A Review of the Hyporheic Zone, Stream Restoration, and Means to Enhance Denitrification

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A Review of the Hyporheic Zone, Stream Restoration, and Means to Enhance Denitrification

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Abstract

The hyporheic zone is the subsurface area below and adjacent to a stream where groundwater mixes with stream water, through vertical, lateral, and longitudinal flows. The hyporheic zone connects the stream to uplands and other terrestrial environments. It is a zone of distinct faunal communities, high biological diversity and ecological complexity, and is the site of chemical processing and transformations of ground- and stream waters. The hyporheic zone is important to the overall ecosystem ecology of the stream, and it can influence stream water chemistry. Flows, reactions, and biota in the hyporheic zone are heterogeneous and patchy, making it difficult to clearly describe the ecotone in a straightforward, general way. Nitrogen processing, especially denitrification, appears to be widespread in the hyporheic zone.

The hyporheic zone, as with most aquatic systems, is often impacted by human activities. Stream restorations rarely consider potential effects on the hyporheic zone, but careful project choices can enhance the condition of the hyporheic zone, and so increase uptake of nitrogen by stream-associated environments as partial mitigation of continuing and increasing releases of reactive nitrogen, potentially reaping short-term benefits to estuarine environments that might not be as quickly realized from source control measures.

1. **The Hyporheic Zone: Definition**

The hyporheic zone is the area below and adjacent to the streambed where surface water and groundwater mix (Fig. 1). It is not at one single, easily defined place, but rather is a diverse set of elements (Boulton et al. 2010) that define an ecotone (Williams et al. 2010), and its attributes vary considerably over time and space (Williams 1993; Poole et al. 2006; Kaser et al. 2009; Zlotnik et al. 2011), so that its defining attribute may be its dynamism (Boulton et al. 2010). It is also difficult to monitor so as to produce data with wide applicability (O'Connor and Harvey 2008). Because the hyporheic is an often overlooked, underappreciated element of the environment, we begin with a general discussion of the traits of this ecologically important area. This lengthy exposition of hyporheic zone attributes and its ability to host denitrification will support details of hyporheic zone impairments, and the means by which the hyporheic zone can be remediated, especially to promote denitrification. Other reviews in this journal (Birgand et al. 2007; Garcia et al. 2010; Kadlec 2012) have addressed aspects of the growing aquatic nitrogen pollution problem, and it is our intention to supplement this literature. Potentially, this body of work will affect management decisions relating to restorations, although it has been noted that few stream restorations are directly shaped by scientific research and reports (Bernhardt et al. 2007).

The hyporheic zone has been described differently in terms of its hydrology, geochemistry, and ecology. Hydrologically, the hyporheic zone is the interstitial spaces adjacent to the stream bank and below the streambed that are saturated and contain some portion of channel water (White 1993), especially when modified to "<98% stream water and >10% groundwater" (Triska et al. 1989b; Boulton et al. 2010). Water quality that results from mixing stream and groundwater in the subsurface can be distinct from both (Fraser and Williams 1998; Hill et al. 1998; Hayashi and Rosenberry 2002), and can be further modified by biologically-mediated redox reactions (Jones and Holmes 1996; Storey et al. 1999). The hyporheic zone was first identified as a region with unique biota (Orhigdan 1959), some associated with streams or groundwater (Boulton 2007), but many others that are distinctive (Boulton et al. 2010). The sum of conditions create a transition area between two distinct ecological regions, and it has been suggested (Williams et al. 2010) that it fits the definition of an "ecotone" (Odum 1971). Water flow is the dominant driver of most processes, and so here the hyporheic zone is almost always considered as the mixing zone for groundwater and stream water.

2. Hyporheic Zone Attributes

2.1 Hyporheic Zone Hydrology

A useful simplification is to consider that essentially all baseflow of streams was once groundwater (Williams 1993; Winter 2000; Hayashi and Rosenberry 2002; Sophocleous 2002), although it is clearly not true in all particulars (e.g., Seitzinger et al. 2006). The subsurface is key for stream flows and, generally, flow through the hyporheic zone is from groundwater to surface water. At some point bankward and downward from the stream, all flow is classified as being "groundwater;" but, at the stream-sediment boundary, assuming permeable sediments create hydraulic conductivity, there is bidirectional flow between stream and sediments (Triska et al. 1993), even if only driven by diffusive flows.

Any small difference in pressure between subsurface and surface water causes interchange between them: upwelling zones where subsurface water enters into the stream; and, downwelling zones where stream water enters the hyporheic zone. These zones can range from cm^2 to km^2 in area (Reidy and Clinton 2004), although most are limited in extent (Runkel et al. 2003). Under steady-state conditions, discharge dominates at stream banks and the streambed closest to the bank, so that downward hyporheic flow is most likely near the center of the streambed; residence time in the subsurface is less near banks and greatest at the streambed center (Boano et al. 2009). Most stream reaches are comprised almost entirely of discharging zones (Conant 2004), although continuous areas of discrete upward and downward flows increase downstream (Gooseff et al. 2006).

Flows originating in the stream can be oriented longitudinally (along the stream path) or laterally into the stream bank. Small vertical head differences between the stream and the subsurface waters create longitudinal flows (Williams, 1993; Holmes et al. 1996; Olsen and Townsend 2003). The standard model for steady-state flows has head-driven inflows at the top of riffles and outflows (including groundwater discharges) concentrated at the foot of riffles and the head of pools (Williams 1993; Hill et al. 1998; Hayashi and Rosenberry 2002; Kaser et al. 2009). Transitory changes in stream conditions (including flooding of previously dry areas) from phenomena including run-off from precipitation, snow-melt, larger scale flooding, and/or spates from dams, can create significant head differences over larger areas (Poole et al. 2006; Boano et al. 2007; Kaser et al. 2009; Maier and Howard 2011). Such “event flows” may actually define most flow conditions in many streams.

Longitudinal flows can be also initiated by flow-driven pressure differences on bedforms, with upstream faces serving as points where surface water enters the subsurface (Thibodeaux and Boyle 1987); this is called “advective pumping” (Worman et al. 2002), and was first described by Vaux (1968) analytically. Thus, increased flow rates in the stream can drive greater exchange (Fraser and Williams 1998), without any changes in surface water-subsurface water head differences. Pressure variations associated with turbulent flow can be sufficient to cause hyporheic exchange even in the absence of substantial bedform relief (Packman et al. 2004). The frequency of bed sediment reworking means particular bedform conditions may not be maintained at any one location over any long period of time (Dole-Olivier et al. 1997; Fowler and Death 2001; Fischer et al. 2005; Poole et al. 2006; Boulton et al. 2010; Robertson and Wood 2010; Stubbington 2012), underscoring the dynamic nature of the phenomenon.

Lateral flows may be driven by transitory elevated stream heights (creating "bank storage"), follow paleochannels across flood plains, or be created by head differences between meanders in the stream path (Triska et al. 1993, Wroblicky et al. 1998; Sophocleous 2002; Cardenas 2008, 2009). Hill et al. (1998) also attributed lateral flows to head differences stemming from riffle-pool sequences (which create differences between stream and subsurface water elevations) and considerable flow appears to occur parallel but outside of stream channels – in the “alluvial” aquifer (Poole et al. 2008). Lateral flows have been less studied due to their longer residence times (Runkel et al. 2003). Most studies of lateral hyporheic flows focus on meander-driven flows, which appear to be greatest from hinge points, and may be persistent even in settings with large groundwater discharge rates (Cardenas 2009). Channel sinuosity leads to variable and irregular flows through point bars, too (Cardenas 2008). Horizontal flow rates, whether lateral or longitudinal, span a wide range from 1 cm d^{-1} to 43 m d^{-1} (Kaplan and Newbold 2000).

Bank storage is a special kind of hyporheic zone feature. Bank storage is created when stream water absorbs into side sediments, often because stream levels rise faster than water tables following precipitation (Gulley et al. 2011). This dynamic storage returns stored water as water levels fall in the stream, buffering stream flow rates. Some bank storage can be formed through subsurface flows in the vadose zone associated with precipitation. Although bank storage is often connected to the water table, it is more a stream than groundwater feature

(Brunke and Gonser 1997). More bank storage occurs when stream flows are larger and where sediment hydraulic conductivity is greater (Wroblicky et al. 1998).

Most flow in the subsurface is downstream (Poole et al. 2006) with a hierarchical mixture of long and short flow paths that have different residence times in the subsurface (Fig. 2) (Poole et al. 2008). These paths result from the synchronous mixtures of processes that are primarily vertical (along the flowpath of the stream) or horizontal (across meanders) and result from the heterogenous distributions of sediments (Liao and Cirpka 2011). This means that “mean” or “median” measures of residence may be misleading, as the range (minutes to months to years) (Harvey and Wagner 2000; Reidy and Clinton 2004) is affected by whether flow is through, say, a ripple or across a major meander.

The amount of stream water in a reach that enters the hyporheic zone has been estimated over several orders of magnitude, from <<1% to as much as 460% (Jones and Holmes 1996; Burkholder et al. 2008), partially because of the undefined length of a "reach", but also because factors affecting hyporheic exchange are so variable. High sediment conductivity, streambed roughness, and low groundwater pressures result in more surface water exchange, and fine bed sediment and high groundwater pressures result in much less exchange (Duff et al. 2008). Surface water tracer experiments often generate substantial “tails” (retarded tracer not advected with general stream flows), which has been interpreted as evidence of much mixing between the stream and subsurface (Worman et al. 2002). A model of a New England river estimated 50% of run-off entered the hyporheic zone at least 2.5 times, but also found a 3:1 ratio for time in the main channel to time in the hyporheic zone (Stewart et al. 2011).

