**0.93**		
Log ($NO_3 + NO_2$)		0.35	**0.86**

[a] Trophic state component; [b] Ammonium component; [c] Nitrate component

land class PC (PC4), the forest/soils PC (PC5), and the dilution ratio PC (PC6) (Table 4).

3.3. Lake water quality

Study lakes in the Twin Cities metropolitan area (Table 6) fall within the classification of Group III lakes (specific conductance 141–501 μmhos $\cdot$ cm^{-1}), which occur on calcareous substrates throughout most of Minnesota and are characterized by the deposition of marl to the sediments at conductance $> 190\ \mu mhos \cdot cm^{-1}$ (Gorham et al. 1983). Our study lakes are typical of Twin Cities metropolitan area lakes, the majority of which are hardwater, alkaline (80–180 mg $CaCO_2 \cdot L^{-1}$), and eutrophic ($> 10\ \mu g$ chl $a \cdot L^{-1}$, < 1.5 m Secchi depth, $> 30\ \mu g$ total $P \cdot L^{-1}$; Oberts 1981). Ratios of total N:total P suggest that most of the study lakes are phosphorus-limited; only 3 lakes (Cynthia, Golden, and Spring Lakes) have total N:total P levels less than 12, the boundary between N and P limitation (Dillon and Rigler 1974). Chloride levels were higher (4–144 $mg \cdot L^{-1}$) than is typical of Group III lakes (0.7–1.8 $mg \cdot L^{-1}$), suggesting the presence of contamination from road salt. Of the heavy metals measured, only zinc levels (1–24 $\mu g \cdot L^{-1}$) were well below the maximum allowable (270–520 $\mu g \cdot L^{-1}$) and short-term (47 $\mu g \cdot L^{-1}$) guidelines for protecting freshwater aquatic life (EPA 1980).

3.4. Principal components analysis *of* water quality

Principal components analysis of the reduced water quality matrix for 28 of the original 39 site-years identified three principal components which explained 84.1% of the variance. The first water quality principal component (WQPC1) was related to lake trophic state and was highly correlated positively with total P, total and organic N, and chlorophyll a, and negatively associated with Secchi depth ($p < .05$; Table 7). Diamond and Cynthia Lakes, with high positive WQPC1 values, were highly eutrophic (0.36–0.60 $mg \cdot L^{-1}$ total P, 4.0–7.2 $mg \cdot L^{-1}$ total N, 280–309 μg chl $a \cdot L^{-1}$, 0.2–0.4 m Secchi depth), as compared to White Bear Lake (1980,1987) with a low value of WQPC1 (0.02 $mg \cdot L^{-1}$ total P, 0.64–0.68 $mg \cdot L^{-1}$ total N, 7–11 μg chl $a \cdot L^{-1}$, 3.1–3.2 m Secchi depth). The second PC (WQPC2) was significantly correlated with log NH_4, while WQPC3 was significantly correlated with log ($NO_3 + NO_2$) (Table 7). In the following text, WQPC1 will be refered to as lake trophic state.

3.5. Multiple regression and partial correlation analyses

Multiple regression analysis demonstrated that trophic state variables were correlated with watershed-scale (PC1), land-use (PC2, PC5), and wetland (PC4) components (Table 8). Positive correlations between trophic state and PC1 or PC2

Table 8. Results of multiple regressions with growing season epilimnetic water quality averages as dependent variables and landscape principal components plus mixing ratio as independent variables. Dependent variable abbreviations and units defined in Table 6. Trophic state variables were correlated with watershed-scale (PCI), land-use (PC2, PC5) and wetland (PC4) components.

