

# Chapter 10

## Phosphorus and Nitrogen Dynamics in Riverine Systems: Human Impacts and Management Options



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### 10.1 Introduction

Water chemistry constitutes one key factor for the ecological state of streams and rivers as it determines the composition of the media in which the aquatic organisms live. Among the various chemical substances dissolved in water, phosphorus (P) and nitrogen (N) are particularly important for the management of riverine systems. These two macronutrients are essential components of all organisms and are closely linked to the aquatic carbon cycle, determining both the primary production and the microbial mineralization of organic matter in aquatic systems. The industrialization and intensification of agricultural production during the twentieth century has resulted in the nutrient enrichment and eutrophication of many freshwaters in Europe and the USA, impairing the water quality of rivers, lakes, and aquifers (Grizetti et al. 2011). Among others, eutrophication is responsible for toxic algal blooms, water anoxia, and habitat and biodiversity loss in freshwater ecosystems and poses direct threats to humans by impairing drinking water quality (Smith and Schindler 2009). Nutrient enrichment causes severe problems in coastal zones and can even affect the climate through increased greenhouse gas emissions. Despite current improvements in wastewater treatment from industrial and municipal sources in Europe (Kroiss et al. 2005), phosphorus and nitrogen remain of concern for river managers especially in regions where intensive urban or agricultural land use results in pollution of aquatic systems through diffuse nutrient inputs. Diffuse sources challenge the management of nutrients in riverine systems by requiring a combination of mitigation measures on both the catchment and the reach scale (Mainstone and Parr 2002).

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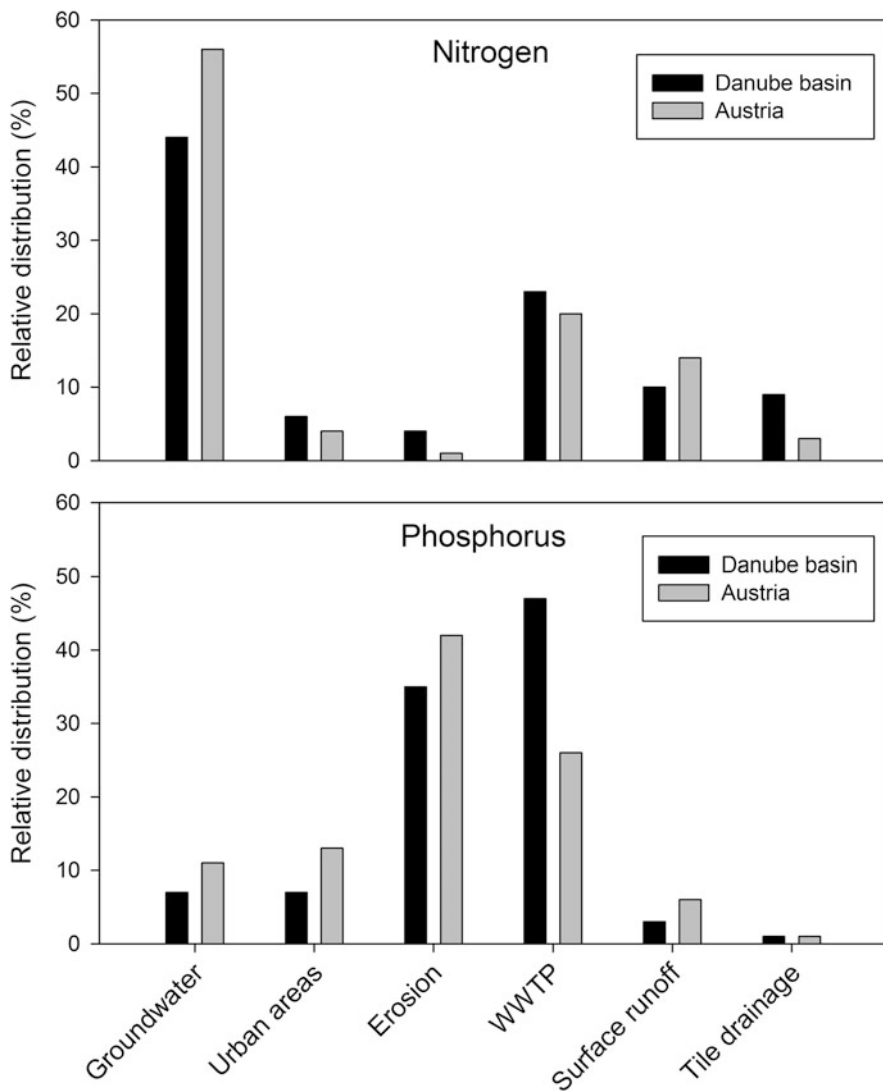
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The following chapter describes dominant input pathways and transformation processes for these two nutrients in streams and rivers and deals with the various human impacts on these processes which impair the nutrient retention function of riverine systems, with specific reference to the Danube River basin. The chapter provides an overview of measures for the mitigation and management of diffuse nutrient inputs in both the river and the riparian zone. Technical treatment of wastewater and point sources is not further addressed in this book chapter and can be found in, e.g., Tchobanoglous et al. (2003). In addition, we address consequences of river restoration measures on the nutrient uptake and release in running waters.

