

A COMPARATIVE STUDY OF NITROGEN AND PHOSPHORUS CYCLING IN TIDAL AND NON-TIDAL RIVERINE WETLANDS

Jos T. A. Verhoeven¹, Dennis F. Whigham^{1,2}, Richard van Logtestijn¹, and Jay O'Neill²

¹ *Department of Geobiology, Section of Landscape Ecology*

University of Utrecht

P.O. Box 800.84

3508 TB Utrecht

The Netherlands

E-mail: j.t.a.verhoeven@bio.uu.nl

² *Smithsonian Environmental Research Center*

Box 28

Edgewater, Maryland, USA 21037

Abstract: This paper describes a study of nutrient dynamics in 12 tidal and non-tidal freshwater riverine wetlands in The Netherlands, Belgium, and Maryland (USA). The purpose of the study was to investigate the relationships between nutrient cycling processes in riverine wetlands that were geographically separated, that were dominated by different types of vegetation, and that had different hydrodynamics. We also compared restored and natural riverine wetlands. The results showed distinct differences in interstitial water chemistry between the sites in Maryland and Europe. No such regional differences were found in the soil variables, except for soil phosphorus, which was higher in The Netherlands. Soil organic matter, total nitrogen and phosphorus content, and bulk density were higher in tidal freshwater wetland soils. Forested wetland soils had higher organic matter and total nitrogen and lower bulk density and total phosphorus than soils from wetlands dominated by herbaceous species. Restored wetlands had lower soil organic matter and total soil nitrogen and phosphorus than similar types of natural riverine wetlands. There were no differences in nutrient-related process rates nor plant nutrient concentrations in tidal versus non-tidal riverine wetlands. Lower nitrogen and phosphorus concentrations in plants at the restored sites suggest that nutrient uptake by vegetation may be poorly coupled to rates of nutrient cycling during early stages of vegetation development. A principal components analysis of the data identified groupings of soil and water variables that were similar to those that had been previously identified when we applied the same methods to peatlands that were also geographically widely separated. Results of the study demonstrate that the techniques that we have been using are robust and repeatable. They are especially useful for making general comparisons of nitrogen and phosphorus cycling when there are limitations on the number of wetland that can be sampled. The approach that we have developed may also be used to calibrate and refine nutrient cycling models that are incorporated into wetland assessment procedures.

Key Words: The Netherlands, Maryland, Belgium, riverine, tidal, non-tidal, wetlands, vegetation, mineralization, denitrification, geographic comparison, decomposition

INTRODUCTION

For the past several years, we have been using standard methods to develop a regional and global approach to examine patterns of nutrient cycling in various types of wetlands to evaluate factors that are related to rates of nutrient cycling (Verhoeven et al. 1994, 1996). Our primary goal has been to use standard methodology to develop a global data set that could be used to compare nitrogen (N) and phosphorus (P) dynamics in similar types of wetlands that are geographically separated (i.e., on different continents) and that experience different levels of human perturbation.

This goal is of interest because it would allow us to evaluate factors that control N and P cycling under differing climatological conditions and under situations where human impacts on wetlands may be quite different (e.g., different levels of atmospheric loading of N, drainage). A secondary goal has been to develop a methodology that could be used to evaluate and refine functional assessment models that are used to assess nutrient cycling in wetlands. The HGM (hydrogeomorphic) and FAWE (Functional Analysis of European Wetland Ecosystems) approaches to functional assessment are two examples of assessment systems that are currently being developed and that include nu-

trient cycling models (Brinson 1993, Maltby *et al.* 1996, Brinson *et al.* 1998).

There are strengths and weaknesses in the HGM and FAEWE approaches to wetland assessment. A strength of the HGM approach is that variable scaling and selection are based on sampling many wetlands chosen to represent the range of natural and anthropogenic activities that impact the wetland class. A strength of the FAEWE approach is that the models are based on detailed scientific studies. A weakness of both approaches is that they have been developed without rigorous testing of the assumptions that the variables that are used to assess ecological functions are correlated with ecological processes that control the functions. It is unlikely, however, that it will ever be possible to have adequate funding to test the assumptions of the assessment models across a range of sites that would be representative of the different conditions that exist for a specific wetland class or hydrogeomorphic unit. The standardized and relatively rapid approach to measurements of nutrient cycling that we have implemented may serve to identify key components of nutrient cycling that can be used and calibrated in functional assessment models.

This paper reports results of a comparative study of riverine wetlands in the USA and Europe. Our objective was to compare N and P cycling in natural, created, and restored tidal and non-tidal riverine wetlands. We compared freshwater tidal wetlands with non-tidal riverine wetlands because the former have been purported to be among the most productive types of wetlands in the temperate zone (Odum *et al.* 1984, 1988, Simpson *et al.* 1983), and less is known about nutrient cycling in them in comparison to non-tidal riverine wetlands. We compared restored to natural wetlands because it is important to develop a better characterization of how ecological processes develop in restored wetlands (Middleton 1999). To achieve our objective, N- and P-related process rates were simultaneously measured in a standardized manner in wetlands bordering freshwater tidal and non-tidal rivers in The Netherlands, a freshwater tidal wetland in Belgium, and freshwater tidal and non-tidal wetlands in Maryland and the District of Columbia, USA. The approach involved measuring rates of N and P release or uptake in incubated soil samples during the period of active growth of vegetation in the early summer, measuring relative rates of decomposition, and denitrification during the same period and relating the process measurements to soil and interstitial water variables (Verhoeven *et al.* 1994, 1996). The wetlands selected differed in flooding frequency (twice daily bi-directional flooding in freshwater tidal wetlands versus only periodic and unidirectional flooding in non-tidal wetlands) and vegetation (dominance of woody versus herbaceous

vegetation). We are also able to use data collected in Maryland to contrast N and P cycling in restored freshwater tidal and non-tidal wetlands compared to nearby natural wetlands of the same types.

STUDY SITES

Field work was conducted in 1995 between 30 May–10 July in Europe and 15 May–25 June in Maryland. Nomenclature for plants follows Van der Meijden *et al.* (1984) for the European wetlands and Radford *et al.* (1968) for Maryland. We studied wetlands along two rivers in The Netherlands. Two wetlands were located in the river forelands ('uiterwaarden') in the Millingerwaard along the Waal River. The river forelands are positioned between the high 'winter dykes' that protect the hinterland from flooding, and the low 'summer dykes' that directly border the river channel. The two wetlands flood about once every two years during high winter river discharges. Wetlands at both sites had developed in abandoned clay pits, areas on the floodplain where clay had been extracted at least 50 year ago. Shallow surface water is present for most of the year at both wetlands. One wetland (NRH) was dominated by *Phragmites australis* (Cav.) Trin. ex Steudel, while the other (NRF) was vegetated with a 30-year-old forest dominated by *Salix alba* L. and *Salix purpurea* L. and a dense understory of *Urtica dioica* L. and *Rubus* sp. Two other Dutch sites were freshwater tidal wetlands along the Oude Maas between Rotterdam and Spijkenisse. One wetland (NTH) was dominated by herbaceous species (*Scirpus lacustris* L., *Scirpus maritimus* L., *Senecio paludosus* L.) and the other (NTF) by woody taxa (*Alnus glutinosa* (L.) Gaertner, *Salix* spp.) The depth of tidal flooding at NTH was ~ 1 m. Soils at NTF were continuously saturated but did not flood with each tide, and the depth of flooding was less than 1 m. A freshwater tidal wetland in Belgium (BTH) was located on the Schelde River near Hingene. The wetland was dominated by *Phragmites australis*, and it flooded twice daily to ~ 1 m.

