Wetlands and Aquatic Processes

Nutrient and Sediment Removal by a Restored Wetland Receiving Agricultural Runoff

Thomas E. Jordan,* Dennis F. Whigham, Kirsten H. Hofmockel, and Mary A. Pittek

ABSTRACT

Few studies have measured removal of pollutants by restored wetlands that receive highly variable inflows. We used automated flow-proportional sampling to monitor the removal of nutrients and suspended solids by a 1.3-ha restored wetland receiving unregulated inflows from a 14-ha agricultural watershed in Maryland, USA. Water entered the wetland mainly in brief pulses of runoff, which sometimes exceeded the 2500-m^3 water holding capacity of the wetland. Half of the total water inflow occurred in only 24 days scattered throughout the two-year study. Measured annual water gains were within 5% of balancing water losses. Annual removal of nutrients differed greatly between the two years of the study. The most removal occurred in the first year, which included a three-month period of decreasing water level in the wetland. In that year, the wetland removed 59% of the total P, 38% of the total N, and 41% of the total organic C it received. However, in the second year, which lacked a drying period, there was no significant ($p > 0.05$) net removal of total N or P, although 30% of the total organic C input was removed. For the entire two-year period, the wetland removed 25% of the ammonium, 52% of the nitrate, and 34% of the organic C it received, but there was no significant net removal of total suspended solids (TSS) or other forms of N and P. Although the variability of inflow may have decreased the capacity of the wetland to remove materials, the wetland still reduced nonpoint-source pollution.

THROUGHOUT THE WORLD, fluvial discharges of nutrients and sediments have increased due to increasing fertilizer applications to croplands (e.g., Turner and Rabalais, 1991; Jordan and Weller, 1996; Howarth et al., 1996), increasing concentration of livestock waste production (e.g., Sims and Wolf, 1994), and land-cover changes that enhance erosion (Woodward and Foster, 1997). Such nonpoint sources of pollutants have had significant detrimental effects on freshwater and coastal ecosystems. Nonpoint-source discharges contribute about two-thirds of the nitrogen and one-quarter of the phosphorus inputs to Chesapeake Bay (Correll, 1987), one of the world's largest estuaries. Increases in the inputs of both nitrogen and phosphorus to Chesapeake Bay have led to excessive phytoplankton production (Malone et al., 1986, 1988; Boynton et al., 1982; Correll, 1987; Jordan et al., 1991a,b; Gallegos et al., 1992) that has contributed to the demise of submerged aquatic vegetation (Kemp et al., 1983) and the increase in the extent of hypoxic waters (Taft et al., 1980; Officer et al., 1984).

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Preserving or restoring wetlands may help reduce nonpoint-source pollution. Wetlands can act as filters removing particulate material, as sinks accumulating nutrients, or as transformers converting nutrients to different forms, such as gaseous compounds of nitrogen (N) and carbon (C) (Richardson, 1989). Recent research has shown that constructed or restored wetlands can remove sediments and nutrients from nonpoint sources, including agricultural discharges (e.g., Fleischer et al., 1994; Mitsch, 1994; Raisin and Mitchell, 1995; Whigham, 1995; Jordan et al., 1999). Widespread restoration of wetlands has been suggested as part of a plan for reducing nitrogen releases from the Mississippi River basin (Mitsch et al., 2001).

Nutrient removal by constructed wetlands has been extensively studied for their use in wastewater treatment (Hammer, 1989; Kadlec and Knight, 1996). However, wetlands constructed for wastewater treatment usually receive measured and controlled inflows of wastewater. Also, the outflows from wastewater treatment wetlands are usually monitored to check the wetland's performance. Therefore, much is known about the capabilities and design criteria of such wetlands (Hammer, 1989; Kadlec and Knight, 1996). Much less is known about the nutrient and sediment removal capabilities of natural and restored wetlands that receive unregulated inflows. Unregulated flows are more difficult to measure than the regulated flows. Automated sampling is generally required to quantify unregulated event-driven fluxes (e.g., Kovacic et al., 2000; Braskerud, 2002).

In general, the ability of a wetland to trap or transform nutrients increases as the water retention time increases. Models incorporating the effects of water retention time are used in designing treatment wetlands (Kadlec and Knight, 1996). Similar effects of water retention in natural and restored wetlands have been suggested by several studies (e.g., Mitsch et al., 1995; Carleton et al., 2001). Water retention time may vary widely with weather and season in wetlands with unregulated inflows. Variability of water flow may diminish the ability of wetlands to remove nutrients and sediments, as removal capacities may be temporarily overwhelmed during short-lived high flow events (e.g., Kovacic et al., 2000).

Wetlands are being restored in agricultural watersheds to provide wildlife habitat as well as improve water quality (Whigham, 1995). Some restorations involve minimal alterations of drainage, which produce

Abbreviations: TN, total nitrogen; TNH_4^+, total ammonium; TOC, total organic carbon; TON, total organic nitrogen; TOP, total organic phosphorus; TPO_4^{3-}, total phosphate; TSS, total suspended solids.

wetlands with highly variable inflow rates (e.g., Magner et al., 1995). The objective of this study was to quantify the removal of nutrients and sediments of one such wetland under a highly variable flow regime. This wetland, on the Chesapeake Bay shore, receives cropland runoff and may serve as a model for systems mitigating nonpoint-source nutrient discharges toward the goal of 40% reduction set by the Chesapeake Bay Program (1997). We hypothesized that the wetland would remove nutrients and sediments although the variable inflow rate would reduce the removal efficiencies compared with similar wetlands with more constant inflow rates. To test this hypothesis, we monitored fluxes of water, nutrients, and sediments into and out of the wetland for two years using an automated sampling system to permit observation of unpredictable episodes of high flow.

MATERIALS AND METHODS

Study Site

The study wetland is on Kent Island, Maryland, which is part of the Delmarva Peninsula on the eastern shore of the Chesapeake Bay (Fig. 1). Much of the surrounding land is in agriculture, primarily corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] production. The 14-ha watershed of the 1.3-ha wetland was 18% forest and 82% cropland planted to corn in 1995 and 1997, and to soybean in 1996. The average slope of the watershed is less than 1%. The soils in most of the study area are Typic Endoaquults of the Othello soil series (fine-silty, mixed, active, mesic Typic Endoaquult) but the watershed also includes Aquic Hapludults of the Mattapex series (fine-silty, mixed, active, mesic Aquic Hapludult) (Matthews and Reybold, 1966). These soils have a silt loam texture with a moderate or moderately slow permeability. The silt loam surface layers are underlain by a silty clay loam subsurface horizon with 18 to 30% clay at 0.2 to 1 m below the surface forming an extensive and continuous aquiclude. Because of the low permeability of the soils and the low topographic relief, most croplands in the study area are drained by ditches or by plowed channels that discharge water into wetlands, streams, riparian forests, or directly into the Chesapeake Bay. Artificial drainage has converted some wetlands to croplands.

The study wetland (called "Barnstable 1" by Jordan et al., 1999) had been artificially drained cropland before being restored to wetland in 1986 by the Chesapeake Wildlife Heritage as part of their program to provide wildlife habitat and improve the quality of runoff from agricultural fields. During restoration, a layer of soil was removed to create a depression that was <1 m deep. Some of the excavated soil was used to create a low dike to retain water at depths of <1 m, averaging 0.2 m. After excavation, topsoil was returned to the surface and wetland vegetation was established by natural succession. The three most dominant macrophyte species in the wetland were blunt spikerush [*Eleocharis obtusa* (Willd.) Schult.], water-purslane [*Ludwigia palustris* (L.) Elliott], and American bulrush [*Schoenoplectus americanus* (Pers.) Volkart ex Schinz & R. Keller] (Jordan et al., 1999). Whigham et al. (2002) describe the patterns of biomass and nutrient distribution in the vegetation over a three-year period in this wetland and in 11 other restored wetlands in the area. Emergent vegetation covered a maximum of 70 to 90% of the wetland surface during the growing season and 10 to 20% of the surface during the nongrowing season.

