BioScience Advance Access published February 25, 2015

Geographically Isolated Wetlands are Important Biogeochemical Reactors on the Landscape

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Wetlands provide many ecosystem services, including sediment and carbon retention, nutrient transformation, and water quality improvement. Although all wetlands are biogeochemical hotspots, geographically isolated wetlands (GIWs) receive fewer legal protections compared with other types of wetlands because of their apparent isolation from jurisdictional waters. Here, we consider controls on biogeochemical functions that influence water quality, and estimate changes in ecosystem service delivery that would occur if these landscape features were lost following recent US Supreme Court decisions (i.e., Rapanos, SWANCC). We conclude that, despite their lack of persistent surfacewater connectivity or adjacency to jurisdictional waters, GIWs are integral to biogeochemical processing on the landscape and therefore maintaining the integrity of US waters. Given the likelihood that any GIW contributes to downstream water quality, we suggest that the burden of proof could be shifted to assuming that all GIWs are critical for protecting aquatic systems until proven otherwise.

Keywords: geographically isolated wetlands, connectivity, adjacency, biogeochemistry, wetland protection

Wetlands exist along a continuum of hydrologic connectivity to surrounding upland and aquatic ecosystems. The transport and retention of carbon (C), nutrients, and other solutes in wetlands are mediated through a vast array of hydrologic and biogeochemical processes across this continuum. These processes, in turn, provide valuable ecosystem services such as C sequestration, removal of reactive nitrogen (N), phosphorus (P) sorption and storage, and sediment retention (Craft and Casey 2000, Badiou et al. 2011, Wolf et al. 2013). Wetlands across the landscape, as embedded depressions or intermediary elements and the terrestrial–aquatic interface, reduce the amount of C, N, P, and sediment that reaches downstream aquatic systems, thereby helping to maintain the physical, chemical, and biological integrity of the nation's waters, which is the overarching goal of the Clean Water Act (CWA).

The effectiveness of pollutant retention and transformation by wetlands varies across the hydrogeomorphic continuum and is influenced by a wetland's degree of connectivity with nearby aquatic ecosystems and the surrounding landscape (Craft and Casey 2000, Racchetti et al. 2011, Marton et al. 2014). An important and often undervalued class of wetlands is the class of geographically isolated wetlands (GIWs) defined by Tiner (2003, p. 495) as "hydrophytic plant communities surrounded by terrestrial plant communities or undrained hydric soils surrounded by nonhydric soils." GIWs are formed by natural forces that create depressions on the landscape wherein precipitation, near surface (i.e., interstitial) water, or groundwater create saturated soil conditions for sufficient duration for hydric soils and hydrophytic plant assemblages to develop (Mitsch and Gosselink 2000). Tiner (2003) identified iconic examples of GIWs, including the prairie potholes of the upper Midwest of the United States and central Canada, playas of the southwestern United States and Mexico, woodland vernal pools of New England and eastern Canada, the Carolina Bays of the Atlantic Coastal Plain, cypress domes of the southeastern coastal plain, and California vernal pools, amongst others. Tiner (2003) also identified special areas of interest in the contiguous United States wherein geographically isolated wetlands may be particularly common (figure 1).

Tiner (2003) pointed out that the term *isolation* may not be appropriate across geographical, hydrological, and ecological perspectives, which implies that although a wetland might be isolated in the sense that it is surrounded by upland, it may still exchange organisms, water, solutes, and energy with the surrounding landscape and downstream ecosystems. Despite what the term *geographically isolated* suggests, these wetlands cannot be uniformly classified into even one hydrological type as there is substantial variation

BioScience XX: 1–11.
doi:10.1093/biosci/biv009
Advance Access publication XX XXXX, XXXX

