Nutrient Cycling and Retention in Natural and Constructed Wetlands

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RESTORED WETLANDS IN CROP FIELDS CONTROL NUTRIENT RUNOFF

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1. Introduction

Anthropogenic sources of nitrogen (N) and phosphorus (P) to Chesapeake Bay have been purported to be responsible for major changes in ecosystem structure and function. Increased N and P loading (Taft *et al.* 1980, Officer *et al.* 1984, Conley & Malone 1992) have led to excessive phytoplankton production causing depletion of dissolved oxygen and declines in submersed aquatic vegetation (Malone *et al.* 1986, Boynton *et al.* 1982, Gallegos *et al.* 1992, Kemp *et al.* 1983). The sources of N and P inputs to Chesapeake Bay (hereafter referred to as the Bay) are numerous as it receives runoff from approximately 50 rivers which drain its 178,000 km2 watershed. Many point sources of pollution (*e.g.*, sewage plant effluents) have been identified and actions taken to reduce nutrients and toxins in discharges. Atmospheric inputs of N, from within and outside of the watershed, are also significant (Jordan & Weller 1996). One of the main sources of N inputs to the Bay is non-point source runoff, primarily from croplands (Jordan *et al.* 1997a, b). Large amounts of P also enter the Bay from croplands but most of the P loading to the Bay is associated with suspended sediments (Jordan *et al.* 1997a, b) which can have numerous sources.

A wide range of projects are currently underway to restore the ecological health of the Bay. A goal of the Chesapeake Bay Program, the largest coordinated restoration effort in the United States, is to reduee nutrient inputs to the Bay by 40% by the year 2000. Toward this goal a variety of initiatives and approaches are being used to reduce non-point sources of N and P. For example, farmers are encouraged to use less fertilizer and pesticides and to adopt praetices that reduce soil erosion. Also, incentives for restoring riparian forests have been implemented because both wetland and non-wetland riparian forests are known to improve water quality by removing N, P, and sediments (Pinay & Decamps 1988, Lowrance *et al.* 1995). However, restoration of riparian forests is often less attractive to farmers than restoration of emergent wetlands, in part because emergent wetlands provide the added benefit of attracting wild waterfowl. Recent research has shown that constructed or restored herbaceous wetlands can also be used to remove sediments and nutrients from non-point sources including agricultural discharges (*e.g.*, Fleischer *et al.* 1994, Mitsch 1994, Raisin & Mitchell 1995, Whigham 1995). Wetlands may act as filters, removing particulate material, or as sinks, accumulating nutrients, or as transformers, converting nutrients to different forms, ineluding gaseous forms of N and C (Richardson 1989).

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In this paper we report results of an ongoing study to examine the potential to restore emergent wetlands in agricultural fields as an effective means for reducing N and P runoff to the Bay. The project takes place on the Delmarva peninsula on the Eastern Shore of the Bay (Fig. 1), a very flat landscape that has a high percentage of the land in agriculture, primarily corn (maize) and soybeans (soya). Soils throughout the study area typically have a very high clay content and an impermeable clay layer near the soil surface. For crops to be sucessfully grown in these soils, excess water is removed from fields to avoid waterlogging. Most fields in the study area contain a network of connected drainage channels, created by plowing and ditching, that discharge water into wetlands, streams, riparian forests, or directly into the Bay. The drainage networks effectively convert non-point runoff into point source pollution. Ditches that are at the lowest topographic position within an agricultural field are typically in areas that were onee wetland habitat. Seven wetlands restored in such areas are the subject of our study. By comparing the concentrations of N, P, organic C, and suspended solids in water entering and leaving the wetlands, our study assessed removal of these materials from agricultural runoff.

