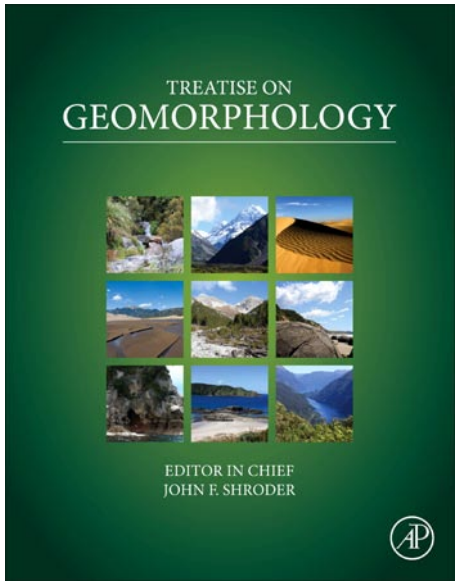
Provided for non-commercial research and educational use only.
Not for reproduction, distribution or commercial use.

This chapter was originally published in the *Treatise on Geomorphology*, the copy attached is provided by Elsevier for the author's benefit and for the benefit of the author's institution, for non-commercial research and educational use. This includes without limitation use in instruction at your institution, distribution to specific colleagues, and providing a copy to your institution's administrator.



All other uses, reproduction and distribution, including without limitation commercial reprints, selling or licensing copies or access, or posting on open internet sites, your personal or institution's website or repository, are prohibited. For exceptions, permission may be sought for such use through Elsevier's permissions site at:

http://www.elsevier.com/locate/permissionusematerial

Noe G.B. (2013) Interactions among Hydrogeomorphology, Vegetation, and Nutrient Biogeochemistry in Floodplain Ecosystems. In: John F. Shroder (ed.) Treatise on Geomorphology, Volume 12, pp. 307-321. San Diego: Academic Press.

© 2013 Elsevier Inc. All rights reserved.

Author's personal copy

12.21 Interactions among Hydrogeomorphology, Vegetation, and Nutrient Biogeochemistry in Floodplain Ecosystems

GB Noe, US Geological Survey, Reston, VA, USA

Published by Elsevier Inc.

12.21.1	**Floodplains and Their Essential Interactive Processes**	308
12.21.2	**The Template of Hydrogeomorphology in Floodplains**	308
12.21.2.1	History of Hydrogeomorphic Concepts in Rivers	308
12.21.2.2	Hydrogeomorphic Controls on Floodplain Ecosystems	309
12.21.3	**Controls of Vegetation in Floodplains**	310
12.21.3.1	Hydrogeomorphic Controls of Vegetation	310
12.21.3.2	Influence of Vegetation on Hydrogeomorphology	311
12.21.3.3	Biogeochemical Controls on Vegetation	311
12.21.3.4	Other Biota	311
12.21.4	**Controls of Nutrient Biogeochemistry in Floodplains**	312
12.21.4.1	Hydrogeomorphic Controls of Nutrient Biogeochemistry	312
12.21.4.2	Influence of Vegetation on Nutrient Biogeochemistry	314
12.21.4.3	Biogeochemical Controls on Hydrogeomorphology?	314
12.21.5	**Case Studies**	314
12.21.5.1	Hummock/Hollow Geomorphology in Peatlands and Floodplains	314
12.21.5.2	Coastal Plain Floodplains	315
12.21.5.3	Montane Floodplains	316
12.21.5.4	Desert Streams	316
12.21.6	**Conclusions**	317
References		317

Glossary

Ecogeomorphology The integration of hydrology, fluvial geomorphology and ecology to study riverine systems (Thoms and Parsons 2002).

Floodplain A unit of riparian systems that occur on alluvial surfaces and are sometimes inundated with flooding surface water.

Hydrogeomorphology The integrated study of hydrology and geomorphology.

Hydrologic connectivity: The exchange of water between floodplain and river channel.

Riparian It is related to the interface of aquatic and upland ecosystems.

Abstract

Hydrogeomorphic, vegetative, and biogeochemical processes interact in floodplains resulting in great complexity that provides opportunities to better understand linkages among physical and biological processes in ecosystems. Floodplains and their associated river systems are structured by four-dimensional gradients of hydrogeomorphology: longitudinal, lateral, vertical, and temporal components. These four dimensions create dynamic hydrologic and geomorphologic mosaics that have a large imprint on the vegetation and nutrient biogeochemistry of floodplains. Plant physiology, population dynamics, community structure, and productivity are all very responsive to floodplain hydrogeomorphology. The strength of this relationship between vegetation and hydrogeomorphology is evident in the use of vegetation as an indicator of hydrogeomorphic processes. However, vegetation also influences hydrogeomorphology by modifying hydraulics and sediment entrainment and deposition that typically stabilize geomorphic patterns. Nitrogen and phosphorus biogeochemistry commonly influence plant productivity and community composition, although productivity is not limited by nutrient availability in all floodplains. Conversely, vegetation influences nutrient biogeochemistry through direct uptake and storage as well as production of organic matter that regulates microbial biogeochemical processes. The biogeochemistries of nitrogen and phosphorus cycling are very sensitive to spatial and temporal variation in hydrogeomorphology, in particular floodplain wetness and sedimentation. The least-studied interaction is the direct effect of biogeochemistry on hydrogeomorphology, but the control of nutrient availability over organic matter decomposition and thus soil permeability and elevation is likely important. Biogeochemistry also

Noe, G.B., 2013. Interactions among hydrogeomorphology, vegetation, and nutrient biogeochemistry in floodplain ecosystems. In: Shroder, J. (Editor in Chief), Butler, D.R., Hupp, C.R. (Eds.), Treatise on Geomorphology. Academic Press, San Diego, CA, vol. 12, Ecogeomorphology, pp. 307–321.

Author's personal copy

has the more documented but indirect control of hydrogeomorphology through regulation of plant biomass. In summary, the defining characteristics of floodplain ecosystems are determined by the many interactions among physical and biological processes. Conservation and restoration of the valuable ecosystem services that floodplains provide depend on improved understanding and predictive models of interactive system controls and behavior.

12.21.1 Floodplains and Their Essential Interactive Processes

The value of floodplains to society highlights the importance of understanding the controls on their functions. Frequently cited ecosystem services provided by floodplains include flood reduction, water-quality improvement, habitat, food production, and recreation. Costanza et al. (1997) found that the value of the ecosystem services provided by floodplains was among the highest of any ecosystem on the Earth, providing about 10% of total global ecosystem services that occur on a mere 0.3% of the Earth's surface. However, most floodplains are impacted by human modifications to hydrology and geomorphology (Hupp et al., 2009) and the rest are sensitive to future human impacts (Tockner and Stanford, 2002). Ecological restoration of floodplains through the reintroduction of flooding has the potential to increase the function, resiliency, and values of impacted floodplains (Sparks, 1995; Galat et al., 1998). Future impacts of altered precipitation and sea level due to climate change, as well as human water use and land-cover change, on river discharge, flood characteristics, and salinity make an improved understanding and predictive ability even more essential for successful management of floodplain ecosystems. Effective management of floodplain ecosystems to conserve, restore, or augment their ecosystem services requires thorough understanding of the processes that control system dynamics.

Hydrologic and geomorphic ('hydrogeomorphic' for short), vegetative, and nutrient biogeochemical processes interact to control the dynamics of floodplain ecosystems (**Figure 1**). These essential interactive processes and their feedbacks define riparian systems (Naiman and Décamps, 1997) and have been termed 'functional ecogeomorphology' (Fisher et al., 2007). The direct and indirect feedbacks between and among these three broadly aggregated processes are, for the most part, well documented but make it difficult to study and to predict pattern and process in floodplains. These processes and feedbacks also are at the core of the valuable ecosystem services that floodplains provide to society. Successful river restoration depends on restoring the geomorphic complexity of floodplains (Petts et al., 1992) through reintroduction of hydrogeomorphic processes (Beechie et al., 2010) – including the direct effects as well as the indirect interactions of hydrogeomorphic, vegetative, and biogeochemical processes. The direct effect of nutrient biogeochemistry on the hydrogeomorphology of floodplains is perhaps less obvious than the other interactions, but specific examples and concepts exist to support this feedback. To begin discussion of these fundamental processes and interactions, the four-dimensional (4D) nature of riverine hydrogeomorphology will first be described.

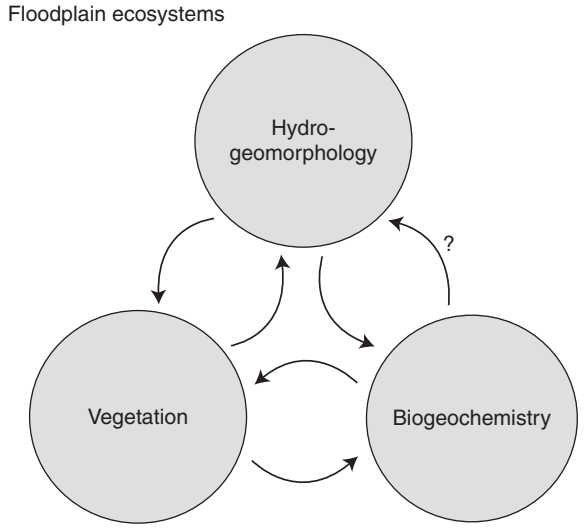


Figure 1 The interaction of hydrogeomorphic, vegetative, and biogeochemical processes in floodplain ecosystems. Of these interactions, the direct effects of biogeochemistry on hydrogeomorphology are the least understood.

12.21.2 The Template of Hydrogeomorphology in Floodplains

12.21.2.1 History of Hydrogeomorphic Concepts in Rivers

Hydrogeomorphic concepts in river science have progressed through time to include the four dimensions of riverine hydrogeomorphology, namely longitudinal, lateral, vertical, and temporal gradients (**Figure 2**) that are the fundamental controls of ecosystem structure and processes in streams and rivers (Ward, 1989a; Poole, 2010). Leopold et al. (1964) provided the foundational conceptual and quantitative synthesis of the interaction of hydrology and geomorphology in fluvial systems. Subsequent concepts in river hydrogeomorphology identified upland control (Hynes, 1975), then added longitudinal (Vannote et al., 1980), lateral (Junk et al., 1989), vertical (Stanford and Ward, 1993), and temporal dimensionality (Amoros et al., 1987). The dendritic network approach has also been applied to stream landscapes (Fisher, 1997).

The longitudinal model of ecological structure and function in rivers, the river continuum concept (RCC), predicts continuous gradients in physical conditions, biotic communities, and ecological processes from headwaters to the mouth (Vannote et al., 1980). By constrast, the flood pulse concept is a lateral model that predicts that the hydrogeomorphic characteristics of flood pulses control both vegetative and animal communities and biogeochemical cycles in river–floodplain systems (Junk et al., 1989). The RCC does not include the lateral dimension of rivers that is critically important to ecosystem dynamics particularly in large rivers (Ward, 1989b). Thus, the longitudinal RCC does not suffice to explain biotic

Author's personal copy

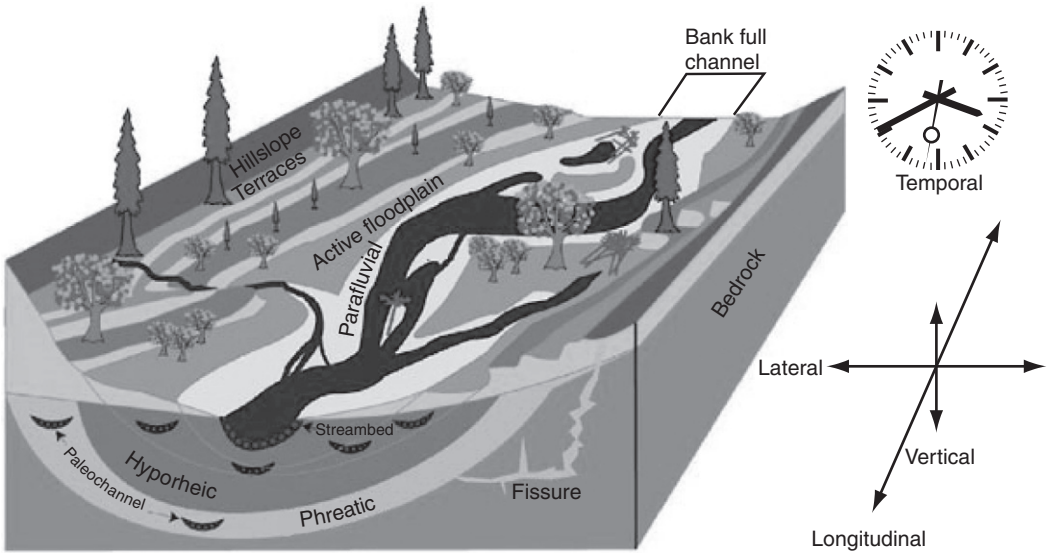


Figure 2 Example of an alluvial river–floodplain corridor showing the longitudinal, vertical, and lateral dimensions that change through time to influence the interactions of hydrogeomorphology, vegetation, and nutrient biogeochemistry. Modified with permission from National Research Council, 2002. Riparian Areas. National Academy Press, Washington, DC.

assemblages or biogeochemistry of rivers with lateral floodplain connections (Sedell et al., 1989). The importance of lateral connectivity is demonstrated by human modifications to rivers, including dams, flood protection levees, and channelization, that minimize lateral connectivity and, as a result, diminish the ecosystem services of river–floodplain systems (Ward and Stanford, 1995; Hupp et al., 2009).