2.2 Hyporheic Zone Geochemistry

Interactions between surface waters and the subsurface can lead to geochemically-driven changes in important stream attributes (Bencala 1983). Redox chemistry, the set of reactions requiring electron acceptors (these molecules become “reduced”) and electron donors (these moelcules become “oxidized”), is key. Carbon (C) in its various forms plays a key role in redox reactions in biologically active systems. In the presence of oxygen (O_2), the compound most likely to become reduced (accept electrons) is O_2. In the absence of O_2 (lower redox states), other compounds act as terminal electron donors. As redox potentials decrease, the electron acceptors that become thermodynamically favored are, in order: nitrate (NO_3^-) to dinitrogen (N_2) or ammonia (NH_3); manganese from valence state +4 to valence state +2; iron from valence state +3 to +2; sulfate (SO_4^{-2}) to sulfide (HS^-); and carbon dioxide (CO_2) to methane (CH_4). Most redox reactions are microbially-mediated (Hedin et al. 1998).

In shallow flow groundwater systems, the groundwater usually has short residence in the subsurface and typically is well-oxygenated (Gold et al. 2001; Storey et al. 1999) because it usually does not contain great enough dissolved organic carbon (DOC) concentrations to support sufficient metabolism to consume much O_2 over short periods of time (Gold et al. 2001). Therefore, it is often only older groundwater associated with longer, deeper flow paths that may be anoxic (Malcolm et al. 2003; Robertson and Wood 2010) and can support redox reactions using terminal electron acceptors other than O_2.

It is generally assumed that stream waters are well-oxygenated (Robertson and Wood 2010), although low dissolved oxygen (DO) surface water conditions can be created behind natural or artificial dams or in backwaters and channels, where organic matter accumulates and water advection is low (Baker et al. 2000). Therefore, absent any local biological consumption of O_2, downwelling zones (stream water) are usually better oxygenated than upwelling zones (groundwater).

Hyporheic flow paths are thought to encounter enough organic C to support sufficient respiration to deplete O_2 in the shallow subsurface (Jones and Holmes 1996); generally, hyporheic zone O_2 concentrations are inversely related to residence time (Findlay 1995). This, combined with increased contributions from groundwater, means upwelling water is often much lower in DO than downwelling water.

The location of the hyporheic zone at the interface between the upland and the stream suggests it will affect stream chemistry, especially nutrient dynamics. The hyporheic zone is a transition from open water to water-sediment conditions; in it, electron donors and acceptors change over a patchy mosaic; and, parcels of water appear to cycle back and forth in and out of the zone. These factors produce effects on overall stream chemistry (Hedin et al. 1998; Dahm et al. 1998; Baker et al. 2000). The assumed presence of a redox gradient associated with DO depletion allows the hyporheic zone to be considered a geochemical “hot spot,” where changes in redox conditions in the presence of diverse chemical substrates (associated with sediments or in solution) fosters chemical transformations (Hedin et al. 1998), particularly denitrification (Baker et al. 2000). One by-product of respiration is CO_2, and so pH values in the hyporheic zone are often different from both groundwater and the stream (other reactions, many enhanced by sediment-water connectivity, can also result in pH changes) (Runkel et al. 2003). The distribution of reaction sites is spotty over both time and space (Hedin et al. 1998); shifts in subsurface flows due to changes in stream or groundwater flows can cause relocation of reaction sites. These shifts may be predictable, especially if seasonal patterns (flood, drought, precipitation) are the drivers of changes in flow (Dahm et al. 1998).

2.3 Hyporheic Zone Biology

It is difficult to disentangle the chemistry of the hyporheic zone from its biology; in addition, stream remediation project managers often focus on biological attributes, including invertebrate populations which depend on hyporheic conditions (Bernhardt et al. 2007; Lake et al. 2007). Thus, we will briefly discuss some of the more notable biological attributes of the hyporheic zone.

The hyporheic zone serves two important, overt ecological purposes for stream fauna: refuge for invertebrates in times of disturbance; and prime egg incubation sites. Both stem from the perceived greater environmental stability of the hyporheic zone compared to open waters (Orghidan 1959), a function of water velocity and temperature. Water velocity decreases upon entering the hyporheic zone, by factors as much as 10^{-3}, compared to surface water (Brunke and Gonser 1997), creating shelter for stream invertebrates when water column currents increase (Boulton et al. 1998). Krause et al. (2011) describe temperature as the "master variable" that controls all other hyporheic zone processes. Water temperature fluctuation is generally less than in surface waters, and its variability decreases with increasing depth and distance from stream water infiltration sites (Brunke and Gonser 1997). Thus, the asynchronous pulses of water from the hyporheic moderate stream water temperatures: on short time scales night time lows are warmed, and daytime highs are reduced; on longer flow paths, hyporheic flows equilibrate with groundwater temperatures, further mitigating daily or seasonal temperature fluctuations (Poole et al. 2008). Careful measurements of temperature changes in streams have even been used to quantify hyporheic exchange rates (Westhoff et al. 2011). Consistent temperatures are advantageous for salmonid spawning, and subsurface temperatures are more constant in upwelling zones (van Grinsven et al. 2012).

Organism sheltering generally occurs in downwelling areas, allowing benthic populations to rapidly recover from events (Dole-Olivier et al. 1997), although major floods require longer recovery periods (Maier and Howard 2011). Stubbington (2012) notes refuge utility is a function of interactions between sediment types, taxon, the kind of disturbance, and whether the flight from disturbance is active or passive, so that *which* organisms seek refuge *where* and *when*, while determinable, is not consistent. In addition, at least some of these disturbance events affect the hyporheic environment, altering the refuge and its functionalities (Boulton 2007; Robertson and Wood 2010).

Changes in sediments, water flows, and associated conditions make the hyporheic zone a patchwork of small, differentiated habitats. Broad generalizations of benthic sediment patterns, for instance, might include downwelling gravel patches at the head of a riffle, upwelling gravel areas dominated by hyporheic flows at the riffle foot, with upwelling groundwater immediately adjacent (although perhaps associated with finer sediments of the pool), and other areas of sediment and flows associated with meander erosion (Boulton 2007).

Hyporheic zone organisms include microbes (Findlay and Sobczak 2000), meiofauna (Hakenkamp and Palmer 2000), and macrofauna (Boulton 2000). Most faunal characterizations concentrate on insect instars (transient members of the ecosystem) (Boulton et al. 2010). For many biologists, the hyporheic zone is defined by the class of micro- and macro-invertebrates called “hyporheos”: crustaceans, segmented worms, flatworms, rotifers, water mites, and juvenile stages of aquatic insects (Williams and Hynes 1974). The eggs and alevin of salmonid fish are also members, and often are the focus of hyporheic zone management programs; the areas they live, excavated and then backfilled by adults, are “redds” (Environment Agency 2009). Differences in overall hyporheos distributions are a function of DO (Hakencamp and

Palmer 2000), but also are affected by grain size variations and vertical flow patterns (Olsen and Townsend 2003).

Bacteria are important elements of the hyporheic zone ecology, and can create biofilms. Biofilms foster the creation of micro-environments – small anaerobic zones in otherwise oxygenated settings, for instance – that appear to be required for reactions such as denitrification (Storey et al. 1999). Biofilms create specialized environments due to expressed enzymes and the restricted size of pore space environments, but the supply of nutrients and dispersion of wastes is controlled by the rate of advected waters passing them (Findlay and Sobczak 2000).

3.0 Hyporheic Zone Nitrogen Transformations: Research Findings

Krause et al. (2011) summarized current research on N in streams, finding it can be transformed, mobilized, or returned to the atmosphere at different rates over relatively small scales, and that these processes differ for particular streams. The concept of nutrient “spiraling” (Webster and Patten 1979) is helpful: it illustrates, in this case, N moving from organisms to a variety of reservoirs, and being carried (predominantly) downstream via diverse pathways, including subsurface routes, with repetitive cycling of flows, forms, mineralization, and organism uptake. The retention of nutrients for at least some time is necessary in order to maintain ecosystem processes in streams, and so the spiraling concept illuminates stream ecological dynamics (Triska et al. 1989a, Ensign and Doyle 2006). In headwater streams, groundwater is the primary source of N, although much N enters the system as organic N from leaf litter and sediment inputs (Duff and Triska 2000). Transformation from organic to inorganic forms is expected, along with considerable lags due to incorporation into organisms or sorption onto sediments, so that the transport of N is considerably delayed along each stream reach compared to a non-reactive tracer (like chloride). Nitrogen spiraling (Mulholland and DeAngelis

2000) describes the repeated transformation of N from inorganic to organic forms, and its track from the main body of the stream into sediments. The spiral has “uptake length” (the distance an atom travels before being biologically retained) and “turnover length” (the distance traveled by the atom as organic matter) (Duff and Triska 2000). Hyporheic exchange should reduce uptake length and increase turnover length by increasing interaction between sediments and the water column (Mulholland and DeAngelis 2000). Denitrification is, in a sense, a form of completion of the spiraled pathway as the N-atoms are thus lost to the system.