In Maryland, one non-tidal riverine forested wetland and two freshwater tidal wetlands were sampled at the Jug Bay Wetland Sanctuary on the Patuxent River. The non-tidal forested wetland (ARF1) was located approximately 1 km upstream of the freshwater tidal wetland. Dominant trees were *Acer rubrum* L., *Betula nigra* L., *Liquidambar styraciflua* L., and *Liriodendron tulipifera* L. The dominant understory shrubs were *Viburnum dentatum* L. and *Lindera benzoin* (L.) Blume. One tidal wetland sampled at Jug Bay (ATS) was a shrub-scrub habitat dominated by *Alnus rugosa* (Du Roi) Sprengel, *Rosa palustris* Marshall, *Cornus amomum* Miller, and scattered *A. rubrum* and *Ce-*

phalanthus occidentalis L. Common herbaceous species were *Impatiens capensis* Meerb., *Peltandra virginica* (L.) Kunth., *Sagittaria latifolia* Willd., *Leersia oryzoides* (L.) Swartz., *Polygonum arifolium* L., *Typha latifolia* L., and *Sparganium* sp. The depth of twice-daily tidal flooding in the shrub-scrub wetland was less than 30 cm. The second tidal wetland at Jug Bay (ATH) was dominated by *Nuphar advena* Aiton., *Pontederia cordata* L., *T. latifolia*, *P. virginica*, and *P. australis*. The flooding regime at the site was similar to the one at ATS.

Two restored tidal freshwater wetlands were sampled in Maryland. One (ATH1*) was located along the Anacostia River, a heavily-polluted tributary of the Potomac River in the District of Columbia. The site had originally been a tidal wetland but had been dredged many years ago to create a tidally influenced lake. The site had been restored to a freshwater tidal wetland in 1992–1993 by raising the level of the substrate using coarse grained material (e.g., gravel) with a thin covering (10–50 cm) of fine sediments that had been dredged from the Anacostia River. Wetland vegetation was planted to provide initial colonization of the site but extensive recruitment and spread of natural vegetation had occurred by the time that we sampled the site in 1995. Dominant species were *T. latifolia*, *Acnida cannabina* (L.) J.D. Sauer, *Bidens laevis* (L.) BSP, *L. oryzoides*, *P. cordata*, *P. virginica*, *Scirpus americanus* Persoon, and *I. capensis*. The depth of the twice daily tidal flooding averaged ~ 50 cm. The second restored tidal freshwater wetland (ATH2*) was located 7 km upstream of Jug Bay on the Patuxent River. The restoration was completed in 1993. The wetland was similar to the Anacostia River wetland in that the substrate consisted of gravel overlain with a thin (10–20 cm) layer of fine dredged sediments. Dominant species were *T. latifolia*, *Eleocharis obtusa* (Willd.) Schultes, *A. cannabina*, *Ranunculus scleratus* L., *Echinochloa crus-galli* (L.) Beauvois, *Epilobium coloratum* Biehler, *Polygonum punctatum* Ell., *B. laevis*, *Ludwigia palustris* var. *americana* (DC.) Fernald & Griscom, *Juncus effusus* L., and *Cyperus diandrus* Torr. Scattered seedlings of *Salix* sp. and *Populus* sp. were present. Tidal flooding averaged between 30–50 cm.

Two non-tidal riverine wetlands were also sampled in Maryland. Both wetlands were located on the Patuxent River in Anne Arundel County in a suburban area between Washington, DC and Baltimore, MD. One wetland (ARF2) was a floodplain forest dominated by a mature canopy of *A. rubrum* and *Fraxinus pennsylvanica* Marshall. *Viburnum dentatum* was the dominant understory shrub. *Saururus cernuus* L., *I. capensis*, *Boehmeria cylindrica* (L.) Swartz, *Cinna arundinacea* L., and *L. oryzoides* were common herbaceous

species, and *Rhus radicans* L. was also abundant. The wetland experienced periodic overbank flooding from the Patuxent River and overland discharge from an adjacent watershed. The second Patuxent River wetland was a restored wetland (ARF*) that was located approximately 500 m from ARF2. Dominant herbaceous taxa at ARF* were *J. effusus*, *Cyperus* sp., *Panicum* sp., *Lycopus virginicus* L., *Arthraxon hispidus* var. *cryptantherus* (Hackel) Honda, and *Bidens bipinnata* L.. There was no evidence of surface flooding of the ARF* site, but pools of water, most likely from precipitation, were present in depressions on the wetland surface. Soil texture was spatially variable and consisted of areas with substrates that had a high clay content interspersed with areas with gravelly or sandy substrates. The substrate contained very little organic matter, and the litter layer on the soil surface was absent or poorly developed.

METHODS

At each wetland, 5 sampling locations were randomly chosen. Relative decomposition was evaluated using the cotton strip assay (Harrison et al. 1988). Duplicate cotton strips were placed vertically into the soil to a depth of 30 cm at each sampling location at the start of each field period. One cotton strip from each sampling location served as a control and was immediately removed and returned to the laboratory where they were washed in tap water followed by a second wash with distilled water. The cotton strips were then air dried. The second cotton strip in each pair was retrieved after 6 weeks, gently washed with tap water, rinsed in distilled water and air dried. Loss of cotton tensile strength (TSL) was measured on the dried samples (Maltby 1988).

Measurements of net N mineralization and P release or uptake were made using 6-week *in situ* soil incubations following procedures described and tested by Verhoeven et al. (1990, 1994, 1996) and Van Oorschot et al. (1997). Fresh (time = 0) and incubated (time = 6 weeks) soil material was extracted with 0.2 M KCl (available ammonium and nitrate) and 0.1 M HCl in combination with 0.03 M NH_4F , available phosphate, following the procedure described by Richardson and Marshall (1986). Extracts were analyzed for ammonium, nitrate and phosphate with a Skalar continuous-flow analyzer. Uptake and release were calculated as mgN or mgP/m²d.

Denitrification was measured at the start and at the end of the 6-week period by 24-h incubations of soil using the acetylene blockage technique following procedure described in De Klein and Van Logtestijn (1994). Glass jars (diameter 9.5 cm) were filled with four undisturbed samples of the upper 10 cm of soil

that was collected with a 2.6-cm-diameter soil corer using plastic film to avoid compaction. The samples were wrapped in aluminum foil before being placed into the jars to prevent them from collapsing into a slurry. Jars were taken to the lab where they were flushed with nitrogen (N_2) gas to remove the air before they were sealed. The lids had two silicon rubber septa, through which 50 ml of acetylene was added to the headspace with a syringe. After 24 h of incubation at field temperature (about 15°C), 2 ml of gas were extracted from the headspace of the containers and N_2O concentrations were determined with ECD (Electron Capture Detection) gas chromatography. Denitrification was calculated as mgN/m^2 .

Interstitial soil water samples were collected at the end of the 6 week soil incubation periods from holes created with a soil corer. Electrical Conductivity (EC) and pH were measured in the field. Acid washed polyethylene bottles, filled completely to minimize contact with air, were used to transport the samples to the laboratory where they were stored at 4°C . Concentrations of dissolved nitrate (NO_3^-), ammonium (NH_4^+), phosphate (PO_4^{3-}), sulfate (SO_4^{2-}), calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^+), potassium (K^+), aluminum (Al^{3+}), iron (Fe), bicarbonate (HCO_3^-), and chlorine (Cl^-) were determined on $2\ \mu\text{-Millipore}$ filtered samples with a Skalar continuous-flow analyzer. The water table was measured in the holes relative to soil surface at the beginning and at the end of the field period.

Soil temperatures were measured at 5 cm below the soil surface using soil thermistors. Soil redox potential was measured using platinum-calomel probes that were placed into the soil one week before the start of the 6-week field period at a depth of 5 cm below soil surface at each sampling location. Both temperature and redox potential were measured at the beginning and at the end of the study periods, and the average values of these 2 measurements were used in the data analysis.

