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CUMULATIVE EFFECTS ON WETLAND LANDSCAPES: LINKS TO WETLAND RESTORATION IN THE UNITED STATES AND SOUTHERN CANADA

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Abstract: The cumulative effects of human actions on wetland ecosystems motivate current efforts at wetland restoration. They also have created in part the context within which restorations are undertaken. Using modern hydrogeological understanding of wetland-landscape linkages, I argue that restorations should begin with a cumulative impact analysis for the entire region in which the restoration is proposed. The analysis, however, should not focus merely on number of hectares of wetlands lost or degraded. It should be based on the concept of templates for wetland development. These templates are the diversity of settings created in specific landscapes by the complex interactions of hydrogeologic factors and climate. They control key hydrologic variables and hydrologically influenced chemical variables that cause specific wetland types to form and to be maintained through time. They also determine in large part the biogeochemical cycling characteristics specific to different types of wetlands. They thus account for both the biological and functional diversity of wetlands. A cumulative impact assessment for restoration purposes should identify the kinds, numbers, relative abundances, and spatial distribution of wetland templates in a region—both past and present. These past and present profiles of the wetland landscape can be used to make decisions regarding the type and location of restorations. Matching type and location to the appropriate hydrogeologic setting will maximize the probability of success for individual projects. Regional wetland diversity can be restored if individual restoration decisions about wetland type and location are made in light of the diversity of templates in past and present regional profiles.

Key Words: biological diversity, cumulative effects, functional diversity, hydrology, hydrogeologic classification, hydrogeomorphic classification, landscape assessment, restoration, wetlands

INTRODUCTION

The term "cumulative effects" refers to a critical concept within the lexicon of environmental regulation and impact assessment. It attempts to capture what we see all around us but fail to regulate or predict: the collective result through time of the numerous and varied human-caused impacts on the environment. Even though individual human actions may have only minor effects, the concept of cumulative effects recognizes that the net effect of past, present, and future actions may be significant (Beanlands et al. 1986, Preston and Bedford 1988). Thus, the essential difference between conventional and cumulative impact assessment (CIA) lies in the manner in which spatial and temporal boundaries of the evaluation are established. The boundaries drawn in CIA are larger with respect to the number of disturbances, the geographic area, and the time frame considered. Without attention to this larger framework, wetland restoration risks reducing the biological and functional diversity of wetland landscapes.

Cumulative effects have led to several landscape-level patterns that both motivate and constrain efforts to restore wetlands. They have resulted in significant loss of wetland area, disproportionate loss of some wetland types, degradation of remaining wetlands, and a consequent decrease in the diversity of native wetland types and species. These factors motivate restoration efforts. However, cumulative effects also have altered landscapes in which wetland restoration occurs and have thus constrained opportunities for wetland restoration. Wetland restoration has the potential to increase wetland diversity as well as wetland area but only if those planning restorations do so within the context of a cumulative effects analysis that considers landscape patterns of loss and degradation and landscape controls of wetland development. A cumulative effects analysis draws attention to wetlands as landscape elements.

In this paper, I briefly review landscape patterns of wetland loss and degradation for the conterminous United States, the prairie regions of western Canada, and the more densely-populated temperate regions of Canada in southern Ontario and Quebec along the Great Lakes and St. Lawrence River. I then describe the main elements of a conceptual framework that explicitly treats wetlands as landscape elements, in terms of both the landscape factors that control characteristics of individual wetlands and the diversity of wetland types within landscapes. I conclude by presenting the rudiments of a cumulative effects analysis of wetland landscapes that might be undertaken as the basis for planning restorations within specific regions of the United States and temperate Canada.

LANDSCAPE PATTERNS OF LOSS, DEGRADATION, AND RESTORATION

Wetland Loss

The primary motivation for current restoration efforts is the significant loss of wetland area that has occurred through the cumulative effects of thousands of individual decisions to alter wetlands. Estimates of wetland loss in the United States total about 53% of the area that was in wetland at the time of European settlement of the lower 48 states, or about 24.3 ha of wetland lost for every minute since 1780 (Tiner 1984, Dahl 1990, Dahl et al. 1991, Dahl and Allord 1996). Although loss rates have slowed according to recent estimates, annual net losses on non-federal lands totaled 321,003 hectares between 1982 and 1992, or 28,350 to 36,450 hectares a year (Heimlich and Melanson 1995).

Wetland conversions to other land uses have been minimal in the vast boreal, subarctic, and arctic regions of Canada, but the overall picture for the prairie regions and the more densely populated temperate regions along the Great Lakes and St. Lawrence River in southern Canada is similar to that for the U.S. (i.e., 50–55% overall loss of wetland area). Glooschenko et al. (1993) noted major losses in the 3 prairie provinces (Alberta, Saskatchewan, and Manitoba) to draining, filling, and cultivation for agriculture. No overall studies have been completed for the entire prairie region, but they cite Schick's (1972) study, which found that only 39% of the pre-European settlement area of wetland remained in the Alberta prairie parkland region. Rakowski and Chabot (1983, as cited in Rubec et al. 1988) reported a decrease of 71% in wetland area between 1928 and 1982 in the 131 km^2 Minnedosa pothole region of Manitoba. Glooschenko et al. (1993) also noted major wetland losses in southern Ontario, which has been well-studied. Snell (1982, as cited in Glooschenko et al. 1993) estimated that 70% (over 1 million ha) of the pre-European settlement wetland area in 38 counties in southern Ontario had been lost by the late 1960s, largely to agriculture. Between 81 and 98% of wetlands in the three most southwestern Ontario counties have been lost. Whillans (1982) reported 75–100% loss of marshes along Lake Ontario near Hamilton and Toronto since settlement.

The pattern reported for southern Ontario also holds for all Canadian urban-centered regions, except Atlantic Canada (Rubec et al. 1988). For Pacific, western, and the eastern temperate regions of Canada, wetland loss ranges from 30 to 98%, with most urban-centered regions showing losses in excess of 75% (Rubec et al. 1988). Rubec et al. (1988) provide a summary of available estimates of wetland land-use conversions in Canada.

Degradation of Remaining Wetlands

Numerous individual human actions in and near wetlands also have had the cumulative effect of degrading many remaining wetlands, especially those in agricultural regions, along the shores of the Great Lakes, and in urban areas. Extreme examples have and will become the object of restoration. The form and extent of degradation will impose constraints on restoration efforts.

Degradation comes in many forms and degrees of severity. Although evident to all serious observers of wetlands, degradation is far more difficult to quantify than loss of area. Documented examples of typical forms of degradation include (a) the fragmentation, partial diking and filling, and industrial development within the River Raisin drowned river mouth on Lake Erie (Herdendorf 1987), New York harbor, and other ports (Pinder and Witherick 1990); (b) several different toxic substances in sediments of Great Lakes wetlands; (c) extensive invasion by exotic species (Malecki et al. 1993, Doren and Jones 1997); (d) pesticides in prairie wetlands (Neely and Baker 1989); (e) nutrient enrichment of previously nutrient-poor wetlands and subsequent loss of species richness and/or endemic species (Ehrenfeld 1983, Ehrenfeld and Schneider 1991, Morris 1991); (f) decrease in average area of individual wetlands and loss of connectivity among wetlands and other ecosystems in southern U.S. bottomland hardwood swamps (Lee and Gosselink 1988, Gosselink et al. 1990); and (g) loss of marsh-edge vegetation to cultivation in the prairie provinces. For example, Millar (1981, as cited in Glooschenko et al. 1993) reported that by 1979, 84% of the wetlands on sampling transects in the prairie provinces of Canada had been altered by human activities.

Alteration of Wetland Watersheds and Landscapes

Aside from the economic and political constraints that often dominate restoration projects, the greatest constraint on restoration efforts will come from cumulative alteration of landscapes within which wetland restoration sites occur. The reasons for this are obvious—wetlands are functionally interdependent with other landscape units (Bedford and Preston 1988). Properties of the overall landscape, such as land-surface slope and topography, the distribution of glacial deposits, percent area in wetlands, and spatial configuration of different land-use types, determine the kinds and characteristics of inputs to wetlands. The quantity, chemistry, flow rates, and timing of water inputs, the types and quantities of sediments, nutrients, and pollutants entering the wetland, as well as the plant and animal species that move or disperse into the wetland, all reflect landscape characteristics.

