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3.1   Introduction

Over the past 40 years, riparian corridors have been held up as an iconic interface between land and water. Yet, for several centuries, these riverine environments have been subject to intense anthropogenic disturbance such as urban development, clear-cutting for roads and agriculture, diking, straightening, and dredging. In the early 1980s, ecologists and biogeochemists demonstrated the importance of these vegetated ribbons alongside stream networks. Stream ecologists underlined the key role of riparian zones in structuring and fuelling riverine food webs (Vannote et al. 1980). In the meantime, from a more terrestrial perspective, biogeochemists unveiled the role of riparian forests in buffering stream nitrate input from upslope (Peterjohn and Correll 1984). These pluridisciplinary approaches have improved our understanding of the multiple functions of riparian ecosystems and led to the formalization of a more mechanistic definition of the ecotone concept (Weaver 1960). Hence, Holland et al. (1991) proposed to define an ecotone as a "zone of transition between adjacent ecological systems, having a set of characteristics uniquely defined by space and time scales, and by the strength of the interactions between adjacent ecological systems".

The growing understanding of riparian zones as part of the stream ecosystem builds on a perspective that Hynes advocated in his seminal paper *The Stream and its Valley* (Hynes 1975). Yet, this interface vision focused at the landscape scale has hidden a more complex biogeochemical functioning of riparian ecosystems that extends beyond a single interface between land and water, and which has led to some management simplifications and misinterpretations. For instance, several studies in the 1990s showed that the nitrate buffering capacity of the riparian zone was proportional to its width (see Mayer et al. 2007 for a review). These observations led to the establishment of minimal widths for riparian buffer strips, which are already used as a guideline for best management practices by many state environmental agencies (Lee et al. 2004). However, the potential for reducing

nitrate loads is highly variable among riparian zones. Later studies demonstrated that this buffer capacity is a function of water table elevation and nitrate residence time within the system (Hefting et al. 2004; Ocampo et al. 2006). Although riparian zone efficiency to remove nitrate was once thought to be independent to its location along the stream network, it is now clear that contact length between terrestrial upslope and riparian zone is the key driver for nitrate removal. Therefore, riparian zones adjacent to small streams, which comprise the greatest proportion of stream length, should have management priority (Pinay et al. 2015).

While nitrate can be permanently removed from the riparian zone as an inert gas to the atmosphere through microbial denitrification, there is no such mechanism for phosphorus. Although phosphorus may be temporarily stored in plants and soils, it is inevitably released back to subsurface waters under waterlogged conditions; riparian zones are thus typically inefficient in reducing phosphorus loads to streams (Nair et al. 2015). Climatic conditions can also strongly influence the potential for riparian zone denitrification by controlling groundwater table depth and moisture in surface soil layers (Butturini et al. 2003; Pinay et al. 2018; Poblador et al. 2017). Hence, widespread ideas are usually taken as *totems* during the management and restoration of riparian ecosystems. For instance, the pervasive potential of riparian zones as green nutrient filters, or "the wider, the better", do not take into account the complex interplay between topography, hydrology, soil, and vegetation along spatial gradients, and how these interactions and climatic patterns ultimately influence riparian biogeochemical processes.

In fact, riparian zones are constituted of four interfaces with different biogeochemical drivers and effects on stream water quality and function (Figure 3.1). The interface between: (i) upslope and wetland, (ii) wetland and stream surface water, (iii) stream surface water and groundwater, and (iv) stream surface water and the atmosphere. The location and the intrinsic characteristics of each of these interfaces sustain and/or constrain different physical and biogeochemical processes, which ultimately dictate whether riparian ecosystems act either as sources or sinks of the different elements considered. The following chapter will consider each of these four interfaces and their main physical, hydrological, and biological controls.

