AGU PUBLICATIONS

Water Resources Research

REVIEW ARTICLE

10.1002/2015WR017617

Special Section:

The 50th Anniversary of Water Resources Research

Key Points:

- Hydrologic exchange prolongs physical storage and reactive processing in river corridors
- Consequences for water quality, ecological metabolism, and wildlife habitat provision
- Challenge linking small-scale physical drivers with larger-scale ecological consequences

Correspondence to:

J. Harvey,
jwharvey@usgs.gov

Citation:

Harvey, J., and M. Gooseff (2015), River corridor science: Hydrologic exchange and ecological consequences from bedforms to basins, *Water Resour. Res.*, *51*, 6893–6922, doi:10.1002/2015WR017617.

Received 25 MAY 2015
Accepted 22 JUL 2015
Accepted article online 27 JUL 2015
Published online 4 SEP 2015

River corridor science: Hydrologic exchange and ecological consequences from bedforms to basins

Jud Harvey[1] and Michael Gooseff[2]

[1]U.S. Geological Survey, Reston, Virginia, USA, [2]Institute of Arctic and Alpine Research, University of Colorado, Boulder, Colorado, USA

Abstract Previously regarded as the passive drains of watersheds, over the past 50 years, rivers have progressively been recognized as being actively connected with off-channel environments. These connections prolong physical storage and enhance reactive processing to alter water chemistry and downstream transport of materials and energy. Here we propose river corridor science as a concept that integrates downstream transport with lateral and vertical exchange across interfaces. Thus, the river corridor, rather than the wetted river channel itself, is an increasingly common unit of study. Main channel exchange with recirculating marginal waters, hyporheic exchange, bank storage, and overbank flow onto floodplains are all included under a broad continuum of interactions known as "hydrologic exchange flows." Hydrologists, geomorphologists, geochemists, and aquatic and terrestrial ecologists are cooperating in studies that reveal the dynamic interactions among hydrologic exchange flows and consequences for water quality improvement, modulation of river metabolism, habitat provision for vegetation, fish, and wildlife, and other valued ecosystem services. The need for better integration of science and management is keenly felt, from testing effectiveness of stream restoration and riparian buffers all the way to reevaluating the definition of the waters of the United States to clarify the regulatory authority under the Clean Water Act. A major challenge for scientists is linking the small-scale physical drivers with their larger-scale fluvial and geomorphic context and ecological consequences. Although the fine scales of field and laboratory studies are best suited to identifying the fundamental physical and biological processes, that understanding must be successfully linked to cumulative effects at watershed to regional and continental scales.

1. Introduction

The measure of a river is one of the primary occupations of hydrology, and it comprises much more than the cross-section average width, depth, and velocity. Typical two-dimensional velocity-area-based measurements of river discharge do not capture the complex, three-dimensional flows of a river interacting with undulating banks, side cavities, low-lying riparian areas, and permeable sediments and floodplains lying far outside of the river's wetted width [*Ward*, 1989; *Junk et al*., 1989; *Dunne et al*., 1998; *Knighton*, 1998; *Winter et al*., 1998]. Together, these features (river channels, fluvial deposits, riparian zones, and floodplains) form an inseparable unit—the river corridor [*National Research Council*, 2002]. A river, after all, is not a pipe, and a complete picture of transport, storage, reaction, and biological productivity and diversity of river networks requires consideration of exchanges of relatively fast moving waters through the river's thalweg with the more slowly flowing surface and subsurface waters adjacent to and directly beneath the channel [*Bencala*, 1993; *Stanford and Gonser*, 1998].

River water moves in and out of the main channel along pathways that are perpendicular to the channel's main axis. These "hydrologic exchange flows" are difficult to measure [*Boano et al*., 2014; *Kondolf and Piègay*, 2003], yet quantifying fluxes perpendicular to the axis of flow is no less important than the river's downstream flow, or exchanges with the atmosphere and deeper groundwater (Figure 1). Greater contact of river water with geochemically and microbially rich sediments is of particular significance, because of the increased opportunities for reactive transformations of organic carbon, nutrients, and other energy-rich substrates [*Battin et al*., 2008; *Dahm et al*., 1998; *Brunke and Gonser*, 1997]. In fact, these exchanges can explain the downstream influence of chemical reactions on the quality of receiving waters [*Findlay*, 1995; *Marzadri et al*., 2014; *Harvey et al*., 2013].

The river corridor perspective views hydrologic exchange flows and the resulting enhancement of biogeochemical processing as being integral to supporting healthy levels of aquatic metabolism [*Helton et al.*, 2011; *Lautz and Fanelli*, 2008; *Mulholland et al.*, 2008; *Fischer et al.*, 2005; *Grimm and Fisher*, 1984] and other functions that provide ecosystem services (e.g., water quality improvement) valued by humans [e.g., *McClain et al.*, 2003]. Biogeochemical reactions of organic carbon and nutrients not only regulate nutrient budgets and support stream ecosystems but also strongly influence the fate of toxic metals and mining contaminants [e.g., *Von Gunten and Lienert*, 1993; *McKnight et al.*, 1988], organic contaminants from industrial and military sources [e.g., *Kim et al.* 1995], and emerging contaminants such as pharmaceuticals [e.g., *Fick et al.*, 2009] and contaminants in wastewater from unconventional oil and gas development [*Warner et al.*, 2013]. However, uncertainties in basic physical parameters of flowing waters, such as width and depth [*McDonnell and Beven*, 2014], residence time [*Green et al.*, 2009], and river sediment characteristics [*Hall et al.*, 2013] are now some of the key limitations in understanding functions of river networks including CO_2 and N_2O emissions at nationwide and global scales [*Battin et al.*, 2008; *Beaulieu et al.*, 2011; *Butman and Raymond*, 2011]. River corridor science has a rich interdisciplinary heritage, and this paper provides a vision for future research while also reviewing historical highlights as published in the pages of *Water Resources Research* and many other journals.

1.1. Hydrologic Connectivity as a Key Organizing Concept

In this article, we take a broad view of river corridor transport processes in order to emphasize commonalities and promote integration. Hydrologic connectivity is a key concept that integrates longitudinal transport with vertical and lateral exchanges in river corridors [*Stanford and Ward*, 1988, 1993], watersheds [*Jencso et al.*, 2010], and aquatic ecosystems [*Larsen et al.*, 2012]. The concept of connectivity is playing an increasingly important role by influencing the development of scientifically based policies for protecting the valuable functions of river corridors [e.g., *Nadeau and Rains*, 2007; *Alexander et al.*, 2007]. We use the term "hydrologic exchange flows" to specifically refer to lateral and vertical exchanges of water, materials, and energy between rivers and their surrounding marginal surface and subsurface waters. All near-river exchanges are included, regardless as to whether exchange flows remain on the surface or cross the sediment interface, or whether exchanges are driven by steady or dynamic river flow conditions. Hydrologic exchange of relatively fast moving water of the river's thalweg with zones of recirculating surface water behind boulders, logs, or other roughness features, and recirculation in lateral cavities at channel banks are included [*Jackson et al.*, 2013a], as is hyporheic flow of river water driven through subsurface flow paths vertically beneath the channel and laterally beneath the banks and marginal barforms of the river [*Boano et al.*, 2014; *Storey et al.*, 2003; *Cardenas and Wilson*, 2007]. Bank storage of water caused by rising and falling river stages that drives exchange flows laterally into and out of emergent bedforms, barforms, and banks [*Dudley-Southern and Binley*, 2015; *Larsen et al.*, 2014; *Boano et al.*, 2013; *Gerecht et al.*, 2011] is included. Also included are exchange flows with riparian areas and floodplains that are activated when rising rivers spill over low points on banks [*Wondzell and Swanson*; 1999; *Bridge*, 2009].

