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Pollutant removal from stormwater runoff by palustrine wetlands based on comprehensive budgets

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Abstract

Comprehensive budgets of total suspended solids (TSS), total phosphorus (TP), total zinc (the most prevalent metal in urban runoff), and fecal coliforms (FC) were developed for two palustrine freshwater wetlands in Washington, USA. These budgets were based on input (inflow, precipitation, groundwater) and output (outflow, evaporation). One wetland received runoff from a primarily urbanized watershed (187 ha), whereas the other watershed (87 ha) was mostly rural and forested. Annual loadings to the urban wetland for TSS, TP, and zinc were 107, 0.63 and 0.43 kg ha^{-1} yr^{-1}, respectively, based on watershed area. Corresponding loadings to the nonurban wetland were 30, 0.62 and 0.08 kg ha^{-1} yr^{-1}. High TP input from groundwater at the nonurban wetland (82% of the TP input) resulted in overall loadings comparable to the urban wetland. Fecal coliform loadings were $4.2 \cdot 10^{10}$ and $1.4 \cdot 10^{9}$ organisms ha^{-1} yr^{-1} for the urban and nonurban wetlands, respectively. Mean annual removal percentages (over the two-year study period) for TSS, TP, zinc, and FC were estimated at 14, 8, 31, and 49%, respectively in the urban wetland (2 ha). Corresponding removal rates in the nonurban wetland (1.5 ha) were 56, 82, 23, and 29%, respectively. Pollutant removal was influenced by season, flow conditions, residence time, pollutant source (surface versus groundwater), and pollutant state (solid versus dissolved). Results for the nonurban wetland showed that groundwater inputs were significant for TP loadings and removal.

Keywords: Pollutant removal; Wetland; Zinc; Phosphorus; Fecal coliforms; Urban runoff

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1. Introduction

1.1. Pollutant dynamics in palustrine wetlands

Sediments, nutrients, metals, and bacteria originating from diffuse (nonpoint) sources can adversely affect the water quality and beneficial uses of receiving waters. Excess quantities of these substances from urban, agricultural, and forest activities result from vegetation removal, chemical usage, poor management practices, and increased runoff volumes. Agricultural sources have been identified as the most pervasive nonpoint source, but urban runoff is the next most commonly reported nonpoint problem (U.S. EPA, 1984). Better information about treatment techniques can improve the application of control programs aimed at reducing the effects of nonpoint pollutants. The use of natural or constructed wetlands for runoff treatment is one technique that has shown promise for nonpoint source pollution control (Baker, 1992; Hammer, 1992; Knight, 1992).

It is well established that wetlands can improve water quality under certain circumstances (Kadlec and Kadlec, 1979; Nichols, 1983; Horner, 1986; Martin, 1988). In particular, there has been a lot of research in the use of wetlands for wastewater treatment (Chan et al., 1981; Heliotis, 1982; U.S. EPA, 1985; Hammer, 1989). More recently, increased interest has been focused on the use of natural or constructed wetlands for treatment of nonpoint source pollution (Martin, 1988; Fleischer et al., 1989; Stockdale, 1991; Baker, 1992). Wetlands are considered most often for treatment of urban runoff (Strecker et al., 1992; Schueler, 1993) and mine drainage, but interest in using wetlands for treatment of agricultural runoff is increasing (Hammer, 1992; Rodgers and Dunn, 1992).

Sedimentation is one of the principal mechanisms of pollutant removal in wetlands. The retention of suspended solids in wetlands is controlled by particle size, hydrologic regime, flow velocity, wetland morphometry, residence time, and storm surges (Boto and Patrick, 1979; Kranck, 1984; Schubel and Carter, 1984). Hydraulic resistance from the vegetation and soil decreases the velocity of water entering a wetland and enhances the settling and deposition of suspended solids. While wetlands remove solids under many circumstances, they also can release solids due to scouring, plant dieoff, or algal washout. For this reason, it is desirable to have data from a variety of flow conditions, including baseflow and storm events, to assess accurately the overall balance of sediments.

Nutrients are removed, released, and transformed in wetlands by a number of mechanisms (Nichols, 1983; Nixon and Lee, 1985; Mitsch and Gosselink, 1986). Nutrients in particulate form (e.g., particulate phosphorus) often are removed by sedimentation. Significant amounts of dissolved phosphorus can be taken up by vegetation during the growing season, but can also be released once vegetation decays (Richardson et al., 1978). Dissolved phosphorus can also participate in a number of precipitation, adsorption, and complexation reactions (Lee et al., 1975).

Heavy metals in urban runoff occur in both soluble and particulate forms. Some metals, such as zinc, tend to be more soluble in water and therefore more mobile. In wetlands, ion exchange, precipitation, and plant uptake are the primary removal mechanisms for soluble metals; however, groundwater infiltration can also be important if the wetland recharges groundwater. Other metals, such as lead, tend to adsorb to sediments and other particles and thus are transported along with solids. These metals are removed with solids from the water column when velocities decrease, accumulating in the bottom sediments of wetlands. Metals in wetland sediments may create toxic conditions for fish and other aquatic life through introduction into the food web (Kadlec and Kadlec, 1979).

Fecal coliforms, an indicator of pathogens (disease-bearing organisms) in surface waters, also tend to be associated with sediments and other particles (Bott, 1973). Thus, like solids, the concentrations of fecal coliforms can be reduced in wetlands through physical settling of particles. Fecal coliforms can also die off in surface waters from exposure to low water temperatures or plant excretions.

1.2. Puget Sound Wetlands and Stormwater Management Research Program

The Puget Sound Wetlands and Stormwater Management Research Program was established to determine the effects of urban stormwater on wetlands and the effect of wetlands on the quality of urban stormwater. The results of the research are intended for use in developing sound policies and guidelines for effectively managing both urban wetlands and stormwater in concert for optimum resource protection. There are two primary components of the research program: (1) a study of the long-term effects of urban stormwater on wetlands (King County Resource Planning, 1987), and (2) a study of the water quality benefits to downstream receiving waters as urban stormwater flows through wetlands. This paper presents results from the second component of the research program. The purpose of the work presented here was to examine pollutant removal in natural wetlands.

2. Methods and materials

2.1. Study area description

Two palustrine (isolated freshwater) wetlands were intensively monitored for the water quality benefit study: Bellevue 3I (B3I) and Patterson Creek 12 (PC12) (Fig. 1). The B3I wetland is located in Bellevue, WA, USA, at the headwaters of Richards Creek. The wetland is about 2 ha in size and contains primarily forested and scrub-shrub wetland classes (Fig. 2a). The B3I wetland watershed is approximately 187 ha in area and highly urbanized. Land uses include single- and multi-family residential, commercial, transportation and isolated mixed forests. Runoff to the wetland is mostly piped and flows year round. The PC12 wetland is located in unincorporated King County, east of Lake Sammamish. The wetland is about 1.5 ha in size and contains scrub-shrub, emergent and open water wetland classes (Fig. 2b). The PC12 wetland watershed is approximately 87 ha in area and mostly rural. The watershed is primarily forested with some isolated single-family