3.2   The Boundary between Upslope and the Riparian Wetland

Three seminal papers published at about the same time demonstrated the role of riparian zones as buffers of diffuse nitrate fluxes from upslope to their draining stream (Jacobs and Gilliam 1985; Lowrance et al. 1984; Peterjohn and Correll 1984). They developed a similar nitrogen mass balance approach between subsurface nitrate inputs from upslope through the riparian zone's wetland. They found that nitrate was disappearing within a few metres of its transit through the upslope wetland interface. They inferred that this buffering capacity was under the control of plant nitrogen uptake and microbial denitrification process. It was confirmed a few years later that eventually heterotrophic denitrification was the mechanism by which nitrate was removed as a gas (dinitrogen N_2 and nitrous oxide N_2O) from the ecosystem (Pinay and Décamps 1988; Pinay and Labroue 1986). Indeed, the
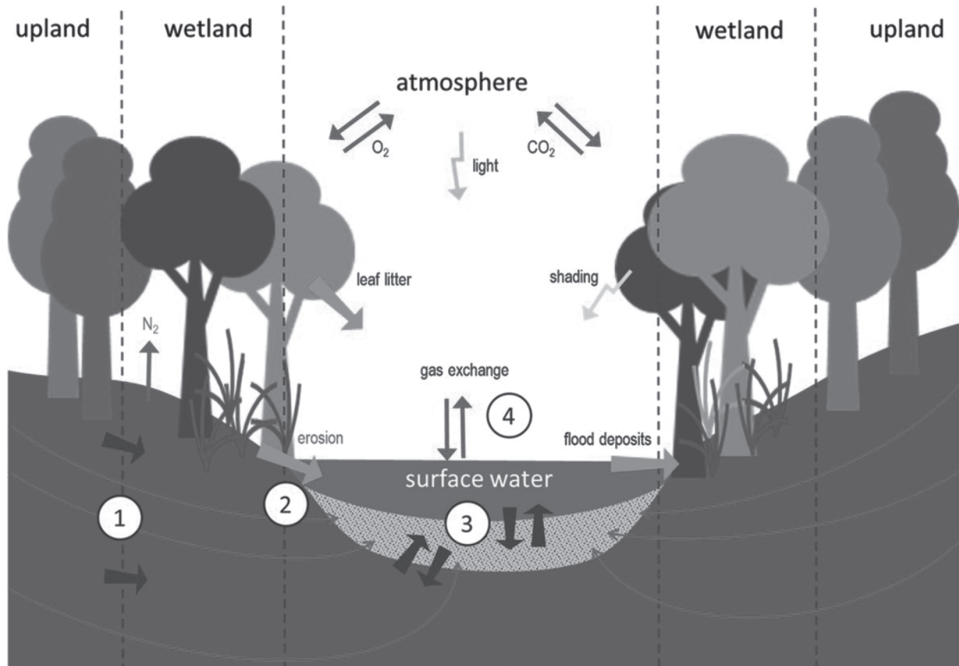
Figure 3.1   Schematic representation of the four interfaces that can be distinguished within the riparian ecotone. (1) The interface between upslope and wetland where N-rich subsurface waters from the hillslope can intersect C-rich riparian soils and promote denitrification and N gas emissions. (2) The interface between wetland and stream surface water, subjected to large sediment exchange during soil erosion and flood deposition processes, and where leaf litter inputs fuel stream metabolism. (3) The interface between stream surface water and groundwater, where vertical hydrological exchange controls biogeochemical processes within the hyporheic zone. (4) The interface between stream surface water and the atmosphere, control point of gas exchange fluxes and light inputs to the stream.

upslope boundary presents the necessary conditions for denitrification to occur (Knowles 1982), i.e. anaerobiosis caused by soil water-logging conditions, the presence of bioavailable organic matter from riparian forest leaves and roots exudates, and nitrate input from upslope subsurface flow. Wetlands classically present both soil anaerobic conditions and organic matter accumulation (Brinson 1993). Yet, denitrification would occur only if the third ingredient, i.e. nitrate, is added (Figure 3.2a). This has the implicit assumption that the wetland environment has to be connected to a source of nitrate for denitrification to occur. Hence, it might sound paradoxical in a world overloaded with nitrate, but the factor limiting denitrification in most wetlands and riparian zones is nitrate inputs (Figure 3.2b). This finding inspired the concept of the *biogeochemical hot spot*, a location showing disproportionally high reaction rates relative to the surrounding area, as it occurs in the upland–wetland interface of many riparian zones (McClain et al. 2003).

   Once nitrate supply is met, the second factor limiting denitrification is the residence time of nitrate in the riparian zone. If subsurface flow is faster than denitrification reaction rate, then nitrate reduction will not be complete, and a portion will escape towards the stream (Sabater et al. 2003). In this context, Ocampo et al. (2006) used the Damköhler ratio,
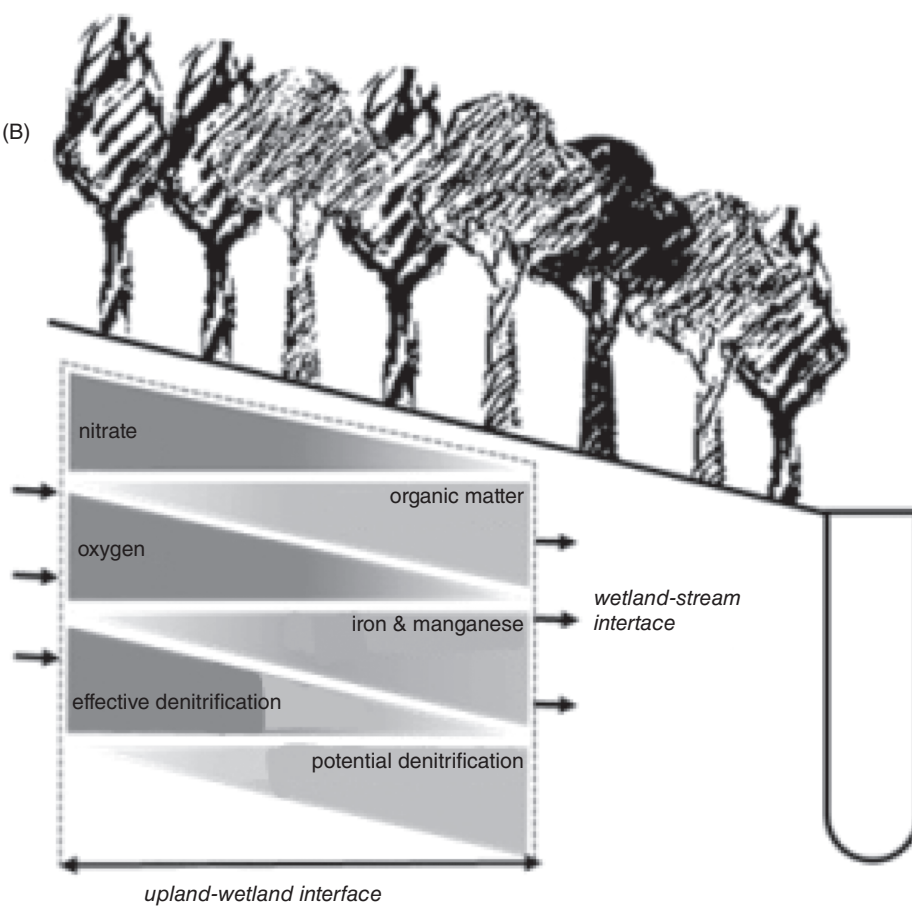
(A)

