17 WETLANDS

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17.1 Ecosystem Description

The U.S. Clean Water Act defines wetlands as "those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions". Frequent water saturation of the soil creates anaerobic conditions and results in slow decomposition of organic matter. Wetlands in the United States support more than 4200 native plant species, of which 121 are federally threatened or endangered (NRCS 2009). Wetlands also offer key ecosystem services, such as sequestering carbon (C), providing habitat, regulating flood, maintaining water quality, and stabilizing coastal slope (US EPA 1993). About 107.7 million acres of wetlands are widely distributed in the conterminous United States (Fig. 17.1), 95 percent of which are freshwater wetlands and 5 percent are estuarine or marine wetlands (FWS 2005).

Depending on geology, position in the watershed, and climate, very different types of wetland ecosystems may develop. Wetlands have numerous classifications in North America. In general, marshes (dominated by herbaceous species) and swamps (dominated by woody species) occur on mineral soils or shallow peats. Peatlands, including ombrotrophic (raised) bogs, minerotrophic (true) fens, and transition (poor fens), are true peat-accumulating communities (Vitt 1994). Wetlands can be divided into three hydrology-based categories that are useful for discussing nitrogen (N) sources and cycling: bogs; fens, marshes, swamps, and riparian wetlands; and intertidal wetlands.

Ombrotrophic bogs are acidic, moss-dominated freshwater wetlands that receive water inputs primarily via precipitation. They develop where precipitation exceeds evapotranspiration and where there is some impediment to drainage of surplus water (Mitsch and Gosselink 2000). Bogs are especially common in the cool boreal zones of North America.

Fens, freshwater marshes, freshwater swamps, and riparian wetlands are characterized by ground and surface water inputs that are often on the same order of magnitude as precipitation (Koerselman 1989). Fens generally receive some drainage from surrounding mineral soils and, in many ecological aspects, are transitional between bogs and freshwater marshes. Freshwater marshes are characterized by tall graminoids, while freshwater swamps are forested wetlands dominated by various species (e.g., bald cypress [*Taxodium distichum*], pond cypress [*Taxodium ascendens*], white-cedar [*Chamaecyparis thyoides*], or red maple, [*Acer rubrum*]). Riparian wetlands are linear systems along rivers and streams and formed by the flooding of these streams or rivers.

Intertidal wetlands (e.g., salt marshes and mangrove swamps) develop on sheltered coasts or in estuaries where they are inundated by water with a periodicity that reflects the combination of tidal cycles and freshwater inputs (e.g., precipitation and ground water). Marine water input depends on the proximity to the coast; wetlands that occur nearest to the coast are the most saline and those towards the inland are primarily freshwater habitats (Archibold 1995, Mitsch and Gosselink 2000).

17.2 Ecosystem Responses to N Deposition

Known responses to N enrichment are generally derived from nutrient-addition studies in the field and observations along gradients of N deposition. A variety of ecological endpoints are evaluated, such as altered soil biogeochemistry, increased peat accumulation, elevated primary production, changes in plant morphology, changes in plant population dynamics, and altered plant species composition (US EPA 2008). Morris (1991) suggested the effect of N deposition on wetland ecosystems depends on the fraction of rainfall (a proxy for atmospheric N deposition) in its total water budget; the sensitivity to N deposition was suggested as bogs >