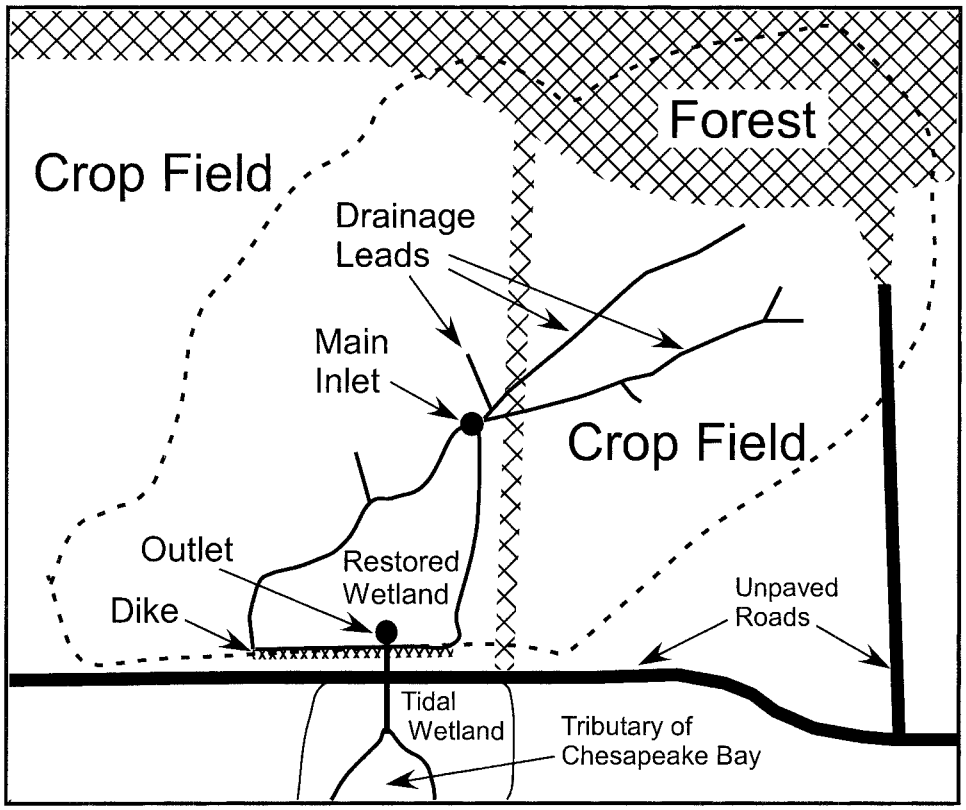

Fig. 1. The study wetland and its watershed (outlined by dashed lines). Automated samplers were located at the labeled outlet and main inlet points. The restored wetland drained through a tidal wetland into a tidal tributary of Chesapeake Bay.

Water enters the wetland via drainage leads carrying surface runoff from the surrounding watershed and via precipitation directly on the wetland surface. Water leaves the wetland via the standpipe drain installed in the dike and via evapotranspiration. When the water was deep enough to flow out the drain, the entire 1.3-ha area of the wetland was submerged and lacked well-defined flow channels. Ground water exchanges are negligible due to the impermeable layer of clay within 0.5 m of the soil surface. We concluded that the clay layer blocks water infiltration, because clay sampled from beneath inundated areas was dry.

Measuring Water Flow

We used automated instruments to measure water flow and to sample water entering and leaving the wetland from 8 May 1995 through 12 May 1997. The instruments included a depth sensor consisting of a float and counter weight suspended in a stilling well that was connected to the impounded water near the wetland drain. A CR10 datalogger (Campbell Scientific, Logan, UT), housed atop the stilling well, recorded the position of the float to monitor water depth. Outflowing water passed over a 120° V-notch weir at the drainpipe. The outflow rate was calculated from the depth of water in the V-notch.

The total rate of water input to the wetland from runoff and direct precipitation combined was calculated by summing the rate of outflow and the rate of increase in water volume held in the wetland, with decrease in volume treated as negative increase. Thus, if water volume in the wetland remained constant, then the total water input rate was assumed to equal the outflow rate; if water volume increased, then the total input rate was assumed to equal the outflow rate plus the rate of increase in volume; and if the water volume decreased, then the total input rate was assumed to equal the outflow rate minus the decrease in volume. This method of calculating the total input of water yielded negative values for input when there was no surface water flow and evapotranspiration decreased the water volume in the wetland. Therefore, we interpreted negative values of total input as indicative of zero input from runoff and precipitation. We also assumed that evapotranspiration was negligible during periods of precipitation and runoff input. The volume of water in the wetland was calculated from water depth and the areas enclosed in 10-cm elevation contours within the wetland basin, which was surveyed with a Total Station CTS-2/2B (Topcon, Tokyo, Japan).

The water input from precipitation directly on the wetland surface was calculated from the surveyed wetland area (1.3 ha) and the precipitation volume measured with standard rain gauges at the wetland and at the Wye Research Center (WRC), 13 km from the wetland. The WRC precipitation data were obtained from the Maryland State Climatologist. The amount of water the wetland received from watershed runoff was calculated by subtracting the direct precipitation input from the measured total water input from runoff and precipitation combined. Evapotranspiration from the wetland was estimated using data from a standard weather-bureau evaporation pan at the Smithsonian Environmental Research Center (SERC), 25 km from the wetland.

Sampling Water

The datalogger controlled pumps that collected separate samples of inflowing runoff and outflowing water in volumes proportional to the respective flow rates. This produced volume-weighted composite samples that represented the water quality of the inflowing runoff and outflowing water. The logger calculated the rates of outflow and total inflow (the sum of runoff plus precipitation) every 15 min. For the purposes of controlling sample pumping, total inflow was assumed to be proportional to runoff. When the amount of outflow or total inflow since the last pumping exceeded a certain threshold, the logger activated the appropriate pump to collect a volume of water proportional to the amount of flow since the last pumping. Thus, the frequency of pumping and the amount of water pumped each time could both vary. During high flow events, the pumps could be activated as often as every 15 min. Increasing the frequency of pumping during high flow events was important because concentrations can change rapidly during runoff events. However, it was also necessary to vary the volume pumped each time because the flow threshold for triggering pumping could be exceeded by different amounts during different 15-min measurement cycles. The pumps sampling inflowing runoff and outflow were controlled independently since inflow and outflow usually differed in timing and rate. The logger recorded each time when each pump was activated and how long it was pumping for each sampling. The signal to activate the inflow pump was transmitted via wire from the logger to the inflow pump located near the drainage ditch that collects the runoff from about 70% of the wetland's watershed. Inflowing runoff was sampled from about 5 cm above the bottom of this ditch. Previous comparisons showed that runoff carried by the main ditch had similar chemical composition to runoff entering from two other points (Jordan et al., 1999). Outflowing water was pumped from the water column near the V-notch weir at the wetland drain (Fig. 1). A submersible impeller pump (Model 1P811A; Teel) was used to sample outflow but a self-priming peristaltic pump (Model LG100; Little Giant Pump Co., Oklahoma City, OK) was needed to sample inflow because the inlet ditch usually dried up between runoff events. At both the inflow and outflow sampling points, the samples were pumped through plastic tubing, which was first rinsed with stream water. The pumped sample stream was split between two carboys, one with about 3 mL of sulfuric acid per liter of sample added as a preservative and one without preservative. In addition to the automatically collected composite samples, grab samples were collected whenever there was water flowing at the inlet or outlet during the weekly visits to retrieve the composite samples.

Each week the composite samples that accumulated during the week were brought into the laboratory for analysis. The acid-preserved samples were analyzed for nutrient concentrations. The measured nutrient concentrations represent the total of dissolved nutrients plus particulate nutrients that were dissolved by the acid preservative. The unpreserved samples collected by the automated instruments were analyzed for total suspended solids (TSS). Unpreserved grab samples were analyzed for pH and conductivity immediately after return to the laboratory.

The content of N, P, and organic C in bulk precipitation was measured in an ongoing monitoring program at the Smithsonian Environmental Research Center (e.g., Correll et al., 1994; Jordan et al., 1995). After each event of more than 5 mm of precipitation, samples of bulk precipitation were collected with a 28-cm-diameter polyethylene funnel and bottle. These samples were analyzed by the same methods as for the wetland water samples.

Chemical Analyses

Standard techniques were used for analysis of nitrogen (N) and phosphorus (P) compounds. Samples for total inorganic nutrients (including originally dissolved species and those dis-

solved by the acid preservative) were filtered before analysis with prewashed 0.45-μm Millipore (Bedford, MA) filters. Total Kjeldahl nitrogen (TKN), total P, and total organic C were measured on unfiltered samples. Total P was digested to phosphate with perchloric acid (King, 1932). Phosphate in the digestate and total phosphate (TPO_4^{3-}) in undigested samples were analyzed by reaction with stannous chloride and ammonium molybdate (American Public Health Association, 1995). The TKN was digested with sulfuric acid, Hengar granules, and hydrogen peroxide (Martin, 1972). The resultant ammonia was distilled and analyzed with a Dionex (Sunnyvale, CA) ion chromatograph. In undigested aliquots, total ammonium (TNH_4^+) was oxidized to nitrite by alkaline hypochlorite (Strickland and Parsons, 1972), dissolved nitrate was reduced to nitrite by cadmium amalgam, and then the nitrite was analyzed by reaction with sulfanilamide (American Public Health Association, 1995). We present data on the sum of nitrite and nitrate concentrations, which we refer to as NO_3^-. From results of the above analyses we calculated total nitrogen (TN) by adding NO_3^- to TKN, total organic nitrogen (TON) by subtracting TNH_4^+ from TKN, and total organic phosphorus (TOP) by subtracting TPO_4^{3-} from total P.