At some point, all of a river's flow has been exchanged with marginal, off-channel surface or subsurface waters. For example, *Richey et al.* [1989] estimated that the Amazon's main stem, the world's largest river, exchanges approximately 25% of its average annual flow with floodplains, an amount equal to more than twice the discharge of the Mississippi River. Flood waters spill over levees and traverse lower lying areas on the floodplain surface until returning to the river at a point downstream [*Mertes*, 1997]. Floodwaters also may remain ponded long after a flood, draining slowly back to the river after the flood recedes or, in some cases returning to the river by recharge to subsurface flow paths [*Jung et al.*, 2004]. The Mississippi River, itself the world's fifteenth largest river, exchanges all of its flow with subsurface (hyporheic) flow paths, including lateral exchanges beneath the river's meandering banks [*Kiel and Cardenas*, 2014] and vertical exchanges beneath the submerged bedforms on the bottom of the river [*Gomez-Velez and Harvey*, 2014]. These dynamic interactions between longitudinal and lateral fluxes within river corridors have far-reaching effects on water chemistry and ecology of downstream receiving waters.

This paper presents a vision of river corridor science by reviewing progress and outlining challenges in characterizing hydrologic connectivity, exchange flows, and related hydroecological processes. We highlight the unification of concepts, which brings together surface and subsurface hydraulic processes in river corridors at the very fine scale of turbulent transfer all the way up to larger-scale exchanges driven by basin-scale geologic controls. We discuss and compare useful abstractions of exchange flows, such as lateral and

10.1002/2015WR017617

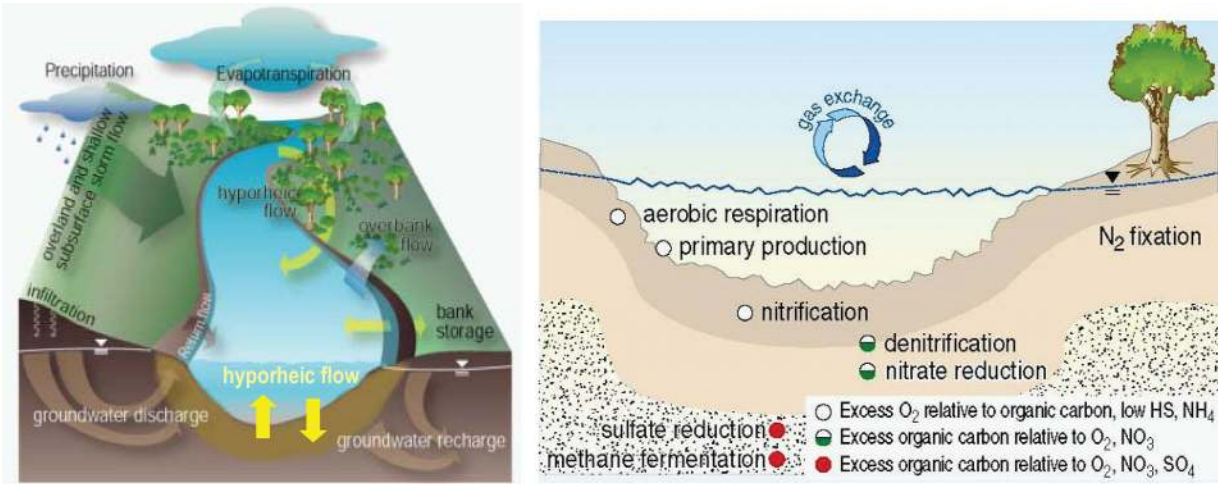

Figure 1. (a) The river corridor viewpoint emphasizes the interactive fluvial and hydrogeomorphic elements and hydrologic exchange flows between them as well as (b) mixing of chemically distinct waters in near-stream zones and resulting enhancement of biogeochemical reactions (left plot modified from *National Research Council* [2002]).

vertical hydrologic exchange flows, and contrast these with other water flows across landscapes including cross-valley flows and underflows. The vertical and lateral component of exchange flows are distinguished by either having a flow component perpendicular to the channel that is dominantly vertical beneath the channel bed, or dominantly lateral beneath the channel banks (Figure 2a), in addition to having flow components parallel to the channel's main axis. Other categorizations are more complicated and highlight the interface with inflows from runoff, subsurface stormflow, and deeper groundwater flow (Figure 2b).

We also emphasize how detailed classifications of hydrologic exchange flows can be useful, such as partitioning surface from subsurface exchange components of total exchange. For example, it can be important to distinguish *hyporheic* exchange from surface water exchange flows because of the increased biophysical opportunities for enhanced chemical reactions in waters that are in close contact with sediments [*Battin et al*., 2008]. Understanding the intermittency of hydrologic exchange flows also is important, and begins with distinguishing the types of exchange flows that are active even when river flow is steady (e.g., hyporheic flow) compared with other types (e.g., bank storage, overbank flooding) that are specifically activated by spates or floods. Even within a single category such as hyporheic exchange, it is important to differentiate between vertical exchanges beneath small (centimeter-scale) bedforms from much larger (hundreds of meter-scale) exchange flows beneath channel meanders, and beneath large bars or river islands. Exchange flows vary with river size and type, and with flow condition, and so approaches are needed that can cross scale while remaining faithful to the underlying physics and biology without becoming overwhelmed by the complexities.

The challenges facing river corridor investigations are many, including applying knowledge gained across the orders of magnitude in spatial scale, from the small scales of individual exchange flows at which controlling processes are understood to the much larger scales at which the water quality and ecological conditions are cumulatively affected (Figure 3). Consequently, although it has become increasingly feasible, for example, to predict hyporheic flow and other types of exchange flows at small spatial scales [*Boano et al*., 2014; *Jackson et al*., 2013a], there has been relatively little progress applying that knowledge to ecologically relevant scales of river reaches and watersheds (but see *Marzadri et al*. [2014], *Kiel and Cardenas* [2014], and *Gomez-Velez and Harvey* [2014]). As always, heterogeneity in the types and sizes of buried alluvium [*Heeren et al*., 2010, 2014; *Menichino et al*., 2014], and heterogeneity of barforms, grain sizes, and subsurface hydraulic conductivity [*Aubeneau et al*., 2015; *Cardenas et al*., 2004], and interactions with other types of roughness features (bioroughness, e.g., aquatic vegetation and downed wood in rivers) [*Jackson et al*., 2013a; *Wohl et al*., 2015] are crucial in their effect on hydrologic exchange flows and fate of dissolved and suspended materials in rivers. Another challenge is developing practical tools of measuring hydrologic exchange fluxes that avoid the potential bias of using a single technique, since no matter what method is used, it has a limited range of sensitivity that may only detect a portion of the water and chemical fluxes crossing hydrologic