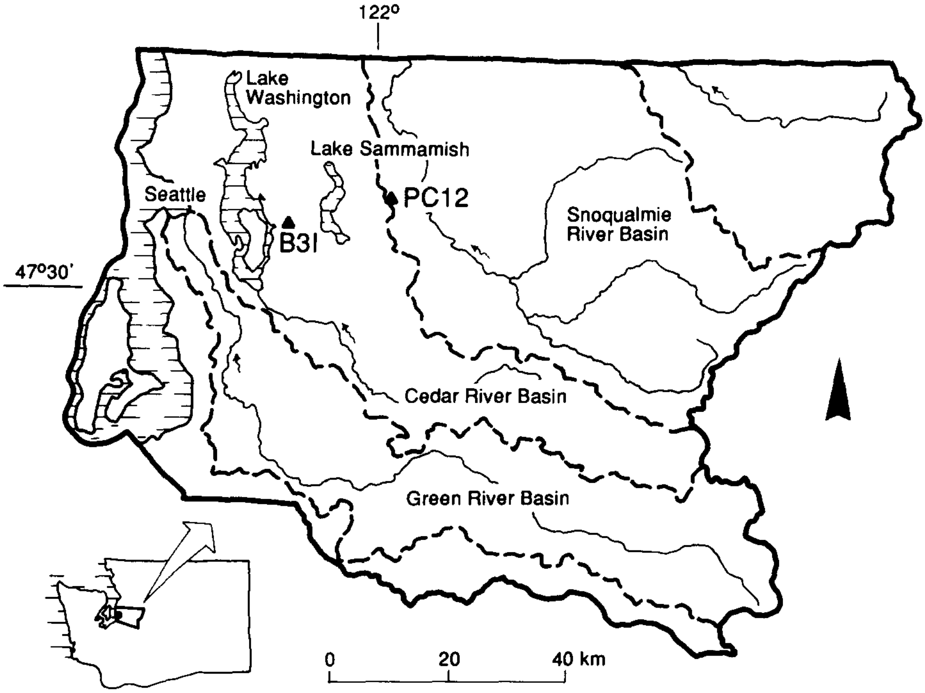

Fig. 1. Location map of the B3I and PC12 wetlands, King County, Washington, USA.

housing. The drainage system is mostly natural and flows from approximately October to June, depending on the year.

2.2. Field measurements and sampling

The study was designed to provide a complete water quantity and quality balance. For both wetlands, there was continuous flow recording at the inlet and outlet, continuous precipitation recording, and regular piezometric surface recording around the wetland (Reinelt et al., 1993). Water quality analyses were performed for baseflow and storm samples at the inlet and outlet, precipitation and groundwater samples. Wetland B3I was studied over a 2-year period from June 1988 to May 1990. Wetland PC12 was studied over only a 20-month period (October 1988 to May 1990), because the inlet and outlet flows were negligible during the summer.

Surface water measurements

Continuous water flow measurements were taken at the inlet and outlet of the two wetlands using several different techniques (e.g., V-notch and trapezoidal weirs, channel, full-pipe flow) as described in Reinelt et al. (1990). At three of four stations, stage measurements were made using an American Sigma 8100 volumet-

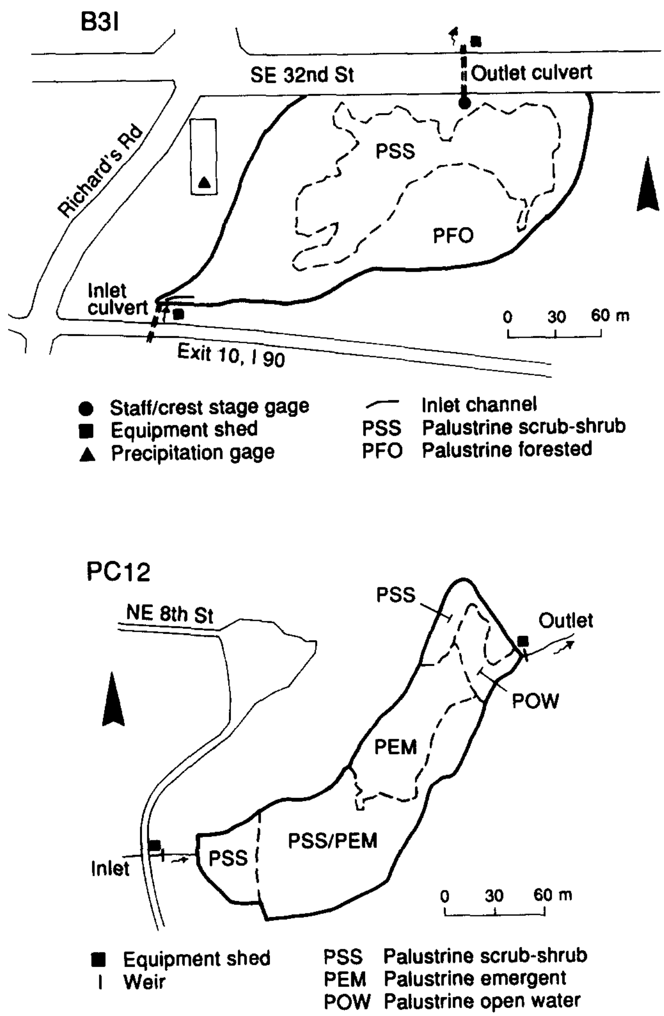

Fig. 2. Site maps of the (a) B3I and (b) PC12 wetlands showing wetland classes and research equipment.

ric flow computer, and flows were calculated using standard weir equations or rating curves (i.e., stage-discharge relation). At the B3I outlet, water flowed through a pipe that was always full. Average velocity (determined by a Marsh–McBirney Model 265 Velocity Modified Flow Meter) was multiplied by the cross-sectional area of the pipe to determine the flow.

Water quality sampling was carried out during baseflow and storm events at the inlet and outlet of both wetlands using Manning Model S-3000 automatic composite samplers modified for enhanced solids capture. During baseflow conditions 24-h composite samples were collected. Flow-weighted composite samples were collected during storm events to obtain event mean concentrations (EMC). There were a total of 30 baseflow and 18 storm samples collected at B3I; and 20 baseflow and 12 storm samples collected at PC12. More detail on sampling was provided by Reinelt et al. (1990).

Groundwater measurements

Shallow and deep piezometers were installed at both wetlands to aid in the estimation of groundwater flow. Shallow piezometers (1.2 to 3 m), placed along the periphery of the wetland facilitated estimates of horizontal groundwater flow. Deep piezometer clusters (three separate, single-hole piezometers installed at one location with depths ranging from 6 to 18 m) were used to assess vertical groundwater discharge with respect to deeper water-bearing zones. Darcy's empirical law governing groundwater flow was used to estimate the net flow rate of groundwater to each wetland. More detail on field methods and discharge calculations were presented by Surowiec (1989).

Groundwater samples were taken on 12 occasions, distributed approximately bimonthly during 1988 and 1989. Samples were extracted from the piezometers using a hand pump and plastic tubing. To obtain sufficient sample volume from the shallow piezometers, composites samples were usually made from two nearby piezometers. The deep piezometer samples were taken from a single piezometer.

Precipitation measurements

The amount and intensity of precipitation were measured at a single location in both watersheds. An event recorder (Weathermeasure Model 6114), connected to a tipping-bucket gage, was used to record each 0.25 mm of precipitation. These data were used to compute the total daily precipitation, and the duration and intensity of storm events. The daily precipitation was multiplied by the wetland area to estimate the water volume from direct precipitation inputs. Precipitation samples were taken from a plastic bag lining a garbage can near the outlet of both wetlands.