3.1 Denitrification

Denitrification measurements are affected by the development of techniques that accurately capture data but are not universally used. Direct measurements of N_2 creation can be made in aquatic systems using gas chromatography, and changes in N_2:argon (Ar) ratios can be measured using membrane inlet mass spectrometry (Davidson and Seitzinger 2006). However, many often-cited papers used the acetylene inhibition technique, which often understates denitrification rates. Acetylene inhibition has great advantages in that it is simple to conduct and can support many measurements over small spaces, which is useful to measure a patchy, inconsistent phenomenon. The method can be confounded by slow diffusion into fine sediments, the presence of sulfide, and insensitivity to low concentrations of NO_3^-. Wide testing of cores and other disturbed samples instead of making in situ measurements also affects the usability of many acetylene inhibition results (Groffman et al. 2006; also see Powell and Bouchard 2010). Use of N^{15} tracer techniques to track denitrification result in higher estimates of denitrification than would be “expected” for acetylene inhibition approaches, given understandings of how site conditions affect denitrification (Mulholland et al. 2004). Another approach is to track potential denitrification by inducing conditions that lead to denitrification and measuring losses of N from the system, which lead to overestimates of actual denitrification.

The necessary elements for hyporheic zone denitrification are subsurface organic C, low O_2 concentrations, and bacterial biofilms to metabolize the organic matter. Surface-subsurface exchange flows create organic C pools from DOM and particulate organic matter (POM). Entrained dissolved inorganic nitrogen (DIN) is then transformed by hyporheic bacteria into N_2 (Boyer et al. 2006; Harvey et al. 2011; Zarnetsky et al. 2011). The hyporheic zone is not uniform in sediment size, O_2 availability, temperature, and other parameters, creating discrete zones of denitrification instead of the entire zone being a NO_3^- sink (Craig et al. 2010). The controls on denitrification have been found to be different under differing conditions. Sometimes a particular factor (nitrate concentration or carbon availability or grain size) is the variable that best describes differences in rates of denitrification, but often there is a complicated interplay among the factors so that no one parameter can predict changes in nitrogen concentrations.

So, for instance, hyporheic zone hydrology and stream N export are linked, but not in a consistent manner (Zarnetske et al. 2011). Generally, increased water residence time in stream environments with suitable denitrification conditions results in increased denitrification rates (Seitzinger et al. 2006; Flewelling et al. 2012; Mayer et al. 2010), a relationship characterized as the Damkohler number (the residence time:reaction time ratio) (Gu et al. 2008a). Thus, at five low gradient, high N concentration streams, residence time correlated with denitrification (Puckett et al. 2008), and comparisons across differing stream conditions found hydrologic residence time increased denitrification rates (Kaushal et al. 2008). Flowpaths where at least 2 days were needed to traverse a 30 cm thick zone where denitrifying activity was greatest had complete N-removal; shorter residence times resulted in less N-removal (Flewelling et al. 2012).

Generally, it is expected that long hyporheic zone residence times will increase denitrification (Hill et al. 1998), and in many sedimentary environments residence time correlates to flowpath lengths, and the effect of increasing flowpaths is often to reduce N concentrations. Still, denitrification of injected N into a gravel bar was largely completed in 1 hr travel time; very little measurable dentrification occurred farther along the flow paths (Pinay et al. 2009). Most river NO_3^- (60-80%) was removed in the first 50 m of hyporheic flowpaths for a river in a boreal forest floodplain (Cliverd 2008). In the Platte River, depletion of DO occurred not in shallow sediments but 30 cm below the subsurface-stream interface, which was assumed to be the result of denitrification (in part) (Duff and Triska 2000).

A process-free model, the transient storage model, has been used for more than 25 years to estimate water retained rather than advected in stream reaches. Strictly speaking, it simulates in-stream storage (such as pools and back flows); but its results have been interpreted as including some flow through the hyporheic zone (typically, shorter duration shallow-flow pathways) (Bencala and Walters 1983; Boano et al. 2007). Use of a fluorescent tracer (resazurin) that is transformed by microbial respiration into another fluorescent tracer (resorufin) can differentiate between biologically active and inactive storage areas, and help interpret the degree of hyporheic exchange associated with transient storage (Haggerty et al. 2008; Gonzalez-Pinzon et al. 2012). For some, transient storage poorly explains N processing (Hall et al. 2002; Lautz and Siegel 2007) but it has produced good correlations (although these N-reductions may have resulted from benthic not hyporheic functions) (Ensign and Doyle 2005). Denitrification rates were found to be greatest at "separation zones": still pools found in the lee of point bars, where water storage occurs, and greater residence time in contact with fine, organic-rich sediments apparently leads to more sediment denitrification (Opdyke et al. 2006). Alternately, increased flow causes less contact time and creates smaller sediment surface:water volume ratios, and so must result in less denitrification (Ranalli and Macalady 2010). Slowed stream flow was responsible for greater N-uptake in a restored stream compared to pre-alteration conditions, with the greatest rates observed in a side pool (Bukaveckas 2007). Conversely, observations at human-impacted streams in the southwest U.S. found that denitrification at these sites was greater than that measured at reference sites, although connectivity and interchange with the hyporheic zone was greater at reference streams (Crenshaw et al. 2010). Thus, Botter et al. (2010) argue that factors other than sediment contact times are also important in determining N-removal efficiencies.

One of these is sediment quality. Decreasing sediment grain size theoretically adds to 1) surface area availability for bacteria; 2) overall residence time; 3) in particular, slower transport rates of dissolved N and C through potential reaction sites; and 4) retention of C to fuel reactions (Baker et al. 2000). Fine-grained sediments were associated with greater dentrification in one study (Opdyke et al. 2006), and residence time in the denitrification zone controlled N-reduction extent, with residence time being inversely proportional to sediment hydraulic conductivity (Flewelling et al. 2012). Sediment surface area:stream volume ratios predicted denitrification in small streams (Peterson et al. 2001) and were used to explain N-uptake dynamics (Ranalli and Macalady 2010), under the assumption that benthic processes control denitrification reactions. Stream flow rate variations across seasons (greater in spring, less in winter) controlled denitrification because more water and faster flows decreased contact time with sediments (Alexander et al. 2009).

Alternately, availability of NO_3^- was said to be the major control on dentrification (Duff and Triska 1990); denitrification in high flow downstream areas of the Elbe River were greater where NO_3^- concentrations in the river were higher (Fischer et al. 2005). More commonly, denitrification rates are related but not proportional to available NO_3^- concentrations. The Lotic Intersite Nitrogen experiment (LINX II) at 72 streams in 8 US regions used N^{15} tracers and found that denitrification increased with NO_3^- concentrations, but the efficiency of the reactions decreased, meaning the relative proportion of N removed was less in higher NO_3^- streams. Smaller streams lost efficiency more rapidly than larger streams, perhaps because they remove a "maximal amount" of N, so that downstream larger order streams have unfilled assimilative capabilities (Mulholland et al. 2008). The absolute amount of NO_3^- denitrified comparing seasonal flows in two streams was greater under higher NO_3^- concentration conditions – but not proportionally to increases in inputs (Alexander et al. 2009). Nitrate concentrations in streams, when comparing agricultural, urban, and forested land uses, were greatest in the agricultural areas, and denitrification rates were also greatest in agricultural area stream sediments. But, increases in denitrification did not compensate for increases in N, so stream reaches leaving agricultural and urban areas had higher NO_3^- concentrations (Inwood et al. 2005).

Carbon is essential to denitrification, but it may not be the rate-determining factor, either considered theoretically (Bardini et al. 2012) or through extensive measurements (Bohlke et al. 2009). Generally, the amount of C in sediments has been found to regulate denitrification, as C is needed to fuel reactions that consume DO. It is also needed for the metabolic reactions that cause denitrification, so that C concentrations were the best correlation for denitrification in one urban setting (Mayer et al. 2010). Augmenting subsurface C where both groundwater and stream water had low NO_3^- concentrations increased denitrification rates (Triska et al. 2007). Sediment instability in the Elbe River meant C concentrations were well distributed with depth, and so high denitrification rates were measured throughout the hyporheic zone. Denitrification rates were lower outside of the main channel where sediment C concentrations were lower (Fischer et al. 2005). When stream NO_3^- concentrations exceeded half of the saturation concentration, sediment C content was a good predictor of denitrification rates. At less than half-saturation concentrations, stream NO_3^- concentrations best predicted denitrification rates (Arrango et al. 2007).

At least one study concluded denitrification was controlled by temperature (Alexander et al. 2007) but no temperature effect was found by Triska et al. (2007). Because the hyporheic zone buffers temperatures, most likely temperature is usually not a dominant factor affecting denitrification processes. Nonetheless, seasonal effects (seasons correlate to temperature changes) were found where lateral flow through a gravel bar resulted in some denitrification, especially in summer (Deforet et al. 2009). However, flow rate variations and/or groundwater inputs appeared to be the underlying cause of a seasonal component in denitrification rate differences for two streams draining agricultural areas (Bohlke et al. 2009). A seasonal effect was associated with snowmelt flooding, which had been expected to increase N exports from a high gradient mountain stream, but because the higher flow rates increased hyporheic exchange, NO_3^- removal rates were much higher during floods. Although the mass of N exported was greater, it was not proportional to the increase in flow (Hall et al. 2009). Seasonal differences in flows and nitrate inputs, based on a regression on 300 published measurements, were found to explain differences in N removal (low flow and low N-inputs resulted in higher N-removal rates) (Alexander et al. 2009). Daily fluctuations in N-removal in open waters were found, but this was related to fluctuations in both NO_3^- and C concentrations due to photosynthetic organism elemental cycling (Heffernan and Cohen 2010) rather than diurnal temperature patterns; open water conditions often affect subsurface conditions with a time delay.