Vertical hyporheic exchange is an important hydrologic dimension affecting river–floodplain systems. Hyporheic exchange can affect nutrient transformations and subsidies in rivers and floodplains (Holmes et al., 1994). This groundwater exchange can also influence the transport of water, heat, and other material and structures riverine vegetation and biogeochemistry gradients (Stanford and Ward, 1993; Valett et al., 1994; Stanford et al., 2005). In addition, lateral hyporheic exchange between channel and floodplain is important to system hydrology and biogeochemistry (Triska et al., 1993).

Temporal variation in longitudinal, lateral, and vertical transport processes defines the fourth dimension in river systems. The timing of hydrologic variation, known as the flow regime, is a characteristic that distinguishes among river ecosystems (Poff et al., 1997). Natural temporal variation can generate large changes in the extent and ecological functions of rivers (Stanley et al., 1997). Human modifications to the flow regime impact river ecosystem dynamics with the response depending on local geomorphic setting (Poff et al., 2010). Bornette et al. (2008) added the temporal aspect of hydrogeomorphic regimes to a model of fluvial hydrosystems in order to emphasize patch dynamics and biotic diversity following physical disturbances.

Thus, river ecology is controlled by the 4D nature of the hydrogeomorphic characteristics of the system with its inherent spatial complexity and patchiness (Thorp et al., 2006). For example, the telescoping ecosystem model is a 4D concept that was developed to describe temporal variation in longitudinal processes, such as nutrient spiraling, due to stream interaction with vertical and lateral subsystems (Fisher et al., 1998). The four dimensions also control river-system biodiversity (Ward, 1998). For any dimension, hierarchy theory and consideration of scale have been used to understand river-system dynamics and show how hydrogeomorphic processes that occur at a larger scale influence ecologic processes at the focal scale (ecogeomorphology; Thoms and Parsons, 2002).

12.21.2.2 Hydrogeomorphic Controls on Floodplain Ecosystems

Riparian systems are defined as the interface of aquatic ecosystems and uplands (Gregory et al., 1991; National Research Council, 2002; Naiman et al., 2005). Floodplains are a specific unit of riparian systems that occur on alluvial surfaces and are sometimes inundated with flooding surface water. By definition, riparian and floodplain systems are functions of their geomorphic position in landscapes and the temporal and spatial variation in the presence and actions of water (Hupp and Osterkamp, 1996). Consequentially, floodplains are ideal systems to study how hydrogeomorphic processes control and interact with ecological processes (Steiger et al., 2005; Tockner et al., 2010). Not all riparian systems, including floodplains, are wetlands; wetlands are characterized by the presence of surface water or shallow subsurface water, hydric soils, and vegetation adapted to flooding (Mitsch and Gosselink, 1993). Some floodplain locations may lack these traits and be either wetter (e.g., oxbow lakes) or drier (e.g., floodplains in urbanized catchments) than a wetland. However, the influence of water is fundamental to wetland ecosystems; therefore, wetlands will be used as an analog for understanding floodplains. It is very well known that hydrology and geomorphology are responsible for regulating many ecosystem functions of wetlands in general (Brinson, 1993) and floodplains in particular (Hupp, 2000). Specifically, geomorphic setting, water source,

Author's personal copy

and hydrodynamics classify wetlands into different functional groups (Brinson, 1993) and determine their functions in watersheds (Brinson, 1988; Whigham et al., 1988).

Flow regime, sediment size and supply, and flow competence (the ability of flow to move sediment of different sizes) determine the hydrogeomorphology of floodplains (Church, 2002). Hydrogeomorphology, specifically, the timing, duration, and velocity of flooding all influence community structure, productivity, and nutrient cycling in floodplains (Conner and Day, 1982). Floodplains, in turn, influence watershed hydrology through transient water storage and high evapotranspiration rates (Conner and Day, 1982). Water origin also determines the physical and chemical attributes of floodplains and influences the structure and function of the ecosystem (Amoros and Bornette, 2002). The area of floodplain that is directly influenced by river water is determined by system geomorphology and antecedent hydrology but does not encompass the entire extent of a floodplain (Mertes, 1997). In addition to river water, important sources of flood water at some times and locations in floodplains include precipitation, groundwater, and upland runoff that may also limit direct inputs of river water to some floodplain locations.

Hydrogeomorphology influences ecosystem dynamics through the provision of flood-derived resources, known as flood subsidy (Odum et al., 1979), which is broader than the flood pulse concept. Open wetland ecosystems, whose geomorphology supports rapid hydrologic exchange with streams or estuaries, have greater vegetative productivity and faster biogeochemical cycling due to greater nutrient inputs than closed wetland ecosystems (Hopkinson, 1992). Floodplains on large rivers experience predictable flood inputs of water, sediment, carbon, and nutrients that, together with floodplain spatial geomorphic configuration, determine many ecological processes (Lewis et al., 2000). Thus, the physical configuration of river–floodplain systems controls their biogeochemical functions and biotic communities. More details on how floodplain hydrogeomorphology influences floodplain vegetation and biogeochemistry are presented below.

12.21.3 Controls of Vegetation in Floodplains

12.21.3.1 Hydrogeomorphic Controls of Vegetation

Many chapters in this volume provide extensive reviews of vegetation in floodplain ecosystems. In this chapter, a brief review of the interaction between hydrogeomorphology and vegetation and the influence of biogeochemistry on vegetation are presented.

Vegetation is responsive to hydrogeomorphic setting and processes in floodplains. This relationship is the basis for using vegetation as an indicator of current and historic hydrologic processes on the different geomorphic surfaces of floodplains (Brinson, 1993; Gurnell, 1997; Osterkamp and Hupp, 2010). For example, dendrogeomorphic analyses and interpretation of species distribution can identify hydrogeomorphic processes and estimate rates of geomorphic change in floodplains (Hupp and Bornette, 2003). Hydrogeomorphology influences floodplain vegetation through multiple processes at physiologic, population, community, and ecosystem levels.

Physiologic and metabolic stresses from flooding and soil saturation restrict plant species and communities to specific locations along the moisture gradient within the floodplain (Wharton et al., 1982). The elevation of the soil relative to water level creates strong spatial gradients of soil oxygen depletion in floodplain wetlands (Dwire et al., 2006). Plant species differentially tolerate the physiologic stresses associated with the lack of oxygen, which include anaerobic metabolism and the toxicity of reduced metals (Ernst, 1990). In the short term, acclimation processes include changes in physiology, anatomy, and morphology in response to the lack of oxygen (McKelvin et al., 1998).

Wetland plants also respond to flooding stresses at the longer timescale of population dynamics through acclimation, mortality, or avoidance by seed dispersal and selective germination cues, all of which lead to species-distribution patterns. Flood energy is an important mechanism for moving plant propagules and influencing plant population dynamics in riparian landscapes. Flood magnitude and seasonality affect the dispersal of propagules in floodplains (Gurnell et al., 2004). Soil surface elevation relative to flood magnitude determines the number, diversity, and floristic composition of propagules delivered to floodplains (Moggridge and Gurnell, 2009). Hydrochory, the dispersal of seeds by water, can be a more important vector for seed dispersal than wind, particularly for delivery of allochthonous propagules (Moggridge and Gurnell, 2009). After dispersal to a floodplain location, floodplain microtopography influences plant germination and establishment (Blood and Titus, 2010).

Hydrogeomorphology also influences vegetative community composition (Wharton et al., 1982; Hupp, 2000; Naiman et al., 2010). Gradients in soil surface elevation relative to local hydrology determine species composition and cause turnover in species assemblages across the geomorphic features of floodplains. Floodplains with intermediate flood frequency and greater microtopographic variation have greatest plant species richness (Pollock et al., 1998). Spatial patches with less flow energy and greater discharge of hyporheic groundwater are also associated with greater plant species richness within floodplains (Mouw et al., 2009). The relative roles of plant competition and facilitation as structuring forces in plant communities are a function of abiotic stress gradients (Callaway and Walker, 1997; Bornette et al., 2008). Thus, plant competition for resources is more likely to occur at higher elevations in floodplains where stress from anoxia, water velocity, and debris impacts occurs less frequently and with less magnitude.

Finally, the hydrogeomorphology of floodplains influences their rates of primary production. Through history, floodplains have been renowned for their fertility resulting in large-scale conversions to agricultural production (Brown, 1997). The subsidy of water and nutrients to floodplains results in natural plant communities with high net primary productivity (Conner and Day, 1982), although not always greater than adjacent upland communities (Megonigal et al., 1997). In support of the flood pulse concept, Conner and Day (1982) reported higher net primary production in forested floodplains with flowing seasonal flooding than those with stagnant flooding. Greater plant growth rates occur in areas of floodplains with hyporheic discharge likely due to both

Author's personal copy

greater water and nutrient availability (Mouw et al., 2009). Total production (litterfall, wood, and root production) is greatest with wetter conditions along a floodplain moisture gradient due mostly to enhanced wood production (Burke et al., 1999; Clawson et al., 2001). However, belowground production can also decrease with greater wetness (Day and Megonigal, 1993; Clawson et al., 2001), and Megonigal et al. (1997) documented a decrease in aboveground production with prolonged inundation due to plant stress.

12.21.3.2 Influence of Vegetation on Hydrogeomorphology

Hydrogeomorphology clearly influences vegetation pattern and process on floodplains, but vegetation also influences hydrologic and geomorphic processes in these interactive systems (Simon et al., 2004). The establishment of vegetation on bare surfaces generally stabilizes geomorphic surfaces and thereby influences hydrology (Osterkamp and Hupp, 2010). The fossil record demonstrates that the evolution of land plants, and in particular rooting, changed the hydrogeomorphic patterns of rivers likely due to increased geomorphic stability (Davies and Gibling, 2010). The importance of vegetation to hydrogeomorphic processes has resulted in the labeling of vegetation as the bioengineer of floodplains (Gurnell and Petts, 2006).

Vegetation generally impacts geomorphology by influencing the balance between deposition and erosion in floodplains, as well as by directly contributing to soil surface elevation in organic soil systems. Riparian and floodplain vegetation affects bank erosion by modifying roughness and hydraulics, including turbulence, flow diversion, infiltration, and evapotranspiration (Tabacchi et al., 2000; Simon et al., 2004). Vegetation also directly impacts bank erosion by impacting water balance and mechanical reinforcement (Gurnell, 1997; Gray and Barker, 2004). On floodplains, dense vegetation increases roughness and decreases water velocity and the shear stress on soils and sediments to levels that may be below the critical entrainment threshold resulting in reduced erosion (Smith, 2004; Larsen et al., 2011) and enhanced sediment deposition (Ross et al., 2004). Vegetation on islands in rivers also increases geomorphic stability and influences hydrology (Schumm, 1985; Osterkamp, 1998). Similarly, large woody debris influences hydrogeomorphic stability as well as vegetative succession (Naiman et al., 2010). By contrast, less vegetated floodplains can be subject to large-scale geomorphic change following large, eroding floods (Griffin and Smith, 2004). The removal of riparian vegetation also dramatically increases erosion rates (Vincent et al., 2009). Finally, floodplain soils are typically thought of as mineral dominated but some floodplains have high soil organic content (median of swamps = 43%, Appendix B in Bedford et al., 1999). Aboveground and belowground production by floodplain vegetation, along with allochthonous inputs, directly supplies the high organic content of soils in some floodplains and thus influences their soil elevation and hydrogeomorphology.

