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The cumulative effect of wetlands on stream water quality and quantity. A landscape approach

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Abstract. A method was developed to evaluate the cumulative effect of wetland mosaics in the landscape on stream water quality and quantity in the nine-county region surrounding Minneapolis–St. Paul, Minnesota. A Geographic Information System (GIS) was used to record and measure 33 watershed variables derived from historical aerial photos. These watershed variables were then reduced to eight principal components which explained 86% of the variance. Relationships between stream water quality variables and the three wetland-related principal components were explored through stepwise multiple regression analysis. The proximity of wetlands to the sampling station was related to principal component two, which was associated with decreased annual concentrations of inorganic suspended solids, fecal coliform, nitrates, specific conductivity, flow-weighted NH_4, flow-weighted total P, and a decreased proportion of phosphorus in dissolved form ($p < 0.05$). Wetland extent was related to decreased specific conductivity, chloride, and lead concentrations. The wetland-related principal components were also associated with the seasonal export of organic matter, organic nitrogen, and orthophosphate. Relationships between water quality and wetlands components were different for time-weighted averages as compared to flow-weighted averages. This suggests that wetlands were more effective in removing suspended solids, total phosphorus, and ammonia during high flow periods but were more effective in removing nitrates during low flow periods.

Introduction

'Cumulative impact,' the incremental effect of an impact added to other past, present and reasonably foreseeable future impacts, has been an area of increasing concern to regulatory agencies because the piece-meal loss of wetlands over time has seriously depleted wetland resources (Williamson et al. 1986; Preston & Bedford 1988). The legal mandate for cumulative impact assessment has existed for a decade (Council on Environmental Quality 1978), but the empirical data and assessment methodologies needed to make regulatory judgments about cumulative impacts to wetlands have been lacking until recently (Gosselink & Lee 1987).

To assess the cumulative impact of wetland loss at the watershed scale, techniques are needed to assess the cumulative *effect* of multiple wetlands on watershed functions. This is important because the cumulative function of all wetlands in a watershed may be different than the additive function of the individual wetlands themselves. If a non-linear relationship exists between wetland abundance and cumulative wetland function, the ecological ramifications of identical impacts would be different at different points along a cumulative disturbance gradient. Also, wetland effects which are locally significant may not be significant at a sampling point far downstream (Ogawa & Male 1986). Therefore, an approach must be used which evaluates the cumulative effects of wetlands in a landscape context.

This paper describes a method we have developed and used to examine the relationship between watershed mosaics and the water quality and flow output from those watersheds, focusing on the role of wetlands as a watershed component. A premise of this study is that the collective function of a watershed mosaic can be predicted by attributes of that mosaic (Preston & Bedford 1988), provided that the attributes and functions of the mosaic are measured at a suitable spatial and temporal scale (Allen et al. 1984). At the landscape scale used in this study, aerial photography and a Geographic Information System (GIS) were used for measuring watershed attributes, while seasonal and annual downstream water quality and flow averages were used as indicators of watershed function. Multivariate statistical techniques, which can evaluate many variables simultaneously, provided the means for relating the watershed attributes (independent variables) to watershed functions (dependent variables).

Most of the watershed attributes used in this study involved area measurements. This is important to the understanding of cumulative impact, because loss of wetland area by drainage or filling has been the most pervasive impact to wetlands in the United States (Tiner 1984). Although other types of impacts affect wetland functions, wetland loss is the most damaging type of impact which can occur because it eliminates all functions of a wetland. The importance of the cumulative impact of wetland loss has already been recognized in regulatory actions: EPA used cumulative loss of wetland acreage as a justification in at least two 404(c) vetoes (Hirsch 1988).

The purpose of this paper is to present the method developed, and to illustrate its use. Specific goals are:

— To empirically relate watershed attributes to downstream water quality and flow, focusing on wetlands as a watershed component, and
— To use those relationships to identify wetland classes most important to water quality maintenance and flow reduction.

Methods

Study Site selection

The Minneapolis–St. Paul metropolitan area, a 8075 km^2 region, was chosen for study (Fig. 1). The area lies primarily in the North Central Hardwood Forests

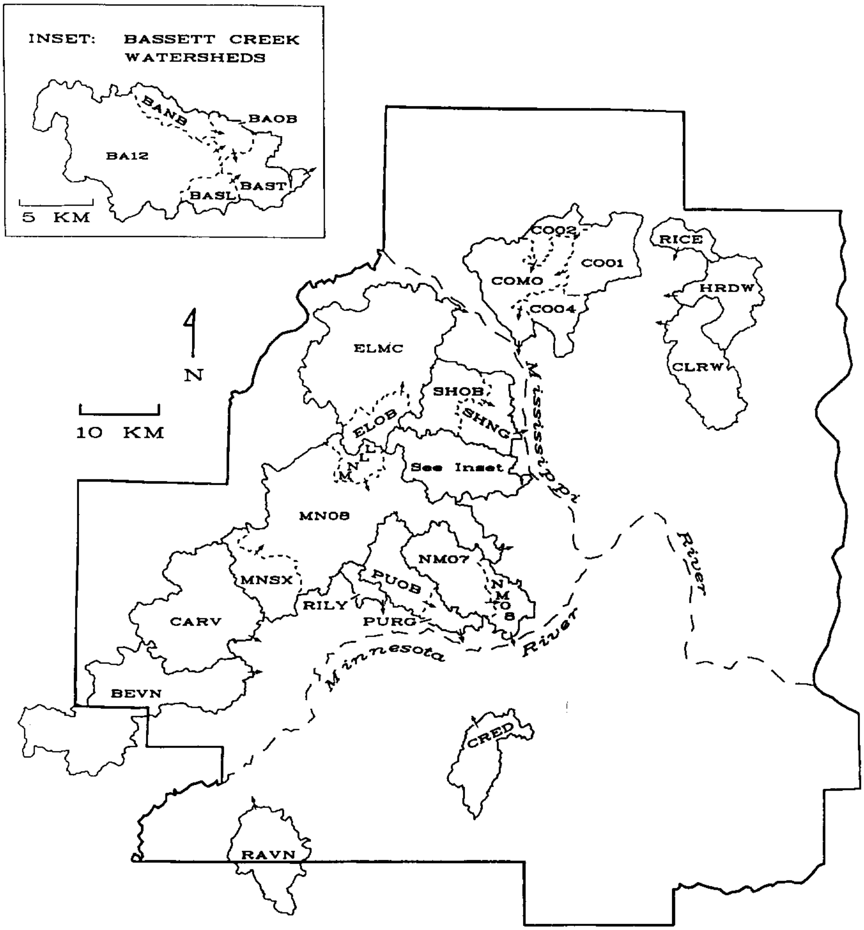


Fig. 1. Watersheds studied in the Minneapolis St. Paul Metropolitan Area (—— = major water boundary, – – – = watershed boundary within a major watershed, → = sample site at watershed outlet, indicating flow direction). Watershed codes and dates sampled are listed in Table 1.

ecoregion, with portions extending into the Western Corn Belt Plains (Omernik 1986). Wetlands in this region have been subjected to many developmental pressures from agriculture and urban expansion, so that by 1969 only about half of the pre-settlement wetland area remained (Anderson & Craig 1984). Wetlands now constitute about 7.6% of the region (Owens & Meyer 1978). Although the majority are herbaceous (Werth et al. 1977; Owens & Meyer 1978), a variety of wetland classes occur within the region.

Fifteen major watersheds covering 2073 km^2 were selected as study sites (Fig. 1). Because the primary objective was to relate watershed attributes to downstream water quality and flow, watersheds were selected for which there were both:
— stream monitoring data collected at least monthly, and
— concurrent aerial photographs of the monitored area (Table 1).

Table 1. Watershed area and data sources used for water quality analyses. Unless otherwise specified, both field data and aerial photos were available for the years indicated. Site-year numbers are used in Fig. 3.

Major watershed site description	Watershed code	Year	Site-year number	Area (Ha)	Agency collecting data
Bassett Creek	BAST	1966	01	10666	MN Pollution Control Agency
Oberts study	BAOB	1980	02	8293	Metropolitan Council (Oberts 1981)
Station 12	BA12	1975	03	6669	Barr Engineering
North branch	BANB	1975	04	993	Barr Engineering
Sweeney L. br.	BASL	1984	05	776	Barr Engineering
Bevens Creek	BEVN	1980	06	21652	Metropolitan Council (Oberts 1981)
Carver Creek	CARV	1980	07	17370	Metropolitan Council (Oberts 1981)
Clearwater Cr.	CLRW	1980	08	10222	Hickok & Associates
	CLRW	1987	09	10222	Hickok & Associates
Coon Creek	COON	1966	10	24456	MN Pollution Control Agency
Station 1	COO1	1980	11	7527	USGS (Arntson & Tornes 1985)
Station 2	COO2	1980	12	2787	USGS (Arntson & Tornes 1985)
Station 4	COO4	1980	13	4142	USGS (Arntson & Tornes 1985)
Credit River	CRED	1980	14	6221	Metropolitan Council (Oberts 1981)
Elm Creek	ELMC	1970	15	27629	MN Pollution Control Agency
	ELMC	1975	16	27629	MN Pollution Control Agency
Oberts study	ELOB	1980	17	3960	Metropolitan Council (Oberts 1981)
Hardwood Cr.	HRDW	1980	18	7296	Hickok & Associates
	HRDW	1987	19	7296	Hickok & Associates
Minnehaha Cr.	MNHA	1960*	20	37496	MN Pollution Control Agency

Six Mile Cr.	MNSX	1975	21	6 275	Hickok & Associates
Six Mile Cr.	MNSX	1980	22	6 275	Hickok & Associates
Six Mile Cr.	MNSX	1984	23	6 275	Hickok & Associates
Long L. outlet	MNLL	1975	24	2 754	Hickok & Associates
Long L. outlet	MNLL	1980	25	2 754	Hickok & Associates
Long L. outlet	MNLL	1984	26	2 754	Hickok & Associates
Nine Mile Cr.	NMIL	1975	27	11 683	Barr Engineering
Station 7	NM07	1966	28	8 180	MN Pollution Control Agency
Station 7	NM07	1984	29	8 180	Barr Engineering
Purgatory Cr.	PURG	1975	30	8 538	Barr Engineering
Oberts study	PUOB	1980	31	6 396	Metropolitan Council (Oberts 1981)
Raven Stream	RAVN	1980	32	8 477	Metropolitan Council (Oberts 1981)
Rice Creek	RICE	1966	33	2 639	MN Pollution Control Agency
Riley Creek	RILY	1973#	34	1 559	MN Pollution Control Agency
	RILY	1980	35	1 559	MN Pollution Control Agency
	RILY	1984	36	1 559	MN Pollution Control Agency
Shingle Creek	SHNG	1966	37	11 391	MN Pollution Control Agency
Oberts study	SHOB	1980	38	6 094	Metropolitan Council (Oberts 1981)

* 1957 aerial photography used.
# Landscape data from 1968 and 1975 interpolated to match 1973 water quality data.

Most of the major watersheds had a single sampling site (Fig. 1). For those seven major watersheds in which there were multiple sample sites, all stream monitoring data were spatially and/or temporally distinct (Fig. 1, Table 1), so that water sampled at a given site was never resampled further downstream. When multiple sample years existed for the same watershed or portion thereof, we used data sets separated in time by at least four years to minimize potential autocorrelation. To ensure that the watershed analysis included only those lands which could have affected surface water draining to a particular sampling site, watersheds were delineated for each unique sampling location. In this way, watershed conditions were both spatially and temporally paired with water quality and flow.