Some studies have tried to capture the apparent interplay among multiple factors. So, a forested area had increased denitrification; this was linked due to increased debris inputs, which slowed stream flows and increased organic matter content on the stream bottom (although it was not determined if the N was lost at the stream-surface sediment interface or in the hyporheic zone) (Weigelhofer et al. 2012). A study of 18 streams found links among seasonality, NO_3^- concentrations, sediment C content, and denitrification rates. Sediment C was the best predictor of denitrification rates, but stream NO_3^- concentrations were highest in winter, when the greatest denitrification rates were measured. Agricultural land uses resulted in higher stream NO_3^- concentrations compared to urban areas (Arrango and Tank 2008). Biological activity was increased with warmer water in summer, but denitrification was greater in winter due to increased available N from greater groundwater discharge rates. The flux of denitrified N per unit streambed area was inversely related to hyporheic flow rates, suggesting that residence time was important, and, denitrification was higher for areas with finer sediments (which also contained more C) (Bohlke et al. 2009).

Denitrification mechanisms create some of these complications. Biofilms generate zones where local conditions can vary tremendously from bulk water states. This is a potential mechanism for patchy biochemistry, such as denitrification; this is true even though there are general oxidizing states in the overall flow line (Mulholland et al. 2004). Variations at these small scales fit with accounts of sudden changes in NO_3^- concentrations over distances <1 m (Storey et al. 1999); if small, idiosyncratic single sites are determinants of reactions, notions like the Damkohler number that apply to bulk conditions (Gu et al. 2008a) would not be pertinent, although remaining descriptive of changes in water quality at the reach scale.

Downwelling zones are hotspots for denitrification (Holmes et al. 1996). Highest denitrification rates were found in these areas, although rates can be inhibited by higher DO concentrations associated with downwelling, and short residence times make it difficult for microbial respiration to deplete the available DO. Denitrification was minimal in upwelling zones due to a lack of NO_3^- (Storey et al. 2004). In an N-rich agricultural stream, downwelling areas resulted in losses of DO and NO_3^-, while more anoxic upwelling areas transported NH_4^+ from groundwater to the stream (albeit in reduced concentrations), so that the hyporheic zone was a sink for NO_3^- and a source of NH_4^+ for the stream (Hill et al. 1998). Similarly, in a gravel bed of an NO_3^--rich stream denitrification rapidly commenced as DO was reduced below 1 mg L^{-1}, and NH_4^+ concentrations increased (Peyrard et al. 2011).

Floods enhance denitrification, as they increase flow through the hyporheic zone and expand its extent (Cliverd 2007); however, they also have the potential to reduce reactions by changing biochemical conditions – "washing out" the needed redox state (Gu et al. 2008a), which also may occur with high flow-high groundwater conditions (Ranalli and Macalady 2010). Low flow conditions correlated well with higher denitrification rates in Baltimore. The water table was closely linked to stream flow rates, so that head in the aquifer decreased with decreased flow, causing less hyporheic discharge, and greater hyporheic zone residence time – thus, greater denitrification (Mayer et al. 2010).

In a desert stream, nitrification and denitrification occurred in the subsurface; denitrification rates were higher in the banks than the sub-benthic hyporheic sediments (Holmes et al. 1996). A modeling solution of bedform-induced flows suggested that in the flow cells created by hyporheic exchange, the shallower portion of each cell would be a nitrifying area and the deeper portion would denitrify. Consumption of DO (due to available C) would determine the relative proportions of each (Bardini et al. 2012). And, at one stream where denitrification occurred in the hyporheic zone, the hyporheic zone (as determined by comparisons to groundwater chemistry) was very shallow and was not the locus for most denitrification of the incident groundwater. More denitrification occurred below the hyporheic zone, as O_2-rich groundwater became depleted of DO, after coming into contact with buried organic C. Therefore, only tracking hyporheic zone processes may lead to underestimates of total subsurface reactivity (Stelzer et al. 2011).

Generally, despite recent advances in theory (Botter et al. 2010) and field techniques (Haggerty et al. 2009), because hyporheic exchange rates are often not well-defined over larger scales, quantifying the impact of the hyporheic zone on N-attenuation is difficult beyond single site evaluations (Krause et al. 2011). Reports of large reductions include 12 mg N L^{-1} to 0.1 mg L^{-1} over 30 cm of flow path (Gu et al. 2008b), 21% removal of total inorganic N over the entire river network (nearly 75% removal by all river processes) (Stewart et al. 2011), 30% of NO_3^- additions removed in a 300 m reach (Triska et al. 1989b), differences of >80% between groundwater N concentrations (approximately 15 mg N L^{-1}) and stream water concentrations (mean of 2 mg N L^{-1}) (Gu et al. 2007), and overall losses of N in the subsurface in the vicinity of 90% (by various mechanisms that varied with depth) (Lansdowne et al. 2012). High denitrification rates (2.0-16.3 mg m^{-2} h^{-1}) were reported for agricultural streams in three varying settings (but note NO_3^- concentrations increased over the reaches due to larger inputs from groundwater) (Duff et al. 2008).

3.2 Nitrification

Nitrification occurs when bacteria oxidize NH_3 to NO_3^- (Hedin et al. 1998); conditions that allow for nitrification include the presence of: 1) O_2; 2) NH_4^+; and 3) a carbon source (to allow for bacterial growth and reproduction) (Triska et al. 1993). However, nitrifying bacteria are relatively inefficient, and so on theoretical grounds alone will only constitute a small portion of microbial production compared to other heterotrophs (Storey et al. 1999). There have been reports that the hyporheic zone is an area of net nitrification, transforming organic N or NH_3-N to NO_3^- (Boulton et al. 1998).

Generally, subsurface waters have higher NH_4^+ concentrations than surface waters do. This suggests, given overall subsurface to surface transport, that the hyporheic zone generally nitrifies some NH_4^+ (Baker et al. 2000). Triska et al. (1993, 2007) describe the organic-rich subsurface therefore as a patchwork of zones where nitrification and denitrification occurred discretely, depending on the species of N and the redox status of the sediments.

Nitrification requires relatively large amounts of O_2 to occur, and so nitrifiers may be an important link in the reduction of DO that may allow for denitrification. Coupled nitrification-denitrification will only occur with a substantial change in redox potential (from +200 mV to -200 mV), which implies a change in time and distance; however, these kinds of coupled reactions appear to occur across short distances in sewage plant trickling filters, which use biofilms to treat sewage, and so it has been hypothesized similarly linked reaction sites could exist in the interstices of the hyporheic zone (Storey et al. 1999). This apparently was the case for one stream where pressure dynamics changed the extent of the hyporheic zone so that it often extended beyond its "permanent" depth. When the zone was extended, NO_3^- concentrations sometimes increased in the stream, and sometimes decreased – suggesting that the particular relations between nitrification and denitrification were contingent on small local variability (Krause et al. 2009). A modeling study suggested that the upper portion of flow in the hyporheic zone would support nitrification, while deeper flows would be more likely to result in denitrification (assuming needed distributions of N-species, C, and DO) (Bardini et al. 2012).

In an anthropogenically-impacted, N-limited, losing, desert stream, NO_3^- concentrations increased following hyporheic zone residence, with the largest increases in NO_3^- concentrations being found in the summer at the head of the flowpath (Holmes et al. 1994). The hyporheic zone was also found to be a net source of NO_3^- in the alluvial system of an arctic tundra stream (Greenwald et al. 2008). In a stream with low DIN concentrations, organic N (as DOM and POM) was first ammonified and then nitrified; there was no measurement of denitrification, although DO concentrations decreased along the hyporheic flowpaths. All reactions appeared to occur within the first 10 cm of flow (Harvey et al. 2011). Nitrification was found to be affected by temperature, much more so than denitrification, so that with low temperatures, net denitrification was found, and high temperatures led to a net increase in NO_3^- (Triska et al. 2007). Concurrent nitrification and denitrification were measured in the Elbe River; nitrification occurred in sediments closest to the bank, and declined with sediment depth due to decreasing DO concentrations (Fischer et al. 2005).

Jones and Holmes (1996) suspected that hyporheic zones in N-poor streams are generally a NO_3^- source, and hyporheic zones in N-rich streams act as an N sink. Duff and Triska (2000) agreed nitrification is more important in low-N streams, but thought nitrification was a pathway to transform organic N to forms that could subsequently be denitrified, as part of subsurface metabolic processes. Arrango and Tank (2008) measured substantial nitrification in agricultural streams with high NO_3^- concentrations, and the occurrence of peak nitrification did not accord with peak denitrification, suggesting no explicit linkages.

4. The Riparian Zone and the Hyporheic Zone

Although the riparian zone is imprecisely defined, for many it excludes classic wetlands with open water (swamps and marshes) (*sensu* Mitsch et al. 2012). Rather, it is characterized by specific vegetation communities and is physically located between uplands and the stream. Underground water flow in the riparian zone tends to be dominated by groundwater (surface water inputs from bank storage or infiltrating flood water can be important at times), and riparian zone water tables tend to be very shallow (within 1-2 m of the ground surface) (Dahm et al. 1998; Hill 2000; Kaplan et al. 2010). The hyporheic zone is not defined by any surface vegetation features, and so there can be some overlap between these two features. Typically, some bank storage and lateral flows between meanders have been classified as both hyporheic and riparian zone waters (Duval and Hill 2007; Pinay et al. 2009); some prefer not to distinguish between the two processes (Vidon et al. 2010).