12.21.3.3 Biogeochemical Controls on Vegetation

The characteristics of vegetation can vary with the nutrient biogeochemistry of floodplain ecosystems, depending on nutrient availability. The productivity of wetland vegetation is generally determined by nutrient availability as well as hydrology (Brinson et al., 1981). Nutrient availability on some floodplains can be high enough to result in net primary production that is not limited by nutrients (Spink et al., 1998; Wassen et al., 2003). Yet some floodplains still have nutrient limitation of productivity (Lockaby and Walbridge, 1998). Some large wetland and floodplain landscapes with low allochthonous nutrient and mineral sediment loading, like limnogenous peatlands, are oligotrophic, and nutrient availability highly constrains plant productivity, community composition, and diversity (Bridgham et al., 1996; Noe et al., 2001; King et al., 2004).

The sensitivity of primary production to nutrient availability is supported by the observation that maximum litterfall production on floodplains occurs at a nitrogen (N) to phosphorus (P) mass ratio of 12 in litterfall, indicating sufficient supply of both N and P, and decreases with changes away from this optimal ratio (Lockaby and Walbridge, 1998). Plant productivity in pristine European floodplains is N limited, but nutrient limitation is generally absent in regulated and impacted floodplains due to high anthropogenic nutrient inputs (Antheunisse et al., 2006). In addition, discharge of nutrient-rich groundwater to floodplains increases basal area and decreases leaf carbon (C) to N ratios (Harner and Stanford, 2003) and also increases woody vegetation growth rates (Mouw et al., 2009). Furthermore, macrophyte and phytoplankton production in floodplain water bodies are enhanced by river nutrient import (Hamilton and Lewis, 1987; Heiler et al., 1995; Knowlton and Jones, 1997; Maine et al., 2004). In a forested floodplain, the lowest soil N availability and the lowest plant productivity were both found with fluctuating inundation compared to wetter or drier floodplain zones (Burke et al., 1999). Despite the larger number of studies of the effects of nutrient availability on aboveground production in floodplains, in general little knowledge exists of how belowground production in wetlands responds to nutrient enrichment. In terrestrial forests, fine root production increases with N availability (Nadelhoffer, 2000).

12.21.3.4 Other Biota

It should be noted that other biota in addition to vegetation are important to floodplain dynamics and support complex interactions with biogeochemistry and hydrogeomorphology. For example, beaver are widely regarded as ecosystem engineers and keystone species that can impact the physical, chemical, and biotic environment of aquatic and terrestrial ecosystems (Johnston and Naiman, 1987; Naiman et al., 1988). Marine-derived nutrients from salmon have been shown to fuel aquatic production, food webs, and riparian vegetation productivity (Naiman et al., 2002). Grazing of riparian vegetation can extensively change channel geomorphology, floodplain connectivity, and floodplain plant communities (Beschta and Ripple, 2006). Of course, soil microbes also are critical to ecosystem processes through their participation in biogeochemical cycles (Megonigal et al., 2004).

Author's personal copy

12.21.4 Controls of Nutrient Biogeochemistry in Floodplains

12.21.4.1 Hydrogeomorphic Controls of Nutrient Biogeochemistry

Hydrology and geomorphology create circumstances for augmented biogeochemical reactions by regulating when and where transport pathways converge and mix organic matter (electron donors) and electron acceptors (McClain et al., 2003; Vidon et al., 2010). One such hot spot is floodplains, where hot moments also occur. Floodplains are located at depositional and low-velocity geomorphic positions in riverine landscapes where high primary productivity and large pools of organic matter (hot spot) mix with transport pathways of reactive and particulate N and P (often during flood pulses; hot moments). These hydrogeomorphic properties are the ultimate reason why floodplains are important regulators of water quality. For example, denitrification in river corridors is enhanced where nitrate is delivered to carbon-rich and oxygen-poor environments along lateral floodplain and vertical hyporheic gradients (Pinay et al., 1994).

River–floodplain hydrology, as influenced by floodplain geomorphology, is generally accepted as the dominant control of biogeochemical process rates in floodplains (Lockaby and Walbridge, 1998; Hill, 2000). Hydrogeomorphology determines key aspects of floodplains: inundation characteristics and water velocity. These, in turn, determine the hydraulic residence time of transport pathways in the floodplain and the loading rate of material to the floodplain, together with the amount of material in source waters. All of these determine the deposition rates of material and biotic processing rates. Thus, hydrogeomorphology determines biogeochemical sinks, sources, and transformations in floodplains.

In particular, inundation dynamics are one basic determinant of biogeochemical processes in floodplains. For example, the biogeochemistry of floodplain soils is highly responsive to drying and wetting events (Baldwin and Mitchell, 2000). Decomposition rates, and associated mineralization or immobilization of N and P, are a function of inundation in wetlands (Brinson et al., 1981). Lockaby et al. (1996) experimentally manipulated inundation on a floodplain and found that decomposition rates were highest with temporary, aerobic flooding compared to nonflooded and longer duration flooding. Floodplain wetness also affects N and P retranslocation, the removal of nutrients from tissues prior to senescence, in deciduous floodplain trees (Clawson et al., 2001). Because the nutrient content and lability of organic matter can influence microbial decomposition of organic matter and N and P cycling (Aerts, 1997), the effects of inundation patterns can propagate through other floodplain ecosystem processes. Water-table depth and fluctuation also are influential in determining redox and N cycling. Denitrification rates are greater in restored floodplains that have re-established hydrology than those that do not have re-established hydrology and have infrequent surface-water inundation (Hunter et al., 2008). Similarly, fluctuations of surface water level increase with connectivity to streams and support greater rates of N sedimentation and ammonification in restored and natural floodplains (Wolf, K.L., Noe, G.B., Ahn, C., U.S. Geological Survey, unpublished data). Ammonification rates are greater with the water table near the soil surface (<10 cm below), nitrification rates are greater at the lowest water table (>30 cm below), and denitrification rates are greatest in the middle where greater nitrate availability and lower soil redox coincide (water table 10–30 cm below the soil surface; Hefting et al., 2004). Johnston et al. (2001) found that hydrology was more important than soil characteristics in determining nutrient cycling in floodplains, with hot spots of nitrate production and denitrification on natural levees. Flooding also deposits refractory P and releases labile P from riparian soils (Fabre et al., 1996). Finally, floodplains may vary from being sinks, sources, or transformers of nutrients depending on the seasonality of inundation (Noe and Hupp, 2007).

Hydrologic connectivity sets the template for river–-floodplain interaction and the capacity for floodplains to retain material (Heiler et al., 1995). The exchange of water between river and floodplain is an essential aspect of floodplains that regulates internal biogeochemical cycling as well as their capacity to influence riverine water quality. Only material loaded to a floodplain has the possibility of being retained or processed in the floodplain. In the case of surface water, flood pulses and water exchange between channel and floodplain are critical (Junk et al., 1989). Floodplains along more incised reaches (Stubblefield et al., 2006) or channelized and artificially leveed reaches (Noe and Hupp, 2005) have less connectivity between the main channel and floodplains and, therefore, trap less sediment and associated P, N, and C. Riverine floodplains, having connection to rivers with larger watershed areas and material loading, have greater long-term P accumulation rates than nonconnected depressional wetlands (Craft and Casey, 2000). It should be noted that not only overbank flooding is important – flow pulses below bank full also have important influence on ecosystem productivity and spatial heterogeneity in river–floodplain systems (Tockner et al., 2000).

Surficial flowpaths route water and associated materials into, through, and out of floodplains. Retention of particulate N and P is strongly influenced by hydrogeomorphology and vegetation through regulation of sedimentation rates in floodplain flowpaths (Walbridge and Struthers, 1993; Steiger and Gurnell, 2002; Noe and Hupp, 2005; Hupp et al., 2008; Larsen et al., 2011). Greater sedimentation and associated C, N, and P accumulation occur where water rich in sediment and particulates and dissolved nutrients enter floodplains (Johnston et al., 1984; Sánchez-Carrillo et al., 2001; Ross et al., 2004; Knösche, 2006; Fink and Mitsch, 2007; Kronvang et al., 2007). Lower elevation soil surfaces in floodplains also support greater sedimentation and C, N, and P accumulation rates (Olde Venterink et al., 2006; Stoeckel and Miller-Goodman, 2001). For example, near-channel riparian areas that were lower in elevation trapped more sediment, C, and N than more extensive but higher floodplains (Brunet et al., 1994). These spatial patterns of nutrient accumulation in floodplains propagate through other ecosystem processes, exemplifying the interactions among hydrogeomorphology, nutrient biogeochemistry, and vegetation. Wassen et al. (2003) found that floodplain plant biomass, plant N and P storage, and soil N and P mineralization rates all increased with proximity to the channel. Increased deposition of sediment N and P on floodplain soils leads to enhanced rates of soil N and P

Author's personal copy

mineralization and also greater litterfall N fluxes (Noe, G.B., et al., U.S. Geological Survey, unpublished data).

In wetlands in general, water residence time in flowpaths is a master controller of biogeochemical reaction rates (Kadlec and Wallace, 2009). In floodplains, water residence time is critical to nutrient retention and transformation rates once streamflow occurs over a connected floodplain flowpath. Floodplain backwaters with shorter residence times have less percent retention of nitrate compared to longer residence time flowpaths; nitrate uptake rates are maximal at intermediate nitrate-loading rates because moderate residence time permits some uptake in the upper reaches and transports nitrate to downstream reaches for further uptake (James et al., 2008). Nitrate concentrations are heterogeneous across the floodplain during periods of low flow, when limited hydraulic connectivity and longer residence time permit depletion of nitrate pools in water bodies away from the main channel, and more homogenous during high flow, when nitrate-rich main channel waters may be distributed across floodplains, and shorter residence times restrict nitrate uptake (Heiler et al., 1995; Tockner et al., 1999; Richardson et al., 2004). In addition, slow-moving groundwater associated with shallower hydraulic gradients is also associated with greater nitrate removal than faster-moving groundwater in riparian ecosystems (Sabater et al., 2003).

Inorganic nutrients transported from the river channel to floodplain lakes fuel primary productivity and are converted into organic nutrient forms (Hamilton and Lewis, 1987; Heiler et al., 1995; Knowlton and Jones, 1997; Tockner et al., 1999; Hein et al., 2003; Maine et al., 2004). During low-flow periods, a side arm of the Danube River was dominated by autochthonous primary production, whereas during flooding, autochthonous carbon was exported to the main channel and allochthonous carbon was imported to the floodplain (Preiner et al., 2008). However, short-hydroperiod floodplains can exhibit the opposite pattern of organic nutrient retention and inorganic nutrient export to the river during flooding (Noe and Hupp, 2007).

Groundwater flowpaths in river–floodplain corridors have a large role in regulating nutrient transport, redistribution, and retention (Stanford et al., 2005.) Discharge of hyporheic groundwater on the floodplain can be enriched in nutrients (Harner and Stanford, 2003). Groundwater discharge increases tree growth and increases lability of tree leaves (decreases C:N; Harner and Stanford, 2003), which should lead to enhanced N mineralization rates. Hydrology, geomorphology, and soil characteristics determine the retention of groundwater nitrate transported from uplands through floodplains, with shallow groundwater transported slowly through wet and organic floodplain soils having high capacity for nitrate removal (Gold et al., 2001). Denitrification can also occur in hyporheic flowpaths of the riparian zone (Triska et al., 1993).

Hydrogeomorphology has an additional influence on P biogeochemistry through metal biogeochemistry. Trace-metal transformations (including those involving P) are generally a function of topography, flooding, and vegetation in floodplains (Du Laing et al., 2009). The retention of dissolved P, in particular, is determined by the geochemistry of Fe, Al, and Ca, as well as organic matter, which, in turn, are influenced by hydrology and redox (Walbridge and Struthers, 1993). Specifically, concentrations of amorphous (noncrystalline) Al and Fe best predict the P adsorption capacity of soils in wetlands in general (Richardson, 1985). The large spatial variation in these parameters within and among floodplain wetlands results in patchy P dynamics (Bridgham et al., 2001). Amorphous Fe and Al have complex but different distribution patterns and associations; amorphous Fe was more abundant at lower elevations in floodplains and inputs were predominantly from flooding, whereas amorphous Al was associated with organic matter and inputs were predominantly from uplands (Darke and Walbridge, 2000). However, Fe–P and Al–P did not change with experimental floodplain flooding, although inundation did release microbial P that increased soil P availability (Wright et al., 2001). Chacon et al. (2005) also found greater labile P in drier zones associated with microbes, but less amorphous Fe and Al in flooded floodplains. Riparian wetlands with differing clay and noncrystalline Al and Fe content have differing P sorption capacity (Bruland and Richardson, 2004). Watershed urbanization increases the crystallinity of riparian Fe, and decreases soil P sorption capacity, likely due to increased deposition of crystalline mineral sediment eroded from terrestrial soils (Hogan and Walbridge, 2007). At the other extreme, riparian soils with very high organic content but little Al and Fe can have less P sorption capacity (Richardson, 1985; Lyons et al., 1998). In conclusion, it is necessary to consider Fe and Al inputs, fractionation, and transformations to predict floodplain P dynamics and retention.