Landscape alterations particularly likely to influence restoration efforts include (a) hydrologic modifications within the watershed, such as dams, dikes, levees, drainage ditches, and channelization of water courses; (b) land-use changes; and (c) changes in the spatial distribution of wetlands with respect to other ecosystems and land uses in the landscape. Land-use changes, such as conversion of forest to agriculture and agriculture to residential or commercial, affect the delivery of water, nutrients, sediments, and pollutants to wetlands as to other water bodies (e.g., Poiani et al. 1996). Several studies have reported high correlations between nutrient loading to downstream waters and the proportion of agriculture in the watershed (e.g., Dillon and Kirschner 1975, Hill 1978, Beaulac and Reckhow 1982, Jordan et al. 1997a, 1997b).

The spatial configuration of wetlands in many areas has been altered in a number of ways that will affect restoration efforts. The number of wetlands in an area and the distance from each other affect the dispersal of plant propagules and animals to the site. The connectivity of wetland types to each other and other ecosystem types influences animal use (Harris 1988) and movement of water and solutes. Location of wetlands within a watershed, especially relative to stream order, determines wetland effects on water quality and flood attenuation (Brinson 1993a). Whigham et al. (1988) and Correll et al. (1992) have shown how the spatial mosaic of wetlands and other communities in the Rhode River Estuary affect the delivery of nutrients to the stream. Johnston et al. (1990) showed how the position of wetlands in the watershed and their proportion of the landscape cumulatively affect stream water quantity and quality.

Non-random Cumulative Effects

The pattern produced by cumulative loss of wetlands is not random with respect to wetland type. Some types of wetlands have suffered greater losses than others. For example, species-rich sedge meadows, wet prairies, and other palustrine wetlands saturated or flooded for only a part of the growing season were easily and quickly drained for agriculture. By the mid-1950s, few of these types remained in central Wisconsin (Curtis 1959) and probably elsewhere in the midwestern United States (Prince 1997) and the prairie provinces of Canada. In coastal California, shallow-water habitats (salt marsh and intertidal flats) have decreased by 85% while subtidal areas have shown little net change since about 1856 (Macdonald 1990, as cited in Zedler 1996).

A pattern of disproportional loss also is evident for forested wetlands in the United States (Johnston 1994). Between the 1950s and 1970s, forested wetlands constituted 54% of all wetland losses in the U.S. Between the mid-1970s and mid-1980s, they accounted for 95% of all losses (Dahl et al. 1991). The cessation of fires previously set by American Indians (Jean and Bouchard 1991), stabilization of water levels on two of the Great Lakes and elsewhere (Keddy and Reznicek 1985), ground-water withdrawals in arid regions with riparian wetlands (Stromberg et al. 1996), and the practice of letting cattle graze wetlands along streams all select against some specific types of wetlands. The net effect has been a shift in the relative proportion of different types of wetlands in the landscape.

Non-random Restoration

The net result of selective wetland loss and degradation, favoring persistence of some types and decline of others, has been to homogenize the landscape with respect to wetland diversity, providing a second motivation for wetland restoration. Wetland restoration could increase landscape heterogeneity by restoring lost types. In practice, however, it contributes to landscape homogenization because restoration projects favor some wetland types over others (Kusler and Kentula 1990). Several evaluations of wetlands restored or created as mitigation for wetland losses permitted under Section 404 of the U.S. Clean Water Act found that emergent marshes and open water habitat were preferentially created, even in regions where the type was not common (Kentula et al. 1992). Dahl and others (1991) reported that the only wetland type increasing in the U.S. was "ponds." One of the types most often created, freshwater emergent marshes, especially those dominated by cattails (*Typha* spp.), is also the type toward which human disturbance of wetlands often tends to move wetlands. The pattern is so common in Canada that Keddy (1990) has presented a model, which he calls centrifugal, with a diversity of wetland types all being driven by various human disturbances toward a central single type, the cattail marsh.

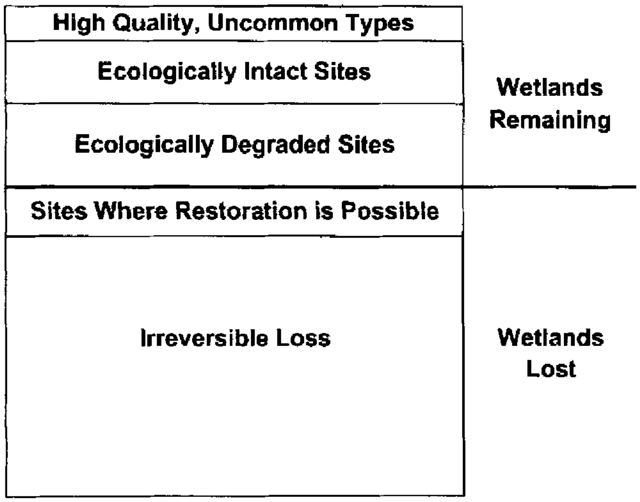

Figure 1. Conceptual representation of the status of the total wetland resource of the United States and southern Canada as affected by loss and degradation. The entire box represents the total wetland resource (i.e., the area in wetland at the time of European settlement). This resource is then divided into areas representing approximately the proportion lost and remaining. The estimate of proportion lost is based on Dahl et al. (1991) for the conterminous United States and on two summaries (Rubec et al. 1988, Glooschenko et al. 1993, see text) of available estimates of wetland conversion in the agricultural and more heavily-populated areas of southern Canada: the prairie region of southern Alberta, Manitoba, and Saskatchewan, the temperate region along the Great Lakes and St. Lawrence River of southern Ontario and Quebec, and all Canadian urban-centered regions except those in Atlantic Canada. Remaining wetlands then are divided further into areas representing approximately the proportion of high quality, uncommon types of wetlands, ecologically intact wetlands, and ecologically degraded wetlands. Wetlands lost are divided into irreversible loss and wetland sites where restoration is still possible. These latter two divisions are only rough approximations and are not based on systematic estimates; see text under sub-heading "degradation of remaining wetlands."

These cumulative effects on the total wetland resource (Figure 1) both motivate and constrain restoration efforts. They also should inform our decisions about the types and locations of the wetlands we attempt to restore. A large proportion of wetland area has been lost. Of this area, some is irreversible loss (e.g., areas occupied by cities, airports, and harbors; areas where extensive drainage has lowered groundwater levels), while some can be restored (e.g., former agricultural wetlands). Of what remains as wetland, most is degraded in some way. Only a relatively small portion consists of high quality sites, uncommon types, or ecologically intact sites. These should be protected and maintained. Decisions about restoration lie in the region of the chart marked "reversible loss." The remainder of this paper proposes a framework for choosing among those sites in restoration decisions.

TEMPLATES FOR WETLAND RESTORATION

Wetlands can be restored as self-maintaining ecosystems only if they are properly placed in the landscape. Because the landscape in which a wetland sits mediates delivery of water, mineral ions, plant growth-limiting nutrients, and sediments, proper placement means placement with respect to the different combinations of climatic and hydrogeologic factors that control these inputs (i.e., hydrogeologic setting) (Winter 1988, 1992). The concept of hydrogeologic setting captures landscape characteristics that control development and maintenance of wetlands. A closely related precursor to this concept is the idea of templates for wetland development (Moore and Bellamy 1974). Bedford (1996) previously used this concept as the basis for defining hydrologic equivalence in wetland mitigation. Here I apply the concept to wetland restoration and illustrate the concept with examples of wetland-landscape linkages that show the critical role these linkages play in determining both the type of wetland that develops and how the wetland functions biogeochemically.

Hydrogeologic and Hydrogeomorphic Frameworks

Scientific thinking about what controls development and maintenance of wetlands has evolved profoundly in the last two to three decades. It has moved from a single, autogenic, largely biologically-determined model to a more pluralistic paradigm. In the present view, physical and chemical factors both within and outside the wetland itself interact with biological processes to determine wetland characteristics (Heinselman 1970, van der Valk 1981, Bridgham and Richardson 1993, Glaser et al. 1997, Stromberg et al. 1997). For example, in a recent paper on what controls wetland type and distribution in Canada, Halsey et al. (1997) concluded that climatic variables and bedrock geology of the area were the primary determinants, even for these peat-dominated systems.