Water quality analyses

All baseflow and stormflow composite samples were analyzed for total suspended solids (TSS), total phosphorus (TP), total zinc (Zn), and fecal coliform

Table 1
Laboratory analysis specifications for the measured variables

Variable	Method	Equipment
Total suspended solids (TSS)	Gravimetric	Mettler Type H 80 analytical balance
Total phosphorus (TP)	Automated heteropoly blue after persulfate digestion	Alpkem RFA Model 300
Total zinc (Zn)	Inductively coupled plasma (ICP)	Jarrell-Ash Model 61
Fecal coliforms (FC)	Membrane filter	-

Table 2
Watershed land use information from the GIS for Bellevue (B3I) and Patterson Creek (PC12) study wetlands in Bellevue, Washington, USA

Land use	Code	Area (ha)	Percent of watershed
B3I Wetland			
High-density single family	113	77.3	42.1
Low-density multi family	115	22.4	12.2
Commercial	120	32.0	17.4
Industrial	131	0.6	0.3
Freeway	144	5.0	2.7
Utility	155	0.5	0.3
Commercial facility	160	23.3	12.7
Open land	190	22.6	12.3
Total		183.7	100.0
PC12 Wetland			
Low-density single family	111	19.9	23.5
Open land	190	1.1	1.3
Forest-mixed	430	63.6	75.2
Total		84.5	100.0

bacteria (FC). Groundwater samples were analyzed for TP, Zn, and FC. Precipitation samples were analyzed for TP and Zn. All sample bottles were prepared in accordance with U.S. EPA (1983) specifications. The laboratory analysis methods, including analytical instruments used, are shown in Table 1.

2.3. Watershed delineation and land use classification

The wetland watersheds were delineated using contour information available on USGS topographic maps. Field checking verified the location of uncertain boundaries. Land use information was obtained from aerial photos and entered into a geographic information system (GIS) for easy analysis (Reinelt et al., 1991). Table 2 summarizes the land use data for both wetland watersheds. The B3I watershed contains 88% urban land uses, while the PC12 watershed contains 24% urban land (all low density single-family residential).

2.4. Comprehensive pollutant budgets

Information on the comprehensive water balance for the two wetlands is presented in Reinelt et al. (1993). All components of the balance were obtained from field measurements. Surface and groundwater flows, precipitation, evapotranspiration and wetland storage either were measured continuously or estimated from discrete measurements.

Pollutant loading and removal estimation

The transport or loading of TSS, TP, Zn, and FC at the inlet and outlet of a wetland was calculated as:

$$L(t) = Q(t) \cdot C(t), \tag{1}$$

where $L(t)$ = estimated pollutant mass flux for time t, $Q(t)$ = volume of water for time t, and $C(t)$ = estimated pollutant mass per unit volume for time t.

A daily time step was chosen to carry out the loading estimates for purposes of applying pollutant concentrations to flow. In Eq. 1 mean daily flow was directly measured for each day and, therefore, the values were considered to be reasonably accurate. The pollutant concentrations, on the other hand, were measured less frequently as noted above and estimates were made for those days or events not measured. Thus, considerable uncertainty may be associated with these daily estimates.

One of the following four methods was used to estimate daily inlet and outlet concentrations for the four variables at each wetland: (1) multiple or (2) simple linear regression analysis, (3) arithmetic or (4) geometric seasonal or annual mean concentrations. Multiple linear regression analysis was used if correlations occurred between pollutant concentration and hydrologic factors expected to affect such concentrations (e.g., flow, precipitation). Simple linear regressions were applied if correlations existed between concentrations at the inlet and outlet of a wetland, or between concentrations of two variables. Seasonal or annual mean concentrations were used if there was no apparent relationship between pollutant concentrations and other factors. Seasons were determined based on literature guidance and judgments relative to expected plant uptake or release of substances in the Pacific Northwest (Reinelt and Horner, 1990). The seasons, dates and classifications used are given in Table 3. A summary of the methods used to estimate daily concentrations for the two wetlands, four variables and flow conditions is given in Table 4.

To estimate net removal or release in a wetland, the daily inlet and outlet concentration estimates were multiplied by mean daily flow according to:

$$\text{Removal} = \left(Q_{(\text{in})} \cdot C_{(\text{in})}\right) - \left(Q_{(\text{out})} \cdot C_{(\text{out})}\right). \tag{2}$$

A negative value for net removal corresponds to pollutant release. Pollutant

Table 3
Seasons, dates, plant status, and expected uptake and release of substances for study wetlands

Seasons	Dates	Plant status	Uptake/release
1	March 1–May 15	early growing	some uptake expected
2	May 16–August 15	main growing	high uptake expected
3	August 16–September 30	decline	gradual release expected
4	October 1–November 15	decay	high release expected
5	November 16–February 28	dormant	continued release expected

Table 4
Summary of methods used to estimate daily concentrations for the two study wetlands, four variables and flow conditions (SM = seasonal or annual arithmetic mean, GM = seasonal or annual geometric mean, MR = multiple linear regression, SR = simple linear regression)

Station	Estimation method [a]			
	TSS	TP	Zn	FC
B3I Wetland				
Inlet (baseflow)	SM	SM	SM	GM
Inlet (storms)	MR	SR	SM	GM
Outlet (baseflow)	SM	SM	SM	GM
Outlet (storms)	SR	SR	SM	GM
Precipitation	–	SM	SM	–
Groundwater	–	SM	SM	GM
PC12 Wetland				
Inlet (baseflow)	SM	SM	SM	GM
Inlet (storms)	SM	SM	SM	GM
Outlet (baseflow)	SM	SM	SM	GM
Outlet (storms)	SM	SM	SM	GM
Precipitation	–	SM	SM	–
Groundwater	–	GM	SM	GM

[a] TSS = total suspendend solids; TP = total phosphorus; FC = fecal coliforms.

removal or release was calculated by season, flow conditions, and on an annual basis to examine its variation.

3. Results and discussion

3.1. Summary characteristics of the two wetlands

A summary of the natural and hydrologic characteristics during the study for both wetlands is given in Table 5. Mean daily flows at the inlet of the two wetlands are shown in Figs. 3a and c. The hydrologic reaction to storms of the two wetlands are typical of the respective watershed land uses. The reaction of B3I inlet flows to storms is fast and dramatic. Flows increase almost immediately because of the large impervious land area and piped storm drain system. Similarly, when storms end, the flow recedes quickly to near baseflow conditions. The PC12 inlet flow, on the other hand, reacts relatively slowly to storms, with the receding limb of the hydrograph extending much longer. Significant inflows occurred at PC12 only from October to June; however, there was water in the wetland year round.

The monthly distributions of precipitation for the two wetlands are shown in Figs. 3b and d. Nearly 80% of the annual precipitation occurred between October and March. The maximum daily precipitation occurred on January 9, 1990 (approximately 80 mm at both sites).