Other soil characteristics influence nutrient biogeochemistry in floodplains. The proximity to current or previous channels in river–floodplain ecosystems determines the texture and elevation of alluvial floodplain soils (Stanturf and Schoenholtz, 1998). In addition to determining metal biogeochemistry, soil variability associated with floodplain microtopography directly influences rates of N and P fluxes. For example, potential rates of soil N mineralization flux are greater in finer texture soils of riparian geomorphic zones due to greater N pools compared to coarser-textured soils (Bechtold and Naiman, 2006). Hot spots of nitrate production and denitrification have been found on natural levees with coarser-textured soils (Johnston et al., 2001). However, flooding or rain after dry periods can flush large amounts of soil nitrate that accumulated from mineralization and nitrification in oxidized soils (Bechtold et al., 2003). The larger particle size and lower organic content of soils in floodplain natural levees compared to finer-textured backswamp deposits result in less P sorption in sandier levees (Axt and Walbridge, 1999; Bridgham et al., 2001).

Finally, the geologic setting of catchments also influences the biogeochemistry of floodplains. Floodplains on alluvial versus blackwater rivers differ in chemistry and sediment supply (Wharton et al., 1982; Junk and Furch, 1993; Hupp, 2000; Schilling and Lockaby, 2006; Anderson and Lockaby, 2007). Blackwater rivers draining Coastal Plains are characterized by lower suspended sediment load, ionic strength, pH, less inorganic material, and more organic matter than alluvial rivers draining Piedmont or montane physiographic provinces (Beck et al., 1974). As a result, sedimentation rates on floodplains of blackwater rivers are among the lowest recorded (Hupp, 2000). Floodplain soils of blackwater rivers are more organic, acidic, and N and P deficient, and have lower cellulose and lignin decomposition than those of alluvial rivers (Stanturf and

Author's personal copy

Schoenholtz, 1998; Entry, 2000). Alluvial rivers are characterized by more P and a lower N:P in floodplain litterfall that is closer to optimum for plant productivity (Schilling and Lockaby, 2006) and, as a result, have greater vegetative productivity (Junk and Furch, 1993; Lockaby and Walbridge, 1998; Schilling and Lockaby, 2006) than blackwater rivers.

12.21.4.2 Influence of Vegetation on Nutrient Biogeochemistry

The vegetation of floodplains directly affects nutrient biogeochemistry by storing nutrients in biomass as well as indirectly by producing organic matter that supports microbial processing of nutrients. Although not permanently retained, plant uptake of nutrients into biomass can be a large magnitude flux. Riparian vegetative uptake can range from 13% to 99% of total measured system N retention in riparian buffers across Europe (uptake + denitrification; Hefting et al., 2005). In some floodplains, similar quantities of P enter the floodplain and are recycled through the vegetation of the floodplain (Mitsch et al., 1979; Yarbro, 1983). Similarly, standing stocks of N and P stored in herbaceous biomass are similar to the annual inputs of N and P in wetlands (Olde Venterink et al., 2002). However, storage in plant tissues can be a small proportion of experimental N and P additions to floodplains (Brinson et al., 1984).

Vegetation also indirectly influences biogeochemistry through the production of food and electron receptors for microbes. The production of roots, exudates, and detritus influences the populations, communities, and activities of soil microbes including denitrifiers and symbiotic root microbes (Tabacchi et al., 2000). The lability of detritus, due in part to species differences in lignin content as well as nutrient stoichiometry, influences nutrient cycling (Melillo et al., 1982; Lockaby and Walbridge, 1998; Burke et al., 1999). Portions of the N and P stored in floodplain plant litter are mineralized by microbes into inorganic nutrient forms that are recycled through plant uptake or exported from the floodplain (Noe, G.B., et al., U.S. Geological Survey, unpublished data).

12.21.4.3 Biogeochemical Controls on Hydrogeomorphology?

Nutrient biogeochemistry indirectly influences floodplain hydrogeomorphology through nutrient control of vegetation (see the previous section). Because vegetation in many floodplains is nutrient limited, increased nutrient availability will increase plant productivity. This increase in productivity results in greater aboveground biomass. Aboveground biomass has a large influence on water velocity, sediment entrainment thresholds, and deposition rates in floodplains (Harvey et al., 2009). Belowground biomass influences soil surface elevation by modifying erosion rates (Wynn and Mostaghimi, 2006), although little is known about nutrient controls on belowground production.

Does nutrient biogeochemistry only indirectly influence the hydrogeomorphology of floodplain ecosystems through control of vegetation (**Figure 1**)? No – nutrient controls on decomposition rates could directly influence hydrogeomorphology through controls on soil elevation and permeability. Other mechanisms for direct control are possible, but in general very little information exists on this topic.

One possible mechanism for direct control of biogeochemistry on hydrogeomorphology is through changes in soil macropores and permeability. The amount and degree of decomposition of soil organic matter influence soil physical properties and processes, including bulk density, water-holding capacity, and hydraulic conductivity, dominantly (Stanturf and Schoenholtz, 1998). Macropores increase water and solute fluxes through wetland soils (Harvey and Nuttle, 1995). Many soil macropores are formed in channels left by decomposing roots (Beven and Germann, 1982). Changes in the decomposition rate of the refractory organic matter that stabilizes macropores and smaller pores due to increased nutrient availability could reduce permeability and have large impacts on the water balance of floodplain wetlands. Changes in root production and subsequent macropore formation by altered nutrient availability could indirectly influence hydrology as well, especially during the rising and falling limbs of inundation events.

The elevation of the soil surface relative to site hydrology is an essential trait in wetland ecosystems. For example, sufficient accretion of the soil surface is necessary for coastal wetlands to remain stable during rising sea levels (Callaway et al., 1996; Morris et al., 2002). Soil organic matter production is a critical component of soil accretion in wetlands and may be enhanced by increased nutrient availability (Craft and Richardson, 1993; Nyman et al., 2006). Importantly, decomposition of soil organic matter in wetlands also is generally limited by nutrient availability, either through bioavailable N or P inputs to soil microbes or through regulation of the nutrient content, stoichiometry, and lability of detritus (Brinson et al., 1981). Nutrient enrichment increases respiration in wetland soils (Amador and Jones, 1995) as well as decreases the pool of C in some wetland soils (Morris and Bradley, 1999; Wigand et al., 2009) but increases C storage in others (Craft and Richardson, 1993). Most research on this topic has not occurred in floodplains. However, these processes are relevant to floodplains because floodplain soils can have a large organic component as previously described. Clearly, more research is needed to determine the generalized effects of nutrient availability on organic matter fluxes and thus soil elevation in floodplains.

12.21.5 Case Studies

The following case studies describe some well-known examples of floodplain ecosystems with strong interactions among hydrogeomorphology, vegetation, and nutrient biogeochemistry.

12.21.5.1 Hummock/Hollow Geomorphology in Peatlands and Floodplains

Scale-dependent feedback processes among hydrogeomorphology, vegetation, and biogeochemistry can generate pattern in ecosystems (Rietkerk and van de Koppel, 2008). The patterned hummock and hollow microtopography in some wetlands and floodplains is a clear example of these interactive

Author's personal copy

processes. The hummock and hollow pattern in peatlands is controlled by scale-dependent feedback processes: greater evapotranspiration on hummocks creates groundwater transport toward the hummock from hollows, transporting nutrients to hummocks that increase vegetative production, and soil surface elevation, and further increase evapotranspiration on hummocks (Eppinga et al., 2008). Alternatively, greater plant production on hummocks increases the deposition of sediment and associated nutrients which then increase local plant production and soil elevation (Larsen et al., 2007).

Hummock and hollow microtopography has large implications for wetland ecosystem processes. The greater heterogeneity occurring with hummocks and hollows increases diversity of wet-meadow plants (Werner and Zedler, 2002) and soil microbes in floodplains (Ahn et al., 2009). Hummocks have different pore-water chemistry than hollows (Stribling et al., 2007). Root production is greater on floodplain hummocks (Jones et al., 2000), suggesting that higher elevations are hot spots for nutrient and water uptake. Additions of hummock and hollow microtopography to created wetlands increase plant diversity and spatial heterogeneity in soil chemistry (Bruland and Richardson 2005; Moser et al., 2007; Moser et al., 2009).

Hummock and hollow microtopography is common in tidal freshwater to brackish forested floodplains (Day et al., 2007). This microtopography creates large differences in water-table elevation relative to the soil surface in tidal freshwater floodplains and is the dominant gradient controlling plant distribution and species richness in these frequently inundated wetlands (Rheinhardt, 2007). Tree species in tidal freshwater floodplains occur preferentially on hummocks (and likely help build and maintain hummocks) compared to hollows in the more stressful, wetter areas of the floodplain (Duberstein and Conner, 2009). However, little is known about the nutrient biogeochemistry of hummocks compared to hollows in tidal freshwater floodplains (Anderson and Lockaby, 2007).

Microtopographic heterogeneity in floodplains, and wetlands in general, clearly has importance to the structure and function of these ecosystems. A critical research need is to better understand the interacting physical and biological processes that create, maintain, and destroy this geomorphology, particularly in response to disturbances. For example, the peatlands of the Florida Everglades historically supported but have lost much of their microtopographic patterning (mimicking river–floodplain hydrogeomorphology) following human modification of system hydrology; identification of the processes that will restore this pattern is a priority (Larsen et al., 2011).

12.21.5.2 Coastal Plain Floodplains

The extensive floodplains in the Coastal Plain physiographic province also demonstrate the importance of feedbacks to ecosystem pattern and process. Coastal Plains are predominantly flat, depositional landscapes with wide alluvial valleys located downstream from sloping erosional landscapes (Markewich et al., 1990). The low gradient of Coastal Plain rivers promotes high depositional rates of transported river sediment load, creating meandering channels with extensive areas of alluvial floodplains (Wharton et al., 1982; Hupp, 2000; Naiman et al., 2010). Forested floodplain wetlands were abundant along the small and large rivers of the Coastal Plain in the Southeastern United States (Conner and Day, 1982; Wharton et al., 1982; Sharitz and Mitsch, 1993; Messina and Conner, 1998). However, conversion to agricultural land use and logging has impacted a large portion of their areal extent (Conner and Day, 1982; Sharitz and Mitsch, 1993).

The fluvial dynamics of Coastal Plain river–floodplain systems generate complex geomorphic patterns, including natural levees, crevasses and their splays, backswamps, ridges, swales, sloughs, abandoned channels, oxbow lakes, and point bars (Wharton et al., 1982). Coastal Plain floodplains also have distinct hydrology. Sweet and Geratz (2003) found that Coastal Plain rivers reach bank full discharge on average every 0.19 year, 8 times more frequently than other rivers. As a result, Coastal Plain floodplains are frequently inundated. Over 50% of the floodplain of a Coastal Plain floodplain is inundated more than 15% of the average year (Benke et al., 2000). Floodplains along Coastal Plain reaches of rivers typically have very high sediment-accumulation rates where rivers originate outside the Coastal Plain or have disturbed catchments with high sediment loads (Hupp, 2000; Noe and Hupp, 2005; Phillips and Slattery, 2006; Kronvang et al., 2007; Hupp et al., 2008). By contrast, floodplain ecosystems along blackwater rivers that originate within the Coastal Plain have different structure and function compared to those along alluvial rivers that originate outside the Coastal Plain, as previously discussed.

Plant communities change with geomorphic position on Coastal Plain floodplains due to differential species tolerance to inundation and anoxia (Wharton et al., 1982; Sharitz and Mitsch, 1993; Conner, 1994) and successional age of dynamic geomorphic surfaces (Shankman, 1993). Annual productivity of forested wetlands in the Southeastern Coastal Plain of the United States varies greatly depending on site hydrology and disturbance (Conner, 1994). Although early research found high production in Coastal Plain floodplain forests that were subsidized by fluctuating hydrology, it is now thought that productivity in floodplains is similar to uplands and decreases with flooding depth during the growing season (Megonigal et al., 1997). However, it was found no differences in aboveground production among wetter and drier sites on two floodplains. A large amount of the high litterfall production may be decomposed, especially in wetlands with fluctuating hydrology (Brinson et al., 1981), compared to terrestrial ecosystems in temperate and arid regions (Aerts, 1997). Nonetheless, a large quantity and proportion of the detritus production on forested floodplains of the Coastal Plain is exported to the main channel of the river during floods (Cuffney, 1988).