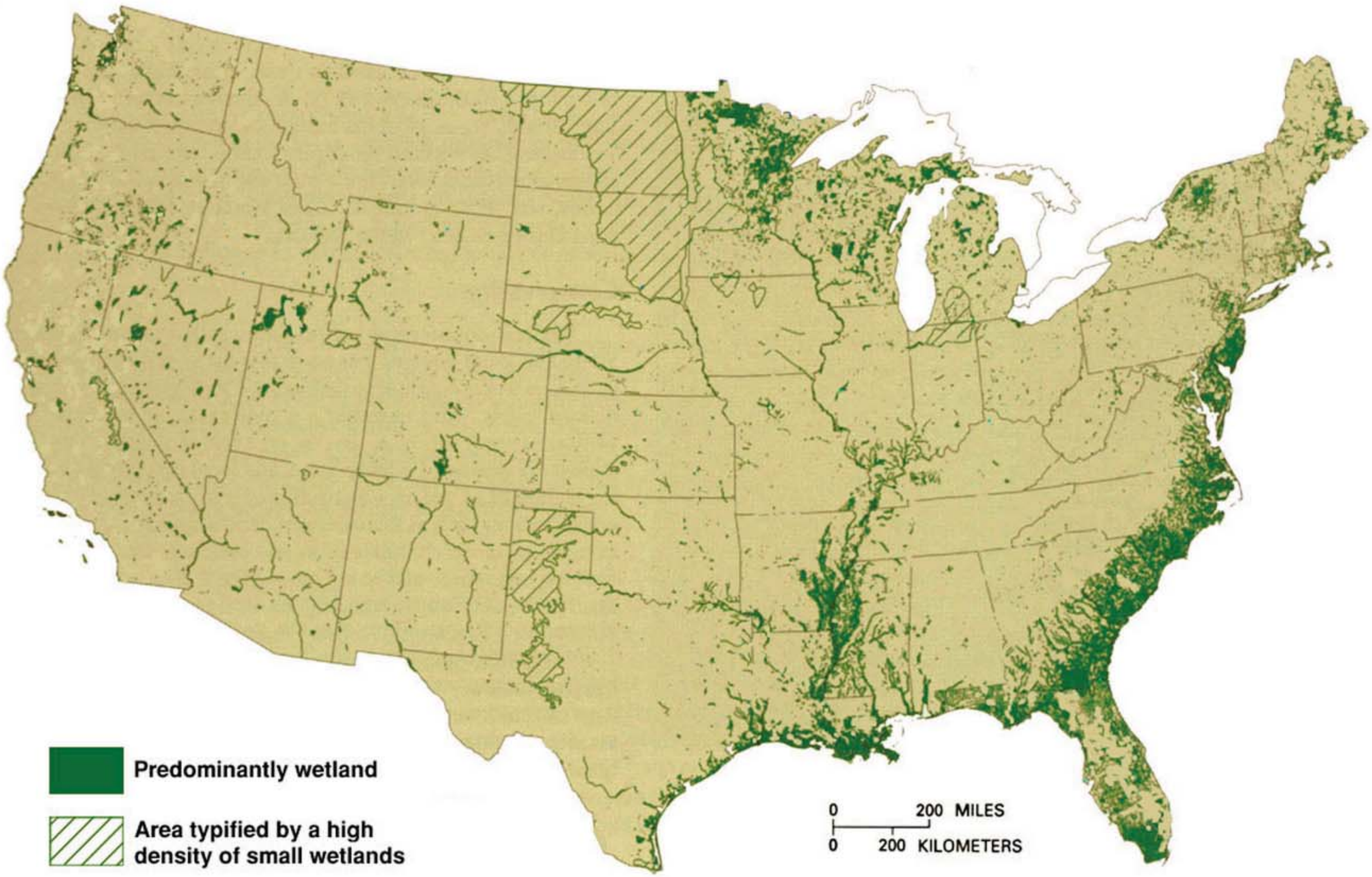

Fig 17.1—Distribution of wetlands in the United States. Source: Dahl and Allord 1997.

fens, marshes, swamps > intertidal wetlands. At the most sensitive end of the spectrum, ombrotrophic bogs receive exogenous nutrients (from external sources) primarily from precipitation, and the species in these ecosystems are adapted to low levels of N (Bridgham et al. 1995, 1996; Shaver and Melillo 1984). Intertidal wetlands are at the other end of this spectrum; in these ecosystems, N loading from marine/estuarine water sources exceed that from atmospheric inputs by one or two orders of magnitude (Morris 1991). Building on these early observations by Morris (1991), it is now generally accepted that nutrient budgets in wetland ecosystems are tightly linked with local hydrology (Bobbink et al. 2003). In general, N deposition is expected to have a larger effect on ecosystems with more closed N cycles, such as bogs, than on those with open N cycles, such as intertidal wetlands. Most data for U.S. wetlands indicates nutrient-related effects, unlike Europe, where acidification effects in wetlands are well documented.

17.3 Range of Responses Observed

Both freshwater and estuarine wetlands tend to be N-limited ecosystems (LeBauer and Tresseder 2008, US EPA 1993). Most studies evaluate short-term fertilization with high N loading that typically exceeds atmospheric deposition. Therefore, it is difficult to assess how long-term N deposition will affect N limitation, especially considering that the degree of N limitation varies among wetlands across the United States (Bedford 1999).

17.3.1 Freshwater Wetlands

Freshwater wetlands harbor large numbers of plant species that have evolved under N-limited conditions. These include quillworts (*Isoetes* spp.), of which three species are federally endangered; insectivorous plants such as the endangered green pitcher (*Sarracenia oreophila*); and the genus *Sphagnum*, of which there are 15 species listed as endangered by some eastern states. An extensive literature survey on temperate North American wetlands found that, in general, plant communities under high nutrient supply had low species richness when compared among different wetland types (Bedford et al. 1999).

Sphagnum moss. The genus *Sphagnum* dominates ombrotrophic bogs and some nutrient poor fens in the northern United States and Canada. The species composition of bogs in both countries is similar, therefore, the Canadian data are included in the following discussion. These mosses efficiently capture atmospheric N deposition with retention rates between 50 and 90 percent; much of the variation is due to the depth of the water table (Aldous 2002a, 2002b). Studies conducted on four species of *Sphagnum* in Maine (2 to 4 kg N $ha^{-1} yr^{-1}$ wet deposition) and New York (10 to 13 kg N $ha^{-1} yr^{-1}$ wet deposition) document that higher wet deposition of N resulted in higher tissue N concentrations and greater net primary productivity (NPP), but lower bulk density (Aldous 2002a; Table 17.1). Note that these results are for wet deposition and not total deposition of N.

A study of *Sphagnum fuscum* in six Canadian peatlands showed a weak, although significant, negative correlation between NPP and modeled total N deposition when deposition levels were greater than 3 kg N $ha^{-1} yr^{-1}$ (Vitt et al. 2003; Table 17.1). A study of 23 ombrotrophic peatlands in Canada with total modeled N deposition ranging from 2.7 to 8.1 kg N $ha^{-1} yr^{-1}$ showed that peat accumulation increases linearly with N deposition. However, in recent years this rate has begun to slow, indicating a limited capacity for N to stimulate accumulation (Moore et al. 2004; Table 17.1).

In summary, documented responses to N deposition by *Sphagnum* mosses in the United States and Canada are limited to changes in N tissue concentration, bulk density, and NPP. There is additional information from European ecosystems that prolonged N deposition may ultimately lead to N saturation, leaching of excess N, and ultimately a shift in community composition. For example, N additions to a boreal bog in Europe resulted in a collapse of the *Sphagnum* mat followed by increased growth of ericaceous shrubs and the sedge, tussock cottongrass (*Eriophorum vaginatum*) (Wiedermann et al. 2007). In more southerly bogs in Europe, excess N permitted an invasion of vascular plants normally held at bay due to low N concentration because of the N-filtering capacity of mosses (Lamers et al. 2000).