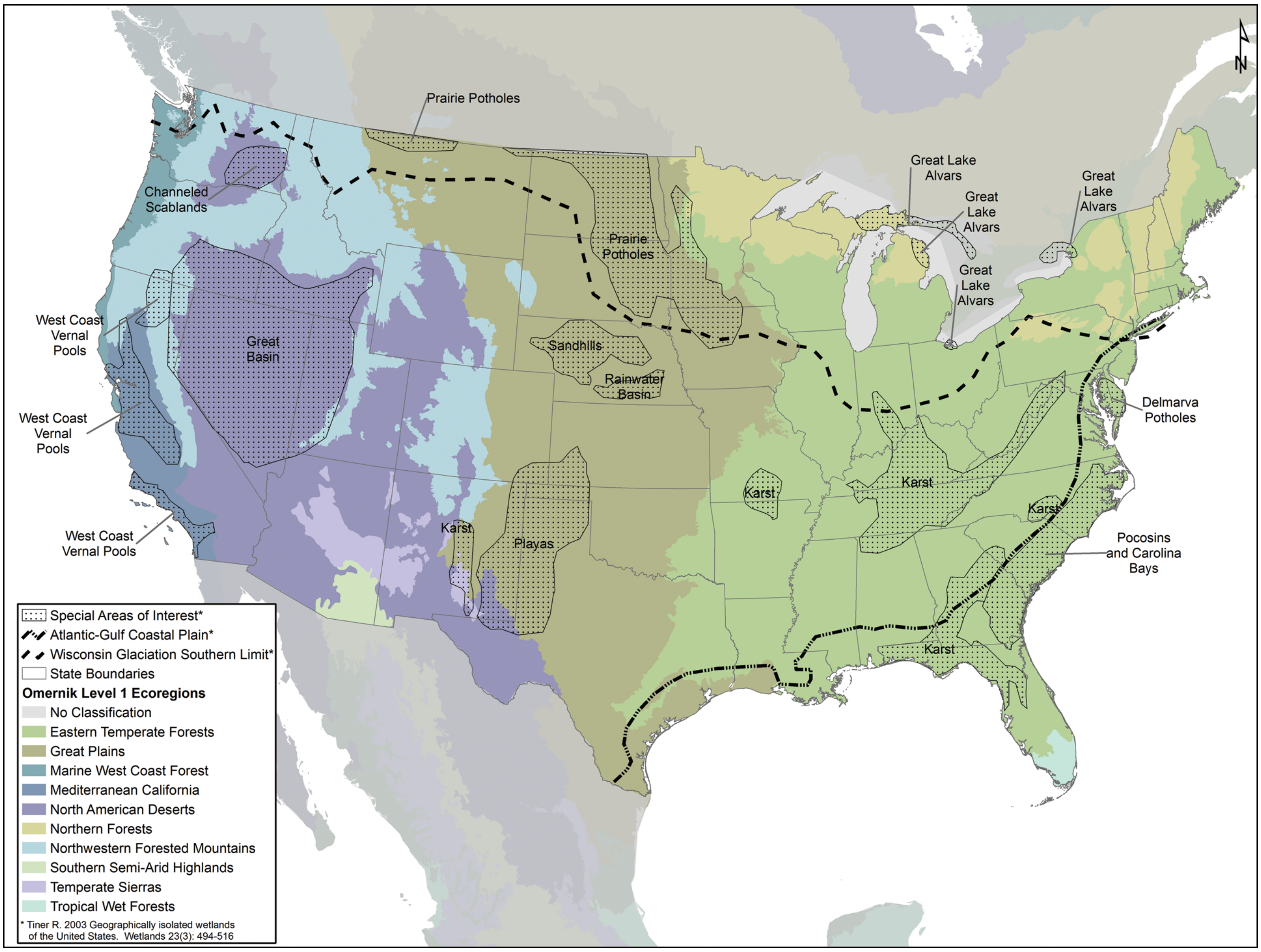

Figure 1. Generalized regions potentially containing geographically isolated wetlands. Source: Adapted with kind permission of Springer Science+Business Media from figure 3 of Tiner (2003).

both within wetlands over time and among wetlands, as the degree of connectivity via surface and subsurface hydrologic flow paths changes (Leibowitz 2003). Tiner's definition is seemingly simple to apply at broad scales using remotely sensed data and GIS resources, but in reality the adjacency, connectivity, and isolation of a wetland relative to other landscape units (particularly navigable waters) is exceedingly difficult to determine owing to intermittent surface-water, groundwater, or biogeochemical connectivity (Creed et al. 2003, Leibowitz and Vining 2003, Lang et al. 2012).

It is because of this complexity and at times uncertainty in defining hydrological (or other) connectivity of GIWs to nearby waters that have blurred their inclusion as part of the nation's waters. The CWA's primary goal is to restore and maintain the physical, chemical, and biological integrity of US waters. How these waters are defined (e.g., navigable, adjacent) directly influences how wetlands are protected at the federal level. The US Supreme Court's ruling in 2001 in *Solid Waste Agency of Northern Cook County v. US Army Corps of Engineers*, 531 US 159 (2001) (SWANCC) changed the criteria by which wetlands are protected (Downing et al. 2003). Specifically, the use of wetlands by migratory birds was ruled to be inadequate as the sole determinant in assessing CWA jurisdiction for an isolated, intrastate, non-navigable water. This decision, and the subsequent Rapanos ruling (Rapanos v. United States, US 126 S. Ct. 2006), diminished the reach of the CWA by determining that federal jurisdiction may extend, in part, only to those GIWs adjacent to traditional navigable waters or those found to have a substantial effect, or a "significant nexus" with traditional navigable waters, which are protected under the CWA. This becomes further complicated considering that wetlands can be considered adjacent even if they are separated from jurisdictional waters by manmade barriers, dikes, and natural river berms.

However, even in the absence of obvious hydrological connectivity, an individual GIW, GIW complex (a series of proximally situated GIWs), or a host of GIWs in a regional setting, may still provide functions to maintain and improve the quality of traditional navigable waters. This water-quality service derives from the many biogeochemical processes that occur in wetlands, which in turn minimize pollutant loading to federally protected waters, including traditional navigable waters. The ability to perform these functions depends on several factors, including water residence time, proportion of the watershed occupied by wetlands, the mechanisms by which freshwater wetlands interact with sediments, nutrients, and the landscape, and the degree of hydrological connectivity between the wetland and the drainage system (Powers et al. 2012). This water quality nexus between GIWs and traditional navigable waters may, in fact, be strongest where hydrological connectivity is least obvious, as those GIWs connected to traditional navigable waters with slow flow paths may play a disproportionately large role in nutrient and pollutant retention (Racchetti et al. 2011, Powers et al. 2012, Marton et al. 2014). In this article, we reinforce the importance of GIWs in providing water quality improvement functions for waters of the United States, particularly by emphasizing their capacity for nutrient and sediment retention. Our goal is not to underplay the importance of more connected wetlands and those with direct adjacency to traditional navigable waters. Rather, we chose to focus on the importance of biogeochemical processes in GIWs, though wetlands across the connectivity continuum provide a vast array of ecosystem services. First we use a survey of the literature to show that GIWs host high rates of biogeochemical processing that, in turn, result in reduced nutrient and contaminant loading to downstream waters. Next we discuss the variety of ways they are hydrologically linked to traditional navigable waters, and the challenges (and opportunities) for getting realistic and reasonable inventories of GIWs. We then provide estimates of the rates of GIW loss in watersheds and use basic mass–balance principles to highlight how these losses result in greater pollutant delivery to traditional navigable waters. Here, we maintain the convention of defining GIWs as those wetlands that are completely surrounded by uplands while acknowledging that this construct dichotomously (and, therefore, inaccurately) classifies wetlands that exist along a connectivity gradient.