The hydrogeomorphic complexity of Coastal Plain floodplains makes them difficult to study but determines their essential biogeochemical characteristics (Lockaby and Walbridge, 1998). Thus, geologic, geographic, geomorphic, and hydrologic characteristics of these floodplains imprint biogeochemical processes. The blackwater rivers that originate in the Coastal Plain have low pH and abundant organic matter with Al and Fe complexes (Beck et al., 1974). Coastal Plain floodplains are therefore generally rich in organic matter complexed with Al and Fe which enhances their ability to retain inorganic P (Darke and Walbridge, 2000). The crystallinity, complexation with organic matter, or oxidation state of

Author's personal copy

Fe and Al is sensitive to inundation and anoxia, which should decrease P retention in wetter locations (Darke and Walbridge, 2000). Flooding increases soil P availability, but the mechanism was loss of microbial P instead of solubilization of Fe–P or Al–P fractions with waterlogging (Wright et al., 2001).

Within Coastal Plain floodplains, lower elevations accumulate finer sediments and are enriched in soil P, N, and C (Stoeckel and Miller-Goodman, 2001). Soil N availability is also greater in wetter portions of these floodplains (Burke et al., 1999). The fine texture of Coastal Plain floodplain soils results in much higher P sorption capacities than the sandier regional upland soils (Walbridge and Struthers, 1993). The overall high sedimentation rates on Coastal Plain floodplains correspond to high accumulation and retention rates of the C, N, and P associated with sedimentation (Noe and Hupp, 2009; Hupp et al., 2009). Riverine Coastal Plain floodplain wetlands have greater soil P accumulation rates compared to nonriverine depressional wetlands, although sedimentation mass, and C and N accumulation rates were similar (Craft and Casey, 2000). However, the percent of river nitrogen retained by floodplains may be insignificant in Coastal Plain rivers with high dissolved nutrient loads (van der Lee et al., 2004).

12.21.5.3 Montane Floodplains

Montane rivers are characterized by high channel gradients, which lead to high hydraulic energy, dynamic channels, large sediment size, and high bedload/suspended load ratios (Schumm, 1985). Channel geomorphic pattern within montane alluvial floodplain reaches is a function of catchment slope and channel constriction (Beechie et al., 2006). Higher gradient reaches of montane floodplain rivers are generally braided, transitioning to island braided and then meandering with decreasing gradient at lower elevations (Schumm, 1985). This change in pattern is associated with increasing turnover of floodplain surfaces at higher elevations due to more frequent channel migration (Beechie et al., 2006). Within the island-braided domain of channel pattern in montane river–floodplain systems, Ward et al. (1999) proposed a model of island development that incorporates feedbacks between vegetation and hydrogeomorphology. Island persistence is a function of vegetation density, organic matter accumulation, and the frequency of high discharge (Ward et al., 1999). River regulation can change channel geomorphology from multi-channel braided to single-channel anastomosing (Ward et al., 1999).

The habitat template for biotic communities in montane floodplains is a function of their highly dynamic geomorphology that generates high geomorphic and hydrologic diversity (Poole et al., 2002). The extensive hyporheic zone of montane river–floodplain systems supports large flows, extensive and biodiverse subterranean foodwebs, and important biogeochemical transformations (Stanford and Ward, 1993). Groundwater discharge locations on montane floodplains increase nutrient availability and periphyton and macrophyte production and lower plant C:N (Harner and Stanford, 2003; Stanford et al., 2005; Mouw et al., 2009). Surface flow events at bankfull or below bankfull are important for creating ecosystem heterogeneity in montane floodplains, particularly in braided rivers with larger amounts of channel edge (Tockner et al., 2000). Seasonal variation in discharge and spatial heterogeneity in water sources create river channel expansion and contraction cycles that control spatiotemporal heterogeneity, water quality, primary productivity, and food webs (Stanford et al., 2005; Malard et al., 2006).

The frequency and intensity of scouring critically influence the distribution of vegetation in montane floodplains (Fetherston et al., 1995). Scour holes provide lentic habitat in a lotic environment (Stanford et al., 2005). Scouring at lower elevations limits the survival of seedlings, whereas coarser sediment at higher elevations also limits the survival of seedlings by decreasing water availability (Gage and Cooper, 2004). Flood damage to riparian vegetation is greater in steeper gradient reaches of montane floodplains and increases plant species diversity (Hupp, 1982).

12.21.5.4 Desert Streams

Desert stream ecosystems also are illustrative examples of the interaction of hydrogeomorphology, vegetation, and biogeochemistry. In fact, they have been used to describe the functional ecomorphology of streams (Fisher et al., 2007). Less infiltration in upland desert soils generates more overland flow and less groundwater flow to desert riparian zones (Martí et al., 2000). As a result, desert rivers are more hydrogeomorphically active and dynamic than their temperate counterparts with higher rates of upland and bank erosion and episodic sediment transport (Ffolliott et al., 2004). Marked temporal variation in the absence of surface water highlights the importance of drying processes in intermittent channels (Stanley et al., 1997). At the other end of hydrologic extremes, frequent flash flooding resets ecosystem succession (Fisher et al., 1982). The distribution of riparian vegetation is closely linked to groundwater depths and the frequency of discharge events (Hupp and Osterkamp, 1996). Stream entrenchment lowers riparian groundwater tables resulting in desertification of wetland plant communities in arid regions (Stromberg et al., 1996; Chambers et al., 2004). Longitudinal gradients in riparian soil moisture are typically opposite of most temperate climates, with greater soil moisture at higher watershed elevations due to greater evapotranspiration and less precipitation at lower elevations (Ffolliott et al., 2004).

Research in arid stream ecosystems has shown that hydrologic exchange between surface and subsurface waters has important implications for ecological and biogeochemical processes. Algal productivity is commonly nitrogenlimited in desert streams (Grimm and Fisher, 1986). Hyporheic flowpaths are hot spots of nitrate production in desert stream ecosystems due to high nitrification rates coupled with mineralization of surficial dissolved or particulate organic N (Holmes et al., 1994) in microbially active subsurface sediments (Grimm and Fisher, 1984). Reaches with hyporheic discharge, which is determined by stream geomorphology, therefore have greater primary production due to greater nitrate loading (Valett et al., 1994).

Riparian vegetation stabilizes the geomorphology of desert streams. Desert riparian zones (both in-channel and floodplain) exhibit alternative stable states where abundant vegetation reduces geomorphic change during flooding, resulting in increased productivity and further stabilization from catastrophic erosional events that remove vegetation (Heffernan,

Author's personal copy

2008). Periods of high macrophyte abundance lead to shallower and wider channels with greater hydraulic retention times in the channel (Harvey et al., 2003). Stream algal productivity also is more resilient to hydrologic disturbance in areas of higher primary production associated with hyporheic upwelling (Valett et al., 1994).

12.21.6 Conclusions

The many and potentially nonlinear interactions among hydrogeomorphology, vegetation, and nutrient biogeochemistry, nested in the four dimensional framework of river–floodplain systems, make floodplains difficult to study and gain insight. This complexity also makes quantitative predictions difficult. Yet this complexity provides an opportunity to understand the many linkages between physical and biotic processes in ecosystems. Current river–floodplain system concepts and understandings have clearly advanced to include the substantial spatial and temporal heterogeneity and interactive processes inherent to floodplains.

Mechanistic models of floodplains are needed to better manage floodplains and should include hierarchical, nonequilibrium, physical, and biological processes (Johnson et al., 1995). Including interactions among hydrogeomorphology, vegetation, and nutrient biogeochemistry has improved predictive modeling of floodplain systems – a concrete demonstration of the utility of considering these interactions. However, quantitative models of floodplain ecosystem dynamics are rare. The conceptual advances described in this chapter are integral to the few quantitative floodplain ecosystem models. For example, integrated and interactive models of hydrology, geomorphology, vegetation, and N and P cycling have successfully described ecosystem dynamics in natural, drained, and created floodplain wetlands (van der Peijl and Verhoeven, 2000; Wang and Mitsch, 2000). A model with floodplain geomorphology, hydrology, and N cycling was used to demonstrate the effects of dams and artificial levees on floodplain nitrate removal (Gergel et al., 2005). Hydrologic exchange between shallow and deep habitats in a modeled floodplain complex enhances system food-web production by coupling N regeneration and phytoplankton growth (Cloern, 2007). Finally, Helton et al. (2010) concluded that river biogeochemical models need to include hydrologic exchange between main channel and floodplain and the resulting biogeochemical processes to predict N transport and cycling. Further development of holistic quantitative models is needed to predict the future state of floodplain–river systems and their ecosystem services under altered hydrogeomorphic regimes.

References

Aerts, R., 1997. Climate, leaf litter chemistry and leaf litter decomposition in terrestrial ecosystems: a triangular relationship. Oikos 79, 439–449.

Ahn, C., Gillevet, P., Sikaroodi, M., Wolf, K., 2009. An assessment of soil bacterial community structure and physicochemistry in two microtopographic locations of a palustrine forested wetland. Wetlands Ecology and Management 17, 397–407.

Amador, J.A., Jones, R.D., 1995. Carbon mineralization in pristine and phosphorus-enriched peat soils of the Florida Everglades. Soil Science 159, 129–141.

Amoros, C., Bornette, G., 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology 47, 761–776.

Amoros, C., Roux, A.L., Reygrobellet, J.L., Bravard, J.P., Pautou, G., 1987. A method for applied ecological studies of fluvial hydrosystems. Regulated Rivers 1, 17–36.

Anderson, C.J., Lockaby, B.G., 2007. Soils and biogeochemistry of tidal freshwater forested wetlands. In: Conner, W.H., Doyle, T.W., Krauss, K.W. (Eds.), Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, Dordrecht, pp. 65–88.

Antheunisse, A.M., Loeb, R., Lamers, L.P.M., Verhoeven, J.T.A., 2006. Regional differences in nutrient limitation in floodplains of selected European rivers: implications for rehabilitation of characteristic floodplain vegetation. River Research and Applications 22, 1039–1055.

Axt, J.R., Walbridge, M.R., 1999. Phosphate removal capacity of palustrine forested wetlands and adjacent uplands in Virginia. Soil Science Society of America Journal 63, 1019–1031.

Baldwin, D.S., Mitchell, A.M., 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river–floodplain systems: a synthesis. Regulated Rivers: Research and Management 16, 457–467.

Bechtold, J.S., Edwards, R.T., Naiman, R.J., 2003. Biotic versus hydrologic control over seasonal nitrate leaching in a floodplain forest. Biogeochemistry 63, 53–72.

Bechtold, J.S., Naiman, R.J., 2006. Soil texture and nitrogen mineralization potential across a riparian toposequence in a semi-arid savanna. Soil Biology and Biochemistry 38, 1325–1333.

Beck, K.C., Reuter, J.H., Perdue, E.M., 1974. Organic and inorganic geochemistry of some coastal plain rivers of the southeastern United States. Geochimica et Cosmochimica Acta 38, 341–364.

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

Beechie, T.J., Liermann, M., Pollock, M.M., Baker, S., Davies, J., 2006. Channel pattern and river–floodplain dynamics in forested mountain river systems. Geomorphology 78, 124–141.

Beechie, T.J., Sear, D.A., Olden, J.D., et al., 2010. Process-based principles for restoring river ecosystems. BioScience 60, 209–222.

Benke, A.C., Chaubey, I., Ward, G.M., Dunn, E.L., 2000. Flood pulse dynamics of an unregulated river floodplain in the southeastern US Coastal Plain. Ecology 81, 2730–2741.

Beschta, R.L., Ripple, W.J., 2006. River channel dynamics following extirpation of wolves in northwestern Yellowstone National Park. USA. Earth Surface Processes and Landforms 31, 1525–1539.

Beven, K., Germann, P., 1982. Macropores and water flow in soils. Water Resources Research 18, 1311–1325.

Blood, L.E., Titus, J.H., 2010. Microsite effects on forest regeneration in a bottomland swamp in western New York. Journal of the Torrey Botanical Society 137, 88–102.

Bornette, G., Tabacchi, E., Hupp, C., Puijalon, S., Rostan, J.C., 2008. A model of plant strategies in fluvial hydrosystems. Freshwater Biology 53, 1692–1705.

Bridgham, S., Pastor, J., Janssens, J., Chapin, C., Malterer, T., 1996. Multiple limiting gradients in peatlands: a call for a new paradigm. Wetlands 16, 45–65.

Bridgham, S.D., Johnston, C.A., Schubauer-Berigan, J.P., Weishampel, P., 2001. Phosphorus sorption dynamics in soils and coupling with surface and pore water in riverine wetlands. Soil Science Society of America Journal 65, 577–588.