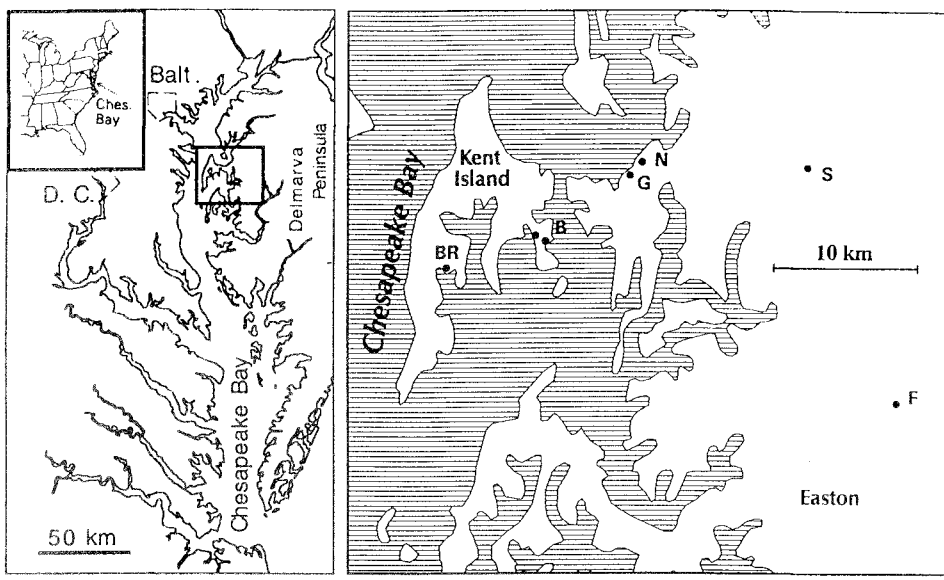

Fig. 1. Inset: Location of Chesapeake Bay on the east coast of the United States. Left: Chesapeake Bay and the cities of Baltimore and Washington, D. C. The box outlines the area of the study wetlands on Kent Island and the Delmarva Peninsula. Right: Locations of the study wetlands on Kent Island and the Delmarva Peninsula near the town of Easton. The different wetlands are Braun (BR), Barnstable (B, two wetlands), Gerber (G), Nesbit (N), Sultenfuss (S), and Foster (F).

2. Methods

2.1. Study sites

The study wetlands are in watersheds dominated by croplands (Table 1). The wetlands were restored by the Chesapeake Wildlife Heritage as part of their program to provide wildlife habitat and improve the quality of runoff water from agricultural fields. The restored wetlands were created 1-8 years before this study began. During restoration, a thin layer of soil was removed to create a shallow depression. Some of the excavated soil was used to create a dam to retain water. After excavation, top soil was returned to the surface and wetland vegetation was established by natural succession.

Water enters the wetlands mostly through drainage ditches and leaves through standpipe drains installed in the dams. Most of the water flow through the wetlands is surface runoff associated with heavy rain. There is little or no groundwater flow because all of the wetlands, except the Nesbit site, are underlain by a layer of clay within 0.5 m of the soil surface. The clay layer seems to block water infiltration, because clay sampled from beneath inundated areas appears dry. Water levels in the wetlands typically remain close to tops of the standpipes during late fall, winter, and spring. When the wetlands were full, water depths were generally less than 0.5 m and nowhere more than 1 m. In summer, evapotranspiration lowers the water levels, thus exposing most of the sediment that is submerged the rest of the year. Occasionally, rain storms partly refill the wetland during the summer.

A variety of emergent and submerged macrophytes colonized the wetlands soon after restoration. At the time of our study, 40 macrophyte species were found in the wetlands. Each of the 7 wetlands had a different combination of macrophytes, but *Eleocharis obtusa* was the most dominant species in 3 wetlands and the second most dominant in one of the wetlands (Table 2). *Echinochloa crusgalli* was among the top 3 dominant in 3 of the wetlands; and *Ludwigia palustris*, *Potamogeton diversifolius*, and *Panicum virgatum* were among the top 3 dominants in 2 of the wetlands. None of the 7 wetlands had a completely unique assortment of species comprising the three most dominant (Table 2).

Table 1. Date of restoration of the study wetlands, areas (ha) of the wetlands and their watersheds, and percentages of cropland in each watersheds.

	Wetland		Watershed	
Name	Date Restored	Area (ha)	Area (ha)	% Cropland
Barnstable 1	1986	1.3	14	85
Barnstable 10	July, 1992	2.6	20	70
Braun	Sept., 1992	2.3	12	80
Foster	Oct., 1993	0.49	4.3	60
Gerber	July, 1990	0.40	3.6	30
Nesbit	July, 1989	0.40	10	95
Sultenfuss	Nov., 1992	1.2	19	95

Table 2. The three most dominant macrophyte species in the wetlands in 1994 based on importance values ([relative frequency+relative cover]/2) in permanent plots (personal communication, A. Peppin).

Wetland	Most Dominant Macrophyte Speeies		
	First	Second	Third
Barnstable 1	*Eleocharis obtusa*	*Ludwigia palustris*	*Scirpus americanus*
Barnstable 10	*Potamogeton diversifolius*	*Panicum virgatum*	*Echinochloa crusgalli*
Braun	*Panicum virgatum*	*Xanthium strumarium*	*Setaria glauca*
Foster	*Eleocharis obtusa*	*Alisma plantago-aquatica*	*Rotala ramosior*
Gerber	*Eleocharis obtusa*	*Scirpus mucronatus*	*Digitaria ischaemum*
Nesbit	*Ludwigia palustris*	*Eleocharis obtusa*	*Echinochloa crusgalli*
Sultenfuss	*Leersia oryzoides*	*Potamogeton diversifolius*	*Echinochloa crusgalli*

2.2. *Sampling and analysis*

We sampled water draining from the 7 wetlands and water entering through up to 3 drainage leads per wetland for up to 10 dates, depending on the wetland, from February, 1994 through May, 1995. We designed our sampling to cope with the episodic and unpredicable nature of water flow through the wetlands. We installed polyethylene sampling bottles in drainage leads and near the outlets of the wetlands that would trap storm runoff. When possible, we also visited the wetlands during periods of runoff to collect samples by hand. However, because runoff was so intermittent, we could not collect complete sets of inflow and outflow samples from all the wetlands on each sampling date. Initially, we measured concentrations of dissolved inorganic nutrients, including phosphate, ammonium, and nitrate (plus nitrite). In the fall of 1994, we began measuring concentrations of suspended sediments, particulate nutrients and organic C, N and P.