Table 17.1—Response to N inputs of biological and chemical indicators for freshwater wetlands

Site	N Input *kg N ha^{-1} yr^{-1}*	Indicator	Responses	Reference
Peatlands:				
23 ombrotrophic peatlands in Canada	2.7 to 8.1	Peat accumulation	Peat accumulation increases linearly with N deposition: $y = 2.84x + 0.67$, $r^2 = 0.32$, $P<0.001$, where x is wet atmospheric N deposition from 1990-1996 (g m^{-2} yr^{-1}), y is 50-year peat N accumulation (g m^{-2} yr^{-1}). However, in recent years this rate has begun to slow, indicating limited capacity for N to stimulate accumulation	Moore et al. 2004
Six Canadian peatlands	3	*Sphagnum fuscum*	At sites with deposition levels ranging from 0.7 to 152 kg N ha^{-1} yr^{-1}, a weak, although significant, negative correlation between NPP and N deposition when deposition levels were greater than 3 kg N ha^{-1} yr^{-1}: $y = 150 - 3.4x$, $R^2=0.01$, $P=0.04$, where x is N deposition rate (kg N ha^{-1} yr^{-1}), y is NPP (g m^{-2} yr^{-1}).	Vitt et al. 2003
Poor fen in NW Ontario, Canada	4.6	*Sphagnum magellanicum*	Growth was stimulated by 4.6 kg N ha^{-1} yr^{-1} as NO_3^--N. However, additional N loading (18 kg N ha^{-1} yr^{-1} as NO_3^--N and 18 kg N ha^{-1} yr^{-1} as NH_4^+-N) did not further stimulate growth	Rochefort et al. 1990
Two bogs, Adirondacks (NY)	10 to 13 wet deposition	*Sphagnum* spp. *Polytrichum commune*	Comparison of low (2 to 4 kg N ha^{-1} yr^{-1}) and high N deposition level (10 to 13 kg N ha^{-1} yr^{-1}) showed elevated tissue N concentrations, greater NPP and lower bulk density under the higher level of deposition	Aldous 2002a
Lindsey Brook Bog, Vanceboro Bog (ME)	2 to 4 wet deposition			
Two bogs, Adirondacks (NY)	10 to 13 wet deposition	*Sphagnum* spp.	Main findings of the ^{15}N tracer experiments are 1) the two Adirondack sites had a greater N% translocation of N to the growing capitula than the Maine sites; 2) 11% of the tracer was retained in Maine and 80% in the Adirondacks; 3) atmospheric deposition is a portion of the *Sphagnum* N budget, the remaining N likely comes from N mineralization	Aldous 2002b
Lindsey Brook Bog, Vanceboro Bog (ME)	2 to 4 wet deposition			
Pitcher plant:				
Hawley Bog (MA) Molly Bog (VT)	6.8	Pitcher plant *(Sarracenia purpurea)*	Based on empirical demographics, an increase above 6.8 kg N ha^{-1} yr^{-1} would cause substantial increase in the extinction risk	Gotelli and Ellison 2002
26 bogs in Massachusetts and Vermont, US	9.76 to 16.42	Pitcher plant; [NH_4] of soil pore water	Relative keel size of *S. purpurea* increased from 0.4 to 0.7 with soil pore water NH_4 concentration increase from 0.1 to 0.5 mg L^{-1}	Ellison and Gotelli 2002
Hawley Bog (MA) Molly Bog (VT)	10 to 14	Pitcher plant	When populations occurring under low deposition (0-1.4 kg N ha^{-1} yr^{-1}) are compared to those at higher deposition (10 to 14 kg N ha^{-1} yr^{-1}), negative population growth rate is observed under higher deposition	Gotelli and Ellison 2006
Molly Bog (VT)	35	Pitcher plant	Relative keel size of *S. purpurea* was compared in two nitrogen addition treatments (0 and 35 kg N ha^{-1} yr^{-1}). Keel size increased from 0.2 to 0.8 under the higher loading	Ellison and Gotelli 2002
Prairie wetland:				
Controlled mesocoms at the University of Minnesota Horticultural Research Center (MN)	0,120, and 480	11 species in sedge meadow community common to prairies	N addition as nitrate-N 1) increased community biomass; 2) decreased the root:shoot ratio from 0.82 at 0 addition to 0.48 at 120 kg N ha^{-1} yr^{-1}, no further reduction was observed at the highest N addition level; 3) decreased community diversity and evenness between the control and highest N addition.	Green and Galatowitsch 2002