Total organic carbon (TOC) was analyzed as chemical oxygen demand by drying samples at 60°C, followed by reaction with potassium dichromate in 67% sulfuric acid at 100°C for 3 h (Maciolek, 1962; American Public Health Association, 1995). Organic carbon was calculated from the amount of unreacted dichromate measured colorimetrically (Maciolek, 1962; Gaudy and Ramanathan, 1964).

Total suspended solids (TSS) were measured by filtering the nonacidified samples through prewashed, preweighed 0.40-μm Nuclepore filters (Whatman, Maidstone, UK), after which the filters were dried and reweighed.

Measurements of pH were made on air-equilibrated samples with an expanded range pH meter and a ROSS electrode (Thermo Orion, Beverly, MA). Conductivity was measured with a Model 32m conductivity meter (YSI, Yellow Springs, OH).

Statistical Analyses

To assess whether annual net fluxes were statistically significant, we calculated the 95% confidence limits around the annual net fluxes using the bootstrap technique (Efron, 1982). Differences among annual fluxes may arise due to the variability among weekly fluxes, because a few weeks with high flux can dominate the calculation of the annual flux. The bootstrap technique measures the consequences of randomly including or excluding certain weekly fluxes from the calculation of the annual flux. In other words, it accounts for the chance occurrence of weeks with differing water and nutrient flow within a given year. The bootstrap procedure begins by creating 1000 sets of data by randomly selecting data points from the original data set, replacing the selected points so they can be chosen again. In this case, the data points are the weekly net fluxes within the one-year or combined two-year periods. We included only weekly net fluxes for weeks when measurements were available for both influx and outflux. We did not include weeks for which net flux was estimated due to missing measurements. Each of the data sets created by the bootstrap procedure has the same number of samples as the original data set. The means of the created sets are calculated, and the 2.5 and 97.5 percentiles of these means represent the 95% confidence limits of the original mean (Efron, 1982). If the confidence limits of the net flux do not overlap zero, then the net flux is significantly different from zero at the $p < 0.05$ level. The bootstrap analysis and all other statistical analyses were performed using the Statistical Analysis System (SAS Institute, 1989).

RESULTS

Water Flow

Water depth and volume in the wetland varied throughout the study. When the wetland was full, runoff events caused short-lived peaks in water depth, reaching up to 42 cm above the bottom of the weir (Fig. 2). The water level dropped below the bottom of the weir preventing surface outflow during the summer of 1995, but water level stayed above the bottom of the weir during most of the summer of 1996 (Fig. 2). The differences in summer water levels reflect differences in summer rainfall, which amounted to 110 mm in July–August 1995 versus 270 mm in July–August 1996. Although rainfall differed greatly between the summers, the total annual precipitation was similar for the two years of the study: 1090 mm for the first year and 1150 mm for the second, amounts close to the long-term annual mean for the area, 1090 mm (Correll et al., 1994). The volume of water in the wetland ranged from 6900 m^3 when depth was 42 cm above the bottom of the V-notch weir to less than 350 m^3 during the 1995 summer draw-down when depths were more than 40 cm below the bottom of the weir (Fig. 2). When the water depth reached the bottom of the weir without spilling over, the wetland was at its maximum water holding capacity of 2500 m^3.

Most flow events lasted only a few hours, as illustrated by the flow rates during the fourth largest event of our study (Fig. 3). Inflow from precipitation and runoff combined was typically more rapid than outflow (e.g., Fig. 3). The duration of inflow was influenced by the duration of the rain event as well as the rate of runoff from the watershed.

Measurements of the total inflow rate from runoff plus precipitation were less precise than measurements of outflow rates because the total inflow rate was calculated from the outflow rate and the rate of change of water level in the wetland. Water level changes of a millimeter or less implied flows of substantial volumes of water. Thus, variance in measuring water level introduced noticeable variance into our measurements of inflow, as in the flow event of 13 Aug. 1996 when inflow rates appear to fluctuate rapidly by a few hundredths

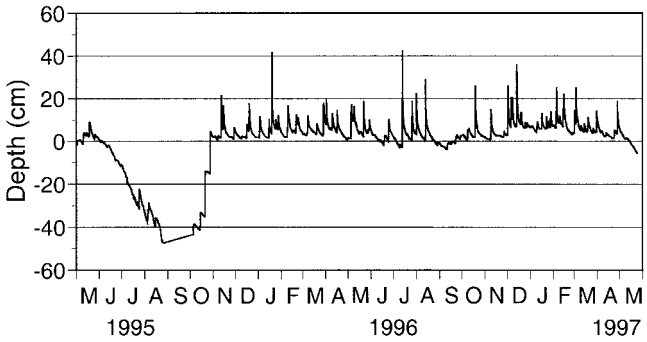

Fig. 2. Water depth vs. time. Depth is relative to the bottom of the V-notch in the weir at the wetland drain. Thus, outflow occurs only when depths are above zero.

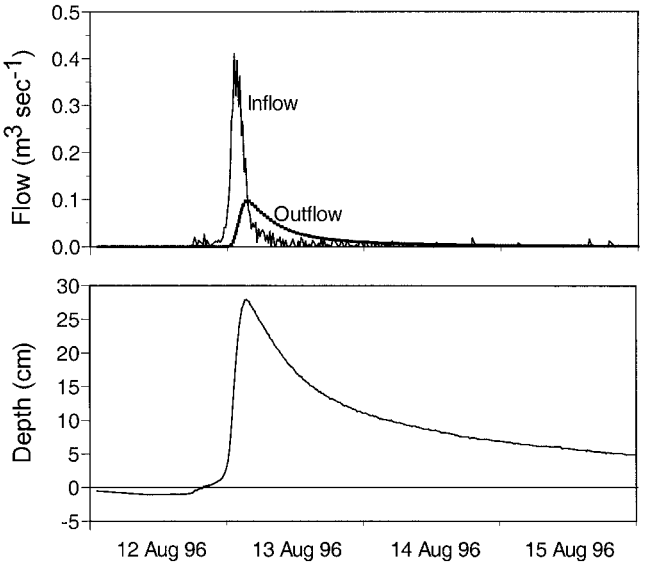

Fig. 3. Total inflow (from runoff and direct precipitation combined) and outflow of water and depth relative to bottom of V-notch in weir during a four-day period in August 1996. A rainfall and runoff event began late on 12 August.

of m^3 s^{-1} (Fig. 3). Although this short-term variance introduces uncertainty in instantaneous measurements of inflow rate, it has little effect on measurements of total daily inflow from precipitation plus runoff.

Inflow to the wetland was episodic, depending almost entirely on rain events. Half of the total water inflow occurred in only 24 d scattered throughout the two-year study. During eight of those days, outflow exceeded 2500 m^3, the water holding capacity of the wetland (Fig. 4). Even on a weekly basis, total water inflow to the wetland was very uneven (Fig. 5). More than half of the total annual inflow occurred during only 12 weeks scattered throughout the study.

The hydraulic loading rate (inflow divided by wetland area) and the water detention time (wetland volume divided by inflow rate) varied with changes in wetland volume and inflow rate. To calculate water detention time, we estimated the average wetland volume during a given time period from the average outflow rate and the relationships among outflow rate, depth, and volume. Calculated over the first and second years of our study, hydraulic loading rates were 12 and 20 mm d^{-1}, respectively, and detention times were 19 and 12 d, respectively. However, maximum hydraulic loading rate

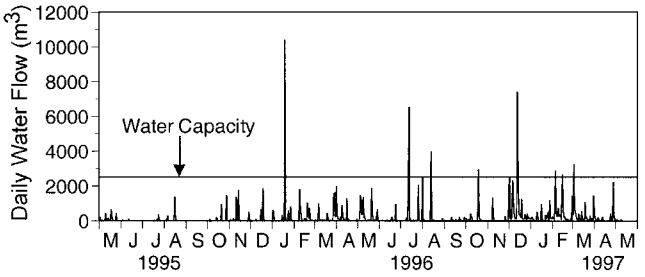

Fig. 4. Daily outflow of water vs. time for the wetland. The horizontal line at 2500 m^3 indicates the water capacity (i.e., the volume held by the wetland before outflow occurs).