Standard techniques were used for analysis of N and P species. Samples to be analyzed for dissolved substances were filtered with prewashed 0.45 μm Millipore filters. Total P in filtered and unfiltered samples was digested to phosphate with perchlorie acid (King 1932). Phosphate in the digestate and dissolved phosphate (DPO_4) in filtered samples were analyzed by reaction with stannous chloride and ammonium molybdate (APHA 1989). Total Kjeldhal N was digested with sulfuric acid, Hengar granules, and hydrogen peroxide (Martin 1972). The resultant ammonia was distilled and analyzed by Nesslerization (APHA 1989). In undigested aliquots, dissolved ammonium (DNH_4) was oxidized to nitrite by alkaline hypochlorite (Strickland & Parsons 1972), dissolved nitrate was reduced to nitrite by cadmium amalgam, and nitrite was analyzed by reaction with sulfanilamide (APHA 1989). We present data on the sum of nitrite and nitrate concentrations, which we refer to as NO_3. Phosphate and ammonium bound to particles were extracted by collecting particles on 0.4 mm Nuclepore filters, and then rinsing with 1 M KCl (Keeney & Nelson 1982) to extract particulate ammonium (PNH_4), or with 0.5 N H_2SO_4 (Correll & Miklas 1975) to extract particulate phosphate (PPO_4). The extracts were analyzed with the same methods used for DNH_4 and DPO_4.

From results of the above analyses we calculated particulate organic N (PON) and P (POP), and dissolved organic N (DON) and P (DOP). PON was calculated by subtracting Kjeldhal N in filtered samples and PNH_4 from Kjeldhal N in unfiltered samples. Similarly, POP was calculated by subtracting the total P in filtered samples and PPO_4 from total P in unfiltered samples. DON was calculated by subtracting DNH_4 from Kjeldahl N in filtered samples. Likewise, DOP was calculated by subtracting DPO_4 from total P in filtered samples.

Dissolved and particulate organic carbon (DOC and POC) were analyzed by drying samples at 60°C, followed by reaction with potassium dichromate in 67% sulfuric acid at 100°C for 3 h (Maciolek 1962). Organic carbon was calculated from the amount of unreacted dichromate measured colorimetrically (Maciolek 1962, Gaudy & Ramanathan 1964).

Total suspended solids (TSS) were measured by filtering through prewashed, preweighed 0.4 μm filters, rinsing with distilled water to remove salts, drying, and reweighing.

3. Results and Discussion

Concentrations of total suspended solids and all forms of P, N, and organic C differed greatly among sampling dates (*e.g.*, Fig. 2). In many cases, there were too few samples to reveal seasonal patterns. However, at 5 of the wetlands, dissolved inorganic nutrients were sampled on 7-10 different dates throughout the year and these nutrients showed no clear seasonal pattern (*e.g.*, Fig. 2). The extreme variability in concentrations among runoff events may be caused by variable dilution from different runoff volumes, farming activities, or differences in soil conditions prior to the runoff event.

Because there were no clear temporal patterns, we averaged data from different dates to compare concentrations among different nutrient forms, among different wetlands, and between inflowing and outflowing water. We tested the statistical significanee of differences between inflowing and outflowing water using Wilcoxon's signed-ranks test (Sokal & Rohlf 1981). We selected a non-parametric test because sporadic high concentrations resulted in a non-normal distribution of differences in concentration. For the test, we first calculated the difference between inflowing and outflowing concentration for each sampling date and then determined the probability that the differences for all the dates for each wetland are statistcally different from zero. Pairing comparisons by date helps resolve differences when variance among dates is high.

Usually inflowing water had higher nutrient concentrations than outflowing water, suggesting that the wetlands remove nutrients from the water (Figs. 3 and 4). For most of the wetlands, mean DPO_4 concentrations were higher in inflowing than outflowing water. This difference was significant ($p<0.05$) for 4 out of 7 wetlands (Fig. 3). Similarly, the concentration of DNH_4 was significantly higher in inflowing than outflowing water at two wetlands (Fig. 3). NO_3 concentrations were signifieantly higher in inflowing water at 3 of the wetlands (Fig. 3).