Measurements of soil variables included extractable N and P, total N and P, organic matter (OM), bulk density and water content. The variables were measured on samples collected at the end of the 6-week study period at each location. Extractable N and P were determined as described above for the incubated samples. Organic matter was determined as loss on ignition at 550°C . Total N and P were measured after acid digestion according to a salicylic-acid thiosulphate modification of the Kjeldahl method (Page *et al.* 1982).

At the end of the 6-week field periods, all herbaceous vegetation was harvested from a $50 \times 50\ \text{cm}$ quadrat positioned adjacent to each of the locations used for the incubations. Samples were dried to con-

stant weight at 60°C , weighed, ground in Wiley mills, and analyzed for N and P using the same procedure described for total soil N and P.

Results were analyzed using SAS (1985). The similarity among wetlands based on process rates and environmental factors was evaluated with cluster analysis. Stepwise multiple regression was used to correlate variables for which rates were measured with the soil and water chemistry data. These parameters were further investigated for mutual relationships with a principal component analysis in the SAS 'FACTOR' procedure. Five factors were retained in the analysis on the basis of the criterion that each retained factor explained at least 10% of the total variance and had an eigenvalue >1 .

RESULTS

Interstitial Water

Interstitial water variables (Table 1) showed continental patterns, with lower pH, Ca, Mg, SO_4 , and HCO_3 and higher Fe concentrations in Maryland wetlands. Ammonium concentrations were also lower in interstitial water from the Maryland wetlands, but the differences were not statistically different. The cluster analysis of interstitial water variables resulted in separate groupings for the European and Maryland wetlands (Figure 1a). The European cluster was further subdivided into one subgroup composed of the two Dutch wetlands dominated by herbaceous vegetation which had lower interstitial Ca and Mg (Table 1). The second subgroup was composed of the two Dutch forested wetlands and the *Phragmites*-dominated site in Belgium: all with significantly higher NO_3 , Ca, Mg, and SO_4 . The Maryland cluster was also subdivided into two groups: one formed by wetlands dominated by trees or shrubs and one dominated by herbaceous species. The subgroup formed by the herbaceous wetlands had higher pH, calcium, chloride, and bicarbonate concentrations.

Soil

Soil parameters were highly variable among and within wetlands, and the only significant differences were that the restored wetlands (ATH1*, ARF*) had lower NH_4 and NO_3 , total N, O.M., and higher bulk density (Table 1). Cluster analysis of the soil data grouped together the 3 wetlands dominated by forests (Figure 1b). The exception was ARF2 (Figure 1b), which for no clear reason grouped with two of the restored wetlands. Forested wetlands were separated from the other wetlands because they had higher redox and lower O.M. (Table 1). The ATH and BTH wet-

Table 1. Interstitial water and soil variables for the sites studied. Values are means of 5 samples; values with the same letter are not significantly different ($P < 0.05$). Concentrations in water as mg/L, redox (Eh) as mV, extractable N or P as mgN/kg or mgP/kg dry soil, total N or P as gN/kg or gP/kg dry soil, organic matter as g/g dry soil, bulk density as kg/L. Codes: A—Maryland, USA, B—Belgium, N—The Netherlands, T—tidal wetland, R—non-tidal river floodplain; H—herbaceous, S—scrub, F—forested. * restored wetland.

Site	ATH1*	ATH2*	ATH	ATS	ARF1	ARF2	ARF*	BTH	NTH	NTF	NRH	NRF
Water variables												
pH	6.7 ^{abc}	6.6 ^{bc}	6.5 ^{bc}	6.2 ^{cd}	4.9 ^{de}	5.5 ^{de}	5.3 ^{de}	7.3 ^a	6.7 ^{abc}	7.1 ^{ab}	7.0 ^{ab}	7.2 ^{ab}
NH ₄	1.580 ^{ab}	3.250 ^a	3.640 ^a	1.040 ^{ab}	1.486 ^{ab}	1.318 ^{ab}	2.287 ^{ab}	1.188 ^{ab}	0.804 ^b	0.808 ^b	0.912 ^{ab}	0.360 ^b
PO ₄	0.072 ^{ab}	0.030 ^b	—	0.590 ^a	0.052 ^{ab}	0.112 ^{ab}	0.086 ^{ab}	0.370 ^{ab}	0.172 ^{ab}	0.678 ^a	0.014 ^b	0.096 ^{ab}
NO ₃	0.160 ^{ab}	0.178 ^{ab}	0.046 ^b	0.142 ^{ab}	0.138 ^{ab}	0.126 ^{ab}	1.517 ^{ab}	0.186 ^{ab}	0.016 ^b	0.172 ^{ab}	0.056 ^b	1.210 ^{ab}
K	6.27 ^a	3.17 ^a	4.94 ^a	2.21 ^a	1.52 ^a	2.06 ^a	6.23 ^a	3.32 ^a	4.77 ^a	3.26 ^a	2.75 ^a	2.87 ^a
Ca	45.1 ^{cd}	28.5 ^d	24.9 ^d	10.1 ^d	10.5 ^d	8.6 ^d	12.8 ^d	148.3 ^a	82.3 ^c	161.3 ^a	92.3 ^{bc}	131.7 ^{ab}
Mg	3.4 ^d	6.1 ^{cd}	5.2 ^d	2.4 ^d	3.0 ^d	2.1 ^d	4.4 ^d	13.0 ^{ab}	10.7 ^{bc}	20.8 ^a	10.9 ^b	15.1 ^a
Fe	1.74 ^b	2.22 ^b	9.24 ^a	2.22 ^b	1.13 ^b	2.23 ^b	3.57 ^{ab}	0.18 ^b	0.31 ^b	0.08 ^b	0.13 ^b	0.11 ^b
Al	0.024 ^b	0.024 ^b	0.154 ^{ab}	0.060 ^b	0.290 ^a	0.58 ^b	0.173 ^a	0.042 ^b	0.014 ^b	0.012 ^b	0.004 ^b	0.008 ^b
SO ₄	10.3 ^e	21.2 ^c	—	18.7 ^c	35.7 ^c	13.7 ^c	38.4 ^c	64.9 ^{bc}	62.1 ^c	216.4 ^a	31.3 ^c	153.1 ^{ab}
Cl	33.9 ^{cd}	39.3 ^b	36.3 ^b	11.9 ^{bc}	5.4 ^{cd}	17.7 ^{bc}	16.8 ^{bc}	100.7 ^a	73.0 ^a	83.5 ^a	94.5 ^a	93.5 ^a
HCO ₃	134.1 ^d	110.6 ^d	118.1 ^d	22.9 ^e	2.3 ^e	15.1 ^e	11.8 ^e	356.4 ^b	263.0 ^e	491.8 ^a	304.3 ^{bc}	322.1 ^{bc}
Soil variables												
Redox	174.4 ^d	108.9 ^{abcd}	-34.3 ^d	0.9 ^{cd}	405.4 ^a	191.0 ^{ab}	319.0 ^{bc}	219.9 ^{abcd}	-34.6 ^d	78.8 ^{abcd}	-13.3 ^d	351.0 ^{ab}
tot N	0.21 ^e	1.47 ^{de}	12.19 ^a	12.10 ^b	2.10 ^{cde}	1.90 ^{cde}	0.74 ^e	6.99 ^b	0.66 ^e	13.52 ^a	5.68 ^{bc}	5.25 ^{bcd}
tot P	0.32 ^{de}	0.72 ^{cde}	1.48 ^c	1.26 ^{cd}	0.68 ^{cde}	0.66 ^{cde}	0.16 ^e	7.01 ^a	0.46 ^{cde}	5.0 ^b	1.25 ^{cd}	1.40 ^c
extr NH ₄	5.25 ^{ab}	18.42 ^{ab}	23.52 ^a	16.90 ^{ab}	8.85 ^b	7.84 ^{ab}	4.38 ^b	2.18 ^b	2.63 ^b	14.02 ^{ab}	15.54 ^{ab}	3.19 ^b
extr NO ₃	0.33 ^c	0.45 ^c	0.97 ^{bc}	0.82 ^{bc}	0.53 ^c	0.65 ^c	0.28 ^c	1.41 ^{abc}	0.08 ^c	3.10 ^{ab}	0.57 ^c	3.48 ^a
extr PO ₄	39.9 ^c	41.7 ^c	277.8 ^{ab}	96.8 ^{bc}	35.3 ^c	9.5 ^c	3.0 ^c	296.7 ^a	56.2 ^c	114.8 ^{bc}	4.9 ^c	14.2 ^c
O.M.	0.04 ^d	0.10 ^{cd}	0.45 ^a	0.44 ^a	0.08 ^{cd}	0.10 ^{cd}	0.04 ^d	0.22 ^{bc}	0.06 ^{cd}	0.35 ^{ab}	0.16 ^{cd}	0.15 ^{cd}
bulk density	1.23 ^{ab}	1.17 ^{abc}	0.17 ^g	0.20 ^g	0.95 ^{cd}	0.86 ^d	1.38 ^a	0.39 ^{efg}	1.12 ^{bc}	0.21 ^{fg}	0.44 ^{ef}	0.45 ^e