All data were summarized as 'site-years,' which represent watershed conditions in time. For watershed attribute variables (Table 2), a site-year value is a spatial summary statistic for the watershed, derived from aerial photos taken during the year in question (e.g. wetlands as a percentage of watershed area in 1980). For water quality and flow variables (Table 3) a site-year value is an average of the data from flow measurements or water quality samples taken at the mouth of the watershed during the year in question (e.g. average chloride concentration for all water samples collected in 1980). There were 38 site-years for each watershed attribute variable (Table 2), but fewer site-years for water quality variables because the different collecting agencies did not analyze for all the same parameters (Table 3). Each site-year was used as a separate case in the statistical analyses. Site-years are indicated in the text by an alpha-numeric code, the last two digits of which indicate the year of sampling (e.g. BAOB80).

Wetland and land use data

Existing aerial photography was used to document the location and extent of wetlands (defined as per Cowardin et al. 1979) for the years of water quality and flow record (Table 1). Photo enlargements were used whenever possible. Stereoscopic magnification ($3\times$) was required to interpret the 1957 air photos, which were only available as 9×9 contact prints. National Wetlands Inventory maps were used for 1980.

Because water quality and runoff is potentially affected by all the land within a watershed, both wetland and upland cover types were mapped for each date of photography. Mapping was done by U.S. Public Land Survey quarter–quarter sections, each covering a 40 acre (400×400 m) land area. Each cell was classified by the land use which constituted the majority of its area: agriculture, forest, urban/residential, lake, or wetland. Therefore, the minimum possible area detected by the mapping resolution was 14–20 acres (a third to a half of a quarter–quarter section). If the major cover type was wetland, it was further classified into one of nine wetland categories based on U.S. Fish and Wildlife Service criteria (Cowardin et al. 1979). The value for each quarter–quarter section was recorded on 1:24 000 USGS topographic maps, which were used for computer digitizing.

Two raster format Geographic Information System (GIS) programs run on a IBM PC/AT were used to enter and measure the landscape variables (Johnston et al. 1988). Digital data files of soil and topographic variables were obtained for the region from the Minnesota State Planning Agency. Land use, watershed boundaries, and streams were digitized from USGS topographic maps. Streams were classified by stream order (Morisawa 1968), and measured as line vectors directly from the X and Y coordinates in the digitized file. Where agricultural ditching had altered the drainage pattern between two site-years, the different stream configurations were digitized separately. The ERDAS GIS was used to extract individual watersheds from the regional data files, and to compute average soil and topographic variables (Table 2) for each site-year (Johnston et al. 1988).

Previous workers have demonstrated that wetland/upland (Whigham & Chitterling 1988; Johnston et al. 1984) and stream/riparian zone (Peterjohn & Correll 1984; Schnabel 1986) edges have an important effect on the flux of water and materials in the landscape, an effect which diminishes with increasing distance from the edge (Osborne & Wiley 1988). Therefore, the GIS was used to extract land use data from 175 m fringe zones on either side of streams, and 400 m upland fringe zones surrounding wetlands (Table 2). The fringe zones were also used to determine the proportion of the stream corridor in different stream order classes.

Wetlands in the headwaters of a watershed may have less of an influence downstream than do wetlands closer to the sampling site (Ogawa & Male 1986). Therefore, an index of average wetland stream order position relative to that of the sampling station (RELWTPOS) was developed as a simple means of estimating wetland distance upstream:

$$\frac{\sum_{i=1}^{j} (j - i)Ai}{\sum_{i=1}^{j} Ai} \qquad (1)$$

where j = stream order of sample point

Ai = area of ith order wetlands.

Two indices of watershed shape, elongation ratio and compactness ratio (USGS 1978), were used. A sequential comparison index (Cairns et al. 1968) was used to quantify the diversity of land use adjacent to streams by dividing the number of runs (i.e. a string of adjacent cells with identical classification) by the number of cells bisected by the stream.

Water quality data

Water quality data were obtained primarily from the STORET computerized database (U.S. Environmental Protection Agency, Office of Water and Hazardous Materials), supplemented with data collected by consulting firms for local

Table 2. Watershed attribute variables (n = 38 site-years). Landscape variables measured by the GIS.

Watershed variables	Code	Units[a]	Mean	S.D.	Min	Max
Total wetland	WTLD	%W	16.1	10.3	1.6	52.3
Herbaceous wetland	HERB	%W	9.3	7.1	0.0	39.6
Herbaceous wetland, seasonally flooded	HERBSF	%W	5.9	4.6	0.0	24.9
Herbaceous wetland, semipermanently flooded	HERBSP	%W	3.3	3.8	0.0	23.7
Woody wetland, seasonally flooded	WDYSF	%W	3.8	4.0	0.0	16.9
Wetland/lake fringe	WLKEFR	%W	2.0	1.9	0.0	7.0
Wetland/upland fringe, agricultural	WAGRFR	%W	18.9	8.8	0.0	33.3
Wetland/upland fringe, urban	WURBFR	%W	5.8	5.2	0.0	17.5
Total wetland fringe	WFRG	%W	32.3	8.2	11.2	47.5
Wetland distance upstream[b]	RELWTPOS	# stream orders	1.3	0.7	0.0	2.6
Total watershed area	AREA	km^2	88.9	812	7.7	375.0
Lake area	LAKE	%W	5.9	5.3	0.0	21.8
Lake fringe area	LKFRG	%W	9.7	7.5	0.0	22.9
Stream fringe, agricultural	STAGRPFR	%S	18.9	8.8	0.0	33.3
Stream fringe, forested	STFORPFR	%S	5.1	3.3	0.0	12.3
Stream fringe, urban	STURBPFR	%S	5.8	5.2	0.0	17.5
Stream fringe, wetland	STWTLDFR	%S	29.3	13.2	4.5	61.5
Stream fringe, 1st order streams	STR1FR	%S	57.1	14.5	12.5	100
Stream fringe, 2nd order streams	STR2FR	%S	27.4	14.3	0	87.4
Stream fringe, 3rd order streams	STR3FR	%S	13.9	12.8	0	47
Stream fringe	STRFRG	%W	31.8	8.1	11.9	45.2
Sequential comparison index	SCI	# runs/# seq'l pixels	6.7	1.9	2.9	11.6

Average soil K-factor (erodibility)	KFCTR		0.3	0.05	0.2	0.4
Average soil surface permeability	SRFPRM	cm min^{-1}	3.0	0.42	2.4	3.8
Average available soil P	SOILP	Index (1–3)	2.5	0.58	1.0	3.0
Average soil pH	SLPH	S.U.	6.0	0.32	5.8	6.9
Average watershed slope	AVSLP	degrees	0.8	0.37	0.1	1.4
Maximum watershed slope	MAXSLP	degrees	6.3	2.2	2	11
Channel slope	CHNLSL	m/km	1.6	1.0	0.1	5.0
Average watershed elevation difference[b]	AVDIF	m	23.3	11.3	4.8	49.1
Maximum watershed elevation difference[b]	MAXDIF	m	53.9	26.1	13.0	109.0
Elongation ratio[c]	ELNG		0.8	0.20	0.4	1.4
Compactness ratio[d]	CMPCT		1.7	0.20	1.3	2.2

[a] %w = Data expressed as percentage of total watershed area. %s = data expressed as percentage of stream fringe area.
[b] Difference relative to gauging station.
[c] Ratio of diameter of a circle of equal area to the basin length.
[d] Ratio of perimeter of basin to circumference of circle of equal area.

Table 3. Water quality variables used to analyze the cumulative impact of wetland loss. N = number of site-years for which water quality data were available.

Description	Code	Units	N	Time-weighted Annual avgs	
				Mean	S.D.
PHYSICAL PARAMETERS					
Instantaneous flow	FLOW*	cfs	23	22	15
	LOGFLOW	log10 (cfs)	23	0.95	0.26
pH	PH	S.U.	24	7.88	0.30
Suspended solids, total	TSS	mg/L	32	54	92
	LOGTSS	log10 (mg/L)	32	1.3	0.45
Suspended solids, inorganic	TSIS	mg/L	15	19	12
	LOGTSIS	log10 (mg/L)	15	0.96	0.24
Suspended solids, volatile	SSVOL	mg/L	15	11	7
	LOGVSS	log10 (mg/L)	15	0.91	0.22
	FROPR	SSVOL/TSS	15	0.48	0.14
MAJOR IONS					
Total chloride	CL	mg/L	35	34	22
Specific conductivity (@ 25 C)	SPCOND	umhos/cm @ 25 C	23	524	157
	LGSPCND	log10 (umhos/cm)	23	2.68	0.14
NUTRIENTS					
Ammonia-nitrogen, dissolved or total	NH4	mg/L	28	0.34	0.17
	SQRTNH4	mg/L	28	0.54	0.56
	FRNH4	NH4/TN	14	0.13	0.04
Nitrate-nitrogen, dissolved or total	NO3	mg/L	17	0.45	0.51
	SQRTNO3	mg/L	17	0.56	0.32
Nitrite-nitrogen, dissolved or total	NO2	mg/L	8	0.024	0.009
Nitrate-N + nitrite-N	NOX	mg/L	14	0.7	0.6
	SQRTNOX	mg/L	14	0.7	0.3
	FRNOX	NOX/TN	14	0.28	0.2

Nitrogen, dissolved	DN	mg/L	11	2.4	1.4
	LOGDN	log10 (mg/L)	11	0.48	0.14
	FRDN	DN/TN	11	0.93	0.26
Nitrogen, dissolved organic	DON	mg/L	11	1.4	1.1
	FRDON	DON/TN	11	0.51	0.10
Nitrogen, total = TKN + NO3 + NO2	TN	mg/L	14	2.4	1.3
	LOGTN	log10 (mg/L)	14	0.43	0.17
Nitrogen, total Kjeldahl	TKN	mg/L	16	1.9	0.9
Nitrogen, total organic = TN − NH4 − NOX	TON	mg/L	13	1.7	1.0
	FRON	TON/TN	10	0.65	0.14
Orthophosphate-P, total or dissolved	OPO4	mg/L	11	0.073	0.071
	SQOPO4	mg/L	11	0.22	0.10
	FRSRP	OPO4/TP	9	0.41	0.30
Phosphorus, dissolved	DP	mg/L	8	0.3	0.32
	FRDP	DP/TP	8	0.55	0.24
Phosphorus, total	TP	mg/L	34	0.25	0.24
	SQRTTP	mg/L	34	0.45	0.20
OTHER					
Chemical oxygen demand	COD	mg/L	9	63	17
	LOGCOD	log10 (mg/L)	9	1.73	0.14
Fecal coliform	FCOL	colonies/100 ml	26	4523	1.7E + 04
	LOGFCOL	log10 (col/100 ml)	26	2.2	9.9
Organic carbon, total	TOC	mg/L	9	21	6
Total lead	PBTOT	ug/L	14	39	86
	LOGPB	log10 (ug/L)	14	0.89	0.24

* Nonweighted average.

watershed management organizations (Table 1). Sampling intensities ranged from routine monthly sampling (MPCA) to continuous or event-based sampling (Oberts 1981).