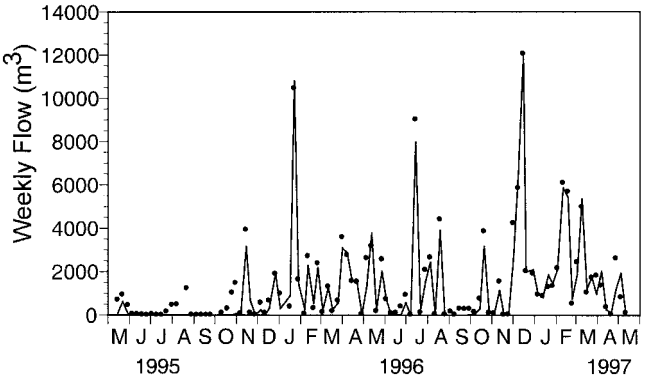

Fig. 5. Total inflow (points) and outflow (line) of water vs. time during the sampling intervals, which are approximately one week long.

was much higher and minimum detention time much lower when calculated over shorter time scales. For example, on 19 January 1996, when daily inflow reached its maximum of 10 400 m^3 (Fig. 4), the hydraulic loading rate was 800 mm d^{-1} and the detention time was 0.51 d. During the weekly water-sampling interval with highest inflow (9–16 Dec. 1996, 12 000 m^3; Fig. 5), the hydraulic loading rate was 130 mm d^{-1} and the detention time was 2.2 d. In contrast, during the 71 d and seven water-sampling intervals when there was no inflow, the hydraulic loading rate was zero and detention time was incalculable.

Our estimates of annual water gains and losses in the wetland come within 5% of balancing (Table 1). This close agreement supports our assumption that the underlying clay layer prevented ground water exchanges. Runoff from the watershed was the main source of water input. The total input from runoff and rainfall over the two years (150 200 m^3) was about 60 times the water holding capacity of the wetland (2500 m^3). Surface outflow was the main water loss. Surface flows were especially dominant in the second year of the study, which had a wetter summer than the first year. Annual net change in standing water volume was orders of magnitude smaller than the surface flow. Our estimate of evapotranspiration, assumed equal to pan evaporation, is probably the least certain component of the water

Table 1. Water gains and losses for the wetland. Evapotranspiration is assumed to equal pan evaporation. Net gain in stored water is the net increase in the water volume held in the wetland at the end of the time period compared to the beginning of the period. The negative net gain during Year 2 represents a net loss or decrease in stored volume. The amount by which gains exceed losses (the excess gain) represents the imbalance of the budget. The imbalance may result from errors in measurements or from an unaccounted loss of water from the wetland, such as seepage through the dike.

	Year 1	Year 2	Both years
		m^3	
Gains			
Runoff from watershed	41 184	79 931	121 115
Direct precipitation	14 174	14 888	29 062
Losses			
Surface outflow	42 321	86 174	128 495
Evapotranspiration	9 865	8 697	18 562
Net gain in stored water	564	−437	127
Excess gain	2 608	385	2 993

budget. However, pan evaporation (Table 1) was very similar to potential evapotranspiration estimated from the Thornthwaite equation (Veihmeyer, 1964), which was 770 mm yr^{-1} or 9995 m^3 yr^{-1} for the whole wetland.

Concentrations of Nutrients and Suspended Solids

The concentrations of materials in inflowing runoff and outflowing water varied greatly from week-to-week with no clear seasonal pattern (Fig. 6, 7, 8). Concentrations in inflowing runoff were generally higher than in outflowing water. This tendency was most consistent for TOC. Both TOC and TON concentrations were consistently much higher in inflowing runoff than in outflowing water in the spring and summer of 1996 (Fig. 6). However, after the fall of 1996, TON concentrations in runoff dropped below concentrations in outflow, while TOC concentrations in runoff decreased but remained higher than TOC concentrations in outflow (Fig. 6). For some materials, unusually high concentrations occurred sporadically. For example, in the first year, inflowing runoff had exceptionally high concentrations of TOP (Fig. 6) and TPO_4^{3-} (Fig. 7) for a few weeks and exceptionally high concentration of NO_3^- in one week (Fig. 8). The highest NO_3^- concentration occurred immediately after the prolonged period of low flow in the summer of 1995 (Fig. 8). Concentrations of NO_3^- remained elevated from November 1995 through February 1996.

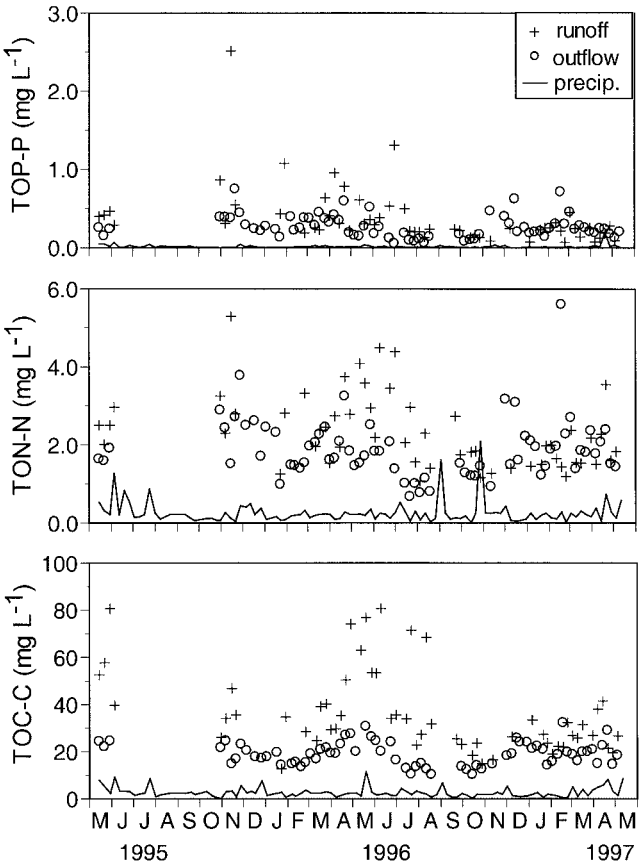

Fig. 6. Concentrations of total organic nitrogen (TON), phosphorus (TOP), and carbon (TOC) in inflowing runoff (+), out flowing water (o), and precipitation (line) in mg L^{-1} N, P, or C.

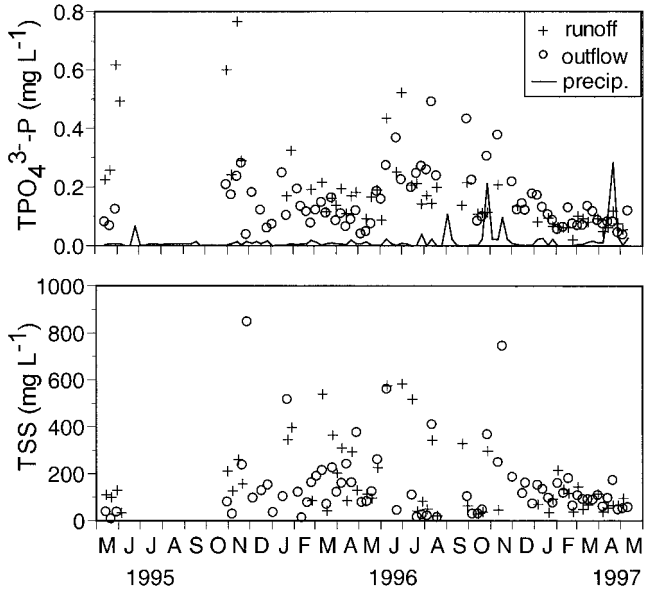

Fig. 7. Concentrations of total phosphate (TPO_4^{3-}) and total suspended solids (TSS) in inflowing runoff (+), out flowing water (o), and precipitation (line) in mg L^{-1} P or TSS.

Based on analyses of grab samples, pH and conductivity were lower in inflowing runoff than in outflowing water. In inflowing runoff, pH averaged 5.2 (SD = 0.93) and conductivity averaged 10.6 mS m^{-1} (SD = 39). By comparison, in outflowing water, pH averaged 6.8 (SD = 0.40) and conductivity averaged 12.3 mS m^{-1} (SD = 36).