(B)

upland-wetland interface
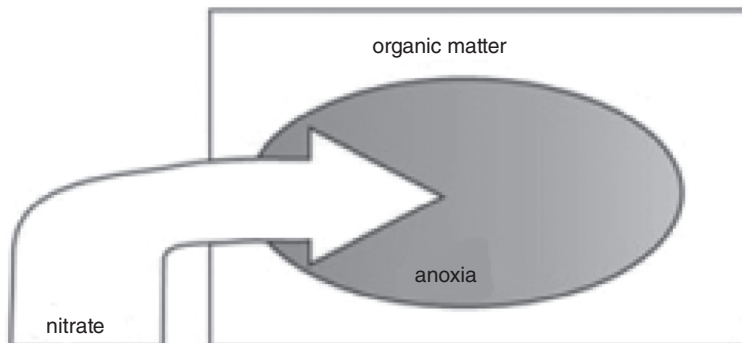
Figure 3.2   (a) Illustration of a biogeochemical hot spot for denitrification where nitrate flux intersects an anaerobic organic carbon-rich soil pool; and (b) major biogeochemical gradients along the upland–wetland interface and hypothetical associated effective and potential denitrification. Adapted from Pinay (1986) and McClain et al. (2003).

i.e. rate of nitrate input to rate of denitrification, to characterize and estimate the nitrate buffering capacity of riparian zones. They showed that the balance between transport and reaction is essential for determining the proportion of nitrate denitrified in riparian zones. Based on those findings, they proposed to use the Damköhler ratio as an indicator to define how wide a riparian should be, or to identify the most efficient riparian buffers within the landscape. In practice, riparian subsurface flows are so heterogeneous that heavy hydrogeological monitoring is needed to predict the local residence time of nitrate, which limits the possibility to use this type of approach in environmental management plans. Nonetheless, this research shed light on why riparian zone width is a poor surrogate of subsurface travel time, and thus, an insufficient criterion to warrant high nitrate retention in riparian zones. While predicting the nitrate buffering capacity of riparian zones at the local scale is highly uncertain, the likelihood of denitrification occurrence clearly increases with increasing the length of the upslope–wetland interface at the landscape scale. In other words, the higher the length of contact between upslope and riparian zone, the higher the potential for denitrification. The second management lesson to learn is that preserving and restoring riparian zones along small stream orders would be more efficient to remove upslope nitrate input than preserving large riparian areas at particular locations. This is because the former strategy will contribute to elongate the upslope–riparian wetland interface, while the nitrate retention efficiency of the latter will mostly rely on the local features of the selected riparian zone.

Heterotrophic denitrification is perhaps the biogeochemical process that naturally contributes the most to the permanent removal of reactive nitrogen from the biosphere. Yet, it should be mentioned that denitrification can also contribute to nitrous oxide emission, a potent greenhouse gas, since it is an intermediary product of the chain of reduction reactions from nitrate to nitrogen gas. Nitrous oxide emission can be particularly important at the beginning of soil rewetting during rainfall or a flooding event because of limited activation of N_2O-reductase enzyme by denitrifying bacteria (Firestone et al. 1980). Additional denitrification pathways have been identified such as Anammox – the microbial oxidation of ammonium to nitrogen under anaerobic conditions (Jetten 2001), nitrifier denitrification (Wrage et al. 2001), and autotrophic denitrification, which uses inorganic substrates as electron donors. The existence of these different biogeochemical pathways highlights the complexity of soil microbial communities and their large potential to respire oxidized nitrogen compounds. Increased attention has also been given to the interaction between the nitrogen and sulphur cycles (Burgin and Hamilton 2008), and also to carbon–nitrogen interactions, with a wide range of research focused on the type and form of carbon needed to drive denitrification and the sustainability of these electron donor sources (Newcomer et al. 2012). Research on the interaction between the nitrogen and iron cycles was also undertaken, proposing the ferrous wheel concept with the oxidation of ammonium to nitrite using ferric iron as an electron donor, and further nitrite reduction into nitrogen gas through classic bacterially-mediated denitrification (Clément et al. 2005). The ferrous wheel concept widened our understanding of the nitrogen processing within riparian zones by demonstrating that alternative biogeochemical pathways, independent of organic carbon sources, can occur and substantially contribute to the gaseous purge of nitrate from riparian soils.