Seasonal and biennial or annual averages were calculated for all water quality parameters using non-weighted and time-weighted data. Where instantaneous flow measurements were made at the time of sample collection, flow-weighted averages were also computed. The USGS water year (October 1 through September 30) was used as the annual time unit. Where possible, we combined water quality data for the year of aerial photo coverage with the previous year's data to lessen the influence of year-to-year climatic variation on water quality.

On the basis of rankit plots and the Wilks–Shapiro statistic (STATISTIX 1987) for raw and transformed water quality variables, we transformed several variables using logarithmic ($\log_{10}[X + 1]$) or square root calculations to achieve a normal distribution of the variables (Table 3).

Water flow data

The potential effect of wetland loss on flood magnitude was estimated by applying empirical equations developed for southern Minnesota (Jacques & Lorenz 1988) to each site-year. These equations were based on empirical relationships between calculated flood magnitudes for given recurrence intervals (2, 5, 10, 25, 50, and 100-year) and selected watershed variables (drainage area, main channel slope, percent storage in lakes and wetlands, percent lake area, and mean annual runoff). The equations were derived from flow data collected at 149 southern Minnesota stream-gauging stations having 10 or more years of record. Equations derived for this region have the form:

$$Q_n = mA^i(St + 1)^j S^k R^p \qquad (2)$$

where Q_n = estimated peak flow (cfs) for n-year recurrence interval,
A = drainage area in square miles,
St = percentage of drainage area as lakes, ponds, and wetlands,
S = main channel slope in feet per mile,
R = mean annual runoff, and
m, i, j, k, p = empirically determined coefficients.

Statistical analyses

We performed a principal components analysis without rotation (Norusis 1988) to reduce the 33 initial watershed variables (Table 2) to a smaller number of principal components. To relate these watershed variables with water quality parameters, we performed stepwise multiple regression analyses using the principal components derived from the watershed variables as independent variables, and water quality variables as dependent variables (Norusis 1988). Principal components are ideal variables to use in multiple regression because they are mathematically uncorrelated and, hence, problems of multicollinearity of independent variables are reduced (Tatsuoka 1971). Independent variables were included in regression equations based on the magnitude of partial correlations with the dependent variable. Selected variables were then included or rejected on the basis of F-tests with criteria of $p < 0.05$ or $p < 0.10$, respectively (p. B-227, Norusis 1988). To reduce the probability that regression results would reflect spurious correlations, we restricted the number of independent variables included in equations so that the case-to-variable ratio was $\geqslant 5:1$.

Partial correlations 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.

Details of the methods used are in Johnston et al. (1988, 1989).

Results

Characteristics of the wetland mosaic

Wetlands constituted between 5 and 37% of the landscape in the 15 major watersheds studied (Fig. 2a). The majority of the wetlands were herbaceous, but shrub and forested wetlands constituted about a third of the wetland area in watersheds of the Anoka glacial outwash plain (i.e. COON, RICE, HRDW, CLRW). The percentage of each watershed in semipermanently to permanently flooded wetlands was relatively uniform among watersheds, averaging about 6% (Fig. 2b). However, the percentage of each watershed in seasonally flooded wetlands ranged from 4 to 27%. Temporarily flooded wetlands were uncommon. Agriculture was the most abundant land use bordering the wetlands studied, as well as the most abundant land use in the watersheds as a whole (Fig. 2c).

Of the five watersheds having sequential data (ELMC, NMO7, RILY, CLRW, and HRDW: Table 1), all but RILY experienced wetland losses over time. Watersheds with the largest short-term losses were CLRW, HRDW, and NMO7, which lost 1.0 to 1.3% of their wetland area per year. Wetland gains in the RILY watershed were small (+6.0 ha between 1968 and 1975).

Principal components analyses

All principal components with eigenvalues greater than 1.0 were retained in subsequent analyses. Thus, the 33 original watershed variables were reduced to eight principal components that still explained 86.5% of the variance (Table 4).

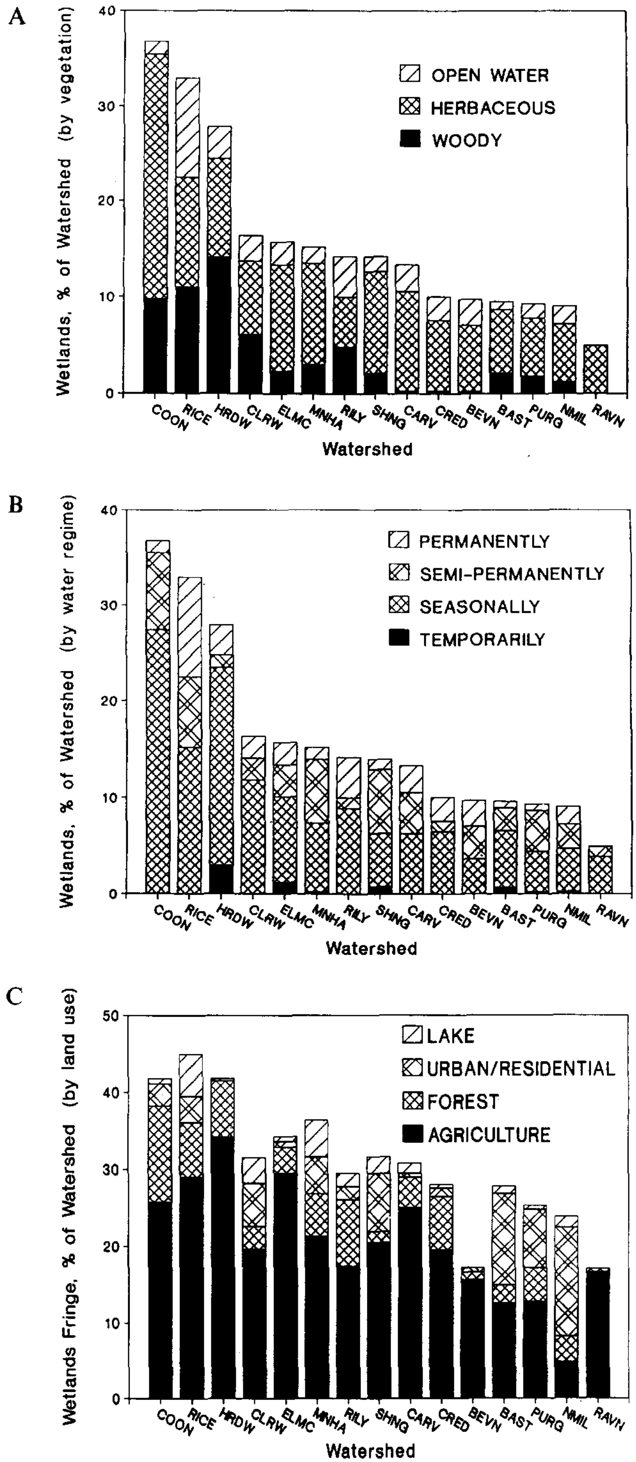

A
Wetlands, % of Watershed (by vegetation)
OPEN WATER
HERBACEOUS
WOODY
0
10
20
30
40
COON
RICE
HRDW
CLRW
ELMC
MNHA
RILY
SHNG
CARV
CRED
BEVN
BAST
PURG
NMIL
RAVN
Watershed
B
Wetlands, % of Watershed (by water regime)
PERMANENTLY
SEMI-PERMANENTLY
SEASONALLY
TEMPORARILY
Watershed
C
Wetlands Fringe, % of Watershed (by land use)
50
LAKE
URBAN/RESIDENTIAL
FOREST
AGRICULTURE
Watershed

Table 4. Variance of watershed variables explained by first eight principal components.

Rank	General interpretation of principal component	Eigenvalue	Variance explained (%)	Cumulative variance (%)
1	Wetland extent	9.8	29.7	29.7
2	Wetland proximity, watershed area	4.5	13.6	43.3
3	Agricultural/urban land use	4.1	12.6	55.9
4	3rd order streams, watershed diversity, soil pH	3.1	9.4	65.3
5	Forested stream fringe	2.5	7.7	73.0
6	Elongated headwater watersheds	2.1	6.3	79.3
7	Soil erodibility, forest	1.3	4.1	83.4
8	Herbaceous marsh extent	1.0	3.1	86.5

The communality, or fractional variance of the original variables explained by linear combinations of the eight factors was greater than 0.7 for all variables except for watershed area (0.64), so the watershed variables were well represented by principal components. If the communality had been low for some watershed variables, their effects would not have been represented by relationships between principal components and water-quality variables.

Interpretations of the principal components were based on the correlation coefficients ($p < 0.05$) between watershed variables and principal components (Table 5). Three of the principal components, PC1, PC2, and PC8, were associated with wetland variables. The first principal component (PC1), an indicator of wetland extent, was positively correlated with the relative area of wetlands, the dominant wetland types (e.g. herbaceous and woody wetlands), and stream fringe, and negatively correlated with watershed slope variables. Hence, watersheds with relatively high positive values for PC1 are relatively flat and have the highest proportion of wetlands per unit area. The unexpectedly high values for stream fringe were due to drainage ditches associated with flat watersheds having a high proportion of wetlands (e.g. Coon Creek watersheds).

The second principal component (PC2) was negatively correlated with relative wetland position (RELWTPOS), watershed area (AREA), average watershed elevation difference (AVDIF), and percent third order stream fringe (STR3FR), and positively correlated with landscape diversity (SCI) and soil phosphorus (SOILP; Table 5). Watersheds with low PC2 values had wetlands predominantly in the headwaters and far removed from the watershed outlet (e.g. BEVN, RAVN). These watersheds were generally large, included more third- and fourth-order streams, had relatively low soil P values, and had lower land use diversity (predominantly agricultural) than watersheds with high values for PC2. Because maximum stream order tended to be higher for larger watersheds, there was a tendency for large watersheds (> 15 000 ha) to have high

← *Fig. 2.* Wetland characteristics of major watersheds. 1980 site-year data used if available for entire watershed; most recent data used for other major watersheds. (*A*) Proportion of wetland in watershed, by vegetation type. (*B*) Proportion of wetland in watershed, by hydrologic type. (*C*) Proportion of wetland/upland fringe in watershed, by upland land use.

Table 5. Significant ($P < 0.05$) correlation coefficients between principal components and watershed variables.