Two dominant expressions of the current view are the hydrogeologic (LaBaugh et al. 1987, Siegel 1988, Winter 1988, 1992, Novitzki 1989, Roulet 1990, Shedlock et al. 1993) and hydrogeomorphic (HGM) (Brinson 1993a, 1993b) frameworks within which wetlands are classified and studied. The two views are related but contribute slightly different perspectives relevant to restoration. When combined, they offer a comprehensive basis for classifying the templates for wetland restoration in a given geographic region. Winter (1992) has developed a scheme for North America that identifies 24 major "type settings" based on possible combinations of physiography and climate. Brinson et al. (1995) have produced guidelines for identifying HGM types for riverine wetlands. Furthermore, Brinson (1993a) has elaborated how wetland position along various landscape gradients influences wetland functioning.

The hydrogeomorphic (HGM) perspective is implicitly landscape-based but places greater emphasis on the wetland itself than does the hydrogeologic approach (Brinson 1993b). The rationale for the classification was to simplify the vast diversity of wetlands in terms of their biogeochemical processes and functioning. Although the HGM approach has generated considerable debate in terms of its applicability to wetland assessment for mitigation purposes, the classification on which it is based is technically sound and offers insights to those planning restorations. The classification's three components—geomorphic setting, water source, and hydrodynamics—are similar to what the hydrogeologic approach explains by reference to a larger system of surface- and ground-water flows. Geomorphic setting refers to the topographic location of wetlands within the surrounding landscape (e.g., riverine wetlands located on low-gradient alluvial flood plains). In contrast to hydrogeologic setting, it places the emphasis on the shape and location of the wetland itself rather than the surface and subsurface characteristics of the landscape that drive water-flow systems. Because its focus is wetland functioning, the HGM classification also elaborates the component of hydrodynamics within the wetland to a greater degree than do discussions of the hydrogeologic approach. It differentiates the direction of flow and strength of water movement within the wetland in terms of the ability of water to transport sediments, nutrients, and other substances to and from the wetland, as well as within it.

The hydrogeologic perspective is explicitly landscape based. It emphasizes regional climate and hydrogeologic setting as primary determinants of wetland characteristics and encourages a view of wetlands as hydrologic units of the landscape continuous with larger-scale surface- and ground-water systems. Climate drives the larger hydrologic system, determining the precipitation and evapotranspiration patterns that ultimately move surface and ground water into and out of wetlands. Hydrogeologic setting (HGS) refers to those local and regional factors that determine wetland hydrology and chemistry. It encompasses the position of the wetland in the landscape with respect to the flows of surface and ground water, and the geologic characteristics that control the flow and chemistry of water: (1) surface relief and land surface slope; (2) thickness and permeability of soils; and (3) the composition, stratigraphy, and hydraulic properties of the underlying geologic materials (Winter 1988, 1992). HGS mediates climate-driven water delivery to wetlands in terms of both quantity and chemistry.

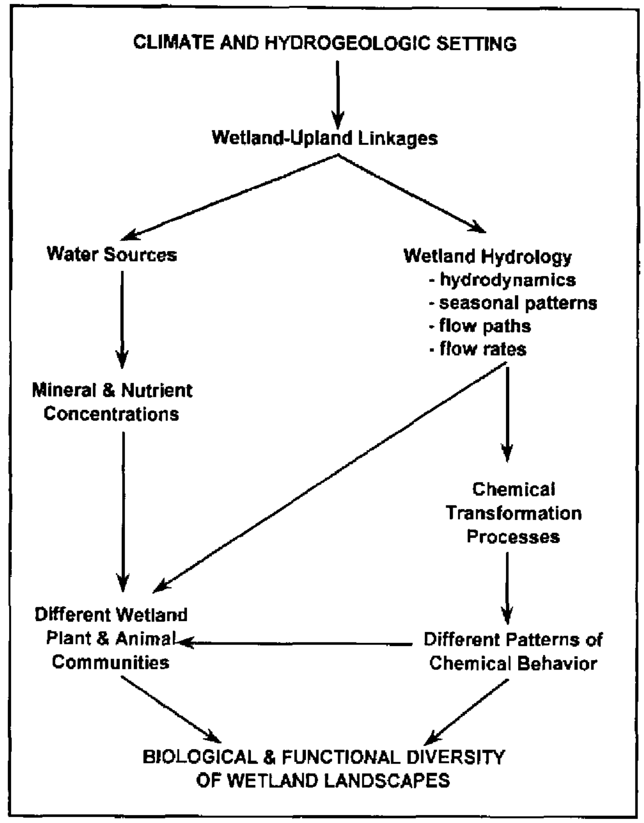

Figure 2. Conceptual model of the relationship between wetland templates (climate and hydrogeological setting) and the biological and functional diversity of wetland landscapes.

In terms of restoration, the hydrogeologic perspective identifies factors outside the wetland that determine which type(s) of wetland can be restored and maintained in a given setting. The key hydrologic and hydrologically-influenced geochemical variables that lead to the formation of specific types of wetlands are a function of regional climate and hydrogeologic setting (Figure 2). Together, these two factors govern the types and strength of linkages between wetlands and their watersheds: (a) the relative importance of different water sources (precipitation, ground water, surface water) for a wetland; (b) the elemental composition and nutrient status of waters entering wetlands; and (c) the spatial and temporal dynamics of water within a wetland and between it and adjacent systems. Changes in the position of the water table, surface- and ground-water flow paths, and rates of water flow strongly influence biogeochemical processes through their effects on reduction-oxidation status of wetland soils, surface- and ground-water interactions, and transport of mineral ions and nutrients to and within wetlands. Recent discussions of wetland classification, whether in terms of species composition or functional characteristics, distinguish wetland types on the basis of these aggregate hydrologic and hydrochemical variables (e.g., Brinson 1988, Vitt 1994, Bedford 1996, Anderson and Davis 1997). In fact, the distinctive characteristics long used to classify peatlands reflect the weakening linkage to the surrounding landscape as the peatland surface rises above the influence of surface- and groundwater inputs, and the peat body itself exerts strong control over water movement. Nonetheless, regional climate, topography, and geological substrates determine where peatlands develop (Almquist-Jacobson and Foster 1995, Halsey et al. 1997).

Wetland-Landscape Linkages

These perspectives link wetland characteristics to landscape characteristics. They do so along fundamental lines of understanding that reflect a synthesis of modern hydrogeology, biogeochemistry, and ecosystem biology. They are the counterweight to decades in which vegetation classification dominated efforts to group wetlands into appropriate categories for study. As such, neither the hydrogeologic nor HGM classification uses vegetation (either indicator species nor community types) as a primary criterion for placing wetlands into categories. Rather, vegetation is seen as the outcome of the long-term interaction of climate and landscape factors controlling wetland hydrology and hydrochemistry.

One fundamental link between wetlands and their landscapes is the effect of mineralogical composition of geological deposits and soils on the formation of specific types of wetlands through their effects on wetland hydrology and water chemistry (Siegel 1983, Wilcox et al. 1986, Siegel and Glaser 1987, Rey Benayas et al. 1990, Thompson et al. 1992, Rey Benayas and Scheiner 1993, Shedlock et al. 1993). For example, Bridgham and Richardson (1993) showed that bog-like vegetation can develop even in areas fed by ground water if the water moves through geological substrates that yield little in the way of mineral cations or nutrients (i.e., dilute waters resembling precipitation, the supposedly sole source for the surface water of bogs) (see Siegel 1983, 1988, Glaser et al. 1997). Many other studies of specific sites have related fen hydrology and geochemistry to hydrogeologic setting (Wilcox et al. 1986, Siegel and Glaser 1987, Gilvear et al. 1993, Shedlock et al. 1993, Komor 1994, Almendinger and Leete 1998). In all of these studies, distribution of distinctive plant associations corresponded to hydrologic and geochemical gradients maintained by local and regional hydrogeologic conditions. Variation in the salinity and soil chemistry of prairie wetlands, and consequently their plant species composition, is a partial function of the position of the wetland in the landscape with respect to local and regional ground-water flow systems and regional patterns of surface relief (Kantrud et al. 1989, Richardson et al. 1994).

Other recent studies have specifically linked abiotic factors controlled by hydrogeologic setting to plant community composition and diversity within and among wetlands. For example, several studies have identified gradients in water-table fluctuations, base cation availability, and pH that control patterns of species composition and richness (Vitt and Chee 1989, Johnson and Leopold 1994, de Mars et al. 1997, Pollock et al. 1998). Komor (1994), Boeye and Verheyen (1994), Boeye et al. (1994), and Almendinger and Leete (1998) have shown the strong influence that high flux rates of calcium-rich ground water have on species richness and the occurrence of rare species in fens. In addition, an extensive literature links wetland plant diversity to nutrient availability (reviewed in Bedford et al. 1999), and recent studies have shown that landscape-controlled variables (e.g., water-table dynamics, surface- and ground-water flow paths) drive N and P availability in wetlands (Boeye et al. 1994, Verhoeven et al. 1996, Hill and Devito 1997). Rey Benayas et al. (1990) and Rey Benayas and Scheiner (1993) emphasize particularly the effects of landscape-scale gradients in hydrology and geochemistry on plant species diversity.

