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EFFECTIVENESS OF A COASTAL WETLAND IN REDUCING POLLUTION OF A LAURENTIAN GREAT LAKE: HYDROLOGY, SEDIMENT, AND NUTRIENTS

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Abstract: The ability of coastal wetlands of the Laurentian Great Lakes to reduce pollution from tributaries has not been documented in detail or over multiple seasons. This study developed a surface-water budget for a coastal wetland along Lake Erie and estimated monthly, annual, and storm-related exports of total suspended solids and selected nutrients from the wetland. Water-budget measurements included precipitation, evaporation, surface discharge into the wetland, and net surface discharge into Lake Erie. Water samples collected upstream and downstream and composite dryfall-precipitation samples were analyzed for total suspended solids (TSS), total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate + nitrite nitrogen (NO_{2+3}), ammonia nitrogen (NH_3), total Kjeldahl nitrogen (TKN), soluble reactive silica, chloride, and specific conductance. Seasonal and storm-related concentration patterns and a wide variation in monthly, seasonal, and annual loads from the tributary into the wetland were typical of streams draining the western Lake Erie basin. All substances reached higher maximum concentrations upstream than downstream; however, median monthly time-weighted mean concentrations of TP, TSS, NH_3, and TKN were higher downstream. Concentrations without discharge data were inadequate to estimate removal rates. Annual loads of TSS, NH_3, and TKN increased during passage through the wetland, whereas those of TP, SRP, NO_{2+3}, and soluble reactive silica decreased. During storm runoff events, various proportions of TP, SRP, TSS, NO_{2+3}, and soluble reactive silica were removed, despite brief hydraulic residence times, whereas more NH_3 exited than entered. Wetlands occupying the flooded lower reaches of Great Lakes tributaries collectively are probably important in maintaining and enhancing the water and sediment quality of the lakes. Water levels throughout the Laurentian Great Lakes have decreased in recent years; consequently, wetland areas with standing water and hydraulic residence times have decreased, probably reducing the effectiveness of the wetlands in mitigating pollution.

Key Words: nonpoint-source pollution, pollution abatement, Laurentian Great Lakes, Lake Erie, Old Woman Creek, coastal wetland, phosphorus, nitrogen, sediment, wetland hydrology, nutrient

INTRODUCTION

Wetlands of many types have been shown to be effective in processing sediment, nutrients, and organic pollutants entering them from tributaries (Kadlec and Knight 1996, Moustafa et al. 1998, Reddy et al. 1999). Similarly, coastal wetlands of the Laurentian Great Lakes probably play a beneficial role in reducing the concentrations and loads of materials entering the lakes from their watersheds. However, no detailed studies have been conducted to confirm that role.

The impact of pollutants on the Great Lakes could be mitigated in several ways: (1) pollutants entering the wetlands could be transformed through chemical, physical, or biological processes into less bioavailable or less toxic forms prior to entering the lakes; (2) pollutants could move into long-term sinks within the wetlands; and (3) pollutants could move into short-term storage within the wetlands, thereby altering the temporal patterns of pollutant loadings and concentrations entering the lakes. In the absence of wetlands that can intercept and process tributary water, the forms, amounts, and timing of pollutant inputs into the open waters of the Great Lakes and their bays directly match those of the outputs from the tributaries.

The degree of pollution mitigation that takes place within a wetland is likely to vary with stream discharge, from season to season, and from year to year, depending on local climatic conditions and the prevailing water level of the adjacent lake. Previous studies suggest that two Lake Erie wetland systems of greatly different sizes, Old Woman Creek wetland and Sandusky Bay, both seem to function in the ways listed above under different conditions (Richards and Baker 1985, Klarer and Millie 1989). However, those functions were measured indirectly on the basis of differences in pollutant concentrations between the upstream and downstream ends of the systems. The absence of discharge measurements at the lake interface of either system precluded the development of mass balance budgets.

The use of concentration differences alone to determine removal efficiencies is satisfactory only when the inflow and outflow volumes are equal and when the input concentrations are constant. Reliance on concentration data in the absence of discharge data can lead to large errors in estimates of the differences between the inputs of materials to, and their outputs from, wetlands. This is particularly true of coastal wetlands whose outlets to the lake are intermittently blocked by a barrier beach, as is the case with numerous tributaries along the southern shore of Lake Erie and parts of the other Laurentian Great Lakes.

Furthermore, during droughts, upstream inputs of pollutants yield only small quantities of materials at concentrations that have no direct relationship with their concentrations near the wetland outlet, either because export of preceding water masses is blocked by a barrier beach or because, in the absence of a barrier beach, lake water has become a major component of the downstream water mass via intermittent flow reversals. The inflow and mixing of lake water with wetland water results in a lowering of the concentrations of various chemical constituents of the water column. Thus, a quantitative understanding of the functioning of coastal wetlands with regard to pollutant processing requires not only information on concentrations but also detailed knowledge of the discharge into and out of the wetland.

The general hypothesis of this study was that under most hydrologic conditions, Lake Erie coastal tributary wetlands function as sinks and transformers of most materials and thereby substantially reduce the impact of those materials on the chemistry and biology of the receiving lake. The objective was to characterize the dynamics of total suspended solids and nutrients and the efficiency of pollution mitigation over a range of hydrologic conditions on the scales of years, seasons, months, and storm runoff events.

STUDY AREA

Old Woman Creek (OWC) wetland on the south shore of Lake Erie (Figure 1) was selected for study because it represents many coastal wetlands of Lake Erie. It is a relatively unmodified riverine-palustrine (Cowardin et al. 1979), drowned river mouth (Keough et al. 1999) wetland consisting of the flooded downstream channel and floodplain of the creek. During the period of field collections (October 1987–September 1990), the submerged area of the wetland varied from ~550,000 m^2 to 620,000 m^2, depending on the water level, and the volume from ~360,000 m^3 to 600,000 m^3.

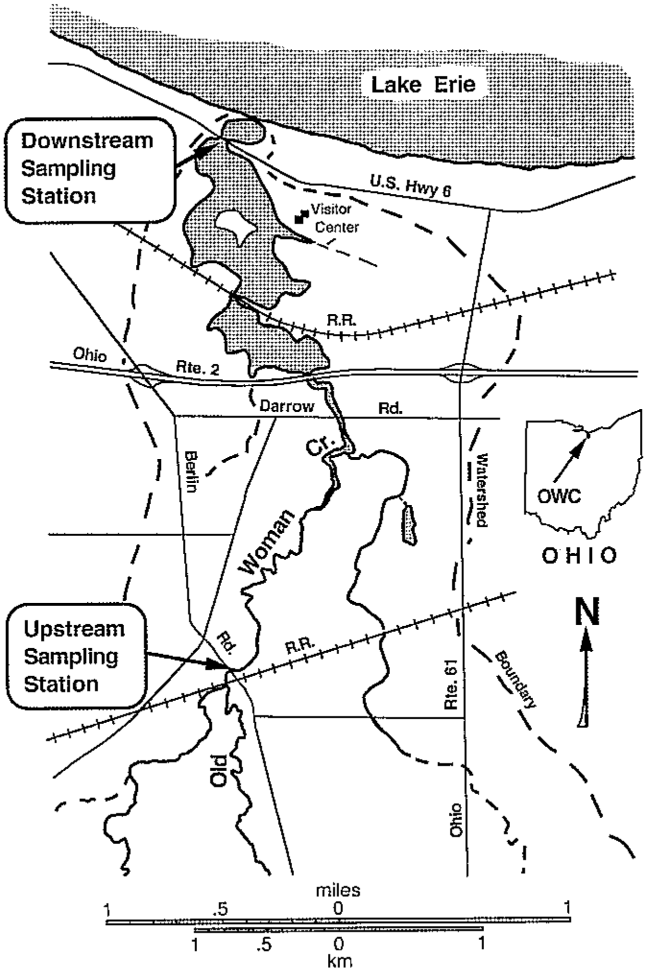

Figure 1. Lower part of the Old Woman Creek watershed, showing locations of the upstream and downstream sampling and gaging stations.