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

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

Brinson, M.M., Bradshaw, H.D., Kane, E.S., 1984. Nutrient assimilative capacity of an alluvial floodplain swamp. Journal of Applied Ecology 21, 1041–1057.

Brinson, M.M., Lugo, A.E., Brown, S., 1981. Primary productivity, decomposition and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics 12, 123–161.

Brown, A.G., 1997. Alluvial Geoarchaeology: Floodplain Archaeology and Environmental Change. Cambridge University Press, Cambridge.

Bruland, G.L., Richardson, C.J., 2004. A spatially explicit investigation of phosphorus sorption and related soil properties in two riparian wetlands. Journal of Environmental Quality 33, 785–794.

Bruland, G.L., Richardson, C.J., 2005. Hydrologic, edaphic, and vegetative responses to microtopographic reestablishment in a restored wetland. Restoration Ecology 13, 515–523.

Author's personal copy

Brunet, R.C., Pinay, G., Gazelle, F., Roques, L., 1994. Role of the floodplain and the riparian zone in the suspended matter and nitrogen retention in the Adour River, south-west France. Regulated Rivers Research and Management 9, 55–63.

Burke, M.K., Lockaby, B.G., Conner, W.H., 1999. Aboveground production and nutrient circulation along a flooding gradient in a South Carolina Coastal Plain forest. Canadian Journal of Forest Research 29, 1402–1418.

Callaway, J.C., DeLaune, R.D., Patrick, W.H., 1996. Chernobyl 137Cs used to determine sediment accretion rates at selected Northern European coastal wetlands. Limnology and Oceanography 41, 444–450.

Callaway, R.M., Walker, L.R., 1997. Competition and facilitation: a synthetic approach to interactions in plant communities. Ecology 78, 1958–1965.

Chacon, N., Dezzeo, N., Muñoz, B., Rodríguez, J.M., 2005. Implications of soil organic carbon and the biogeochemistry of iron and aluminum on soil phosphorus distribution in flooded forests of the lower Orinoco River, Venezuela. Biogeochemistry 73, 555–566.

Chambers, J.C., Tausch, R.J., Korfmacher, J.L., Germanoski, D., Miller, J.R., Jewett, D., 2004. Effects of geomorphic processes and hydrologic regimes on riparian vegetation. In: Chambers, J.C., Miller, J.R. (Eds.), Great Basin Riparian Ecosystems: Ecology, Management, and Restoration. Island Press, Washington, DC, pp. 196–231.

Church, M., 2002. Geomorphic thresholds in riverine landscapes. Freshwater Biology 47, 541–557.

Clawson, R.G., Lockaby, B.G., Rummer, B., 2001. Changes in production and nutrient cycling across a wetness gradient within a floodplain forest. Ecosystems 4, 126–138.

Cloern, J.E., 2007. Habitat connectivity and ecosystem productivity: implications from a simple model. American Naturalist 169, E21–E33.

Conner, W., 1994. Effect of forest management practices on southern forested wetland productivity. Wetlands 14, 27–40.

Conner, W.H., Day, J.W., 1982. The ecology of forested wetlands in the southeastern United States. In: Gopal, B., Turner, R.E., Wetzel, R.G., Whigham, D.F. (Eds.), Wetlands Ecology and Management. National Institute of Ecology and International Scientific Publications, New Delhi, pp. 69–87.

Costanza, R., d'Arge, R., de Groot, R., et al., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Craft, C.B., Casey, W.P., 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia. USA. Wetlands 20, 323–332.

Craft, C.B., Richardson, C.J., 1993. Peat accretion and phosphorus accumulation along a eutrophication gradient in the northern Everglades. Biogeochemistry 22, 133–156.

Cuffney, T.F., 1988. Input, movement and exchange of organic matter within a subtropical coastal blackwater river–floodplain system. Freshwater Biology 19, 305–320.

Darke, A.K., Walbridge, M.R., 2000. Al and Fe biogeochemistry in a floodplain forest: implications for P retention. Biogeochemistry 51, 1–32.

Davies, N.S., Gibling, M.R., 2010. Cambrian to Devonian evolution of alluvial systems: the sedimentological impact of the earliest land plants. Earth-Science Reviews 98, 171–200.

Day, F., Megonigal, J., 1993. The relationship between variable hydroperiod, production allocation, and belowground organic turnover in forested wetlands. Wetlands 13, 115–121.

Day, R.H., Williams, T.M., Swarzenski, C.M., 2007. Hydrology of tidal freshwater forested wetlands of the southeastern United States. In: Conner, W.H., Doyle, T.W., Krauss, K.W. (Eds.), Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, New York, NY, pp. 29–63.

Du Laing, G., Rinklebe, J., Vandecasteele, B., Meers, E., Tack, F.M.G., 2009. Trace metal behaviour in estuarine and riverine floodplain soils and sediments: a review. Science of the Total Environment 407, 3972–3985.

Duberstein, J.A., Conner, W.H., 2009. Use of hummocks and hollows by trees in tidal freshwater forested wetlands along the Savannah River. Forest Ecology and Management 258, 1613–1618.

Dwire, K., Kauffman, J., Baham, J., 2006. Plant species distribution in relation to water-table depth and soil redox potential in montane riparian meadows. Wetlands 26, 131–146.

Entry, J.A., 2000. Influence of nitrogen on cellulose and lignin mineralization in blackwater and redwater forested wetland soils. Biology and Fertility of Soils 31, 436–440.

Eppinga, M.B., Rietkerk, M., Borren, W., Lapshina, E.D., Bleuten, W., Wassen, M.J., 2008. Regular surface patterning of peatlands: confronting theory with field data. Ecosystems 11, 520–536.

Ernst, W.H.O., 1990. Ecophysiology of plants in waterlogged and flooded environments. Aquatic Botany 38, 73–90.

Fabre, A., Pinay, G., Ruffinoni, C., 1996. Seasonal changes in inorganic and organic phosphorus in the soil of a riparian forest. Biogeochemistry 35, 419–432.

Fetherston, K.L., Naiman, R.J., Bilby, R.E., 1995. Large woody debris, physical process, and riparian forest development in montane river networks of the Pacific Northwest. Geomorphology 13, 133–144.

Ffolliott, P.F., DeBano, L.F., Baker, M.B., Neary, D.G., Brooks, K.N., 2004. Hydrology and impacts of disturbance on hydrologic function. In: Baker, M.B., Ffolliott, P.F., DeBano, L.F., Neary, D.G. (Eds.), Riparian Areas of the Southwestern United States: Hydrology, Ecology and Management. CRC Press, Boca Raton, FL, pp. 1–9.

Fink, D.F., Mitsch, W.J., 2007. Hydrology and nutrient biogeochemistry in a created river diversion oxbow wetland. Ecological Engineering 30, 93–102.

Fisher, S.G., 1997. Creativity, idea generation, and the functional morphology of streams. Journal of the North American Benthological Society 16, 305–318.

Fisher, S.G., Gray, L.J., Grimm, N.B., Busch, D.E., 1982. Temporal succession in a desert stream ecosystem following flash flooding. Ecological Monographs 52, 93–110.

Fisher, S.G., Grimm, N.B., Martí, E., Holmes, R.M., Jones, J.B.J., 1998. Material spiraling in stream corridors: a telescoping ecosystem model. Ecosystems 1, 19–34.

Fisher, S.G., Heffernan, J.B., Sponseller, R.A., Welter, J.R., 2007. Functional ecomorphology: feedbacks between form and function in fluvial landscape ecosystems. Geomorphology 89, 84–96.

Gage, E., Cooper, D., 2004. Constraints on willow seedling survival in a Rocky Mountain montane floodplain. Wetlands 24, 908–911.

Galat, D.L., Fredrickson, L.H., Humburg, D.D., et al., 1998. Flooding to restore connectivity of regulated, large-river wetlands. BioScience 48, 721–733.

Gergel, S.E., Carpenter, S.R., Stanley, E.H., 2005. Do dams and levees impact nitrogen cycling? Simulating the effects of flood alterations on floodplain denitrification. Global Change Biology 11, 1352–1367.

Gold, A.J., Groffman, P.M., Addy, K., Kellogg, D.Q., Stolt, M., Rosenblatt, A.E., 2001. Landscape attributes as controls on ground water nitrate removal capacity of riparian zones. Journal of the American Water Resources Association 37, 1457–1464.

Gray, D.H., Barker, D., 2004. Root-soil mechanics and interactions. In: Bennett, S.J., Simon, A. (Eds.), Riparian Vegetation and Fluvial Geomorphology. American Geophysical Union, Washington, DC, pp. 113–124.

Gregory, S.V., Swanson, F.J., McKee, W., Cummins, K.W., 1991. An ecosystem perspective of riparian zones. BioScience 41, 540–551.

Griffin, E.R., Smith, J.D., 2004. Floodplain stabilization by woody riparian vegetation during an extreme flood. In: Bennett, S.J., Simon, A. (Eds.), Riparian Vegetation and Fluvial Geomorphology. American Geophysical Union, Washington, DC, pp. 221–236.

Grimm, N.B., Fisher, S.G., 1984. Exchange between interstitial and surface water: implications for stream metabolism and nutrient cycling. Hydrobiologia 111, 219–228.

Grimm, N.B., Fisher, S.G., 1986. Nitrogen limitation in a Sonoran Desert stream. Journal of the North American Benthological Society 5, 2–15.

Gurnell, A., 1997. The hydrological and geomorphological significance of forested floodplains. Global Ecology and Biogeography 6, 219–229.

Gurnell, A., Petts, G., 2006. Trees as riparian engineers: the Tagliamento River, Italy. Earth Surface Processes and Landforms 31, 1558–1574.

Gurnell, A.M., Angold, P.G., Goodson, J.M., Morrissey, I.P., Petts, G.E., Steiger, J., 2004. Vegetation propagule dynamics and fluvial geomorphology. In: Bennett, S.J., Simon, A. (Eds.), Riparian Vegetation and Fluvial Geomorphology. American Geophysical Union, Washington, DC, pp. 209–210.

Hamilton, S.K., Lewis, W.M., 1987. Causes of seasonality in the chemistry of a lake on the Orinoco River floodplain, Venezuela. Limnology and Oceanography 32, 1277–1290.

Harner, M.J., Stanford, J.A., 2003. Differences in cottonwood growth between a losing and a gaining reach of an alluvial floodplain. Ecology 84, 1453–1458.

Harvey, J.W., Conklin, M.H., Koelsch, R.S., 2003. Predicting changes in hydrologic retention in an evolving semi-arid alluvial stream. Advances in Water Resources 26, 939–950.

Harvey, J.W., Nuttle, W.K., 1995. Fluxes of water and solute in a coastal wetland sediment. 2. Effect of macropores on solute exchange with surface water. Journal of Hydrology 164, 109–125.

Harvey, J.W., Schaffranek, R.W., Noe, G.B., Larsen, L.G., Nowacki, D.J., O'Connor, B.L., 2009. Hydroecological factors governing surface water flow on a low-gradient floodplain. Water Resources Research 45, W03421.