The biogeochemistry of GIWs

The condition of traditional navigable waters is inextricably linked to storage and exchange functions in the contributing watershed. Surfacewater features within the watershed, such as rivers, lakes, wetlands and aquifers are sinks to which water and solutes drain. As such, their collective functions are what control the quality and integrity of traditional navigable waters (Baron et al. 2002). Wetlands in particular improve water quality of downstream or other waters by retaining sediments, nutrients, and other dissolved pollutants, which they can do either by storing water for vegetation-mediated evapotranspiration or recharging aquifers, and therefore exporting no pollutants or by exporting water after sediment, nutrients, and other pollutants have been retained or transformed (Hansson et al. 2005). GIWs have disproportionately large biogeochemically reactive perimeters relative to their area, similar to headwater streams that make them more effective biogeochemical reactors per unit area of wetland, and greatly influence the delivery of reactive N, P, and other pollutants to downstream traditional navigable waters (Peterson et al. 2001). GIWs remove or retain pollutants at rates comparable to or higher than those of wetlands in which hydrologic connectivity to traditional navigable waters is more rapid (Fischer and Acreman 1999, Whigham and Jordan 2003). Even when rates in GIWs are lower than in more hydrologically connected systems, GIWs can be efficient at processing incoming nutrients. For example, Dierberg and Brezonik (1983) and Deghi and Ewel (1984) found that GIWs that received wastewater inputs removed more than 90% of incoming nutrients. Wolf and colleagues (2013) measured greater sedimentation and nutrient inputs from stream-connected wetlands relative to precipitation-fed wetlands, though they found comparable rates of nitrification, denitrification, and P mineralization between the two systems. Racchetti and colleagues (2011) found that denitrification rates in river-connected wetlands were up to two orders of magnitude greater than isolated wetlands. However, denitrification in the isolated wetlands was limited by available NO_3^-, with 60% to 100% of the denitrification supported by water-column NO_3^-, suggesting that GIWs were capable of high denitrification following NO_3^- inputs.

Despite the large amount of variability in efficacy, wetlands generally have greater nutrient retention and processing potential than surrounding uplands and thereby are better suited to buffer adjacent aquatic systems (Reddy et al. 1999). Although they receive fewer inputs of water, and from different sources, relative to floodplain wetlands, GIWs can have comparable organic C and N accumulation rates. For example, Craft and Chiang (2002) measured sediment accumulation of rates of 951 and 1289 grams (g) per square meter (m^2) per year in depressional and floodplain wetlands, respectively, in the Dougherty Plain in southwestern Georgia. The rates of organic C, total N, and total P accumulation did not significantly differ between the more isolated (i.e., depressional) and the more connected (i.e., floodplain) wetlands. Craft and Casey (2000) reported significantly greater organic and inorganic N and labile P in depressional wetlands than in the adjacent upland and ecotone systems. However, high sedimentation rates potentially could fill in depressional basins, thereby limiting the amount of time this service could be provided by a given wetland.

Furthermore, benefits to water quality improvement functions do not apply to natural wetlands only. Both natural and restored GIWs provide beneficial functions that improve water quality. Marton and colleagues (2014) found that restored depressional wetlands in the agricultural midwestern United States were able to sorb P and remove N via denitrification, although at rates lower than natural depressional wetlands. Furthermore, both processes in the restored wetlands exceeded rates measured in nearby agricultural soils

Table 1. Published rates of key geographically isolated wetland biogeochemical processes.

Target	Process	Rates (in grams per square meter per year)	Geography	References
Sediment	Storage	230	Virginia	Wolf et al. 2013
		120–950	Georgia	Craft and Casey 2000
		500–3600	North Dakota	Freeland et al. 1999
Carbon	Storage	21–70	Georgia	Craft and Casey 2000
		270	Canadian Prairies	Badiou et al. 2011
		317	Ohio	Bernal and Mitsch 2012
Phosphorus	Storage	0.01	Florida	Dierberg and Brezonik 1983
		0.01	North Dakota	Freeland et al. 1999
		0.08–0.25	Georgia	Craft and Casey 2000
		0.11–5.0	Florida	Dunne et al. 2007
	P Mineralization	0.10	Virginia	Wolf et al. 2013
Nitrogen	Denitrification	0.8–2.8	Florida	Dierberg and Brezonik 1983
		1.5–5.3	Georgia	Craft and Casey 2000
		0.25–28	Italy	Racchetti et al. 2011
	Nitrification	1.3	Virginia	Wolf et al. 2013
	N Mineralization	3.0	Virginia	Wolf et al. 2013

from which the wetlands were restored. Although restored GIWs can perform the functions of natural wetlands, it often takes a considerable amount of time for restored GIWs to reach their potential and to function at rates comparable to natural GIWs (Woltemade 2000, Badiou et al. 2011, Marton et al. 2014).