The concentrations of some materials were serially correlated (i.e., there were multiweek trends in concentrations). We assessed the serial correlation for each material from the correlation between its concentration in each week with the average of its concentrations in

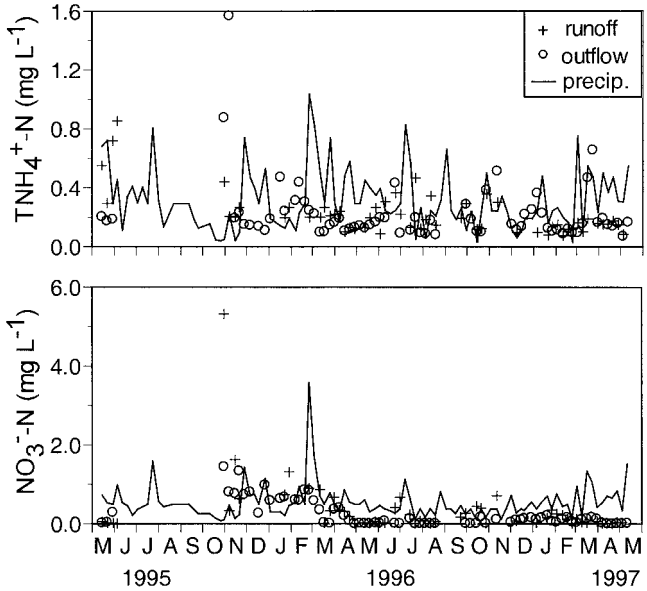

Fig. 8. Concentrations of total ammonium (TNH_4^+) and nitrate plus nitrite (NO_3^-) in inflowing runoff (+), out flowing water (o), and precipitation (line) in mg L^{-1} N.

the weeks immediately before and after. The concentrations of all of the materials in outflow except TSS had significant serial correlation (Pearson, $p < 0.05$). Among outflow concentrations, NO_3^- had the strongest serial correlation with an r^2 of 0.76, TPO_4^{-3} and TOC had the next highest serial correlations with r^2 values of 0.37 and 0.33, respectively, and correlations for other materials ranged down to $r^2 = 0.10$ for TON. Materials in inflowing runoff showed less serial correlation than materials in outflowing water. Only two materials in inflowing runoff had significant serial correlation: TPO_4^{-3} and TOC with r^2 values of only 0.18 and 0.29, respectively.

In some cases the concentration of a material in outflow was correlated with its concentration in inflowing runoff. Such correlations are more likely as the volume of outflow increases, indicating more rapid passage of water through the wetland. We analyzed correlations of runoff and outflow concentrations for weeks with outflow volumes of more than 500 m^3 (i.e., more than 20% of the water holding capacity of the wetland). These represented 47% of the weeks studied and accounted for 97% of the outflow that occurred. During these weeks, concentrations in inflowing runoff and in outflowing water were significantly correlated (Pearson, $p < 0.05$) for NO_3^-, TPO_4^{-3}, TSS, TNH_4^+, and TOC, with r^2 values of 0.46, 0.25, 0.24, 0.18, and 0.18, respectively.

Correlations between concentrations and water flow rates might be expected due to the effects of flow on erosion, resuspension, or dilution. However, the only significant correlations (Pearson, $p < 0.05$) we observed were for TNH_4^+ and TOC in inflowing runoff, which were negatively correlated with inflow rate ($r^2 = 0.11$ and 0.12, respectively). The weak negative correlations suggest a slight tendency for high water flows to dilute the materials. Relationships between concentration and flow rate may be difficult to demonstrate with our weekly data because weeks differ not only in the total amount of flow but also in how the flow is distributed during the week. For example, a week with only moderate total flow may include a short period of very high flow that could affect concentrations in the weekly composite samples. Therefore, event-based sampling might be needed to reveal correlations between concentrations and flow rates. However, conditions antecedent to the flow event, such as soil saturation, may also influence the effect of water flow rate and thereby obscure correlations between flow and concentration.

Concentrations of materials other than TNH_4^+ and NO_3^- were usually much higher in inflowing runoff than in precipitation (Fig. 6 and 7). Therefore, precipitation falling directly onto the wetland surface dilutes most materials entering the wetland via runoff. This dilution is important, although only about one-fifth of the water entering the wetland enters via direct precipitation (Table 1). Rather than diluting TNH_4^+ and NO_3^-, direct precipitation represents a considerable source of those materials to the wetland because their concentrations were similar in precipitation and runoff (Fig. 8).

Fluxes of Nutrients and Suspended Solids

We calculated fluxes of materials based on weekly concentrations and water flows. Our concentration measurements represented most of the water flow volume but were more complete for outflow than for inflow. The percentage of outflow volume for which we measured concentrations was 90% for TSS, 96% for TOC, and 99% for forms of N and P. The percentage of inflowing runoff for which we measured concentrations was 69% for TSS and 74% for other materials.

When concentration data were missing, we substituted annual flow-weighted mean concentrations, which were calculated from all the available measurements. To avoid bias in weekly net flux calculations, we also substituted flow-weighted means for measured concentrations when concentrations in the opposing flow were not available. This usually applied to weeks when concentrations were measured in outflow but not in inflowing runoff. In such cases we would substitute mean concentrations for outflow as well as for runoff. We followed this procedure because runoff concentrations of NO_3^-, TPO_4^{-3}, TSS, TNH_4^+, and TOC correlated with their respective outflow concentrations. Thus, substituting a mean concentration for only one of the flow directions could create an artificial imbalance of concentrations leading to a less accurate estimate of net flux.

The protocol for filling missing data was selected because it could be applied consistently for all the missing data and because alternate protocols would require predictions based on weak correlations. In some cases, missing concentrations of NO_3^-, TPO_4^{-3}, TSS, TNH_4^+, and TOC could have been predicted from the correlations between their concentrations in runoff and outflow. However, those correlations had r^2 values of 0.46 to 0.18 and would therefore provide a poor basis for making predictions. Moreover, a measured concentration was not always available to use for predicting a missing concentration in the opposing flow. Similarly, the weak serial correlations for outflow concentrations would yield only imprecise predictions of missing values and could be applied in only a limited number of cases.

The total annual influxes of nutrients and TSS differed between the two years of our study, but generally not as much as would be expected from the difference between the annual water flows. Influxes of TOP and NO_3^- were actually greater in the first year than in the second year (Fig. 9). Influxes of other nutrients were greater in the second year than in the first year (Fig. 9), but not by a factor of two, which was the difference between the annual water inflows from runoff (Table 1). Clearly, the increases in inflows of water from runoff in the second year were offset by decreases in concentrations of materials in the inflowing runoff. Direct atmospheric deposition provided <4% of the total annual influx of most materials but provided 28 to 33% of the influx of TNH_4^+ and 18 to 36% of the influx of NO_3^- (Fig. 9). Unlike influxes, the annual outfluxes of TOP, TPO_4^{3-}, TON, and TOC were elevated in the second year to about the same extent as the outflow of water (Fig. 9, Table 1). However, this was not true for NO_3^-