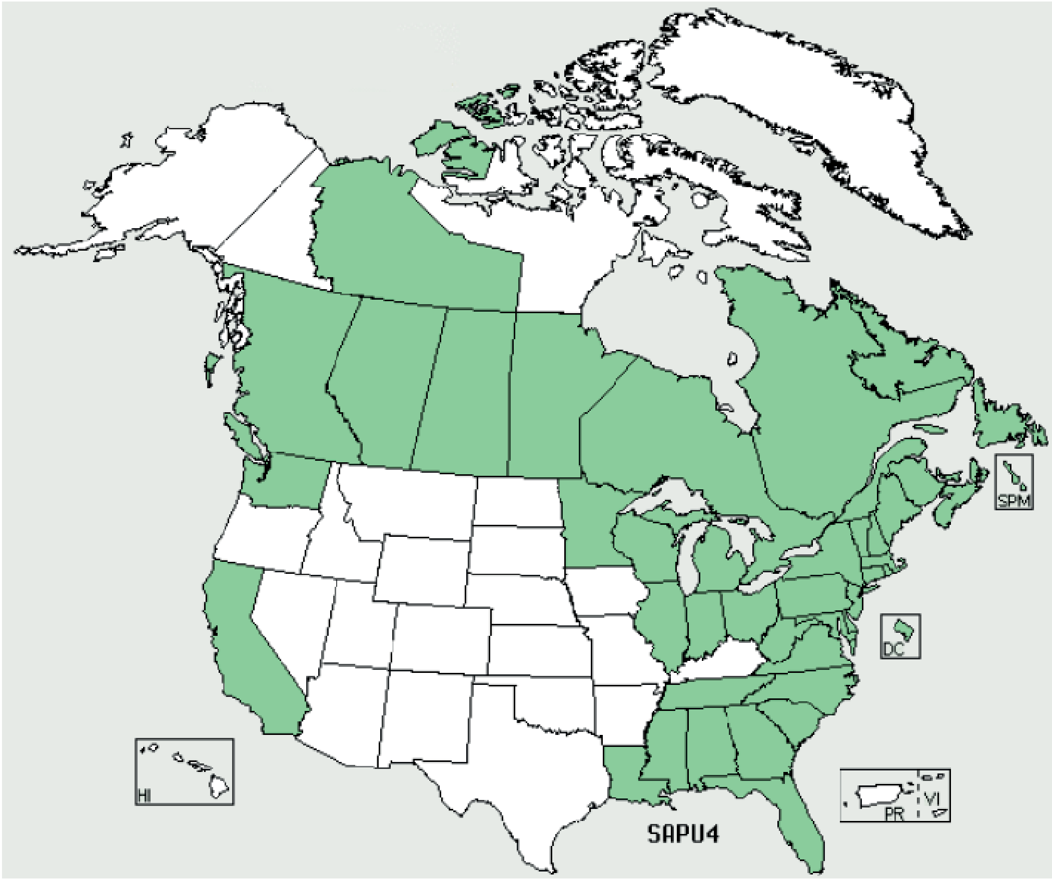

Figure 17.2—Distribution of purple pitcher plant in North America. Green denotes areas where purple pitcher plant is known to occur. Sources: USDA 2009.

Purple pitcher plant. Purple pitcher plant (*Sarracenia purpurea*), also known as northern pitcher plant, is long lived (30 to 50 years) and widely distributed in bogs, fens, and swamps across Canada and the eastern United States (Fig. 17.2; Ellison and Gotelli 2002). Purple pitcher plant has adapted to nutrient poor environments and is very sensitive to increasing N input. Purple pitcher plant was listed as vulnerable in New York, threatened in Florida, Maryland, and Michigan, and endangered in Illinois (USDA 2009). In a study of purple pitcher plant in Vermont and Massachusetts, Ellison and Gotelli (2002) conducted a series of N addition experiments by augmenting N availability to leaves with solution of 0, 0.1, and 1 mg L^{-1} nitrogen as ammonium chloride (NH_4Cl) for one growing season. Population growth rates, estimated by demographic survey, were positive for 0 and 0.1 mg N L^{-1} additions (equal to atmospheric deposition of 0 to 1.4 kg N ha^{-1} yr^{-1})[23] and negative for 1 mg N L^{-1} additions (equivalent to 14 kg N ha^{-1} yr^{-1})[23] (Gotelli and Ellison 2006; Table 17.1). Based on the annual demographic information, a nonstationary matrix model forecasted that the extinction risk within the next 100 years was small if N deposition rate was maintained at 4.5 to 6.8 kg N ha^{-1} yr^{-1}.[23] However, a small annual increase (1 to 4.7 percent) in N deposition rates would substantially increase the extinction risk (Gotelli and Ellison 2002).

Increasing N availability not only reduced population growth of purple pitcher plant, but also dramatically altered plant morphology. Purple pitcher plant produces carnivorous leaves (pitcher) and photosynthesis efficient

[23]N treatments were selected to represent annual N deposition measured at the nearest monitoring sites of National Atmospheric Deposition Program (NADP). The unit of N treatments reported in the publication was precipitation-weighted mean concentrations (mg N L^{-1}), from which we calculated the level of deposition (kg N ha^{-1} yr^{-1}) using the equation: *deposition= precipitation-weighted mean concentrations* × a*nnual precipitation*. More detailed information on N deposition is available on the NADP website: http://nadp.sws.uiuc.edu/sites/ntnmap.asp

leaves (phyllodia). Nitrogen enrichment was shown to stimulate the photosynthesis rate and increase the production of phyllodia relative to pitcher (Ellison and Gotelli 2002). The field N deposition simulation experiment (treatments ranged from 0 to 35 kg N ha^{-1} yr^{-1})[23] revealed a positive linear relationship between N deposition level and relative keel size (keel width/total width, an indication of the relative size of the pitcher tube, in which prey are digested). This correlation was supported by the field surveys of 26 sites across Massachusetts and Vermont (Ellison and Gotelli 2002), and 39 sites across Canada and the eastern United States (Ellison et al. 2004). The relative keel size of northern pitcher plant increased with increasing ammonium (NH_4^+) concentration in soil water, and may be used as a bioindicator ($\log [NH_4^+] = -1.57 + 1.78x$ relative keel size).

Greenhouse Gas Flux. A recent meta-analysis evaluated the effects of N enrichment on biogenic greenhouse gas (GHG) flux from terrestrial ecosystems (Liu and Greaver 2009). In wetlands, N enrichment increased methane (CH_4) emission and nitrous oxide (N_2O) emission, but had no effect on net ecosystem exchange of C or CH_4 uptake. These results are the average response for N addition experiments on wetlands located around the world. Data is limited for the United States; there are just four field addition experiments on freshwater wetlands. Of these, three studies demonstrated a response to N addition. These are summarized in Table 17.2. In the first study, N addition of 20 and 60 kg ha^{-1} yr^{-1} in combination with 6.7 and 20 kg ha^{-1} yr^{-1} phosphorus (P) had no effect on carbon dioxide (CO_2) or CH_4 emissions from a Minnesota peatland fen (Keller et al. 2005). There is evidence that N addition increases N_2O flux from swamp forest in Louisiana and mangrove forest in Florida. However, the lowest N addition level was 100 kg ha^{-1} yr^{-1}, making it difficult to determine if N deposition would also increase N_2O flux (Lindau et al. 1994, Delaune et al. 1998, Whigham et al. 2009). A 5-year addition study indicated that net ecosystem exchange of CO_2 with the atmosphere of bogs in Ottawa, Ontario, was reduced by 45 percent with an addition of 64 kg N ha^{-1} yr^{-1} (Bubier et al. 2007).