## 10.2 Historic and Current Emission Situation in the Danube River Basin

Between the 1950s and the 1980s, the emissions of nitrogen and phosphorus into the rivers of the Danube basin increased by more than the twofold as a result of industrialization, urbanization, and intensification of agriculture (Kroiss et al. 2005; Grizetti et al. 2011). Since the 1990s, slight-to-moderate mitigations of nutrient inputs have been achieved by improving wastewater treatment via the implementation of collection systems and new technologies mainly in Germany and Austria and by reducing industrial discharges in the lower Danube countries. However, trends are not consistent throughout European water bodies. In particular, emissions from diffuse sources originating from agricultural areas remain elevated (Kroiss et al. 2005). The significance of diffuse sources for phosphorus and nitrogen inputs to riverine systems in Austria and the Danube basin is shown in Fig. 10.1. While wastewater treatment plants account for about 20–26% of nutrient emissions to Austrian rivers, diffuse inputs via groundwater, soil erosion, and surface runoff (including urban areas) play a key role in delivering nitrogen and phosphorus to riverine systems (BMLFUW 2017). Currently, only 6% of Austrian streams and rivers have a moderate to high risk for failure in water quality due to point sources, but about 25% are threatened in their water quality by diffuse sources.

While point sources are relatively easy to control, the management of diffuse sources requires an integrative approach on multiple levels, comprising (1) minimization of emissions in the catchment (see Chap. 13), (2) nutrient retention in riparian buffer zones and floodplains, and (3) the control of the nutrient cycling within the river channel (Mainstone and Parr 2002). The increasing application of good management practices in agricultural catchments during the last decade has resulted in the mitigation of nutrient loads in rivers, especially as regards nitrogen (Kronvang et al. 2005; Oenema et al. 2005). However, the effects of catchment measures on river water quality are often less effective than expected due to nitrogen and phosphorus legacies in soils and groundwater from past land use activities (Sharpley et al. 2014). Therefore, catchment management needs to be supported by on-site measures in both the riparian zone and the channel. Effective nutrient management within the riverine system, in turn, requires a profound understanding of the various biogeochemical processes phosphorus and nitrogen undergo in running waters.



**Fig. 10.1** Sources of nitrogen and phosphorus to riverine systems in Austria and the Danube basin (Data from Kroiss et al. 2005; BMLFUW 2017)

### 10.3 Forms and Sources of Phosphorus and Nitrogen

Phosphorus occurs in aquatic systems in four basic forms: dissolved inorganic phosphorus (usually referred to as soluble reactive phosphorus (SRP) which is immediately bioavailable), dissolved organic phosphorus (DOP; e.g., P-esters), particulate organic phosphorus (POP; in detritus and living biomass), and particulate

inorganic phosphorus (PIP; e.g., iron- or aluminum-bound phosphorus on particles) (Allan and Castillo 2007). SRP is biologically available for both aquatic primary producers and microorganisms, while the particulate and soluble organic fractions must undergo chemical or biological transformations to SRP before being bioavailable.

Nitrogen occurs in freshwater ecosystems in three forms: dissolved inorganic nitrogen (DIN), including ammonium ( $\text{NH}_4\text{-N}$ ), nitrate ( $\text{NO}_3\text{-N}$ ), and nitrite ( $\text{NO}_2\text{-N}$ ), dissolved organic nitrogen (DON; e.g., amino acids, polypeptides), and particulate organic nitrogen (PON) (Allan and Castillo 2007). In addition, nitrogen occurs in gaseous forms as dinitrogen gas  $\text{N}_2$  and nitrous oxide ( $\text{N}_2\text{O}$ ), a potent greenhouse gas. DIN is biologically available for both aquatic primary producers and microorganisms, whereby  $\text{NH}_4\text{-N}$  is preferentially taken up by the aquatic community due to lower physiological costs (Birgand et al. 2007).