Hill and Devito (1997) recently presented the most cogent summary published to date of how wetland-landscape interactions regulate wetland biogeochemistry and the source-sink functions of wetlands. In an overview of research conducted by Alan Hill and Nigel Roulet over the past decade in central and southern Ontario, they develop a strong conceptual and data-rich picture of the hydrologic and chemical linkages between uplands and wetlands. Working in three wetlands dominated by similar vegetation but in different hydrogeological settings, they show how geological deposits within the basins ultimately control the different patterns of chemical behavior observed among the wetlands. Wetlands that might be classified similarly according to vegetation behaved differently in terms of biogeochemistry and nutrient retention because their basins differed hydrogeologically, notably in terms of type and depth of glacial till.

Hill and Devito (1997) make several points relevant to restoration efforts, especially if the goal of restoration is to improve downstream water quality. First, element cycling and retention is determined by spatial and temporal patterns of anoxia and aeration of surface peat associated with landscape-mediated water-table fluctuations. They show how, in glaciated terrain, till depth regulates water sources and seasonal patterns in water-table elevation. Seasonal variations in the proportion of the total water budget coming from ground water, shallow surface flow, and precipitation differed among basins with different till depths. These translated into highly different patterns in water-table fluctuation and in the season and duration of anoxia in surface sediments. Basins with greater till depth sustained more continuous ground-water inputs throughout the year and more stable water levels relative to basins with thin layers of till. Because reduction-oxidation reactions regulate biogeochemical processes in wetlands, including rates of decomposition and most element transformations, wetlands with more stable water levels exhibited chemical behaviors that differed from those with widely fluctuating water levels. For example, basins with continuous ground-water inputs supported the development of peat deposits. Wetlands where water tables frequently dropped below the surface had high rates of net nitrogen mineralization, whereas wetlands with more seasonally uniform water levels showed net immobilization of nitrogen. Branfireun et al. (1996) showed that these upland-wetland hydrologic connections are not restricted to wetlands located in areas with glacial deposits but also occur on a local scale in wetlands occurring in confined bedrock basins.

Second, Hill and Devito (1997) demonstrate how different hydrologic pathways connecting uplands to wetlands influence wetland water chemistry. Two of the swamps they studied received water from two distinct ground-water flow systems. The chemistry of water entering the swamps from the local source differed substantially from that of the regional source. Concentrations of calcium, magnesium, potassium, sodium, dissolved oxygen, and nitrate nitrogen in one swamp and of sulfate in another differed for the two ground-water components. The chemistry of these input waters strongly influenced the chemistry of output waters.

Third, Hill and Devito (1997) describe how hydrologic pathways within wetlands are controlled by hydrogeological setting and influence element transformations within and export from wetlands. The pathways that water takes through a wetland determine seasonal water residence time within the wetland, extent of contact with peat, and degree of mixing with other water sources. For example, water crossing the swamp in streamlets, formed where springs and seeps emerged from the surrounding upland, had little contact with surface peat and generally showed low denitrification rates. However, nitrate-nitrogen concentrations of outlet water were lower than input water because this local ground water mixed within the swamp with regional ground water that entered the swamp through deep peat and was low in nitrate nitrogen.

The major message of Hill and Devito's (1997) paper for restoration efforts is that wetland classifications based on vegetation and substrate characteristics are limited in terms of predicting biogeochemical functioning. Results from the studies they describe tell those interested in restoring wetland functions that "a focus on interactions between hydrology and chemistry within the context of basin hydrogeology can explain differences in the role of wetlands as sources, sinks, or transformers of mineral elements" (p. 228). Although knowledge that such linkages exist is not new (Gorham 1957), until recently, the relationships have been inferred without data to support them or to give them specific meaning. The data put together by Hill, Roulet, and their students provide detail on how basin hydrogeology drives biogeochemical processes in wetlands through effects on seasonal changes in water-budget components, seasonal patterns of water-level fluctuation, flow rates, and flow paths of water to, and within, wetlands.

A Diversity of Biological and Functional Templates

The templates for wetland restoration are the many different combinations of climate and hydrogeologic setting that occur in a given geographic region. The large number of ways in which these combinations occur leads to the diversity of plant communities and specific chemical behaviors shown by wetlands in a given region (Figure. 2). The kinds, number, and spatial distribution of these templates will vary by region. On a coarse scale, Winter's (1992) classification recognizes only 24 generic types for North America. On a finer scale, however, far greater diversity exists, in terms of both biological and functional templates within specific geographic regions.

The high biological diversity of templates is evident in the many different wetland plant communities and vegetation types that have been identified and in the numerous regional classification systems that have been developed (NRC 1995). The cumulative landscape-level diversity that arises as a function of the diversity of biological templates within a single wetland community type may not be as evident. For example, Reschke (1990) classified herbaceous fens within New York State in part according to their geomorphic position (i.e., sloping fens, coastal plain poor fens). These fens tend to be relatively rich in plant species on an individual basis. From the standpoint of maintaining biological diversity, however, a landscape view presents a far richer picture (Figure 3). No individual type exceeded 100 species per $100m^2$. At the landscape level where all types are considered, the cumulative number of different species for fens rose from about 40 to over 375 per $100m^2$ (data adapted

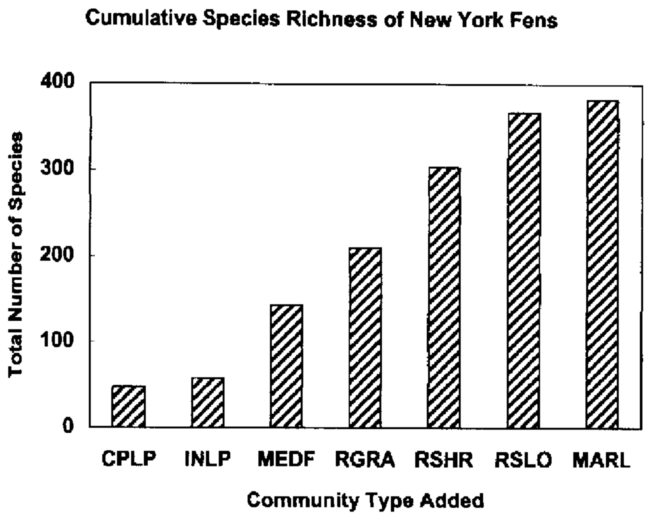

Figure 3. Cumulative plant species richness for several types of fens in New York State. Number of species is given as number per 100 m². Values were calculated by adding sequentially for each type the number of species that had not been recorded for other types (i.e., new species occurrences). Codes for fen types are as follows: CPLP = coastal plain poor fen, INLP = inland poor fen, MEDF = medium fen, RGRA = rich graminoid fen, RSHR = rich shrub fen, RSLO = rich sloping fen, MARL = marl fen. Types are according to Reschke 1990. See Slack (1994) and Reschke et al. (1990) for methods.

from Reschke et al. 1990 and Slack 1994). Maximum diversity required several different fen templates.

The functional diversity of templates, however, has no such readily catalogued indicator. It can be outlined in principle (Bedford 1996) because data like those reported by Hill and Devito (1997) are becoming increasingly available. The three swamps they discuss all fit within one of Winter's categories—terraces within riverine valleys. Within the Canadian system (Zoltai 1988), which is based largely on vegetation and substrate type, all three are classified as coniferous treed swamps or stream swamps. Yet, the nutrient-retention efficiency of these vegetatively similar systems differed markedly because of differences in hydrogeologic setting and wetland-upland linkages. With the benefit of Hill and Devito's results, the fundamental principles governing surface- and ground-water flow between wetlands and lakes now well understood (Winter and Woo 1990), and an increasing number of similar studies on wetland-upland interactions (e.g., Phillips and Shedlock 1993, Grieve et al. 1995, DeVito et al. 1997, Glaser et al. 1997), these differences in hydrogeological setting might have been deduced from topographic and surficial geology maps, as they might be for region-specific assessments for restoration purposes.