17.3.2 Intertidal Wetlands

Nitrogen loading to intertidal wetlands tends to be much higher than loading to freshwater wetlands because, in addition to atmospheric deposition, N sources may include land runoff, ground water, marine water, and waste-water effluent. Nitrogen load to one salt marsh in Narragansett Bay, Rhode Island, was calculated to be 10,253 kg N ha^{-1} yr^{-1} (Wigand et al. 2003), an extremely high value. Compared to N brought in by tidal water and ground water (typically 565 to 668 kg N ha^{-1} yr^{-1}), the N deposited directly to the surface of coastal marshes (~ up to 30 kg N ha^{-1} yr^{-1}) is a small proportion of the total N budget (Morris 1991). However, atmospheric deposition can be the primary N source in some regions. For example, in southern New England estuaries, where N loading (mean =167 kg N ha^{-1} yr^{-1}) was much lower than other U.S. estuaries (mean =893 kg N ha^{-1} yr^{-1}), direct atmospheric deposition to the estuary surface was the largest N source, accounting for 37 percent of the total N loading (Latimer and Charpentier 2010). This information suggests that the impact of direct atmospheric deposition of N may vary among regions. Indirect atmospheric deposition—N deposited to the watershed and transported via surface or ground water—could be another major source of the total N load to coastal marshes. For example, the modeled contribution from the atmosphere (36 million kg N yr^{-1}) was about 21 to 30 percent of the total N loading of 170 million kg N yr^{-1} (Castro et al. 2003, Driscoll et al. 2003) for Chesapeake Bay waters, and 16 percent of the total N loading of 0.029 million kg N yr^{-1} for southern New England estuaries (Latimer and Charpentier 2010) Future studies are needed to determine the role of direct and indirect atmospheric N deposition on the nutrient budget of intertidal wetlands.

Regardless of its source, N enrichment is shown to alter the structure and function of intertidal wetland ecosystems (Table 17.3) by causing increased primary production (Darby and Turner 2008a, Mendelssohn 1979, Tyler et al. 2007, Wigand et al. 2003); invasion of nonnative species (Tyler et al. 2007); altered competition between native species (Crain 2007, Mendelssohn 1979, Wigand et al. 2003); loss of

Table 17.2—Greenhouse gas flux in response to N enrichment for freshwater wetlands.

Site	N Input *kg N ha^{-1} yr^{-1}*	Indicator	Responses	Reference
Mer Bleue Bog, Ottawa, ON	64 +PK	NEE (net ecosystem exchange of carbon)	After 5 years nutrient treatment, the rate of maximum NEE was significantly reduced (45%) under the highest levels of nutrient addition (64 kg N ha^{-1} yr^{-1} +5.0 kg P ha^{-1} yr^{-1} + 63 kg K ha^{-1} yr^{-1}), but was not changed under other nutrient levels, compared to control	Bubier et al. 2007
Swamp forest, St. James Parish, LA	100	N_2O emission	Addition of NO_3 and NH_4 to the columns increased N_2O production 56% and 15% over the control, respectively	Delaune et al. 1998
Swamp forest, Spring Bayou Wildlife Management Area, LA	100, 300 NH^{4+} - N; 100, 300 NO^{3-} -N	N_2O emission	N_2O emission was largely stimulated by NO_3^-addition but not NH_4^+; Total N_2O evolution was estimated to be 0.44,0.36,2.83 and 10.45 kg ha^{-1} yr^{-1} for the 100 NH_4^+, 300 NH4+, 100 NO_3- and 300 kg NO_3-N ha $^{-1}$ yr^{-1} treatments, respectively	Lindau et al. 1994
Mangroves forest, Indian River Lagoon, FL	100	N_2O emission	Averaged across the three experimental years (2001 -2003), N_2O emission was almost six times higher in fertilized plots (0.028 mg N_2O-N m^2 hr^{-1}), compared to that in control plots (0.005 mg N_2O- N m^2 hr^{-1})	Whigham et al. 2009

sensitive species (Latimer and Rego 2010); increased herbivore damage to plants (Bertness et al. 2008); stimulated evapotranspiration (Howes et al. 1986); altered microbial community and pore water chemistry (Caffrey et al. 2007); and altered carbon allocation between roots and shoots (Darby and Turner 2008b). It is unclear if these effects could occur in response to current levels of deposition alone because most studies apply N additions that are several orders of magnitude greater than atmospheric deposition.

Spartina. *Spartina* is an aggressive perennial cordgrass native to the coasts of the Atlantic Ocean. The dense root mats of *Spartina* trap sediment, elevating the substrate, and creating new shoreline. Nitrogen enrichment has been shown to alter the ecophysiological processes and community composition of *Spartina* salt marshes. This work is based on N addition levels that simulate total loading, not deposition alone. Evidence from Mendelssohn et al. (1979) indicates 280 kg N ha^{-1} yr $^{-1}$ alters growth and biomass of *Spartina*. Howes et al. (1986) found N addition (416 kg N ha^{-1} yr $^{-1}$) increased live aboveground biomass, leaf area coverage, and evapotranspiration of smooth cordgrass (*S. alterniflora).* Under N-enriched condition, saltmeadow cordgrass (*S. patens)* is often out-competed and replaced by smooth cordgrass (Emery et al. 2001). Wigand et al. (2003) examined plant community structure of 10 salt marshes over a 2-year period, with N ranging from 2 to 10,253 kg N ha^{-1} yr $^{-1}$, in Narragansett Bay. Several dose response curves based on their findings are given in Figure 17.3. Species richness decreased linearly with increasing N loading. With N loading higher than 63 to 400 kg N ha^{-1} yr $^{-1}$, the density of saltmeadow cordgrass and short smooth cordgrass rapidly decreased and the density of tall smooth cordgrass increased.