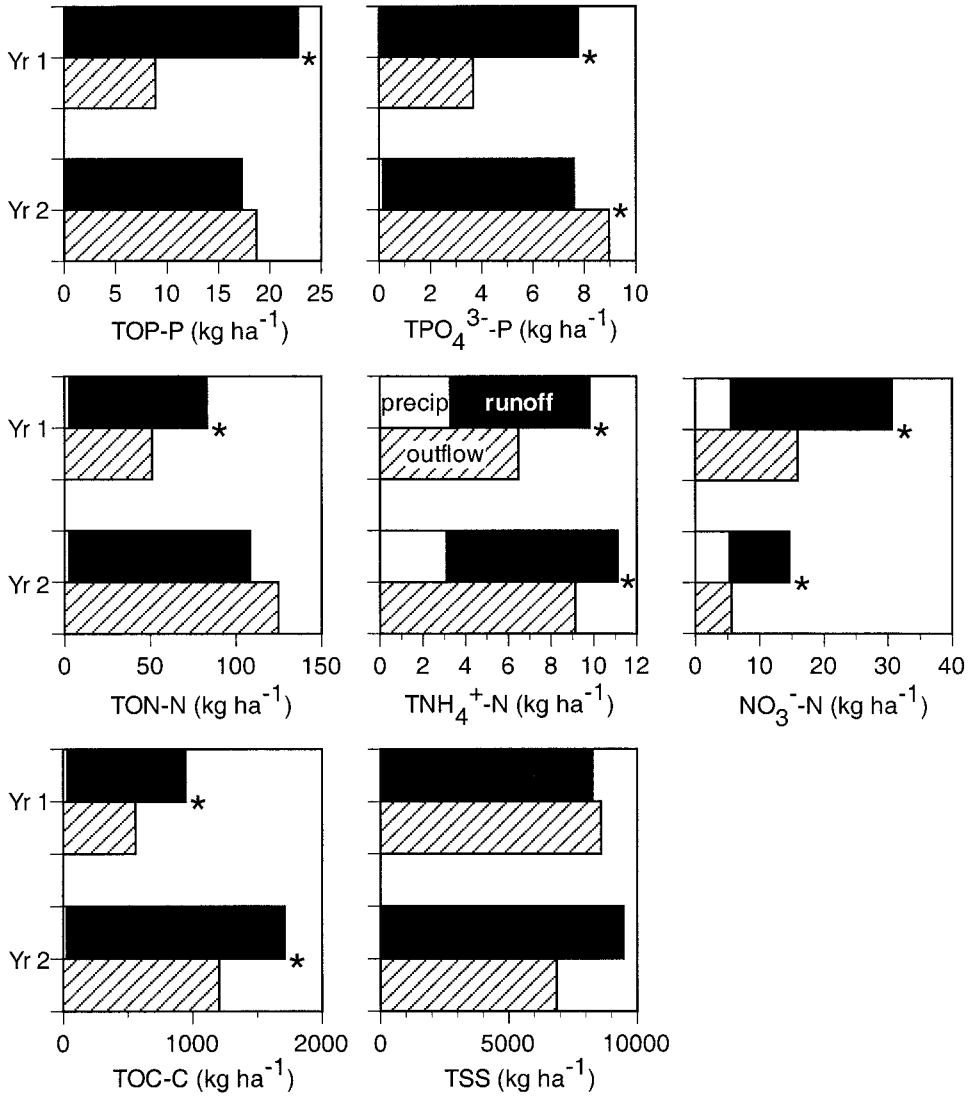

Fig. 9. Annual total inflows and outflows of materials during the two years of the study. Inflows from runoff (solid bar) and direct precipitation (unshaded bar) are shown by stacked bars, and outflow by a cross-hatched bar, as labeled for total ammonium. Asterisks indicate statistically significant differences between inflow (precipitation plus runoff) and outflow ($p < 0.05$, bootstrap). TNH_4^+, total ammonium; TOC, total organic carbon; TON, total organic nitrogen; TOP, total organic phosphorus; TPO_4^{3-}, total phosphate; TSS, total suspended solids.

and TSS, which flowed out of the wetland in lesser amounts during the second year than during the first (Fig. 9). In the first year, influxes of TOC and all forms of P and N exceeded outfluxes, suggesting a net removal of these materials by the wetland. In the second year, there were apparent net releases of TOP, TPO_4^{3-}, and TON from the wetland, while other materials appeared to be removed by the wetland.

The net influxes of materials varied from week to week with no clear seasonal pattern (Fig. 10). For many materials, there were exceptionally high net influxes or net outfluxes during a few of the weeks. These few weeks with exceptionally high net fluxes accounted for a high proportion of the annual net flux. For example, exceptionally high net influx of TOP, TPO_4^{3-}, TON, NO_3^-, and TOC occurred during the week ending 15 Nov. 1995. This week was characterized by moderately high water flows (the highest for that month; Fig. 4 and 5), and high material concentrations in inflowing runoff (Fig. 6, 7, 8). Other exceptional net fluxes included the large net influxes in July 1996 and large net releases in February 1997 (Fig. 10).

Because large net fluxes occur sporadically in different weeks, it is difficult to judge whether the wetland is a long-term source or sink of nutrients or TSS. The chance occurrence of one week with high flux can have a strong influence on the annual net flux. This underscores the importance of using continuous automated sampling to observe the effects of rare but critically important events.

We assessed the statistical significance of annual net fluxes by using the bootstrap technique (see Materials and Methods section) to measure the consequences of randomly including or excluding different weekly net fluxes from the calculation of annual net flux. The bootstrap analysis highlights the differences between the two years. In the first year, the net influxes of all forms of P, N, and organic C were statistically different from zero

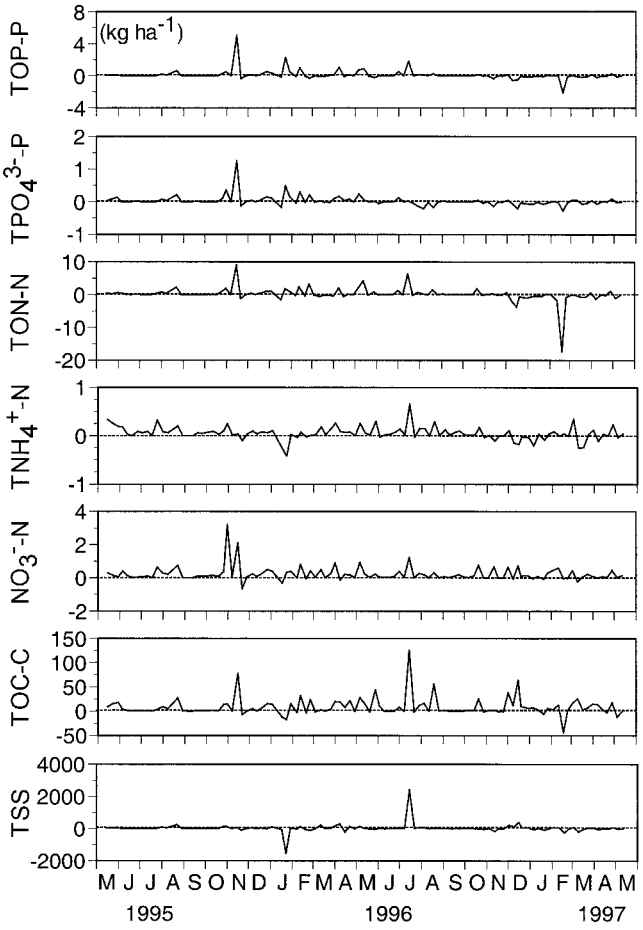

Fig. 10. Net removal of materials by the wetland, calculated by subtracting the outflow from the inflow for each sampling interval. Intervals were usually about one week long. Negative removal represents a net export of material from the wetland.

(Table 2). In the second year, only the net influxes of NO_3, TNH_4^+, and TOC, and the net release of TPO_4^{3-}, were statistically different from zero (Table 2). For the combined two-year period, only the net influxes of NO_3, TNH_4^+, and TOC were statistically different from zero (Table 2).

The two years of the study also differed in the percentage of inflowing material that was removed in the wetland (Table 2). In the first year, 59% of the inflowing TP, 38% of the inflowing TN, and 41% of the inflowing TOC were removed in the wetland. In contrast, in the second year, only 30% of the inflowing TOC was removed and the net fluxes of TN or TP were not statistically different from zero.

When calculating net flux for weeks when concentration was measured in outflow but not in inflowing runoff, we substituted annual flow-weighted mean concentrations for both outflow and inflow. As mentioned, we think this produces the best estimate of net flux because it avoids possible biases that may arise due to correlations between concentrations in outflow and inflowing runoff. Another way to deal with missing concentration would be to only substitute mean concentrations when no concentration measurement was available. Annual net fluxes calculated that way (with minimal substitutions) generally agreed well with net fluxes calculated by our preferred method. For annual mass per area net fluxes that were statistically different from zero ($p < 0.05$, bootstrap) the differences between the alternate estimates were <5% except for three materials. Differences greater than 5% were as follows: compared with using minimal substitutions, our preferred method predicted 16% less net influx of TOC in the first year, and 13% less net influx of TPO_4^{-3} and 74% more net influx of TNH_4^+ in the second year. Calculated with minimal substitutions, the net influx of TNH_4^+ in the second year is 1.1 kg N ha^{-1} yr^{-1}. By comparison, our preferred estimate is 2.0 kg N ha^{-1} yr^{-1} (Table 2). This is a small absolute difference compared with the net influxes of other forms of N (Table 2). Thus, the major conclusions of our study would not be changed by estimating net influxes with minimal substitutions.

DISCUSSION

Water Flow

Accurate measurements of water flows are essential for calculating material fluxes through wetlands. The

Table 2. Net fluxes of nutrients and total suspended solids in the wetland for Year 1, Year 2, and both years combined. Net flux is calculated by subtracting outflux from influx. Therefore, positive net flux indicates net removal and negative net flux indicates net export. Also shown are average annual net fluxes in other surface-flow wetlands reported in literature summarized by Kadlec and Knight (1996, p. 731).