dependent variable	n	adjusted r^2	F-stat	regression constant	regression coefficients associated with independent variables*** pc1	pc2	pc3	pc4	pc5	pc6	mixratio
log,, total P	33	0.28	5.2**	– 1.24	0.13	0.12		−0.12			
log,, total N	29	0.30	7.0**	0.17		0.07			−0.12		
log,, organic N	28	0.36	8.7**	0.08		0.11			−0.16		
arcsin (ON/TN)	28	0.25	5.4*	1.03		0.09			−0.10		
	30	0.15	5.8*	– 1.6							0.06
$\log_{10}$ (NOx/TN)	31	0.14	6.0*	– 1.8							0.07
$\log_{10}$ NH_4-N	13	0.39	8.7*	– 1.97	0.14						
$\log_{10}$ NH_4-N/TN	28	0.40	10.0**	– 1.14		−0.15			0.24		
total N:total P	29	0.28	6.5**	28	·5			3			
log,, chl. ***a***	28	0.24	9.6**	1.48		0.20					
Secchi depth	32	0.23	5.7**	1.5		– 0.3		0.3			
color	11	0.76	16.5**	20	8			– 7			
log,, turbidity	8	0.48	7.5*	0.37						0.31	
$\log_{10}$ total **SS**	21	0.59	8.2**	1.4		0.13	0.24		– 0.18		−0.15
log,, volatile SS	20	0.24	6.9*	0.89					−0.17		
log,, chloride	22	0.31	5.9**	1.41		0.14			−0.14		
log,, total lead	18	0.35	5.7*	0.35		0.11			– 0.07		
zinc	19	0.29	8.4**	11			4				
WQPC1	28	0.49	9.8**	−0.2	0.3	0.4			– 0.6		
WQPC3	28	0.23	9.3**	−0.9							0.25

*p < .05
**p < .01
***PC1 = Wetlands/watershed area PC; PC2 = PC; PC3 = PC; PC4 = Herbaceous wetland type; PC5 = Forest/soils PC; PC6 = Dilution ratio PC.

and the negative correlation between trophic state and PC5 were explained by the influence of agricultural or forested land-use, topography, and wetland position within the watershed. Lake trophic state was negatively correlated with forested land-**use** with other PC5-related variables held constant (Table 9). Trophic state was positively correlated with agricultural lake-fringe and maximum elevation difference when the number of lakes upstream and wetland extent were held constant. Lake trophic state also was positively correlated with wetland distance upstream with other factors influencing PC1 held constant (Table **9).** Conversely, Iakes with proximal wetlands, *i.e.* relatively close upstream, had a lower trophic state.

The relationship between lake total or organic N and PC2 also was explained by the influence of agricultural land-use. The partial correlation between agricultural lake fringe and organic or total nitrogen was significant with other PC2-related variables held constant (Table 9). Total and organic nitrogen were both negatively correlated with the forest/soils component (PC5) but the effect of original variables related to PC5 could not be separated through partial correlation analysis. Elevation difference was significantly correlated with total phosphorus or chlorophyll *a* with other PC2-related variables held constant.

Our evidence for the effect of agriculture on lake trophic state in the Twin Cities metropolitan area is based strictly on significant correlations, and thus is subject to errors in our interpretation of cause/effect relationships. It is possible, for example, that agricultural zones remaining in the face of strong

Table 9. Partial correlations (rp) between lake water-quality variables and original watershed variables for 33 lake watersheds in Minneapolis-St. Paul metropolitan area (n = 38 site-year observations).

independent variables			
water-quality	watershed	partial correlation, r_p	original watershed variables held constant for partial correlation
WQPCl (trophic state)	FOR	−0.43*	SOILP, SOILPH
WQPCl (trophic state)	AGLKFR	0.49**	OLAKES, WTLND
WQPC1 (trophic state)	ELEVDIFF	0.47*	OLAKES, WTLND
WQPCl (trophic state)	WTLDDIST	0.41*	WTLND, CAREA, AVWTLDSZ
Total nitrogen	AGLKFR	0.39*	OLAKES, WTLND, ELEVDIFF
Organic nitrogen	AGLKFR	0.45*	OLAKES, WTLND, ELEVDIFF
Total phosphorus	ELEVDIFF	0.36*	OLAKES, AGLKFR
Chlorophyll *a*	ELEVDIFF	0.50*	OLAKES, AGLKFR
Color	WTLND	0.80**	CAREA, WTLDDIST, AVWTLDSZ
Color	STRFRG	0.81**	HBSPLKFR, HRBSP
Secchi depth	STRFRG	0.50**	HBSPLKFR, HRBSP
Total suspended solids	OLAKES	−0.52*	AGLKFR, WTLND, ELEVDIFF
log,, total lead	WTLD	−0.60*	ELEVDIFF, OLAKES, AGLKFR
log,, total lead	FOR	−0.52*	SOILP, SOILPH

* $p < 0.05$, ** $p < 0.01$

developmental pressures are located on the most fertile land, and that lakes in these areas have a naturally high trophic state. We tested this hypothesis through partial correlation analysis. Agricultural lake-fringe area (Log AGRLKFR) was significantly correlated with lake trophic state when available soil phosphorus was held constant ($p < .01$), but there was no significant effect of soil phosphorus on trophic state when agricultural lake fringe area was held constant ($p > .05$; Table 9).