Eelgrass. Eelgrass, genus *Zostera*, is submerged aquatic vegetation that provides important habitat for many wildlife species, such as fish, shellfish, sea turtles, and waterfowl. Eelgrass beds are also important for sediment deposition and substrate stabilization. The massive decline of eelgrass has been widely documented in estuaries across Atlantic and Pacific coast (Latimer and Rego 2010, Ward et al. 2003). Excess N loading is one of the primary disturbances causing the rapid loss of eelgrass habitat. Latimer and Rego (2010) estimated the relationship between eelgrass extent and predicted watershed-derived N loading for 62 shallow estuaries along the coasts of Connecticut, Rhode Island, and Massachusetts. They found that N loading greater than 50 kg N ha^{-1} yr^{-1} lead to a significant decline of eelgrass coverage, and almost all eelgrass habitats were lost at loading levels higher than 100 kg N ha^{-1} yr^{-1}.

Table 17.3—Responses of coastal marsh ecosystems to N fertilization. Note an asterisk (*) indicates that N addition units were converted to kg N ha^{-1} yr^{-1} when different units were given in the publication.

Site	N Input	Indicator	Responses	Reference
Walden Creek, NC	0, 280, 560, and 1120 kg N ha^{-1} yr^{-1}	Smooth cordgrass (*Spartina alterniflora*) (short and tall)	1) increased the growth of short smooth cordgrass, but had no effect on tall smooth cordgrass; 2) NH_4^+ showed higher growth stimulation on short smooth cordgrass than NO_3^- 3) biomass production of short smooth cordgrass increased linearly with N addition	Mendelssohn et al. 1979
Great Sippewissett Marsh, MA	26 and 76 kg N $ha^{-1}wk^{-1}$ from June to Sept (~416 and 1216 kg N ha^{-1} yr^{-1}*)	Smooth cordgrass	N addition 1)increased live aboveground biomass 100 and 270% in the low and high addition, respectively; 2) increased leaf area coverage; 3) doubled the rate of evapotranspiration	Howes et al. 1986
San Diego Bay, CA	15 g N m^{-2}·wk^{-1} from March to Aug 1995 and 1996 (~3600 kg N ha^{-1} yr^{-1}*)		N addition 1) increased plant tissue N concentration; 2) did not increase soil N concentration	Boyer et al. 2000
Narragansett Bay, RI	N gradient from 2 to 10,253 kg N ha^{-1} yr^{-1}	Saltmeadow cordgrass (*Spartina patens*); smooth cordgrass (short and tall)	N addition 1) reduced species richness; 2) decreased the density and extent of saltmeadow cordgrass; 3) increased the density and extent of tall smooth cordgrass	Wigand et al. 2003
Tomales Bay, CA	N addition of 15 g N m^{-2} urea fertilizer every two weeks from March 2000-July 2001 (~2100 to 3000 kg N ha^{-1} yr^{-1}*)	Saltgrass (*Distichlis spicata*), Virginia glasswort (*Salicornia virginica*), marsh rosemary (*Limonium californicum*), Canadian sandspurry (*Spergularia caadensis*), arrowgrass (*Triglochin sp*).	N addition 1) increased soil nitrogen; 2)increased height, plant tissue N concentration in all species; 3)increased biomass *of* saltgrass, marsh rosemary, Canadian sandspurry, arrowgrass; 4) increased percent cover of all species at one or more times during the experiment	Traut 2005

San Francisco Bay (CA); Willapa Bay (WA)	Added 800 kg N ha^{-1} yr^{-1}	Smooth cordgrass; Hybrid *Spartina* (*S. alterniflora* x *S. foliosa*); Virginia glasswort	1) increased biomass, stem density and the rate of habitat invasion; 2) promoted the spread of invaded species (hybrid *Spartina*); 3) decreased the density of native species (Virginia glasswort)	Tyler et al. 2007
Nags Creek (RI)	2 g N m^{-2} added once per month from May-Aug for 3 years (~80 kg N ha^{-1} yr^{-1}*)	Saltmeadow cordgrass	N addition 1) increased bacterial production; 2) increased pore water NH_4^+ and NO_3^- concentrations; 3) increased the uptake of dissolved inorganic phosphorous; 4) inconclusive effects on mineralization and denitrification rates	Caffrey et al. 2007
Cocodrie (LA)	N additions made once per month from April- Aug 2004 (~0, 230, 465, 930, 1860 and 3720 kg N ha^{-1} yr^{-1}*)	Smooth cordgrass	N addition 1) increased the aboveground biomass and stem density; 2) did not change live belowground biomass	Darby and Turner 2008b
York River (ME); Nonesuch River (ME)	N addition of 1630 kg N ha^{-1} yr^{-1} for 3 years	Salt, brackish and oligohaline marshes	N addition 1) increased the productivity of all three types of marshes by 100% after 3 years; 2) lowered C:N and increased N:P in plant tissue in all marsh-types; 3) had no significant effect on species composition of salt and brackish marshes, however a combination of N and P fertilization changed the species composition of oligohaline marshes	Crain 2007
Narragansett Bay (RI)	Added 30% N by weight of 60 g m^{-2} per month from May to August 2005 and 2006 (~720 kg N ha^{-1} yr^{-1}*)	Smooth cordgrass	N was added with phosphorous and potassium. The herbivore consumption of primary production was small in pristine marshes, but increased to 50-75% in eutrophic marshes	Bertness et al. 2008
62 New England estuaries	N gradient from 24.2 to 928 kg N ha^{-1} yr^{-1}	Eelgrass	N loading greater than 50 kg N ha^{-1} yr^{-1} led to a significant decline of eelgrass coverage, and almost all eelgrass habitats were lost at loading levels higher than 100 kg N ha^{-1} yr^{-1}.	Latimer and Rego 2010