Author's personal copy

Heffernan, J.B., 2008. Wetlands as an alternative stable state in desert streams. Ecology 89, 1261–1271.
Hefting, M., Clément, J.C., Dowrick, D., et al., 2004. Water table elevation controls on soil nitrogen cycling in riparian wetlands along a European climatic gradient. Biogeochemistry 67, 113–134.
Hefting, M.M., Clement, J.-C., Bienkowski, P., et al., 2005. The role of vegetation and litter in the nitrogen dynamics of riparian buffer zones in Europe. Ecological Engineering 24, 465–482.
Heiler, G., Hein, T., Schiemer, F., Bornette, G., 1995. Hydrological connectivity and flood pulses as the central aspects for the integrity of a river–floodplain system. Regulated Rivers: Research and Management 11, 351–361.
Hein, T., Baranyi, C., Herndl, G.J., Wanek, W., Schiemer, F., 2003. Allochthonous and autochthonous particulate organic matter in floodplains of the River Danube: the importance of hydrological connectivity. Freshwater Biology 48, 220–232.
Helton, A.M., Poole, G.C., Meyer, J.L., et al., 2010. Thinking outside the channel: modeling nitrogen cycling in networked river ecosystems. Frontiers in Ecology and the Environment. http://dx.doi.org/10.1890/080211.
Hill, A.R., 2000. Stream chemistry and riparian zones. In: Jones, J.B., Mulholland, P.J. (Eds.), Streams and Groundwaters. Academic Press, London, pp. 83–107.
Hogan, D.M., Walbridge, M.R., 2007. Urbanization and nutrient retention in freshwater riparian wetlands. Ecological Applications 17, 1142–1155.
Holmes, R.M., Fisher, S.G., Grimm, N.B., 1994. Parafluvial nitrogen dynamics in a desert stream ecosystem. Journal of the North American Benthological Society 13, 468–478.
Hopkinson, C.S., 1992. A comparison of ecosystem dynamics in freshwater wetlands. Estuaries and Coasts 15, 549–562.
Hunter, R.G., Faulkner, S.P., Gibson, K.A., 2008. The importance of hydrology in restoration of bottomland hardwood wetland functions. Wetlands 28, 605–615.
Hupp, C.R., 1982. Stream-grade variation and riparian-forest ecology along Passage Creek, Virginia. Bulletin of the Torrey Botanical Club 109, 488–499.
Hupp, C.R., 2000. Hydrology, geomorphology and vegetation of Coastal Plain rivers in the south-eastern USA. Hydrological Processes 14, 2991–3010.
Hupp, C.R., Bornette, G., 2003. Vegetation as a tool in the interpretation of fluvial geomorphic processes and landforms in humid temperate areas. In: Kondolf, M., Piegay, H. (Eds.), Tools in Fluvial Geomorphology. Wiley, Chichester, pp. 269–288.
Hupp, C.R., Demas, C.R., Kroes, D.E., Day, R.H., Doyle, T.W., 2008. Recent sedimentation patterns within the central Atchafalaya Basin, Louisiana. Wetlands 28, 125–140.
Hupp, C.R., Osterkamp, W.R., 1996. Riparian vegetation and fluvial geomorphic processes. Geomorphology 14, 277–295.
Hupp, C.R., Pierce, A.R., Noe, G.B., 2009. Floodplain geomorphic processes and environmental impacts of human alteration along Coastal Plain rivers. USA. Wetlands 29, 413–429.
Hynes, H.B.N., 1975. The stream and its valley. Verh. Internat. Verein Limnol. 19, 1–15.
James, W.F., Richardson, W.B., Soballe, D.M., 2008. Effects of residence time on summer nitrate uptake in Mississippi River flow-regulated backwaters. River Research and Applications 24, 1206–1217.
Johnson, B.L., Richardson, W.B., Naimo, T.J., 1995. Past, present, and future concepts in large river ecology. BioScience 45, 134–141.
Johnston, C.A., Bridgham, S.D., Schubauer-Berigan, J.P., 2001. Nutrient dynamics in relation to geomorphology of riverine wetlands. Soil Science Society of America Journal 65, 557–577.
Johnston, C.A., Bubenzer, G.D., Lee, G.B., Madison, F.W., McHenry, J.R., 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. Journal of Environmental Quality 13, 283–290.
Johnston, C.A., Naiman, R.J., 1987. Boundary dynamics at the aquatic-terrestrial interface: the influence of beaver and geomorphology. Landscape Ecology 1, 47–57.
Jones, R.H., Henson, K.O., Somers, G.L., 2000. Spatial, seasonal, and annual variation of fine root mass in a forested wetland. Journal of the Torrey Botanical Society 127, 107–114.
Junk, W.J., Bayley, P.B., Sparks, R.E., 1989. The flood pulse concept in river–floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences 106, 110–127.
Junk, W.J., Furch, K., 1993. A general review of tropical South American floodplains. Wetlands Ecology and Management 2, 231–238.
Kadlec, R.H., Wallace, S., 2009. Treatment Wetlands. Lewis Publishers, Boca Raton, FL.
King, R.S., Richardson, C.J., Urban, D.L., Romanowicz, E.A., 2004. Spatial dependency of vegetation–environment linkages in an anthropogenically influenced wetland ecosystem. Ecosystems 7, 75–97.
Knösche, R., 2006. Organic sediment nutrient concentrations and their relationship with the hydrological connectivity of floodplain waters (River Havel, NE Germany). Hydrobiologia 560, 63–76.
Knowlton, M.F., Jones, J.R., 1997. Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity. Wetlands 17, 468–475.
Kronvang, B., Andersen, I., Hoffmann, C., Pedersen, M., Ovesen, N., Andersen, H., 2007. Water exchange and deposition of sediment and phosphorus during inundation of natural and restored lowland floodplains. Water, Air, and Soil Pollution 181, 115–121.
Larsen, L.G., Aumen, N., Bernhardt, C., 2011. Recent and historic drivers of landscape change in the Everglades ridge, slough, and tree island mosaic. Critical Reviews in Environmental Science and Technology 41(S1), 344–381.
Larsen, L.G., Harvey, J.W., Crimaldi, J.P., 2007. A delicate balance: Ecohydrological feedbacks governing landscape morphology in a lotic peatland. Ecological Monographs 77, 591–614.
van der Lee, G.E.M., Olde Venterink, H., Asselman, N.E.M., 2004. Nutrient retention in floodplains of the Rhine distributaries in The Netherlands. River Research and Applications 20, 315–325.
Leopold, L.B., Wolman, M.G., Miller, J.P., 1964. Fluvial Processes in Geomorphology. W.H. Freeman and Company, San Francisco.
Lewis, W.M., Hamilton, S.K., Lasi, M.A., Rodriguez, M., Saunders, J.F., 2000. Ecological determinism on the Orinoco floodplain. BioScience 50, 681–692.
Lockaby, B.G., Murphy, A.L., Somers, G.L., 1996. Hydroperiod influences on nutrient dynamics in decomposing litter of a floodplain forest. Soil Science Society of America 60, 1267–1272.
Lockaby, B.G., Walbridge, M.R., 1998. Biogeochemistry. In: Messina, M.G., Conner, W.H. (Eds.), Southern Forested Wetlands: Ecology and Management. Lewis Publishers, Boca Raton, FL, pp. 149–172.
Lyons, J.B., Gorres, J.H., Amador, J.A., 1998. Spatial and temporal variability of phosphorus retention in a riparian forest soil. Journal of Environmental Quality 27, 895–903.
Maine, M.A., Sune, N.L., Bonetto, C., 2004. Nutrient concentrations in the Middle Parana River: effect of the floodplain lakes. Archiv fur Hydrobiologie 160, 85–103.
Malard, F., Uehlinger, U., Zah, R., Tockner, K., 2006. Flood-pulse and riverscape dynamics in a braided glacial river. Ecology 87, 704–716.
Markewich, H.W., Pavich, M.J., Buell, G.R., 1990. Contrasting soils and landscapes of the Piedmont and Coastal Plain, eastern United States. Geomorphology 3, 417–447.
Martí, E., Fisher, S.G., Schade, J.D., Grimm, N.B., 2000. Flood frequency and stream-riparian linkages in arid lands. In: Jones, J.B., Mulholland, P.J. (Eds.), Streams and Ground Waters. Academic Press, San Diego, CA, pp. 111–136.
McClain, M.E., Boyer, E.W., Dent, C.L., et al., 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6, 301–312.
McKelvin, M.R., Hook, D.D., Rozelle, A.A., 1998. Adaptation of plants to flooding and soil waterlogging. In: Messina, M., Conner, W. (Eds.), Southern Forested Wetlands. Lewis Publishers, Boca Raton, FL, pp. 173–204.
Megonigal, J.P., Conner, W.H., Kroeger, S., Sharitz, R.R., 1997. Aboveground production in southeastern floodplain forests: a test of the subsidy-stress hypothesis. Ecology 78, 370–384.
Megonigal, J.P., Hines, M.E., Visscher, P.T., 2004. Anaerobic metabolism: linkages to trace gases and aerobic processes. In: Schlesinger, W.H. (Ed.), Biogeochemistry. Elsevier-Pergamon, Oxford, pp. 317–424.
Melillo, J.M., Aber, J.D., Muratore, J.F., 1982. Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. Ecology 63, 621–626.
Mertes, L.A.K., 1997. Documentation and significance of the perirheic zone on inundated floodplains. Water Resources Research 33, 1749–1762.
Messina, M.G., Conner, W.H., 1998. Southern Forested Wetlands: Ecology and Management. CRC Press-Lewis Publishers, Boca Raton, FL.
Mitsch, M.J., Dorge, C.L., Wiemhoff, J.R., 1979. Ecosystem dynamics and a phosphorus budget of an alluvial cypress swamp in southern Illinois. Ecology 60, 1116–1124.
Mitsch, M.J., Gosselink, J.G., 1993. Wetlands. Van Nostrand Reinhold, New York, NY.
Moggridge, H.L., Gurnell, A.M., 2009. Hydrological controls on the transport and deposition of plant propagules within riparian zones. River Research and Applications 26, 512–527.
Morris, J.T., Bradley, P.M., 1999. Effects of nutrient loading on the carbon balance of coastal wetland sediments. Limnology and Oceanography 44, 699–702.

Author's personal copy

Morris, J.T., Sundareshwar, P.V., Nietch, C.T., Kjerfve, B., Cahoon, D.R., 2002. Responses of coastal wetlands to rising sea level. Ecology 83, 2869–2877.

Moser, K., Ahn, C., Noe, G.B., 2007. Characterization of microtopography and its influence on vegetation patterns in created wetlands. Wetlands 27, 1081–1097.

Moser, K.F., Ahn, C., Noe, G.B., 2009. The influence of microtopography on soil nutrients in created mitigation wetlands. Restoration Ecology 17, 641–651.

Mouw, J.E.B., Stanford, J.A., Alaback, P.B., 2009. Influences of flooding and hyporheic exchange on floodplain plant richness and productivity. River Research and Applications 25, 929–945.

Nadelhoffer, K.J., 2000. Research review: the potential effects of nitrogen deposition on fine-root production in forest ecosystems. New Phytologist 147, 131–139.

Naiman, R., Bechtold, J., Beechie, T., Latterell, J., Van Pelt, R., 2010. A process-based view of floodplain forest patterns in coastal river valleys of the Pacific Northwest. Ecosystems 13, 1–31.

Naiman, R.J., Bilby, R.E., Schindler, D.E., Helfield, J.M., 2002. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. Ecosystems 5, 399–417.

Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28, 621–658.

Naiman, R.J., Décamps, H., McClain, M.E., 2005. Riparia. Academic Press, San Diego, CA.

Naiman, R.J., Johnston, C.A., Kelley, J.C., 1988. Alteration of North American streams by beaver. BioScience 38, 753–762.

National Research Council, 2002. Riparian Areas. National Academy Press, Washington, DC.

Noe, G.B., Childers, D.L., Jones, R.D., 2001. Phosphorus biogeochemistry and the impact of phosphorus enrichment: why is the Everglades so unique? Ecosystems 4, 603–624.

Noe, G.B., Hupp, C.R., 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. Ecological Applications 15, 1178–1190.

Noe, G.B., Hupp, C.R., 2007. Seasonal variation in nutrient retention during inundation of a short-hydroperiod floodplain. River Research and Applications 23, 1088–1101.

Noe, G.B., Hupp, C.R., 2009. Retention of riverine sediment and nutrient loads by Coastal Plain floodplains. Ecosystems 12, 728–746.

Nyman, J.A., Walters, R.J., Delaune, R.D., Patrick, W.H., 2006. Marsh vertical accretion via vegetative growth. Estuarine, Coastal and Shelf Science 69, 370–380.

Odum, E.P., Finn, J.T., Eldon, H.F., 1979. Perturbation theory and the subsidy-stress gradient. BioScience 29, 349–352.

Olde Venterink, H., Pieterse, N.M., Belgers, J.D.M., Wassen, M.J., de Ruiter, P.C., 2002. N, P, and K budgets along nutrient availability and productivity gradients in wetlands. Ecological Applications 12, 1010–1026.

Olde Venterink, H., Vermaat, J.E., Pronk, M., et al., 2006. Importance of sediment deposition and denitrification for nutrient retention in floodplain wetlands. Applied Vegetation Science 9, 163–174.

Osterkamp, W.R., 1998. Processes of fluvial island formation, with examples from Plum Creek, Colorado and Snake River, Idaho. Wetlands 18, 530–545.

Osterkamp, W.R., Hupp, C.R., 2010. Fluvial processes and vegetation – glimpses of the past, the present, and perhaps the future. Geomorphology 116, 274–285.

van der Peijl, M.J., Verhoeven, J.T.A., 2000. Carbon, nitrogen, and phosphorus cycling in river marginal wetlands; a model examination of landscape geochemical flows. Biogeochemistry 50, 45–71.