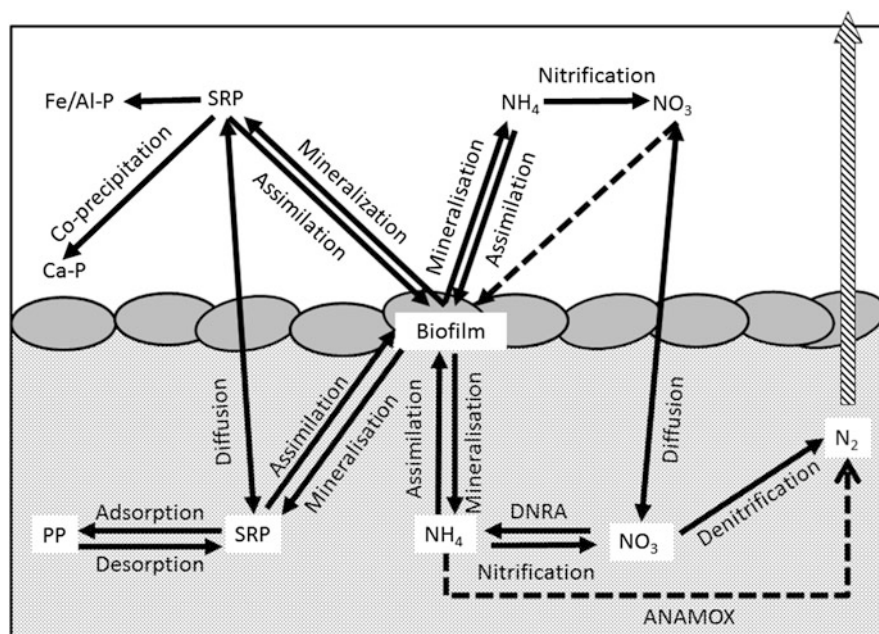
Natural sources for phosphorus and nitrogen comprise leaching from terrestrial soils and plant material during decomposition, release of P from weathering rocks, atmospheric deposition of  $\text{N}_2$  (precipitation and dry fallout), and biological N fixation through cyanobacteria (Mainstone and Parr 2002; Bernot and Dodds 2005; Birgand et al. 2007; Withers and Jarvie 2008). As nutrient inputs from natural sources are generally low, pristine streams usually show SRP concentrations  $<10 \mu\text{g SRP L}^{-1}$ , while average DIN concentrations may amount to  $0.1 \text{ mg NO}_3\text{-N L}^{-1}$ ,  $0.015 \text{ mg NH}_4\text{-N L}^{-1}$ , and  $0.001 \text{ mg NO}_2\text{-N L}^{-1}$  (Allan and Castillo 2007). DON can reach proportions of 40–90% of total nitrogen and, thus, often constitutes a major component in pristine systems.

Anthropogenic sources import significant amounts of nutrients into streams and rivers which lead to the eutrophication of the aquatic system and impair its ecological state. The main anthropogenic inputs to riverine systems include increased atmospheric deposition of  $\text{N}_2$  due to the burning of fossil fuels, cultivation of N-fixing crops, municipal wastewater and industrial effluents, and agricultural fertilizers (Bernot and Dodds 2005; Burgin and Hamilton 2007). Depending on their temporal and spatial extent, input pathways can be distinguished as (1) point sources, largely in form of municipal wastewater and industrial effluents, (2) nonpoint or diffuse sources from agricultural areas, and (3) intermediate sources, such as runoff from impervious surfaces (Withers and Jarvie 2008). Municipal wastewater is usually dominated by dissolved inorganic and, thus, bioavailable phosphorus and nitrogen (Mainstone and Parr 2002; Withers and Jarvie 2008), the concentrations of which depend on the efficiency of the sewage treatment. Point sources constitute permanent and localized delivery pathways, which are comparatively easy to control. Diffuse inputs from agricultural land use include agricultural fertilizers and increased soil leachate and erosion due to tillage. As SRP and ammonium easily adsorb to charged soil particles, these nutrients enter streams and rivers mainly via soil erosion (Craig et al. 2008; Withers and Jarvie 2008) (Figs. 10.1 and 10.3). By contrast, nitrate is highly soluble and mobile. Excess  $\text{NO}_3\text{-N}$  from agricultural areas is, thus, usually leached to groundwater or drainage waters and transported into river systems via subsurface flow paths (Grizetti et al. 2011) (Figs. 10.1 and 10.3). Intermediate sources include runoff from various urban areas and farmyards and vary greatly in nutrient amounts and composition. Both diffuse and intermediate sources occur mainly during storm

events and are difficult to control. Stormwater management structures in urban areas, such as vegetated ponds and wetlands, bio-retention devices, and porous pavements, can help to control both water and nutrient fluxes to urban streams (Bernhardt and Palmer 2007).

## 10.4 Nutrient Cycling in Streams and Rivers

Once in the aquatic system, phosphorus and nitrogen undergo numerous biogeochemical transformations (Fig. 10.2). Biotic transformations include the autotrophic and heterotrophic uptake of nutrients from the water, their assimilation into biomass, and their release by excretion and microbial decomposition (Reddy et al. 1999; Bernot and Dodds 2005; Birgand et al. 2007). In deep and slow-flowing rivers and floodplain channels, nutrient uptake by macrophytes and emergent plants plays a major role in nutrient cycling. Plants can take up SRP and DIN from soil or sediment pore water via roots or directly from the water column (Birgand et al. 2007). While nutrient storage in aboveground plant tissue is usually short-term, resulting in the release of nutrients after the vegetation period, belowground storage in roots and rhizomes may provide long-term storage, depending on the hydrological situation, the vegetation type, the physicochemical properties of the water, and the climate.