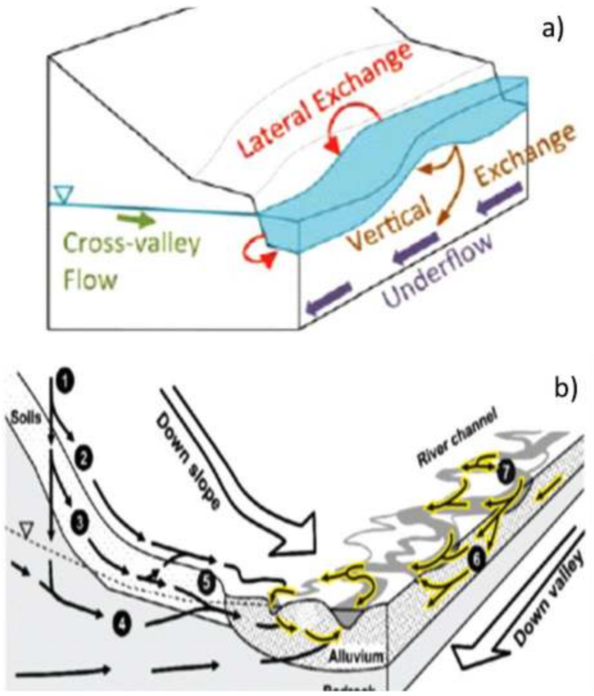

Figure 2. Comparatively simple and complex perspectives of river corridor inflows and hydrologic exchange flows through surface and subsurface pathways. (a) The first viewpoint identifies hydrologic exchange flows (with lateral and vertical components) and separates them from cross-valley flow and underflow. (b) The second viewpoint identifies river fluxes and hydrologic exchange fluxes (yellow highlighting) and separates them from hillslope processes such as infiltration (1), overland flow (2), subsurface stormflow (3), groundwater recharge and discharge (4), and floodplain exfiltration (5). Conceptual diagrams in Figures 2a and 2b are from *Ward et al.* [2013] (used with permission) and modified from *Poole* [2010], respectively.

interfaces [*Cook and Herczeg*, 2000; *Harvey and Wagner*, 2000]. An example of methodological limitations is detecting fluxes using decay of radioisotopic solute tracer along a flow path, which only provides results at sampling time scales that are similar to decay time scales. Lastly there is the challenge of quantifying "hot spots and moments" and "effective discharge" for biogeochemical processing of nutrients and organic matter in watersheds to understand river corridor functions and [e.g., *McClain et al.*, 2003; *Doyle et al.*, 2005]. We review the history and progress in these and other river corridor concepts and we also review the tried and true field practices with a view toward emerging techniques such as novel applications of geophysical techniques [*Briggs et al.*, 2014; *Lowry et al.*, 2007], thermal imaging [e.g., *Dugdale et al.*, 2015; *Briggs et al.*, 2013b], and satellite observations [e.g., *Alsdorf et al.*, 2007; *Townsend and Foster*, 2002]. Practical examples relating to management of river corridors are considered, such as concepts to quantify and characterize the chemical reactions of major concern and their relation to practical aspects of the conjunctive management of surface and groundwater management (e.g., bank filtration).

It is not enough to know that hydrologic exchange flows are effective in delivering reactants such as organic matter, dissolved nutrients, and oxygen into contact with microbe-rich sediments. We can also determine the specific roles of the various types of hydrologic exchange flows and predict where and when these are significant to cumulative outcomes for river water quality and ecology at the watershed and regional scale. Also there is much that can be done to strengthen the scientific basis of river corridor protections (e.g., riparian setbacks) and actions that are effective in restoring river corridor functions (e.g., improved storm water retention and channel restoration) [e.g., *Roley et al.*, 2012a, 2012b]. A predictive modeling framework for scientifically informed river corridor management, although becoming more accessible, is still in its infancy.

2. Origins of a River Corridor Perspective

Early concepts of environmental hydrology generally did not address the river corridor. Rather, hydrologic concepts were typically developed in isolation for rainfall-runoff, groundwater, and channel flow problems. Watershed hydrology, for example, emphasized fast transport through surface and shallow subsurface pathways of hillslopes into channels on time scales of a few minutes to a few days [*Loague*, 2010], whereas groundwater basin hydrology emphasized slower transport through deeper subsurface flow paths, eventually discharging to channels on time scales of tens of days to millennia [*Anderson*, 2008] (Figure 4, top). Meanwhile, the analysis of river flow routing and water quality emphasized flow, transport, and dispersive mixing in main channels [*Cunge*, 1969; *Fischer et al.*, 1979], generally without considering water inflows and exchanges with groundwater, riparian, floodplain, or hyporheic environments. Early scientific advancements were made in sanitary engineering that would influence reactive transport modeling in rivers including the Streeter-Phelps oxygen sag model [*Streeter and Phelps*, 1925]. Only later did traditional watershed and groundwater basin perspectives move away from depicting