There is substantial variability in biogeochemical processing rates of GIWs

There is substantial variability in the biogeochemical processing rates of GIWs as a function of their spatial and temporal properties, as well as their geographic location (table 1). Spatial properties include the size, shape and position of a GIW within the watershed (table 2), whereas temporal properties include the changes in environmental factors such as soil moisture, temperature, and reduction–oxidation (redox) potential, all of which respond to changing meteorological and hydrological conditions. For example, reduced hydrologic connectivity results in greater residence times, thereby allowing increased biogeochemical processing (e.g., denitrification, sedimentation) (Woltemade 2000, Powers et al. 2012). Furthermore, greater shape complexity and a higher perimeter to area ratio leads to greater overall biogeochemical processing through more rapid changes in soil moisture and redox potential (Hefting et al. 2004, Ligi et al. 2013).

Wetland size is likely to be an important control on nutrient retention. Multiple studies have shown that small streams and lakes are more efficient at nutrient retention or removal than are larger rivers and lakes. For example, dissolved organic C concentrations, rates of organic C sequestration in sediments, surface carbon dioxide and methane concentrations, and losses to the atmosphere have declined sharply with increasing lake size (Bastviken et al. 2004, Downing 2010). One study found that organic C burial in lakes of various sizes ranged from 17 kilograms (kg) of C per m^2 per year to a low of 0.15 kg C per m^2 per year, with significantly greater rates of burial in smaller lakes (Downing et al. 2008). Although these studies are not specific to wetlands, and comparable studies in wetlands are rare, similar relationships are expected to hold in wetlands owing to the principle that the large amount of reactive area relative to the ecosystem's size results in more incoming nutrients being processed, despite the fact the most wetland assessments were formed with emphasis and importance placed on larger wetlands and total cumulative area (Adamus et al. 1991). One particularly suggestive meta-analysis of 418 observations from 186 wetland sites worldwide that revealed a negative correlation between wetland size and water quality (Ghermandi et al. 2010), meaning that smaller wetlands held and exported cleaner water relative to larger wetlands. The results from this meta-analysis could also have been influenced by topography of the sites, underlying geology, and frequency of inundation, all of which could influence water retention and reaction time, and therefore changes in wetland surface and ground water quality (Rains et al. 2008, Rains 2011, Nilsson et al. 2013). Interestingly, early wetland evaluation procedures and techniques focused on cumulative wetland coverage and larger wetlands that could potentially provide a more diverse set of ecosystem services (Adamus et al. 1991).

The length and shape of the edge of GIWs are important as they play a critical role in controlling biogeochemical process rates. A greater diversity of biogeochemical processes is found within the frequently wetted and dried wetland edge relative to the uplands and wetland interior, with the

Table 2. Morphormetric factors influencing geographically isolated wetlands' (GIWs) water quality improvement potential.

Factor	Effect	References
Size (surface area) of wetland	Small wetlands are shown to be more effective at phosphorus retention, whereas large wetlands are more effective at retaining nitrogen because of an increase in water retention time.	Hansson 2005, Ghermandi et al. 2010, Downing et al. 2010
Shape of wetland edge (i.e., the ratio of shoreline to surface area)	Rates of water loss from GIWs vary with the length of shoreline per unit area. This water loss causes wetland water levels to fluctuate, modifying the size of the transitional zone at wetland edges and the biogeochemical processes that occur in these transitional zones.	Hefting et al. 2004, Ligi et al. 2013
Shape of wetland profile (i.e., convex vs. concave)	Frequency of wetting and drying cycles at the edge of GIWs is greater in concave profiles compared with convex ones. Wetting and drying of sediments increases the leaching and desorption of phosphorus from sediment organic matter, therefore releasing phosphorus in hydrological export. Wetting and drying of sediments also increases denitrification, therefore removing nitrate in hydrological export.	Galloway and Branfireun 2004, Creed et al. 2013
Position within watershed	GIWs at different positions have different effects on water quality – upper reach GIWs are effective at removing sediment, midreach GIWs are effective at retaining phosphorus, and lower reach GIWs are hydrologic flow attenuators and are effective at removing nitrogen.	Cohen and Brown 2006, Denver et al. 2014
Slope of watershed	GIWs located in watersheds with steep slopes and shallow soils are more likely to receive a larger input of nutrients from surface runoff and therefore have higher water quality improvement potential.	Andersson and Nyberg 2008, Denver et al. 2014
Abundance within watershed	Increasing number of GIWs within a watershed increases the potential to remove nutrients. A minimum of 2 to 7% of wetlands per watershed area has been found to improve water quality.	Mitsch and Gosselink, 2000, Verhoeven et al. 2006

diversity of processes being strongly influenced by soil moisture distribution (Creed et al. 2013). Furthermore, a larger wetland perimeter to area ratio is often correlated directly to biogeochemical processing, that is, a greater perimeter to area ratio means that more GIW area is exposed and is subjected to periodic wet–dry cycles contributing to coupled nitrification–denitrification (Hefting et al. 2004). Figure 2 shows the conceptual relationship between wetland edge complexity, hydrologic connectivity, and N and P processing potential. For example, wetlands with a complex edge (greater perimeter:area ratio) have a greater amount of biogeochemically reactive surfaces. Minimal hydrologic connectivity, in systems such as wet meadows, will have a greater water residence time allowing for increased nutrient processing. On the opposite end of the spectrum, hydrologically connected wetlands will have minimized residence time which will reduce nutrient retention and processing, and wetlands with a lower perimeter:area ratio will have less reactive surfaces. This is not to discount the valuable ecosystem services provided by adjacent wetlands or those with greater connectivity to traditional navigable waters. Rather, this is to emphasize that GIWs are not insignificant landscape components with respect to nutrient processing and water quality improvement functions.