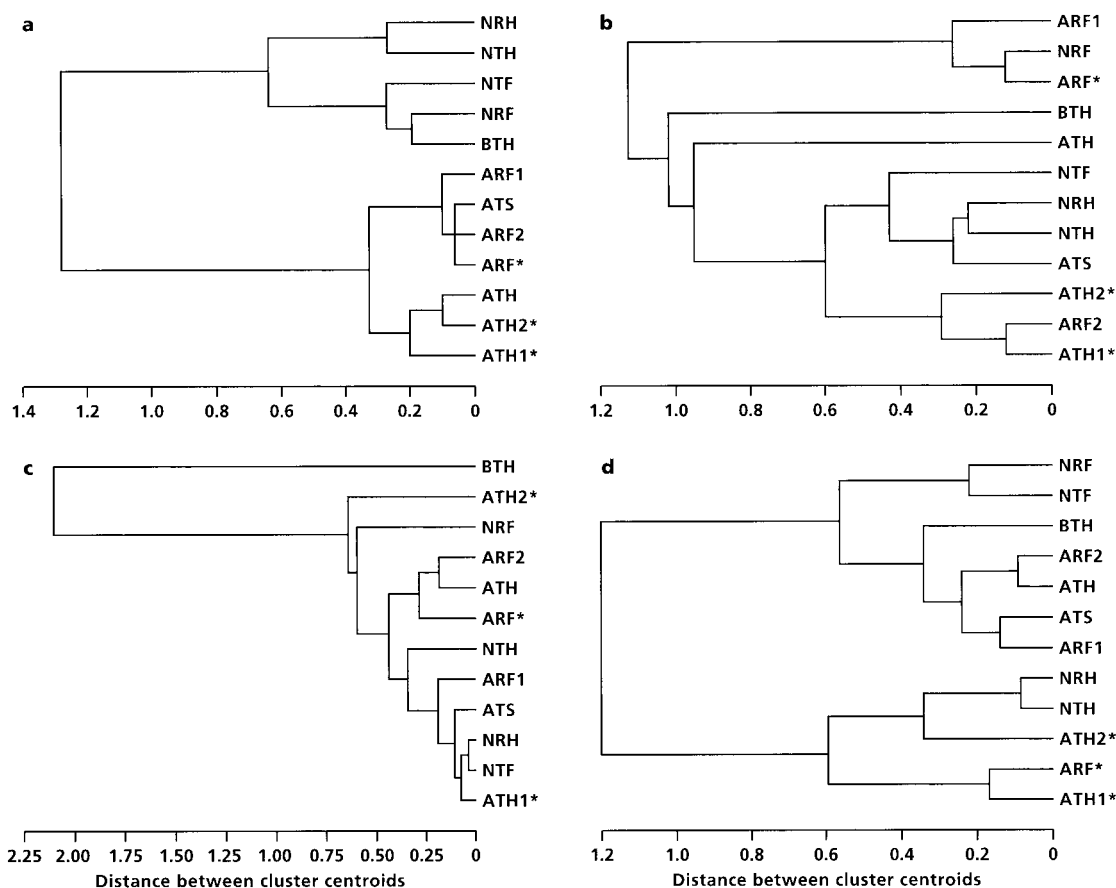


Figure 1. Dendrograms indicating the distance between cluster centroids. For a key to the sites see Table 1. a. Cluster analysis based on interstitial water variables (pH, Ca, Mg, Fe, Al, Cl, PO₄, NH₄, NO₃). b. Cluster analysis based on soil variables (organic matter, bulk density, redox, extractable ammonium, extractable phosphate, total soil N, total soil P). c. Cluster analysis based on process rates (cotton TSL, N mineralization, P release, denitrification). d. Cluster analysis based on plant N and P concentrations.

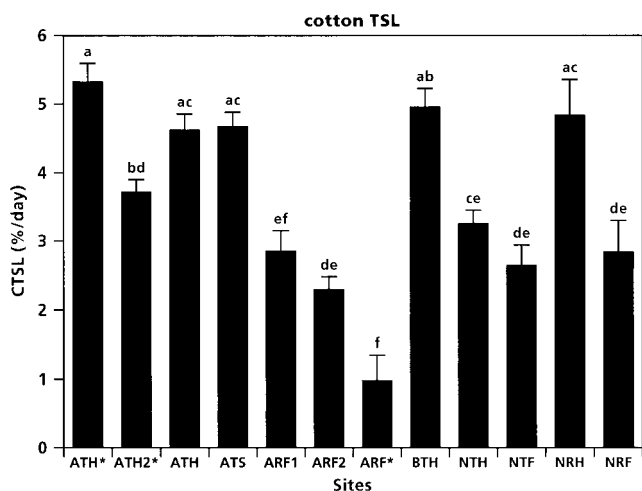


Figure 2. Cotton tensile strength loss (% per day) with SE in the soil of the sites studied during the spring field period. For a key to the sites see Table 1.

lands were also separated from all others in the cluster analysis because they had high extractable PO₄ concentrations (Table 1).

Process Rates

Cotton tensile strength loss was greater in most of the wetlands dominated by herbaceous vegetation and the scrub-shrub wetland at Jug Bay (Figure 2). The TSL was very low for one of the restored (ARF*) wetlands (Figure 2). Nitrogen mineralization rates were greatest in the freshwater tidal wetland in Belgium (BTH), one of the forested wetlands in The Netherlands (NRF), and two restored wetlands (ARF*, ARH*) in Maryland (Figure 3). Denitrification rates were low in all wetlands, with the exception of the Belgian site (BTH) and one of the forested sites (NRF) in Holland (Figure 4). The restored wetlands had both high (ARF*, ATH2*) or low (ATH1*) N mineralization rates compared to similar types of natural wetlands (ARF and ATH, respectively). Phosphorus re-

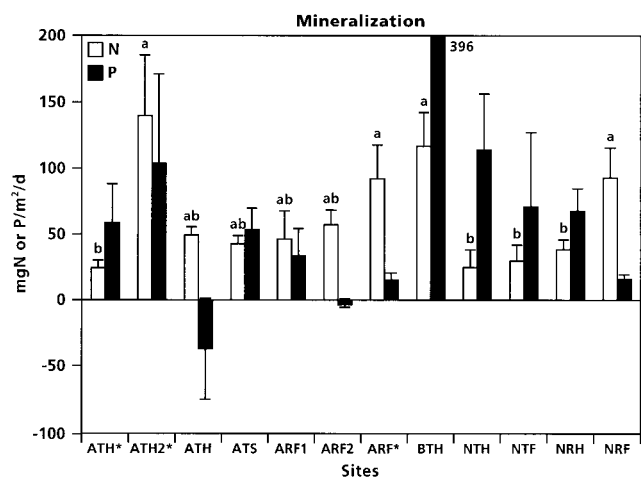


Figure 3. N mineralization (mg N m⁻²d⁻¹) and P release (mg P m⁻²d⁻¹) with SE in the soil of the sites studied.