The OWC watershed overlies shale and sandstone bedrock formations. The modern soils are mixed with glacial till and glacial lacustrine deposits (Buchanan 1982). During the 1980s, land use in the watershed consisted of 51% cropland, primarily corn, soybeans, and wheat; 21% woodland; 16% grassland; 3% vineyards and orchards; and 10% other uses (e.g., water, urban areas, transportation). The only town in the watershed is Berlin Heights (pop. 800), situated about 9 km upstream from the wetland and served by private septic systems (Unpubl. data, Erie County SWCD).

Of the total watershed area of 68.9 km^2, 85% is drained by OWC, a second-order tributary. A first-order stream drains an additional 7.5%, and 6% of the watershed drains into the wetland via intermittent tributaries. The wetland surface accounts for 1.4% of the watershed area. An upstream water-level gage and water sampling station were situated on OWC where it drains 83% of the watershed about 1 km upstream from lake-level influences (Figure 1). A downstream water-level gage and water sampling station were situated near the confluence of OWC with Lake Erie. The creek and wetland open to the lake through a natural barrier beach. Storm runoff events create an open channel between the wetland and the lake, whereas periods of heavy surf close the barrier beach and prevent the surface exchange of water. In most years, the mouth of the wetland is closed during much of the summer and fall because low stream discharges are insufficient to maintain an opening in the beach. During periods when the beach is open, seiches and storm surges in Lake Erie cause constant but erratic oscillations in the water level of the wetland and often force lake water into its downstream end.

During the record high lake levels of the mid-1980s, immediately prior to this study, the depth of the wetland was as much as 1.5 m in many areas, with a maximum depth near 3 m beneath two bridges. Lake level controlled the wetland water level to ~3 km south of the lakeshore.

Aquatic macrophytes were dominated in turbid open-water areas by water lotus (*Nelumbo lutea* (Willd.) Pers.), which covered as much as one-third of the water surface in late summer (Whyte et al. 1997). A swamp dominated by an open overstory of ash (*Fraxinus* sp.) occupied much of the area south of the railroad causeway to Darrow Road (Figure 1) and was bordered on the north by a small sedge meadow (*Scirpus fluviatilis* (Torr.) Gray). The swamp floor and wetland margins were populated primarily by cattails (*Typha* spp.), sedges, reed (*Phragmites australis* Trin., (Cav.) ex Steudel), and arrowhead (*Sagittaria* spp.). Other than the swamp and sedge meadow, the wetland was characterized all year by open water with a mud bottom mostly devoid of emergent and submergent vegetation except for the water lotus in summer and fall.

METHODS

Hydrology

Precipitation and Evaporation. A recording precipitation gage and an evaporation pan (Belfort® recording pan followed by a Class A evaporation pan) were operated in an open field adjacent to the visitor center (Figure 1). Because evapotranspiration (ET) rates in summer are often several times greater than evaporation rates alone due to a complex of factors (Wetzel 1983, Maidment 1993), and pan evaporation coefficients may vary from less than 0.4 in summer to >1.3 in winter (Shaw 1988) as the result of biotic and abiotic interactions (Lott and Hunt 2001), the evaporation readings provided only a rough estimate of ET in the annual water budget. For comparison with pan evaporation, ET was estimated for two periods (July–October 1989 and June–July 1990) when the barrier beach was closed. This was done by subtracting the outflow volume, suddenly released when the beach opened, from the total tributary inflow during the period when it was closed. For each period, a monthly average ET was calculated from the difference in inflow and outflow volumes, and the results were applied to other months of the 1990 water year (1 October 1989–30 September 1990) by extrapolation and fractionally weighting the colder months. Those results were compared to total monthly pan evaporation from April through October of 1988–1990.

Discharge. A flow rating curve was developed for the upstream station (USGS Station 04199155) by the U.S. Geological Survey (USGS) using data from a pressure transducer. At the downstream station (USGS Station 04199165), a bubble gage operated by the USGS recorded changes in water level, which were used to determine direction of flow and volume of water exchanged at that station (Krieger 1993). Most discrete storm hydrographs appearing in the Berlin Road stage data from December 1987 through August 1990 during periods when the barrier beach was open were analyzed to determine residence time of the storm water in the wetland.

Chemistry

Water samples were collected at the upstream and downstream stations via submersible pumps suspended near the middle of the channel that pumped water continuously into a sink housed within each all-weather sampling building. An Isco® automatic water sampler pumped one sample every 8 h from the sink into a plastic bottle and maintained the samples at 4°C. The samples were retrieved weekly and were returned the same day to the laboratory for analysis. During low-flow periods, ≤ 1 sample/d was analyzed. During periods of high discharge, as many as 3 samples/d were analyzed in order to document concentration changes in greater detail. From April 1988 through September 1990, 1,019 upstream samples and 1,131 downstream samples were analyzed.

A stainless steel sink that drained into a large glass carboy was placed in an open field near the visitor center for collection of dryfall and precipitation. After each precipitation event, the water and particulates in the carboy were mixed and a portion was poured into a sample bottle. Composite dryfall-precipitation samples were analyzed identically to surface-water samples. All samples were analyzed for total suspended solids (TSS), total phosphorus (TP), soluble reactive phosphorus (SRP), nitrite + nitrate nitrogen (NO_{2+3}), ammonia nitrogen (NH_3), total Kjeldahl nitrogen (TKN), soluble reactive silica, chloride, and conductivity following methods prescribed by U.S. EPA (1979).

Data Analysis

Discharge. Upstream discharge (q_i) at every hour was obtained from the USGS rating curve. A rating curve could not be developed near the mouth of the wetland because of flow reversals and the absence of a relationship between stage and discharge. Therefore, the net volume of water exchanged between the wetland and the lake was calculated for each time interval of interest based on (1) the direction of change (up or down) in wetland elevation during the interval, (2) the volume of water exchanged, based on the stage-volume and stage-area relationships (Herdendorf and Hume 1991), and (3) the volume of water calculated to have entered the wetland via surface discharge from the watershed (Krieger 1993).

Precipitation and loads directly onto the surface of the wetland were calculated for several storm runoff events and were compared to the loads exported during the runoff events. Infiltration and seepage were excluded from the water-budget calculations because those processes are not sufficiently quantified at OWC wetland, although Matisoff and Eaker (1992) estimated that ground-water input varied from ~0 to ~0.7 m^3/s during and following the 1988 drought. During a low-water period in April 2000, an extensive visual survey indicated no seeps or springs outside the shallow channel running through the mudflats (D. M. Klarer, pers. commun., 5 April 2000).