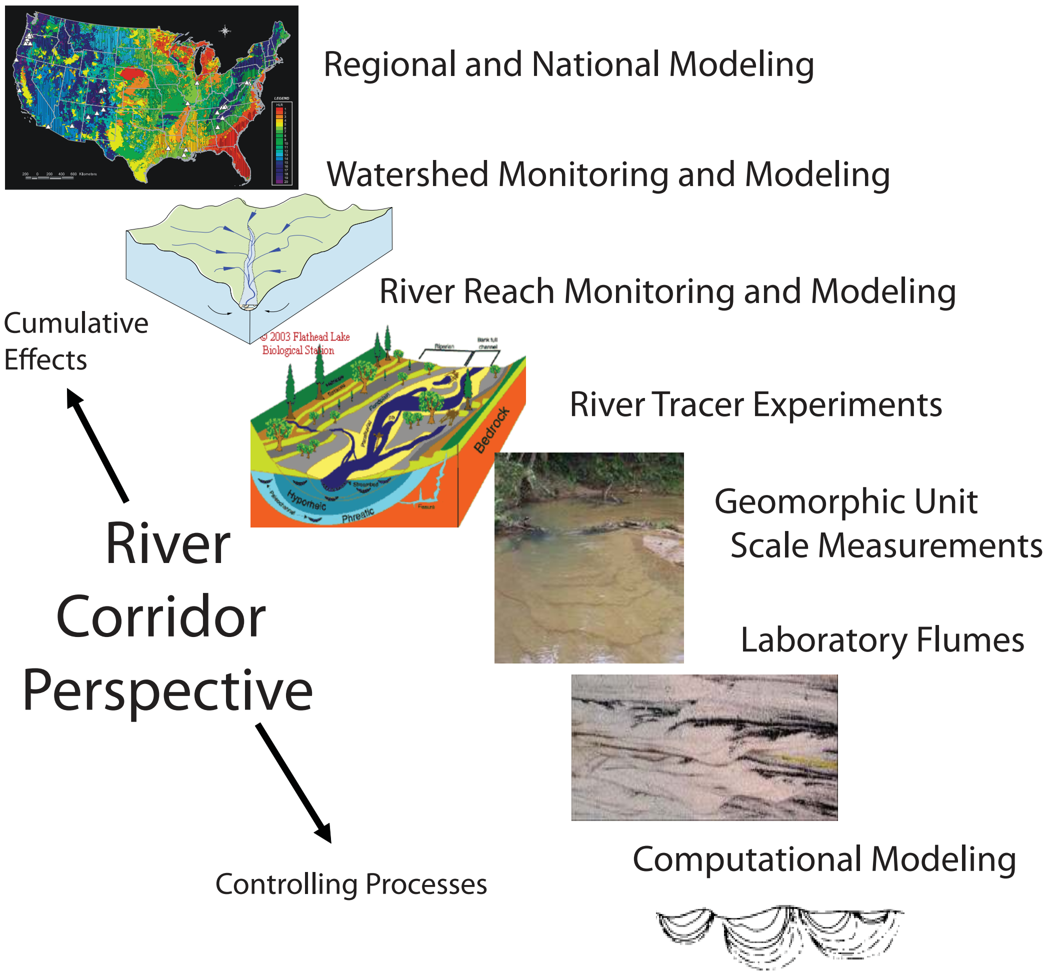

Figure 3. Ten orders of magnitude of inquiry of river corridor processes showing relation between small-scale understanding of the fundamental processes affecting biogeochemical reactions, habitats, etc. with larger-scale understanding of cumulative effects on water quality and ecological function in river basins, hydroclimatic regions, and nationwide.

streams as drains on the landscape toward recognizing the bidirectional exchanges near the river and hydroecological interactions that affect in-stream water chemistry and ecology (Figure 4, bottom). Some early contributions by hydrologists included identifying bank storage exchange of river water responding to dynamic river stages [*Pinder and Sauer*, 1971; *Moench et al.*, 1974], induced river recharge from groundwater pumping [*Chen*, 2001; *Hunt*, 1999; *Glover and Balmer*, 1954], overbank flow and floodplain exchange [*Mertes*, 1997; *Knighton*, 1998], and integrated reactive transport modeling in rivers exchanging water with surrounding subsurface waters [*Bencala*, 1983].

The origins of the river corridor perspective were in concepts that linked aquatic ecology with fluvial geomorphic principles. The River Continuum Concept (RCC) hypothesized predictable relations among food webs, aquatic metabolism, and fluvial geomorphology [*Vannote et al.*, 1980; *Minshall et al.*, 1985]. In other words, the hydrogeomorphic template of rivers matters to organisms. The RCC took into account the differing potential for autochthonous and allocthonous inputs of organic matter along the continuum from small to large rivers, as well as interactions between river size, overhang of riparian vegetation above the channel, and effects on light penetration and water temperature. Originally, a static view that emphasized longitudinal changes through river networks, the RCC viewpoint was extended by other concepts that emphasized dynamics, most notably the Flood Pulse Concept (FPC) of *Junk et al.* [1989] that highlighted the role of flood movement onto and off of riparian and floodplain zones where organic and inorganic materials are stored and transformed and where aquatic organisms feed or take refuge.

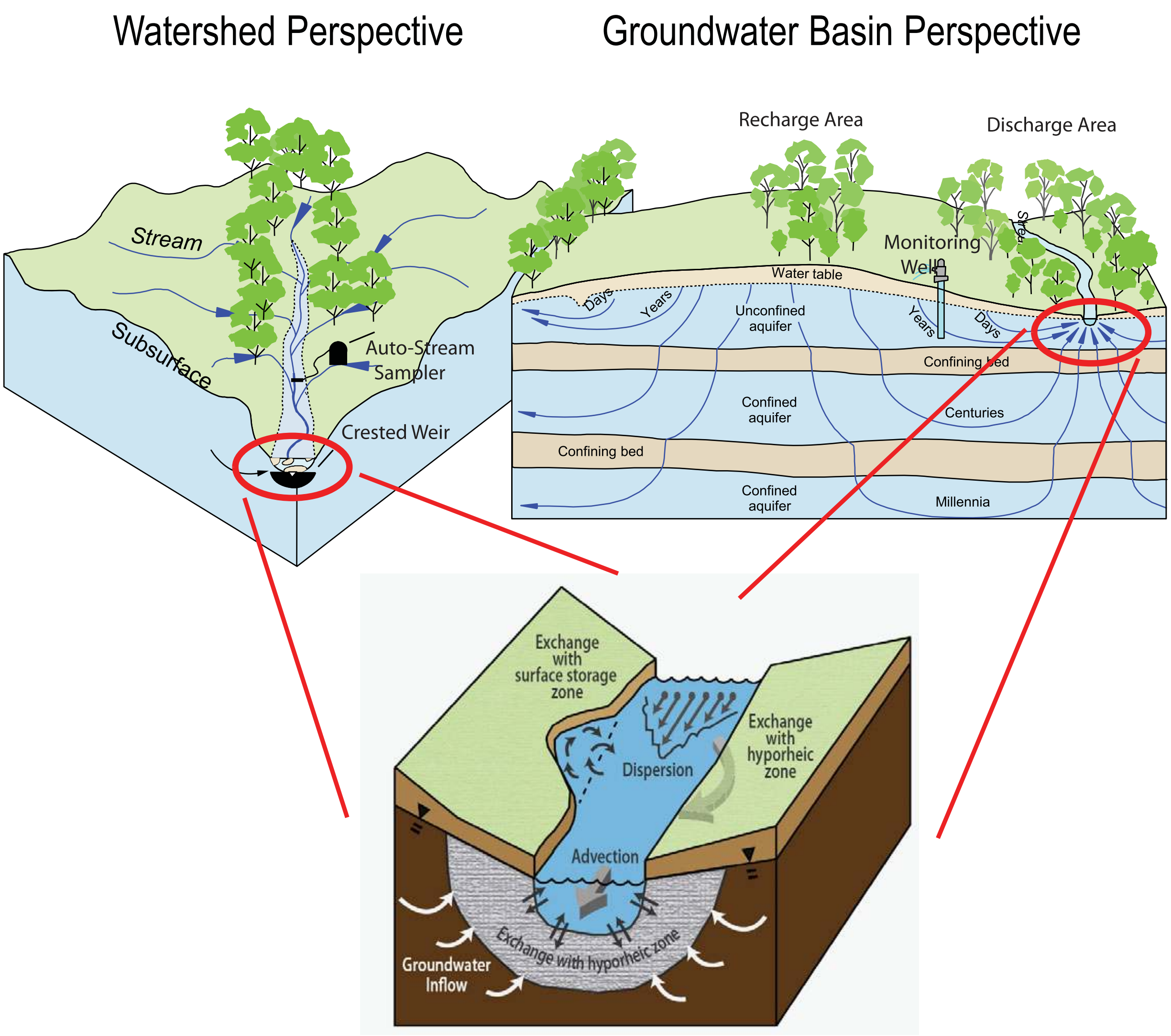

Figure 4. The river corridor in relation to watershed and groundwater basin perspectives of hydrologic transport.