		PC1	PC2	PC3	PC4	PC5	PC6	PC7	PC8
PC1	WTLD	0.91							
	STWTLDFR	0.89							
	HERB	0.85							
	HERBSF	0.80			0.34				
	WDYSF	0.80							
	AVSLP	−0.76							
	STRFRG	0.72		−0.34					
	MAXSLP	−0.71	−0.39						
	MAXDIF	−0.64	−0.46		0.50				
	CHNLSL	−0.62		−0.38		−0.41			
	*HERBSP	0.61							0.54
	*WFRG	0.56	0.47	0.45	0.42				
PC2	RELWTPOS	0.47	−0.68						
	AREA		−0.64	0.35					
	AVDIF	−0.57	−0.61		0.44				
	*SCI		0.59		0.51				
	SOILP	−0.59	0.56						
	*STR3FR		−0.54		0.56				
PC3	STURBPFR	−0.57		−0.68					
	WAGRFR	0.60		0.66					
	STAGRPFR		−0.42	0.64	−0.43				
	WURBFR	−0.50		−0.59		0.40			
	LAKE			0.58					
	WLKEFR		0.49	0.56		0.38			
	*CMPCT			0.42	0.36		−0.38	0.36	
PC4	SLPH		−0.50	0.36	−0.56				
	*STR3FR		−0.54		0.56				
	*SCI		0.59		0.51				
	*WFRG	0.56	0.47	0.45	0.42				
	*CMPCT			0.42	0.36		−0.38	0.36	
PC5	*STFORPFR					−0.66		0.43	
	*STR2FR					0.55	−0.60		
	*STR1FR					−0.52			
PC6	ELNG			0.38			0.64		
	*STR1FR			−0.34		−0.52	0.57		
	*STR2FR					0.55	−0.60		
	*CMPCT			0.42	0.36		−0.38	0.36	
PC7	KFCTR	−0.53		0.51				−0.50	
	*STFORPFR							0.43	
	SRFPRM	0.57		−0.44				0.41	
	*CMPCT			0.42	0.36		−0.38	0.36	
PC8	*HERBSP	0.61							0.54
	LKFRG	−0.38	0.46			0.39	−0.36		0.38

* Listed in more than one principal component grouping.

RELWTPOS values (1.93 to 2.59) and for small watersheds (< 5000 ha) to have low RELWTPOS values (< 1.0). However, those watersheds within the intermediate size range of 5000–15 000 ha represented a wide range of RELWTPOS values (0.43 to 2.32).

Only PC8 was associated with a single wetland type, the proportion of herbaceous marsh in the watershed (HERBSP; Table 5). This component was also associated with percent lake fringe (LKFRG), but not with percent lake area (LAKE). Site-years with high PC8 values (e.g. CO0180, BANB75) often had semi-permanent herbaceous wetlands associated with shallow lakes having irregular perimeters, and thus relatively large areas of lake fringe. Site-years 18 and 19 (HRDW80, HRDW87) had the lowest PC8 values, reflecting the relative lack of semi-permanently flooded wetlands in the HRDW watershed (Fig. 2b).

The other principal components were related primarily to upland watershed attributes. Relationships between these principal components and downstream water quality are discussed in another manuscript (Detenbeck et al., in preparation), so they are only briefly described here. Three principal components, PC3, PC5, and PC7, were highly correlated with land use variables. PC3 was positively correlated with the proportion of agriculture (WAGRFR, STAGRPFR) and negatively correlated with the proportion of urban land (WURBFR, STURBPFR) in the watershed. PC5 and PC7 represented the association of forested stream fringe area with headwater reaches and with certain soil types (Table 5). PC5 was positively associated with second-order stream fringe (STR2FR) and negatively associated with first-order (STR1FR) and forested (STFORPFR) stream fringe area. High values of PC7 represented watersheds with a high percentage of forested stream fringe (STFORPFR), high soil permeability (SRFPRM), and low soil erodibility (KFCTR).

The remaining two principal components, PC4 and PC6, were related to watershed morphometry. PC4 was positively correlated with percent third-order stream fringe, landscape diversity (SCI), and soil pH. PC6 represented a combination of watershed elongation and the predominance of headwater streams (STR1FR).

Plotting the first two principal component values for each site-year illustrated the differences among watersheds (Fig. 3). Site-years 10 (COON66), 11 (CO0180), and 12 (CO0280) had the highest PC1 values, reflecting the large proportion of wetland in the Coon Creek major watershed (Fig. 2a). Site-years with the highest PC2 values (24–26) were located in the MNLL watershed, in which wetlands tended to be concentrated near the sample site (i.e. low RELWTPOS). The wetlands in the elongated Bevens Creek watershed, however, (site-year 06: BEVN80), tended to be located far from the sample site (i.e. high RELWTPOS). Although PC2 was also related to watershed area (Table 5), the site-year with the largest watershed area (20: MNHA60) had an intermediate PC2 value, comparable to those of watersheds with much smaller areas (01: BAST66, 14: CRED80).

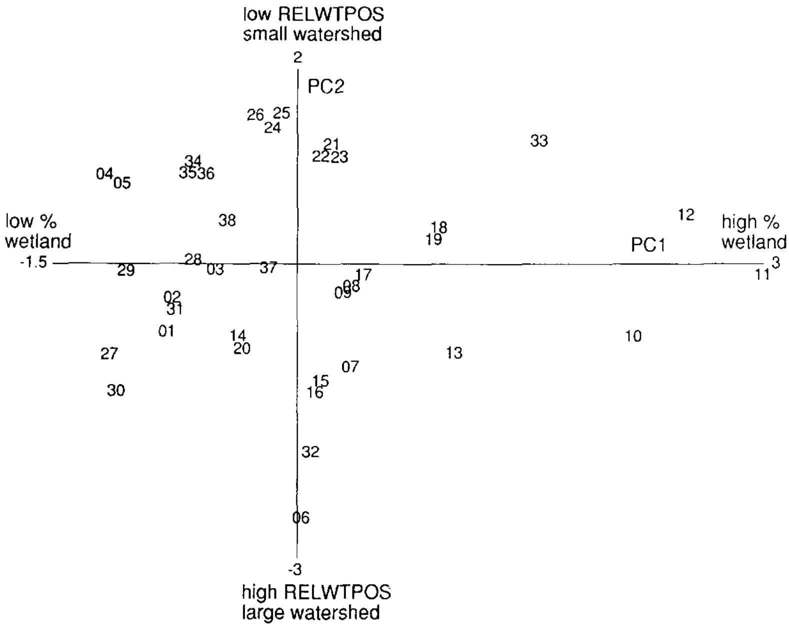


Fig. 3. Scores for principal component 1 vs. principal component 2, by site-year. Site-year codes are listed in Table 1. Principal components were computed for each site-year based on the 33 watershed variables listed in Table 3.

Effect of averaging technique on assessment of water quality function

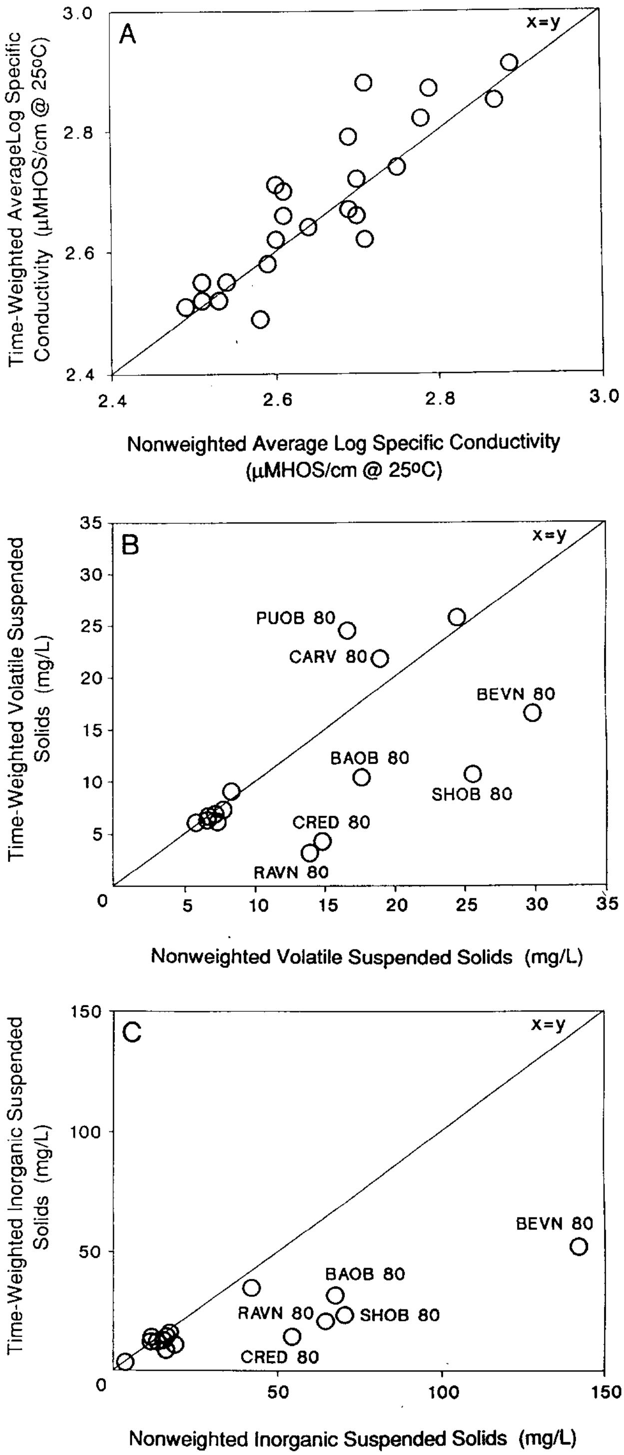
Water quality samples gathered monthly represent different time intervals than event-based samples, which are collected more during high flow than low flow periods. Therefore, non-weighted averages of routine monthly sampling can differ from those of event-based samples. Since our water quality data included both event-based and routinely sampled site-years, we used time-weighting to make the data more comparable.

Time-weighting had little effect on some variables, such as specific conductivity (Table 6, Fig. 4a). However, non-weighted averages would have overestimated suspended solids (Fig. 4b, c) because event-based sampling was concentrated around high-flow events associated with high loadings of suspended solids. The effect of time-weighting on average ammonium concentrations varied by watershed: non-weighted averages from event-based samples underestimated ammonium for the Coon Creek watersheds, in which ammonium values were highest for the infrequently sampled low-flow periods (Fig. 4d).

Where instantaneous flow measurements were made at the time of sampling, flow-weighted averages were computed to express the mean rate of export from watersheds. Flow rates affect the concentrations of some water quality variables, such as inorganic suspended solids, due to the relationship between water velocity and material transport (Morisawa 1968). Although most time-weighted and flow-weighted averages were highly correlated, flow-weighting produced

Table 6. Correlations between non-weighted (NW), time-weighted (TW), and flow-weighted (FW) averages of water quality parameters.