The abundance and arrangement of GIWs in a watershed play a significant role in their functionality. A modeling study in low-relief watersheds characteristic of the southeastern United States coastal plain revealed that small headwater GIWs effectively removed sediment, medium reach GIWs retained P, whereas large wetlands in coastal or bottomland areas primarily served as hydrologic flow attenuators and were effective at denitrification, though water quality improvement was greatest when all wetland sizes were present (Cohen and Brown 2007). Although their conditions (i.e., redox, available C) are conducive for denitrification, the nitrate removal efficiency of an individual GIW is dependent on its position in the landscape that controls its ability to intercept nitrate-laden water (Woltmeade 2000, Denver et al. 2014, Marton et al. 2014). However, even when GIWs do not intercept NO_3^- from groundwater, they still improve water quality downstream by being situated to become hydrologically connected in the event of watershed development and landscape modification. For example, perched-precipitation wetlands in Alaska have much higher rates of groundwater recharge than other wetlands in the surrounding area (Rains 2011), and the losses of these system would likely divert precipitation and snowmelt elsewhere, thereby creating surfacewater connections among other wetlands. Furthermore, vernal pools in California maintain a degree of surfacewater connectivity that exist throughout the year (Rains et al. 2008) and drainage and ditching of these sites would alter the hydraulic head dynamics and modify connectivity throughout the wetland complex and surrounding areas. Collectively, GIWs throughout the landscape can provide significant

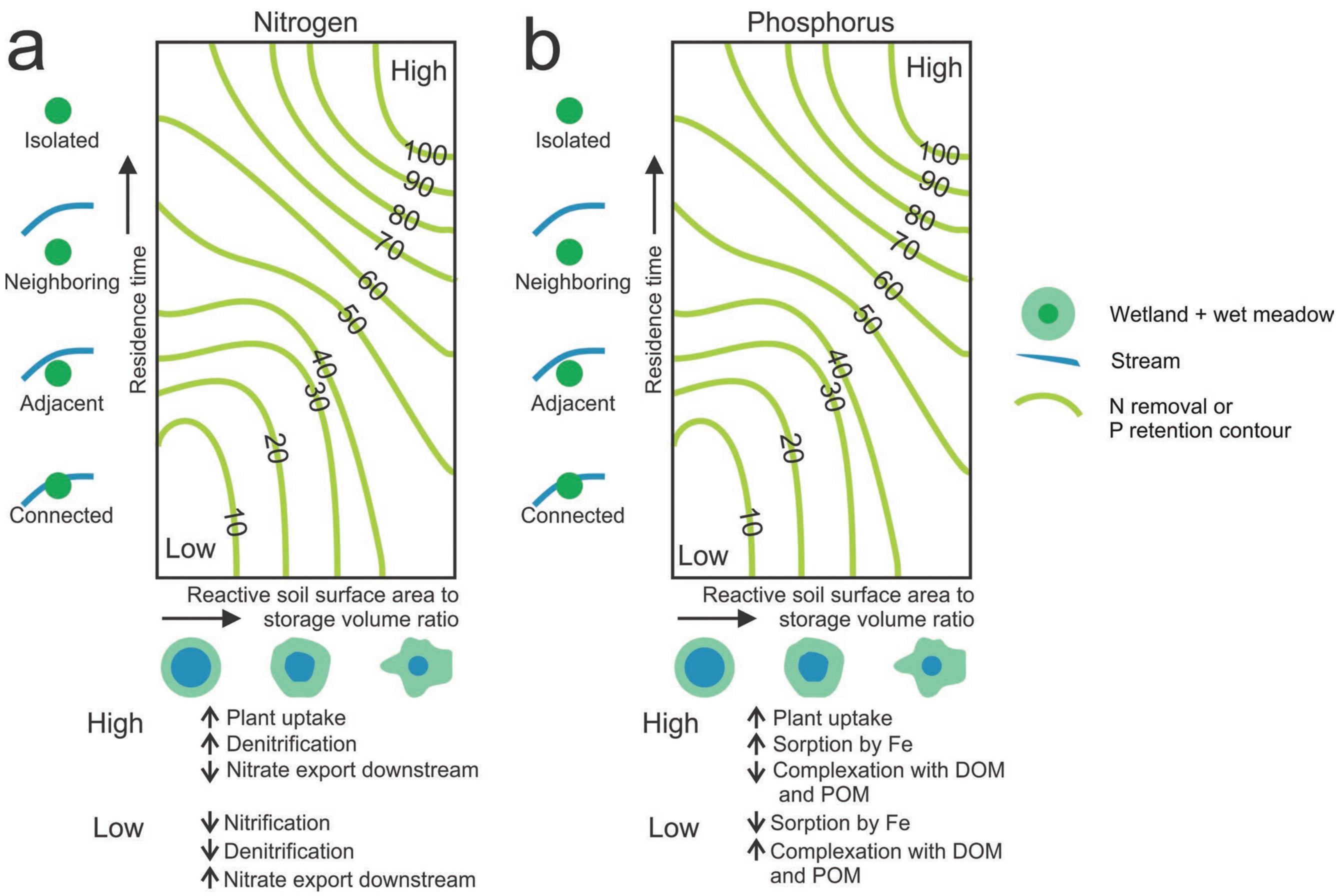

Figure 2. Conceptual relationship between the magnitude of key nitrogen (N) and phosphorus (P) biogeochemical processes (e.g., nitrification, denitrification, sorption by iron [Fe] interaction with dissolved and particulate organic matter [DOM, POM, respectively], plant uptake) and the perimeter: the area ratio and adjacency of geographically isolated wetlands. The contours represent the theoretical percentages of incoming N and P removed or retained by the wetland.

water quality improvement functions, and the value of these systems in aggregate exceeds the value of individual GIWs. Because of this aggregate value, an individual GIW's membership in a wetland complex could theoretically constitute a significant nexus with traditional navigable waters.