We had at most 5 pairs of measurements of inflowing and outflowing concentrations for total suspended solids and for particulate and organic forms of P, N, and C. With so few pairs, it is not possible to achieve a significance level of $p<0.05$ with

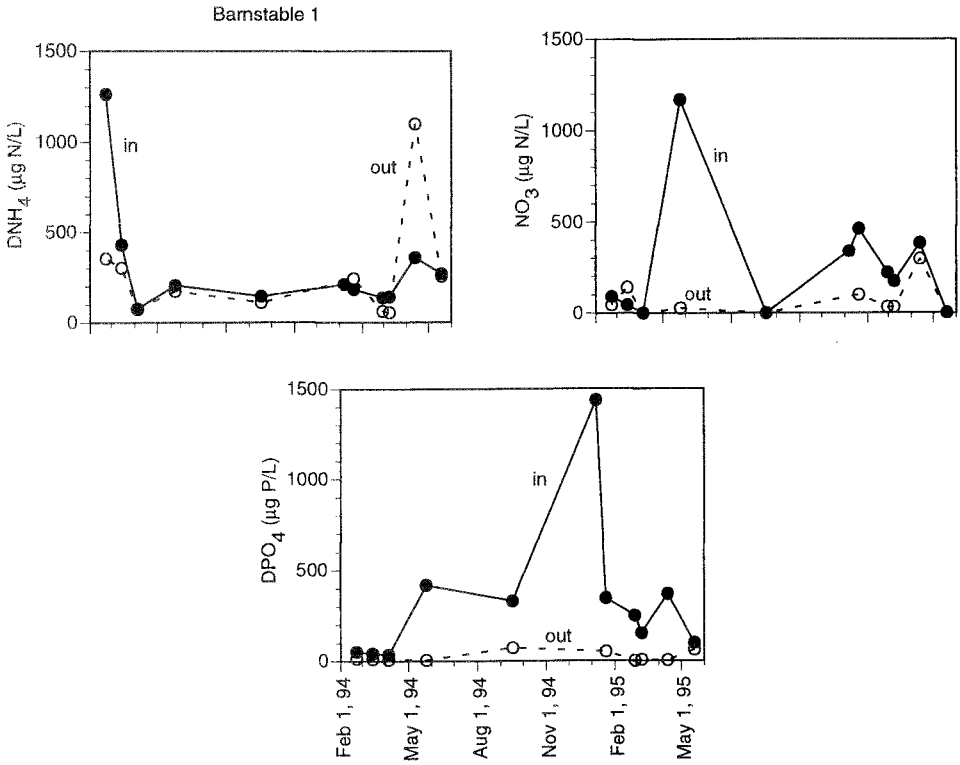

Fig. 2. Concentrations of DNH_4, NO_3, and DPO_4 in water entering (filled circles, solid lines) and leaving (open circles, dashed lines) the Barnstable 1 wetland versus time. Concentrations entering are averages of samples taken from 1-3 points of entry.

Wilcoxon's signed-ranks test. However, with 5 pairs, if the differences between inflowing and outflowing concentrations are all of like sign, then p=0.0625, which is close to the traditional significanee cutoff. In all cases where this level of significance was achieved, inflowing concentrations were higher than outflowing concentrations. This occurred for POP at one wetland, for DON at three wetlands, and for DOC at 2 wetlands, out of a total of 4 wetlands whcre these nutrients were measured (Fig. 4). In general, inflowing concentrations cxceeded outflowing concentrations more for dissolved nutrients and TSS than for particulate forms of N, P, and organic C (Figs. 3 and 4).

For four wetlands, we could compare concentrations of different forms of P, N, and C. In general, the most abundant form of P was POP while the least abundant was DOP in both inflowing and outflowing water (Figs. 3 and 4). In some eases, POP concentrations were more than 10 times DOP concentrations (Fig. 4). N was usually most abundant in organic forms, except at one wetland whieh had unusually high NO_3 concentrations (Figs. 3 and 4). PNH_4 was usually the least abundant form of N (Figs. 3 and 4). Organic C was usually more abundant in dissolved than in particulate form (Fig. 4).

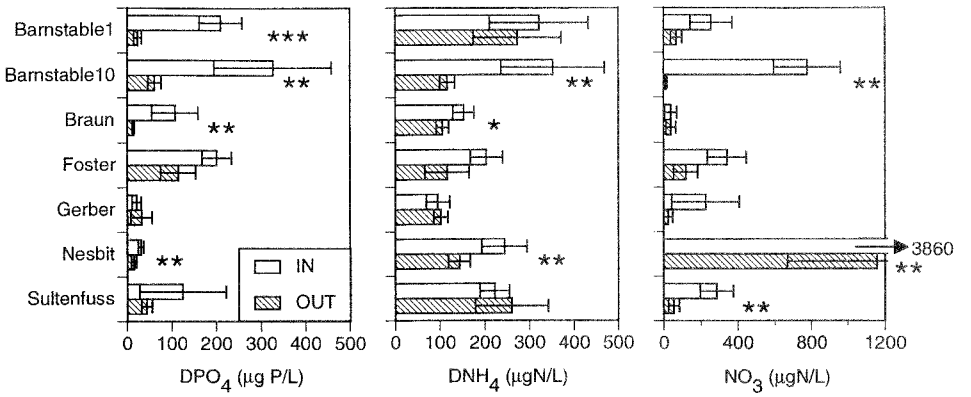

Fig. 3. Mean coneentrations of DPO_4, DNH_4, and NO_3 in water entering (open bars), and leaving (shaded bars) seven different wetlands. Brackets are ± standard errors. Asterisks indicate levels of significance of differences between inflowing and outflowing concentrations (*** $p<0.01$, ** $0.01<p<0.05$, * $0.05<p<0.07$). Significance levels were calculated by Wilcoxon's signed ranks test.