Variable	Pearson correlation coefficient		
	NW VS. TW	NW VS. FW	TW VS. FW
CL	0.77	0.67	0.61
COD	0.89	0.67	0.65
LOGCD	0.91		
DN	0.96	0.86	0.76
LOGDN	0.90		
DP	0.99	0.99	0.99
SPCOND	0.91	0.65	0.54
LGSPCND	0.89		
FCOL	0.99	0.97	0.79
LOGFCOL	0.98		
FRDN	0.92	0.83	0.69
FRDON	0.69	0.90	0.60
FRDP	0.99	0.86	0.95
FRNH4	0.46	0.26	0.25
FRNOX	0.93	0.76	0.55
SQFRNOX	0.94		
FRON	0.93	0.92	0.92
FRSRP	0.89	0.98	0.94
TKN	0.77	0.84	0.64
UNH4	0.95	0.97	0.85
NH4	0.75	0.81	0.57
SQRTNH4	0.76		
NO3	0.99	0.89	0.87
SQRTNO3	0.99		
NOX	0.91	0.89	0.65
SQRTNOX	0.92		
PBTOT	0.96	0.97	0.99
LOGPB	0.90		
PH	0.94	0.96	0.73
OPO4	0.98	0.91	0.95
SQROPO4	0.98		
SSTOT	0.93	0.71	0.52
LOGTSS	0.95		
TSIS	0.89	0.64	0.39
LOGTSIS	0.85		
TN	0.95	0.95	0.90
LOGTN	0.98		
TOC	0.82	0.24	−0.48
TON	0.96	0.95	0.97
TP	0.96	0.91	0.81
SQRTTP	0.95		
SSVOL	0.63	0.70	0.10
LOGVSS	0.84		
SQRTNO2	0.77		


A
x=y
Time-Weighted AverageLog Specific Conductivity (µMHOS/cm @ 25°C)
Nonweighted Average Log Specific Conductivity (µMHOS/cm @ 25°C)
B
x=y
PUOB 80
CARV 80
BEVN 80
BAOB 80
SHOB 80
CRED 80
RAVN 80
Time-Weighted Volatile Suspended Solids (mg/L)
Nonweighted Volatile Suspended Solids (mg/L)
C
x=y
BEVN 80
BAOB 80
RAVN 80
SHOB 80
CRED 80
Time-Weighted Inorganic Suspended Solids (mg/L)
Nonweighted Inorganic Suspended Solids (mg/L)

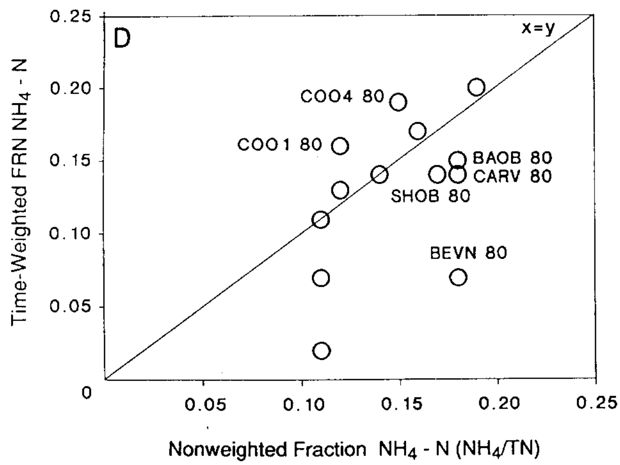


Fig. 4. Time-weighted vs. non-weighted annual averages for selected water quality parameters. Each circle represents a different site-year. The diagonal line plotted on each graph indicates where $x = y$ (i.e. non-weighted values equal time-weighted values). (*A*) Log specific conductivity (LGSPCND). (*B*) Volatile suspended solids (SSVOL). (*C*) Total inorganic suspended solids (TSIS). (*D*) Ratio of NH_4 to total N (FRNH4).

different average values than time-weighting for the following water quality variables: FRDON, FRNH4, FRNOX, NH4, TOC, SSTOT, TSIS, and SSVOL (Table 6).

Relationships between watershed and water quality variables

Stepwise multiple regressions were run separately for all annual and seasonal time-weighted or flow-weighted average water quality variables with $n > 5$. Although the principal components representing agricultural/urban land use (PC3) and forested stream fringe (PC5) were significantly related to some water quality variables (Detenbeck et al. in preparation), only those regressions which included PC1, PC2, or PC8 are shown in Tables 7 and 8.

Most time-weighted annual averages of water quality were significantly related ($p < 0.05$) to one or more of the wetland-related principal components (PC1, PC2, and PC8: Table 7). The wetland principal components were not significantly related to annual averages for FRDON, FRNH4, FRSRP, SQRTNH4, LOGVSS, and SQRTN02, but they did explain a significant amount of variation in seasonal averages. Explanatory variables retained in regressions often varied among seasons.

The sign of regression coefficients (Tables 7 and 8) indicates whether the principal component was associated with an increased or decreased value for the water quality parameter. In general, a negative coefficient indicates improved water quality (e.g. decreased nutrients or suspended solids concentrations),

Table 7. Results of stepwise multiple regressions between principal components and time-weighted water quality variables. Only those regressions which included PC1, PC2, or PC8 are shown.

Dependent variable	Season	Adjusted R^2	Std err	n	F	Constant b	Regression coefficients, Wetland-related components			Coefficients for other components retained
							PC1	PC2	PC8	
LOGTSS[b]	AN	0.38	0.37	32	10.3**	1.23	0.22			−0.2 PC3
LOGTSS[b]	SP	0.52	0.39	22	8.5**	1.32	0.17	−0.25		−0.28 PC3
LOGTSS[b]	SU	0.50	0.38	22	22.3**	1.36	0.31			
LOGTSS[b]	FA	0.42	0.48	21	8.2**	1	0.27			−0.21 PC3
LOGTSS[b]	WI	0.21	0.63	18	5.6*	1.2	0.3			
TSIS	AN	0.26	11	15	5.9*	14		−7.6		
SSVOL	SP	0.57	5.50	7	9.1*	12		8.26		
SSVOL	SU	0.63	11	7	11.3*	21		19		
FROPR	SP	0.30	0.16	15	6.9*	0.49		0.12		
TOC	AN	0.43	4.57	9	6.9*	21			8	
TOC	SU	0.46	6.70	8	7.1*	19			12	
LOGCOD	AN	0.49	0.10	9	8.6*	1.73			0.2	
LOGCOD	SP	0.86	0.06	9	17.8**	1.67			0.13	−0.09 PC5
LOGCOD	SU	0.74	0.11	8	21.0**	1.75			0.35	
LGSPCND	AN	0.55	0.09	23	14.5**	2.69	−0.06	−0.07		
LGSPCND	SP	0.45	0.11	23	9.9**	2.65	−0.07	−0.07		
LGSPCND	SU	0.44	0.12	21	8.8**	2.67	−0.08	−0.05		
LGSPCND	FA	0.49	0.11	17	8.6**	2.77	−0.07			−0.07 PC4
CL	AN	0.19	20	35	9.0**	34.54	−10			
Log10 (CL)[a]	SP	0.55	0.20	33	40.7**	1.38	−0.22			−6.0 PC3
CL	SU	0.17	16	32	7.3*	27	−7			−17.0 PC4
CL	FA	0.32	35	23	6.2**	30		−22		
LOGTN	SP	0.75	0.07	9	25.3**	0.44		−0.11		

LOGTN	WI	0.84	0.09	8	38.1**	0.57		−0.21		
TKN	AN	0.42	0.73	16	11.9**	1.7		−0.7		
TKN	SP	0.75	0.20	16	16.1**	1.63		−0.13		0.25 PC3, −0.16 PC5
TKN	WI	0.33	3.2	11	6.0*	2		−3		
TON	SP	0.83	0.28	14	22.5**	1.6	1		0.46	−0.2 PC3
TON	SU	0.36	1.06	11	6.7*	2.1			1.8	
LOGDN[c]	SU	0.70	0.10	6	12.8*	0.50			0.24	
DON	AN	0.73	0.63	10	24.8**	0.6		−1.2		−0.6 PC6
DON	SU	0.77	0.38	6	17.9*	0.8		−0.7		
DON	FA	0.68	0.40	6	11.5*	1.1	1.3			
FRDON	SU	0.50	0.06	9	8.9*	0.46	0.05			
SQRTNO3	AN	0.56	0.22	17	7.7**	0.66		−0.27		−0.17 PC4, 0.17 PC7
SQRTNOX	SP	0.67	0.20	19	13.3**	0.65		−0.11		−0.16 PC5, 0.14 PC7
SQRTNO3	SP	0.25	0.23	17	6.2*	0.46	−0.18			
SQRTNO3	SU	0.26	0.24	15	5.9*	0.39	−0.10			
FRDP	AN	0.82	0.11	7	27.6**	0.38		−0.25		
FRDP	FA	0.81	0.24	7	26.7**	0.3		−0.5		
SQOPO4	SU	0.88	0.05	6	37.9**	0.27	0.35			
SQOPO4	FA	0.88	0.02	6	37.9**	0.17	0.12			
FRSRP	FA	0.60	0.19	9	13.0**	0.79	−0.22			
LOGPB	AN	0.40	0.20	13	9.1*	0.98	−0.15			
LOGPB	SP	0.59	0.23	10	13.7**	1.10	−0.21			
LOGPB	WI	0.41	0.41	13	9.3*	1.1	−0.3			
LOGFCOL	AN	0.31	0.84	26	12.0**	2.1		−0.5		
LOGFCOL	SP	0.27	0.91	26	10.2**	2.3		−0.5		
FLOW	SU	0.26	7.37	13	5.2*	15		−8		

[a] Dependent variable log-transformed to stabilize variance.
[b] Relationship not significant ($p > 0.05$) when 1980 Coon Creek subwatersheds are removed from the analysis.
[c] Regression restricted to only one independent variable because of limited degrees of freedom.
* $p < 0.05$ **$p < 0.01$

Table 8. Results of stepwise multiple regressions between principal components and flow-weighted water quality variables. Only those regressions which included PC1, PC2, or PC8 are shown.

Independent variable	Season	Adjusted R^2	Std err	*n*	*F*	Constant b	Regression coefficients Wetland-related components			Coefficients for other components retained
							PC1	PC2	PC8	
SSTOT[b]	SU	0.63	91.00	8	13.0*	37	98			
TOC	SP	0.63	6.10	6	9.3*	39	−23			
LGSPCND	AN	0.35	0.11	17	9.7**	2.62	−0.08			
LGSPCND	SP	0.34	0.12	17	9.1**	2.60		−0.08		
LGSPCND	SU	0.55	0.14	13	8.4**	2.70	−0.13	−0.07		
LGSPCND	FA	0.92	0.05	10	51.3**	2.72		−0.09		0.09 PC3
CL	AN	0.35	24	23	7.0**	38	−17			11.0 PC4
CL	SP	0.34	14.00	20	11.0**	29	−11			
CL	SU	0.40	20.00	16	6.0*	35	−12	−13		
TN	AN	0.63	1.936	12	19.4**	2.8		−2.6		
TN	SP	0.56	2.11	12	15.2**	2.39		−2.5		
TKN	AN	0.46	1.20	16	13.9**	2.1		−1.3		
TON	AN	0.55	1.15	11	13.0**	1.5		−1.3		
FRDN	AN	0.66	0.21	10	18.5**	0.83	0.24			
DN	SP	0.62	0.84	7	11.0*	1.3		−1.1		
FRDN	SU	0.71	0	8	17.8**	0.8	0.32			
DON[a]	AN	0.74	0.37	10	13.6**	0.8		−0.6		−0.5 PC6
FRDON	SP	0.73	0.08	9	23.0**	0.72		0.15		
NOX	AN	0.71	1	11	13.5**	0.50		−1.4		
FRNOX	AN	0.78	0.12	12	39.7**	0.3	0.18			
NOX	SP	0.58	1.34	16	21.5**	1.2		−1.3		
NOX	SU	0.51	0.81	14	7.8**	0.6	0.9		−0.6	
FRNOX	SU	0.65	0.25	10	17.6**	0.33	0.27			

NO3	AN	0.40	0.25	11	7.7*	0.7		−0.4		
NH4	AN	0.73	0.13	18	24.1**	0.39		−0.13		0.13 PC3
NH4[a]	WI	0.79	0.23	11	20.1**	0.72		−0.24		0.29 PC3
TP	AN	0.43	0.29	23	18.2**	0.43		−0.25		
TP	SP	0.38	0.26	22	14.1**	0.34		−0.21		
FRDP	AN	0.53	0.16	8	8.9*	0.36		−0.18		
OPO4	FA	0.68	0.01	9	18.3**	0.031			−0.01	
FRSRP	FA	0.80	0.12	9	16.9**	0.67	−0.16		−0.07	
LOGFCOL	AN	0.39	0.90	13	7.9*	2.6		−0.6		
LOGFCOL	SP	0.68	0.69	10	17.8**	2.6		−0.7		
LOGFCOL	SU	0.71	0.76	9	15.7*	2.7		−0.9		
PH	AN	0.68	0.23	11	11.8**	7.9	−0.29		0.2	
PH[a]	SU	0.72	0.38	8	19.1**	8.0	−0.5			

[a] Number of independent variables restricted because of limited degrees of freedom.
[b] Regression not significant ($p > 0.05$) without Coon Creek subwatersheds.
* $p < 0.05$
** $p < 0.01$

while a positive coefficient indicates reduced water quality. For some parameters, however, an increased or decreased value may have a fairly neutral effect on water quality (e.g. specific conductivity).