With climatic variability and an increase in climatic extremes, an important aspect of water quality improvement functions of wetlands lies in understanding the temporal dynamics of the nutrient retention potential (Sass et al. 2008). For example, GIWs might be intermittently connected to the stream network during high flow conditions and transport significant loads of nutrients or contaminants. Time series data representing periods of relatively high and low inflow indicate that wetland performance is highly sensitive to water retention time with nutrient retention and transformation increasing with retention time (Woltemade 2000). Temporal dynamics in water flow also affect the redox conditions that regulate nutrient retention and release patterns. Phosphorus retention and release are affected by intermittent wetting and drying cycles, because frequent wet–dry cycles increase the extractable soil P pool and thereby inhibit P retention capacity in GIWs (Bhadha and Jawitz 2010). In contrast, rewetting of dried soils strongly stimulated denitrification, but did not affect nitrogen mineralization rates (Venterink et al. 2002).

GIWs can reach their biogeochemical saturation capacity

A GIW's ability to retain sediments, nutrients, and other pollutants is dependent at least partially on the degree to which it is open to hydrologic, biogeochemical, and biological fluxes with surrounding agricultural, forest, and urban landscapes. Human development has reduced the number of GIWs while simultaneously creating greater need for the ecosystem services they provide, thereby strengthening the value of the remaining GIWs. However, increases in nutrient flows from the surrounding landscape may exceed a GIW's ability to retain and transform nutrients thereby altering the system's ecology (Fischer and Acreman 1999). The vulnerability of natural wetlands to excess nutrient loading varies depending on their antecedent nutrient regime. For example, wetlands that are nutrient poor (oligotrophic) have smaller critical loads in comparison to nutrient-rich (eutrophic) wetlands. Excess nutrient flows may cause GIWs to become a net source of greenhouse gases to the atmosphere or a net source of nutrients to the downstream ecosystem (Hefting et al. 2013). For instance, Bhadha and Jawitz (2010) reported that internal loading (i.e., P in the soils within the GIW) accounted for 18% of the P “entering” two studied historically isolated wetlands in southern Florida; a shallow ditch draining these wetlands, which eventually connected to Lake Okeechobee, accounted for 49% of the P outflow, therefore creating a potential P source.

Connectivity between GIWs and navigable waters

Connections between GIWs and other aquatic and terrestrial systems are manifest as the transfer of materials, energy, or organisms. Geographically isolated wetlands can be connected to jurisdictional waters via either groundwater linkages or surfacewater fill and spill mechanisms. Groundwater connectivity between GIWs occurs when the wetlands are connected to each other via a regional groundwater flow system and is dependent on the terrain slope, hydraulic conductivity, and composition of the medium through which the water will flow (Mitsch and Gosselink 2000, Rains et al. 2008, Rains 2011, Nilsson et al. 2013). Nilsson and colleagues (2013) found that the majority of GIWs in west-central Florida served as groundwater recharge zones, suggesting a hydrologic connection despite the lack of an apparent surfacewater connection, whereas Rains (2011) found similar results in ponds in closed-basin depressions in Alaska. Surfacewater fill and spill mechanisms occur in depressional landscapes during wet years where wetlands fill with water. Eventually, some of these wetlands fill beyond their storage capacity and overflow, forming intermittent hydrologic connections between the overflowing wetland and surrounding topographically lower wetlands or downstream waters (Liebowitz and Vining 2003). These ephemeral hydrological connections are less obvious than perennial, more permanent, surfacewater connections but still facilitate the movement of nutrients and water from a wetland to the adjacent stream network.

Freshwater wetlands have a continuum of connectivity to the drainage network—from rarely connected along slow hydrological flowpaths (i.e., “isolated”) to permanently connected along faster ones (Liebowitz 2003). Where a wetland exists along this continuum defines the degree to which it acts as a biogeochemical processor to improve water quality. For GIWs to have a measurable biogeochemical effect on downstream waters, a direct surfacewater connection is not required. In fact, a hydrologic connection may actually lessen the water quality improvement functions by reducing nutrient retention and transformation capacity by reducing the water residence time within the wetland (Powers et al. 2012). This lower residence time would lead to greater nutrient and pollutant export to downstream waters. Furthermore, differences in hydraulic heads of GIWs and the surrounding terrestrial landscape can influence groundwater discharge–recharge dynamics, subsurface solute transport, and water–rock interactions (Rains et al. 2008, Rains 2011, Nilsson et al. 2013).