Riparian zones are often found at the base of hills, where the surface topography intercepts the water table, or, at least, comes close to doing so. There is often an accumulation of fine, C-enriched sediments at the base of the slopes (Hill et al. 2004). The flatter portion of the riparian zone should accumulate C-rich sediments from flood overflows, which should increase closer to the riverbank (Kellogg et al. 2005). Riparian zones have been identified as hot spots for denitrification (Holmes 2000), although these denitrification zones are often only meters wide (McClain et al. 2003), often at the uphill edge of the zone (Schipper et al. 1993), and at or near the groundwater discharge point into the stream, as well (McClain et al. 2003; Flewelling et al. 2012). Hydric (low redox state) soils, created by high organic content, low O_2 transfer rates from the ground surface, and saturated conditions, support denitrification, and are a signature element of riparian zones (Gold et al. 2001). Nitrogen removal from groundwater can be as much as 100% for riparian zones (Dosskey 2002), but conditions that lead to substantial denitrification are patchy and are not found in all shallow groundwater transition zones near streams (Stutter et al. 2012).

5. Hyporheic Zone Management Issues: Research Findings

5.1 Degradation of the Hyporheic Zone

Anthropogenic degradation of the hyporheic zone results because it lies between surface water and groundwater, two resources exploited by humans and both intentionally and inadvertently affected by their activities. Impacts to the hyporheic zone often affect water exchange and may poison bacteria and invertebrates (Hancock 2002). Direct changes to streams and groundwater flows, such as through water withdrawals and discharges, or to physical morphology such as with dams, channeling, and shoreline and bottom hardening cause impacts to chemical and biological functions, too (Pringle and Triska 2000). Indirect effects come with mining activities, urban and industrial discharges, changes in land use, and agriculture and forestry practices, including removal of sediment and/or water, impairment of surface and/or groundwater quality, disruption of hydrological connectivity between the hyporheic zone and the surface and groundwater systems, and changes in hyporheic biota (Boulton 2007).

Changing stream flow or groundwater heads will affect the hyporheic zone. Flow patterns are generally defined by head differences (Sophocleous 2002), which are affected by changes in stream conditions and groundwater levels (Packman and Bencala 2000; Gu 2008a). Advective pumping can be increased by higher flow rates (Fraser and Williams 1998). Flooding across previously dry areas (Maier and Howard 2011) or drying of previously flowing areas (Gu et al. 2008a, b) can reverse or substantially change hyporheic flow patterns. Lateral flows are also affected by changes in the stream flow or groundwater head, as these will change bank storage and may affect the head differences between meanders (Triska et al. 1993; Wroblicky et al. 1998; Sophocleous 2002; Cardenas 2009).

Constructing dams can induce channel migration and bank erosion, moving the stream away from its original course, and, as a result, changing relationships with the hyporheic zone. Downstream erosion, a common feature for dams, could reduce the size of the hyporheic zone (Hancock 2002). Dams change sedimentation rates and sediment flushing, which can affect the interstitial spaces of the hyporheic zone. If particulates are trapped by the dam, there may be fewer inputs of organic matter downstream, which could affect microbial respiration rates and geochemical reactions (Environment Agency 2009). Dams can either increase or decrease temperatures downstream, with the controlling factors being the size of the impoundment and its management (Webb et al. 2008); temperature is a key element in hyporheic ecological processes (Krause et al. 2011). Releases of water from dams also change rates by which surface-subsurface exchange occurs; rapid changes resulting from dam spates may not allow organisms to accommodate to the new conditions. In addition, subsurface residence time may be substantially reduced under higher flows (Maier and Howard 2011). Dams also affect general downstream groundwater head levels, and the biology and geochemistry of water from reservoirs can be very different from that in native streams (Pringle and Triska 2000).

Simplification of bedforms and channels due to canalization or other channeling and constraining of stream flows reduces exchange potentials between the stream and the subsurface. A smooth stream bottom minimizes advective pumping (Packman et al. 2004; Poole et al. 2006). A stream with fewer meanders had less lateral flow (Cardenas 2009) and overall less connectivity with the subsurface (Crenshaw et al. 2010); all of these lead to a reduced portion of stream water entering the hyporheic zone (Dahm et al. 1998). Fewer stream obstacles mean decreased transient storage (Ensign and Doyle 2005). Straightening channels decreases the overall amount of sediment area per linear distance traversed by the stream, and so decreases water exchange and associated subsurface reactions (Opdyke et al. 2006). In addition, canalization of waterways changes subsurface entry points into the stream, so groundwater may not flow through the riparian zone (Gold et al. 2001). Urban environments are characterized by altered stream channels; in one, Groffman et al. (2005) found that although substantial denitrification appeared to occur, a lack of debris accumulations limited the number of locations where proper reaction conditions could occur. Streams where channels have been modified often have greater erosion rates; locations with higher erosion rates or where flows were constrained and/or straightened, were characterized as being less likely to retain nutrients (Dahm et al. 1998).

Mining in a stream basin can add excess silt, introduce heavy metals, and change channel morphological features. Runoff can introduce additional silt to the hyporheic zone leading to colmation (the clogging of interstitial spaces), which limits surface water exchange, and so decreases hyporheic zone O_2 and nutrients. Mining that occurs directly in a stream increases colmation by causing sediment resuspension (Hancock 2002). Overall, the occurrence and amount of colmation is affected by stream bed transport properties, as small differences in velocity affect settling and resuspension (Rehg et al. 2005). Mining activities that change stream pH (by exposing sulfidic minerals) increase dissolved metals concentrations, and could prove to be toxic to the hyporheos (Hancock 2002). The hyporheic zone was shown to immobilize manganese from copper mining in Arizona (Harvey and Fuller 1998) and, because changes in redox zonation occur generally in hyporheic zones and residence time in reactive sediments promotes sorption, generally the hyporheic zone does a fair job in removing many metals of concern from mining waste-impacted groundwater and surface water (Gandy et al. 2007).

Changes in channel geomorphology associated with in-stream mining, such as widening or deepening of the channel with the removal of sediments, can cause loss of riffle-pool sequences and river bends, and lower floodplain water levels, thus also changing hyporheic flows (Hancock 2002).

Negative impacts to the hyporheic zone from urban and industrial activities come from effluents, stormwater, and other discharges, as well as from groundwater pollution, and general colmation effects. Nutrients from effluents and stormwaters increase N concentrations; discharged metals and organics may affect the hyporheos; and colmation results from excess sediment inputs (Hancock 2002). High levels of sewage-polluted groundwater prompted a faunal composition change in one hyporheic zone (Mallard et al. 1994), a finding which does not support a more general hypothesis that the hyporheic zone can serve as a refuge from pollution for stream invertebrates (Hancock 2002). Inputs of sewage-derived DOM into groundwater systems caused a change in invertebrate community structure to more pollutant-tolerant organisms (Hartland et al. 2011). It is thought that urban environments support a less diverse hyporheic biology that has less production (Environment Agency 2009). Overall, however, determining impacts to biota from pollution is hampered by a lack of detailed information for many subsurface taxa (Hakencamp and Palmer 2000); nonetheless, it has been proposed that larger hyporheic zone organisms would be suitable for use as biomarkers, as their distributions are affected by pollutants in streams (Boulton 2000).

Water quality impairments in many streams correlate to the amount of agriculture in the surrounding basin. Agriculture (including range activities which often affect fluvial landscapes) and forestry introduce excess nutrients and silt to stream ecosystems, change vegetation distributions and the physical landscape (including stream morphology and positioning), discharge pollutants of various kinds, and also alter flows in the hyporheic zone through groundwater and surface water extraction (Pringle and Triska 2000). Unregulated forestry has been found to reduce inputs of large wood, alter riparian zone vegetation (leading to hyporheic zone effects), and increase sedimentation (Environment Agency 2009). Any residual poor practices will also have some impacts similar to these. Nutrients may be introduced to aquatic systems through fertilizers, waste from livestock, and ash from forestry waste management. Augmented nutrient levels in streams lead to reduced DO and can change hyporheic conditions from oxidizing to reducing (Hancock 2002). In many agricultural areas adjacent to streams, the streams are physically modified (channelized and tiled) to drain high water tables or encourage run-off to prevent saturated soils; this was found to diminish riparian and hyporheic cycling of N (Triska et al. 2007), and also affected the general ecology of the impacted streams (Freeman et al. 2007). Anoxic conditions associated with stream degradation may increase denitrification, a potential environmental benefit, but only if net N removal equals or exceeds N inputs (Boulton et al. 1997).

Agriculture and forestry can increase colmation. Generally, the loss of riparian vegetation (from field expansion or livestock browsing or trampling) can lead to bank collapse, burying the hyporheic zone and limiting parafluvial exchange. Native riparian vegetation was found to support a more diverse and abundant hyporheos than pasture land (Boulton et al. 1997). Deforested riparian areas have narrower streams with less bed roughness and higher stream velocities. This was thought to lead to lower denitrification rates because of less connectivity to subsurface processes (Sweeney et al. 2004). Livestock moving through streams can affect the hyporheic zone by contributing nutrients through waste, compacting gravel and clogging interstitial spaces, resuspending sediments, and consuming or trampling riparian vegetation

(Hancock 2002). Sediment inputs are also increased by near-stream construction (Hester and Gooseff 2010). Erosion generally changes bedform conditions; the introduction of substantial sediments into streams can lead to sand slugs, which, while comprising new hyporheic habitat, are not natural features, and have not proved amenable to restoration projects intending to restore habitat heterogeneity (Lake et al. 2007). Generally, agricultural impacts to streams are thought to result in simplified hyporheic population structures that have less overall production (Environment Agency 2009).

5.2 Stream Restoration, Nitrogen Dynamics, and the Hyporheic Zone

Environmental management requires making selections from a suite of goals, which are developed from identified and sought values and functions for the restoration site. Natural systems do not have such pre-selected goals, per se, although our analyses often impute intentions and directions to them. However, when we take steps to undo our effects on a system, we must choose the directions and aims for the project, as our general alterations of the world make it impossible to simply return to pre-anthropogenic conditions.