Residence Time. Mean hydraulic residence time of water in the wetland was calculated as

$$\overline{RT} = \bar{V}/\bar{Q} \tag{1}$$

where $\bar{V} = (\Sigma v_i)/t$, $\bar{Q} = (\Sigma q_i)/t$, v_i is the wetland volume (m^3) at hour i, q_i is the discharge (m^3/s) for the entire upland watershed at hour i, and t is the total number of hourly readings during the storm hydrograph.

Mean Concentrations. The *simple mean* chemical concentration of a substance during a time period can be calculated from all samples collected during the period:

$$\bar{c} = (\Sigma c_i)/n \tag{2}$$

where c_i is the concentration in the i^{th} sample, and n is the number of samples.

Because samples were collected at unequal time intervals, *time-weighted mean concentrations* (TWMCs), rather than simple means, were calculated to provide a more accurate estimate of the mean concentrations during storm runoff events (Richards and Baker 1993):

$$TWMC = (\Sigma c_i t_i)/\Sigma t_i \tag{3}$$

where c_i is the concentration in the i^{th} sample, and t_i is the time represented by the i^{th} sample, computed as one-half the time between the i^{th} sample and the immediately preceding sample, plus one-half the time between the i^{th} sample and following the sample (Baker 1988).

Removal efficiency (*RE*) based on concentrations was calculated as

$$RE(\%) = \frac{\bar{c}_u - \bar{c}_d}{\bar{c}_u} \times 100 \tag{4}$$

where $\bar{c}_u$ and $\bar{c}_d$ are the upstream and downstream TWMC, respectively, during the period of interest.

Upstream Loads. Every analyzed water sample characterized chemical concentrations during a time interval ranging from 8 h to several days. For the time interval, t_i, represented by the i^{th} sample, the load (L) of each substance was calculated as

$$L = \Sigma c_i \bar{q}_i t_i \tag{5}$$

where c_i is the concentration in that sample; q_i is the mean discharge (m^3/s) computed from all the hourly stages included in the interval, and t_i is the same as in Eq. (3). The total load for the entire period of interest (e.g., month, storm, season) was obtained by summing the individual loads (L) for all samples during the period.

Downstream loads. During a *falling* stage interval, discharge through the mouth of the wetland was calculated as

$$D = D_{cr} + P - E + \Delta V \tag{6}$$

where D_{cr} is the discharge from the total watershed via the tributaries (m^3), which is the mean hourly discharge at the Berlin Road gaging station times a factor of 1.187 to account for unmeasured discharge from the remainder of the watershed (Krieger 1993); P is the volume of precipitation during the interval (m^3), which equals the measured precipitation (m) times the mean area (m^2) of the wetland during the interval; E is the evapotranspiration during the interval (m^3), which equals the evapotranspiration (m) times the mean area (m^2) of the wetland during the interval, with the beginning and ending areas determined from the hypsographic relationship (Herdendorf and Hume 1991); and $\Delta V = |V_b - V_e|$, where V_b is the volume of water in the wetland at the beginning of the interval and V_e is the volume at the end of the interval, with volumes

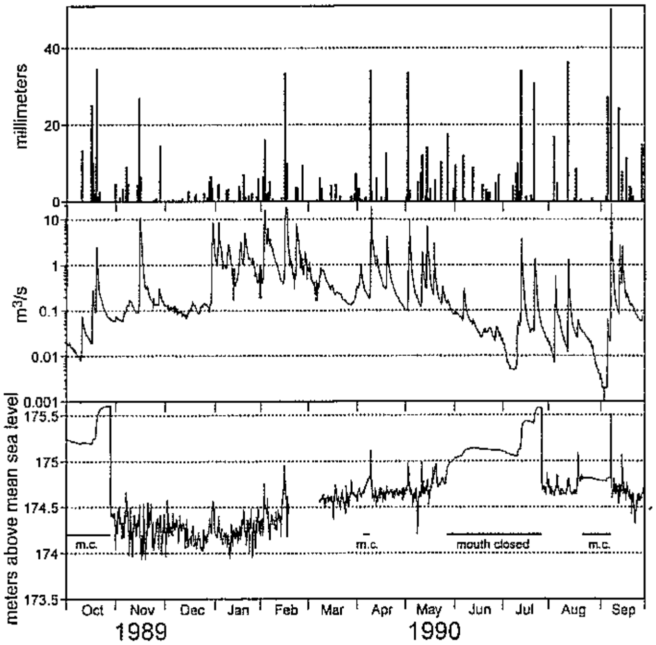

Figure 2. Precipitation at Old Woman Creek Wetland (upper graph), upstream discharge at Berlin Rd. (middle graph), and water level at the mouth of the wetland (lower graph), October 1989 through September 1990. Horizontal bars show periods when barrier beach closed the mouth (m.c.).

determined by the stage-volume relationship (Herdendorf and Hume 1991).

Because P and E were estimated to be equivalent to only ~2%-3% of annual surface discharge into the wetland (see below), both terms were always assigned a value of zero.

During a *rising* stage interval, discharge was calculated as

$$D = D_{cr} + P - E - \Delta V \qquad (7)$$

where P and E again were always assigned a value of zero.

Once the change in volume was determined for each rising and falling interval, chemical concentrations were assigned to the interval based on one or more samples taken during the interval. For rising intervals, samples were chosen that had a low conductivity (<35 mS/m), therefore representing the lake water chemistry (Krieger 1993). Thus, the load of materials from the lake into the wetland could be quantified. If no sample with a low conductivity occurred during the rising stage interval because of the location of the sampling station ~300 m from the lake, the low-conductivity sample nearest that interval was used. For falling intervals, all samples were used that had a conductivity >35 mS/m, thus representing wetland water (Krieger 1993). In computing both input and output loads for a given time period, it was necessary to calculate a total upstream load, via iterations and summation of Eq. 5, and a net downstream load, using both Eq. 6 and Eq. 7 as determined by the alternately rising and falling wetland stage.

RESULTS

Hydrology

A severe drought persisted from spring through fall of 1988. Almost all rainfall events through September 1988 were less than 20 mm and resulted in little or no increase in stream discharge. Total precipitation April–September 1988 was 278 mm, while total pan evaporation was 955 mm, averaging 7.3 mm/d in June and >4 mm/d the other months from April–August. Surface discharge into the wetland fell below 0.005 m^3/s in June 1988 and remained essentially zero through September. The drought ended in October 1988, and precipitation followed normal seasonal patterns most months through September 1990. Total precipitation in the 1989 water year was 766 mm, and was 881 mm in the 1990 water year (Figure 2). Monthly average pan evaporation both summers was ≤3.7 mm/d except 4.2 mm/d in July 1990. Discharge decreased to an annual minimum during summer both years, dropping below 0.01 m^3/s only briefly in late summer and fall 1989 and during two more-extended periods in summer 1990 (Figure 2). The period of greatest sustained discharges was approximately December–April in all three years.