The recent development of hydrologic indices for non-tidal wetlands also may help those interested in restoring a diversity of wetland functions within a specific geographic region. Lent et al. (1997) have attempted to systematize present understanding of wetland-landscape linkages by developing two sets of hydrologic indices that quantify key hydrologic variables affecting wetland biogeochemistry, major-ion chemistry, and hydroperiod. The first set, called water-budget indices, characterizes the relative sizes of wetland water inputs and outputs. It quantifies what Brinson (1993a) presented as a ternary diagram showing the relative importance only of water inputs. Combining the proportions of both inputs and outputs, Lent et al. (1997) derived 9 basic hydrologic types of wetlands. Interestingly, "types" based on vegetation (e.g., marshes, swamps, bogs, fens) did not all group together into single hydrologic "types."

The second set, derived from the water-budget indices, consists of two indices termed hydrologic-interaction indices. These indices provide quantitative indications of the strength and type of water exchanges between a wetland and the larger ground-water and surface-water systems of which it is a part (Lent et al. 1997). These indices quantify the hydrologic portion of what Brinson (1993a) presented as conceptual categories (e.g., donor, receptor, conveyor) for classifying wetlands according to exchanges of elements and sediments as these are affected by water exchanges.

The interaction index I quantifies the degree of interaction between a wetland and the landscape. Wetlands with low I values have water budgets dominated by precipitation and evaporation and occur in isolated hydrogeologic positions in the landscape. The water budgets of wetlands with high I values are strongly dominated by ground- and surface-water inputs and outputs. These wetlands are flow-through systems and occupy hydrogeologic settings where strong inflows and outflows can occur.

The source/sink index S quantifies the nature of the interaction (i.e., whether the wetland is a net hydrologic source or sink with respect to its landscape) and whether the relationship is a strong or weak one. In addition to hydrogeologic setting, climate plays a strong role in differentiating strong sources from strong sinks. Wetlands that are net hydrologic sources have ground- and surface-water outputs greater than ground- and surface-water inputs. They thus tend to develop in humid climates, whereas strong sink wetlands are found in arid climates.

These hydrologic indices (Lent et al. 1997) begin to enumerate the high degree of variation that exists among wetlands in hydrologic functioning. They only implicitly address the diversity in biogeochemical functioning. However, they do provide those interested in restoration with a tool for quantifying two key outcomes of wetland templates—the proportion of wetland water budgets provided by different sources and the extent to which the wetland is linked hydrologically with up-gradient and down-gradient systems. Biogeochemical functioning depends on and follows from these hydrologic outcomes. As these tools are combined with the conceptual tools presented by Winter (1988, 1992) and Brinson (1993a and b), persons involved in restoration will have a clearer and clearer picture of the rich diversity of templates that allow for the development of biologically and functionally diverse wetland landscapes.

CUMULATIVE EFFECTS ANALYSIS FOR WETLAND RESTORATION

Implicit in a hydrogeological view of wetland landscapes is the need to increase the temporal and spatial scale of analysis for planning and implementing wetland restorations. Decisions about which types of wetlands to restore and where they should be located should follow from a cumulative effects analysis of the region within which the restoration will take place. A focus only at the scale of the individual project may result in a gain of wetland area, but it risks lowering further the biological and functional diversity of wetlands in a given region. It also may risk failure for the individual project. If planning does not take into account the minimal requirement that wetlands be properly placed with respect to larger surface- and groundwater systems, or the extent to which these systems have been previously modified by human activities, the desired hydrologic regime and water chemistry may not be achieved.

These points have been made previously in terms of cumulative impact assessment for wetland landscapes (Bedford and Preston 1988, Preston and Bedford 1988) and permit decisions involving compensatory mitigation (Bedford 1996, Zedler 1996). Methods for a landscape-level assessment are outlined in Bedford and Preston (1988). Gosselink et al. (1990) and Abbruzzese and Leibowitz (1997) provide specific and expanded examples. The approach is analogous for a cumulative effects analysis for wetland restoration.

Scales and Boundaries

The essence of a cumulative effects analysis is that the boundaries for analysis are drawn more broadly in both time and space than those used in conventional impact assessment, which focuses on site-specific and project-specific effects (Beanlands et al. 1986). How broadly to bound the analysis depends on both ecological principles and practical considerations. When restorations are done within the context of regulatory mitigation, practical considerations often may override all other considerations (Race and Fonseca 1996). Those restoring wetlands outside regulatory constraints, however, have the opportunity to operate with ecological principles more firmly in the foreground of their thinking.

Boundaries in Time. Restoration implies some effort to return an area to a condition resembling the self-regulating ecosystem that existed prior to disturbance, including how it was integrated into the landscape (NRC 1992). The time frame for any cumulative effects analysis for restoration purposes, therefore, necessarily includes reference to past conditions, not only for the wetlands of an area but also for the watersheds of which they are a part. The decision on how far back to go in time is not straightforward. The goals of the restoration as well as data availability will dictate choices.

Methods for establishing past conditions include the records of early land surveyors (e.g., Tans 1976), aerial photographs, botanical records, and historical impact assessment (Showers 1996). Land surveyors' records, which exist for much of the United States, provide a rich source of information on the areal extent of wetlands in the mid- to late 1800s but do little to differentiate more than the major types of wetlands. They contain more detailed information on the terrain surrounding wetlands than on the wetlands, which often were difficult to traverse. Most parts of the United States have aerial photographs taken every decade by the Agricultural Stabilization and Conservation Service and dating back to the late 1930s. These photos allow wetlands to be differentiated to class (e.g., evergreen or deciduous swamp, emergent marsh). The landscape settings of particular types of wetlands also can be extracted from these photos. Botanical records available in the herbaria of museums, colleges, and universities often date back to the late 1800s and provide specific data on species composition of sites visited by local botanists.

All of these methods for establishing historical conditions are time-consuming and offer useful but limited information about the numbers, kinds, and geographic distribution of wetland templates in a region. A more direct approach is to infer past conditions by a combination of (a) fundamental principles of hydrogeology and the formation of wetlands (Winter 1992); (b) fundamental principles defining hydrogeomorphic class (Brinson 1993b); (c) existing knowledge of regional topography and surficial geology; and (d) use of the best remaining examples of wetlands occurring in each hydrogeologic/hydrogeomorphic setting in a region as reference wetlands. That is, use the landscape to define templates. Once the different types of templates are identified, characterize their hydrology, chemistry, and species composition by matching them to reference wetlands associated with each template. Brinson and Rheinhardt (1996) and Cole et al. (1997) have outlined various aspects of this approach. The approach will be difficult to implement in regions where topography and geomorphological processes have been modified (e.g., where dikes, dams, and levees have been constructed) or where few wetlands of any type remain as reference sites.

Boundaries in Space. Defining the areal extent of the analysis should be based on a non-hierarchical approach in which multiple scales of analysis are considered (Bedford and Preston 1988, Bedford 1996). Ecological considerations dictate several spatial scales that are not necessarily nested, while practical and political realities dictate still other non-overlapping boundaries. This multiple-boundary dilemma cannot be solved but only recognized. It means that those planning restorations will need to consider several "landscapes."

From an ecological perspective, different wetland functions mean different criteria for setting boundaries for an assessment. Hydrologic functions dictate the use of major watersheds (i.e., the drainage basins for large rivers). The U.S. Geological Survey divides the United States into 18 accounting units or major drainage basins, which are further divided into cataloging units consisting of smaller watersheds of about 1800 km^2. Lee and Gosselink (1988) recommended the use of accounting units for cumulative impact assessments for bottomland hardwood swamps of the southern U.S. The smaller cataloging units may be more appropriate for wetlands less dependent on the flows of major rivers. Lee and Gosselink (1988) and Bedford (1996) discuss why watersheds are appropriate units for cumulative impact assessment.

Nutrient removal and other biogeochemical functions also call for using watershed boundaries but further require attention to differences within watersheds in soils and geological materials that affect water chemistry. In the United States, this can be accomplished by using both watershed boundaries and Omernik's (1987) ecoregions, which are defined on the basis of landform, land use, and potential natural vegetation. For Canada, ecodistricts consisting of areas with similar climate and physical landscapes have been identified and mapped for the provinces of Alberta, Manitoba, and Saskatchewan (Veldhuis et al. 1996, as cited in Halsey et al. 1997), and the framework has been developed for the rest of Canada (Ecological Stratification Working Group 1995, as cited in Halsey et al. 1997).