3.3   The Boundary Between Wetland and Surface Water

The lateral interface between surface water and riparian wetland is one of the most dynamic and complex in river corridors. These environments facilitate reciprocal exchanges of energy, water, nutrients, and organisms across the terrestrial-aquatic gradient (Baxter et al. 2005; Nakano and Murakami 2001). Small changes in water table elevation can restructure the extent and reverse the direction of these exchanges (Raymond et al. 2016; Zarnetske et al. 2018), which further depend on seasonal variation in organismal phenology (Fellman et al. 2009; Gücker et al. 2016; Saurer et al. 2014). Consequently, understanding the wetland–surface water interface requires multi-scale monitoring and interdisciplinary approaches (Abbott et al. 2017; Lee-Cullin et al. 2018).

One of the most important functions of the wetland–surface water interface is export of terrestrial carbon to surface water networks, where it exists as dissolved (DOM) and particulate organic matter (POM). The input of POM from riparian zones increases reach hydraulic heterogeneity, slowing flow and increasing in-channel transient storage, leading to greater opportunities for microbial processing (Ensign and Doyle 2005; Roberts et al. 2007a). DOM is a fundamental component of the global carbon cycle (Lupon et al. 2020; Wologo et al. 2021), and forms the base of many stream food webs (Acuña et al. 2007). While sources of DOM vary, much of it derives from riparian leaf litter and soil inputs that provide the dominant fuel for in-stream respiration and secondary production, especially in small and intermediate-size streams (Ledesma et al. 2015; Zarnetske et al. 2018). Stream ecologists integrated the importance of riparian leaf litter inputs for stream food webs and biogeochemical processing into classical conceptual models such as the River Continuum Concept (Vannote et al. 1980) and the nutrient spiralling theory (Newbold et al. 1981, 1983). Reach-scale experiments have supported these conceptual models, empirically demonstrating that streams rapidly process allochthonous organic matter and nutrients from adjacent wetlands (Brookshire et al. 2005; Peterson et al. 2001). Moreover, studies conducting mass balance approaches have shown that in-stream processes largely modify riparian groundwater inputs of nutrients, and that, far from being recalcitrant, dissolved organic carbon from lateral groundwater inputs increase heterotrophic activity in some streams (Lupon et al. 2019).

A paradigm shift in the potent biogeochemical processing capacity of streams has replaced earlier views that carbon and nutrient inputs across the riparian interface (Figure 3.3) were conservatively advected downstream (Brookshire et al. 2009; Goodale et al. 2009; Siegenthaler and Sarmiento 1993). Indeed, the conceptualization of streams as inert pipes connecting terrestrial and marine environments is rapidly shifting towards a perception of these ecosystems as biogeochemical funnels and bioreactors. Global estimates suggest that approximately 75% of the carbon imported by freshwater ecosystems, including both particulate and dissolved forms, is outgassed to the atmosphere as carbon dioxide or sequestered in sediments (Cole et al. 2007; Raymond et al. 2013). Likewise, stream networks remove substantial proportions of nitrogen inputs (Alexander et al. 2000), predominately through denitrification (Seitzinger et al. 2006), where removal efficiency and pathway depend strongly on nitrogen and carbon stream concentrations (Burgin and Hamilton 2007; Mulholland et al. 2008). Critically, the concentrations of necessary reactants and environmental conditions required for in-stream nitrogen removal are controlled by the degree of connectivity along wetland–stream surface water interfaces.
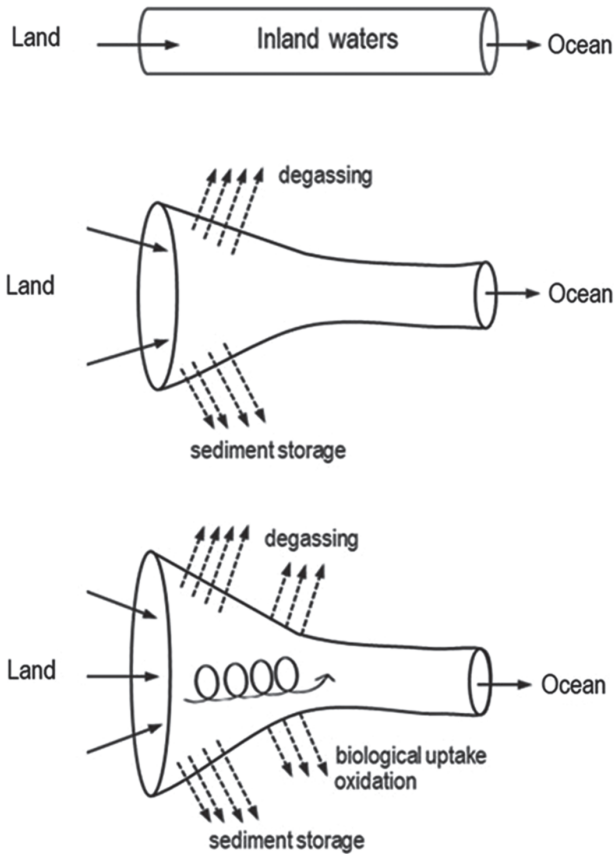
Figure 3.3 Perceptions of dissolved organic carbon in stream ecosystems. Adapted from Cole et al. (2007).