The most common explicit goal for stream restoration projects is to improve habitat for one or more commercial fish species; this is closely related to project rationales to improve habitat for stream macroinvertebrates, either as habitat indicator species or to support the charismatic fish species (Bernhardt et al. 2007; Lake et al. 2007). Often these goals are achieved through alteration of stream morphology (Bond and Lake 2003) – what has been described as the “field of dreams” hypothesis (“if you build it, they will come”) (Palmer et al. 1997). Although management programs seeking to increase salmonid fish populations sometimes specifically seek to improve the hyporheic zone, as these species lay eggs and have young fish that live in the top 5-50 cm of stream sediments (Environment Agency 2009), even comprehensive stream restoration designs usually do not explicitly address any subsurface hydraulic connectivity issues (e.g., Shields et al. 2003).

General restoration efforts for stream and benthic habitats can also enhance the hyporheic zone and affect its ability to transform N, even if not implicitly included in project planning, as hyporheic zone improvements are a byproduct of efforts aimed at other goals (Welti et al. 2012). For instance, one common stream restoration approach to improve fish habitat is to flush fine sediment from benthic gravel areas (Arthington and Pusey 2003), which should also improve connectivity into the hyporheic zone. Adding woody debris to streams is another common surface water ecosystem rehabilitation technique that also helps hyporheic zones. If the log is partially embedded in sediment, across a flowpath, this will create two downwelling areas: one just before the water hits the log and another right at the downstream plunge pool. There will also be an upwelling area shortly after the plunge pool (Fig. 3). This should also reduce colmation, which will improve connectivity (Boulton 2007).

If the hyporheic zone is a foundation for overall stream health, then its significance in restoration plans is thought to be severely underappreciated (Boulton et al. 2010). To address this failing, the British Environment Agency issued a 250 page handbook on science issues associated with the hyporheic zone. One chapter discussed how common stream restoration efforts affect the hyporheic zone. The addition of in-stream deflectors and large wood was identified as the most common activity (43% of projects). This increases hyporheic exchange, increases subsurface DO, and generally enhances subsurface chemical reactions. It tends to redistribute fauna because of habitat changes. Plantings to enhance fish cover was the second most common restoration technique affecting the hyporheic zone (18% of projects), and was thought to create very local changes in flows, chemistry, and habitats. Bed raising and substrate changes (8% of all projects) increases stream connectivity, could enhance chemical reactions, and could have a major effect on habitat types and distributions. Increasing the sinuosity of the stream (6.5% of all projects) increases lateral hyporheic flows and tends to increase subsurface residence time, and creates more diversity of benthic and hyporheic habitats. Removing dams/weirs (6% of all projects) increases lotic environments and may increase exchange processes but probably decreases overall storage times; it causes major shifts in fauna due to habitat change. Removing artificial banks and beds (5% of all projects) causes a substantial increase in exchange and adds the subsurface-banks as potential habitat zones. Creating riffles (4.5% of all projects) increases exchange rates and subsurface residence time, increasing the potential for chemical reactions, and, at a minimum, relocates subsurface habitats (Environment Agency 2009). Although not explicitly mentioned in the handbook, other improvements to riparian zone conditions, such as plantings or vegetation restoration should also indirectly improve hyporheic zone functions.

However, increased connectivity with surface water or groundwater can have negative consequences. Contaminated groundwater can degrade surface water if it is transmitted through the hyporheic zone (Hancock 2002) or contaminated surface water can affect groundwater or hyporheic zone water quality (Environment Agency 2009). Restoring connectivity can also allow invasive species to spread and expose endemic species to new competitors, changing community dynamics (Kondolf et al. 2006). Even increased hyporheic zone denitrification (see below) can have negative consequences: one estimate is that the equivalent of 10% of anthropogenic emissions of nitrous oxide (a potent greenhouse gas) are generated from river denitrification processes (Beaulieu et al. 2011).

One broad stream restoration review identified vertical connectivity with the hyporheic zone as an important element in creating proper ecological functions, but no explicit actions were identified to achieve the connectivity goal (Lake et al. 2007). However, a similar review included specific design elements to improve hyporheic zone functions: creating features such as pools, riffles, steps, log dams, bars, meanders, and side channels, along with in-stream placement of debris dams and large wood, and increasing bed complexity (or at least matching historical patterns), coarsening sediments, and restoring the riparian zone (Hester and Gooseff 2010).

Stream restoration projects aiming to increase bedform heterogeneity will strengthen connections in longitudinal, lateral, and vertical dimensions and increase surface-subsurface exchange flowpaths, although predictive capabilities for such efforts were said to be lacking (Boulton et al. 2010). Several weir variants (cross vanes and J-hooks) were installed in a New York mountain stream to reduce stream erosion, and also to increase hyporheic zone connections. Temperature testing largely corresponded with modeling of the project, suggesting that design water exchange patterns can be largely achieved in practice (Crispell and Endreny 2009). A Nevada project undertaken to restore riparian functions by elevating downcut sections and adding riffles and pools was found to have greater transient storage, as measured by retention time, compared to unrestored areas. Modeling supported longer flow intervals in the hyporheic zone, which suggested denitrification would have also increased (Knust and Warwick 2009). Several small weirs (1.5 m high) were constructed in another stream to mimic beaver dams; a complex flow pattern of shallow pools, plunge pools, glides, and riffles with a variety of sediment distributions and bedforms resulted. Indirect measurements (temperature and water chemistry) along with modeling found distinct areas of inhibited and enhanced hyporheic exchange, with evidence of much denitrification found in downwelling zones, and some more in upwelling areas (Fanelli and Lautz 2008). Conversely, installation of a flat gravel bed, although conformed to the preferred depths used by salmon for spawning, did not replace lost habitat from dam construction. Salmon did not use the artificially formed sediments, and it was suggested that the lack of bedform definition impeded hyporheic flows. Salmon possibly found the space subpar due to the absence of hyporheic environmental modifications (particularly temperature control) (Kondolf et al. 2006).

Instead of proposing in-stream modifications, Vaux (1968), using analytical solutions of flow equations, determined that subsurface flows could be enhanced by changing hydraulic conductivity in sediments (explicitly intended to increase DO availability for salmon alevins). Structural changes included various high or low conductivity blocks of material, or sheet pilings. Ward et al. (2011) simulated the structural changes proposed by Vaux, and, using reasonable rate values derived from high gradient streams, estimated the impacts on processes such as denitrification, respiration, and temperature buffering from various designs. A template to achieve various effects was proposed. It was noted, on a practical note, that high conductivity subsurface features can be difficult to retain as they will have their effectiveness reduced by sediment clogging, but that some of the same results could be achieved through selection of various low conductivity structures (the functions of which are unlikely to be easily degraded).

Currently, there are few broad guides focusing on improving stream N-management, as there are for increasing bank stability and some other stream attributes. One explicit management approach suggested an emphasis on 2^{nd} and 3^{rd} order streams with low flow rates, calling for enhanced C availability and increased transient storage and interchanges with surrounding terrestrial environments (Craig et al. 2008). The program targeted in-stream N, not subsurface concentrations. The low order streams were preferred in accord with N-removal efficiencies identified by Ensign and Doyle (2006). Carbon enhancement was not selected based on any cited studies, but rather to ensure stream metabolism was maintained to allow for denitrification. However, Hartland et al. (2011) determined that enhancing DOM in subsurface environments caused a change in invertebrate populations to more pollution tolerant species, and so this remedial approach for N may have unintended consequences.

A stream restoration project in Maryland that was intended to decrease stormwater-driven erosion also led to improved N-removal rates. Cobbles and boulders and coarse sediments were set into the stream, and features such as point bars, pool-riffle sequences, and meanders were constructed. The riparian zone had trees planted, and banks were cut to be closer to the stream surface in places. Tracer tests found that mean denitrification rates were twice as high for restored areas as unrestored areas, and groundwater and stream water NO_3^- concentrations were lower in the upstream restoration areas. Low bank riparian reaches had greater overall denitrification rates, which was attributed to wider channels and less stream incision creating greater overall system hydrological connections (for both the hyporheic and riparian zones) (Kaushal et al. 2008). Approximately 40% of nitrate loadings were removed, due to "greater whole stream connectivity" and especially to increased residence time (especially in the hyporheic zone, where most dentrification was assumed to occur) (Klocker et al. 2009).

A long-time (ca. 100 yr) channelized stream in Kentucky was relocated to its former floodplain. Its flow patterns were altered by creating meanders and pool-riffle sequences; the restored segment was wider and shallower and approximately 15% longer than the channelized segment had been. Significantly slower flow rates, higher temperatures, greater transient storage areas, and more connectivity with the hyporheic zone were created. Nitrogen uptake was estimated to be 30 times greater than the channelized segment used to have, and approximately an order of magnitude greater than a reference site (which had a thriving, forested riparian zone) (Bukaveckas 2007).

Constructing artificial riffles (adding stones or cobbles) or gravel bars, or recreating meanders are also common habitat restorations. Constructed riffles and a constructed step in N-rich agricultural and urban streams induced additional hyporheic exchange, with clear downwelling and upwelling trends (Fig. 4). The hyporheic zone at the restored sites was a NO_3^- sink; the streams had steeper longitudinal hydraulic head gradients and coarser substrates than reaches with natural riffles and steps, suggesting the restored sites had enhanced NO_3^- removal capabilities (Kasahara and Hill 2006). At another site, a constructed gravel bar and re-meandered stream reach caused enhanced lateral hyporheic exchange flow. Vertical exchange was increased at the gravel bar by adding a riffle-pool sequence. The need to manipulate sediments in restorations was underlined, especially in agricultural and urban settings where fine-grained sediments predominate and cause colmation (Kasahara and Hill 2007). Construction of baffles also lead to increased denitrification – however, the effect was thought to be due to increased transient storage due to stream velocity decreases, and not increased hyporheic zone exchange (Ensign and Doyle 2005). Adding debris dams and gravel bars to streams in urban and suburban settings caused greater denitrification rates, more than other management steps, even when compared to forested reference sites. These sites supported organic-rich matrices, which seemed to be the key factor for added N-losses (Groffman et al. 2005).