Continually changing inflow rates and wetland water storage volumes resulted in variable residence times within the wetlands. During storms, the B3I wetland

Table 5
Natural and hydrologic characteristics of the B3I and PC12 wetlands (note: Study period was 2 years for B3I and 20 months for PC12)

Variable (unit)	B3I wetland	PC12 wetland
Dominant land type	Urban	Forest
Drainage area (ha)	187	87
Wetland area (ha)	2	1.5
Effective wetland area (ha)	0.22	0.38
Total precipitation (mm)	1813	1934
Precipitation volume (m^3) in drainage area	$3.4 \cdot 10^6$	$1.7 \cdot 10^6$
Mean daily inlet flow (m^3/s)	0.042	0.021
Maximum daily inlet flow (m^3/s)	0.75	0.22
Days with measurable flow during study	730	493
Total flow during study (m^3)	$2.7 \cdot 10^6$	$0.9 \cdot 10^6$
Wetland storage volume (m^3) [a]	400–5000	600–7000
Runoff/precipitation ratio	0.80	0.53
Residence time for mean daily flow (hours)	3.3	20
January 9, 1990 residence time (hours)	1.9	9

[a] Wetland storage volume varies depending on season and flow conditions.

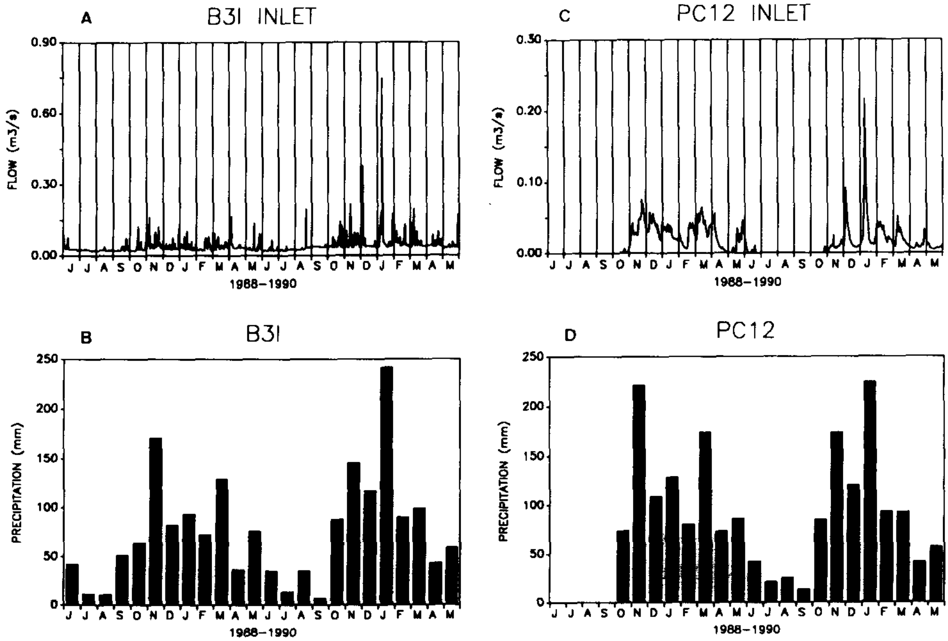

Fig. 3. Hydrologic characteristics of the B3I and PC12 wetlands during the study period. (a) Mean daily flow at B3I, (b) monthly precipitation at B3I, (c) mean daily flow at PC12, (d) monthly precipitation at PC12 (note: scales are different for flow at B3I and PC12).

storage increased significantly because of the constriction at the outlet culvert. However, since flows were also higher, the residence time often decreased (cf. Table 5 for January 9, 1990 storm). The storage of the PC12 wetland varied by season, but it also increased during storms. A beaver dam at the outlet of PC12 partially constricted outlet flows until it was badly damaged by the January 9, 1990 storm.

Channelization in both wetlands reduced the areal distribution of water and residence times. In the B3I wetland, many bifurcating channels meander throughout the wetland. The total surface area of these channels during baseflow conditions is approximately 0.22 ha. Thus, during much of the year this represents the effective wetland area in terms of pollutant removal. In PC12, there is considerable channelization in the upper scrub-shrub and emergent zones. However, in the lower portion of the wetland, there is less gradient and considerable pooling in the emergent and open water zones. The water surface area for mean water depth is approximately 0.38 ha. Approximate residence times for mean daily flows were calculated from estimates of water storage volume for mean conditions (Table 5).

3.2. Concentration estimates for nonsampled occasions

There is often a relationship between TSS concentrations and a variety of hydrologic factors (McElroy et al., 1976; Horner and Mar, 1982). These factors include the amount, intensity, or duration of precipitation events, and water flow. In urban areas, where the buildup and washoff of pollutants from impervious surfaces play an important role, the antecedent days and precipitation during preceding events can also influence pollutant concentrations. For the B3I wetland, all of these factors were considered in the development of a regression model to estimate inlet TSS EMCs for storm events. The multiple linear regression shown in Eq. 3 was found to produce the best estimates:

$$\mathrm{EMC}_{(\mathrm{TSS})} = a_0 + a_1 \cdot I + a_2 \cdot D + a_3 \cdot P + a_4 \cdot Q, \tag{3}$$

where $\mathrm{EMC}_{(\mathrm{TSS})}$ = estimated EMC for inlet TSS in mg/l; a_0, a_1, a_2, a_3, a_4 = regression coefficients; I = precipitation intensity in mm/h; D = duration of precipitation in hours; P = preceding week precipitation intensity in mm/day; and Q = mean daily flow rate in m^3/s.

Eleven storm data records were used in the regression of Eq. 3, resulting in an R^2 of 0.90. Data from samples collected during rare events (e.g., one-time snowmelt event during 1989, an extreme event immediately following a high flow event) were not included in the analysis, because these events were not considered representative of the events to which the regression model was applied. A plot of modeled TSS versus measured TSS is shown in Fig. 4a.

There was a positive correlation for intensity and duration, and a negative correlation for preceding week precipitation and flow. The positive correlation with intensity and duration seems reasonable because there is usually an increase in pollutant washoff with more intense and longer storms. A negative correlation with preceding week precipitation occurs because rainfall during the preceding

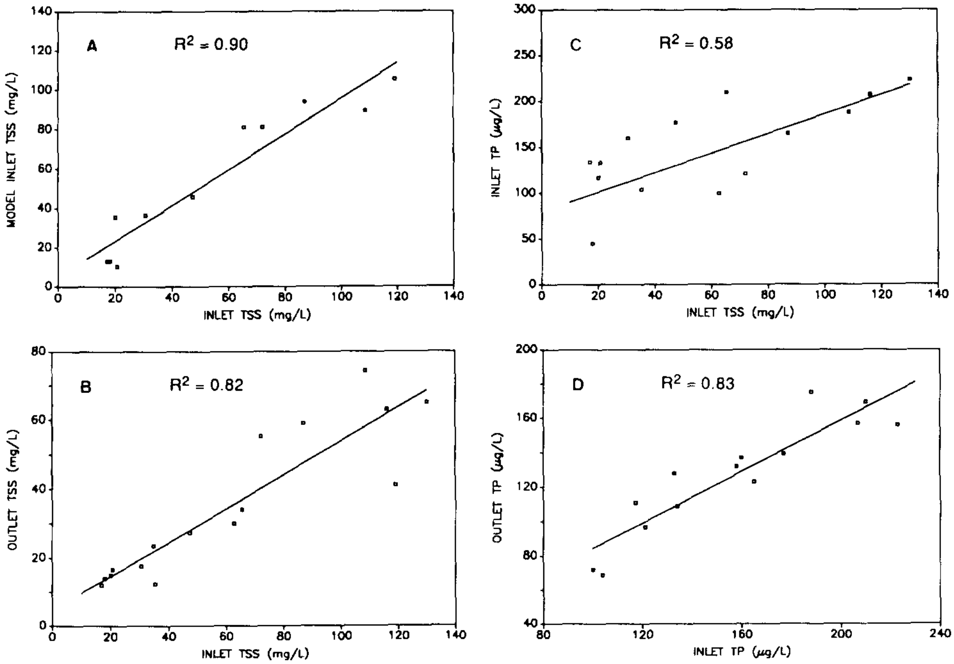

Fig. 4. Relationships between modeled variables at B3I inlet and outlet: (a) model inlet TSS vs. measured inlet TSS, (b) outlet TSS vs. inlet TSS, (c) inlet TP vs. inlet TSS, and (d) outlet TP vs. inlet TP.

week reduces the pollutants available for mobilization by the next storm. The negative correlation with flow is more confusing. With increasing flow, there is a positive correlation with TSS, but as flows increase further, there is a dilution effect, because most easily mobilized solids are already removed. An overall negative correlation with flow, however, occurs in the regression of Eq. 3.