The existing wetland regionalization scheme (Zoltai 1988) could serve as the starting point for analyzing patterns within broad ecological regions rather than political units of Canada. It recognizes 7 broad climatic zones affecting wetland formation—arctic, subarctic, temperate, prairie, boreal, oceanic, and mountain. Each zone is further divided but, again, largely on the basis of climate (see individual chapters in National Wetlands Working Group 1988). The zones strongly parallel the distribution of potential natural vegetation in Canada. Differences in landform, geological formations, or soil types within the zones do not form part of the regionalization but could be overlaid on the wetland regions by using Bostock's (1970, as cited in National Wetlands Working Group 1988) map of the physiographic regions of Canada and existing soils maps. For example, this process would divide Canada's Eastern Temperate Wetland Region into the Laurentian Uplands, St. Lawrence Lowlands, and Appalachian Highlands. The differences among these physiographic regions in terms of topography and the composition of geological materials will produce wetlands with different characteristics.

Landscape Diversity and Landscape Profiles

Diversity typically is measured in terms of number of species and their relative abundances. If this diversity is ultimately a function of the diversity of wetland templates in a given region (Rey Benayas et al. 1990, Rey Benayas and Scheiner 1993, Vitt and others 1995), then measuring landscape diversity means identifying the kinds, numbers, relative abundances, and distribution of different wetland templates (i.e., a demography of wetlands and wetland templates) (Bedford 1996). Halsey et al. (1997) recently completed such an analysis for the present distribution of wetlands in the province of Manitoba.

For restoration, a comparative sense of the present versus the past "demography" of wetlands and wetland templates in a region is an essential tool for planning restorations. A comparative analysis would reveal which types have suffered the most extensive losses, as well as the existing availability of templates within which specific types of wetlands could be restored. It should inform decisions about which types to restore and which types of restorations are possible in a given region. Some templates, such as river floodplains and Great Lakes shorelines, have been so modified that restoration to conditions at the time of pre-European settlement will be difficult to achieve.

The basis for comparing past and present conditions is the preparation of landscape profiles (Bedford 1996). The profiles would consist of the elements identified in Table 1. They could be formally or informally compiled by reference to resources existing for much

Table 1. Elements of landscape profiles to be prepared for purposes of planning wetland restorations.

WETLAND TEMPLATES	
Catalog and Map of Wetland Templates	Geographical Analysis of Templates
Hydrogeomorphic classes	Proportion lost and location
Hydrogeologic and climatic settings	Proportion modified and location
Wetland-landscape linkages	Proportion remaining and location
WETLAND ECOSYSTEMS	
Catalog of Wetland Ecosystems	Geographical Analysis of Wetland Ecosystems
Types	Proportion lost and location
Relative abundances of each type	Proportion modified or degraded and location
Uncommon, unique, rare types	Proportion remaining and location
Sites of unusual natural heritage value	

of the United States and southern Canada: (a) maps of the physiographic provinces of a state or province, which reflect major land-form types, surficial geology, and potential vegetation; (b) topographic maps from which a wetland's topographic position in the landscape (e.g., headwaters, river valley bottom), approximate surface watershed, ratio of watershed to wetland size, local land form (e.g., kettle, through-valley, hill slope), and maximum possible flow path length can be drawn; (c) maps of bedrock and surficial geology from which the types of geological deposits surrounding and underlying wetlands can be identified; (d) soil surveys from which soil types and shallow substrate composition of the surface watershed can be identified; and (e) aerial photographs from which surface-water inflows and outflows can be identified. The likelihood that a wetland receives ground water from more than one flow system can be deduced from its topographic position in the watershed and the type of terrain in which it occurs (e.g., hummocky, flat terraces), which can be deduced from topographic maps. The use of geographic information systems in preparing the profiles would allow changes in the resource to be tracked through time and would provide a powerful tool for communicating with the public and decision- makers.

However, even if resources don't allow detailed profiles to be prepared, any attempt to summarize the type of information listed above and in Table 1 will enhance decision-making and increase the probability that restorations will contribute to the goal of restoring wetland landscape diversity. The work by Halsey et al. (1997) for the entire province of Manitoba shows that such a broad-scale perspective on wetland diversity can be obtained. Thompson et al. (1992) developed the foundations for such a landscape view of Iowa fens. The general nature of the link between hydrogeological setting and the biological and functional diversity of wetlands is sufficiently well established that waiting for specific development of the relationships for all regions is not justified. The type of contextual, landscape-scale thinking outlined here can be infused into decision-making merely by acknowledging its relevance to any goal that includes restoration of wetland biological or functional diversity.

CONCLUSIONS

Wetland restoration is an act of human creativity. By definition, it seeks to replace what has been lost. By definition then, it should be undertaken with knowledge of what has been lost. That knowledge consists of more than the number of hectares of wetland lost. It consists more fundamentally of a knowledge of relationships among landscape units. Wetlands are hydrologically, chemically, and biologically linked to the landscapes in which they occur. Restoring the biological and functional diversity of wetlands means understanding that specific types of wetlands develop in specific places in the landscape. It also means understanding that seemingly similar types can have quite different biogeochemical behaviors because of how they are linked to their watersheds. As Brinson (1993a) pointed out, what controls much of the variation that exists among wetlands is obscured if those planning restoration fail to look beyond the wetland boundary. By looking beyond the boundary of individual wetlands, those planning wetland restorations are more likely to see the relationships necessary for restoration efforts to succeed. By looking at the wetland landscape as a whole, those planning restorations may choose to enhance rather than diminish regional wetland diversity.

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LITERATURE CITED

Abbruzzese, B. and S. Leibowitz. 1997. Environmental auditing: a synoptic approach for assessing cumulative impacts to wetlands. Environmental Management 21:457–475.

Almendinger, J. E. and J. H. Leete. 1998. Regional and local hydrogeology of calcareous fens in the Minnesota River Basin, USA. Wetlands 18:184–202.

Almquist-Jacobson, H. and D. R. Foster. 1995. Toward an integrated model for raised-bog development: theory and field evidence. Ecology 76:2503–2516.

Anderson, D. S. and R. B. Davis. 1997. The vegetation and its environments in Maine peatlands. Canadian Journal of Botany 75: 1785–1805.

Beanlands, G. E., W. J. Erckmann, G. H. Orians, J. O'Riordan, D. Policansky, M. H. Sadar, and B. Sadler (eds.). 1986. Cumulative environmental effects: a binational perspective. Canadian Environmental Assessment Research Council, Ottawa, Ontario, and U.S. National Research Council, Washington, DC, USA.

Beaulac, M. N. and K. H. Reckhow. 1982. An examination of land use—nutrient export relationships. Water Resources Bulletin 18: 1013–1022.

Bedford, B. L. 1996. The need to define hydrological equivalence at the landscape scale for freshwater wetland mitigation. Ecological Applications 6:57–68.

Bedford, B. L. and E. M. Preston. 1988. Developing the scientific basis for assessing cumulative effects of wetland loss and degradation on landscape functions: status, perspectives and prospects. Environmental Management 12:751–771.

Bedford, B. L., M. R. Walbridge, and A. Aldous. 1999. Patterns in nutrient availability and plant diversity of temperate North American wetlands. Ecology 80:2151–2169.

Boeye, D., L. Clement, and R. F. Verheyen. 1994. Hydrochemical variation in a ground-water discharge fen. Wetlands 14:122–133.

Boeye, D. and R. F. Verheyen. 1994. The relation between vegetation and soil chemistry gradients in a groundwater discharge fen. Journal of Vegetation Science 5:553–560.

Branfireun, B. A., A. Heyes, and N. T. Roulet. 1996. The hydrology and methylmercury dynamics of a Precambrian Shield headwater peatland. Water Resources Research 32:1785–1794.

Bridgham, S. D. and C. J. Richardson. 1993. Hydrology and nutrient gradients in North Carolina peatlands. Wetlands 13:207–218.

Brinson, M. M. 1988. Strategies for assessing the cumulative effects of wetland alteration on water quality. Environmental Management 12:655–662.

Brinson, M. M. 1993a. Changes in the functioning of wetlands along environmental gradients. Wetlands 13:65–74.

Brinson, M. M. 1993b. A hydrogeomorphic classification for wetlands. Technical Report WRP-DE-4, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS, USA.

Brinson, M. M., F. R. Hauer, L. C. Lee, W. L. Nutter, R. D. Rheinhardt, R. D. Smith, and D. Whigham. 1995. A guidebook for application of hydrogeomorphic assessments to riverine wetlands. U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, MS. USA. Wetlands Research Program Technical Report WRP-DE-11.