Indirect effects on the hyporheic zone may be achieved through alternative restoration efforts. Forested riparian zones, for instance, were associated with greater hydrologic retention times in stream reaches, apparently from slowing stream flows through debris additions. The debris may have increased hyporheic zone inputs or created surface backwaters. In any case, N-reductions greater than degraded, non-forested areas were measured (Weigelhofer et al. 2012). Stream fencing can be useful in preventing cattle from encroaching on the riparian zone (Vidon et al. 2010); not only might that lead to indirect hyporheic zone benefits from a restored riparian buffer, but keeping cattle out of a stream is a good direct hyporheic zone remediation activity. In the Danube River, changes to channels to restore more natural flow conditions increased surface-subsurface connectivity, and resulted in greater rates of denitrification (among other enhanced hyporheic zone functions) (Welti et al. 2012). It has also been suggested that restoring variable flow conditions in controlled streams can improve nutrient uptake, as this may increase contact with subsurface C pools (Faulkner 2008).

Agricultural drains are designed to have flat bottoms and steep, unvegetated sides to facilitate water flows. These ditches lose functionality as they erode and with increasing plant colonization, and the narrow-bottomed, vegetated, and often benched ditches that result have been called “2-stage” ditches. Testing of sediments found that the benches in 2-stage ditches function like floodplains, and have good denitrification potential (greater than sediments in 1-stage ditches). Although this potential decreases some effects associated with excessive fertilizer use, drains also foster direct transport of excess N from fields to streams; overall, it is likely that areas with drains have quicker transport of more N to surface waters than areas that are not drained (Powell and Bouchard 2010). In one setting, a bioreactor was installed in a ditch instead of relying on natural deterioration of the ditch structure. The woodchip bioreactor generated impressive denitrification rates, estimated to exceed those associated with natural wetlands in the region by a factor of 40 (Robertson and Merkley 2009).

There appear to be correlations among land use, channelization, and the hyporheic zone’s ability to retain NO_3^-, but the exact linkages have not been made yet (Robertson and Wood

2010). Although some studies have quantified the effect of stream restoration on nutrient dynamics, it is difficult to determine general effects that extend beyond the specific examples (Bukaveckas 2007). Clearly, understanding denitrification better is an important element in the construction of accurate watershed nutrient management plans (Davidson and Seitzinger 2006). Because denitrification is limited in time and space within the hyporheic zone, meeting the definition of “hot spots”/“hot moments,” it may not be possible to manage specific stream elements to create increased denitrification rates. Instead, increasing overall stream-hyporheic zone connectivity may be the most feasible means of achieving the desired end (McClain et al. 2003), although such a restoration approach becomes a "black box" solution, resistant to further analysis.

6. Conclusions

Regional mass balances (e.g., Howarth et al. 1996; van Breeman et al. 2002) find that sources of N to the environment exceed identified sinks, and so denitrification is assumed to account for the lost N, based on data collected in experimentation over physically small spaces and short durations (Grimm et al. 2003). Use of N_2:Ar ratios and very precise direct measurements of N_2 appear to be resolving some of the analytical issues (Laursen and Seitzinger 2002; McCutchan et al. 2003). However, historically, it has very difficult to measure denitrification well at any scale, from the regional to site-specific. This has led to the invocation of “hot-spots,” variable over time and space, to account for inabilities to repeat measurements or to find the expected phenomenon that is predicted by mass-balance and other modeling (Boyer et al. 2006). Denitrification of groundwater N in the hyporheic zone has been consistently found for streams across the U.S.; the amount of denitrification is site specific, but generally relates to residence time in the reaction zone beneath the stream (Puckett et al. 2008). The absolute impact of the hyporheic zone is a function of still poorly determined relationships defined by Findlay (1995): short residence times with high reaction activity lead to as much alteration of water chemistry as longer residence times with slower reaction rates. Long residence times imply that not very much water volume can be processed through the subsurface. Short reaction times allow for greater volumes to be treated, but then require resolution of conundrums such as quick depletion of DO (which appears to require residence time), DOC availability to fuel reactions, and whether small biofilm zones can suffice to explain how otherwise well-oxygenated sediments can host denitrification. Denitrification in the hyporheic zone occurs in spatially discrete zones, and requires specific geologic and nutrient conditions. Although best estimates are that river basins are the site of significant denitrification (for instance, van Breeman et al. 2002), a skeptical analysis of extent of hyporheic conditions could conclude that there is often not enough upwelling and downwelling relative to the size of the stream to generally create meaningful effects on stream N-cycling.

Riparian zones cannot be separated from the hyporheic zone, given their close spatial and functional proximity in many streams. Riparian zones have been described as poor “end-of-pipe solutions” for increasing nutrient content in run-off and groundwater; where conditions are suboptimal (deep groundwater flow paths, non-hydric sediments), only minor (<10%) N-reduction can be expected (Stutter et al. 2012). The strongest correlations for N reductions in streams have been found to wetlands acreage, not riparian or hyporheic conditions (McClain et al. 2003). In fact, factors other than denitrification potential (such as land use, population density, soil quality, and N atmospheric deposition rates) correlate much better with stream N concentrations (Smith et al. 2008).

This supports the proposition that augmenting subsurface denitrification is unlikely to be more effective at reducing stream N-concentrations than reducing input N concentrations (Ranalli and Macalady 2010). Various land use programs have been proposed to achieve lower N-inputs (Howarth 2005; Silgram et al. 2005; Hiscock et al. 2007), with one estimate being that major changes in N-loading in 25% of headwater streams could "easily" lead to 10-15% reductions in river discharge N loadings (Alexander et al. 2007). However, most input control programs have not been able to achieve their goals (Boesch et al. 2001; Howarth 2005). One of the few clear reductions in the delivery of N to coastal waters occurred in the Black Sea in the 1990s. This was not due to management success, but rather reflected the substantial, negative impacts of economic chaos on agriculture in the former Soviet Union (Howarth 2005).

General prescriptions to reduce world-wide releases of reactive N by 25-30% include controlling emissions from fossil fuel combustion, increasing efficiency of N applications to crops, improving animal waste management, and, in cities without sewage treatment, treating at least half of all human septic wastes (Galloway et al. 2008). Howarth (2005) identified steps to be taken in the U.S. that could reduce coastal impacts from increased N releases. These included source reduction steps, and additional treatment possibilities including:

1) changing agricultural drainage systems so as to improve nutrient uptake

This has been identified as feasible and generally creating few impacts to overall agricultural output. Mostly this kind of project appears to require changes in perception of desired aesthetics and some changes in general ditch management (Birgand et al. 2007), although more substantial projects are also feasible (Robertson and Merkley 2009). Agricultural drainage systems do not affect N that was exported directly to groundwater, however.

2) adding wetlands to riverine systems wherever feasible and desired.

It has been argued that above-ground, flow-through marshes are the most effective means of reducing NO_3^- concentrations, especially if flow short-circuiting is avoided through careful design (Kadlec 2012). Greater removal efficiencies (although greater space requirements are needed too) can be achieved using constructed subsurface wetlands (Garcia et al. 2010), although subsurface treatment is most beneficial when pathogen exposure is a major concern (Kadlec 2012). Marsh projects like these can be monitored and assessed more easily than less intrusive changes to foster subsurface NO_3^- removal in riparian and/or hyporheic zones. Marsh construction requires large expanses of space, however, and may not be the landscape feature that is possible, needed, or desired in all settings.

3) restoring riparian areas as is possible.

A nation-scale modeling exercise in England suggested that substantial attenuation of N-inputs through subsurface reactions is possible in many lotic environments (although certainly not all) (Smith et al. 2009). Bayesian simulations using literature search denitrification values suggested that basic riparian restoration techniques would lead to approximately 25% more N-assimilation in restored reaches compared to impacted reaches (Faulkner 2008). Still, although higher concentrations of NO_3^- lead to higher denitrification rates, the increase in denitrification is not proportional to increases in inputs (Alexander et al. 2009), and so only mitigates (not resolves) the issue of increasing N-releases.

The degree that restoration efforts should focus on the hyporheic zone and its potential for denitrification is not clear. Estuarine N-loads are a function of prior loadings in the upriver region: e.g., water quality in the Gulf of Mexico is closely linked to historical fertilizer applications throughout the Mississippi-Missouri River basin (Alexander et al. 2007). Degradation and alteration of headwater streams (in particular) was identified as a major element in coastal hypoxia due to a loss of nutrient processing capabilities (Freeman et al. 2007). This seems to imply that restoration efforts in these areas, assuming that denitrification potential is part of the selected approach, could be effective in improving a major regional problem. But, determining the impact of a potential denitrification zone depends on the degree and reliability of the connection between N-source and the denitrification zone (McClain et al. 2003). There is good evidence that denitrification occurs in the riparian and hyporheic zones, reducing groundwater N inputs and mitigating stream NO_3^- concentrations. Denitrification in riparian zones requires groundwater to be funneled through particular small regions of the streamside environment, and many groundwater pathways do not intersect these zones. Hyporheic processes depend on generating micro-scale patches of favorable conditions, or appear to be governed by Freundlich reaction kinetics: denitrification increases as ambient concentrations of NO_3^- increase, but denitrification rate increases are not fast enough so as to keep pace with the increases in the stream water NO_3^-. Slower flow rates through sediments compared to flow rates in the stream and the relatively small volume of the hyporheic zone imply that, in most settings, water in a particular reach cannot have much residence time in the subsurface. Thus, there can only be a limited role for these environments as checks on increasing stream NO_3^- content and deliveries to marine systems. Increasing treatment of water through wetlands appears to return greater dividends than enhancing subsurface NO_3^- treatment potentials.