The RCC and FPC concepts were developed in the spirit of testable hypotheses, although quantification of hydrologic exchange flows came relatively slowly. Early on the prevailing hydrologic instrumentation tended to be ill suited to measuring hydrologic exchange flows and impacts on water quality and ecology. Velocity-area-based measurements of river discharge, and typical groundwater wells (e.g., 5 or 10 m deep), provide end members but little in the way of useful information about interactions between the river's main channel and marginal surface water areas, or the permeable sediments and floodplains lying far outside of the river's wetted width [*Ward*, 1989; *Dunne et al*., 1998].

River corridor science gained momentum when hydrologists and aquatic ecologists combined their tools to quantify nutrient and contaminant spiraling through river networks. Beginning several decades ago, the traditional hydrologic instrumentation began to give way to new experimental approaches that could trace and quantify the ecological influence of hydrologic exchange flows [*Stream Solute Workshop*, 1990]. A major turning point in quantifying hydrologic exchange flows and their biogeochemical consequences came in the form of the Nutrient Spiraling Concept [*Newbold et al*., 1981, 1982]. Nutrient spiraling explains how downstream movement of nutrients (the concept is now applied to chemical contaminants and suspended sediments as well) is lagged by physical and biological storages, and that biophysical opportunities for chemical reactions are enhanced by storage. These and similar works were responsible for catalyzing decades of experimental work in rivers with solute tracers [e.g., *Battin et al*., 2008; *Boano et al*., 2014]. Tracer experiments revealed how solutes being rapidly transported in the main channel can enter storage within slowly moving marginal waters, or may be taken up by plants and aquatic organisms and spend time within biota until later released with waste products or after the organism or plant dies. Use of isotopic tracers such as ^{32}P and $^{15}NH_4$ for measuring spiraling through physical storage and food webs was particularly innovative [*Newbold et al*., 1981; *Peterson et al*., 2001]. Such processing has the potential to strongly affect downstream concentrations and transport loads as well as rates of transformation of reactive substances.

In addition to methodological advancements, concepts for hydrologic exchange flow also have advanced with scaling concepts such as the Hyporheic Corridor Concept of *Stanford and Ward* [1993], the Telescoping Ecosystem Concept [*Fisher et al*., 1998], the Natural Flow Regime concept [*Poff et al*., 1997], and the Effective Discharge Concept [*Doyle et al*., 2005], which expand the context for quantifying particular types of physical storage and biological processing across a spectrum of flow conditions. These and other emerging concepts, such as the lung versus gill models of hyporheic exchange [*Sawyer et al*., 2009], are now contributing as key test beds for quantifying the close contact between rivers and their surrounding off-channel waters. Most recently, hydrologists have contributed to integrated river management assessments including designing more effective management practices for protection and restoration of water quality and biological resources [*Bourg and Bertin*, 1993; *Von Gunten and Lienert*, 1993; *Fleckenstein et al*., 2004; *Ward et al*., 2011; *Hester and Gooseff*, 2010, 2011; *Bernhardt et al*., 2005].

3. Fluvial-Geomorphic and Ecological Drivers of Hydrologic Exchange Flows

The fluvial and biogeomorphic template of a river corridor is shaped by floods and sediment supply and evolves continuously [*Wohl et al*., 2015]. The connectivity of river networks is controlled in large part by the hydraulic forces that dissipate water's energy as it interacts with geomorphic and biological roughness elements on the streambed and at channel margins [*Jackson et al*., 2013b; *Prestegaard*, 1983] that resist the flow and force water to move laterally through pathways across the channel and beneath the surface. Resistance to exchange flows through the subsurface is controlled by the grain-size distribution and layering of river sediment which determines the sediment's hydraulic conductivity [*Cardenas et al*., 2004; *Brunke*, 1999]. Biological features such as woody and herbaceous vegetation also contribute to hydraulic resistance to surface flows [*Griffin et al*., 2005; *Harvey et al*., 2009], which can focus energy dissipation and reshape the sediment bedforms that steer and modulate surface and subsurface exchange flows.

Many of the recent conceptual advancements in river corridor science can be loosely categorized following *Montgomery* [1999] as Process Domain Concepts [e.g., *Poole et al*., 2006; *Boano et al*., 2014; *Wohl et al*., 2015]. These focus attention on how climate, geological, and topographic drivers produce predictable channel morphology, sediment grain size, and flows that shape the physical template for flowing aquatic ecosystems. Figure 5 is one example of a Process Domain Concept that illustrates relations between many of the mechanisms that control both surface and subsurface hydrologic exchange flows. The spatial extent of exchange and the associated time that river water spends in storage are positively related and scale approximately with the size of bedforms, barforms, and other roughness features such as downed wood in channels [*Kasahara and Wondzell*, 2003; *Stonedahl et al*., 2010; *Sawyer et al.,* 2011], as well as frequency, size, and duration of spates and floods [*Martin and Jerolmack*, 2013]. The scaling ranges across many orders of magnitude from the vertical circulation beneath submerged ripples and dunes on the channel bed to much larger exchange fluxes driven by meanders, bedrock undulations, and channel bank height variations that introduce variability in both surface water and groundwater exchanges with areas outside of the main channel and overbank flow [*Jung et al*., 2004; *Wörman et al*., 2007; *Stonedahl et al*., 2013].

At the basin scale, geologic factors such as valley slope and the extent and texture of the valley fill and alluvial sediments are important controls on hydrologic exchange flows [*Larkin and Sharp*, 1992; *Woessner*, 2000]. The size and frequency of channel-forming flows and texture of bed sediments determine the channel's geometry, including bankfull width, sinuosity, and types, sizes, and spacing of bedforms and barforms, all of which influence hydrologic exchange flows [*Aubeneau et al*., 2015; *Martin and Jerolmack*, 2013]. Hydrologic exchanges vary with time during spates and floods [*Ward et al*., 2013; *Zimmer and Lautz*, 2014] and punctuated discharge and water level variations in rivers regulated for hydropower also generate fluctuating surface-subsurface exchange flows [*Sawyer et al*., 2009]. Vertical exchange of water with the streambed