Three events at B3I were well outside the applicable range of the inlet TSS regression model (the two largest storms: December 4, 1989, and January 9, 1990, and a high intensity, short duration storm on August 15, 1989). The model resulted in negative TSS estimates for the two large storms because of the high flows, and an improbably high TSS estimate for the summer storm. Rather than trying to estimate EMCs associated with these occasions, these dates were removed from the analysis. The two large storms accounted for 3.6% of the total 2-year flow volume at B3I, but they represented only 0.3% of the study period. The total inlet loading estimates presented below for TSS and TP at B3I are thus slightly low because these three events were not included in the analysis. For storm events at the B3I wetland, inlet EMCs for TP, and outlet EMCs for TSS and TP were modeled by the simple linear regression shown in Eq. 4:

$$\mathrm{EMC}_{(\mathrm{est})} = b_0 + b_1 \cdot \mathrm{EMC}_{(\mathrm{meas})} \tag{4}$$

Table 6
Summary of mean concentration data for the two wetlands and four water quality variables (all values are arithmetic means, except those shown as geometric means in Table 4; ranges given in parentheses)

Station	Pollutant concentrations [a]			
	TSS (mg/l)	TP (μg/l)	Zinc (μg/l)	FC (org/100 ml)
B3I Wetland				
Inlet (base)	4 (1–11)	75 (35–91)	26 (7–51)	441 (2–3900)
Inlet (storms)	58 (18–120)	172 (90–220)	95 (38–137)	746 (200–4900)
Outlet (base)	7 (1–15)	72 (39–95)	25 (4–59)	182 (30–1840)
Outlet (storms)	34 (10–75)	127 (70–180)	57 (33–79)	379 (120–4000)
Precipitation	–	68 (16–97)	163 (26–418)	–
Groundwater	–	42 (26–100)	8 (4–15)	5 (2–16)
PC12 Wetland				
Inlet (base)	2 (0–20)	31 (9–66)	9 (4–15)	20 (2–480)
Inlet (storms)	4 (1–32)	28 (7–57)	11 (4–32)	31 (2–550)
Outlet (base)	3 (0–17)	47 (5–88)	9 (4–39)	15 (2–350)
Outlet (storms)	2 (1–5)	25 (2–48)	10 (4–28)	16 (2–130)
Precipitation	–	30 (9–75)	19 (11–58)	–
Groundwater	–	365 (230–900)	9 (4–19)	2 (1–6)

[a] See footnote to Table 4.

where $EMC_{(est)}$ = estimated EMC for inlet TP, outlet TSS or TP; b_0 and b_1 = estimated regression coefficients; and $EMC_{(meas)}$ = measured EMC for inlet TSS or TP.

All storm data were used for the simple regression of outlet and inlet TSS EMCs (Fig. 4b, $R^2 = 0.82$, $n = 16$). Sediment has often been viewed as a carrier of other pollutants (Zison, 1980; Horner and Mar, 1982), including phosphorus, metals and bacteria. Inlet storm EMCs for TP were estimated from inlet storm TSS EMCs (Fig. 4c, $R^2 = 0.58$, $n = 14$). Finally, outlet storm TP EMCs were estimated from inlet storm TP EMCs (Fig. 4d, $R^2 = 0.83$, $n = 14$).

There was no apparent relationship between storm zinc or FC EMCs and flow or other water quality variables. In fact, the storm zinc concentrations at B3I were quite stable, regardless of storm size, and seasonal differences. The storm mean was 95 μg/l, and the coefficient of variation (CV) was 0.25, indicating a tight grouping of storm values. Conversely, storm FC values were highly variable, ranging from 2 to 14400 organisms/100 ml (org/100 ml), with a geometric mean of 746 org/100 ml. Table 6 provides summary concentration data for the four variables for all inputs and outputs at the B3I wetland.

For the PC12 wetland, inlet and outlet TSS, TP, zinc, and FC values were generally lower than in B3I for both storm and baseflow conditions. These lower concentrations and minimal storm effects reflect the nonurbanized conditions in the basin. There were no apparent relationships between flow and other variables for this wetland. Thus, for all four variables, seasonal or annual arithmetic or geometric mean values were applied to flow directly. The high groundwater TP concentration at PC12 was unexpected. The geometric mean TP value (of 6 samples taken from deep piezometers over a one-year period) was 365 $\mu g/l$. This is nearly 10 times the surface inflow concentrations, and also nine times the TP values from the B3I wetland groundwater. The geometric mean SRP values for the same 6 samples were 206 $\mu g/l$, indicating that a substantial fraction of the phosphorus was in dissolved form. Recent data from nearby deep well sampling carried out as a part of new development projects confirmed the high TP concentrations. Geometric means of 569 $\mu g/l$ from seven wells and 737 $\mu g/l$ from six wells were reported by Beak Consultants, Inc. (1992) and W&H Pacific, Inc. (1992), respectively. The suspected source of these high TP concentrations is phosphorus-rich soil deposits, perhaps associated with historic lake beds. Table 6 provides summary concentration data for the four variables for all inputs and outputs at the PC12 wetland.

3.3. Pollutant loading and removal

Discrepancies are likely between estimated and true removal for a given event or day. Past studies using similar types of empirical models, however, have shown that these discrepancies are reduced when grouping estimates over longer time periods, groups of storms or seasons (Chui, 1981; Little et al., 1983). For these reasons, the results presented below are for annual and seasonal estimates only.

Total and component annual inputs (baseflow, storms, groundwater, precipitation) and outputs (baseflow, storms), and percent removal estimates for both wetlands and the four variables are given in Table 7. The removal percentages for TSS and TP of 14 and 8%, respectively, at B3I are quite low given the high inlet loadings. These low removal rates likely result from short residence times, high water velocities and little available storage capacity for sedimentation. Visible deposition of sediments in the wetland demonstrates that TSS removal is occurring; however, over time there is likely to be a reduction in storage capacity and removal potential.