**Fig. 10.2** Dominant sources and transformation processes for nitrogen and phosphorus in running waters (after Mainstone and Parr 2002; Revsbech et al. 2006; Withers and Jarvie 2008)

Enhanced nutrient levels and increased water temperatures, for example, accelerate microbial decomposition of plant tissue (Withers and Jarvie 2008). Drying and rewetting of floodplain sediments in the course of water level fluctuations have also been shown to foster nutrient release from organic matter (Schönbrunner et al. 2012). In small streams, benthic processes dominate over water column processes. Benthic algae and microorganisms can assimilate nutrients from both the water column and the pore water of the sediments. In the case of phosphorus surplus, algae are capable of luxury phosphorus uptake, i.e., excess uptake that is not immediately needed but can be used subsequently for later growth, while microorganisms can store phosphorus via the formation of polyphosphate compounds (Reddy et al. 1999).

Bacterial mineralization of organic matter results in the release of SRP and  $\text{NH}_4\text{-N}$  to the water column. Under aerobic conditions,  $\text{NH}_4\text{-N}$  is turned into  $\text{NO}_3\text{-N}$  by nitrifying bacteria via nitrification (Bernot and Dodds 2005; Birgand et al. 2007) (Fig. 10.2). Under anaerobic conditions, denitrifying bacteria use nitrate as an electron acceptor to oxidize organic matter, thus reducing  $\text{NO}_3\text{-N}$  via  $\text{NO}_2\text{-N}$  and  $\text{N}_2\text{O}$  to  $\text{N}_2$  (Bernot and Dodds 2005; Birgand et al. 2007; Burgin and Hamilton 2007). This denitrification process is promoted by low oxygen concentrations and high concentrations of organic matter and nitrate, as occur, e.g., in wetland soils, sediments of agricultural streams, and groundwater, and represents a permanent N sink for streams and rivers. Denitrification is often restricted to anoxic microzones in the sediments and can be coupled to nitrification by using nitrate originating from decomposition and subsequent nitrification rather than nitrate imported from external sources (Birgand et al. 2007). In aquatic systems with high organic carbon, but low nitrate concentrations, dissimilatory nitrate reduction to ammonium (DNRA) may become important under anoxic conditions (Birgand et al. 2007; Burgin and Hamilton 2007). However, the significance of DNRA for nitrogen cycling in running waters remains to be investigated yet. Another potential sink for nitrogen is the anaerobic oxidation of ammonium to  $\text{N}_2$  using nitrite (Anammox). So far, Anammox has been mainly observed in anoxic environments with high nitrogen, but low carbon concentrations, such as wastewaters and marine systems (Burgin and Hamilton 2007).

In addition to these biological processes, phosphorus availability in running waters is influenced by various physical and chemical transformations. The adsorption of phosphorus to sediment particles plays a key role in phosphorus cycling, especially in streams and rivers impaired by agricultural land use. Adsorption comprises all physical and chemical processes in which phosphorus is bound to the surface of particles, such as ligand exchange, electrostatic attraction, and ion exchange (Reddy et al. 1999; Withers and Jarvie 2008). Sedimentation of particle-bound phosphorus constitutes an important phosphorus sink in retention zones of rivers, such as pools, floodplain lakes, or impounded sections. However, the adsorbed phosphorus can be released again to the water column, depending on the adsorption capacity of the sediments, which is highest in clay and sand, and on the concentration gradient between the pore water and the water column (Reddy et al. 1999; Mainstone and Parr 2002). In general, phosphorus adsorption occurs at high SRP concentrations in the surface water, while desorption is favored by low

SRP concentrations in the surface water. The zero equilibrium phosphorus concentration ( $EPC_0$ ) is the SRP concentration in the water where no net phosphorus exchange between the water column and the sediments occurs (House 2003). High  $EPC_0$  values are especially evident in streams and rivers in agricultural catchments due to phosphorus overloading of the sediments. As a consequence, sediments function as an internal phosphorus source for the water column during most of the time (Sharpley et al. 2014).

Under aerobic conditions, dissolved inorganic and organic phosphorus may complex with metal oxides and hydroxides to form insoluble precipitates (Reddy et al. 1999; House 2003). This phosphorus is released under anaerobic conditions as may occur in organic-rich sediments of floodplain lakes and agricultural streams and rivers. In addition, phosphorus can coprecipitate with calcite in calcareous waters under high pH conditions resulting from the photosynthetic activity of macrophytes and benthic algae.