River systems and their riparian zones are dynamically linked longitudinally, laterally, and vertically by hydrologic and geomorphic processes (Ward 1989). The intensity of these processes varies from headwater to the river mouth with dominant erosional processes in the upper part and depositional processes in the lower reaches (Sullivan et al. 1987). River floodplains are important sinks for storing sediments and associated nutrients mobilized from upstream catchments during flood events. These transfers of energy, biotic, and abiotic matter in the floodplains are under the control of flood duration, frequency, and magnitude that create a mosaic of geomorphic surfaces influencing the spatial pattern and successional development of riparian vegetation (Roberts and Ludwig 1991; Salo et al. 1986). Flood characteristics control also the nutrient cycling intensity of floodplain soils and their impact on stream nutrient fluxes (Brinson et al. 1984; Mulholland 1992). Flooding directly affects nutrient cycling in alluvial soils by controlling the duration of oxic and anoxic phases (Ponnamperuma 1972; Tabacchi et al. 1998) (Figure 3.4).

For instance, it was found that the net nitrogen mineralization rate was four times greater in a spring-flooded marsh than in a non-flooded one, and that alternate aerobic and anaerobic conditions enhance organic matter decomposition and nitrogen loss through denitrification in flooded soils (Groffman and Tiedje 1988; Reddy and Patrick 1975).
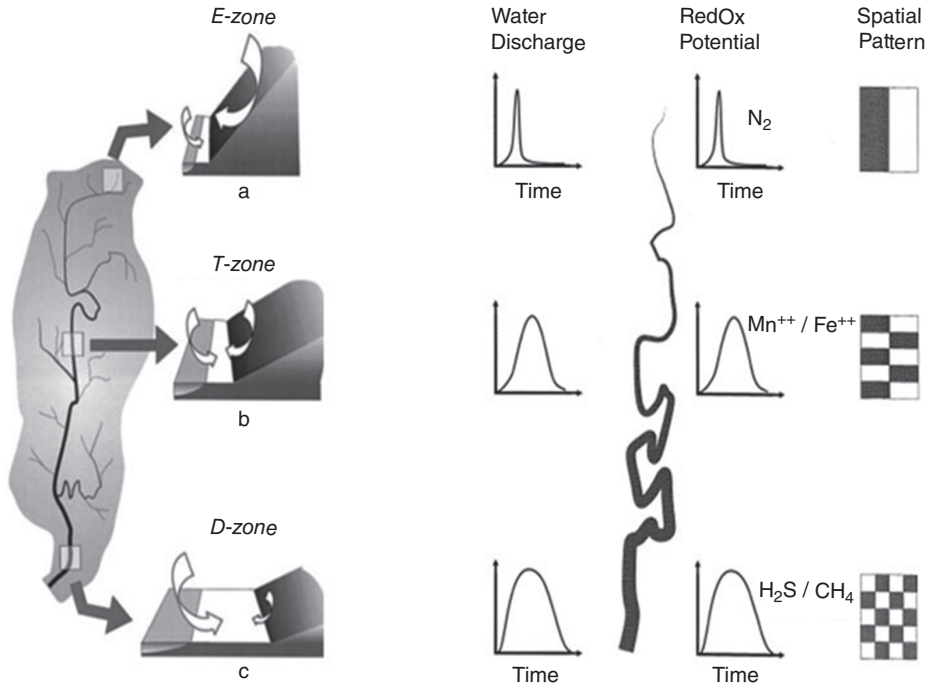
Figure 3.4 Illustration of the erosional (E), transitional (T), and depositional (D) zones along an idealized stream network. Differences in topographic and geomorphic features between these three zones result in different flood characteristics and spatial heterogeneity of soil properties and associated biogeochemical processes. Adapted from Tabacchi et al. (1998).

Flooding duration is controlled by local topography; low areas are flooded more often and longer than higher ones, producing large variations in biogeochemical patterns at a metre scale. Flooding also indirectly affects nutrient cycling in floodplain soils by influencing the soil structure and texture through sediment deposits. Hence, floodplain and stream channel geomorphic and hydrologic processes influence the sorting of flood sediment deposits on a grain size basis creating a mosaic of soils of different textures.

In a study realized on the Garonne River floodplain, Pinay et al. (2000) found that the floodplains' soil grain size could be a good proxy to estimate the likelihood of denitrification activity. Below a threshold of 65% of silt and clay content, the floodplain soils did not present any significant denitrification rates. Above that threshold denitrification increased significantly (Figure 3.5).