The intermittent hydrological connectivity of GIWs, coupled with their high potential for biogeochemical processing, can have large cascading effects on the landscape by reducing nutrient delivery downstream, similar to what is seen in stream networks (Alexander et al. 2007, Freeman et al. 2007). Modifications to GIWs, either through loss or alterations in connectivity to uplands and downstream waters, are likely to lead to fundamental changes in biogeochemical process rates, and ultimately, the ability of GIWs to maintain the biological and chemical integrity of jurisdictional waters at local and regional scales (Woltemade 2000, Baron et al. 2002, Marton et al. 2014). GIWs can hold water for several months each year depending on inputs, outputs, and underlying geology, and loss of these systems through either ditching, draining, or filling in can alter groundwater recharge dynamics, water–rock interactions, and solute transport (Rains et al. 2008, Rains 2011, Nilsson et al. 2013). Though there is significant variation in rates across wetland types and ecoregions, wetlands are, on balance, more biogeochemically reactive than uplands (e.g., Chessman et al. 2010). Therefore, preserving all wetlands within a watershed is the most defensible way to maintain the integrity of jurisdictional waters, at least until we better understand the significance of distance, retention, ecoregion, and wetland type on biogeochemical processes and connectivity (Whigham and Jordan 2003).

GIW inventories: We can’t manage it if we can’t see it

Mapping the extent and connectivity of isolated wetlands at multiple spatial and temporal scales is a first step toward better understanding the aggregate biogeochemical benefits to jurisdictional waters. Efforts to map GIWs are typically confounded by multiple factors, including their size, the presence of obscuring vegetation, and data age (Ozesmi and Bauer 2002, Adam et al. 2010). Because of their small size, many isolated wetlands are not mapped by national (e.g., the National Wetlands Inventory [NWI]) or state or provincial monitoring programs. In the United States, the NWI minimum mapping unit ranges approximately from 0.4 to 1.2 hectares (ha; Tiner 1997), a size too large to map the majority of isolated wetlands (e.g., Burne and Lathrop 2008). Similarly, the Canadian wetland-mapping program has a minimum mapping unit of approximately 1.0 ha (Milton et al. 2003, as cited in Burne and Lathrop 2008).

Nevertheless, researchers are starting to map the extent of potential isolated wetlands in certain areas of the United States For example, Martin and colleagues (2012) used digital elevation models and digital rasters of US Geological Survey topographic maps combined with NWI data to map potential GIWs in a physiographic region of Georgia, finding that almost 20,000 more ha of GIW were identified when using integrated data sources than when using the NWI alone. Statewide mapping of potential woodland vernal pools have been conducted using aerial photography in Massachusetts (Burne 2001) and New Jersey (Lathrop et al. 2005). In Massachusetts, over 29,000 potential vernal pools have been identified (*http://wsgw.mass.gov/data/gispub/shape/state/pvp.exe*) whereas Lathrop and colleagues (2005) identified over 13,000 potential vernal pools in New Jersey. Interstate mapping by Lane and colleagues (2012) found almost 813,200 potentially isolated wetlands covering an area of 1.2 million ha across an eight-state region of the southeastern United States

Water quality improvement functions at the landscape scale

Using estimates of wetland loss from different parts of the United States and the ranges of reported biogeochemical process rates (table 1), we estimate changes in sediment trapping, C sequestration, N and P storage, and nitrogen lost via denitrification by multiplying estimated GIW losses by published areal rates. To estimate changes, we multiplied wetland loss for multiple regions by the minimum and maximum reported process rate values reported in table 1, which provided an estimated range. Each minimum and maximum range of a given process rate (e.g., 21–270 g per m^2 per year of C sequestration) was applied to wetland losses from all regions. Although these estimations are oversimplified and ignore spatial and temporal variability they offer an empirical basis for enumerating the watershed-scale biogeochemical influence of GIWs.

The United States has lost approximately half of the wetlands present before European settlement (Dahl 2011), but wetland loss is not uniform across wetland types (Tiner 2003) as areas formerly rich in GIWs have wetland losses that exceed 90%, and losses continue (Dahl 2011). For example, the North American Prairie Pothole Region is a physiographic region that covers almost 800,000 square kilometers (km^2) straddling Canada and the United States. Historically, GIWs across the prairie pothole region (i.e., prairie potholes) were estimated to cover almost 8 million ha (80,000 km^2; Mitsch and Gosselink 2000); however, more than half of these wetlands have been lost. In the Canadian portion of this region it has been estimated that upward of 70% of the wetlands may have been destroyed (Mitsch and Gosselink 2000), whereas losses can reach closer to 90% US Prairie Pothole states (Dahl 1990).

We estimate that the loss of approximately 4 million ha of GIWs in the Prairie Pothole Region has resulted in an increase of between 5 and 140 teragrams (Tg) per year (1 Tg = 10^{12} g) of sediment entering surface waters and decreases of 0.84–13 Tg per year C sequestration, 0.00040–0.20 Tg per year P storage, and 0.032–0.21 Tg per year denitrification potential. Estimates are admittedly coarse and rates from the literature imperfectly represent the 4 million ha of lost wetlands. But, lost wetland area and high wetland biogeochemical retention function are two patterns that are documented beyond dispute. Our calculations here merge these two patterns to reveal that even rudimentary estimates of lost potential for water-quality protection are huge. Combined, the lost functions provided by GIWs are likely to contribute to further eutrophication of nearby aquatic systems. The trend of wetland loss is continuing if not escalating. Johnston (2013) reported wetland losses of 5200–6200 ha per year from 1980 and 2000 in the Prairie Pothole Region in North and South Dakota alone which led annual decreases of 0.0062–0.22 Tg in sediment trapping and 0.0011–0.020 Tg in C sequestration on the basis of reported sedimentation rates ranging from 120 g per m^2 per year (Craft and Casey 2000) to 3600 g per m^2 per year (table 1; Freeland et al. 1999), which were then applied to losses reported by Johnston (2013). This indicates that despite conservation efforts, wetlands in the region continue to be converted, ditched, or drained.