To summarize, both biotic and abiotic processes significantly influence phosphorus and nitrogen retention in riverine systems, depending on various factors such as hydrology, climate, the activity of primary producers and decomposers, and the loading of the system by organic matter and nutrients. In particular, the role of fine sediment accumulations as potential sink or source for phosphorus has to be taken into account in nutrient management concepts.

## 10.5 Human Impacts on Nutrient Cycling

Due to the various biogeochemical transformation processes, nitrogen and phosphorus are continuously recycled between their inorganic and organic forms as well as among the water column, the sediments, and the biota. The continuous downstream movement of water in streams and rivers transforms these nutrient cycles into spirals (nutrient spiraling concept; see review by Ensign and Doyle 2006). The length of the spirals depends on the nutrient uptake capacity of the river relative to the water transport and represents the efficiency of the aquatic system for nutrient retention. This nutrient retention efficiency depends on two factors: (1) the physical (hydrological) retention of the water within the system and (2) the nutrient demand of the aquatic community.

The hydrological retention determines the contact time and the contact area between nutrients and biogeochemically reactive sites in the riverine systems (Ensign and Doyle 2006). It depends on the three-dimensional connectivity of the river channel with adjacent compartments, namely, the longitudinal linkage of headwaters with downstream reaches, the transversal linkage between channel and riparian areas, and the vertical linkage between surface water and the hyporheic zone (Ward 1989). Time, in the form of hydrological dynamics (e.g., flooding of riparian zones), adds a fourth dimension to the complex nutrient exchange processes in riverine systems.

Regarding the longitudinal aspect, nutrient retention is highest in headwater streams, which are important links between the catchment and downstream reaches (Reddy et al. 1999). Due to their diverse channel morphology and their low discharge, pristine headwaters can retain large amounts of nutrients via benthic uptake, thereby controlling nutrient transport into downstream reaches (Craig et al. 2008). Stream regulation due to urbanization or agricultural land use results in a homogenization of the stream channel and an acceleration of water flow and, thus, decreases the physical retention function of headwater streams (Ensign and Doyle 2005).

The vertical dimension of nutrient retention via the hyporheic zone is especially important in small streams (Boulton 2007). The hyporheic water exchange depends on the porosity of the sediments and on pressure imbalances at the sediment surface induced by local obstacles in the channel. Removal of flow obstacles in the channel, coverage of sediment surfaces with concrete or pavement, and the clogging of sediments due to siltation restrict the hyporheic water exchange and heavily impair the nutrient retention of streams (Boulton 2007). Besides, sedimentation of nutrient-loaded soil particles from agricultural landscapes may lead to internal eutrophication.

In larger streams and rivers, the lateral hydrological connectivity with riparian zones and floodplains determines the efficiency of nutrient retention. Here, the dimension of time gains in importance. Natural water level fluctuations of the river lead to the repeated connection and disconnection of floodplain water bodies with the main river, inducing the frequent exchange of chemically different water sources (Weigelhofer et al. 2015). Floodplains have proven to be especially efficient in trapping nutrients associated with particles (Reddy et al. 1999; Fisher and Acreman 2004). The remobilization of sediments is usually low as old sediments are buried by freshly deposited sediments. However, the role of floodplains in retaining dissolved nutrients is less clear and depends on the hydrology and the balance between uptake and release processes. Floodplain soils may constitute hotspots for denitrification in the case of high organic matter contents and high water tables (Forshay and Stanley 2005). However, the repeated drying and rewetting of floodplain soils can also cause the release of substantial amounts of nitrogen and phosphorus during flooding (Schönbrunner et al. 2012; Weigelhofer et al. 2015). Disconnections of floodplains from the main river, resulting from river regulation, impoundments, and channel incision, as well as hydrological alterations due to land use and climate changes have largely reduced the lateral hydrological connectivity of river-floodplain systems in Europe, thereby depriving rivers of these important retention structures (Hein et al. 2016).

The second aspect of nutrient retention in running waters is the nutrient demand of the aquatic primary producers and decomposers, which is controlled by the specific C–N–P (carbon-to-nitrogen-to-phosphorus) ratio of their bodies compared to the C–N–P ratio of their food (ecological stoichiometry; Cross et al. 2005). In pristine streams and rivers, nitrogen and phosphorus are mainly delivered by terrestrial plant material, which has substantially higher C-nutrient ratios than algae or microorganisms, thus limiting production. In eutrophic streams and rivers, nutrient supply from anthropogenic sources can exceed the demand of the community and lead to the saturation of the aquatic system (Bernot and Dodds 2005). Saturation of



lotic systems occurs as a result of (1) a limited nutrient demand of aquatic organisms as they become limited by other factors (e.g., light, oxygen), (2) the internal release of nutrients due to mineralization and abiotic release processes, and (3) reduced adsorption capacities of overloaded sediments (Bernot and Dodds 2005; Withers and Jarvie 2008). Sediments enriched with nutrients and organic matter from the catchment may serve as internal eutrophication source for the aquatic system as they continuously provide benthic communities with nutrients from below even though external inputs have been reduced. Organic matter accumulations in sediments occur especially in agricultural streams due to manure application, soil erosion, and the mowing of the riparian vegetation.