With the exception of dissolved phosphorus, the regressions using flow-weighted averages had higher adjusted R^2 values than those using time-weighted averages (Tables 7 and 8). This probably represents both the greater effect of wetlands during periods of high flow, and the higher quality of data for watersheds with available flow data. Six of the water quality variables exhibited significant relationships whether expressed as time- or flow-weighted averages (LGSPCND, CL, TKN, DON, FRDP, and LOGFCOL).

Wetland extent (PC1) was associated with lower average values for the following time-weighted variables (Table 7): LGSPCND, CL, LOGPB, spring and summer SQRTN03, and autumn FRSRP. In contrast, PC1 was associated with higher time-weighted averages of LOGTSS, spring TON, fall DON, summer FRDON, and summer and fall SQOP04. For flow-weighted averages, PC1 was also associated with lower annual PH, lower spring TOC, and higher annual FRDN and FRNOX (Table 8).

PC2 was associated with lower annual DON, LGSPCND, TSIS, LOGFCOL, TKN, SQRTNO3 and FRDP, lower spring LOGTSS and SQRTNOX, lower spring and winter LOGTN, lower fall CL, and lower summer FLOW. In contrast, PC2 was associated with higher spring and summer SSVOL, and higher spring FROPR. For flow-weighted averages, PC2 was also associated with lower annual TN, NH4, NOX, TON, and TP, lower summer CL, lower spring DN, and higher spring FRDON.

The marsh component (PC8) was positively associated with time-weighted annual TOC, annual LOGCOD, spring and summer TON, and summer LOGDN, and negatively associated with flow-weighted summer NOX, fall OP04, and fall FRSRP.

Potential effect of wetland loss on flood magnitude

Regional USGS flow equations (Jacques & Lorenz 1988) were applied to watersheds across the metropolitan region to estimate the contribution of flow per unit area of watershed to the 100-year flood (Fig. 5). Watershed storage (i.e. proportion of watershed area in lakes and wetlands) ranged from 1.6 to 52.3%, with approximately half of the site-years having a total storage within the range of 10 to 20% of total watershed area. The overall range in percent storage was sufficient to account for an increase in flood peak of over two orders of magnitude. Most of the site-years considered, however, occurred within a region of the graph ($> 10\%$ storage) in which decreased storage caused only small increases in flood magnitude. Site-years with $\leqslant 10\%$ storage (i.e. BASL84, BANB75, RAVN80, and BEVN80) occurred within a region of the curve where the estimated flood magnitude increased rapidly with loss of storage.

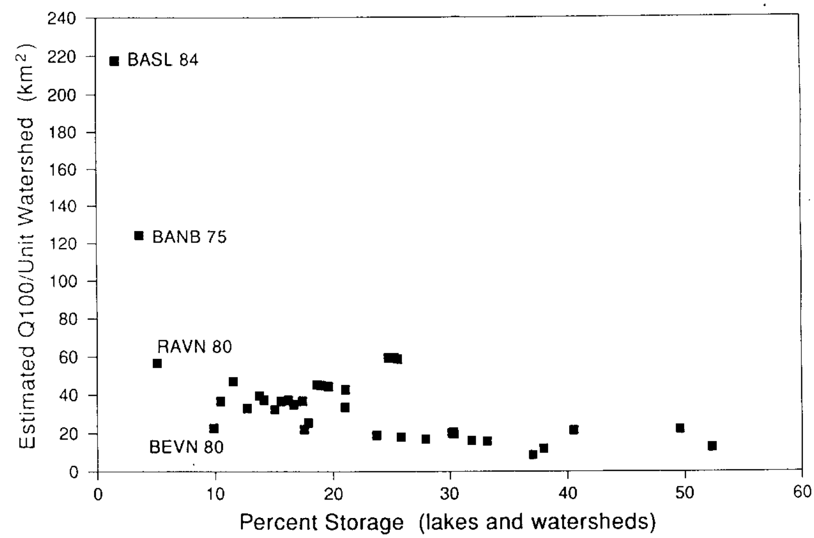


Fig. 5. Estimated contribution to 100-yr recurrence interval flood discharge (Q100) per unit watershed area, as a function of percent storage (lakes + wetlands). Site-years with the lowest percent storage are labeled.

Discussion

Effect of the wetland mosaic on water quality

The three wetland-related principal components, PC1, PC2, and PC8, were retained in a majority of the stepwise multiple regressions. Given the number of regression analyses performed, it is possible that some of these results are spurious. However, our results are generally consistent with wetland processes known to affect water quality in individual wetlands. The numerous significant relationships between water quality variables and these principal components are impressive in view of the myriad of ways in which surface water may interact with its watershed.

The second principal component (PC2) was most frequently selected in the stepwise multiple regressions, particularly those with flow-weighted water quality variables. The predominantly negative coefficients for PC2 (Tables 7 and 8) indicate that watersheds with the highest PC2 values (i.e. lowest RELWTPOS, smallest AREA) had the best water quality (i.e. the lowest concentrations of nutrients). Relating this principal component to watershed variables was complicated, however, because PC2 was related to both wetland (e.g. RELWTPOS) and non-wetland variables (e.g. AREA). The partial correlation analyses helped to distinguish some of these effects. Partial correlation indicated that time-weighted TKN (annual and spring), DON, and FRDP (fall) were primarily a function of watershed size, whereas time-weighted LGSPCND (annual and spring) was primarily a function of relative wetland position (RELWTPOS).

Partial correlation analysis did not reveal any significant ($p < 0.05$) independent relationship of either AREA or RELWTPOS with any other of the 45 water quality parameters explained by PC2 in the stepwise multiple regressions (Tables 7 and 8).

Particulates

The effect of wetlands on particulates differed with the principal component, time of year, and type of suspended solid. PC2 was related to lower annual concentrations of inorganic solids (TSIS), and lower spring LOGTSS. These effects may be due to sediment deposition, especially in seasonally flooded wetlands (Boto & Patrick 1979; Johnston et al. 1984).

In contrast to the trapping of inorganic suspended solids, PC2 was related to higher concentrations of volatile suspended solids (SSVOL) in the spring and summer. The marsh component (PC8) was also related to higher annual TOC and LOGCOD concentrations. Both of these findings imply that wetlands export organic matter. Studies of salt marshes have shown them to preferentially retain the more dense inorganic suspended solids, while exporting organic suspended solids on an annual basis (Settlemyre & Gardner 1977; DeLaune et al. 1979).

Although wetland extent (PC1) appeared to be related to *higher* annual levels of suspended solids, this effect was no longer significant when Coon Creek watersheds (CO01, CO02, CO04) were removed from the regression. Coon Creek watersheds have a high density of drainage ditches, which can increase loadings of suspended solids from streambank erosion (Brown 1988). Thus, the drainage of wetlands, rather than their presence *per se*, appears to be related to higher concentrations of suspended solids.

Fecal coliform

PC2 was significantly related to lower annual fecal coliform concentrations (LOGFCOL), similar to findings that wetlands receiving wastewater discharge decrease fecal coliform concentrations (Tilton & Kadlec 1979; Godfrey et al. 1985). Given that bacteria are usually associated with particulates (Stumm & Morgan 1981), this may be the result of sedimentation in wetlands. Since many of the microorganisms in fecal matter are not able to survive for long periods of time outside of their host organisms, the long particulate retention time in wetlands should promote natural bacterial die-off (Tilton & Kadlec 1979, Hemond & Benoit 1988).

Nitrogen

There were numerous significant relationships between wetland-related components and nitrate concentrations: wetland extent (PC1) was associated with lower time-weighted spring and summer SQRTNO3, and PC2 was associated with lower annual time-weighted SQRTNO3 and flow-weighted NO3 and NOX. Possible mechanisms for this effect include plant uptake or denitrification (Nixon & Lee 1985).

While the net effect of wetlands was to reduce nitrate concentrations, the finding that wetland extent (PC1) was related to an *increase* in annual proportion of flow-weighted soluble nitrogen (FRNOX, FRDN) and summer flow-weighted NOX indicates that nitrate may be flushed from wetlands during periods of high flow. This is consistent with Brown's (1985) observations that the efficiency of wetlands in removing nitrates is related to retention time. Individual wetlands with limited retention times were ineffective at removing nitrates or suspended solids during high flow events, and occasionally increased nitrate concentrations during spring storms. Wetlands could act as a source of dissolved nitrogen during flow events if dissolved organic N produced through decomposition and nitrates produced through nitrification of NH4 inputs were flushed out faster than mineralization or denitrification could proceed.

PC2 was related to lower annual flow-weighted ammonium averages (NH4). This is consistent with the work of Peterjohn & Correll (1984), who found that 89% of the N discharged from an agricultural field over one year was retained by a downslope riparian forest, most of it within 19 m of the cropland–forest border. Brinson and coworkers (1984) found ammonium retention during seasonal flooding of wetland forests, primarily due to interactions between the floodwaters and the forest floor. The sorption of ammonium by wetland soils would occur relatively quickly in comparison to the rate of denitrification, which is diffusion limited (Reddy et al. 1976).

Positive relationships between wetland-related principal components and some of the nitrogen forms were probably due to the solubilization of organic compounds during litter decomposition: PC1 with spring TON, summer FRDON and autumn DON; PC8 with spring and summer TON and summer LOGDN. Increased spring TON concentrations related to PC1 and PC8 were probably due to the flushing of plant litter broken down over winter. The positive relationship between PC8 and TON continued through the summer, possibly a result of slow *Typha* decomposition rates (Davis & van der Valk 1978).

High PC2 values were related to low concentrations of time-weighted DON and TKN, and flow-weighted TON and TN. This relationship was probably due to watershed area, however, rather than relative wetland position. AREA had significant partial correlations ($p < 0.05$) with annual time-weighted TKN ($r = 0.78$) and DON ($r = 0.75$), and spring time-weighted TKN ($r = 0.60$; PC3 and PC5 held constant). This implies that the larger a watershed, the more organic nitrogen it is likely to export. However, it is possible that proximal wetlands contributed to this relationship: long-term studies using soil dating techniques have found high rates of N accumulation in streamside wetlands, primarily in the form of organic N (DeLaune et al. 1978; Johnston et al. 1984).