This relationship between soil grain size and denitrification was confirmed in another pan European study (Pinay et al. 2007). These results confirm the importance of texture on soil nitrogen cycling processes and suggest that soil grain size could be a good proxy at the reach scale to determine where denitrification occurs in a floodplain.
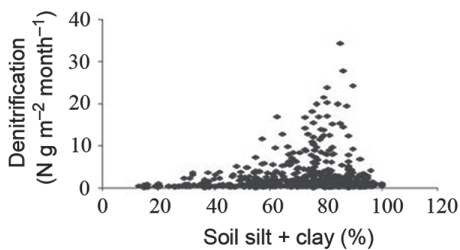

Figure 3.5 Relationship between the floodplain's soil grain size and denitrification in the Garonne River floodplain. Adapted from Pinay et al. (2000).

3.4   The Boundary between Surface and Groundwater

The riparian zone extends beneath the hyporheic zone, a highly dynamic subsurface region that encompasses both the stream channel and the riparian zone itself. The hyporheic zone is defined as the portion of sediments surrounding the stream that continuously exchange water and solutes with the stream (Boano et al. 2014). By definition, the hyporheic zone implies water exchange at relatively small spatial scales, typically from centimetres to metres, and it is characterized by slowly moving waters, which increase the hydrological opportunity for biogeochemical interactions (Battin et al. 2008). Yet, the area of influence of the hyporheic zone can be highly variable across sites depending on the hydromorphological characteristics of the river corridor as well as on a seasonal scale with changing climatic conditions (Harvey and Bencala 1993; Wondzell and Swanson 1996). Groundwater–streamwater exchange can also occur at larger spatial scales, from hundreds to thousands of metres, and this continuous gain and loss of water has large implications for the biogeochemical processing of solutes along river corridors (Covino and McGlynn 2007). In fact, the transition between riparian groundwater, hyporheic zone, and surface stream water can be understood as a continuum of hydrological and biogeochemical conditions, precluding delineation of a physical boundary between these water bodies (Figure 3.6).
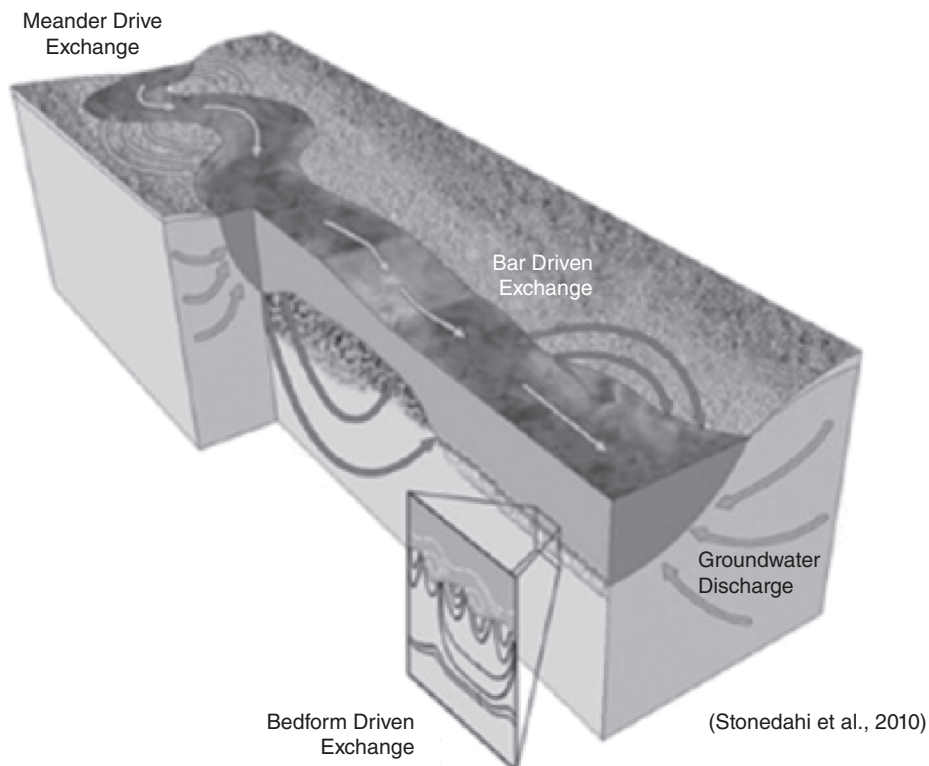

Figure 3.6   Hyporheic flows. Adapted from Stonedahl et al. (2010).

Hydraulic gradients usually promote the movement of water from hillslopes towards the stream, especially in headwaters. Thus, it is often assumed that groundwater preferentially flows from the riparian zone towards the stream. However, rather than being unidirectional, the hydrological exchange between riparian groundwater and stream water is highly dynamic over space and time depending on topography and watershed structure (Covino et al. 2011; Jencso et al. 2009). Further, there can be large mixing of groundwater and stream water in the riparian zone, especially in large alluvial valleys such as reported for the Garonne River in southwest France (Pinay et al. 1998). In semi-arid areas, water losses from the stream towards the riparian zone are commonly observed during low flow periods when hydraulic pressure from hillslope groundwater is small (Shade et al. 2005). Riparian tree evapotranspiration can further control water exchange fluxes at the stream–riparian interface by dropping down groundwater tables and favouring the movement of stream water towards the riparian zone at both daily and seasonal scales (Lupon et al. 2016; Wondzell et al. 2010). This ecohydrological process can exert a strong control on stream discharge in semi-arid areas, favouring stream channel desiccation and the lateral infiltration of stream water up to dozens of metres within the riparian zone, especially during periods with low hydrological connectivity (Bernal et al. 2013; Butturini et al. 2003). This is why planting riparian trees along buffer strips without considering local climate conditions can be a counterproductive management strategy in some cases.