Numerous in-field nutrient addition experiments have determined the retention efficiency of streams for dissolved nutrients (Ensign and Doyle 2006). Decreased uptake efficiencies for dissolved N and P have been mainly reported from agricultural and urban streams as these streams are often subject to both degraded stream morphology and increased nutrient loads (Bernhardt and Palmer 2007). While uptake lengths for ammonium or SRP range from less than 100 m to a few 100 m in oligotrophic headwater streams (e.g., Hall et al. 2002; Gibson et al. 2015), agricultural headwater streams often yield uptake lengths of several kilometers (e.g., Gücker and Pusch 2006; Weigelhofer et al. 2013; Sheibley et al. 2014). Such streams have lost their natural retention function and act as mere transport systems, impairing the water quality of downstream reaches and recipient standing waters.

## 10.6 Potential and Limitations of Mitigation Measures

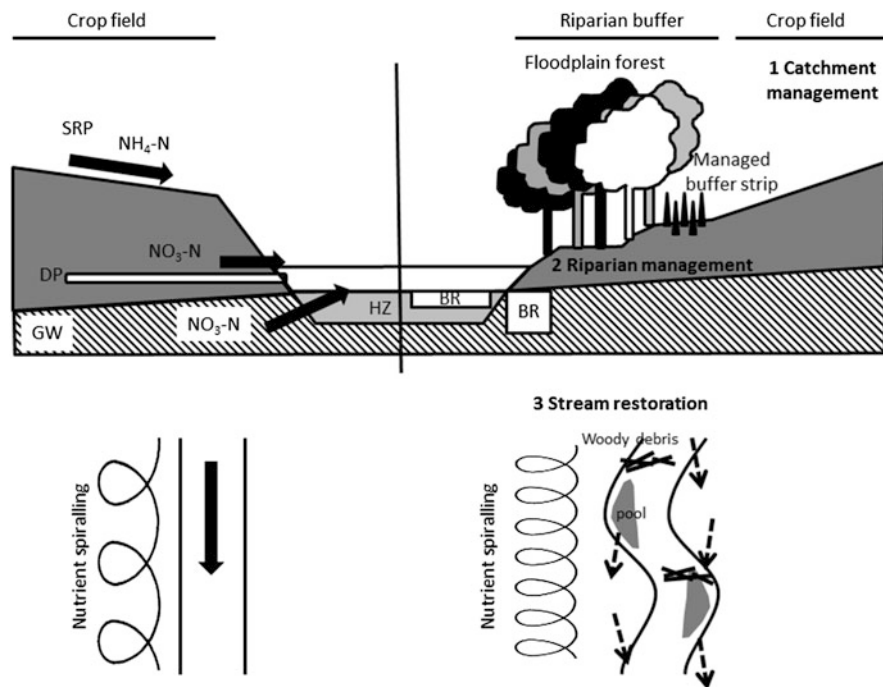
The following chapter focuses on measures within riverine systems, including riparian zones, for the management of diffuse nutrient inputs to streams and rivers (end-of-pipe measures). Measures for treatment of point sources, especially technical measures for wastewater treatment, are not discussed. For management measures in the catchment, we refer to Chap. 13.

*Riparian areas* constitute important interaction and buffer zones for river ecosystems as they control the fluxes of material and energy from the terrestrial catchment and the adjacent groundwater to the surface water (Hoffmann et al. 2009). Floodplains and riparian areas have the ability to retain, transform, and release nutrients, thereby influencing the water quality of the recipient water body (Hoffmann et al. 2009; Roberts et al. 2012). The sink and source function of riparian areas depends on the delivery pathway (surface runoff, drainage water, groundwater, or floodwater), the form of the delivered nutrient (particulate or dissolved), the specific biogeochemical conditions in the riparian area (e.g., soil humidity), the riparian vegetation, and the temperature (Fisher and Acreman 2004). Depending on the groundwater table in the riparian area relative to the water table of the surface water, riparian soils may favor oxic or anoxic processes. The use of riparian areas for nutrient management has led to a variety of initiatives, such as the restoration and reconnection of former floodplains, the establishment of vegetated (managed)

riparian buffer strips, the reestablishment of wetlands on agricultural land, and the installation of denitrifying bioreactors along streambanks and within channels.