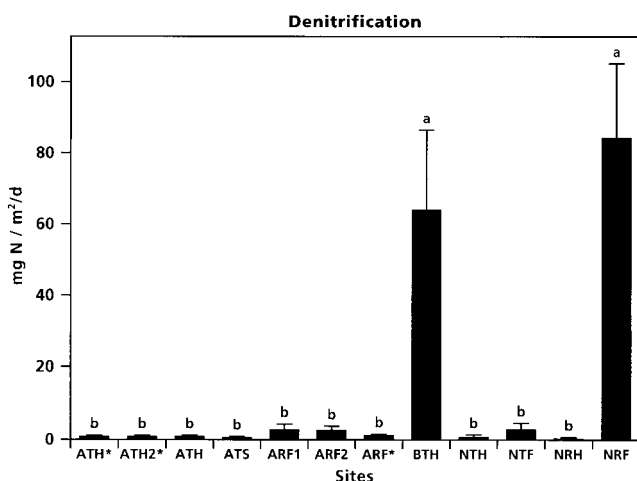


Figure 4. Denitrification (mg N m⁻²d⁻¹) with SE in the soil of the sites studied.

lease was greatest in the Belgium wetland, intermediate in The Netherlands wetlands, and lowest in the Maryland wetlands (Figure 3). Two Maryland sites (ATH, ARF2) had a net P uptake. Cluster analysis of the process rate data showed that the Belgian wetland (BTH) was very different from the other wetlands (Figure 1c) due to the high rates of P release and denitrification. The ATH2* and NRF wetlands were clustered separately from BTH and all other wetlands because of high N and P release rates in both wetlands, low TSL at ATH2*, and high denitrification at NRF. The third cluster was subdivided into two groups that were characterized by low P release (ARF2, ARF*) or P uptake (ATH).

Plant Nutrients

Plant N concentrations were highest in two Dutch forested wetlands (NTF, NRF) compared to the European wetlands dominated by herbaceous species (Figure 5). Plant N concentrations were similar among the Maryland wetlands, with the exception of low values for vegetation in the 3 restored wetlands (Figure 5). Phosphorus concentrations in plant tissues were generally higher in vegetation in forested wetlands and lowest in restored wetlands (Figure 5). Cluster analysis resulted in two main groupings. Low N and P tissue concentration were characteristic of one cluster that included the 3 restored wetlands and the two Dutch herbaceous wetlands (Figures 1d). The second cluster was divided into a subgroup formed by the NRF and NTF wetlands that had very high tissue nutrient concentrations and a subgroup consisting of the ARF* and ATH2* wetlands that had low tissue N and P concentrations.

Comparison of Interstitial Water and Soil Variables

Principle Component Analysis of the interstitial water and soil variables resulted in 5 factors that explained 76.2% of the total variance (Table 2). The first factor was highly correlated with interstitial water variables related to the base status of the wetland (i.e., HCO₃, Ca, Mg, and pH). The second factor was correlated with soil characteristics (organic matter, bulk density, total N, and extractable NH₄). The third factor was correlated with total and extractable P and with interstitial PO₄. Interstitial water concentrations of NH₄, Fe, and K were correlated with the 4th PCA component and NO₃ and redox potential with the 5th factor, respectively. The five PCA components can be characterized as ‘base status’ (i.e., base saturation status), ‘soil N/organic matter’, ‘soil P’, ‘water nutrients’, and ‘redox status’, respectively.

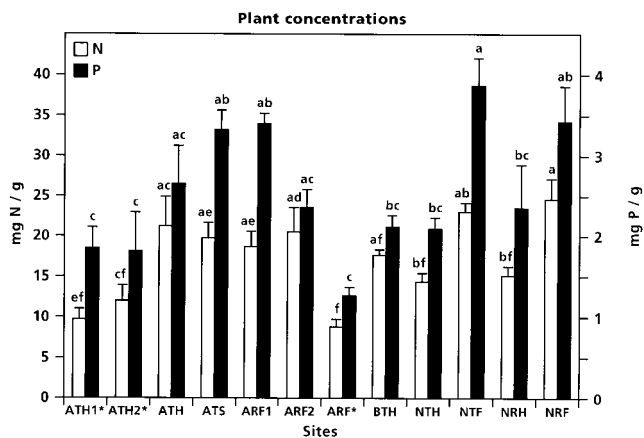


Figure 5. Plant N and P concentrations (mg g⁻¹) in the above-ground vegetation at the sites studied.

Table 2. Principal components of the soil (s) and interstitial water (w) variables.

Principal component	I base status	II soil N/org. matter	III soil P	IV water nutrients	V redox status
Cl (w)	0.91	-0.08	0.20	-0.06	0.04
HCO ₃ (w)	0.90	0.09	0.28	-0.13	0.06
Mg (w)	0.87	0.14	0.21	-0.15	0.21
Ca (w)	0.87	0.05	0.32	-0.21	0.20
pH (w)	0.86	0.14	0.11	0.15	-0.17
SO ₄ (w)	0.60	0.25	0.08	-0.24	0.50
Al (w)	-0.59	-0.14	0.23	0.14	0.39
O.M. (s)	0.13	0.86	0.33	-0.14	0.02
total N (s)	0.28	0.81	0.40	-0.17	0.07
extr. NH ₄ (s)	-0.13	0.69	-0.18	0.23	-0.23
bulk dens. (s)	-0.35	-0.71	-0.35	0.28	-0.08
extr. PO ₄ (s)	0.25	0.10	0.87	0.01	-0.11
total P (s)	0.46	0.19	0.78	-0.11	0.06
PO ₄ (w)	0.04	0.40	0.54	0.05	-0.02
NH ₄ (w)	-0.16	-0.04	-0.01	0.85	-0.01
Fe (w)	-0.38	0.12	-0.10	0.75	0.03
K (w)	0.20	-0.34	0.07	0.54	-0.04
extr. NO ₃ (s)	0.47	0.26	-0.06	-0.18	0.72
NO ₃ (w)	0.07	-0.05	-0.16	0.22	0.71
redox (s)	-0.29	-0.33	0.14	-0.23	0.66
variance expl. (% of total)	28.1	15.1	12.3	10.4	10.3

Comparison of Process Rates with Interstitial Water, Soil, and Plant Variables

A stepwise multiple regression of the process rates and plant nutrient concentrations with the PCA factors showed that the loss of cotton tensile strength was negatively related to 'redox status' (factor 5) and that it explained 22% of the variance (Table 3). Nitrogen mineralization was positively related to 'soil N/organic matter' and 'soil P' (PCA factors 2 and 3, respectively), and it explained 43% of the variance. The 'soil P' and 'base status' factors (factors 3 and 1, respectively) explained 56% of the variance associated with P release from the soil. Denitrification was positively related to 'redox status' (factor 5) of the soil, as well as 'base status' (factor 1) and 'water N and K' (factor 4) concentrations of interstitial water. The three factors explained 60% of the variance in rates of denitrification. 'Redox status', 'soil N/organic matter', and 'water nutrients' (factors 5, 2 and 4) explained about 50% of the variance in both plant N and P concentrations.