However, the situation is hopeful in other parts of the United States. Applying a measure of landscape development intensity within 100 m of each of more than 800,000 GIWs in the southeastern United States, Lane and colleagues (2012) found that approximately 50% of the wetlands would be expected to be in natural or minimally impacted condition. Because they are still intact and have not all been lost or converted, the GIWs mapped by Lane and colleagues (2012) represent significant sediment, organic C, and total N and P sinks. They mapped approximately 1.2 million ha of potentially GIWs, which have the potential to sequester 0.25–3.8 Tg of organic C each year. Furthermore, they can trap 1.4–42.7 Tg of sediment, store 0.00012–0.059 Tg of P, and denitrify 0.0095–0.063 Tg of N. Similarly, the Dougherty Plain in southeastern Georgia has approximately 43,000 ha of mapped GIWs representing cumulative sediment, C, N, and P storage rates of 0.40, 0.023, 0.0020, and 3.4×10^{-5} Tg per year, respectively.

Shifting the burden of proof

Since the SWANCC and Rapanos rulings, the challenge associated with protecting GIWs in part relates to determining whether a wetland had a substantial effect on traditional navigable waters (Downing et al. 2003, Leibowitz 2003). Many GIWs will appear, upon cursory regulatory examination, to have limited connection (e.g., via surface hydrological flowpaths) with a traditional navigable water, despite the fact they are critical in regulating hydrologic gradients within the watershed, and in improving downstream water quality through the interception, transformation, and storage of water quality pollutants (Baron et al. 2002, Racchetti et al. 2011). Currently, whether a wetland exhibits a significant nexus has to be demonstrated on a case-by-case basis unless the wetland is adjacent to a traditional navigable water or has a continuous surfacewater connection. This case-by-case approach undoubtedly leaves many GIWs outside regulatory protection. The lack of an obvious surface hydrological connection between GIWs and jurisdictional waters should not be construed as a lack of quantifiable effect on downstream waters. GIWs significantly protect navigable waters by retaining and removing nutrients and pollutants within the watershed above the downstream water; often, its retention service relies on *limited* hydrological connectivity between GIWs and traditional navigable waters.

On 21 April 2014, the US Environmental Protection Agency and the US Army Corps of Engineers, in response to the SWANCC and Rapanos decisions, proposed clarifications to the CWA that would affect which types of waters would be considered jurisdictional under the Act (see US Army Corps of Engineers and US Environmental Protection Agency "Definition of Waters of the United States under the Clean Water Act," CFR Docket ID No. 79 FR 22188). The clarifications (as of this writing) include reasserting CWA jurisdiction to wetlands adjacent to (i.e., bordering, contiguous, and neighboring) jurisdictional lakes, rivers, and streams. Furthermore, wetlands that are *other waters*, or those that are nonadjacent to waters of the United States, will have jurisdiction assessed on a case-by-case basis. The proposed regulations also allow the evaluation of other waters either alone or in combination with other similarly situated waters in the region to determine whether they significantly affect the chemical, physical, or biological integrity of traditional navigable waters, interstate waters, or the territorial seas. Other waters are similarly situated when they perform similar functions and are located sufficiently close together or sufficiently close to a water of the United States. The fact that CWA jurisdiction may be extended to GIWs on the basis of a watershed assessment of connectivity and GIW effect on downstream waters suggests that watersheds in regions with large amounts of functioning GIWs (such as the prairie pothole region of the Upper Midwest and Canada, California vernal pools, Carolina bays and cypress ponds of the southeastern United States and other GIWs) may gain CWA protections under these new rules should they be finalized. We conclude that that it is important to maintain the "biogeochemical reactors" on the landscapes – the GIWs embedded in landscapes and performing P sorption, denitrification, C mineralization, and a host of other biogeochemical functions. As all wetlands to some degree perform these functions, we suggest that the burden of proof could be shifted from demonstrating a significant effect on downstream waters to assuming that every individual GIW is critical for the protection of aquatic systems within a given landscape and across scales of connectivity until proven otherwise. This shift in the burden of proof, based on the sound science of wetland biogeochemistry and systems ecology, would potentially result in greater wetland area falling under CWA jurisdiction, and ultimately, improved quality and physical, chemical, and biological integrity of the nation's waters, the ultimate goal of the CWA.

Acknowledgments

The idea for this article originated during the "Geographically Isolated Wetlands Research Workshop," convened and cohosted by the US Environmental Protection Agency Office of Research and Development and the Joseph W. Jones Ecological Research Center and held in Newton, Georgia, on 18–21 November 2013. DBL was supported in part by the National Science Foundation under grant no. 1251653. Ellen D'Amico of the Dynamac Corporation, a contractor to the US Environmental Protection Agency Office of Research and Development under contract no. EP-D-06-096, provided support in developing figure 1. This article has been reviewed in accordance with the US Environmental Protection Agency's peer and administrative review policies and approved for publication. Mention of trade names or commercial products does not constitute endorsement or recommendation for use. Statements in this publication reflect the authors' professional views and opinions and should not be construed to represent any determination or policy of the US Environmental Protection Agency.

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