Phosphorus

The ability of wetlands to retain phosphorus has been reported by a number of authors (summaries by Kadlec & Kadlec 1979; Whigham & Bayley 1979; Nixon & Lee 1985), and is a benefit of some wetlands used for wastewater treatment (Nichols 1983; Godfrey et al. 1985). Our results, however, showed both positive and negative relationships between phosphorus concentrations and wetland-related principal components. The marsh component (PC8) was related to lower flow-weighted soluble phosphorus concentrations (OPO4 and FRSRP) in the fall (Table 8), but wetland extent (PC1) was related to *higher* time-weighted summer and fall orthosphosphate concentrations (SQOP04) (Table 7). Although PC2 was related to a lower proportion of phosphorus in dissolved form (FRDP), this was probably due to the effects of watershed area, which had a significant partial correlation ($p < 0.05$) with fall time-weighted FRDP ($r = 0.76$) independent of RELWTPOS effects. However, proximal wetlands may have contributed to the relationship between PC2 and decreased TP (annual and spring flow-weighted) in connection with their sediment trapping properties. The bulk of TP is in particulate form, sorbed to fine silts and clays which are not efficiently trapped in upland riparian areas (Cooper et al. 1986), but which are trapped in floodplain wetlands (Mitsch et al. 1979; Johnston et al. 1984; Brinson et al. 1984; Yarbro et al. 1984; Whigham et al. 1986).

The higher summer and fall orthophosphate concentrations (SQOP04) associated with wetland extent may be due to the leaching of P from senescent wetland vegetation (Richardson et al. 1978; Davis & van der Valk 1978). While this process would not be expected to occur in early summer, Prentki et al. (1978) reported substantial P leaching from wetland vegetation in September, which was included in our 'summer' period.

Other water quality variables

Both PC1 and PC2 were related to lower annual and seasonal specific conductance, an indicator of the total concentration of ionized substances dissolved in water (American Public Health Association 1985). The mechanisms for these effects are not known, but because wetland extent reduced specific conductance throughout the ice-free period, a physical rather than biological mechanism is implied.

PC1 and PC2 were also related to lower chloride concentrations. Bayley et al. (1987) recently reported retention of chloride by wetland ecosystems in Ontario, even though chloride has generally been considered to be so ecologically conservative that it is commonly used as a tracer in mass balance studies. Alternatively, the decrease in chloride concentration could be a dilution effect if the wetlands are discharging groundwater low in chloride to the surface.

PC1 was related to lower annual lead concentrations and flow-weighted pH. Although the retentive capacity of wetlands varies for different metals and wetland types, wetlands are generally considered to be 'sinks' for metals (Oberts

1981; Giblin 1985). Retention mechanisms include precipitation of insoluble metal salts, sorption of metal ions, and uptake by wetland vegetation (Hemond & Benoit 1988). The relationship between wetland extent and pH was probably due to the production of humic acids by the wetlands (McKnight et al. 1985).

Effect of the wetland mosaic on flood storage

A number of authors have related the storage capacity of wetlands and lakes in the drainage basin to flood peak reduction (Novitzki 1979; Carter et al. 1979), and those principles have been applied in models for estimating flood magnitude (Ogawa & Male 1986). The relationship between basin storage (as percentage of basin area in wetlands and lakes) and relative flood flow is non-linear in the empirical models developed by Jacques & Lorenz (1988), so that our data yielded a critical threshold at about 10%. Small wetland losses in watersheds with $< 10\%$ wetlands could have a major effect on flood flows. A similar threshold was found for wetlands in Wisconsin watersheds by Novitzki (1979).

Methodological limitations

While the methods used proved fruitful in exploring relationships between wetland mosaics and water quality function, the large size of the area studied ($> 2073\,km^2$) prevented detailed landscape measurements. The grid resolution and mapping conventions used made it impossible to detect wetlands smaller than about 8 ha. This relatively coarse resolution could bias the results against the effects of small wetlands, which are more likely to be affected by development. However, when our wetland area data were compared with those of Oberts (1981) for the same watersheds, our results did not appear to underestimate total wetland area. Differences in the definition of wetland used in the two studies probably accounts for most of the discrepancies between the data sets.

While the use of water quality data from different sampling agencies and from different years could have biased our results, these biases probably were either negligible or reduced our chances of detecting significant effects of wetlands on water quality. The analysis of unfiltered water samples for dissolved inorganic nutrients probably overestimated the true concentration of ions in solution. However, it was not possible to correct for this source of error without having access to comparable data from both filtered and unfiltered samples. If anything, this source of bias would have caused us to underestimate the role of wetlands in improving water quality, because a greater proportion of the early data was derived from unfiltered water samples. Although some of the early water quality analyses may not have been corrected for color (not mentioned in most data reports), the lack of correction would have overestimated dissolved ions measured spectrophotometrically, therefore reducing the probability of detecting a water quality improvement function for wetlands. Differences in detection limits among different sampling agencies also could have biased our results, but because average water quality values were well above standard detection limits, these biases would have been small relative to the variation in water quality among watersheds.

While the use of principal components complicated interpretation of the results somewhat (e.g. for PC2), principal components analysis is a valuable exploratory tool for evaluating empirical relationships between watersheds and water quality. Principal components analysis reduced numerous possible watershed descriptors, many of which were correlated, to a few independent variables which could be related to water quality and flow. Partial correlation analysis was also useful in examining the contribution of individual variables represented by complex principal components. Principal components analysis is a well-known multivariate statistical technique that can be used to reduce the dimensionality of a complex problem and to explore the relationships among a large number of potential explanatory variables (e.g. Tatsuoka 1971; Ludwig & Reynolds 1988). We have effectively applied principal components analysis to a wide variety of scientific problems including bird-habitat relationships (Niemi & Hanowski 1984), animal morphology (Niemi 1985), and environmental chemistry (Niemi et al. 1987).

Conclusions

Cumulative impact assessment differs substantially from the approach used by existing wetland evaluation systems (Reppert et al. 1979; U.S. Army Corps of Engineers 1980; USFWS 1980; Adamus 1983) because it evaluates the collective function of a group of wetlands, rather than the contribution of an individual wetland. This landscape-scale, long-term approach has only recently been made possible by developments in ecological theory (Allen et al. 1984; Harris 1984; Forman & Godron 1986), methodology (i.e. multivariate statistical analysis), and equipment (i.e. Geographic Information Systems).

Our results indicate the importance of considering wetland position in the landscape when evaluating cumulative function. All wetlands in a watershed do not behave alike with regard to water quality function, which may explain why previous attempts to relate percent wetland to drainage basin water quality have generally been unsuccessful (Whigham & Chitterling 1988). Wetland extent (PC1) was related to decreased concentrations of only three of the time-weighted variables on an annual basis, none of which were nutrients: chloride, lead, and specific conductance. PC2, which was related to wetland proximity, helped to explain decreased concentrations of five annual time-weighted variables (LGSPCND, LOGFCOL, FRDP, SQRTNO3, and TSIS) and three additional flow-weighted variables (NH4, NOX, and TP). Therefore, the position of wetlands in the watershed appears to have a substantial effect on water quality, particularly with regard to sediment and nutrients.

There are several possible explanations for the importance of proximal wetlands to water quality. First, nutrients and sediments which have entered a stream system can only be affected by wetlands downstream of the source, so watersheds with wetlands concentrated upstream (i.e. high RELWTPOS value, low PC2 value) would have less effect on downstream pollution. Second, downstream wetlands may have different characteristics than headwater wetlands which would affect their nutrient retention capacity (e.g. mineral soils which have a greater P sorption capacity, longer duration of flooding which would increase sedimentation potential). Third, the effects of wetlands on water quality may only be detectable if they are close to the sampling station.

These findings do not necessarily mean that wetlands farther upstream from a sampling station are less important to water quality than proximal wetlands; just that their effects on nutrients are not detectable very far downstream, or are offset by downstream inputs. Further work is needed to determine distance relationships between wetlands and downstream water quality.

Wetland type was not distinguished by the principal components analysis, with the exception of the marsh component (PC8). All wetland classes were significantly correlated with PC1 (Table 5), and neither wetland vegetation nor water regime entered into PC2. The marsh component (PC8) explained variation in several water quality variables not explained by the other PCs, implying that differences in vegetation and/or water regime are functionally important. These relationships could be distinguished better by including watersheds with more diverse wetland properties in the analysis (e.g. watersheds with primarily forested wetlands, watersheds with primarily semi-permanently flooded wetlands).

One benefit of this approach is that it allows us to evaluate the cumulative effect of wetlands over an entire year, rather than just the growing season. The few studies which have monitored wetlands in temperate latitudes over all four seasons have shown that biotic nutrient uptake during the growing season is sometimes reversed by releases of nutrients during the dormant season, resulting in no net annual effect (Lee et al. 1975). Our use of annual or biennial averages made it possible to identify these net effects.

This landscape approach to the assessment of cumulative function will be facilitated in the future by the increasing availability of Geographic Information Systems and digitized maps. Wetland scientists and managers can use digital wetland maps from the National Wetlands Inventory, digital soil maps from the Soil Conservation Service, and digital topographic, land use, and stream maps from USGS to characterize watersheds, then statistically relate those watershed parameters to water quality and flow data summarized from the nationwide STORET database. By comparing data from many watersheds representing different degrees of impact, the incremental effect of additional impacts can be determined.

Those wetland managers without a GIS can still benefit from the relationships developed from this analysis, because the variables best correlated with the principal components may provide simple measures for analyzing cumulative impact. For example, our results suggest that watersheds with high PC2 values have a greater influence on nutrients and inorganic suspended solids. Relative wetland position (RELWTPOS) was highly correlated with PC2, and can be easily calculated (Eq. 1) without a GIS. Similarly, the proportion of wetland in a watershed, which may be determined from National Wetland Inventory data, may be a suitable substitute for wetland extent (PC1). While further work is needed to verify the applicability of these relationships to other regions, and to distinguish the effects of individual watershed variables, this type of analysis can provide the tools needed for accurate cumulative impact assessment.

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References

Adamus PR (1983) A method for wetland functional assessment. II. The method. Federal Highway Administration, U.S. Department of Transportation. Report No. FHWA-IP-82-24

Allen TFH, O'Neill RV & Hoekstra TW (1984) Interlevel relations in ecological research and management: some working principles from hierarchy theory. USDA Forest Service General Tech. Rep. RM-110, Fort Collins, CO

American Public Health Association (1985) Standard Methods for the Examination of Water and Wastewater, 16th edition. American Public Health Association, Washington, D.C.