DISCUSSION

Interstitial Water and Substrates

The chemical composition of interstitial soil water showed clear geographic differences (Figure 1a). Maryland wetlands had higher interstitial NH₄, Fe, and Al concentrations and lower concentrations of Ca, Na,

Mg, HCO₃, SO₄, and Cl and a lower pH (Table 1). The geographic pattern is most likely due to differences in the parent rock material, which is the source of water solutes and sediments carried by the tributaries to rivers. Research in the Chesapeake Bay region has demonstrated that differences in water quality parameters can be anticipated when there are regional differences in geologic characteristics. Jordan and colleagues (Jordan *et al.* 1997a, 1997b, 1997c) examined the relationships between land-use patterns and stream water quality in several hundred watersheds throughout the Chesapeake Bay region. They found that watersheds with similar land-use characteristics had different stream water quality parameters and that the differences were related to differences in underlying geology across physiographic regions. Other factors can also influence water quality parameters. The high interstitial Na and Cl in the Dutch wetlands, for example, are probably related to high concentrations of these ions in Rhine water, which is the result of mining activities in France (Van Dijk *et al.* 1994). Regional differences in atmospheric deposition of nitrogen might also result in higher interstitial NH₄ concentrations. The presence of higher interstitial NH₄ in Maryland wetlands was surprising, as N loading rates to Dutch and Belgian rivers would be assumed to be much higher than in Maryland due to higher rates of atmospheric N deposition (Erisman 1991, Isermann 1993, Jordan and Weller 1996). Jordan and Weller (1996), however,

Table 3. Stepwise multiple regressions of process rates versus the principal components of water and soil factors (see Table 2). RC—standardized coefficient.

Step	Variable	Model R ²	F	P	RC
Dependent variable: Cotton tensile strength loss					
1	Factor 5	0.2	13.50	0.0006	-0.46
Dependent variable: N mineralization					
1	Factor 2	0.2	19.16	<0.0001	0.52
2	Factor 3	0.4	14.22	0.0004	0.40
Dependent variable: P release					
1	Factor 3	0.4	49.36	<0.0001	0.70
2	Factor 1	0.5	7.44	0.0088	0.26
Dependent variable: Denitrification					
1	Factor 5	0.4	40.71	<0.0001	0.70
2	Factor 1	0.5	11.26	0.0016	0.33
3	Factor 4	0.6	3.15	0.0826	-0.17
Dependent variable: Plant N content					
1	Factor 5	0.2	12.81	0.0008	0.45
2	Factor 2	0.3	15.20	0.0003	0.43
3	Factor 4	0.5	10.27	0.0024	0.33
4	Factor 1	0.5	3.08	0.0859	0.17
Dependent variable: Plant P content					
1	Factor 2	0.3	22.07	<0.0001	0.55
2	Factor 5	0.4	14.72	0.0004	0.40
3	Factor 4	0.5	8.95	0.0044	0.29

have found that atmospheric loading to the Chesapeake Bay region is relatively high, and all of the wetlands that we studied are located near wastewater treatment facilities, which would be significant sources of nitrogen. Our results also suggest that the chemical characteristics of interstitial water are likely to vary little between wetlands that occur in the tidal and non-tidal portions of rivers. This result is surprising because, at least in Maryland, freshwater tidal wetlands usually are located in the portions of rivers that have greater amounts of urban and suburban development and, therefore, higher nutrient loading rates (Simpson et al. 1983).

The restored sites in Maryland had higher Ca, HCO₃, and NH₄ concentrations than their natural counterparts. Chemical characteristics have almost always been shown to differ when restored or created wetlands have been compared to their natural counterparts, but the differences are not always consistent. Galatowitsch and van der Valk (1996) showed that water in restored wetlands in the Prairie Pothole region had a higher pH and lower levels of alkalinity, conductivity, Ca, and Mg than comparable natural wetlands. Wilson and Mitsch (1996) found that mitigation wetlands in Ohio had lower P, K, Ca, and Mg.

The distinct differences between Maryland and Hol-

land/Belgium in interstitial water were not reflected in a similarly strong regional differences in the physico-chemical soil characteristics (Figure 1b). The one clear regional difference among soils was a higher total soil P content in the Dutch/Belgian sites, which is most likely related to the history of decades of high P loads in the Rhine and Scheldt river basins (Van Dijk et al. 1994, Dogterom et al. 1998). Differences in soil characteristics among the 12 wetlands were most clearly related to water regime and the dominant plant type. Soil organic matter, total N, total P, and bulk density were all higher in tidal wetlands. The regular deposition of nutrient-rich riverine sediments in tidal wetlands might be one reason for the higher N, P, and bulk densities (Simpson et al. 1983, Bowden 1984). Findley et al. (1990), however, found that much of the organic detritus in Hudson River freshwater tidal wetlands was removed by tidal action, suggesting that the tidal wetland soils would not have higher levels of organic matter. Higher N and P in soils of tidal wetlands may also be due to their physical locations near potential urban and suburban sources of nutrients. As indicated earlier, many tidal freshwater wetlands are located in areas where discharges from wastewater treatment facilities are common.

Non-tidal forested wetlands had higher organic mat-

ter and total N and lower bulk density and total P than the herb-dominated wetlands. The presence of nitrogen-fixing alders in some of the forested sites may, in part, explain the higher N levels. A higher organic matter content, higher total N, and lower bulk density of soils in forested wetlands may also be due to low rates of decay of the woody plant tissues compared to the high rates of decomposition of herbaceous species in tidal freshwater wetlands (Odum 1984, Whigham *et al.* 1989), as was also confirmed in our cotton tensile strength loss results. The lower organic matter content of soils in the herbaceous tidal wetland may also be, as noted above, due to tidal export of organic matter (Findley *et al.* 1990).

The restored sites in Maryland differed from their natural counterparts by lower soil organic matter and total soil N and P. Lower organic matter content of soil and lower amounts of N and P have typically been found in restored wetlands. Soil organic matter and bulk density are two soil variables that are typically lower and higher, respectively in restored and created wetlands (Galatowitsch and van der Valk 1996, Stauffer and Brooks 1997, Mitsch *et al.* 1998). Obviously, the use of gravel with only a thin layer of fine river sediments on the surface was at least partially responsible for the observed physical and chemical characteristics of the soil.

N and P Cycling

None of the nutrient-related process rates nor the plant nutrient concentrations showed any clear regional patterns (Figure 1c,d), nor were there differences for tidal versus non-tidal wetlands within the same river systems. These results further demonstrate that the chemical characteristics of water and sediment in individual river systems have a greater impact on nutrient cycling processes than does the flooding regime (*i.e.*, tidal or non-tidal). There were also only minor influences of vegetation type on nutrient-related processes, again indicating the importance of the chemical characteristics of water and sediment in riverine systems.

Surprisingly, process rates in the restored and natural wetlands in Maryland were not significantly different, although significant differences did exist in the factors (*e.g.*, water chemistry and soil variables) that control N and P cycling. Lower N and P concentrations in plants and similar or higher concentrations of soil N and P at the restored sites suggest that plant nutrients were available but were not being taken up in plant biomass. The reasons for this situation are unclear, but the stage of vegetation development may have been important. Mitsch *et al.* (1998) found that several years were required before nutrient concentra-

tions of plants in restored sites reached the same levels as those found in natural wetlands. Lower N and P in the vegetation also suggests that vegetation uptake processes may be poorly coupled to rates of nutrient cycling during early stages of vegetation development in restored wetlands.

Simpson *et al.* (1983) and Odum *et al.* (1984) had suggested that freshwater tidal wetlands would have higher levels of productivity than non-tidal riverine wetlands because of the daily tidal subsidies. Higher rates of production would also be suggestive of higher rates of nutrient cycling if the vegetation was obtaining nutrients primarily from the soils. We found no correlative evidence to suggest that freshwater tidal wetlands had higher nutrient-related process rates even though the tidal wetland soils had higher total N and P. The high level of within-wetland variability in process rates could have obscured any differences between tidal and non-tidal wetlands. Another explanation for the absence of patterns could be that the nutrient release was not related to the total nutrient pool but rather to a limited pool of easily recyclable nutrients. The incubation procedures that we used are only suggestive of longer term (*e.g.*, 6 week) uptake or release of nutrients and would not be a good indicator of short-term turnover rates of N or P (Bowden 1986).