*Vegetated buffer strips* are narrow, tillage-free, uncultivated border zones between agricultural areas and streams (Hoffmann et al. 2009; Roberts et al. 2012). While natural riparian zones vegetated with floodplain forests can also function as buffer zones, vegetated buffer strips are often optimized for nutrient removal as to vegetation type, width, and location, and they can be managed (Vought et al. 1994; Mayer et al. 2007). Vegetated buffer strips aim at reducing P and N inputs from soil erosion and surface runoff via deposition of soil particles, infiltration of water, and the subsequent geochemical and biological retention through sorption, precipitation, plant, and microbial uptake. Studies show that the retention of total phosphorus (TP) may be fairly efficient, depending on the type of vegetation and the morphology of the buffer strip (e.g., slope, width; Mayer et al. 2007), yielding TP retention between 40% and 95% of the original loads (Hoffmann et al. 2009). However, non-managed buffer strips usually provide no permanent P sink. Part of the deposited TP can be remobilized in the soil, e.g., in the course of mineralization of organic matter or P desorption, and be delivered as SRP to the surface water (Reddy et al. 1999; Fisher and Acreman 2004). The plantation and harvest of fast-growing species on buffer strips removes accumulated P and reduces P saturation, thus decreasing also DRP losses in surface runoff (Vought et al. 1994). So far, there is little evidence for significant N removal in riparian zones via plant uptake (Vought et al. 1994). However, riparian zones may be hotspots for nitrate removal in subsurface water due to denitrification, especially if a high water table is maintained in the biologically active soil (Mayer et al. 2007). Sabater et al. (2003) measured annual nitrate removal rates via denitrification between 5% and 30%  $\text{m}^{-1}$  in the riparian zones of 14 streams across Europe. The amounts of nitrate removed by denitrification depend more on the hydrology and the soil of the riparian zone than on buffer width and vegetation type (Vought et al. 1994). In general, complex buffer zones combining different vegetation types, such as grassland and forest communities, are the most efficient structures for nutrient removal and provide additional ecosystem services, such as shading, bank stabilization, increased habitat diversity, and improved microclimate (Mander et al. 2005). The effectiveness of riparian buffer strips for nutrient retention is largely reduced if water from terrestrial areas can circumvent the riparian buffer via subsurface flow or preferential surface flow paths, such as ditches which drain road runoff directly into streams (Mainstone and Parr 2002).

In large streams and rivers, riparian zones expand to *floodplains* surrounding the river channels (Fig. 10.3). The retention capacity of these floodplains is largely determined by the lateral hydrological connectivity. Like riparian buffer strips, natural floodplains can show a high retention capacity for particles as well as a high denitrification potential (Fisher and Acreman 2004; Hoffmann et al. 2009). Pinay et al. (2007), for example, measured denitrification rates in floodplain soils of European rivers of up to 30  $\text{g N m}^{-2} \text{month}^{-1}$ . Floodplains also provide a multitude of ecosystem services apart from nutrient retention, including groundwater replenishment, flood protection, and habitats for a diverse flora and fauna, such as spawning habitats and nurseries for fish (Hein et al. 2016). Therefore, river managers



**Fig. 10.3** Management options for the mitigation of phosphorus and nitrogen in riverine systems at different scales: 1. the catchment; 2. the riparian zone; 3. the stream/river. *BR* denitrifying bioreactors, *DP* drainage pipe, *GW* groundwater, *HZ* hyporheic zone

have to consider possible detrimental side effects of using floodplains for nutrient mitigation. In the case of reconnecting isolated backwaters with the main river, nutrient loading of the floodplain by inflowing river water can affect the species composition in the wetland negatively (Verhoeven et al. 2006). Besides, flooding can increase the terrestrialization of shallow backwaters through increased sedimentation, and it can enhance greenhouse gas emissions due to the inundation of organic-rich floodplain soils. In the case of reestablishing wetlands on former agricultural land, nutrient legacies in the soils have to be considered, too (Reddy et al. 1999).

*Denitrifying bioreactors*, also known as denitrification beds, are one of the newest technologies for edge-of-field nitrate reduction (Schipper et al. 2010; Christianson et al. 2012). Denitrifying bioreactors are porous containers which are filled with an organic carbon source, such as wood chips, sawdust, and straw, in order to facilitate denitrification (Schipper et al. 2010; Christianson et al. 2012). For nutrient management in riverine systems, such bioreactors can be installed within groundwater or drainage water flow paths, along streambanks for diffuse inputs, or directly within the stream channel (Fig. 10.3). Due to the increased hydraulic conductivity of the reactors compared to the surrounding soils or sediments, diffuse inputs are concentrated by the reactor, thereby improving nitrate removal. Long-term nitrate removal

rates of denitrification beds range between 2 and 22 g N m<sup>-3</sup> day<sup>-1</sup> in groundwater, depending on the water residence time, the organic carbon source, nitrate concentrations, water temperature, pH, and the hydrological regime (Schipper et al. 2010). In-stream reactors within an agricultural drainage ditch yielded maximum nitrate removal rates of 160 g N m<sup>-2</sup> month<sup>-1</sup> (Robertson and Merkley 2009). In drainage water with fluctuating flow regimes, denitrifying bioreactors may be less effective as alternating high flow rates and intermittency restrict the denitrification process (Christianson et al. 2012; Weigelhofer and Hein 2015). Detrimental side effects of denitrifying bioreactors on adjacent surface waters and the atmosphere are an increased output of DOC, especially during the initial phase, and the production of N<sub>2</sub>O.