Vertical and lateral hydrological exchange at the stream–riparian interface is accompanied by the exchange of dissolved organic matter and nutrients. The mixing of electron donors and acceptors along thermodynamic gradients from the riparian zone towards the near-stream area can promote intense microbial activity, and host high rates of biogeochemical processes within the hyporheic zone. For instance, Hedin et al. (1998) reported that denitrification at the riparian zone of the Smith Creek, a first-order stream in Michigan (USA), was mostly constrained to a very localized area where horizontal shallow subsurface flow rich in dissolved organic carbon interacts with nitrate-rich vertical upwelling of deep subsurface waters in the near-stream zone. They proposed that those areas of hydrological interaction may act as "control points" for fluxes of nitrogen and other nutrients at the soil–stream interface. In this line of thought, Zarnetske et al. (2011) proposed that thermodynamic gradients, from oxic to reduced conditions, organize along hyporheic flowpaths, so that the prevalence of either nitrification or denitrification is a function of the residence time of water and solutes in the hyporheic zone. There are also beautiful examples illustrating the transfer of water and solutes from the stream towards the riparian groundwater. Pinay et al. (2009) showed that nitrate mineralized from the carcasses of Pacific salmons in a small stream in Alaska (USA) was rapidly taken up by biota along hyporheic flowpaths from the stream towards the riparian zone. In Sycamore Creek (AZ, USA), riparian trees were enriched with the ^{15}N previously added to the adjacent stream, highlighting that solutes in the stream water column travel towards the riparian zone (Shade et al. 2005). Overall, these studies highlight the strong potential for biogeochemical interactions at the surface water–groundwater interface, which requires bidirectional fluxes between surface water and groundwater. The management lesson to learn from these studies is that maintaining and recovering kilometres of free flowing waters along the stream network contributes to naturalize hydrological exchange fluxes between hyporheic and surface waters, and thus, promote ecological and biogeochemical functions at this terrestrial–aquatic interface.

3.5   The Boundary between Stream and Atmosphere

A hierarchy of vertical interfaces controls the exchange of energy and materials between the stream and the atmosphere. Successive attenuation of solar energy by canopy shading, reflectance from the water surface, and absorption in the water column results in a small fraction of light energy entering ecosystems being converted to chemical energy via photosynthesis (Kirk 1994). Hence, the light regime – the timing, mode, and magnitude of photosynthetically active radiation (PAR) available for gross primary production (GPP) – for stream autotrophs differs from terrestrial autotrophs. Stream ecologists hypothesized that light regimes, and therefore GPP, would vary according to riparian shading (Minshall 1978), and that shading patterns were manifest in predictable network patterns of longitudinal increases in channel depth and width (Vannote et al. 1980). Subsequent efforts confirmed that indeed, light regime is the dominant driver of lotic GPP variation across biomes (Mulholland et al. 2001) and over time (Roberts et al. 2007b). However, complexities from riparian phenology (Hill and Dimick 2002) and variation in water column attenuation due to suspended sediment and coloured organic compounds (Davies-Colley and Nagels 2008; Davies-Colley and Smith 2001; Julian et al. 2008) limit the universal predictive capacity of atmospheric PAR for GPP, especially when autotrophs are attached to the stream benthos (Bernhardt et al. 2018). Still, recent improvements in modelling the true light availability for stream benthic autotrophs are increasing our predictive abilities (Julian et al. 2008; Kirk et al. 2020), and are further highlighting how the linked interfaces among atmosphere, riparian vegetation, and stream act in concert to control ecosystem productivity and function.

Stream water temperature represents the balance of energetic fluxes across the surface water–groundwater and surface water–atmosphere interfaces (Hannah and Garner 2015). In headwater streams, groundwater influence limits equilibrium with the atmosphere especially during the summer period (Edinger et al. 1968; Hrachowitz et al. 2010; Kelleher et al. 2012). In reducing solar radiation, riparian vegetation further increases the relative importance of streambed heat flux from groundwater. This is especially noticeable during summer when the net solar radiation is the dominant heat flux input (Hannah et al. 2008; Malcolm et al. 2008). Therefore, riparian forest shading reduces summer maximum daily stream water temperatures (Johnson 2004; Moore et al. 2005). The cooling effect of vegetation has been reported mainly for small streams (Garner et al. 2014; Johnson and Wilby 2015; Moore et al. 2005). It may be possible to maintain daily maximum water temperatures close to 20°C with shade levels of 70% in headwaters (Rutherford et al. 1997). For conditions at midday in July, about 74% decrease can be measured in net energy gain from an open reach under full sun to a full shaded reach (Johnson 2004). The effectiveness of riparian shading depends mainly on channel orientation, canopy density, and within-reach residence times (Garner et al. 2017). For instance, Garner et al. (2017) demonstrated that for reaches under high flow velocity and 30% vegetation density, stream temperatures varied by up to 0.8°C and 2.7 °C for mean and maximum daily temperatures, respectively. Under the same vegetation cover with a low velocity, temperatures varied by 2.7° and 4.3 °C for mean and maximum daily temperatures, respectively. They also showed that an increase in canopy density (from 10 to 90%) could decrease both maximum (≥3.0 °C) and mean (≥1.6 °C) daily temperatures. Riparian clear-cutting can increase maximum daily water temperature by up to 8 °C (Gomi et al. 2006; Johnson and Jones 2000), and can gradually return to pre-harvest temperature after a 15-year regrowth period (Johnson and Jones 2000).