The barrier beach intermittently opened and closed in response to hydrologic events in the watershed and Lake Erie. The beach was closed for about one-fourth to one-half of each year. While it was closed, the water level in the wetland rose or fell according to the balance between runoff from the watershed, direct precipitation onto the wetland, evapotranspiration, and seepage through the beach. The maximum height of the wetland above Lake Erie, ≤1.25 m, was determined by the height attained by the barrier beach. Beach height, in turn, was determined by the number and severity of high surf events while the beach was closed. Overtopping of the beach by a rising wetland water level led to the rapid erosion of the beach and a return within hours to water levels controlled by Lake Erie (e.g., October 1989, Figure 2).

Periods when the barrier beach was open were marked by rapid, sometimes large, oscillations in the water level of the wetland in response to seiches and storm surges on Lake Erie (Figure 2). The levels at those times reflected the average seasonal water levels characteristic of Lake Erie and varied from a minimum in winter to a maximum in summer.

The 21 storm runoff events during periods when the

Table 1. Hydrograph characteristics of 21 storm runoff events when the barrier beach was open.

Storm Began	Duration days	Maximum Discharge m^3/s	Upstream Flux 1000 m^3	Mean Wetland Volume 1000 m^3	Mean Residence Time days
15 Dec 87	1.79	5.85	358	116	0.58
24 Mar 88	3.58	13.22	1087	136	0.47
7 Apr 88	2.17	8.95	730	254	0.80
17 Apr 89	3.08	3.53	427	85	0.66
5 May 89	0.83	1.64	67	90	1.40
6 May 89	2.54	3.21	383	96	0.70
10 May 89	1.21	3.46	268	189	1.08
12 May 89	3.58	5.21	837	137	0.62
23 May 89	2.33	13.58	1015	157	0.39
30 May 89	2.08	11.23	622	151	0.61
3 Jun 89	2.54	19.91	1056	187	0.49
13 Jun 89	1.08	5.90	317	207	0.84
27 Jun 89	1.58	2.77	115	245	3.81
3 Jan 90	4.00	10.32	1330	20	0.07
9 Jan 90	3.38	3.32	628	14	0.08
17 Jan 90	2.92	3.86	490	21	0.11
12 May 90	2.08	2.23	182	167	1.87
15 May 90	1.67	8.37	524	153	0.52
20 May 90	1.71	3.69	210	240	2.33
14 Sep 90	1.63	3.21	210	128	1.11
16 Sep 90	2.00	3.04	214	188	2.01
Mean	2.28	6.50	527	142	0.98
Median	2.08	3.86	427	151	0.66

barrier beach was open from December 1987 through September 1990 ranged in duration upstream from 0.83 d, with a maximum discharge of 1.64 m^3/s and a flux of 67,000 m^3, to 4.0 d, with a maximum discharge of 10.32 m^3/s and a flux of 1,330,000 m^3 (Table 1). The greatest discharge was 19.91 m^3/s during a 2.54 d event.

Residence time during storm events ranged from 0.07 d to 3.81 d (Table 1) and was a function both of the water level in the wetland, as controlled by Lake Erie, and the volume of storm water delivered into the wetland. Residence time was negatively correlated with storm hydrograph duration ($r = -0.51$) and maximum discharge ($r = -0.44$).

Surface inflow accounted for 97.6% of the total input volume to the wetland in the 1990 water year, precipitation supplying the remaining 2.4% (Figure 3). Net surface outflow into Lake Erie accounted for 97.2% of water loss from the wetland, and <3% was lost via evapotranspiration and seepage through the barrier beach.

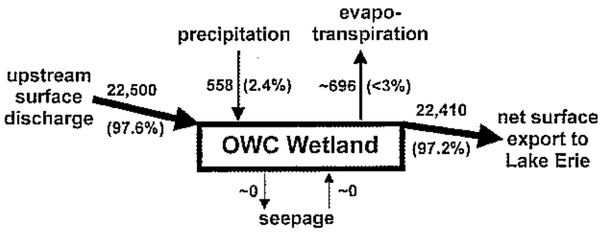

Figure 3. Water budget (1,000 m^3) for Old Woman Creek Wetland for the 1990 water year (October 1989 through September 1990).

Concentrations

Time-series graphs of conductivity and concentrations of TSS and nutrients from November 1988 through September 1990 are presented in Krieger (2001). Concentration patterns are presented here for the period May 1989 through September 1990 for comparison with loads calculated for the same period. All substances reached higher maximum concentrations upstream than downstream; however, median monthly TWMCs of TP, TSS, NH_3 and TKN were higher downstream (Table 2).

The monthly downstream/upstream ratio of TWMCs for each substance varied considerably (Table 3). TP concentrations were almost always higher downstream, with a TWMC ratio of 1.08 to 3.75 except 0.93 in September 1990. Conversely, SRP concentrations were always lower downstream, with ratios of 0.23 to 0.90. NO_{2+3} ratios were also always ≤ 1 (0.02 to 0.92) and displayed a seasonal pattern, with highest ratios November through May (Table 3). NH_3

Table 2. Maximum and minimum conductivity and concentrations recorded for each material April 1988 through September 1990. All units are mg/L except conductivity (mS/m).

	TP	SRP	TSS	NO_{2+3} N	NH_3 N	TKN	Conductivity	Silica	Cl
UPSTREAM (N = 1,019)									
Maximum	2.98	0.25	1,784	65.0	14.6	23.1	127.6	18.8	193.6
Minimum	ND[†]	ND	ND	ND	ND	ND	19.8	ND	6.0
Median Monthly TWMC	0.076	0.011	21.3	3.94	0.074	0.81	72.6	7.03	71.3
DOWNSTREAM (N = 1,131)									
Maximum	2.48	0.12	1,517	17.2	1.33	5.88	105.1	13.5	111.5
Minimum	ND	ND	3	ND	ND	0.19	24.0	ND	10.0
Median Monthly TWMC	0.175	0.005	73.1	0.96	0.152	1.48	51.5	3.92	42.0
DETECTION LIMIT*	0.015	0.015	0.0	0.075	0.076	0.111	0.09	0.028	0.015

* Richards and Baker 1985, Richards and Kramer 2002.

[†] ND, not detected.

ratios were always >1 (1.27 to 8.11) except when very low ratios (0.04 to 0.82) resulted from elevated concentrations upstream in the fall of 1989. TKN ratios also were >1 (1.19 to 2.97) except in the fall of 1989 (0.24 and 0.84). TKN ratios did not reflect the high ratios shown by NH_3, even though TKN measurements include NH_3.

Monthly downstream/upstream ratios of TSS were always >1 (1.02–5.88). Silica ratios were always <1 (0.07–0.94) except July 1989 (1.05). Chloride ratios were always <1 (0.33 to 0.97) and demonstrated a persistent decline from the highest ratios in winter to the lowest ratios in fall (Table 3).

Loads

Atmospheric Loads. The volume of precipitation falling on the surface of the wetland averaged only 1.04% of storm discharge volume, with a median of 0.26% for 14 storms in 1989 and 1990, expressed as the volume derived from the average area of the wetland during each precipitation event. The maximum and minimum equivalents of storm discharge were 4.84% in May 1990 and 0.04% during two events in January 1990.