For some forms of nutrients, the concentrations in water entering our wetlands were markedly different from those in water discharged from watersheds elsewhere on the Delmarva peninsula. Concentrations of N in watershed discharges increase as the proportion of cropland in the watershed increases (Jordan *et al.* 1997a). Watersheds with >60% cropland typically discharge water with NO_3 concentrations of 2000-3000 µg N/L and total organic N (DON+PON) concentrations of 1000-1300 µg N/L (Jordan *et al.* 1997a). By comparison, watersheds with similar percentages of cropland that drain into our wetlands discharged water with much lower concentrations of NO_3 (<1000 µg N/L, Fig. 3) and much higher concentrations of total organic N (>2000 µg N/L, Fig. 4). Concentrations of total organic C (DOC+POC) were also higher in water entering the wetlands (>40 mg C/L) than in discharges from other Delmarva watersheds (<14 mg C/L, Jordan *et al.* 1997a). These differences are probably related to the lack of groundwater flow from the watersheds that drain into the wetlands. Jordan *et al.* (1997c) found that NO_3 concentrations decrease and total organic N and C concentrations increase as the proportion of groundwater in watershed discharge decreases. It is consistent that the one wetland with relatively high concentrations of NO_3 and low concentrations of total organic N in inflowing water (Braun, Figs. 3 and 4) is also the only wetland that seems to receive some groundwater inflow, judging from the lack of impermeable clay near the soil surface and from the persistence of inflow during periods of low rainfall.

For some forms of nutrients, the coneentrations in water entering our wetlands were similar to those found in watershed discharges elsewhere on the Delmarva peninsula. For example, total NH_4 (DNH_4 + PNH_4) coneentrations in inflowing water were similar to those in watershed discharges measured by Jordan *et al.* (1997a). Also, concentrations of P forms and TSS in inflowing water are within the high end of the range reported by Jordan *et al.* (1997a).

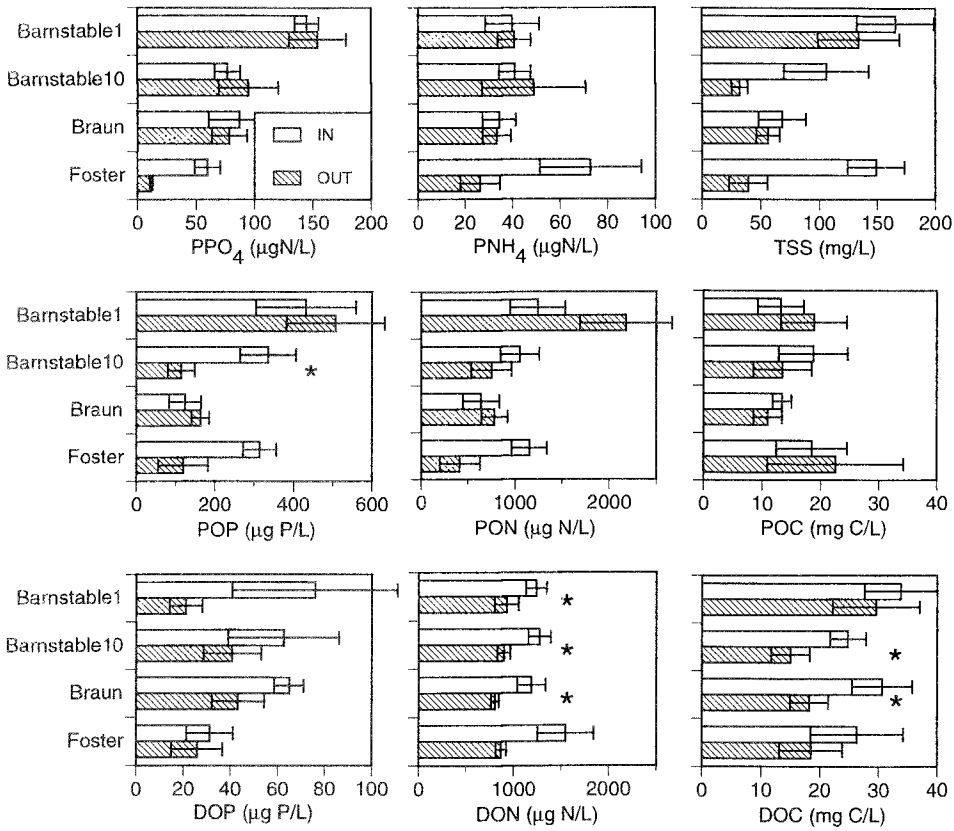

Fig. 4. Mean concentrations of PPO_4, PNH_4, TSS, POP, PON, POC, DOP, DON, and DOC in water entering (open bars), and leaving (shaded bars) four different wetlands. Brackets are ± standard errors. Asterisk indicates that the difference between inflowing and outflowing concentrations is significant at $0.05<p<0.07$ as calculated by Wilcoxon's signed ranks test.