Apart from measures in the riparian zone, *stream restoration* can significantly improve the in-stream retention of dissolved nutrients. Channel reconfiguration, such as channel widening and meandering, and the restoration of structural complexity via the addition of flow obstructions (e.g., debris dams, side pools, and diversification of bed materials) may enhance nutrient uptake by increasing water residence time, promoting contact between the water and the sediment surface, and enhancing the hyporheic water exchange (Bukaveckas 2007; Craig et al. 2008; Hines and Hershey 2011). Woody material on the stream bed additionally increases the nutrient demand of the decomposing microorganisms due to the high C–N and C–P ratios (Roberts et al. 2007). Besides, debris dams provide organic carbon for in-stream denitrification (Craig et al. 2008). For example, the creation of riffles, cross vanes, and step pools within a restored stream shortened NH<sub>4</sub>–N uptake lengths from 200 to 70 m (Hines and Hershey 2011). Bukaveckas (2007) observed reductions in P and N uptake lengths from 1370 to 380 m and from 20 km to 620 m, respectively, after channel reconstruction and reconnection with the floodplain, while Roberts et al. (2007) measured reductions in NH<sub>4</sub>–N uptake lengths of about 50–70% after addition of woody debris in stream channels.

However, the efficiency of stream restoration measures on the in-channel nutrient retention has not yet been evaluated systematically so far, especially in comparison with management measures in the catchment and the riparian zone. Firstly, the majority of stream restoration measures primarily aim at restoring functions other than nutrient retention, such as channel stabilization or habitat diversity. Thus, effects of restoration on nutrient retention are seldom evaluated. Secondly, the tight connection between the various biogeochemical transformations of nutrients in the water and the sediments and the temporal dimension of these processes (e.g., diurnal patterns in primary production) complicates the evaluation of overall nutrient retention. Increased residence time, for example, may increase nutrient uptake by the biota but may also promote sedimentation of fine particles, creating anoxic conditions in the sediments which favor nutrient release (Weigelhofer et al. 2013). Finally, effects of reach-scale restoration measures on the water quality may be distorted by effects of catchment-scale factors, such as the hydrological regime of the catchment, the stream size, and the nutrient loading. For example, the positive effects of stream restoration are usually overshadowed by excessive nutrient loading in agricultural catchments (Weigelhofer et al. 2013) and by altered hydrographs with high stormwater flows in urbanized areas (Bernhardt and Palmer 2007). In those cases, stream restoration concepts should

incorporate the establishment of functional units within the stream course, which possess high nutrient uptake capacities, but often need certain maintenance activities. Examples for such functional units are in-stream sediment traps, in-stream wetlands, and slow-flowing stream reaches with planted submerged macrophytes (Filoso and Palmer 2011; Hines and Hershey 2011; Richardson et al. 2011). In the USA, such functional restoration concepts involving the creation of stream-wetland complexes in lowland streams have proven to successfully increase in-stream nutrient uptake (Filoso and Palmer 2011).

## 10.7 Conclusions and Open Questions

This review shows that efficient mitigation and management of nutrients in riverine systems need measures on both the catchment and the reach scale. On the reach scale, riparian zones are key components for nutrient retention. Stream restoration measures may additionally improve in-stream nutrient retention. However, in catchments with excessive nutrient loading, stream restoration needs a priori reductions of nutrient inputs into the riverine system to avoid detrimental effects on water quality through nutrient release from sediments.

This review also shows that the efficiency of the various measures in the riparian zone and the stream channel can vary widely depending on the environmental conditions. Thus, for a sustainable and efficient management of nutrients in riverine systems, more investigations are needed which evaluate and compare the nutrient retention efficiency of different management measures under varying conditions, considering especially the temporal variability of nutrient transformation processes. In particular, studies need to concentrate on small headwater streams which have the highest potential for nutrient retention due to their strong linkage with surrounding ecosystems. So far, these systems have been largely neglected in restoration efforts in Austria.

Future studies need also to address climate change impacts on nutrient cycling. Increased air and water temperatures may accelerate nutrient cycling in riverine systems, while altered hydrology may significantly influence nutrient input and exchange pathways. In addition, an increased variability in water temperature or water levels may change the natural balance of nutrient transformation processes, thereby impacting the nutrient cycling in the aquatic system. Thus, modern stream restoration needs to consider future changes of environmental conditions for a sustainable mitigation of nutrients in riverine systems.

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