10.1002/2015WR017617

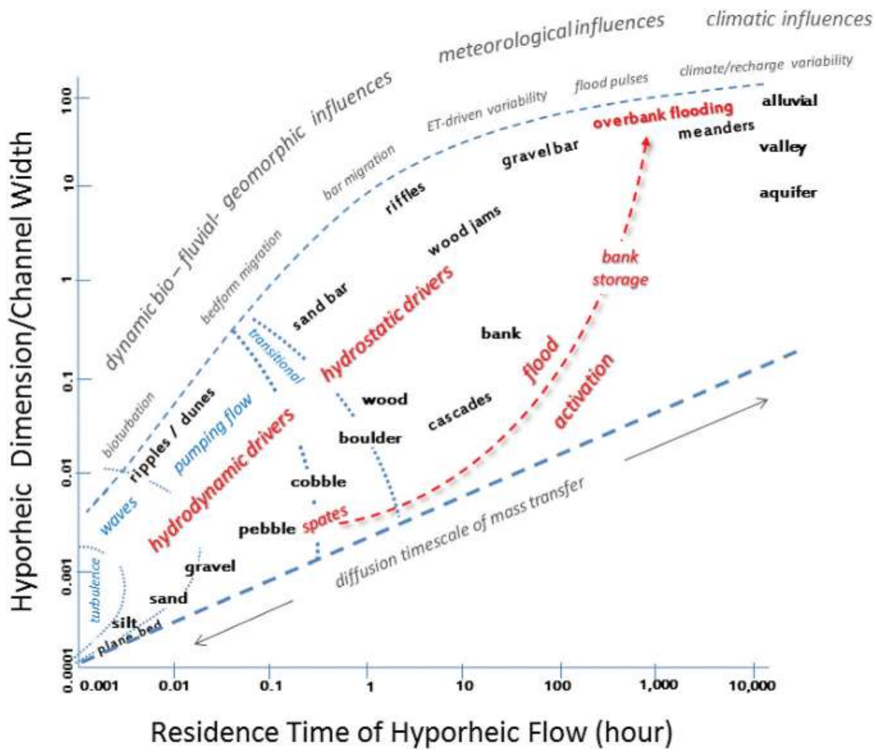

Figure 5. Fluvial, geomorphic, and biological influences in the river corridor. The scaling mainly considers hyporheic flow although the domains are relevant to all types of hydrologic exchange flows. Greater exchanges occur in rivers with geomorphically rough bed and banks. River width is a useful scaling factor however it only crudely explains the complex interactions between flow and geomorphic and ecological features that determine exchange fluxes with storage zones of a given size and residence time (from *Boano et al*., 2014, used with permission).

varies even with minor changes in stream velocities [*Boano et al*., 2013] (e.g., hydrodynamically driven hyporheic fluxes scale with the square of velocity [*Elliott and Brooks*, 1997a]) and will shift in response to flood-driven adjustment of bedforms [*Harvey et al*., 2012].

Within the channel, the surface water's kinetic energy in flow over submerged bedforms imposes kinetic and potential forces on the bed that drives subsurface exchange flows with zones of flow separation behind and beneath dunes, ripples, cobbles, and protruding clusters of grains [*Thibodeaux and Boyle*, 1987; *Savant et al*., 1987], with the redistribution of the fluid's momentum to pressure variations on the streambed causing shallow hyporheic flow that interacts with near-river groundwater heads [*Cardenas and Wilson*, 2007]. A rising and falling river may affect hyporheic flow in unanticipated ways because of complex and interdependent adjustments in river depth, energy slope, and extent of inundation of bar features [*Gariglio et al*., 2013; *Shope et al*., 2012; *Francis et al*., 2010; *Käser et al*., 2009; *Wondzell and Swanson*, 1996; *Harvey et al*., 1996]. Surface water is also exchanged laterally by shear forces driving turbulent exchange across boundaries of flow separation between the fast-flowing main channel from more quiescent waters of the channel margin or behind roughness features [*Jackson et al*., 2013b].

The presence of biological features including biofilms, algae, sticks and leaves, downed wood, and beaver dams adds flow roughness that interacts with sediment features to increase exchange between the main channel and zones of flow separation in off-channel areas and in subsurface hyporheic zones [*Sawyer et al*., 2011; *Battin*, 2000; *Battin and Sengschmitt*, 1999; *Lautz et al*., 2006]. Growth and decay of roots of aquatic and emergent plants in sediments with mixed grain sizes and fine particulate organic matter creates zones of preferential flow through sediment macropores with implications for biogeochemical cycling [*Briggs et al*., 2015; *Harvey et al*., 1995; *Harvey*, 1993]. Near-stream vegetation alters the direction and amount of near-stream subsurface flow by transpiration [*Wondzell et al*., 2010] and riparian and submerged and emergent aquatic vegetation also contribute substantially to flow roughness by shielding sediments from erosion and adding strength to river beds and banks [*Hopkinson and Wynn*, 2009; *Harvey et al.,* 2009].

4. Measuring Hydrologic Exchange Flows

The fundamental physical drivers of hydrologic exchange flows can be isolated and studied under controlled experimental conditions in laboratory flumes, allowing the manipulation of variables one at a time [e.g., *Endreny et al*., 2011; *Packman et al.,* 2000; *Elliott and Brook*s, 1997b]. These approaches sometimes suffer from the artificial nature of the setting, e.g., effects of flume and mesocosm walls, artificial effects of induced flows and stirring, and disruption of sediment structure and chemical characteristics compared to field conditions. However, laboratory measurements have provided an excellent linkage between theory and observations in controlled settings.

An advantage of field measurements is that they are more representative of actual stream conditions which include a wider range of exchange flows which may interact (e.g., smaller-scale flow paths nested within larger scale) [*Poole et al*., 2006] or which may co-occur with little apparent interaction across scales (bedform-scale hyporheic flow independent of pool and riffle-scale hyporheic flow) [e.g., *Stonedahl et al*., 2012]. Nonetheless, hydraulic models may suffer from poor knowledge of boundary conditions and challenges representing complex heterogeneities in surface and subsurface conditions [*Wondzell and Swanson*, 1996] including vegetation and other types of biological roughness [*Jackson et al.,* 2013a] that affect flow.

Consequently, many field researchers use a phenomenological approach to describe transport based on solute tracers, both injected and naturally occurring, and they forgo the measurement of many of the site-specific details in favor of simpler data sets that capture only the most essential details of solute transport that are pertinent to the question being addressed [*Jones and Mulholland*, 2000; *Stream Solute Workshop*, 1990]. Of course, what is essential and pertinent depends to a large extent on the spatial and temporal scales being considered.

Despite the challenges, complementary research in flumes and across many river types has promoted development of useful empirical equations and physically based scaling laws [e.g., *Jackson et al*., 2013b; *O'Connor et al*., 2010; *O'Connor and Harvey*, 2008] based on relatively easily measured physical variables. The topic of hyporheic exchange has been a widely advanced and debated in AGU journals (see recent reviews by *Cardenas* [2015] and *Boano et al*. [2014]), and the evolving role of stream tracers in advancing and integrating surface exchange flows and floodplain processes in river corridors is growing [*Jackson et al*., 2013a; *Wollheim et al*., 2014].