From the differences in nutrient concentrations in inflowing and outflowing water we estimated the percentage of the inflowing material removed from the water passing through the wetland. For this estimation, we assumed that the amount of water entering each wetland is equal to the amount leaving so the net uptake (or net release) of material is proportional to the difference between concentrations in inflowing and outflowing water. Actually, this assumption should lead to underestimation of material removed because evapotranspiration will result in less water leaving the wetland than entering. For example, a later unpublished study found that annually 8-18% of the water entering the Barnstable 1 wetland did not flow out but presumably evaporated. Therefore, a finding that concentrations of materials were the same in inflowing and outflowing water would imply that 8-18% of the materials were trapped within the wetland. Although we do not know the water budgets of the wetlands during the present study, an estimate of nutrient retention based on concentration differences and neglecting evapotranspiration is useful as a conserva-

tive approximation of the relative efficiency of nutrient removal. Thus, for each wetland, we ealeulated the proportion of inflowing nutrient removed as the difference of inflowing-outflowing concentration divided by the inflowing coneentration. We then averaged the pereentages of inflow removed for all the wetlands combined.

Thc average pereentage of inflow removed ranged from 68% for NO_3 to -5% for POC (a negative percentage suggests net release, Fig. 5). Dissolved forms of nutrients were mueh more efficiently removed than particulate forms (Fig. 5). The average pereentages of inflow removed were significantly greater than zero ($p<0.05$, T test) for all the dissolved nutrients except for DNH_4, which was close to the signifcance cutoff ($p=0.070$). In contrast, the percentages removed were not significant for any partieulate nutrients, although average removal of TSS (45% of inflowing TSS) was nearly significant ($p=0.061$). The apparent trapping of TSS without removal of particulate nutrients suggests that the trapped TSS is relatively poor in asssociated PPO_4, PNH_4, and organic P, N, and C.

Our results suggest that restored wetlands could make a substantial contribution toward the Chesapeake Bay Program's goal of lowering nutrient inputs to the Bay by 40%. The average percentages of inflowing dissolved nutrients removed ranged from 25% for DNH_4 to 68% for NO_3. However, the lack of efficient removal of partieulate nutrients lowers the overall efficiency of removing total P, N, and organic C. Based on data from our four most intensively sampled wetlands, the average pereentages of inflowing total P, N, and organic C removed were 43%, 23%, and 18%, respeetively. Of these averages, only the average for total P was significantly greater than zero ($p=0.027$).

Removal effieieneies reported from other wetland studies differ widely, in part due to differenees in the rate of water flow through the wetlands, or to differences in the relative areas of the wetlands and their catchments. Studies reviewed by Verhoeven & van der Toorn (1990) found that natural and eonstructed wetlands receiving wastewater and natural wetlands with high influxes of nutrients removed 50-99% of the ineoming N and 25-98% of the incoming P. Removal of P, which depends on aeeumulation within the wetland, may reach a limit (*e.g.*, Riehardson & Marshall 1986), but removal of N, which can result from denitrification, may persist indefinitely (Verhoeven & van der Toorn 1990). Data summarized by Mitsch & Gosselink (1993) suggest a rough correlation between nutrient loading and percentage removed. Jansson *et al.* (1994), comparing small lakes, ponds, and wetlands, eoncluded that the removal of nutrients increased as the retention time of water within the system inereased. In the wetlands we studied, water retention varies with the frequeney and magnitude of flow events. In the Barnstable 1 wetland, large flow events ean introduee in one day a volume of water equal to or greater than the holding eapaeity of the wetland (unpublished data). Nine sueh large flow events oeeurred within a two year period at the Barnstable 1 wetland (unpublished data). Comparing different wetlands, retention time increases as the volume of the wetland system increases in relation to the catchment area. The effeet of catchment area may aeeount for differenees between the wetlands we studied and those studied by Fleiseher *et al.* (1994). Three of the wetlands they studied received agricultural runoff and were morphologieally similar to ours but covered only 0.3-0.02% of their catchment area and removed only 3-10% of the inflowing N. By comparison, our wetlands eovered 4-16% of their eatchment area (Table 1), and removed about 23%