Anderson, JP & Craig WJ (1984) Growing energy crops on Minesota's wetlands: the land use perspective. Center for Urban and Regional Affairs, University of Minnesota, Minneapolis

Arntson AD & Tornes LH (1985) Rainfall-runoff relationships and water-quality assessment of Coon Creek watershed, Anoka County, Minnesota. U.S. Geological Survey, Water-Resources Investigations Rep. 84-4013, St. Paul, MN

Bayley SE, Schindler DW & Turner MA (1987) Retention of chloride by terrestrial, wetland and lake ecosystems (abstract). American Society of Limnology and Oceanography Annual Meeting, Madison, WI, 14–18 June 1987

Boto KG & Patrick WH Jr (1979) Role of wetlands in the removal of suspended sediments. In: Greeson PE, Clark JR & Clark JE (Eds) Wetland Functions and Values: the State of our

Understanding (pp 479–489). Proc. Nat. Symp. on Wetlands, Lake Buena Vista, FL. Amer. Water Resources Assoc., Minneapolis, MN

Brinson MM, Bradshaw HD & Kane ES (1984) Nutrient assimilative capacity of an alluvial floodplain swamp. J. Appl. Ecol. 21: 1041–1057

Brown RG (1985) Effects of an urban wetland on sediment and nutrient loads in runoff. Wetlands 4: 147–158

— (1988) Effects of wetland channelization on runoff and loading. Wetlands 8: 123–133

Cairns J Jr, Albaugh DW, Busey F & Chanay MD (1968) The sequential comparison index – A simplified method for non-biologists to estimate relative differences in biological diversity in stream pollution studies. J. Water Pollu. Control Fed. 40: 1607–1613

Carter V, Bedinger MS, Novitzki RP & Wilen WO (1979) Water resources and wetlands. In: Greeson PE, Clark JR & Clark JE (Eds) Wetland Functions and Values: the State of our Understanding (pp 344–376). Proc. Nat. Symp. on Wetlands, Lake Buena Vista, FL. Amer. Water Resources Assoc. Minneapolis, MN

Cooper JR, Gilliam JW & Jacobs TC (1986) Riparian areas as a control of nonpoint pollutants. In: Correll DL (Ed) Watershed Research Perspectives (pp 166–192). Smithsonian Institution Press, Washington, DC

Council on Environmental Quality (1978) Recommendations for implementing the procedural provisions of the National Environmental Policy Act. 40 CFR, Sect. 1508.7

Cowardin LM, Carter V, Golet FC & LaRoe ET (1979) Classification of wetlands and deepwater habitats of the United States. Biological Services Program, U.S. Fish and Wildlife Service, FWS/OBS-79/31

Davis CB & van der Valk AG (1978) Litter decomposition in prairie glacial marshes. In: Good RE, Whigham DF & Simpson RL (Eds) Freshwater Wetlands (pp 89–113). Academic Press, NY

DeLaune RD, Buresh RJ & Patrick WH Jr (1979) Relationship of soil properties to standing crop biomass of *Spartina alterniflora* in a Louisiana marsh. Est. Coast. Mar. Sci. 8: 477–488

DeLaune RD, Patrick WH Jr & Buresh RJ (1978) Sedimentation rates determined by Cs-137 dating in a rapidly accreting salt marsh. Nature 275: 532–533

Forman RTT & Godron M (1986) Landscape Ecology. John Wiley & Sons, New York

Giblin AE (1985) Comparison of processing of elements by ecosystems. II. Metals. In: Godfrey PJ, Kaynor ER, Pelczarski S & Benforado J (Eds) Ecological Considerations in Wetlands Treatment of Municipal Wastewater (pp 158–179). Van Nostrand Reinhold, NY

Godfrey PJ, Kaynor ER, Pelczarski S & Benforado J (1985) Ecological Considerations in Wetlands Treatment of Municipal Wastewater. Van Nostrand Reinhold, NY

Gosselink JG & Lee LC (1987) Cumulative impact assessment in bottomland hardwood forests. Center for Wetland Resources Rep. LSU-CEI-86-09 Louisiana State University, Baton Rouge

Harris LD (1984) The Fragmented Forest. The University of Chicago Press, Chicago, IL

Hemond HF & Benoit RJ (1988) Cumulative impacts on water quality functions of wetlands. J. Environ. Manage. 12: 639–654

Hirsch A (1988) Regulatory context for cumulative impact research. J. Environ. Manage. 12: 715–724

Jacques JE & Lorenz DL (1988) Techniques for estimating the magnitude and frequency of floods of ungaged streams in Minesota. U.S. Geological Survey, Water-Resources Investigations Report 87-4170

Johnston CA, Bubenzer GD, Lee GB, Madison FW & McHenry JR (1984) Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. J. Environ. Qual. 13: 283–290

Johnston CA, Detenbeck NE, Bonde JP & Niemi GJ (1988) Geographic information systems for cumulative impact assessment. Photogrammetric Engineering and Remote Sensing 54: 1609–1615

Johnston CA, Detenbeck NE & Niemi GJ (1989) The cumulative effect of wetlands on stream water quality and quantity. Natural Resources Research Institute Technical Report NRRI/TR-89/1

Kadlec RH & Kadlec JA (1979) Wetlands and water quality. In: Greeson PE, Clark JR & Clark JE (Eds) Wetland Functions and Values: the State of our Understanding (pp 436–456). Proc. Nat. Symp. on Wetlands, Lake Buena Vista, FL. Amer. Water Resources Assoc., Minneapolis, MN

Lee GF, Bentley E & Amundson R (1975) Effects of marshes on water quality. In: Hasler AD (Ed)

Coupling of Land and Water Systems. Ecological Studies, Vol. 10 (pp 105–127). Springer-Verlag, NY
Ludwig JA & Reynolds JF (1988) Statistical Ecology. J. Wiley and Sons, New York, NY
McKnight D, Thurman EM, Wershaw RL & Hemond HF (1985) Biogeochemistry of aquatic humic substances in Thoreaus Bog, Concord, Massachusetts. Ecology 66: 1339–1352
Mitsch WJ, Dorge CC & Wienhoff JR (1979) Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. Ecology 60: 1116–1124
Morisawa M (1968) Streams: Their Dynamics and Morphology. McGraw-Hill, NY
Nichols DS (1983) Capacity of natural wetlands to remove nutrients from wastewater. J. Water Pollu. Contr. Fed. 55: 495–505
Niemi GJ (1985) Patterns of morphological evolution in bird genera of New World and Old World peatlands. Ecology 66: 1215–1228
Niemi GJ & Hanowski JM (1984) Relationships of breeding birds to habitat characteristics in logged areas of northern Minnesota. J. of Wildlife Manage. 48: 438–443
Niemi GJ, Veith GD, Regal RR & Vaischnav DD (1987) Structural features associated with persistent and degradable chemicals. Env. Toxicol. Chem. 6: 515–527
Nixon SW & Lee V (1985) Wetlands and water quality. U.S. Army Technical Report Y-85, Washington, DC
Norusis MJ (1988) SPSS/PC+ for the IBM PC/XT/AT. SPSS, Inc., Chicago, IL
Novitzki RP (1979) Hydrologic characteristics of Wisconsin's wetlands and their influence on floods, stream flow, and sediment. In: Greeson PE, Clark JR & Clark JE (Eds) Wetland Functions and Values: the State of our Understanding (pp 377–388). Proc. Nat. Symp. on Wetlands, Lake Buena Vista, FL. Amer. Water Resources Assoc., Minneapolis, MN
Oberts GL (1981) Impact of wetlands on watershed water quality. In: Richardson B (Ed) Selected Proceedings of the Midwest Conference on Wetland Values and Management (pp 231–226). Minnesota Water Planning Board, St. Paul
Ogawa H & Male JW (1986) Simulating the flood mitigation role of wetlands. J. Water Resources Planning and Management 112: 114–128
Omernik JM (1986) Ecoregions of the conterminous United States Map (scale 1:7 500 000). Corvallis, Oregon: U.S. Environmental Protection Agency, Corvallis Experimental Research Laboratory
Osborne LL & Wiley MJ (1988) Empirical relationships between land use/cover and stream water quality in an agricultural watershed. J. Environ. Mgmt. 26: 9–27
Owens T & Meyer M (1978) A wetlands survey of the Twin Cities 7-county metropolitan area – East half. IAFHE FSL Research Report 78-2. University of Minnesota, St. Paul, MN
Peterjohn WT & Correll DL (1984) Nutrient dynamics in an agricultural watershed: Observations of the role of a riparian forest. Ecology 65: 1466–1475
Prentki RT, Gustafson TD & Adams MS (1978) Nutrient movements in lakeshore marshes. In: Good RE, Whigham DF & Simpson RL (Eds) Freshwater Wetlands (pp 169–194). Academic Press, NY
Preston E & Bedford B (1988) Evaluating cumulative effects on wetland functions: a conceptual overview and generic framework. J. Environ. Manage. 12: 565–583
Reddy KR, Patrick WH Jr & Phillips RE (1976) Ammonium diffusion as a factor in nitrogen loss from flooded soils. Soil Sci. Soc. Amer. J. 40: 528–533
Reppert RT et al. (1979) Wetlands values: Concepts and methods for wetlands evaluation. IWR Research Report 79-R-1, U.S. Army Engineering Institute for Water Resources, Fort Belvoir, VA
Richardson CJ, Tilton DL, Kadlec JA, Chamie JMP & Wentz WA (1978) Nutrient dynamics of northern wetland ecosystems. In: Good RE, Whigham DF & Simpson RL (Eds) Freshwater Wetlands (pp 217–241). Academic Press, NY
Schnabel RR (1986) Nitrate concentrations in a small stream as affected by chemical and hydrologic interactions in the riparian zone. In: Correll DL (Ed) Watershed Research Perspectives (pp 263–282). Smithsonian Institution Press. Washington, DC
Settlemyre JL & Gardner LR (1977) Suspended sediment flux through a salt marsh drainage basin. Est. Coast. Mar. Sci. 5: 653–663

STATISTIX (1987) STATISTIX: An Interactive Statistical Analysis Program for Microcomputers. NH Analytical Software, St. Paul, MN

Stumm W & Morgan JJ (1981) Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters, 2nd edition. John Wiley and Sons, NY

Tatsuoka MM (1971) Multivariate Analysis: Techniques for Educational and Psychological Research. John Wiley & Sons, NY

Tilton DL & Kadlec RH (1979) The utilization of a freshwater wetland for nutrient removal from secondarily treated waste water effluent. J. Environ. Qual. 8: 328–334

Tiner RW Jr (1984) Wetlands of the United States: Current Status and Recent Trends. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C.

U.S. Army Corps of Engineers (1980) A Habitat Evaluation System for Water Resources Planning. USCOE, Lower Mississippi Valley Division

U.S. Fish and Wildlife Service (1980) Habitat Evaluation Procedures Workbook. USFWS, Western Energy and Land Use Team

U.S. Geological Survey (1978) National Handbook of Recommended Methods for Water-Data Acquisition. U.S.G.S. Office of Water Data Coordination, Reston, VA

Werth J, Meyer M & Brooks K (1977) A wetlands survey of the Twin Cities 7-county metropolitan area – West half. IAFHE RSL Research Report 77-10, University of Minnesota, St. Paul

Whigham DF & Bayley SE (1979) Nutrient dynamics in fresh water wetlands. In: Greeson PE, Clark JR & Clark JE (Eds) Wetland Functions and Values: the State of our Understanding (pp 468–478). Proc. Nat. Symp. on Wetlands, Lake Buena Vista, FL. Amer. Water Resources Assoc., Minneapolis, MN

Whigham DF & Chitterling C (1988) Impacts of freshwater wetlands on water quality: a landscape perspective. J. Environ. Manage. 12: 663–674

Williamson SC, Armour CL & Johnson RL (1986) Preparing a FWS cumulative impacts program: January 1985 workshop proceedings. U.S. Fish and Wildlife Service Biological Report 85(11.2)

Yarbro LA, Kuenzler EJ, Mulholland PJ & Sniffen RP (1984) Effects of stream channelization on exports of nitrogen and phosphorus from North Carolina coastal plain watersheds. Environ. Mgmt. 8: 151–160