Lake transparency was associated both with wetland/watershed area (PC1) and herbaceous wetland type (PC4) components. Color was positively correlated with wetland extent with other PC1-related variables held constant. Landscapes dominated by wet meadow or herbaceous seasonally-flooded wetland, and with extensive stream-fringe areas (high PC4) contributed more to lake color (and reduced Secchi depth) than did those dominated by cattail marshes. Percent stream-fringe was positively correlated with color and negatively correlated with Secchi depth with other PC4-related variables held constant (Table 9).

Chloride levels were influenced primarily by land-use components. Lakes within landscapes with a high proportion of agriculture relative to wetlands (high PC2) or with sparse forest coverage (low PC5) had higher Cl levels.

Independent variables related to internal (mixing ratio) or external (dilution ratio) loading were significantly correlated with turbidity, total suspended solids, and nitrate. Lakes with low dilution ratios (high PC6) had high turbidity. Those lakes with a tendency for stable stratification periods (high mixing ratio) had high nitrate levels but low suspended solids. In addition, the number of other lakes upstream was negatively correlated with total suspended solids with other PC2-related variables held constant (Table 9).

Of the metals considered, only zinc or lead levels could be explained as a function of landscape components. Lakes in watersheds that were highly agricultural with little urban development (high PC3 values) had higher levels of zinc. However, zinc was significantly correlated with soil K-factor ($r = \mathbf{0.47}$; $p < .05$) and not with % urban or % agricultural land-use. Lakes within agricultural watersheds with sparse wetland coverage (high PC2) or low forest coverage (low PC5) had higher lead levels. The negative partial correlation between log total lead and wetland extent was significant when additional PC2-related variables were held

P than do seasonally-flooded/saturated wetlands. Brown **(1985)** compared nutrient budgets among four different Twin Cities metropolitan area wetlands, and determined that the one impounded wetland had the greatest removal efficiency for suspended solids, total P, and organic N. In addition, many of the streams in the Twin Cities metropolitan area have been channelized, thus reducing the degree of contact between water and stream-fringe wetlands and reducing the efficiency of stream-side wetlands in trapping suspended sediments and associated pollutants (Johnston *et al.* **1990).**

Lake transparency (Secchi depth) was positively associated with the herbaceous wetlands component (PC4) as well as with PC2 (agriculture/wetlands). The association of seasonally-flooded/saturated herbaceous wetlands with low lake transparency and high color helps to explain why these wetlands are associated with higher lake total P levels, but not with a higher trophic state index. The release of organic acids could counteract the effect of higher P loading by limiting lake transparency and thus algal biomass. Among the lakes studied, the relationship between log Secchi depth and log color was significant ($r = -0.66$, $p < .05$), and the range of lake color observed (6–63 PCU) corresponded to a range of >2 meters in Secchi depth. Beaver and Crisman **(1991)** found that empirical equations predicting primary productivity of Florida lakes were improved by first separating clear-water lakes from colored lakes in the analysis. Annual areal gross production responded more to a given level of total phosphorus in clear lakes than in colored lakes (Beaver and Crisman **1991).**

Levels of nitrate + nitrite and the fraction of nitrogen present as nitrate plus nitrite (log (NO_3 + NO_2)/total N) were related only to the mixing ratio of lakes, but not to land-use components. The epilimnion of lakes with more stable stratification periods would have little input of ammonium from the hypolimnion, and ammonium present from external loadings would be readily nitrified under the aerobic conditions in these surface waters (Wetzel **1983).**

4.2. Suspended solids

The total suspended solids concentrations in Twin Cities metropolitan area lakes was influenced by both internal (lake mixing ratio) and external (PC2, agricultural/urban PC3, forest/soils PC5) variables. The negative relationship between log (total suspended solids) and lake mixing ratio probably is the result of the resuspension of sediments during mixing events, which are less frequent in lakes with a high mixing ratio value (Osgood **1988).** Highly erodible agricultural lands (high PC3) can contribute large sediment loadings to surface waters, whereas urban/residential lands contain more impervious surfaces and in the absence of construction activity contribute mainly fine particulates to surface waters (Novotny and Chesters **1981).** The correlation between PC2 and total suspended solids probably was related to the ability of upstream lakes to act as settling basins.