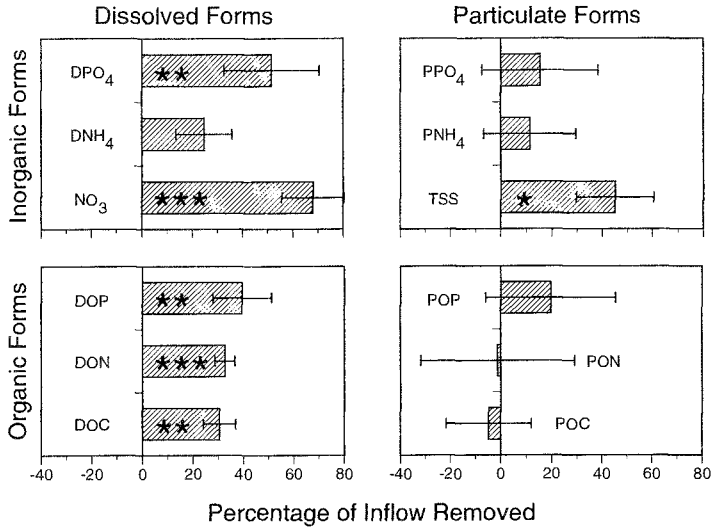

Fig. 5. Mean percentages of inflowing material removed as inferred from differences between inflowing and outflowing concentrations. Percentages for dissolved inorganic forms are averaged for 7 wetlands. Percentages for other forms are averaged for 4 wetlands. Brackets are ± standard errors. Asterisks indicate levels of significanee for difference from zero based on the t test (*** $p<0.01$, ** $0.01<p<0.05$, * $0.05<p<0.07$). TSS is grouped with particulate inorganic forms although it includes particulate organic matter.

of the inflowing N. The efficiency of P removal by wetlands has been sucessfully predicted with a Vollenweider model, which considers both nutrient concentration and water through-flow (Mitseh *et al.* 1995). However, P removal is influenced not only by hydrological loading but also by the chemical composition of wetland sediments (Richardson 1985). Clearly, more research is needed to account for the enormous differences in N and P removal among wetlands.

It is surprising that our restored wetlands can act as depositional environments, trapping TSS, and yet be poor traps for particulate nutrients. Perhaps particle deposition is counter-balanced by export of particulate organic matter produced by wetland maerophytes (e.g., Table 2), periphyton, benthic algae, and phytoplankton. In contrast to our restored wetlands, braekish tidal marshes on the western shore of the Chesapeake Bay trap particulate nutrients and release dissolved nutrients (Jordan *et al.* 1983). The opposite behavior of tidal marshes may compliment the nutrient removal effect of freshwater wetlands just upstream. Our results suggest that it is important to distinguish the fate of particulate and dissolved organic nutrients, but few studies have considered these fractions separately.

To improve estimates of nutrient removal by our wetlands, we need measuremcnts of the amount of water inflow and outflow. Our present estimates, for example, are somewhat low since they ignore evaporation within the wetlands. Also, the high variance in concentrations among sampling dates and the paucity of dates sampled limits the precision of our estimates of nutrient trapping. At the four most intensivcly sampled wetlands, we are now using automated samplers to measure

water inflow and outflow collect samples of inflowing and outflowing water in volumes proportional to flow. This will greatly improve the precision and accuracy of our measurements of nutrient trapping.

Acknowledgements

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References

APHA (American Public Health Association). 1989. Standard methods for the examination of water and wastewater, 17 ed. APHA, Washington, D. C.

Boynton, W.R., Kemp, W.M. & Keefe, C.W. 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Kennedy, V. (ed.), Estuarine Comparisons. pp. 69-90. Academic Press, New York.

Conley, D.J. & Malone, T.C. 1992. Annual cycle of dissolved silicate in Chesapeake Bay: Implications for the production and fate of phytoplankton biomass. Marine Ecology Progress Series, 81, 121-128.

Correll, D. L. & Micklas, J. J. 1975. Phosphorus cycling in a Maryland deciduous forest subjected to various levels of mineral nutrient loading. In: Howell, F. G., Gentry, J. B. & Smith, M. H. (eds.), Mineral Cycling in Southeastern Ecosystems, Energy Research and Development Administration Symposium Series, Conference 740513. pp. 642-657. United States Energy Research and Development Administration, Washington D. C.

Fleischer, S., Gustafson, A., Joelsson, A., Pansar, J. & Stibe, L. 1994. Nitrogen removal in created ponds. Ambio, 23, 349-357.

Gallegos, C.L., Jordan, T.E. & Correll, D.L. 1992. Event-scale response of phytoplankton to watershed inputs in a subestuary: Timing, magnitude and location of blooms. Limnology and Oceanography, 37, 813-828.

Gaudy, A. F. & Ramanathan, M. 1964. A colorimetric method for determining chemical oxygen demand. J. Water Pollut. Control Fed., 36, 1479-1487.

Jansson, M., Andersson, R., Berggren, H. & Leonardson, L. 1994. Wetlands and lakes as nitrogen traps. Ambio, 23, 320-325.

Jordan, T.E., Correll, D.L. & Weller, D.E. 1997a. Effects of agriculture on discharges of nutrients from Coastal Plain watersheds of Chesapeake Bay. Journal of Environmental Quality, 26, 836-848.