For B3I, there were substantial differences between storm and baseflow conditions for TSS and TP removal or release. There was a 29% TSS removal during storm conditions and a 116% release during baseflows. The TSS released during baseflow conditions are probably a combination of fine sediment particles and organic matter (e.g., algae, detritus) that are easily transported by lower flows. For TP, there was a 10% removal during storms and a 14% release during baseflow. The mean soluble reactive phosphorus (SRP) components of TP were 16 and 15% for the inlet and outlet, respectively, during storms. Similar means for baseflows were 45 and 49%. Storm removal likely resulted from sedimentation of particulate phosphorus. Baseflow releases might be explained by solubilization of P in the surficial sediments by reducing conditions, and algal or detritus washout (i.e., export of particulate organic matter after assimilation of soluble phosphorus).

Zinc and FC removals at B3I were 31 and 49%, respectively. Nearly all of the zinc removal occurred during storm events, when concentrations were substantially higher (Table 6). Both particulate and dissolved zinc were removed during storms.

Table 7
Summary of pollutant inputs and outputs for wetlands B3I and PC12 (all values in kg/year except fecal coliforms which are in organisms/year)

Pollutant	Inputs										Outputs						
	Baseflow	%	Storms	%	Ground water	%	Prec.	%	Total	%	Baseflow	%	Storms	%	Total	%	% Removal
B3I																	
TSS	2187	10.9	17787	89.1	–	0.0	–	0.0	19974	100.0	4715	27.3	12546	72.7	17261	100.0	13.6
TP	45.4	38.6	61.3	52.2	10.3	8.8	0.5	0.4	117.5	100.0	51.7	47.6	57.0	52.4	108.7	100.0	7.5
Zinc	16.8	20.8	58.8	73.0	1.8	2.2	3.2	4.0	80.6	100.0	16.4	29.3	39.5	70.7	55.9	100.0	30.6
FC	$3.4 \cdot 10^{12}$	43.2	$4.4 \cdot 10^{12}$	56.7	$9.2 \cdot 10^{09}$	0.1	–	0.0	$7.8 \cdot 10^{12}$	100.0	$1.5 \cdot 10^{12}$	36.7	$2.5 \cdot 10^{12}$	63.3	$4.0 \cdot 10^{12}$	100.0	49.1
PC12																	
TSS	264	10.2	2312	89.8	–	0.0	–	0.0	2576	100.0	437	39.0	684	61.0	1121	100.0	56.5
TP	2.5	4.6	6.8	12.6	44.4	82.4	0.2	0.4	53.9	100.0	2.9	30.5	6.6	69.5	9.5	100.0	82.4
Zinc	1.8	26.1	3.8	55.1	1.0	14.5	0.3	4.3	6.9	100.0	2.1	39.6	3.2	60.4	5.3	100.0	23.2
FC	$3.9 \cdot 10^{10}$	32.0	$8.0 \cdot 10^{10}$	65.9	$2.5 \cdot 10^{09}$	2.1	–	0.0	$1.2 \cdot 10^{11}$	100.0	$3.6 \cdot 10^{10}$	41.3	$5.1 \cdot 10^{10}$	58.7	$8.6 \cdot 10^{10}$	100.0	29.0

Removal mechanisms likely included sedimentation, vegetation filtration, and cation exchange (the exchange of heavy metals for sacrificial cations such as potassium and calcium). For FC, baseflow and storm removal were 57 and 43%, respectively. Reductions in fecal coliforms likely resulted from sedimentation, vegetation filtration, and bacterial die-off.

The areal watershed loadings of TSS, TP, zinc, and FC to PC12 were 28, 19, 15, and 3% of the loadings to B3I, respectively. As noted above, however, groundwater TP loadings to PC12 were very high, resulting in overall TP loadings comparable to B3I. The removal pattern for TSS in PC12 was similar to B3I, with a net storm removal and net baseflow release. The net TSS removal was 57%. The large pooled area near the outlet and longer residence time at PC12 resulted in high TSS removal during storms (70%). Baseflow release of TSS was 66%, but the volume of water was small in comparison to storm volume. Similar to B3I, it is suspected that much of this release was organic matter.

Of the total TP loading to PC12, 82% was estimated to originate from groundwater. The suspected source of these high TP concentrations is phosphorus-rich soil deposits from 5 to 20 m and deeper below the surface. The high groundwater TP loading explains why baseflow outlet TP loadings regularly exceeded inlet loadings. Even during storms, when particulate-P was about 70% of TP, and significant sedimentation was expected, outlet loadings were comparable to inlet loadings. The estimated TP removal of the wetland was 82%. Mechanisms likely important for this removal include soil adsorption, plant uptake, and sedimentation.

Zinc concentrations at PC12 were near or below detection limits on many occasions during both baseflow and storm conditions. Additionally, study mean concentrations at the inlet and outlet were quite comparable (Table 6). For these reasons, it was difficult to estimate removal rates with a high degree of accuracy. The estimated zinc removal of 22% shown in Table 7 is sensitive to shifts in mean concentrations of 1–2 $\mu g/l$. During baseflow conditions, the zinc transport at the

Table 8
Comparison of sediment and phosphorus removal in study wetlands with those in natural wetlands (all values in kg ha^{-1} yr^{-1}, percent removal shown in parentheses)

Location	TSS		TP		Source
	Loading	Removal	Loading	Removal	
B3I Wetland, WA (palustrine)	9990	1360 (13.6)	59	4.4 (7.5)	this study
PC12 Wetland, WA (palustrine)	1720	970 (56.5)	36	29.6 (82.4)	this study
Heron Pond, IL (riparian palustrine)	150000	4470 (3)	800	36 (4.5)	Mitsch (1992)
Old Woman Creek, OH (instream)	–	–	110	11 (10)	Mitsch (1992)

outlet was greater than zinc loading at the inlet, indicating that there was some input of zinc from groundwater or from remobilization in the wetland.

Fecal coliform concentrations and loadings to PC12 were quite low during both baseflow and storm conditions (Tables 6 and 7). Overall, FC removal was estimated at 29%, with removal during baseflow and storm conditions of 12 and 36%, respectively. These removal levels were lower than expected, given the substantial sedimentation in the wetland, and the potential for plant filtration from the dense reed canarygrass throughout much of the wetland.

Examination of the loading and removal rates for TSS and TP in the B3I and PC12 wetlands in comparison with a study of two riparian wetlands (Mitsch, 1992) shows that removal efficiencies increase with decreased loading rates (Table 8). Three of the four wetlands are primarily instream systems with TSS and TP removals of less than 15%. The PC12 wetland with the lowest loading rate and longer residence time had considerably greater TSS and TP removal.

3.4. Seasonal loading and removal

The seasonal loading and removal or release of TSS, TP, zinc, and FC are shown in Fig. 5. As expected, the pollutant loadings to both wetlands followed closely the inflow volumes. Generally, loadings were greatest during season 5 (November to February), the wettest season, followed by seasons 4 and 1, the transition seasons. At the B3I wetland, net removal of TSS occurred during most seasons. The primary exception was season 1, when baseflow releases of TSS exceeded the storm removal. The greatest percent TSS removal occurred during season 3, when inflows were lowest, residence times were highest, and the potential for sedimentation was greatest. For TP, removal was greatest during season 2, when high plant uptake was expected (cf. Table 3). No other season showed substantial TP removal during both years. There was an unexplained release of TP during season 1 of the first year at a time when some plant uptake was expected.

Neither zinc nor FC showed any seasonal trends in terms of removal at B3I. During the 19 storms sampled, zinc removal varied from 13 to 61%, but the coefficient of variation (CV) was 0.33, indicating low variation. There were no trends by season. FC removal was highly variable, ranging from 98% removal to 100% release. The CV of percent removal was 1.92, indicating high variation, but again there were no seasonal trends.