Comparison of Riverine and Fen Data Sets

Our primary long-term goal has been to use the methods that were employed in this study to develop a data base that can be used to compare general patterns of nutrient cycling among and between wetlands that are located in different regions of the world. We have previously compared fens and bogs in Maryland, The Netherlands, and Poland using these procedures (Verhoeven *et al.* 1994, 1996). Table 4 shows the results of a comparison of riverine and fens data sets. For this analysis, we grouped wetlands with similar vegetation structure. In the riverine data set, 3 structural types were distinguished: deciduous trees/shrubs, herbaceous plants, and *Phragmites* dominated wetlands. In the fen data set, 4 types are distinguished: evergreen shrubs, deciduous trees and shrubs, herbaceous plants, and *Sphagnum*-dominated wetlands.

For N and P mineralization, riverine sites dominated by herbaceous species had higher rates than riverine sites dominated by trees, which were in the same range as all of the fen sites. Nutrient concentrations in vegetation were in the same range for riverine and fen wetlands. The fact that most of the process rates were higher in the riverine sites illustrates their generally more nutrient-rich status.

The multivariate analyses of data from the two wetland categories also demonstrates that the standard

Table 4. Cotton tensile strength loss and soil nutrient process rates in (1) fens dominated by evergreen, deciduous, graminoid and Sphagnum plant types and (2) riverine wetlands dominated by deciduous, helophyte and herbaceous plant types. Data are means \pm S.E. Means followed by the same letter are not significantly different (Tukey test after ANOVA, $p < 0.05$). n: number of sampling stations.

	Fen Data Set			River Data Set			
	Evergreen	Deciduous	Graminoid	Sphagnum	Deciduous	Helophyte	Herbaceous
Cotton TSL	1.62 ^a \pm 0.04	1.59 ^{ns} \pm 0.16	1.18 ^a \pm 0.12	0.44 ^b \pm 0.06	3.02 ^b \pm 0.21	4.40 ^a \pm 0.29	4.37 ^a \pm 0.26
N mineralization	27.71 ^{ab} \pm 10.92	67.50 ^b \pm 12.69	20.69 ^b \pm 7.44	41.96 ^{ab} \pm 14.99	54.41 ^b \pm 7.96	98.86 ^a \pm 19.90	33.24 ^b \pm 5.69
P release	30.72 ^{bc} \pm 22.63	82.34 ^d \pm 29.11	3.12 ^{bc} \pm 1.60	11.42 ^{bc} \pm 5.81	34.38 ^b \pm 12.72	189.47 ^a \pm 46.11	45.10 ^b \pm 19.46
Denitrification	0.11 ^b \pm 0.03	3.29 ^{ns} \pm 1.31	0.15 ^b \pm 0.03	0.60 ^b \pm 0.21	18.61 ^a \pm 7.74	21.61 ^a \pm 10.62	0.65 ^b \pm 0.18
n	5	15	20	20	25	15	15

methods that we have chosen offer supporting evidence for the general conclusions that nutrient cycling processes in wetland are controlled by a well-delineated set of different factors that are related to clearly distinguishable soil (base status, N and P status, and redox state) and water (nutrient status) characteristics. In the riverine data set, the five principal components (Table 2) were basically similar to the four principal components that were identified in our earlier study of 12 fens and bogs (Table 2 in Verhoeven et al. 1996). These findings are not unexpected, but the data sets that we have compiled to date offer one of the first tests of these general assumptions using a standard method that is applied across more than one type of wetland as well as across a range of trophic conditions. The results also indicate that the methods that we have applied can be used to examine differences in N and P processing rates among and within different types of wetlands. Nitrogen mineralization and relative decomposition rates were, for example, found to be controlled by different factors in fens than in riverine wetlands, suggesting that for decomposition, redox status is more important in riverine wetlands and base status is more important in fen wetlands (Table 5). On the other hand, denitrification and P release were controlled by the same factors in the riverine and fen wetlands. In both wetland types, there was a positive correlation between denitrification and redox status, indicating that nitrification rather than denitrification is the rate-limiting process (Patrick et al. 1985).

Application to Functional Assessment Methodologies

Our findings also suggest that similar approaches may be used in developing and testing approaches to assess nutrient-related functions in wetland assessment systems. The results of our study confirm the general assumptions of functional assessment models such as FAEWE and HGM, which are based on the prediction that nutrient-related functions should be assessed and tested separately for different classes of wetlands (i.e., riverine versus non-riverine wetlands). Qualitative predictions using a relative scale (i.e., the process occurs at a low, medium, or high rate) may be based on the relationships between process rates and principal components of soil and water variables as described in this paper. Soil organic matter, water-table fluctuation during spring, interstitial water pH, interstitial water NH₄ and PO₄, and total soil P are relatively easy-to-measure indicator variables that are expected to predict the nutrient-related process rates well. This assumption will be further tested as the data set on process rates in different wetland types across geographic regions builds up.

Table 5. Comparison of multiple regressions of process rates and plant nutrient concentrations in the riverine sites with data collected in fens in Maryland, The Netherlands and Poland (see Verhoeven *et al.* 1996).

Process	Step	River Data Set			Fen Data Set		
		Princ. Comp.	Model R ²	Coeff.	Princ. Comp.	Model R ²	Coeff.
Cotton TSL	1	redox status	0.22	-0.46	base status	0.23	0.48
	2				soil P	0.42	0.43
N mineralization	1	soil N/org. m.	0.27	0.52	water nutrient	0.27	0.52
	2	soil P	0.43	0.40			
P release	1	soil P	0.49	0.70	soil P	0.43	0.65
	2	base status	0.56	0.26	water nutrient	0.59	0.42
	3				base status	0.62	-0.17
	4				redox status	0.64	-0.10
Denitrification	1	redox status	0.47	0.70	redox status	0.49	0.70
	2	base status	0.58	0.33	base status	0.56	0.26
	3	water nutrient	0.60	-0.17			
Plant N	1	redox status	0.20	0.45	redox status	0.47	0.68
	2	soil N/org. m.	0.39	0.43	water nutrient	0.66	0.43
	3	water nutrient	0.50	0.33	base status	0.77	0.34
	4	base status	0.53	0.18			
Plant P	1	soil N/org. m.	0.31	0.55	water nutrient	0.26	0.51
	2	redox status	0.47	0.40	soil P	0.52	0.51
	3	water nutrient	0.55	0.28	redox status	0.63	0.33
	4				base status	0.68	0.22

ACKNOWLEDGMENTS

The project was primarily supported by Collaborative Research Grant from the NATO International Scientific Exchange Programmes (grant No. 900425). Additional funding and support were provided by the Smithsonian Institution's Environmental Sciences Program and by Utrecht University. We thank the staff at the Patuxent Wildlife Research Center (Nell Baldacchino), Jug Bay Wetlands Sanctuary (Chris Swarth), and Kenilworth Park and Aquatic Gardens (Gentry Davis, Laura McKenzie) for permission to sample wetlands at their sites. Dr. Hugo Coops and Dr. Jacco van der Nat introduced the tidal sites in The Netherlands and Belgium to us. The manuscript was improved greatly by comments from Mark Brinson, Irv Mendelsohn, and two anonymous reviewers.