Each measurement of contaminant concentrations in precipitation included particulate dry deposition since the time of the preceding sample as well as contaminants present in the rain or snow itself. The atmospheric loads for three storm periods, derived by multiplying sample concentrations by the volume of precipitation onto the wetland, and assuming a maximum area of 633,544 m^2, revealed highly variable inputs of nutrients and TSS (Table 4). Because the volume of precipitation was small compared to the volume of surface discharge during each storm period, and because the concentration of each contaminant (except NH_3) was small relative to its concentration in surface runoff water (Krieger 2001), the atmospheric loads to the wetland during storm events were negligible.

However, during some months of very low base flows, precipitation was equivalent to as much as 20.6% of the volume of creek discharge into the wetland (Table 5). During one or more of those months, the atmospheric loads of TP, SRP, TSS, NO_{2+3}, NH_3, and TKN exceeded 5% of the upstream load from the creek. The atmosphere was a major source of NH_3 and TKN to the wetland throughout the 1990 water year, but it was a minor source of soluble reactive silica and chloride. For the entire year, the volume of precipitation was 2.5% of the upstream discharge while NH_3 input was 6.8% of upstream loads, TKN 1.6%, and each of the other parameters <0.6% (Table 5). Because the atmospheric contributions were so low, only the loads from surface discharge are compared in the remaining presentation.

Monthly and Annual Surface Loads. The upstream and downstream fluxes of surface water varied greatly from month to month and lacked a seasonal pattern during the 18 months for which flux was computed (Table 6). However, during most months when upstream discharge was low, the barrier beach closed the mouth (Figure 2), thereby reducing surface discharge to Lake Erie to zero (Table 6).

Between April 1989 and September 1990, maximum upstream loads of specific materials into the wetland and their maximum net downstream loads into Lake Erie occurred either in May 1989 or January or February 1990 (Table 6). Removal efficiencies varied widely, ranging from 45.7% (net retention) for SRP to −91.8% (net export) for NH_3 (Table 7).

The calculated upstream and downstream surface fluxes of water and chloride in the 1990 water year

Table 3. Comparison of downstream/upstream ratios of time-weighted mean concentrations (TWMCs) and loads for individual months for which complete data are available. Months when downstream discharge was zero (barrier beach was closed) are excluded from mean ratios.

		TP	SRP	TSS	NO_{2+3} N	NH_3 N	TKN	SiO_2	Cl
May 89	TWMC	1.56	0.34	2.13	0.62	7.16	1.35	0.75	0.75
	Load	0.71	0.20	0.80	0.68	6.47	0.95	0.74	1.01
June 89	TWMC	1.08	0.38	1.46	0.35	8.11	1.31	0.52	0.50
	Load	0.72	0.32	0.70	0.62	8.43	1.07	0.92	0.96
July 89[1]	TWMC	2.40	0.71	1.02	0.12	4.26	2.15	1.05	0.55
	Load	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
August 89[1]	TWMC	3.75	0.52	3.75	0.14	4.03	2.97	0.53	0.50
	Load	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
September 89[1]	TWMC	2.74	0.51	3.20	0.02	0.04	0.24	0.67	0.42
	Load	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
October 89[2]	TWMC	2.11	0.41	3.55	0.09	0.17	0.84	0.67	0.47
	Load	2.68	0.41	2.80	0.41	1.29	1.78	2.27	1.37
November 89	TWMC	2.13	0.46	5.88	0.63	0.82	1.35	0.84	0.67
	Load	0.51	0.14	1.88	0.93	1.91	1.48	1.07	1.31
January 90	TWMC[6]	1.39	0.62	2.70	0.92	1.27	1.19	0.94	0.97
	Load	0.98	0.61	1.50	0.74	2.31	1.03	0.79	0.82
April 90[3]	TWMC	2.76	0.90	3.80	0.75	1.50	1.50	0.94	0.77
	Load	1.31	0.74	1.64	0.82	3.42	1.28	0.96	1.01
May 90[4]	TWMC	1.34	0.44	1.78	0.51	1.39	1.24	0.56	0.53
	Load	0.70	0.35	0.67	0.59	1.46	0.93	0.74	0.89
June 90[1]	TWMC	3.67	0.23	3.91	0.16	2.33	2.22	0.19	0.55
	Load	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
July 90[2]	TWMC	2.61	0.30	2.70	0.13	3.11	2.24	0.19	0.55
	Load	0.74	0.13	0.50	0.15	1.31	1.13	0.24	0.94
August 90[4]	TWMC	1.46	0.29	1.80	0.25	1.33	1.45	0.33	0.33
	Load	1.85	1.19	1.34	0.12	11.39	2.46	0.93	0.90
September 90[2]	TWMC	0.93	0.28	1.74	0.29	1.32	1.28	0.51	0.41
	Load	0.55	0.32	0.41	0.48	3.36	0.77	0.86	1.00
Mean TWMC Ratio[5]		1.74*	0.44	2.75**	0.45	2.62	1.38	0.63	0.60***
Mean Flux Ratio[5]		1.07*	0.44	1.22**	0.55	4.14	1.29	0.95	1.02***

[1] Barrier beach was closed the entire month.

[2] Barrier beach was closed at the beginning of the month but open at month's end.

[3] Barrier beach was open at the beginning and end of the month but closed for some period in between.

[4] Barrier beach was open at the beginning of the month but closed before the end of the month.

[5] Mean TWMC ratios and load ratios exclude months when downstream flux was zero.

[6] TWMC was calculated from 3 January 1990.

* $p < 0.05$.

** $p < 0.01$.

*** $p < 0.001$.

Table 4. Volume of precipitation (m^3) and atmospheric loads (kg) of total suspended solids and nutrients to OWC Wetland during three storm runoff periods. Each load is also expressed as a percentage of the storm runoff load into the wetland during the period.

	Precipitation	TP	SRP	TSS	NO_{2+3} N	NH_3 N	TKN	SiO_2	Cl
Three Storms, 23 May–7 Jun 1989									
Atmospheric Load	46,990.0	3.41	0.300	467	31.5	24.8	57.2	0.00	15.1
% Runoff Load	1.0	0.12	0.26	0.023	0.15	10.8	0.45	0.00	0.017
Two Storms, 17–24 Jan 1990									
Atmospheric Load	10,295	0.22	0.028	170	12.9	5.3	7.0	0.00	18.3
% Runoff Load	0.68	0.13	0.30	0.33	0.11	2.5	0.46	0.00	0.019
Two Storms, 12–18 May 1990									
Atmospheric Load	27,838	1.72	0.124	237	44.5	17.3	36.2	30.9	105.1
% Runoff Load	3.3	0.62	2.2	0.058	0.61	5.7	1.9	0.49	0.41

Table 5. Volume of precipitation (m^3) and atmospheric loads (kg) of total suspended solids and nutrients to OWC Wetland during each month of the 1990 water year.