Overall removal of TSS at PC12 was strongly influenced by season 5, when the mean removal was over 70%. Loadings were also substantially higher during season 5. During all other seasons, there was either TSS release or low removal. These releases may be partly explained by algal or detritus washout. TP removal occurred during all seasons, but percent TP removal was greatest during seasons 2 and 3, when loadings were low and plant uptake was high. A substantial portion of the TP loading to the wetland originating from groundwater was probably immobilized in the soil substrate through adsorption and ion exchange. There were not enough data to identify any seasonal trends in zinc or FC removal at PC12. Mean

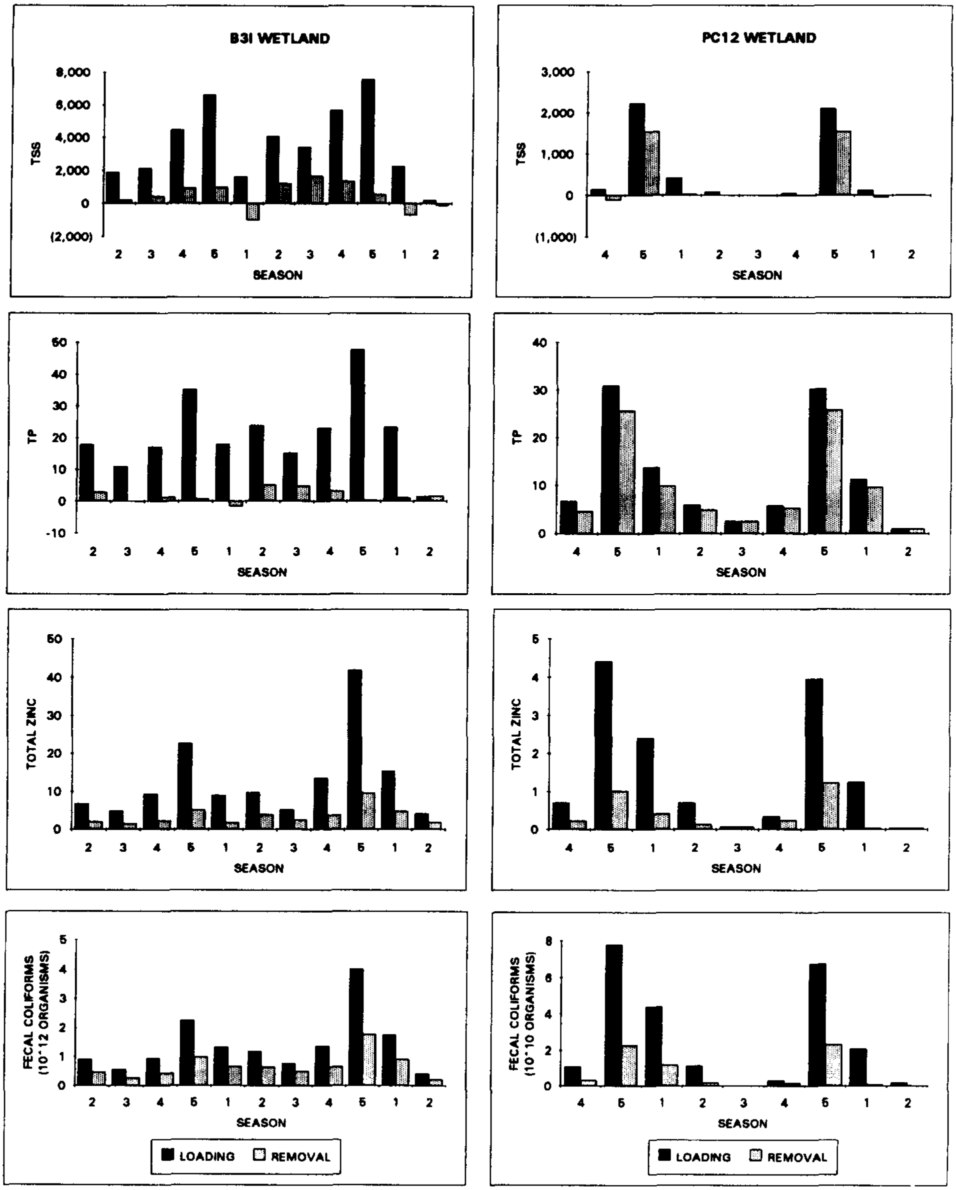

Fig. 5. Seasonal loading and removal (+) or release (−) for TSS, TP, zinc, and fecal coliforms for the B3I and PC12 wetlands (all values in kg/season, except fecal coliforms in organisms/season; note: scales are different for B3I and PC12).

annual removal estimates were made using all data. The zinc and FC loadings to PC12 were 8.5 and 1.5% of the loadings to B3I, respectively. Concentrations were also quite low, including numerous data points below detectable levels.

4. Conclusions

A better understanding of the natural assimilative processes in wetlands can contribute to improved management of existing wetlands and better designs for wetlands constructed for pollutant removal purposes (e.g., for wastewater, stormwater, or agricultural runoff management). This paper examined the removal of TSS, TP, zinc, and fecal coliforms in two palustrine wetlands. The loading and net removal of these pollutants were found to vary by flow conditions (baseflow and storm events), groundwater inputs, season, and residence time. Comprehensive pollutant budgets, based on input (inflow, precipitation, groundwater) and output (outflow, evaporation) calculations revealed the importance of measuring all components when examining pollutant removal in wetlands. Studies that measure pollutant removal only during storm events may overestimate actual removal, because releases during baseflows or during other seasons are not taken into account.

Removal of TSS and TP in the wetland with an urbanized watershed (B3I) occurred only during storm conditions; releases occurred during baseflow conditions, partly as a result of washout of particulate organic matter. The net removal was only 14 and 8% for TSS and TP, respectively. Zinc removal (31%) occurred almost exclusively during storm conditions. Fecal coliforms showed the greatest reductions (nearly 50%), with removal occurring during all flow conditions and all seasons. The low levels of removal for TSS and TP were explained by short residence times, low wetland-to-watershed ratio, high velocities and little capacity for sedimentation. For the wetland with a nonurbanized watershed (PC12), net removals of TSS, TP, zinc, and FC were 56, 82, 23, and 29%, respectively. The higher TSS removal resulted from the longer residence times, lower velocities and, thus, the substantial sedimentation potential of this wetland. The high TP levels in groundwater at PC12 contributed over 80% of the loading. If groundwater flow and sampling had not been carried out, the results of inflow and outflow sampling would have indicated net releases of phosphorus from the wetland (Reinelt et al., 1990). This underscores the importance of performing comprehensive pollutant budget estimates.

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References

Baker, L.A., 1992. Introduction to nonpoint source pollution in the United States and prospects for wetland use. Ecol. Eng., 1: 1–26.

Beak Consultants, Inc., 1992. Water quality and fisheries issues, Chapter 3. In: Beaverdam Master Drainage Plan. King County, WA, pp. 38–46.

Boto, K.G. and W.H. Patrick, Jr., 1979. Role of wetlands in the removal of suspended sediments. In: P.E. Greeson, J.R. Clark and J.E. Clark (Eds.), Wetland Functions and Values: The State of Our Understanding. American Water Resources Association, Minneapolis, MN, pp. 479–489.