Petts, G.E., Large, A.R.G., Greenwood, M.T., Bickerton, M.A., 1992. Floodplain assessment for restoration and conservation: linking hydrogeomorphology and ecology. In: Carling, P.A., Petts, G.E. (Eds.), Lowland Floodplain Rivers: Geomorphological Perspectives. Wiley, West Sussex, pp. 217–234.

Phillips, J.D., Slattery, M.C., 2006. Sediment storage, sea level, and sediment delivery to the ocean by coastal plain rivers. Progress in Physical Geography 30, 513–530.

Pinay, G., Haycock, N.E., Ruffinoni, C., Holmes, R.M., 1994. The role of denitrification in nitrogen removal in river corridors. In: Mitsch, W.J. (Ed.), Global Wetlands: Old World and New. Elsevier, Dordrecht, pp. 107–116.

Poff, N.L., Allan, J.D., Bain, M.B., et al., 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47, 769–784.

Poff, N.L., Richter, B.D., Arthington, A.H., et al., 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biology 55, 147–170.

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

Poole, G.C., 2010. Stream hydrogeomorphology as a physical science basis for advances in stream ecology. Journal of the North American Benthological Society 29, 12–25.

Poole, G.C., Stanford, J.A., Frissell, C.A., Running, S.W., 2002. Three-dimensional mapping of geomorphic controls on flood-plain hydrology and connectivity from aerial photos. Geomorphology 48, 329–347.

Preiner, S., Drozdowski, I., Schagerl, M., Schiemer, F., Hein, T., 2008. The significance of side-arm connectivity for carbon dynamics of the River Danube, Austria. Freshwater Biology 53, 238–252.

Rheinhardt, R.D., 2007. Tidal freshwater swamps of a lower Chesapeake Bay subestuary. In: Conner, W.H., Doyle, T.W., Krauss, K.W. (Eds.), Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, New York, NY, pp. 161–182.

Richardson, C.J., 1985. Mechanisms controlling phosphorous retention capacity in freshwater wetlands. Science 228, 1424–1426.

Richardson, W.B., Strauss, E.A., Bartsch, L.A., Monroe, E.M., Cavanaugh, J.C., Vingum, L., Soballe, D.M., 2004. Denitrification in the Upper Mississippi River: rates, controls, and contribution to nitrate flux. Canadian Journal of Fisheries and Aquatic Science 61, 1102–1112.

Rietkerk, M., van de Koppel, J., 2008. Regular pattern formation in real ecosystems. Trends in Ecology and Evolution 23, 169–175.

Ross, K.M., Hupp, C.R., Howard, A.D., 2004. Sedimentation in floodplains of selected tributaries of the Chesapeake Bay. In: Bennett, S.J., Simon, A. (Eds.), Riparian Vegetation and Fluvial Geomorphology. American Geophysical Union, Washington, DC, pp. 187–208.

Sabater, S., Butturini, A., Clement, J.-C., et al., 2003. Nitrogen removal by riparian buffers along a European climatic gradient: patterns and factors of variation. Ecosystems 6, 20–30.

Sánchez-Carrillo, S., Álvarez-Cobelas, M., Angeler, D.G., 2001. Sedimentation in the semi-arid freshwater wetland Las Tablas de Daimil (Spain). Wetlands 21, 112–124.

Schilling, E.B., Lockaby, B.G., 2006. Relationships between productivity and nutrient circulation within two contrasting southeastern U.S. floodplain forests. Wetlands 26, 181–192.

Schumm, S.A., 1985. Patterns of alluvial rivers. Annual Review of Earth and Planetary Sciences 13, 5–27.

Sedell, J.R., Richey, J.E., Swanson, F.J., 1989. The river continuum concept: a basis for the expected ecosystem behavior of very large rivers? Canadian Special Publication of Fisheries and Aquatic Sciences 106, 49–55.

Shankman, D., 1993. Channel migration and vegetation patterns in the Southeastern Coastal Plain. Conservation Biology 7, 176–183.

Sharitz, R.R., Mitsch, W.J., 1993. Southern floodplain forests. In: Martin, W.H., Boyce, S.G., Esternacht, A.C. (Eds.), Biodiversity of the Southeastern United States: Lowland Terrestrial Communities. Wiley, New York, NY, pp. 311–372.

Simon, A., Bennett, S.J., Neary, V.S., 2004. Riparian vegetation and fluvial geomorphology: problems and opportunities. In: Bennett, S.J., Simon, A. (Eds.), Riparian Vegetation and Fluvial Geomorphology. American Geophysical Union, Washington, DC, pp. 1–10.

Smith, J.D., 2004. The role of riparian shrubs in preventing floodplain unraveling along the Clark Fork of the Columbia River in the Deer Lodge Valley, Montana. In: Bennett, S.J., Simon, A. (Eds.), Riparian Vegetation and Fluvial Geomorphology. American Geophysical Union, Washington, DC, pp. 71–86.

Sparks, R.E., 1995. Need for ecosystem management of large rivers and their floodplains. BioScience 45, 168–182.

Spink, A., Sparks, R.E., 1998. van Oorschot, M., Verhoeven, J.T.A. Nutrient dynamics of large river floodplains. Regulated Rivers: Research and Management 14, 203–216.

Stanford, J.A., Lorang, M.S., Hauer, F.R., 2005. The shifting habitat mosaic of river ecosystems. Verhandlungen des Internationalen Verein Limnologie 29, 123–136.

Stanford, J.A., Ward, J.V., 1993. An ecosystem perspective of alluvial rivers: connectivity and the hyporheic corridor. Journal of the North American Benthological Society 12, 48–60.

Stanley, E.H., Fisher, S.G., Grimm, N.B., 1997. Ecosystem expansion and contraction in streams. BioScience 47, 427–435.

Stanturf, J.A., Schoenholtz, S.H., 1998. Soils and landforms of southern forested wetlands. In: Messina, M.G., Conner, W.A. (Eds.), Southern Forested Wetlands: Ecology and Management. CRC Press-Lewis Publishers, Boca Raton, FL, pp. 123–147.

Author's personal copy

Steiger, J., Gurnell, A.M., 2002. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River. France. Geomorphology 49, 1–23.
Steiger, J., Tabacchi, E., Dufour, S., Corenblit, D., Peiry, J.L., 2005. Hydrogeomorphic processes affecting riparian habitat within alluvial channel-floodplain river systems: a review for the temperate zone. River Research and Applications 21, 719–737.
Stoeckel, D.M., Miller-Goodman, M.S., 2001. Seasonal nutrient dynamics of forested floodplain soil influenced by microtopography and depth. Soil Science Society of America Journal 65, 922–931.
Stribling, J., Cornwell, J., Glahn, O., 2007. Microtopography in tidal marshes: ecosystem engineering by vegetation? Estuaries and Coasts 30, 1007–1015.
Stromberg, J.C., Tiller, R., Richter, B., 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: the San Pedro, Arizona. Ecological Applications 6, 113–131.
Stubblefield, A.P., Escobar, M.I., Larsen, E.W., 2006. Retention of suspended sediment and phosphorus on a freshwater delta, South Lake Tahoe, California. Wetlands Ecology and Management 14, 287–302.
Sweet, W.V., Geratz, J.W., 2003. Bankfull hydraulic geometry relationships and recurrence intervals for North Carolina's coastal plain. Journal of the American Water Resources Association 39, 861–871.
Tabacchi, E., Lambs, L., Guilloy, H., Planty-Tabacchi, A.-M., Muller, E., Décamps, H., 2000. Impacts of riparian vegetation on hydrological processes. Hydrological Processes 14, 2959–2976.
Thoms, M.C., Parsons, M., 2002. Eco-geomorphology: an interdisciplinary approach to river science. In: Dyer, F.J., Thoms, M.C., Olley, J.M. (Eds.), The Structure, Function and Management Implications of Fluvial Sedimentary Systems. International Association of Hydrological Sciences, Wallingford, pp. 113–119.
Thorp, J.H., Thoms, M.C., Delong, M.D., 2006. The riverine ecosystem synthesis: biocomplexity in river networks across space and time. River Research and Applications 22, 123–147.
Tockner, K., Lorang, M.S., Stanford, J.A., 2010. River flood plains are model ecosystems to test general hydrogeomorphic and ecological concepts. River Research and Applications 26, 76–86.
Tockner, K., Malard, F., Ward, J.V., 2000. An extension of the flood pulse concept. Hydrological Processes 14, 2861–2883.
Tockner, K., Pennetzdorfer, D., Reiner, N., Schiemer, F., Ward, J.V., 1999. Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river–floodplain system (Danube, Austria). Freshwater Biology 41, 521–535.
Tockner, K., Stanford, J.A., 2002. Riverine flood plains: present state and future trends. Environmental Conservation 29, 308–330.
Triska, F.J., Duff, J.H., Avanzino, R.J., 1993. Patterns of hydrological exchange and nutrient transformation in the hyporheic zone of a gravel-bottom stream: examining terrestrial-aquatic linkages. Freshwater Biology 29, 259–274.
Valett, H.M., Fisher, S.G., Grimm, N.B., Camill, P., 1994. Vertical hydrologic exchange and ecological stability of a desert stream ecosystem. Ecology 75, 548–560.
Vannote, R.L., Minsall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37, 130–137.
Vidon, P., Allan, C., Burns, D., et al., 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. JAWRA Journal of the American Water Resources Association 46, 278–298.
Vincent, K., Friedman, J., Griffin, E., 2009. Erosional consequence of saltcedar control. Environmental Management 44, 218–227.
Walbridge, M.R., Struthers, J.P., 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the mid-Atlantic region. Wetlands 13, 84–94.
Wang, N., Mitsch, W.J., 2000. A detailed ecosystem model of phosphorus dynamics in created riparian wetlands. Ecological Modelling 126, 101–130.
Ward, J.V., 1989a. The four-dimensional nature of lotic ecosystems. Journal of the North American Benthological Society 8, 2–8.
Ward, J.V., 1989b. Riverine–wetland interactions. In: Sharitz, R.R., Gibbons, J.W. (Eds.), Freshwater Wetlands and Wildlife. USDOE Office of Scientific and Technical Information, Oak Ridge, pp. 385–400.
Ward, J.V., 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83, 269–278.
Ward, J.V., Stanford, J.A., 1995. The serial discontinuity concept: extending the model to floodplain rivers. Regulated Rivers: Research and Management 10, 159–168.
Ward, J.V., Tockner, K., Edwards, P.J., et al., 1999. A reference river system for the Alps: the 'Fiume Tagliamento'. Regulated Rivers: Research and Management 15, 63–75.
Wassen, M.J., Peeters, W.H.M., Olde Venterink, H., 2003. Patterns in vegetation, hydrology, and nutrient availability in an undisturbed river floodplain in Poland. Plant Ecology 165, 27–43.
Werner, K., Zedler, J., 2002. How sedge meadow soils, microtopography, and vegetation respond to sedimentation. Wetlands 22, 451–466.
Wharton, C.H., Kitchens, W.M., Pendleton, E.C., Sipe, T.W., 1982. The Ecology of Bottomland Hardwood Swamps of the Southeast: A Community Profile. US Fish and Wildlife Service, Washington, DC.
Whigham, D.F., Chitterling, C., Palmer, B., 1988. Impacts of freshwater wetlands on water quality: a landscape perspective. Environmental Management 12, 663–671.
Wigand, C., Brennan, P., Stolt, M., Holt, M., Ryba, S., 2009. Soil respiration rates in coastal marshes subject to increasing watershed nitrogen loads in southern New England. USA. Wetlands 29, 952–963.
Wright, R.B., Lockaby, B.G., Walbridge, M.R., 2001. Phosphorus availability in an artificially flooded southeastern floodplain forest soil. Soil Science Society of America Journal 65, 1293–1302.
Wynn, T., Mostaghimi, S., 2006. The effects of vegetation and soil type on streambank erosion, Southwestern Virginia, USA. Journal of the American Water Resources Association 42, 69–82.
Yarbro, L.A., 1983. The influence of hydrologic variations on phosphorus cycling and retention in a swamp stream ecosystem. In: Fontaine, T.D., Bartell, S.M. (Eds.), Dynamics of Lotic Ecosystems. Ann Arbor Science, Ann Arbor, pp. 223–245.

Biographical Sketch

Gregory B. Noe obtained his honors BS in biology from Virginia Polytechnic Institute and State University and PhD in ecology from the Joint Doctoral Program of San Diego State University and University of California, Davis. He is a research ecologist with the National Research Program of the US Geological Survey. His interests focus on wetland ecosystem ecology, including the interactive influences of hydrology, geomorphology, climate, and biology on nitrogen and phosphorus biogeochemistry in fluvial ecosystems, as well as plant community ecology and restoration ecology.