Brinson, M. M. and R. D. Rheinhardt. 1996. The role of reference wetlands in functional assessment and mitigation. Ecological Applications 6:69–76.

Cole, C. A., R. P. Brooks, and D. H. Wardrop. 1997. Wetland hydrology as a function of hydrogeomorphic (HGM) subclass. Wetlands 17:456–467.

Correll, D. L., T. E. Jordan, and D. E. Weller. 1992. Nutrient flux in a landscape: effects of coastal land use and terrestrial community mosaic on nutrient transport to coastal waters. Estuaries 15:431–442.

Curtis, J. T. 1959. The Vegetation of Wisconsin. The University of Wisconsin Press, Madison, WI, USA.

Dahl, T. E. 1990. Wetlands Losses in the United States 1780's to 1980's. U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC, USA.

Dahl, T. E. and G. J. Allord. 1996. History of wetlands in the conterminous United States. p. 19–26. *In* J.D. Fretwell, J.S. Williams, and P.J. Redman (eds.) National Water Summary on Wetland Resources. United States Geological Survey Water-Supply Paper 2425.

Dahl, T. E., C. E. Johnson, and W. E. Frayer. 1991. Status and trends of wetlands in the conterminous United States, mid-1970s to mid-1980s. U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC, USA.

de Mars, H., M. J. Wassen, and H. Olde Venterink. 1997. Flooding and groundwater dynamics in fens in eastern Poland. Journal of Vegetation Science 8:319–328.

Devito, K. J., J. M. Waddington, and B. A. Branfireun. 1997. Flow reversals in peatlands influenced by local groundwater systems. Hydrological Processes 11:103–110.

Dillon, P. J. and W. B. Kirchner. 1975. The effects of geology and land use on the export of phosphorus from watersheds. Water Resources 9:135–148.

Doren, R. F. and D. T. Jones. 1997. Plant management in Everglades National Park. p. 275–286. *In* D. Simberloff, D. C. Schmitz, and T.C. Brown (eds.) Strangers in Paradise: Impact and Management of Nonindigenous Species in Florida. Island Press, Washington, DC, USA.

Ehrenfeld, J. G. 1983. The effects of changes in land-use on swamps of the New Jersey pine barrens. Biological Conservation 25:353–375.

Ehrenfeld, J. G. and J. P. Schneider. 1991. *Chamaecyparis thyoides* wetlands and suburbanization: effects on hydrology, water quality and plant community composition. Journal of Applied Ecology 28:467–490.

Gilvear, D. J., R. Andrews, J. H. Tellam, J. W. Lloyd, and D. N. Lerner. 1993. Quantification of the water balance and hydrogeological processes in the vicinity of a small groundwater-fed wetland, East Anglia, UK. Journal of Hydrology 144:311–334.

Glaser, P. H., D. I. Siegel, E. A. Romanowicz, and Y. P. Shen. 1997. Regional linkages between raised bogs and the climate, groundwater, and landscape of north-western Minnesota. Journal of Ecology 85:3–16.

Glooschenko, W. A., C. Tarnocai, S. Zoltai, and V. Glooschenko. 1993. Wetlands of Canada and Greenland. p. 415–514. *In* D. Whigham, D. Dykyjova, and S. Hejny (eds.) Wetlands of the World: Inventory, Ecology and Management. Volume I. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Gorham, E. 1957. The development of peatlands. Quarterly Review of Biology 32:145–166.

Gosselink, J. G., G. P. Shaffer, L. C. Lee, D. M. Burdick, D. L. Childers, N. C. Leibowitz, S. C. Hamilton, R. Boumans, D. Cushman, S. Fields, M. Koch, and J. M. Visser. 1990. Landscape conservation in a forested wetland watershed: can we manage cumulative impacts? BioScience 40:588–600.

Grieve, I. C., D. G. Gilvear, and R. G. Bryant. 1995. Hydrochemical and water source variations across a floodplain mire, Insh Marshes, Scotland. Hydrological Processes 9:99–110.

Halsey, L., D. Vitt, and S. Zoltai. 1997. Climatic and physiographic controls on wetland type and distribution in Manitoba, Canada. Wetlands 17:243–262.

Harris, L. D. 1988. The nature of cumulative impacts on biotic diversity of wetland vertebrates. Environmental Management 12: 675–693.

Heimlich, R. and J. Melanson. 1995. Wetlands lost, wetlands gained. National Wetlands Newsletter 17(1):23–25.

Heinselman, M. L. 1970. Landscape evolution, peatland types, and the environment in the Lake Agassiz Peatlands Natural Area, Minnesota. Ecological Monographs 40:235–261.

Herdendorf, C. E. 1987. The Ecology of the Coastal Marshes of Western Lake Erie: A Community Profile. U.S. Fish and Wildlife Service, Washington, DC, USA. Biological Report 85(7.9).

Hill, A. R. 1978. Factors affecting the export of nitrate-nitrogen from drainage basins in southern Ontario. Water Resources 12:1045–1057.

Hill, A. R. and K. J. Devito. 1997. Hydrological-chemical interactions in headwater forest wetlands. p. 213–230. *In* C. C. Trettin, M. F. Jurgensen, D. F. Grigal, M. R. Gale, and J. K. Jeglum (eds.) Northern Forested Wetlands: Ecology and Management. Lewis Publishers, Boca Raton, FL, USA.

Jean, M. and A. Bouchard. 1991. Temporal changes in wetland landscapes of a section of the St. Lawrence River, Canada. Environmental Management 15:241–250.

Johnson, A. M. and D. J. Leopold. 1994. Vascular plant species richness and rarity across a minerotrophic gradient in wetlands of St. Lawrence County, New York, USA. Biodiversity and Conservation 3:606–627.

Johnston, C. A. 1994. Cumulative impacts to wetlands. Wetlands 14:49–55.

Johnston, C. A., N. E. Detenbeck, and G. J. Niemi. 1990. The cumulative effect of wetlands on stream water quality and quantity: a landscape approach. Biogeochemistry 10:105–141.

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

Kantrud, H. A., G. L. Krapu, and G. A. Swanson. 1989. Prairie basin wetlands of the Dakotas: A community profile. U.S. Fish and Wildlife Service, Washington, DC, USA. Biological Report 85(7.28).

Keddy, P. A. 1990. Competitive hierarchies and centrifugal organization in plant communities. p. 265–290. *In* J. B. Grace and D. A. Tilman (eds.) Perspectives on Plant Competition. Academic Press, New York, NY, USA.

Keddy, P. A. and A. A. Reznicek. 1985. Vegetation dynamics, buried seeds, and water level fluctuations on the shorelines of the Great Lakes. p. 33–58. *In* H. H. Prince and F. M. D'Itri (eds.) Coastal Wetlands. Lewis Publishers, Inc., Chelsea, MI, USA.

Kentula, M. E., R. P. Brooks, S. E. Gwin, C. C. Holland, A. D. Sherman, and J. C. Sifneos. 1992. An Approach to Improving Decision Making in Wetland Restoration and Creation. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR, USA.

Komor, S. C. 1994. Geochemistry and hydrology of a calcareous fen within the Savage Fen wetlands complex, Minnesota, USA. Geochimica et Cosmochimica Acta 58:3353–3367.

Kusler, J. A. and M. E. Kentula. 1990. Executive Summary. p. xvii-xxv. *In* J.A. Kusler and M.E. Kentula (eds.) Wetland Creation and Restoration: the Status of the Science. Island Press, Washington, DC, USA.

LaBaugh, J. W., T. C. Winter, V. A. Adomaitis, and G. A. Swanson. 1987. Geohydrology and chemistry of prairie wetlands, Stutsman County, North Dakota. U.S. Geological Survey Professional Paper 1431.

Lee, C. L. and J. G. Gosselink. 1988. Cumulative impacts on wetlands: linking scientific assessments and regulatory alternatives. Environmental Management 12:591–602.

Lent, R. M., P. K. Weiskel, F. P. Lyford, and D. S. Armstrong. 1997. Hydrological indices for nontidal wetlands. Wetlands 17:19–30.

Malecki, R. A., B. Blossey, S. D. Hight, D. Schroeder, L. T. Kok, and J. R. Coulson. 1993. Biological control of purple loosestrife. BioScience 43:680–686

Moore, P. D. and D. J. Bellamy. 1974. Peatlands. Springer-Verlag, New York, NY, USA.

Morris, J. T. 1991. Effects of nitrogen loading on wetland ecosystems with particular reference to atmospheric deposition. Annual Review of Ecology and Systematics 22:257–279.