	Precip., mm	Precip., m^3	TP	SRP	TSS	NO_{2+3} N	NH_3 N	TKN	SiO_2	Cl
Oct	93.47	59,217	1.940	0.727	60.5	45.5	27.77	52.73	0.24	26.2
Nov	71.12	45,058	2.807	0.292	1,116.7	41.4	11.61	38.81	3.15	31.6
Dec	25.14	15,927								
Jan	35.55	22,522	2.135	0.036	851.9	29.3	11.59	28.47	0.35	27.9
Feb	93.23	59,065	1.407	0.114	744.3	61.7	27.73	46.12	0.64	40.7
Mar	21.59	13,678	0.435	0.018	179.9	35.6	14.81	25.54	0.12	30.9
Apr	76.98	48,770	0.721	0.100	169.6	52.8	31.78	60.25	0.00	49.6
May	100.83	63,880	3.377	0.159	513.9	113.9	30.83	71.70	78.53	259.7
Jun	57.92	36,695	2.128	0.00	306.0	41.8	13.68	49.05	0.00	15.0
Jul	91.96	58,261	2.308	0.888	38.0	32.6	16.45	40.72	0.00	13.5
Aug	67.05	42,479	0.870	0.00	28.8	36.7	8.89	21.70	0.00	0.0
Sep	146.32	92,700	2.999	0.338	41.4	55.0	13.57	44.48	0.00	39.9
Total	881.16	558,254	21.126	2.672	4,051.0	546.1	208.73	479.56	83.04	535.0

Table 6. Monthly net surface fluxes of water (1,000 m^3) and loads of nutrients and suspended solids (kg) into Old Woman Creek Wetland from the watershed (Up) and into Lake Erie from the wetland (Down).

Month		Water	TP	SRP	TSS	NO_{2+3} N	NH_3 N	TKN	Silica	Chloride
May '89	Up	5,274	2,252	71.5	1,483,653	27,773	222	11,733	35,588	138,316
	Down	5,181	1,588	14.4	1,192,106	18,975	1,436	11,177	26,197	139,395
June '89	Up	2,456	1,147	81.5	899,122	12,246	138	5,504	19,964	68,584
	Down	2,419	822	25.8	627,525	7,614	1,164	5,909	18,327	65,976
July '89	Up	402	128	14.1	129,033	1,207	22	983	3,807	13,024
	Down	0	0	0	0	0	0	0	0	0
August '89	Up	95	6	0.7	1,225	142	3	57	992	7,021
	Down	0	0	0	0	0	0	0	0	0
September '89	Up	113	10	2.6	2,825	540	453	838	811	11,297
	Down	0	0	0	0	0	0	0	0	0
October '89	Up	400	68	12.4	17,587	1,893	321	775	3,184	25,457
	Down	653	181	5.1	49,306	781	413	1,381	7,231	34,762
November '89	Up	1,575	1,318	58.7	220,530	9,920	305	2,946	13,211	70,622
	Down	1,572	667	7.9	415,681	9,184	583	4,353	14,140	92,430
December '89	Up	346	12	4.0	1,113	1,307	202	358	2,580	30,268
	Down	357	13	8.1	−7,621	2,192	339	534	3,156	47,062
January '90	Up	6,432	647	92.5	211,742	47,319	849	5,626	49,482	372,979
	Down	5,521	633	56.3	317,648	34,918	1,961	5,799	38,943	307,219
February '90[1]	Up	5,060	915	143.1	481,199	25,541	323	5,582	30,722	188,664
	Down	6,377	1,330	111.8	800,307	31,410	373	7,674	38,032	264,140
March '90[1]	Up	520	13	1.1	4,562	2,156	88	362	2,168	28,841
	Down	415	66	1.7	56,135	1,262	93	531	1,484	20,279
April '90	Up	3,205	694	56.8	372,967	17,177	241	4,612	19,002	103,188
	Down	3,292	910	42.0	610,486	14,018	825	5,884	18,324	104,306
May '90	Up	2,160	962	25.1	683,901	15,004	651	4,848	13,536	69,684
	Down	2,078	672	8.7	459,327	8,906	952	4,502	9,968	62,179
June '90	Up	187	14	2.3	4,376	577	10	136	935	12,228
	Down	0	0	0	0	0	0	0	0	0
July '90	Up	503	118	17.9	59,775	7,626	141	805	4,545	24,491
	Down	498	88	2.3	29,798	1,127	185	913	1,074	22,979
August '90	Up	206	43	6.1	19,612	436	13	239	1,416	10,119
	Down	132	79	7.2	26,321	51	145	588	1,310	9,114
September '90	Up	1,891	1,009	101.2	622,861	7,777	111	3,834	13,323	37,409
	Down	1,518	558	32.0	252,474	3,761	374	2,934	11,459	37,241

[1] No loading data 17 February through 11 March 1990.

Table 7. Annual loads* and removal efficiencies at Old Woman Creek Wetland (October 1989 through September 1990). Values are kg except water (m^3).

Material	Load to Wetland	Load to Lake Erie	Retained in Wetland (%)
water	22,500,000	22,410,000	0.3
Cl	974,000	1,002,000	−2.9
SRP	521	283	45.7
TP	5,813	3,867	33.5
NO_{2+3} N	136,700	107,600	21.3
SiO_2	154,100	145,100	5.8
TSS	2,700,000	3,010,000	−11.5
TKN	30,120	35,090	−16.5
NH_3 N	3,255	6,243	−91.8

* Data lacking for 17 February–11 March 1990; thus, values represent 0.937 year.

were almost identical (Table 7). Other substances, with the exception of silica, showed either a sizable gain or loss within the wetland. The loads of both TP and SRP were reduced as they passed through the wetland, with only 67% of TP and 54% of SRP exiting into Lake Erie. Likewise, only 79% of NO_{2+3} left the wetland, but 94% of soluble reactive silica was exported. By contrast, more TSS (12%), NH_3 (92%), and TKN (17%) left the wetland than entered (Table 7). However, only 86% of the total N, estimated as NO_{2+3} N plus TKN, left the wetland; the total N load was strongly influenced by the amount of NO_{2+3} N because TKN contributed only 18% of the total N upstream and 25% downstream.

The upstream loads (N = 18) of NH_3 were not significantly correlated with the downstream loads (N = 18) of most of the other substances but were highly correlated with the downstream loads of NO_{2+3} ($r = 0.64$, $P = 0.004$), NH_3 ($r = 0.67$, $P = 0.002$), and chloride ($r = 0.65$, $P = 0.003$), and slightly less correlated with silica ($r = 0.58$, $P = 0.011$). The upstream loads of TP and SRP were highly correlated ($r = 0.70$, $P = 0.001$). TP upstream was highly correlated with TP downstream ($r = 0.95$, $P < 0.001$), but did not show a significant relationship with SRP downstream ($r = 0.36$, $P = 0.140$). TP downstream was relatively weakly correlated, though still significantly, with SRP downstream ($r = 0.57$, $P = 0.018$).

TP upstream was strongly correlated with both TSS ($r = 0.92$, $P < 0.001$) and TKN ($r = 0.93$, $P < 0.001$) upstream, and TP downstream was even more strongly correlated with TSS and TKN downstream ($r = 0.99$ and $P < 0.001$ for both). TSS also showed a highly significant correlation with TKN upstream ($r = 0.93$, $P < 0.001$) and downstream ($r = 0.98$, $P < 0.001$). NO_{2+3} was most strongly correlated with silica ($r = 0.93$, $P < 0.001$ upstream; $r = 0.97$, $P < 0.001$ downstream) and chloride ($r = 0.94$, $P < 0.001$ upstream; $r = 0.99$, $P < 0.001$ downstream), and silica and chloride were strongly correlated with each other ($r = 0.94$, $P < 0.001$ upstream; $r = 0.96$, $P < 0.001$ downstream).