Bott, T.L., 1973. Bacteria and the assessment of water quality. Biological Methods for the Assessment of Water Quality, ASTM STP 528, pp. 61–75.

Chan, E., T.A. Bursztynsky, N. Hantzche and Y.J. Litwin, 1981. The Use of Wetlands for Water Pollution Control. EPA-600/S2-82-086. Municipal Environmental Research Laboratory, U.S. EPA, Cincinnati, OH.

Chui, T.W., 1981. Highway runoff in the State of Washington: model verification and statistical analysis. M.S.E thesis, Department of Civil Engineering, University of Washington, Seattle, WA.

Fleischer, S., I.-M. Andréasson, G. Holmgren, A. Joelsson, T. Kindt, L. Ryber and L. Stibe, 1989. Land Use and Water Quality: A Study of Laholm Bay's Drainage Area. Länsstyrelsen i Hallands län, Sweden (in Swedish).

Hammer, D.A. (Ed.), 1989. Constructed Wetlands for Wastewater Treatment. Lewis, Chelsea, MI.

Hammer, D.A., 1992. Designing constructed wetlands systems to treat agricultural nonpoint source pollution. Ecol. Eng., 1: 49–82.

Heliotis, F.D., 1982. Wetland systems for wastewater treatment: operating mechanisms and implications for design. Inst. for Environ. Studies, Report 117, Univ. of Wisconsin, Madison, WI.

Horner, R.R., 1986. A review of wetland water quality functions. In: Wetland Fuctions, Rehabilitation, and Creation in the Pacific Northwest: The State of Our Understanding. Publ. No. 86-14. Washington State Department of Ecology, Olympia, WA, pp. 33–50.

Horner, R.R. and B.W. Mar, 1982. Guide for water quality impact assessment of highway operations and maintenance. Report No. 14, Washington State Department of Transportation, Olympia, WA.

Kadlec, R.H. and J.A. Kadlec, 1979. Wetlands and water quality. In: P.E. Greeson, J.R. Clark and J.E. Clark (Eds.), Wetland Functions and Values: The State of Our Understanding. American Water Resources Association, Minneapolis, MN, pp. 436–456.

King County Resource Planning, 1987. Detailed planning of the Puget Sound Wetlands and Stormwater Management Research Program. King County, Seattle, WA.

Knight, R.L., 1992. Ancillary benefits and potential problems with the use of wetlands for nonpoint source pollution control. Ecol. Eng., 1: 97–113.

Kranck, K., 1984. The role of flocculation in the filtering of particulate matter in estuaries. In: V.S. Kennedy (Ed.), The Estuary as a Filter. Academic Press, Orlando, FL, pp. 159–175.

Lee, G.F., E. Bentley and R. Amundson, 1975. Effects of marshes on water quality. In: A.D. Hasler (Ed.), Coupling of Land and Water Systems. Springer, New York, NY, pp. 105–127.

Little, L.M., R.R. Horner and B.W. Mar, 1983. Assessment of pollutant loadings and concentrations in highway stormwater runoff. Washington State Department of Transportation, Report No. 17, FHWA WA-RD-39.12.1.

Martin, E.H., 1988. Effectiveness of an urban runoff detention pond-wetlands system. J. Environ. Eng., ASCE 114: 810–827.

McElroy, A.D., S.Y. Chiu, J.W. Nebgen, A. Aleti and F.W. Bennett, 1976. Loading Functions for Assessment of Water Pollution from Nonpoint Sources. EPA-600/2-76-151. Midwest Research Institute. Kansas City, MO.

Mitsch, W.J., 1992. Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecol. Eng., 1: 27–47.

Mitsch, W.J. and J.G. Gosselink, 1986. Wetlands. Van Nostrand Reinhold, New York, NY.

Nichols, D.S., 1983. Capacity of natural wetlands to remove nutrients from wastewater. J. Water Pollut. Control Fed., 55: 495–505.

Nixon, S.W. and V. Lee, 1985. Wetlands and water quality: The role of freshwater and saltwater wetlands as sources, sinks, and transformers of N, P, and heavy metals. U.S. Army Corps of Engineers Waterways Research Station, Vicksburg, MS.

Reinelt, L.E. and R.R. Horner, 1990. Characterization of the hydrology and water quality of palustrine wetlands affected by urban stormwater. Puget Sound Wetlands and Stormwater Management Research Program. King County Resource Planning, King County, WA.

Reinelt, L.E., R.R. Horner and H.B. Wittgren, 1990. Removal of sediments and nutrients in palustrine wetlands receiving runoff from urban and nonurban watersheds. In: L. Reinelt (Ed.), Nonpoint Source Water Pollution Management. Linköping Studies in Arts and Sciences 57. Linköping Univ., pp. 3–25.

Reinelt, L.E., J. Velikanje and E.J. Bell, 1991. Development and application of a geographic information system for wetland/watershed analysis. Computers, Environ. Urban Syst., 15: 239–251.

Reinelt, L.E., M. Surowiec and R.R. Horner, 1993. Urbanization effects on palustrine wetland hydrology as determined by a comprehensive water balance. Wetlands J. (submitted).

Richardson, C.J., D.L. Tilton, J.A. Kadlec, J.P.M. Chamie and W.A. Wentz, 1978. Nutrient dynamics of northern wetland ecosystems. In: R.E. Good, D.F. Whigham and R.L. Simpson (Eds.), Freshwater Wetlands, Ecological Processes and Management Potential. Academic Press, New York, NY, pp. 217–241.

Rodgers, J.H. Jr. and A. Dunn, 1992. Developing guidelines for constructed wetlands to remove pesticides from agricultural runoff. Ecol. Eng., 1: 83–95.

Schubel, J.R. and H.S. Carter, 1984. The estuary as a filter for fine-grained suspended sediment. In: V.S. Kennedy (Ed.), The Estuary as a Filter. Academic Press, Orlando, FL., pp. 81–105.

Schueler, T.R., 1993. Design of Stormwater Wetland Systems. Metropolitan Council of Governments, Washington, DC.

Stockdale, E.C., 1991. The Use of Wetlands for Stormwater Management and Nonpoint Pollution Control: A Review of the Literature. Washington State Department of Ecology, Olympia, WA.

Strecker, E.W., J.M. Kersnar, E.D. Driscoll and R.R. Horner, 1992. The Use of Wetlands for Controlling Stormwater Pollution. Terrene Institute, Washington, DC.

U.S. Environmental Protection Agency, 1983. Methods for Chemical Analysis of Water and Wastes. EPA-600/4-70-020, Environmental Monitoring and Support Laboratory, Cincinnati, OH.

U.S. Environmental Protection Agency, 1984. Report to Congress: Nonpoint Source Pollution in the U.S. Operations Water Planning Division, Washington, DC.

U.S. Environmental Protection Agency, 1985. Freshwater Wetlands for Wastewater Management Environmental Assessment Handbook. EPA 904/9-83-107. Region IV, Atlanta, GA.

W&H Pacific, 1992. Predicted Water Quality Impacts for the Proposed Trossach's Development. Draft EIS, King County, WA.

Zison, S.W., 1980. Sediment-Pollutant Relationships in Runoff from Selected Agricultural, Suburban and Urban Watersheds: A Statistical Correlation Study. EPA-600/3-80-22, Environmental Research Laboratory, Athens, GA.