National Research Council. 1992. Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy. National Academy Press, Washington, DC, USA.

National Research Council. 1995. Wetlands: Characteristics and Boundaries. National Academy Press, Washington, DC, USA.

National Wetlands Working Group. 1988. Wetlands of Canada. Ecological Land Classification Series, No. 24. Sustainable Development Branch, Environment Canada, Ottawa, Ontario, Canada, and Polyscience Publications Inc., Montreal, Quebec, Canada.

Neely, R. K. and J. L. Baker. 1989. Nitrogen and phosphorus dynamics and the fate of agricultural runoff. p. 92–131. *In* A. G. van der Valk (ed.) Northern Prairie Wetlands. Iowa State University Press, Ames, IA, USA.

Novitzki, R. P. 1989. Wetland Hydrology. p. 47–64. *In* S. K. Majumdar, R. P. Brooks, F. J. Brenner and R. W. Tiner, Jr. (eds.) Wetlands Ecology and Conservation: Emphasis in Pennsylvania. The Pennsylvania Academy of Science, Easton, PA, USA.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77:118–125.

Phillips, P. J. and R. J. Shedlock. 1993. Hydrology and chemistry of groundwater and seasonal ponds in the Atlantic Coastal Plain in Delaware, USA. Journal of Hydrology 141:157–178.

Pinder, D. A. and M. E. Witherick. 1990. Port industrialization, urbanization, and wetland loss. p. 234–266. *In* M. Williams (ed.) Wetlands: A Threatened Landscape. Basil Blackwell, Inc., Cambridge, MA, USA.

Poiani, K. A., B. L. Bedford, and M. D. Merrill. 1996. A GIS-based index for relating landscape characteristics to potential nitrogen leaching to wetlands. Landscape Ecology 11:237–255.

Pollock, M. M., R. J. Naiman, and T. A. Hanley. 1998. Plant species richness in riparian wetlands—a test of biodiversity theory. Ecology 79:94–105.

Preston, E. M. and B. L. Bedford. 1988. Evaluating cumulative effects on wetland functions: a conceptual overview and generic framework. Environmental Management 12:565–583.

Prince, H. 1997. Wetlands of the American Midwest. University of Chicago Press, Chicago, IL, USA.

Race, M. S. and M. S. Fonseca. 1996. Fixing compensatory mitigation: what will it take? Ecological Applications 6:94–101.

Reschke, C. 1990. Ecological Communities of New York State. New York Natural Heritage Program. New York State Department of Environmental Conservation, Latham, NY, USA.

Reschke, C., B. Bedford, N. G. Slack, and F. R. Wesley. 1990. Fens of New York State. Abstract, 75th Annual Ecological Society of America Meeting, Snowbird, UT. Bulletin of the Ecological Society of America 71:299.

Rey Benayas, J. M., F. G. Bernáldez, C. Levassor, and B. Peco. 1990. Vegetation of ground water discharge sites in the Douro basin, central Spain. Journal of Vegetation Science 1:461–466.

Rey Benayas, J. M. and S. M. Scheiner. 1993. Diversity patterns of wet meadows along geochemical gradients in central Spain. Journal of Vegetation Science 4:103–108.

Richardson, J. L., J. L. Arndt, and J. Freeland. 1994. Wetland soils of the prairie potholes. Advances in Agronomy 52:121–172.

Roulet, N. T. 1990. Hydrology of a headwater basin wetland: ground water discharge and wetland maintenance. Hydrological Processes 4:387–400.

Rubec, C. D. A., P. Lynch-Stewart, G. M. Wickware, and I. Kessel-Taylor. 1988. Wetland utilization in Canada. p. 380–412. *In* National Wetlands Working Group. Wetlands of Canada. Ecological Land Classification Series, No. 24. Sustainable Development Branch, Environment Canada, Ottawa, Ontario, Canada, and Polyscience Publications Inc., Montreal, Quebec, Canada.

Shedlock, R. J., D. A. Wilcox, T. A. Thompson, and D. A. Cohen. 1993. Interactions between groundwater and wetlands, southern shore of Lake Michigan, USA. Journal of Hydrology 141:127–155.

Showers, K. B. 1996. Soil erosion in the kingdom of Lesotho and development of historical environmental impact assessment. Ecological Applications 6:653–664.

Siegel, D. I. 1983. Ground water and the evolution of patterned mires, Glacial Lake Agassiz Peatlands, northern Minnesota. Journal of Ecology 71:913–921.

Siegel, D. I. 1988. Evaluating cumulative effects of disturbance on the hydrologic function of bogs, fens, and mires. Environmental Management 12:621–626.

Siegel, D. I. and P. H. Glaser. 1987. Ground water flow in a bog-fen complex, Lost River Peatland, northern Minnesota. Journal of Ecology 75:743–754.

Slack, N. G. 1994. Can one tell the mire type from the bryophytes alone? Journal of the Hattori Botanical Laboratory 75:149–159.

Stromberg, J. C., J. Fry, and D. T. Pattern. 1997. Marsh development after large floods in an alluvial, arid-land river. Wetlands 17:292–300.

Stromberg, J. C., R. Tiller, and B. Richter. 1996. Effects of ground-water decline on riparian vegetation of semiarid regions: the San Pedro River, Arizona, USA. Ecological Applications 6:113–131.

Tans, W. 1976. The presettlement vegetation of Columbia County, Wisconsin in the 1830s. Wisconsin Department of Natural Resources, Department of Natural Resources, Madison, WI, USA. Technical Bulletin 90.

Thompson, C. A., E. A. Bettis III, and R. G. Baker. 1992. Geology of Iowa fens. Journal Iowa Academy of Science 99:53–59.

Tiner, R. W. Jr. 1984. Wetlands of the United States: current status and recent trends. U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC, USA.

Van der Valk, A. G. 1981. Succession in wetlands: a Gleasonian approach. Ecology 62:688–696.

Verhoeven, J. T. A., W. Koerselman, and A. F. M. Meuleman. 1996. Nitrogen or phosphorous-limited growth in herbaceous, wet vegetation: relations with atmospheric inputs and management regimes. Trends in Ecology and Evolution 11:494–497.

Vitt, D. H. 1994. An overview of factors that influence the development of Canadian peatlands. Memoirs of the Entomological Society of Canada 169:7–20.

Vitt, D. H. and W.-L. Chee. 1989. Relationships of the vascular plant and bryophyte components of fen vegetation to surface water chemistry and peat chemistry of fens in Alberta, Canada. Vegetatio 92:87–106.

Vitt, D. H., Y. Li, and R. J. Belland. 1995. Patterns of bryophyte diversity in peatlands of continental western Canada. The Bryologist 98:218–227.

Whigham, D. F., C. Chitterling, and B. Palmer. 1988. Impacts of freshwater wetlands on water quality: a landscape perspective. Environmental Management 12:663–671.

Whillans, T. H. 1982. Changes in marsh area along the Canadian shore of Lake Ontario. Journal of Great Lakes Research 8:570–577.

Wilcox, D. A., R. J. Shedlock, and W. H. Hendrickson. 1986. Hydrology, water chemistry and ecological relations in the raised mound of Cowles Bog. Journal of Ecology 74:1103–1117.

Winter, T. C. 1988. Conceptual framework for assessment of cumulative impacts on the hydrology of non-tidal wetlands. Environmental Management 12:605–620.

Winter, T. C. 1992. A physiographic and climatic framework for hydrologic studies of wetlands. p. 127–148. *In* R. D. Robarts and M. L. Bothwell (eds.) Aquatic Ecosystems in Semi-Arid Regions: Implications for Resource Management. N.H.R.I. Symposium Series 7, Environment Canada, Saskatoon, Saskatchewan, Canada.

Winter, T.C. and M. -K. Woo. 1990. Hydrology of lakes and wetlands. p. 159–187. *In* M.G. Wolman and H.C. Riggs (eds.) The Geology of North America, Vol. 0–1, Surface Water Hydrology. Geological Society of America, Boulder, CO, USA.

Zedler, J. B. 1996. Coastal mitigation in southern California: the need for a regional restoration strategy. Ecological Applications 6:84–93.

Zoltai, S. C. 1988. Wetland environments and classification. p. 1–26. *In* National Wetlands Working Group. Wetlands of Canada. Ecological Land Classification Series, No. 24. Sustainable Development Branch, Environment Canada, Ottawa, Ontario, Canada, and Polyscience Publications Inc., Montreal, Quebec, Canada.

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