Surface Loads During Storm Events. To determine the effects of storm runoff events on pollution mitigation by the wetland, four storm periods were investigated. In all four periods, the calculated fluxes of water into the wetland and Lake Erie were essentially equal (Table 8). However, despite very short residence times (Table 1), large proportions of most materials were lost during transit through the wetland (Table 8). The storm period in January 1990 was different from the other three in that there was a net export of most materials. However, in all four storm periods, there was a net retention of SRP, NO_{2+3}, and soluble reactive silica. Two periods showed net retention of chloride, and one showed net export. TP and TSS showed substantial net retention in three periods but net export during the January 1990 storms. NH_3 was exported into Lake Erie in greater quantities than were input to the wetland during three of the four periods, but the pattern did not mirror that of TP and TSS (Table 8).

DISCUSSION

Concentrations versus Loads for Estimating Removal Efficiencies

Concentration data alone are insufficient to calculate the removal efficiencies of coastal wetlands, partly because inflow volumes often do not equal outflow volumes. In addition, water masses entering a wetland may be subject to long detention times before exiting the wetland so that input and output concentrations measured simultaneously do not represent the same water mass (Kadlec and Knight 1996).

Particularly important in comparing the two approaches is the difference in the ratios of chloride. As a conservative ion, chloride is expected to maintain its concentration relatively unchanged as a water mass moves through the wetland (Wetzel 1983, Moustafa et al. 1998). For those months when the downstream flux was >0 (barrier beach was open), the mean ratio of TWMCs of chloride was 0.60, whereas the mean ratio of loads of chloride was 1.02 ($P = 0.001$, $N = 10$, paired-sample $t = -4.77$). Thus, TWMCs indicate that 40% of the chloride was retained or "lost" by the wetland, but loads indicate that all (102%) of the chloride entering the wetland eventually left it (Table 3). The fact that the load ratios of chloride are near unity indicates that the measurement of loads rather than TWMCs is imperative in order to have a correct un-

Table 8. Cumulative inputs (10^6 m^3) of water and loads (kg) of nutrients and suspended solids into Old Woman Creek Wetland and Lake Erie during four time periods consisting of multiple storm runoff events.

	Three Storms, 23 May–7 Jun 1989			Three Storms, 30 Dec 1989–14 Jan 1990			Two Storms, 17–24 Jan 1990			Two Storms, 12–18 May 1990		
	Load to Wetland	Load to Lake Erie	Lake Erie Load/ Wetland Load (%)	Load to Wetland	Load to Lake Erie	Lake Erie Load/ Wetland Load (%)	Load to Wetland	Load to Lake Erie	Lake Erie Load/ Wetland Load (%)	Load to Wetland	Load to Lake Erie	Lake Erie Load/ Wetland Load (%)
water	4.540	4.494	99.0	3.277	3.280	100.1	1.510	1.514	100.3	0.855	0.867	101.4
TP	2,756	1,754	63.6	546	344	63.0	164	200	122.0	278	201	72.3
SRP	115.3	27.3	23.7	74.4	51.1	68.7	9.35	3.80	40.6	5.74	4.54	79.1
TSS	1,989,400	1,327,300	66.7	195,000	146,100	74.9	50,800	113,900	224.2	404,600	162,900	40.3
NO_{2+3} N	21,350	14,580	68.3	25,650	20,510	80.0	11,220	9,100	81.1	7,290	3,670	50.3
NH_3 N	230.4	1,564.0	678.8	551.2	1,595.0	289.4	213.7	268.0	125.4	304.1	287.0	94.4
TKN	12,810	11,000	85.9	3,810	3,280	86.1	1,520	1,770	116.4	1,900	1,550	81.6
SiO_2	32,140	25,340	78.8	24,580	21,570	87.8	12,440	11,500	92.4	6,270	4,110	65.6
Cl	86,400	97,500	112.8	179,100	166,500	93.0	94,600	94,300	99.7	25,400	22,700	89.4

derstanding of the amount—and even the direction—of change in materials delivery to Lake Erie.

In this study, TWMCs greatly overestimated the export of TP and TSS compared to loads, and they greatly underestimated the export of ammonia, whereas both methods yielded similar estimates for some other materials, such as SRP and NO_{2+3} (Table 3). Similarly, in their study of a constructed wetland along the Kissimmee River (Florida), Moustafa et al. (1996) found that removal of TP and total N were greatly underestimated by concentration-based calculations as opposed to calculations based on mass balances.

Mechanisms of Changes in Loads

The loads of SRP, NO_{2+3}, and silica were reduced during passage through the wetland on a monthly basis and during storm runoff events, and Klarer and Millie (1989) similarly reported decreases in concentrations following storms. Biological, chemical, and physical mechanisms interact to modify the load of each material (Heath 1992, Matisoff and Eaker 1992, Tomaszek et al. 1997).

The loads of TP and TSS either increased or decreased during passage through the wetland under differing circumstances. On a monthly basis, which integrated a variety of hydrologic conditions, the TP load decreased, as it did during three of four storm periods. The TSS load also decreased during the same storm periods, but it increased on a monthly basis. The TP and TSS loads were highly significantly correlated ($r = 0.92$ to 0.99), as would be expected because 91% of the total load of P entering the wetland in the 1990 water year was in particulate form (TP less SRP, Table 7). Furthermore, because SRP is taken up by algae and bacteria and converted to TP, the TP-TSS relationship is strengthened during movement through the wetland. In fact, the highest correlation between TP and TSS ($r = 0.99$) was found between the downstream loads of TP and TSS. That monthly TSS loads increased during passage indicates that, although inorganic particulates carried from upstream settle out of the water column upon entering the wetland, plankton is produced throughout the wetland and upon export may more than compensate for the mass of TSS lost at the upper end. Thus, the biogenic/inorganic ratio of TSS is probably greater at export than at import to the wetland.

On a monthly basis, more NH_3 and TKN left the wetland than entered, and this net export of NH_3 was exaggerated during two of the four storm periods (Table 8). This pattern may reflect the release of NH_3 via storm-induced resuspension of the anaerobic organic-rich sediments of the wetland. Tomaszek et al. (1997) attributed ammonia in OWC wetland primarily to remineralization of organic materials in sediment and ex-

Table 9. Unit area yields of western Lake Erie tributaries. Rock Creek and Honey Creek are tributaries of the Sandusky River. The Cuyahoga River, which drains a largely urban and forested watershed into the central basin, is shown for comparison. Basin areas upstream of the sampling points are shown. OWC values are based on the 1990 water year* whereas the values for the remaining streams are based on average annual yields (Baker 1993).