Incorporating better knowledge of these ecotones into stream remediation plans is not pointless, however; understanding the functionalities of these zones better could lead to better crafting of environmental initiatives. On Long Island (New York, U.S.), management concerns regarding NO_3^- concentrations in a shallow lagoonal estuary have focused on direct groundwater discharges to the estuary (Kinney and Valiela 2011). However, although the fresh water entering the estuary is derived from groundwater, most enters the estuary via short stream systems. A focus on improving riparian and hyporheic zone processes in these canalized, heavily altered streams, where sufficient space for wetland construction appears to be lacking, might pay a greater short-term dividend than trying to change overall N-inputs to groundwater (where 20-50 year residence times have been modeled). In this way, rehabilitation of hyporheic zones could reduce estuarine N-loadings within timescales appreciated by funding agencies and politicians.

Therefore, there is virtue in addressing the hyporheic zone, and improving its connectivity with surface waters as stream modifications are made. Even greater returns might be realized by treating the hyporheic and riparian zones together. Although source controls on NO_3^- appear to be the most effective means of reducing NO_3^- inputs to sensitive marine environments, greater water flows through subsurface zones will help to ameliorate increases in NO_3^- releases from agriculture and other human endeavors, especially if short-term effects are desired.

Acknowledgements

Jessica Gurevitch was extremely helpful with her comments, both on the MS project that was the genesis of this paper, and on early drafts of this work. Comprehensive, careful, and thoughtful peer review of the paper enhanced nearly all aspects off our initial submission.

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List of Figures

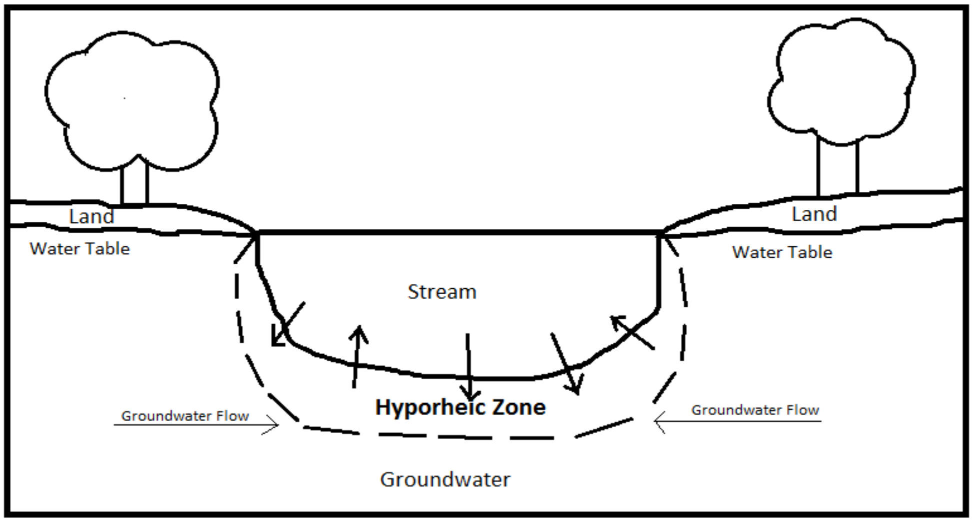

Figure 1. The hyporheic zone

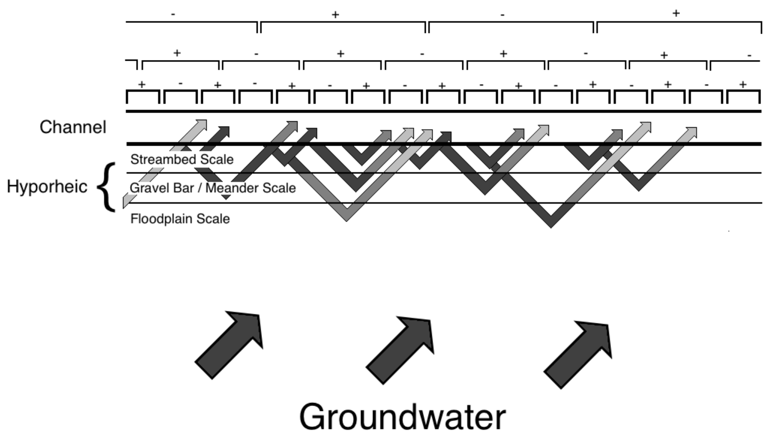

Figure 2. Diverse subsurface flowpaths (adapted from Poole et al. 2008): paths range from very short and shallow (s. to min.) to those that are very long (mos. to yrs.). Temporally longer pathways tend to traverse physically longer subsurface pathways. Groundwater controls the overall directionality of flow.

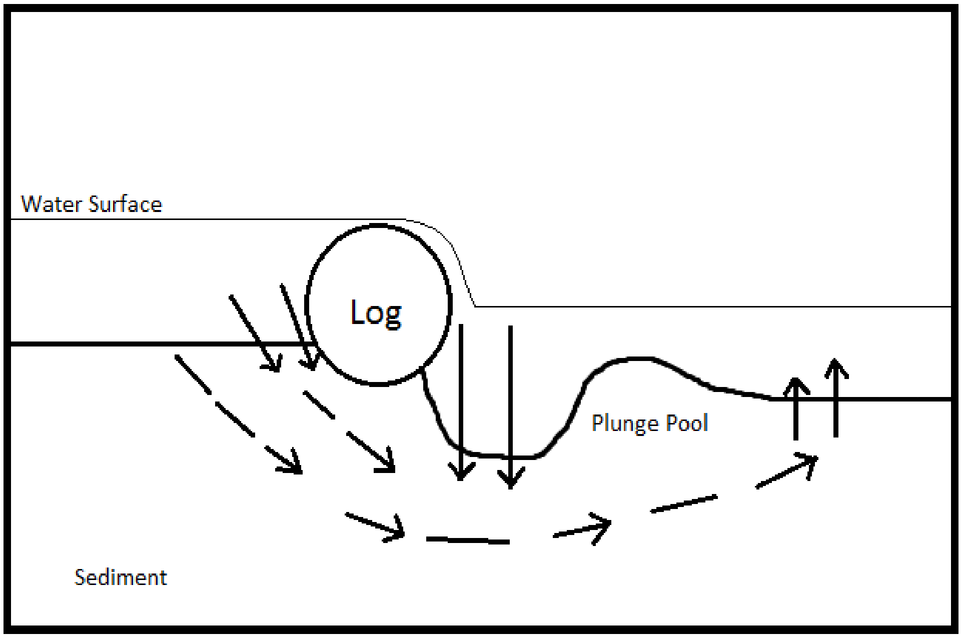

Figure 3. Micro-pressure effects on hyporheic flow, caused by a partially embedded log in a gravel-bed riffle (adapted from Boulton 2007)

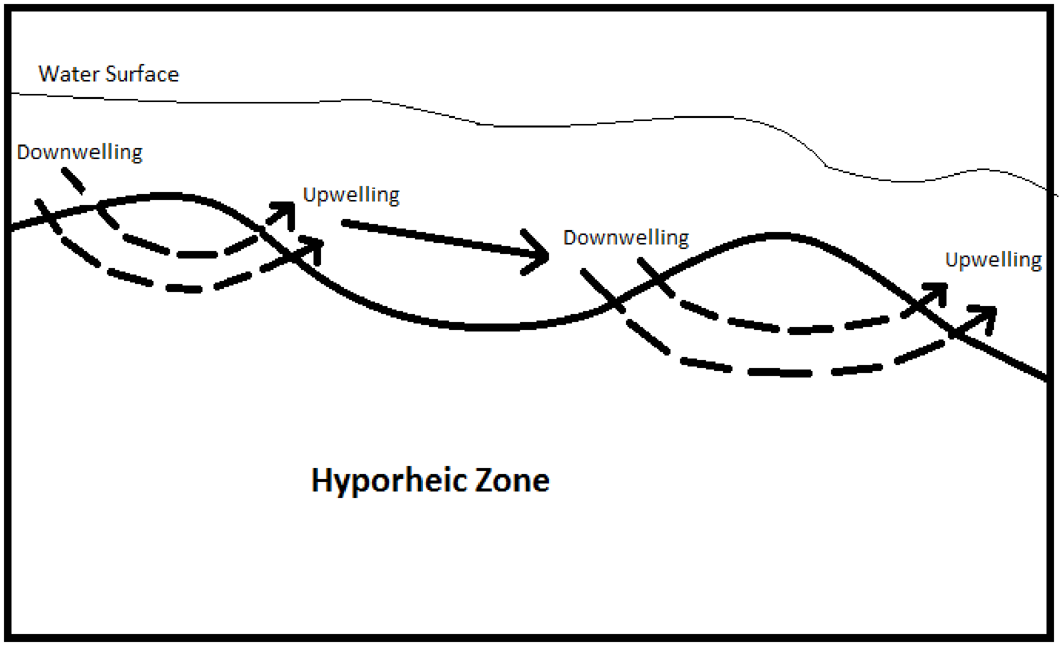

Figure 4. Hyporheic flowpaths through a riffle (adapted from Boulton 2007)