4.1. Hydrometric Methods

Measuring hydrologic exchange flows often requires nonconventional instrumentation designed for deployment in shallow surface and subsurface environments. Surface water flow measurements are difficult in complex channels that are shallow and wide with variable flow speeds around roughness features or through vegetation. As noted before, velocity-area-based measurements of river discharge do not address exchange flows, but they are also problematic for measuring longitudinal discharges in deep and fast-flowing rivers as well as in shallow streams with unstable streambeds—wherever wading or deploying relatively heavy and highly sensitive instruments measurements is difficult, dangerous, or problematic. Acoustic velocimeters originally developed for oceanography have taken a step toward miniaturization over the past few decades are often used for measuring surface flow in river, floodplain, and estuarine environments [*Nikora and Goring*, 1998; *Chanson et al*., 2008; *Harvey et al*., 2009]. Subsurface exchange fluxes can be estimated by measuring hydraulic head gradients in shallow subsurface sediments and pairing these with estimates of the sediment's hydraulic conductivity to compute a flux using Darcy's Law [e.g., *Storey et al*., 2003; *Wroblicky et al*., 1998; *Wondzell and Swanson*, 1996]. The evolution of measurements of hyporheic exchange provides a useful illustration of general advancements in river corridor measurements. In the earliest studies of hyporheic flow, piezometers were not necessarily used and hydraulic head was measured near streams simply by digging shallow pits dug into gravel bars and comparing water levels with the nearby channel [*Bencala et al*., 1984]. Soon after various types of miniaturized drivepoints were developed [*Wroblicky et al*., 1998; *Wondzell and Swanson*, 1996]. Essentially, these were conventional piezometers from hydrogeology that had been miniaturized and hardened for direct driving to shallow depths in river beds without the need for a pilot hole. Drivepoints with short, well-defined screens also function for measuring hydraulic conductivity using slug test approaches to support hydrogeologic modeling [e.g., *Storey et al*., 2003; *Cardenas et al*., 2004; *Käser et al*., 2009; *Stonedahl et al*., 2010]. Drivepoint designs differ considerably in their diameter with larger points mainly being necessary for strength to withstand slide hammering into streambeds with heterogeneous beds containing cobbles [*Geist et al*., 1998]. Finer grained and less heterogeneous streambeds allow lighter weight drivepoint designs (typically 0.375 inch or 1 cm nominal O.D.) that are manually emplaced into sand or sand and gravel beds. However, drivepoints generally are not stable enough under their own weight for shallow subsurface sampling, and their diameter and internal water volume generally require too much purging and disturbance of natural solute gradients to provide solute tracer or chemical samples any shallower than 20 cm beneath the streambed. The need for shallower hyporheic sampling with finer (centimeter-scale) vertical resolution was addressed by further miniaturization and multiplexing to sample six depths simultaneously with centimeter-scale resolution beneath the bed [*Harvey and Fuller*, 1998; *Duff et al*., 1998]. The design provides for minimal surface area exposed to river flow that could change the hydraulic pressures driving hyporheic exchange or cause bed scour, and minimal internal "dead" volume to allow slow pumping of small volume samples without disturbing natural subsurface flow patterns and solute gradients.

Another common hydrometric technique is the use of seepage meters to directly measure water flow across submerged sediment boundaries of lakes, wetlands, and streams. The original design was a steel drum top inserted into sediment with a port to attach a bag to monitor water accumulation or loss [*Lee*, 1977]. Seepage meters were originally designed for well-graded sandy sediments with relatively steep hydraulic gradients in the streambed that overcome the effects of minor surface pressure transients on bag pressures. The design has been modified several times to improve performance by minimizing seepage bag exposure to currents or to replace the seepage bag with electronic (heat-pulse) technology to measure flow. Seepage meters may be problematic in streams and rivers with mixed grain sizes, especially cobbles, and where exchanges fluxes are small, yet they have been used extensively to measure temporal and spatial variability of fluxes [*Kennedy et al*., 2009; *Conant et al*., 2004; *Rosenberry et al*., 2013].

The measurement approaches described above have been used effectively at relatively small scales ranging from a vertical deployment of sensors at a single point to horizontal layouts across larger features such as a gravel bar or meander, and, at most, deployment along a short stream reach. Such measurements reveal important characteristics and controls on hydrologic exchange flows. A chief downside is that data are representative of only a very small portion of a much larger, complex system and generally do not estimate reach-averaged conditions at distances of hundreds of meters to tens of kilometers, i.e., the distance at which the effects on water quality become evident [*Ward et al*., 2014; *Harvey and Wagner*, 2000]. Although it is not impossible to characterize groundwater-surface water interactions by installing many (e.g., hundreds) of drivepoints and modeling results in order to obtain reach-scale averaged exchange flows [e.g., *Wroblicky et al*., 1998; *Baxter and Hauer*, 2000], in most cases the workload is prohibitive. There is a trade-off as instruments become more widely spaced and it becomes more difficult to resolve what often are the dominant drivers of exchange fluxes across centimeter to decimeter-scale bedforms on the streambed [*Stonedahl et al*., 2013; *Wörman et al*., 2007]. *Gomez-Velez and Harvey* [2014] and *Marzadri et al*. [2014] recently developed parsimonious physically based models of hyporheic flow with potential for application in large basin to national scales.

4.2. Tracer-Based Methods

In situations where traditional hydrometric measurements of river flow are not enough to reliably specify hydrologic exchange fluxes, solute tracers (or other tracers, such as heat transport) are often useful. Often, traditional hydrometric measures of river are often combined with environmental tracers [*Cook et al*., 2006; *Gooseff et al*., 2003; *Choi and Harvey*, 2000] to produce spatially averaged measurements of river inflows and outflows from groundwater. Sometimes a coupled water and solute flux balance can be developed where bidirectional exchanges (e.g., hyporheic flow or surface water exchange with slowly flowing marginal areas) can be distinguished from unidirectional inputs or exports of water to locations far outside the reach, e.g. (groundwater or tributaries) [*Payn et al*., 2009; *Ruehl et al*., 2006; *Harvey and Wagner*, 2000]. *Payn et al.* [2012] linked multiple tracer injections in a river to reveal that valley and watershed topography as well as the character of the hillslope-riparian transition controlled river inflows and outflows when river baseflow was relatively high. However, during late summer when baseflow was lowest there was greater control by proximity to geologic fault zones and other aspects of subsurface structure.

4.2.1. Environmental Solute Tracers

Environmental solute and temperature tracing has played an important role in river corridor studies in quantifying groundwater discharge, recharge, and hydrologic exchange flows in river reaches, wetlands, and floodplains. To be useful, environmental tracers should be naturally present at different levels within main channels, off-channel surface waters, hyporheic zones, and groundwater. The variable distribution of environmental tracers provides the sensitivity to calculate mixing between waters from various source areas and exchange fluxes between those areas.

Examples of environmental solute tracers include specific conductivity, Cl^- or other major ions [*Mulholland*, 1992; *Reddy et al.*, 2008; *Larsen et al.*, 2014], and water stable isotopes [*Gooseff et al.*, 2003; *Böhlke et al.*, 1997; *Hunt et al.*, 2005], which are typically used to quantify inflows from various water sources from outside the main channel. They also may help constrain hydrologic exchange fluxes entering or leaving the main channel if, for example, temporal variations in solute tracer concentrations in main channel propagate into off-channel or subsurface waters. Propagation of solute signals into off-channel waters can be used to inversely estimate the flux by fitting of the tracer's lag and attenuation characteristics. *Böhlke et al.* [1997] used the lag and attenuation characteristics of water stable isotopic composition in alluvial groundwater as a basis for calculating exchange fluxes with the Danube River. Other researchers have tested the measurements and modeling of temporal changes in electrical conductivity in a stream and in a nearby observation wells to determine hydrologic exchange fluxes [e.g., *Cirpka et al.*, 2007]. Typically, the technique works best where recharge dominates, as in a situation where pumping of wells occurs adjacent to streams.