	Annual Unit Area Yields, kg/ha					
Substance	Old Woman Cr. 57.2 km^2	Rock Cr. 88.0 km^2	Honey Cr. 386 km^2	Sandusky R. 3,240 km^2	Maumee R. 16,395 km^2	Cuyahoga R. 1,831 km^2
TP	0.844	1.14	1.23	1.41	1.30	1.89
SRP	0.076	0.13	0.20	0.23	0.21	0.32
TSS	392	643	565	760	610	1000
NO_{2+3} N	19.8	11.5	15.4	15.8	16.0	8.7
NH_3 N	0.472					
TKN	4.37	5.3	5.6	6.3	6.0	5.5
SiO_2	22.4					
Cl	141	61.4	62.6	77.8	78.6	466

* Data lacking for 17 February–11 March 1990; thus, values represent 0.937 year.

cretion by aquatic animals. Denitrification of NO_3^- to atmospheric N_2 provides an important route of nitrogen loss from OWC wetland (Wickstrom 1988, Heath 1992, Tomaszek et al. 1997).

Old Woman Creek as a Representative Wetland System

Extensive data sets have been developed for streams draining watersheds of western Lake Erie whose drainage areas differ by more than two orders of magnitude (Baker 1993). Most are largely agricultural watersheds, with row crops occupying as much as 83% of the land area, although some more-eastern watersheds, such as the Cuyahoga River, drain primarily urban and forested landscapes. The annual unit area export (kg/ha) of a given material can be computed for each stream by dividing the total annual export at the sampling station by the watershed area upstream of the station (Baker 1993). Comparison of unit area exports in OWC upstream from the wetland in the 1990 water year with mean annual unit area exports for other Lake Erie tributaries (Table 9) shows that OWC is more similar to the western Lake Erie streams than to the Cuyahoga River. However, its unit area yields of TP, SRP, TSS, and TKN are lower than those of the western Lake Erie streams, probably demonstrating the lower proportion of its drainage area (60%) devoted to row crop agriculture than in the other watersheds (76%-83%, Baker 1993). However, only a single year of record for OWC is being compared with annual averages for the other streams, and although 1990 was not a drought year, it may not have produced unit area yields close to the average. Baker and Richards (2002) reported that the TP export rates of the Maumee and Sandusky watersheds are high relative to those of most large Midwestern watersheds.

Comparison of the ratio of orthophosphate (SRP) to TP also shows that OWC is similar to nearby watersheds. In OWC, 9.0% of TP was in the form of orthophosphate in the 1990 water year, as compared to 11.4% on average in the slightly larger (88.0 km^2) Rock Creek watershed. In all the other watersheds, ranging in size from 386 km^2 to 16,395 km^2, orthophosphate contributed on average 16.3% to 16.9% of TP (Table 9). In the Ohio River Basin, adjacent to the southern boundary of the Lake Erie drainage, orthophosphate contributed 28% of TP on average from 1980 through 1996 (Goolsby et al. 1999), and in Illinois rivers from 1980 through 1997, the contribution was 38% (David and Gentry 2000). Baker and Richards (2002) attributed the lower proportion of orthophosphate in Lake Erie watersheds at least in part to improved farming practices in response to targeted phosphorus reduction programs within the Lake Erie basin.

The wide variation in monthly, seasonal, and annual loads exported from OWC is typical of creeks and rivers draining the western Lake Erie basin (Baker 1988, 1993) and much of the United States (Goolsby et al. 1999). Because of this great variability, related directly to storm hydrology, any attempt to characterize seasonal and annual loads should be conducted over many years (Baker 1988, Heath 1992). This is particularly the case if trends in loads are to be detected (Richards and Baker 1993). For example, although wetland water level (hence area and volume) during this study followed a general seasonal pattern, sporadic storm hydrographs of varying durations and magnitudes are overlain on that pattern, and it is during the storm runoff events that the majority of the materials is delivered to the wetland.

Nutrient concentrations, which are controlled by external and internal loads and transformations, are very important in terms of the physiology of individual species and thus in terms of ecological responses (e.g., productivity, Heath 1992). The seasonal and storm-related patterns of those concentrations in OWC are very similar to the patterns characteristic of other creeks and rivers draining into western Lake Erie (Baker 1988, 1993).

A vast array of wetland types, accompanied by a great diversity in their hydrology and primary nutrient sources, is present along the Great Lakes (Herdendorf 1992, Prince et al. 1992). Long-term studies similar to this one that involve intensive sampling of wetlands for which upland tributaries and seiches play major roles in the wetland hydrology (drowned river mouth wetlands, Keough et al. 1999) have not been published. The timing of delivery of materials and the variability of their concentrations as they are transported down tributaries to wetlands is scale-dependent (Baker and Richards 2000). Furthermore, the residence time—and therefore the time available for nutrient processing within the wetlands—is dependent on a number of variables, including the size of each wetland relative to its drainage area. Thus, nutrient processing would best be compared among wetlands of similar characteristics and size.

The Value of Riverine Coastal Wetlands in Mitigating Pollution

A plethora of studies confirms that most wetlands retain sediment, nutrients, and toxic contaminants or transform the nutrients and toxins into less bioavailable or less harmful forms. This study has confirmed the major role played by a Lake Erie wetland as a partial sink or transformer for sediment and several nutrients, either at all times or under certain hydrologic conditions. If other coastal wetlands surrounding tributary mouths perform the same functions, whether or not to the same degree, their collective importance in mitigating the pollution of Lake Erie must be substantial.

The total loads of phosphorus and other materials into Lake Erie have historically been calculated from measurements of individual tributary loads derived from samples collected at USGS gaging stations upstream from lake-level effects. However, this study and a study of Sandusky Bay by Richards and Baker (1985) reveal that considerable processing and transformation take place between the gaging stations (often located several km upstream from the influence of lake levels) and the open lake waters beyond tributary embayments. Thus, loads to Lake Erie could be estimated more accurately from quantitative data on the processing of materials in the lowermost reaches of each tributary. However, because of the extensive human and financial resources required, such investigations will most likely prove to be impractical. On the other hand, modeling efforts (Dortch and Gerald 1995, Mitsch and Wang 2000) may provide an alternative approach to quantifying the potential benefits of existing and restored wetlands.

The relative amount of materials delivered to a wetland (kg/ha of wetland) is a function of its size (area) in relation to stream discharge (i.e., the area of the watershed ($A_{wetland}/A_{watershed}$)). If a large wetland can remove a greater proportion of pollutants than a small one, an appropriate management approach for reducing the inputs of phosphorus and other contaminants to the Great Lakes would be to increase the functional wetland area. Mitsch and Wang (2000) via modeling results showed that both restored and diked coastal wetlands with direct hydrologic connections to their adjacent tributaries could retain substantial proportions of phosphorus received from the upstream watersheds.

Changes in the water level of Lake Erie result in changes in wetland area on the order of two magnitudes. In concert, the area available for retention and transformation of materials changes. A short-term study of a Lake Champlain wetland found that sediment and nutrient dynamics were influenced by lake level (Clausen and Johnson 1990). Since the late 1980s and early 1990s (this study), the water level of Lake Erie has decreased from record highs in 1986 to near-average levels. Consequently, although total wetland area has increased, the aggregate submersed area of undiked wetlands around Lake Erie has decreased along with hydraulic residence times, probably reducing the capacity of those wetlands to process materials received from upstream.

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