	Net influx				Percentage of influx removed			
Material†	Year 1	Year 2	Both years	Literature	Year 1	Year 2	Both years	Literature
	kg ha^{-1} yr^{-1}				%			
TOP-P	14*	−1.4	6.2	–	61*	−8.3	31	–
TPO_4^{3-}–P	4.1*	−1.4*	1.4	44	53*	−18*	18	41
TP	18*	−2.8	7.6	62	59*	−11	27	34
TON-N	32*	−16	7.9	186	39*	−15	8.2	56
TNH_4^+–N	3.3*	2.0*	2.7*	128	34*	18*	25*	38
NO_3^-–N	15*	9.0*	12*	146	48*	62*	52*	51
TN	45*	−11	17	387	38*	−8.4	14	55
TOC-C	390*	510*	450*	–	41*	30*	34*	71‡
TSS	−330	2600	1100	2560	−4.1	27	13	68

* Significant at the 0.05 probability level.
† TN, total nitrogen; TNH_4^+, total ammonium; TOC, total organic carbon; TON, total organic nitrogen; TOP, total organic phosphorus; TPO_4^{3-}, total phosphate; TSS, total suspended solids.
‡ Measured as biological oxygen demand (BOD_5)

close balance of our water budget (Table 1) suggests that our flow measurements were relatively accurate and that the underlying clay layer blocked ground water flow, as we had assumed. However, the excess water gain in our budget (<5% of the inflow, Table 1) implies that there is a small, unaccounted loss of water. The imbalance may reflect imprecision in our measurements or may be due to seepage into the ground water or through the dike, which we had ignored. Another study of similar wetlands found that seepage through the dike could be substantial, amounting to 27 to 47% of the inflow volume (Larson et al., 2000). The proportions of water budget components can vary greatly among constructed and restored wetlands. As the ratio of watershed area to wetland area increases, the importance of the inflow from the watershed increases. In wetlands with relatively large watersheds, direct precipitation inputs and evapotranspiration may be negligible and outlet flows may approximately equal inlet flows (e.g., Braskerud, 2002).

Comparing Removal Rates

Different wetlands remove materials at widely differing rates (e.g., see reviews by Verhoeven and van der Toorn, 1990; Mitsch and Gosselink, 1993; Whigham, 1995; Mitsch et al., 2000). To put our results in the broadest possible context, we compared removal rates for our wetland with average rates for a wide variety of wetlands reviewed by Kadlec and Knight (1996). By this comparison, our wetland seems to remove nutrients and suspended sediments at below average rates (in mass per area), even in the year when nutrient removal was highest (Table 2). For example, in that year our wetland removed total P at about one-third the average rate and total N at about one-ninth the average rate (Table 2). For some forms of nutrients the differences were even greater (Table 2). However, if we compare the percentages of inflowing nutrients removed in the same year, our wetland does not seem very different from average (Table 2). This may reflect the fact that influxes of materials to our wetland are lower than average due to the large area of our wetland (1.3 ha) relative to the area of its watershed (14 ha).

The concentrations of materials in inflowing water may influence their rates of removal. Often, removal rates are modeled according to first-order kinetics, with removal rates proportional to concentration (Kadlec and Knight, 1996). This could explain why our wetland, which usually had <1 mg NO_3^-–N L^{-1} in inflowing water (Fig. 8), removed NO_3^- at a lower rate than did the wetlands studied by Kovacic et al. (2000) and Hunt et al. (1999), with 9 to 13 mg NO_3^-–N L^{-1} and 3 to 9 mg NO_3^-–N L^{-1} in inflowing water, respectively. The highest NO_3^- concentrations entering our wetland followed the extended dry period from June–October 1995 (Fig. 8). The highest concentration observed (5.3 mg NO_3^-–N L^{-1}) was in the first runoff event after the dry period. After that, inflowing NO_3^- concentrations remained elevated for about 5 mo (Fig. 8). The antecedent dry conditions may have promoted NO_3^- accumulation in the watershed soil by enhancing nitrification while preventing NO_3^- removal via runoff or denitrification. The differences between concentrations of NO_3^- in inflowing runoff and outflowing water tended to be greatest during weeks with higher concentrations in runoff (Fig. 8). This suggests that our wetland may have the capacity to remove NO_3^- at higher rates if runoff entering the wetland had higher NO_3^- concentrations. Jordan et al. (1997a) found that Delmarva watersheds with 80% cropland (the proportion in our wetland's watershed) typically discharge water with about 3 mg NO_3^-–N L^{-1}, 1.2 mg TON L^{-1}, and 4 to 13 mg TOC L^{-1}. By comparison, discharges from our wetland's watershed generally had <1 mg NO_3^-–N L^{-1}, 1 to 5 mg TON L^{-1}, and 15 to 80 mg TOC L^{-1}. These concentrations probably reflect the lack of ground water flow from the watershed draining into the wetland because NO_3^- concentrations decrease and total organic N and C concentrations increase as the proportion of ground water in watershed discharge decreases (Jordan et al., 1997b).

Fluxes of particulate and dissolved materials are likely to differ, but, because of our acid preservative, we could only measure the combined total fluxes of particulate and dissolved materials except for NO_3^-, which is essentially all dissolved. A previous study analyzed dissolved and particulate materials separately in grab-sampled water flowing in and out of our study wetland (Jordan et al., 1999). That study found that inflowing TOC, TON, and TOP were 85, 35, and 15% dissolved matter, respectively. The differences in the proportions of dissolved matter may account for the differences among the temporal variations of TOC, TON, and TOP concentrations (Fig. 6).

Effects of Flow Variability

The low absolute rates of nutrient removal by our wetland (Table 2) may reflect the unregulated inflow. The removal rates reviewed by Kadlec and Knight (1996) are based on a diversity of surface flow wetlands, including constructed wetlands and natural wetlands, receiving water from a variety of municipal and agricultural sources. However, most of the wetlands reviewed had regulated inflows. The few published studies that give absolute rates of TN or TP removal from unregulated inflows (e.g., Table 3) represent a wide variety of wetland types with a wide range of hydraulic loading rates. For example, the wetlands studied by Kovacic et al. (2000) received tile drain effluent leached from cropland soils. Thus, nitrate was the main form of N input and there was little input of particulate matter or P. These wetlands were effective at removing nitrate, probably via denitrification, but were less effective at removing organic N and ineffective at removing P (Kovacic et al., 2000). In contrast, the wetlands studied by Braskerud (2000, 2002) received stream water carrying agricultural runoff with high particulate loads. These wetlands were inefficient at removing nitrate but effective at removing organic N, particulate matter, and P. Similarly, a restored prairie pothole wetland in an agricultural watershed was effective at removing particu-

Table 3. Annual N and P removal by wetlands receiving unregulated inflows. Also shown are wetland areas expressed as a percentage of watershed area, annual mean hydraulic loading rates (inflow/area), and annual mean detention times (volume/inflow rate). Studies are listed in order of hydraulic loading rate. Ranges are shown when more than one wetland or year was studied.

Reference	Area as % of watershed	Hydraulic load	Detention time	Total N removed		Total P removed	
	%	mm d^{-1}	d	kg ha^{-1} yr^{-1}	% of influx	kg ha^{-1} yr^{-1}	% of influx
This study	9	12–20	12–19	−11 to 45	−8.4 to 38	−2.8 to 18	−11 to 59
Kovacic et al., 2000	3–6	17–30	22–38	127–678	27–52	−76 to 8.5	−54 to 80
Magner et al., 1995	2	ND†	ND	ND	ND	1–3	27
Hunt et al., 1999	0.8	97	9.1	1100	37	ND	ND
Raisin et al., 1997	0.05	250	2.0	230	11	28	17
Reinelt and Horner, 1995	1–2	620–720	3.3–20	ND	ND	4.4–30	7.5–82
Braskerud 2000, 2002	0.03–0.4	670–1800	0.39–1.0	500–2850	3–15	170–710	20–44
Fleisher et al., 1994	0.02–0.3	360–4800	0.32–4.2	730–6800	2.6–9.5	ND	ND

† No data.

late matter and P (Magner et al., 1995). Usually P is associated with particulate matter but one of the wetlands studied by Reinelt and Horner (1995) received and removed relatively high amounts of dissolved phosphate from ground water flowing through P-rich deposits.