Jordan, T.E., Correll, D.L. & Weller, D.E. 1997b. Nonpoint source discharges of nutrients from Piedmont watersheds of Chesapeake Bay. Journal of the American Water Resources Association, 33, 631-645.

Jordan, T.E., Correll, D.L. & Weller, D.E. 1997c. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research, 33, 2579-2590.

Jordan, T. E., Correll, D. L. & Whigham, D. F. 1983. Nutrient flux in the Rhode River: Tidal exchange of nutrients by brackish marshes. Estuarine, Coastal and Shelf Science, 17, 651-667.

Jordan, T.E. & Weller, D.E. 1996. Human contributions to terrestrial nitrogen flux. BioScience, 46, 655-664.

Keeney, D. R. & Nelson, D. 1982. Nitrogen- inorganic forms. In: Page, A. L. *et al.* (eds.), Methods of Soil Analysis: Part 2, 2nd Edition. pp. 643-698. American Society of Agronomy, Inc., Soil Science Society of America, Inc., Madison, Wisconsin.

Kemp, W.M., Twilley, R.R., Stevenson, J.C., Boynton, W.R. & Means, J.C. 1983. The decline of submerged vascular plants in upper Chesapeake Bay: Summary of results concerning possible causes. Marine Technology Society Journal, 17, 78-89.

King, E. J. 1932. The colorimetric determination of phosphorus. Biochem. J., 26, 292-297.

Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucas, W. & Todd, A.H. 1995. Water Quality Functions of Riparian Forest Buffer Systems in the Chesapeake Bay Watershed. EPA 903-R95-004, CBP/TRS 134/95, U. S. Environmental Protection Agency, Washington, D.C.

Maciolek, J. A. 1962. Limnological organic analyses by quantitative dichromate oxidation. U. S. Fish Wildlife Serv. Publ.

Malone, T.C., Kemp, W.M., Ducklow, H.W., Boynton, W.R., Tuttle, J.H. & Jonas, R.B. 1986. Lateral variation in the production and fate of phytoplankton in partially stratified estuary. Marine Ecology Progress Series, 32, 149-160.

Martin, D. F. 1972. Marine Chemistry, V. 1. Dekker, New York.

Mitsch, W. J. 1994. The nonpoint source pollution control function of natural and constructed riparian wetlands. In: Mitsch, W. J. (ed.), Global Wetlands: Old World and New. pp. 351-361. Elsevier Science B. V., New York.

Mitsch, W. J., Cronk, J. K., Xinyuan Wu, Nairn, R. W. & Hey, D. L. 1995. Phosphorus retention in constructed freshwater riparian marshes. Ecological Applications, 5, 830-845.

Mitsch, W. J. & Gosselink, J. G. 1993. Wetlands. Academic Press, New York.

Officer, C.B., Biggs, R.B., Taft, J.L., Cronin, L.E., Tyler, M.A. & Boynton, W.R. 1984. Chesapeake Bay anoxia: Origin, development, significance. Science, 223, 22-27.

Pinay, G. & Decamps, H. 1988. The role of riparian woods in regulating nitrogen fluxes between the alluvial aquifer and surface water: A conceptual model. Regulated Rivers: Research and Management, 2, 507-516.

Raisin, G. W. & Mitchell, D. S. 1995. The use of wetlands for the control of non-point source pollution. Wat. Sci. Tech., 32, 177-186.

Richardson, C. J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science, 228, 1424-1427.

Richardson, C. J. 1989. Freshwater wetlands: Transformers, filters, or sinks? In: Sharitz, R. R. & Gibbons, J. W. (eds.), Freshwater Wetlands and Wildlife. pp. 25-46. USDOE Office of Scientific and Technical Information, Oak Ridge.

Richardson, C. J. & Marshall, P. E. 1986. Processes controlling the movement, storage, and export of phosphorus in a fen peatland. Ecological Monographs, 56, 279-302.

Sokal, R. R. & Rohlf, F. J. 1981. Biometry, Second Edition. W. H. Freeman and Company, San Francisco, California.

Strickland, J. D. H. & Parsons, T. R. 1972. A Practical Handbook of Seawater Analysis, 2nd ed. Fish. Res. Bd. Can. Bull., 167. Queen's Printer, Ottawa.

Taft, J.L., Taylor, W.R., Hartwig, E.O. & Loftus, R. 1980. Seasonal oxygen depletion In Chesapeake Bay. Estuaries, 3, 342-247.

Verhoeven, J. T. A. & van der Toorn, J. 1990. Marsh eutrophication and wastewater treatment. In: Patten, B. C. (ed.), Wetlands and Shallow Continental Water Bodies, Volume 1. pp. 571-585. SPB Academic Publishing, The Hague, The Netherlands.

Whigham, D.F. 1995. The role of wetlands, ponds, and shallow lakes in improving water quality. In: Steele K. (ed.), Animal Waste and the Land-Water Interface. pp. 163-172. Lewis Press, Boca Raton, Florida.