Both total and volatile suspended solids were negatively related to PC5. Forested riparian zones can act as filters for sediment from agricultural runoff, retaining as much as **88%** of sediment inputs from adjacent fields (Gilliam *et* ***al.*** **1986).** Forested areas, in general, export less suspended solids than do agricultural or urban areas (Omernik **1976).**

4.3. ***Major*** *ions*

Chloride was positively related to PC2. A similar relationship was demonstrated in our earlier study of the cumulative effect of wetlands on stream water quality (Johnston *et al.* **1990).** The reduction in chloride in watersheds with more wetlands relative to agricultural land is probably the result of dilution by groundwater inputs, as well as the higher loading of fertilizer-derived Cl from agricultural lands (Prochazkova *et al.* **1983).** Alternatively, Cl has been found to be retained within an Ontario bog (Bayley *et* ***al.*** 1987**),** and could be retained by Twin Cities metropolitan area wetlands as well, although it is generally considered to be a conservative element.

4.4. Metals

Lead as a nonpoint source pollutant is associated with areas of high traffic. The level of lead in Twin Cities metropolitan area lakes was related more to watershed sinks than sources, however. A similar negative relationship was found between lead and wetland extent in our previous study of stream watersheds (Johnston *et al.* 1989). The negative relationship between total lead and wetland extent probably is related to the ability of wetlands to retain up to 100% of particulate Pb inputs (Giblin 1985). Unlike many other metals, Pb is relatively immobile in wetland environments (Giblin 1985).

4.5. Color

Seventy-six percent of the variation in epilimnetic color was explained as a function of the wetlands/watershed area and herbaceous wetland type components. Lake color increased as the extent of wetlands (PC1) increased, and as the extent of herbaceous seasonally-flooded/saturated wetlands increased (PC4). Individual wetlands export soluble organic carbon (Kowalczewski 1978), in particular humic acids (McKnight *et al.* 1985).

Lake color has been related to the ratio of muskeg area to lake surface area in Nova Scotia lakes (Gorham *et al.* 1986). To our knowledge, the capacity of herbaceous seasonally-flooded/saturated wetlands to export more highly colored waters than do marshes has not been reported before. Typical marsh vegetation in herbaceous semi-permanent wetlands of the Twin Cities metropolitan area (*Typha* spp., *Phragmites* spp.) has a slow rate of litterfall (<5 to 23% of aboveground production; Mason and Bryant 1975; Gustafson 1976) and a slow decay rate (half-life of 1–2 years). In contrast, wetland vegetation with less supportive tissue, such as the sedges and grasses found in seasonally-flooded/saturated wetlands can be expected to have a higher rate of litterfall and decay rate (van der Valk *et al.* 1978). For example, *Carex aquatilis* litterfall is equal to 100% of net aboveground production and occurs continuously throughout the growing season (Bernard and Gorham 1978). In addition, the longer period of standing water in marshes would expose internally-generated dissolved organic carbon to photolytic degradation for a longer period than in seasonally-flooded wetlands.

4.6. Success and limitations of the landscape-based multivariate approach

We were able to explain a statistically significant proportion of the variability in water quality of Twin Cities metropolitan area lakes for 13 variables and 4 derived variables as a function of six landscape components and the lake mixing regime. However, our ability to predict lake water quality (average $r^2 = 0.34$, $r^2 = 0.14–0.76$) was lower than our ability to predict summer stream water quality in the same region (average $r^2 = 0.74$, range $= 0.17–0.88$) using a similar approach (Johnston *et al.* 1990).

Because of the low flow rates and long water retention time for lakes as compared to streams, lake water quality is affected more by internal processes such as sedimentation, sediment resuspension, and nutrient transformations including regeneration from the sediments, and less by exchanges across the land/water interface than is stream water quality. This disparity increases as lakes become more eutrophic and anoxic sediments release increasing amounts of phosphorus. Thus, water quality reflects the history of nutrient loading to a lake as well as current land-use practices.