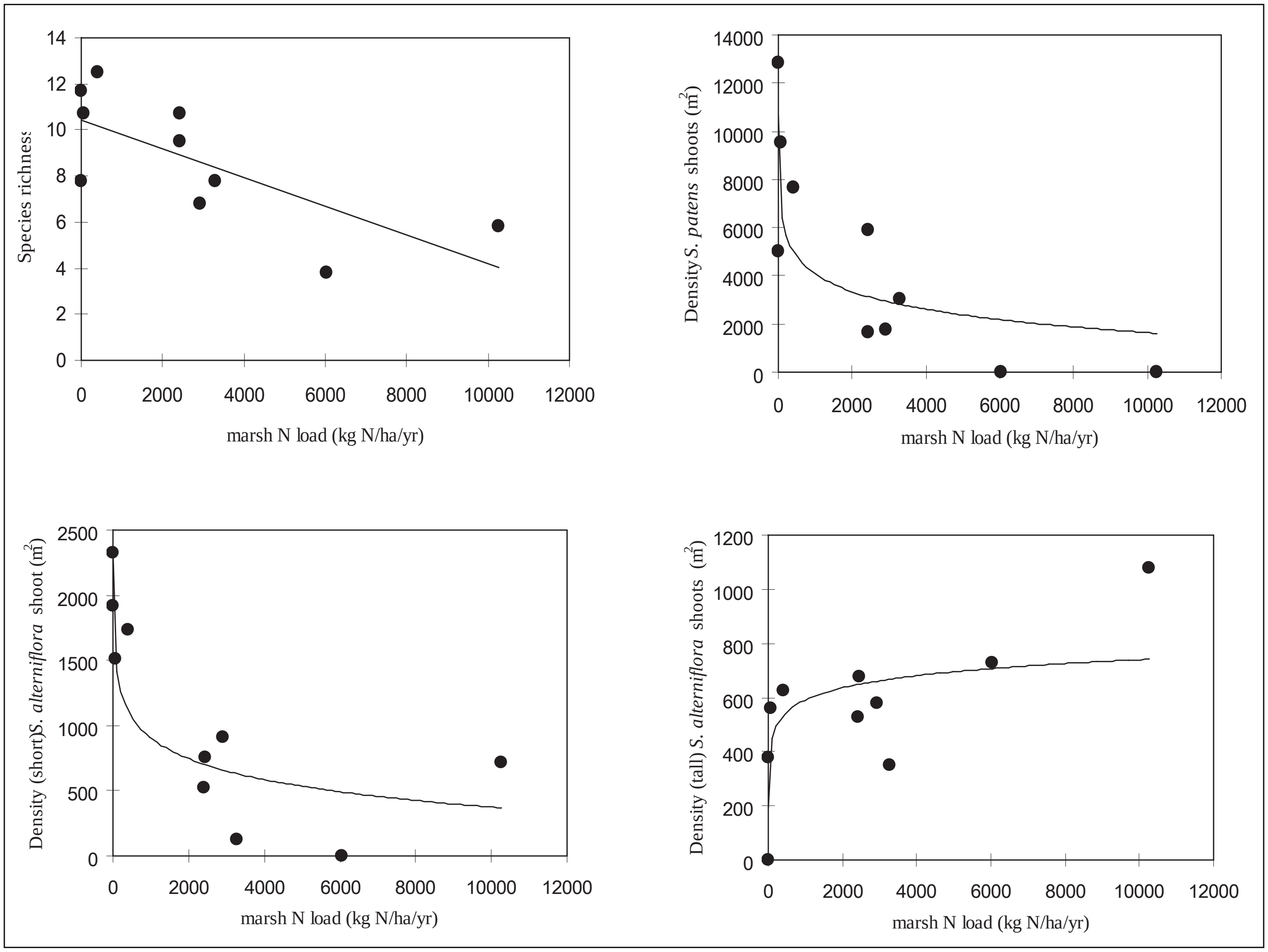

Figure 17.3—The effect of N loading on plant community structure of 10 salt marshes in Narragansett Bay (graphs plotted from data presented in Wigand et al. 2003)

17.4 Critical load estimates

17.4.1 Freshwater Wetlands

There is correlative evidence that *Sphagnum*-dominated freshwater wetlands in the United States are altered by N deposition. The critical load for altered peat accumulation and NPP is between 2.7 and 13 kg N ha^{-1} yr^{-1}, based on observations from Aldous (2002a), Moore et al. (2004), Rochefort et al. (1990) and Vitt et al. (2003). The upper end of this critical load range is based on measurements of wet deposition only (10 to 13 kg N ha^{-1} yr^{-1}) (Aldous 2002a, 2002b) and therefore does not reflect total N loading. There is evidence showing that N deposition alters both the morphology and population dynamics of purple pitcher plant. The empirical evidence suggests a critical load to protect the population of purple pitchers of 10 to 14 kg N ha^{-1} yr^{-1} (Gotelli and Ellison 2006), while matrix modeling to forecast long-term population sustainability based on observations of population demographics suggests a lower value of 6.8 kg N ha^{-1} yr^{-1} (Gotelli and Ellison 2002).

17.4.2 Intertidal Wetlands

The critical load range for atmospheric deposition is difficult to establish for intertidal wetlands because they have open nutrient cycles which are often strongly affected by N loading sources other than atmospheric deposition. Typically, the amount of N added in experimental treatments simulates total N input and therefore far exceeds the amount that U.S. intertidal wetlands would receive by atmospheric deposition. Only two studies (Table 17.3) have addition levels below 100 kg N ha^{-1} yr^{-1}. Based on the results of Wigand et al. (2003), a critical load to protect the community structure of salt marshes is likely to be 63 to 400 kg

Table 17.4—European empirical critical loads of nutrient N for mire, bog and fen habitats (EUNIS class D). Reliability rating: ## reliable, # quite reliable and (#) expert judgment.