Johnson and Wilby (2015) showed that approximately 0.5 km of complete shade is necessary to off-set stream temperature by 1 °C at midday in July in headwaters; whereas at 25 km downstream, 1.1 km of shade is required.

As rivers increase in size, their thermal regimes become dominated by inertial effects and are increasingly insensitive to the height and phenology of surrounding riparian vegetation (Caissie 2006). Yet, few studies have already modelled riparian shading by riparian canopy at any time and location for a given stream depending on stream azimuth and river width, canopy height, and overhang (Li et al. 2012). Integration of such model and dynamics of LIDAR-based riparian vegetation into a physical process-based thermal model showed a decrease in maximum daily water temperature ranging from –3 °C (upstream) to –1.3 °C (downstream) for medium streams (distance from source for 40 km to 300 km) as well as small streams (Loicq et al. 2018).

Despite this buffering effect of the riparian zone on temperature maximum in headwaters, stream water thermal regimes are generally synchronous with air temperature (Beaufort et al. 2020a). Yet, anthropogenic impoundments such as large dams, small reservoirs, and ponds can influence downstream thermal regimes (Beaufort et al. 2020a; Seyedhashemi et al. 2020). In this context, the riparian vegetation cooling effect can mitigate ponds' heating impact on stream thermal regimes (Maxted et al. 2005; Seyedhashemi et al. 2020).

3.6   Impact of Global Change on the Functioning of These Interfaces: Future Challenges and Opportunities

Climate change is one of the greatest challenges facing humanity, and the informed management of natural resources, particularly riparian interfaces, will be a critical component of our response. For example, increasing temperatures – particularly decreased winter minima and increased summer maxima – will shift the hydrological balance, likely reducing snowpack and storage, and increasing potential evapotranspiration. These effects will likely be regionally specific, necessitating continued local monitoring and integration of broad scientific understanding with local stakeholder knowledge. For example, in Europe, increased precipitation is likely to occur in winter in the northern areas, while southern Europe is expected to suffer a decrease in precipitation (recent references, Habets et al. 2013). Compounding the potential for reduced water supply, water demand is expected to increase due to growing irrigation needs and an expanding energy sector that requires water for cooling (Abbott et al. 2019).

As climate change shifts the spatiotemporal balance of energy inputs and water availability at the riparian interface, mismatches between biological and biogeochemical processes may arise. For example, hydrological connectivity of the river network may fragment, decreasing biodiversity due to phenological dependence of organisms on saturating or flowing conditions. Likewise, riparian soil saturation, a requirement for denitrification that limits landscape nitrate loads to streams, may reduce in extent. Land cover and land-use change factors can amplify these mismatches. Agriculture intensification leads to higher/chronic nutrients and pesticides leakages, and the timing of inputs is likely to occur when potential removal rates in the riparian zone are minimized (e.g. in the dormant season, Van Meter et al. 2020). It is therefore critical to maintain, conserve, and re-establish

hydrological and habitat connectivity across the riparian interface to support the biodiverse ecosystems on which we depend.

Critically, the how-to of this important effort, e.g. by preserving and expanding riparian vegetation, is an emerging issue (Dufour et al. 2019), and how we use our knowledge of riparian interfaces to interface with stakeholders and decision-makers remains a difficult, but not insurmountable, challenge. We can use our position of expertise to suggest policy measures, for example, an "ecological re-allotment programme" to restructure the landscape. We are also in dire need of a rehabilitation of public perception of and relationship to riparian corridors; these environments have always been a focus of community life (transportation, recreation, food gathering, and water provisioning). We can conduct outreach programmes that focus on sustainable agriculture and wise application of fertilizers via the RRRR approach (Right type, Right amount, Right place, and Right time).

We can still do more to understand and support our view that riparian zones are critical interfaces in the landscape, with (in)valuable services provided to us at little-to-no cost. While we know that long hydrological travel times through subsurface hot spots of biogeochemical reactivity are the most efficient for removing undesirable solutes from reaching streams, we still lack a landscape perspective on how to design riparian interfaces that maximize this connectivity. How can we improve the drainage and imperviousness of agricultural soils to reduce the rapid bypass of nutrients and pesticides to the stream network? What is the ideal landscape arrangement of hedges, agricultural fields, and riparian zones that maximizes nutrient removal? Global changes bring enormous challenges to environmental scientists, but also unveil grand opportunities to learn, engage, and transform how humans interface with riparian zones.

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