LITERATURE CITED

- Bowden, W. B. 1984. Nitrogen and phosphorus in the sediments of a tidal, freshwater marsh in Massachusetts. *Estuaries* 7:108-118.
- Bowden, W. B. 1986. Nitrification, nitrate reduction, and nitrogen immobilization in a tidal freshwater marsh sediment. *Ecology* 67:88-99.
- Brinson, M. M. (1993). A Hydrogeomorphic Classification for Wetlands. US Army Engineers Waterways Experiment Station, Vicksburg, MS, USA.
- Brinson, M. M., R. D. Smith, D. F. Whigham, L. C. Lee, R. D. Rheinhardt, and W. L. Nutter. 1998. Progress in development of the hydrogeomorphic approach for assessing the functioning of wetlands. p. 393-406. *In* A. J. McComb and J. A. Davis (eds.) *Wetlands for the Future*. Gleneagles Publishing, Adelaide, Australia.
- De Klein, C. A. M. and R. S. P. van Logtestijn. 1994. Denitrification in the top soil of managed grasslands in The Netherlands in relation to soil type and fertilizer level. *Plant and Soil* 163:33-44.
- Dogterom, J., P. J. Van der Wiele, and P. H. L. Buijs. 1998. Diffuse loadings as a yet unsolved problem for receiving water bodies. p. 211-228. *In* P. H. Nienhuis, R. S. E. W. Leuven, and A. M. J. Ragas (eds.) *New Concepts for Sustainable Management of River Basins*. Backhuys Publishers, Leiden, The Netherlands.
- Erisman, J. W. 1991. Acid deposition in the Netherlands. Rijksinstituut voor Volksgezondheid en Milieuhygiëne, Bilthoven. RIVM Report nr. 723001002:1-72.
- Findlay, S., K. Howe, and H. K. Austin 1990. Comparison of detritus dynamics in two tidal freshwater wetlands. *Ecology* 71:288-295.
- Galatowitsch, S. M. and A. G. van der Valk. 1996. Vegetation and environmental conditions in recently restored wetlands in the Prairie Pothole region of the U.S.A. *Vegetatio* 126:89-100.
- Harrison, A. F., P. M. Latter, and D. W. H. Walton. 1988. Cotton strip assay: an index of decomposition in soils. Institute of Terrestrial Ecology, Grange-over-Sands, UK.
- Isermann, K. 1993. Terrestrial, continental, and global aspects of C, N, P and S emissions from agricultural ecosystems. p. 79-121. *In* R. Wollast and F. T. Mackenzie (eds.) *Interactions of C, N, P, and S Biogeochemical Cycles and Global Change*. NATO ASI series. Volume 14. Springer-Verlag, New York, NY, USA.
- Jordan, T. E. and D. E. Weller. 1996. Human contribution to terrestrial nitrogen flux. *BioScience* 46:655-664.
- Jordan, T. E., D. L. Correll, and D. E. Weller. 1997a. Relating nutrient discharges from watersheds to land use and stream flow variability. *Water Resources Research* 33:2579-2590.
- Jordan, T. E., D. L. Correll, and D. E. Weller. 1997b. Effects of agriculture on discharges of nutrients from Coastal Plain watersheds of Chesapeake Bay. *Journal of Environmental Quality* 26:836-848.
- Jordan, T. E., D. L. Correll, and D. E. Weller. 1997c. Nonpoint source discharges of nutrients from Piedmont watersheds of Chesapeake Bay. *Journal of the American Water Resources Association* 33:631-645.
- Maltby, E. 1988. Use of cotton strip assay in wetland and upland environments—an international perspective. p. 140-156. *In* A. F.

- Harrison, P. M. Latter, and D. W. H. Walton (eds.) Cotton Strip Assay: an Index of Decomposition in Soils. Institute of Terrestrial Ecology, Grange-over-Sands, UK.
- Maltby, E., D. V. Hogan, and R. J. McInnes. 1996. Functional analysis of European wetland ecosystems—Phase 1 (FAEWE). European Commission, Office for Official Publications of the European Communities, Luxembourg. EUR 16132.
- Middleton, B. 1999. Wetland Restoration. Flood Pulsing and Disturbance Dynamics. John Wiley & Sons, Inc., New York, NY, USA.
- Mitsch, W. J., X. Wu, R. W. Nairn, P. E. Weihe, N. Wang, R. Deal, and C. E. Boucher. 1998. Creating and restoring wetlands. A whole ecosystem experiment in self-design. *BioScience* 48:1019–1030.
- Odum, W. E. 1988. Comparative ecology of tidal freshwater and salt marshes. *Annual Review Ecology and Systematics* 19:147–176.
- Odum, W. E., T. J. Smith III, J. K. Hoover, and C. C. McIvor. 1984. The Ecology of Tidal Freshwater Marshes of the United States East Coast: A Community Profile, U.S. Fish and Wildlife Service, Washington, DC, USA. FWS/OBS-87/17.
- Page, A. C., R. H. Miller, and D. R. Keeney. 1982. Methods of soil analyses, Part 2, Chemical and Microbiological Properties. American Society of Agronomy, Madison, WI, USA.
- Patrick Jr., W. H., D. S. Mikkelsen, and B. R. Wells. 1985. Plant nutrient behavior in flooded soil. p. 197–228 *In* O. P. Engelstad (ed.) Fertilizer Technology and Use. Soil Science Society of America Inc., Madison, WI, USA.
- Radford, A. E., H. E. Ahles, and C. R. Bell. 1968. Manual of the Vascular flora of the Carolinas. The University of North Carolina Press, Chapel Hill, NC, USA.
- Richardson, C. J. and P. E. Marshall. 1986. Processes controlling movement, storage and export of phosphorus in a fen peatland. *Ecological Monographs* 56:279–302.
- SAS. 1985. SAS/STAT guide for personal computers. SAS Institute, Cary, NC, USA.
- Simpson, R. L., R. E. Good, M. A. Leck, and D. F. Whigham. 1983. The ecology of freshwater tidal wetlands. *BioScience* 33:255–259.
- Stauffer, A. L. and R. B. Brooks. 1997. Plant and soil responses to salvaged marsh surface and organic matter amendments at a created wetland in central Pennsylvania. *Wetlands* 17:90–105.
- Van der Meijden, R., E. J. M. Arnolds, F. Adema, E. J. Weeda, and C. L. Plate. 1984. Standaardlijst van de Nederlandse flora 1983. Rijksherbarium, Leiden, The Netherlands.
- Van Dijk, G., L. van Liere, W. Admiraal, B. A. Bannink, and J. J. Cappon. 1994. Present state of the water quality of European rivers and implications for management. *The Science of the Total Environment* 1994:1–9.
- Van Oorschot, M. M. P., E. Robbemont, M. Boerstal, M. van Strien, and M. van Kerkhoven-Schmitz. 1997. Effects of enhanced nutrient availability on plant and soil nutrient dynamics in two English riverine ecosystems. *Journal of Ecology* 85:167–179.
- Verhoeven, J. T. A., E. Maltby, and M. B. Schmitz. 1990. Nitrogen and phosphorus mineralization in fens and bogs. *Journal of Ecology* 78:713–726.
- Verhoeven, J. T. A., D. F. Whigham, M. van Kerkhoven, J. O'Neill, and E. Maltby. 1994. A comparative study of nutrient-related processes in geographically separated wetlands: towards a science base for functional assessment procedures. p. 91–106. *In* J. W. Mitsch (ed.) *Global Wetlands; Old World and New*. Elsevier, Amsterdam, The Netherlands.
- Verhoeven, J. T. A., A. Keuter, R. van Logtestijn, M. B. van Kerkhoven, and M. Wassen. 1996. Control of local nutrient dynamics in mires by regional and climatic factors: a comparison of Dutch and Polish sites. *Journal of Ecology* 84:647–656.
- Whigham, D. F., R. L. Simpson, R. E. Good, and F. Sickels. 1989. Decomposition and nutrient-metal dynamics of litter in freshwater tidal wetlands. p. 167–187. *In* R. R. Sharitz and J. W. Gibbons (eds.) *Freshwater Wetlands and Wildlife*. USDOE Office of Scientific and Technical Information, Oak Ridge, TN, USA. U. S. Department of Energy Symposium Series No. 61.
- Wilson, R. F. and W. J. Mitsch. 1996. Functional assessment of five wetlands constructed to mitigate wetland losses in Ohio, U.S.A. *Wetlands*:436–451.

Manuscript received 17 March 1999; revisions received 30 June 2000; accepted 5 February 2001.