Empirical models to predict in-lake phosphorus concentrations have included not only a term for external loading, but also terms representing sedimentation and flushing rate (Vollenweider 1968). In lakes with similar morphology and small relatively homogeneous watersheds, the loading term predominates in this relationship. For example, Schindler (1971) was able to predict chlorophyll *a*, total P, and color of lakes in the Experimental Lakes Area simply as a function of volumetric loading: watershed area/lake volume. Similarly, total P loadings and in-lake concentrations have been successfully predicted as a function of the percentage of watershed developed for small urban Twin Cities metropolitan area lakes (Ayers *et al.* **1980;** Walker **1987).**

In analyzing data from a range of oligotrophic to eutrophic lakes across North America, Canfield and Bachmann **(1981)** discovered that the sedimentation coefficient was itself a function of external loading rates. Our empirical models were constrained to those expressing additive linear effects of the principal components on (transformed) water quality variables. Thus we did not consider the possible interaction between external loading and lake morphometry in accelerating the process of lake eutrophication. In contrast, Canfield and Bachmann's model incorporates the decreasing efficiency of net phosphorus sedimentation with increased loading, as modified by lake morphometry.

The prediction of total P and chlorophyll *a* concentrations can be further confounded by the influence of trophic interactions (Carpenter *et* ***al.*** **1985).** For example, Osgood **(1988)** noted that Twin Cities metropolitan area lakes containing flake-forming blue-green algae *(Aphanizomenon* spp.) had higher levels of total phosphorus than expected and suggested that these algae transport phosphorus to the epilimnion from the sediments. Wright and Shapiro **(1984)** have documented the role of large-bodied zooplankton in reducing epilimnetic P by transporting phosphorus downwards during vertical diel migrations. Within developing landscapes, lake trophic structure is often intentionally manipulated by rotenone treatment and/or fish stocking (Wright and Shapiro **1984).** Prediction of lake water quality from watershed characteristics in these settings might be improved by first stratifying lakes according to a trophic structure classification (*e.g.* Paloheimo and Zimmerman **1983;** Zimmerman *et al.* **1983).**

Our multivariate landscape approach has proven to be a rapid, cost-effective means of exploring empirical relationships between lake water quality and watershed characteristics. We were able to explain up to **76%** of the variation in individual water quality variables among the **33** lakes studied as a function of landscape components. We were also able to explain **49%** of the variation in a derived trophic state index (WQPC1), even though most of the lakes examined were already highly eutrophic, and thus were influenced by internal loading. Unlike the empirical (nonlinear) modelling approach described above, our technique has the advantage that it does not require extensive data on external loadings to lakes. Finally, by allowing us to reduce a multitude of landscape variables into a few independent principal components, this methodology has allowed us to evaluate how the extent and placement of wetlands and other landscape features affects lake water quality.

4.7. Implications for watershed management to protect lake water quality

Both the extent and position of wetlands in the landscape mosaic must be considered in developing watershed management plans to protect lake water quality. Within the Twin Cities metropolitan area, the proportion of lake fringe area in agriculture was significantly correlated with lake trophic status, particularly total and organic nitrogen levels. Conversely, urban lake fringe was not a good predictor of lake water-quality, possible because nonpoint source pollutants are routed into lakes through stormwater systems, circumventing the opportunity for riparian treatment of nonpoint source runoff. In both urban and rural landscapes, wetland extent can counteract the nonpoint source loadings of nitrogen, phosphorus, sediment, and lead and thus help to maintain lake ecosystem health.

The negative relationship between total lead and wetland extent in the watershed suggests that wetlands play a significant role in trapping and immobilizing particulate lead in urban runoff. In the Twin Cities metropolitan area, lakes in watersheds with less than **18%** wetland area are predicted to have levels of lead exceeding the EPA criteria denoting short-term toxicity to freshwater aquatic life (Detenbeck *et al.* **1991).** Wetlands should not be considered as permanent sustainable sinks for heavy metals such as lead in urban runoff, however, as little is known concerning the long-term potential for bioaccumulation by wetland biota (*e.g.* Horner **1988;** Stockdale **1991).**

Wetlands should be preserved throughout the watershed, but when nonpoint source loading to a lake within the watershed is a problem, restored or created wetlands could be sited strategically in the lower reaches of a watershed close to the lake of interest. Natural wetlands situated closest to surface waters should receive special protection. Replacement of drained or filled wetlands in areas away from surface waters in the watershed will not achieve the same reduction in nonpoint source loadings to a lake. Although our results suggest that seasonally-flooded streamside wetlands or wet meadows are less effective than cattail marshes in reducing total phosphorus loadings to downstream lakes, these wetlands contribute to lake color, and thus may moderate the response of downstream lakes to a given level of loading by limiting transparency.

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