Ecosystem type	kg N ha^{-1} yr^{-1}	Reliability	Indication of exceedance
Raised bogs and blanket fens	5-10	##	Increased vascular plants, decrease bryophytes, altered growth and species composition of mosses, increased N in peat and peat water
Poor fens	10-20	#	Increases in sedges and vascular plants, negative effects on peat mosses
Rich fens	15-35	(#)	Increases on tall graminoids, decrease diversity
Montane rich fens	15-25	(#)	Increase vascular plants, decrease bryophytes

From Bobbink et al. 2003; Table 6.1

N ha^{-1} yr^{-1}. Caffrey et al. (2007) provide additional evidence that 80 kg N ha^{-1} yr^{-1} alters microbial activity and biogeochemistry. Latimer and Rego (2010) found that eelgrass coverage started to decrease rapidly at N loading higher than 50 kg N ha^{-1} yr^{-1}, with no eelgrass at loading levels higher than 100 kg N ha^{-1} yr^{-1}. Note that these values are the total N loading to salt marshes, including N deposition directly to the marsh surface, as well as N deposited indirectly to the watershed, surface or ground water, and runoff from agriculture, urban areas, and other sources. Additional experimental evidence on ecosystem response to N loads that are similar to the amount of loading due to N deposition is needed to improve the critical load calculation for intertidal wetlands in the United States.

17.5 Comparison to Critical Loads from Europe

The 2003 assessment of N critical loads in Europe designated critical loads for multiple types of wetlands, including raised and blanket bogs, poor fens, rich fens, mountain rich fens, and intertidal wetlands (Bobbink et al. 2003). There are numerous publications on N effects to wetlands in Europe compared to the United States. In general, documented responses include effects on growth and species composition, competition between species, peat and peat water chemistry, decomposition, and nutrient cycling. A brief summary of the European critical loads for wetlands is presented here and summarized in Table 17.4.

Bobbink et al. (2003) assigned a critical load of 5 to 10 kg N ha^{-1} yr^{-1} for bog ecosystems, based on plant community and species responses to N deposition, and indicated that precipitation and P limitation should be used to assign critical loads to individual sites. The observed changes in the plant communities of ombrotrophic bogs included the replacement of *Sphagnum*-forming species with nitrophilous moss species (20 to 40 kg N ha^{-1} yr^{-1} in Dutch bogs) (Greven 1992); the absence of characteristic *Sphagnum* species in British bogs (30 kg N ha^{-1} yr^{-1}) (Lee and Studholme 1992); and reduction in the growth survivorship of characteristic bog species roundleaf sundew (*Drosera rotundifolia*) (10 kg N ha^{-1} yr^{-1} in Swedish bogs) (Bobbink et al 2003, Redbo-Torstensson 1994). The European critical load is similar to the range of critical loads suggested for freshwater wetlands in the United States (Table 17.5).

It is unclear how European critical loads for poor fens, rich fens, and mountain rich fens relate to the critical loads for similar wetlands in North America because the impacts of N loads have not been studied in the United States. European poor fens have a critical load of 10 to 20 kg N ha^{-1} yr^{-1} based on increased sedge and vascular plants and negative effects on peat mosses. The critical load for rich fens in Europe is 15 to 35 kg N ha^{-1} yr^{-1} based on increased tall graminoids and decreased diversity. The critical load for montane rich fens in Europe was 15 to 25 kg N ha^{-1} yr^{-1} based on increased vascular plants and decreased bryophytes. Note that changes in the vegetation composition and structure likely affect fauna species assemblages, such as spiders and beetles living in the originally open bog vegetation, and ground-breeding birds. Increased nutrient

Table 17.5—Empirical critical loads of nutrient N for wetlands. Reliability rating: ## reliable; # fairly reliable; (#) expert judgment

Ecosystem	Critical loads for N deposition *kg N ha^{-1} yr^{-1}*	Reliability	Response	Study
Freshwater wetlands	2.7-13	#	Peat accumulation and NPP	Aldous 2002a, Moore et al. 2004, Rochefort et al. 1990, Vitt et al 2003
Freshwater wetlands	6.8-14	(#)	Pitcher plant community change	Gotelli and Ellison 2002, 2006
Intertidal salt marsh	63-400	(#)	Salt marsh community structure, microbial activity and biogeochemistry	Caffrey et al. 2007, Wigand et al. 2003
Intertidal wetlands	50-100	##	Loss of eelgrass	Latimer and Rego 2010

availability results in an increase of the nutrient content of plant material (Limpens et al. 2003a, Tomassen et al. 2003) and algal growth (Gulati and DeMott 1997, Limpens et al. 2003b), and affects herbivorous, detritivorous, and carnivorous invertebrates (Van Duinen et al. 2004).

The European critical load for salt marshes, based on expert judgment, is 30 to 40 kg N ha^{-1} yr^{-1} (Bobbink et al. 2003), but studies of European salt marshes are limited. High levels of N input (65 to 70 kg N ha^{-1} yr^{-1}) significantly increased biomass production in the Netherlands (Van Wijnen and Bakker 1999); no changes in species composition and in diversity have been observed for the current deposition of 15 to 25 kg N ha^{-1} yr^{-1} at sites in the Netherlands and Germany (Bobbink et al. 2003). The critical load for North American intertidal ecosystems may be closer to these values than to the high levels of N input previously studied (section 17.3.2).

17.6 Future Research Directions and Gaps in Data

An great diversity of wetlands ecosystems are found across Northern America, but experimental studies on the impacts of N enrichment in these wetland systems are scarce. Some data are present for bog systems and intertidal wetland (salt marshes), but even in these ecosystems field additions studies with realistic N loads are rare. Realistic levels of long-term (5 to 10 years) experimental N addition in low background regions, with a high resolution of treatments, are needed to refine and establish reliable critical loads for wetlands. In addition, these studies should be done across a range of wetlands types in the United States, especially in oligotrophic to mesotrophic wetland habitat.

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