The efficiency of nutrient removal by our wetland may be reduced by the temporal variability of the water inflow rates. Most of the inflowing water enters during high flow events that last less than one day and often deliver volumes of water similar to the holding capacity of the wetland (Fig. 4). If inflowing and standing water mix completely, then a flow event with inflow equal to standing volume would cause a discharge of half the standing water and half the inflowing water. Water exchange is even more rapid if mixing is incomplete. Thus, there is a potential for much of the inflowing water to exit the wetland in a matter of hours. The wetland may not have enough time to remove nutrients and suspended sediments from water passing through in brief pulses. For example, Kovacic et al. (2000) noted that the capacity to remove NO_3^- was exceeded during high-flow events in constructed wetlands with unregulated inflow. Increasing the temporal variability of flow may have a similar effect on nutrient removal as increasing the hydraulic loading rate or increasing the ratio of watershed to wetland. For our wetland, the average hydraulic loading rate was about 200 m^3 d^{-1} (15.5 mm d^{-1}) over the two years of the study but hydraulic loading rate was often more than 10 times above average on days with high runoff (Fig. 4). This suggests that the nutrient removal efficiency of our wetland would have been higher if the inflow rate were constant rather than variable. Some of the most detailed studies of wetland use for nutrient removal are of wetlands that receive relatively steady pumped inflows of river water (e.g., Hey et al., 1994; Mitsch et al., 1995, 1998; Moustafa, 1997). The performance of these wetlands may be much better than that of wetlands receiving variable unregulated inflow.

The effects of inflow variability have not been addressed by many studies. However, Raisin and Mitchell (1995) used automated samplers to measure mass balances of N and P during high flow events in three wetlands that receive agricultural runoff. They found that mass balances differed greatly among different high flow events with events in winter causing net releases due to flushing (Raisin and Mitchell, 1995). We could not resolve the effects of individual flow events because our samples were weekly composites. We found large differences in net flux among weeks, with net removal in some weeks and net export in others (Fig. 10). However, we could find no correlations between concentrations or net fluxes of materials and either the total weekly water inflow or the weekly maximum water inflow rate. Event-based automated sampling would probably be better than weekly automated sampling for revealing correlations between water flow and net fluxes of materials. Due to the importance of unpredictable high-flow events, automated sampling is essential for quantifying mass balances for wetlands with variable unregulated inflow. A few studies have used flow-proportional automated sampling (e.g., Reinelt and Horner, 1995; Kovacic et al., 2000; Braskerud, 2002), as we did, while others have used automated sampling at fixed time intervals (e.g., Magner et al., 1995; Hunt et al., 1999), or at selected inflow rates (Raisin and Mitchell, 1995; Raisin et al., 1997).

Carleton et al. (2001) reviewed studies of 49 wetlands receiving unregulated inputs of urban or agricultural runoff and concluded that the wetlands performed similarly to wetlands with regulated flow in removing pollutants. However, they noted that there was high variability of performance that could be related to the temporal variability of inflows. They summarized wetland performance by regressing the percentage of material influx removed versus the ratio of wetland area to watershed area. Such regressions should be interpreted cautiously because the regressed variables are both correlated with the amount of water inflow, which may cause spurious correlations as defined by Kenney (1982) and Garsd (1984). Comparing a few studies that report absolute as well as relative removal rates, we found that the percentage of N and P influx that is removed tends to increase as the hydraulic loading rate decreases and the detention time increases (Table 3). However, net exports of N or P sometimes occurred from wetlands with the lowest hydraulic loading rates (e.g., Table 3: our study and Kovacic et al., 2000). Moreover, the highest absolute rates of nutrient removal were reported for wetlands receiving the highest hydraulic loading rates (e.g., Table 3: Braskerud 2000, 2002; Fleischer et al., 1994).

One factor that can affect nutrient removal is hydraulic efficiency, the degree to which inflowing water is

dispersed over the wetland area (Persson et al., 1999). The even dispersion of inflowing water over the wetland surface maximizes hydraulic efficiency and nutrient removal. In contrast, channeled flow may limit the exposure of inflowing water to the wetland surface and thereby limit nutrient removal. For example, the wetlands studied by Reinelt and Horner (1995, Table 3) carry water in channels that are only 11 to 25% of the total wetland areas. In contrast, our wetland is completely submerged during periods when the water was deep enough to flow over the weir. Thus, there is the potential for the inflowing water to interact with the entire wetland surface before flowing out. However, it is likely that flow was not evenly dispersed over our wetland and, lacking measurements of dispersion, we do not know whether uneven flow may have limited nutrient removal. Persson et al. (1999) discuss design features that can maximize hydraulic efficiency of constructed wetlands.

For wetlands with unregulated variable inflow, nutrient removal may be improved by reducing the variability of the outflow. This could be achieved by designing outflow control structures, such as dikes and drains, to maximize the residence time of water within the wetland. For example, a drain that allows the wetland to slowly empty after a storm inflow prolongs water residence in the wetland by providing holding capacity for later storm inflows. In contrast, a flat-topped standpipe drain, which maintains nearly constant water volume in the wetland, makes the wetland unable to hold additional water from storm inflows.

Seasonal and Interannual Changes

Besides differing among high flow events, nutrient removal may vary at seasonal and interannual time scales. Several studies have observed seasonal changes in NO_3^- removal presumably linked to the temperature dependence of denitrification (e.g., Hunt et al., 1999; Spieles and Mitsch, 2000). For our wetland, variability among high flow events may have obscured seasonal patterns. However, we did observe striking differences between the two years of the study, with the greatest N and P removal in the first year (Table 2). The two years also differed hydrologically, with drying period from June–October in the first year only (Fig. 2). We cannot be certain that differences in nutrient removal are linked to the drying period, but the elevated concentrations of NO_3^- in inflow after the dry period (Fig. 8) may have enhanced NO_3^- removal, as we have discussed. Interannual differences in hydrology and nutrient removal were also noted for a restored prairie pothole wetland (Magner et al., 1995).

Besides interannual differences linked to rainfall, there may be long-term trends in nutrient trapping as the wetland ages. Our study was conducted about a decade after restoration of the wetland when emergent vegetation was well established. After wetland restoration, nutrient removal efficiency could improve as growing vegetation helps trap and hold sediment and produces organic matter to support denitrification (e.g., Mitsch and Carmichael, 1996; Mitsch et al., 2000; Spieles and Mitsch, 2000). During our study, plant biomass in our wetland was not increasing monotonically but varied from year to year in response to variations in precipitation (Whigham et al., 2002). Aboveground biomass and plant N and P were lower in the first year of our study (Whigham et al., 2002) when nutrient retention was highest. This suggests that plant biomass did not limit nutrient removal in our wetland.

Removal of water-borne N and organic C may continue indefinitely if these elements are converted to gaseous forms in the wetland and released to the atmosphere. However, removal of suspended sediment and P may cease sometime after wetland restoration as the wetland fills with sediment and the sediment becomes saturated with P (Richardson, 1989). One multiyear study of constructed wetlands documented an increase in sediment retention in the first four years as vegetation coverage increased (Braskerud, 2001), followed by a gradual decline in organic N removal from 3 to 10 yr after construction (Braskerud, 2002). In a recent review, Mitsch et al. (2000) suggest that sustainable removal rates range from about 5 to 50 kg ha^{-1} yr^{-1} for P and 100 to 400 kg ha^{-1} yr^{-1} for N (with 1000 to 2000 kg ha^{-1} yr^{-1} N removal possible in warm climates). By comparison, the two-year average removal rates for our wetland are near the low end of the sustainable range for P (7.6 kg P ha^{-1} yr^{-1}, Table 2) and below the sustainable range for N (17 kg N ha^{-1} yr^{-1}, Table 2). In contrast, some other constructed wetlands with unregulated inflow (Table 3) removed P (Braskerud, 2000) or N at rates above the sustainable range (Fleischer et al., 1994; Hunt et al., 1999; Braskerud, 2002).

CONCLUSIONS

Our wetland was less effective than many wetlands in removing nutrients but it still made a substantial difference in nutrient delivery to the adjacent waters of Chesapeake Bay. In the first year of our study, our wetland roughly matched the Chesapeake Bay Program (1997) goal of 40% reduction of N input to the Bay and exceeded that goal for P (Table 2). The wetland did not remove N and P in the second year, but over the two years of the study it removed 25% of the TNH_4^+, 52% of the NO_3^-, and 34% of the TOC it received from the watershed (Table 2). Although wetland restoration clearly has potential for reducing nutrient discharges, it is difficult to predict the quantitative effect of wetland restoration because nutrient removal differs greatly among wetlands and varies greatly with brief episodes of high flow, interannual differences in rainfall, and long-term changes in wetland development. Moreover, there is relatively little information for constructed or restored wetlands with unregulated event-driven inflows, which must be monitored with automated sampling. Our study adds to the growing database for such wetlands and suggests that restoring wetlands in agricultural fields will result in significant improvement of runoff.

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