A number of radioisotopes are used in tracing river corridor exchange fluxes [*Cook et al.*, 2006; *Krest and Harvey*, 2003; *Harvey et al.*, 2006]. Recharge fluxes can be quantified by using a radioisotope's well-known decay constant and field estimated equilibrium concentration to quantify hydrologic exchange fluxes and transit times. The analysis typically requires that the tracer be paired with other conservatively transported environmental tracers to account for mixing and dilution [e.g., *Bourg and Bertin*, 1993; *Lamontagne and Cook*, 2007]. Alternatively, two radioisotopes can be expressed as a ratio to account for dilution. Decay time scales of naturally occurring radioisotopes range from days to millions of years: ^{222}Rn (1–10 days), $^{3}H/^{3}He$ (0.1–50 years), ^{224}Ra and ^{223}Ra (3.6–11.4 days), SF_6 (1–40 years), tritium (5–90 years), ^{39}Ar (70–700 years), ^{14}C (200–20,000 years), and ^{36}Cl (10^5-10^6 years). The radioisotopes of greatest interest to studies of hydrologic exchange fluxes and groundwater-surface water interactions are generally those at the short end of this spectrum, including radon [*Cook et al.*, 2006; *Lamontagne and Cook*, 2007; *Hoehn and Cirpka*, 2006], short-lived radium isotopes [*Krest and Harvey*, 2003], and $^{3}H/He$ [*Price et al.*, 2003]. These approaches also provide a foundation for quantifying reactive transport [e.g., *Bourg and Bertin*, 1994].

4.2.2. Injected Solute Tracers

The value of solute tracing to measure river exchange flows is sometimes improved by injecting solutes directly into rivers and then tracking their downstream movement. The injection of solute tracers that are conservatively transported in natural waters have long been used to quantify a river's discharge [*Kilpatrick and Cobb*, 1985] and the rate of dispersive mixing in the longitudinal direction during downstream transport [*Fischer et al.*, 1979]. There has been an increasing emphasis on using tracers to quantify hydrologic and chemical exchange fluxes perpendicular to the channel's main axis [*Stream Solute Workshop*, 1990; *Jones and Mulholland*, 2000]. For example, a tracer can be injected in the stream's thalweg and tracked through surface and subsurface flow paths into marginal waters to determine exchange fluxes [e.g., *Harvey and Bencala*, 1993], or the tracer can be injected directly within a side cavity at the stream's margin, or within the subsurface and then tracked across the interface with the river's main channel [e.g., *O'Connor et al.*, 2010; *Jackson et al.*, 2013b; *Gooseff et al.*, 2013; *Harvey and Wagner*, 2000].

Tracers, such as rhodamine WT, chloride, and bromide, are added either as an instantaneous pulse addition or by injection at constant rate for a specified period of time [*Kilpatrick and Cobb*, 1985; *Wagner and Harvey*, 1997]. Sampling of the breakthrough of the tracer at a point downstream of where the tracer has become mixed with depth and across the width of the river is used to characterize discharge by a method known as "dilution gaging" [*Kilpatrick and Cobb*, 1985]. Breakthrough measurements at points further downstream are useful for characterizing inflows from groundwater, seeps, and tributaries. The net inflow to the river reach is quantified as the difference between dilution gaging estimates at reach endpoints. Importantly, the tracer-based approach often overestimates true discharge at points downstream if losses of river water occur by recharge to groundwater, irrigation takeoffs, etc. If independent measurements of discharge are acquired at reach endpoints, e.g., by velocity gaging or by additional solute tracer injections at all reach endpoints, then both inflows and outflows from the channel may be quantified by solving for two unknowns in the coupled mass balance equations for the river reach, e.g., water mass balance equation and solute tracer mass flux equation [*Payn et al*., 2009; *Ruehl et al*., 2006; *Harvey and Wagner*, 2000].

Temperature tracing of hydrologic exchange fluxes grew enormously in the past few decades with improving sensors available at a decreasing cost, and with development of convenient analytical and numerical solutions for quantifying water exchange fluxes [e.g., *Hatch et al*., 2006; *Swanson and Cardenas*, 2011; *Voytek et al*., 2014]. Most applications involve vertical deployments of temperature sensors in streambeds [e.g., *Hatch et al*., 2006], wetlands [e.g., *Hunt et al*., 1996], and on floodplain surfaces [*Hess et al*., 2011] or in alluvial groundwater [*Johnson et al*., 2005]. Temperature tracing at the scales of individual geomorphic features have provided two-dimensional flow path interpretations, including flow path mapping to define and separate river exchange with hyporheic and groundwater flow paths [*Gariglio et al*., 2013; *Nowinski et al*., 2011; *Shope et al*., 2012; *Francis et al*., 2010]. Measuring vertical fluxes of water across streambeds using heat as a tracer is now often being supplemented with longitudinal distributed temperature sensing applications that follow the main channel to address spatial variability [*Selker et al*., 2006; *Lowry et al*., 2007]. Other recent developments include increased resolution in vertical temperature measurements to define fine-scale subsurface exchange [*Briggs et al*., 2012] and long-term measurements that permit estimation of hyporheic flow dynamics [*Bhaskar et al*., 2012] and temporally varying groundwater-surface water interactions [*Mwakanyamale et al*., 2012; *Briggs et al*., 2012] during floods.

4.2.3. Modeling Reactive Solute Tracers to Characterize Hyporheic Exchange Flows

Often a reactive solute tracer is injected or observed along with a conservatively transported solute tracer for the purpose of quantifying solute reaction [e.g., *Runkel et al*., 1996a, 1996b; *Covino et al*., 2011]. In such cases, a coupled model of transport and reaction in stream corridors can be constructed. An example is the transient storage model, which simulates in-channel advection and longitudinal dispersion in a stream, as well as hydrologic connections with groundwater and with "transient storage zones," i.e., slowly moving surface water at channel sides and in shallow subsurface (hyporheic) waters [e.g., *Thackston and Schnelle*, 1970; *Valentine and Wood*, 1979; *Bencala and Walters*, 1983; *Jackman et al*., 1984; *Castro and Hornberger*, 1991; *Runkel*, 1998; *Harvey and Wagner*, 2000; *Fernald et al*., 2001], as well as reactive processes which may occur at different rates within various hydrologic compartments (Figure 6). In the past few decades, alternative models have been added to the transient storage model family that have more detailed characterization of the storage-exchange processes [e.g., *Wörman*, 2000; *Choi et al*., 2000; *Wörman et al*., 2002; *Haggerty et al*., 2002; *Briggs et al*., 2009; *Neilson et al*., 2010; *Bottacin-Busolin et al*., 2011].

The model conceptualization of transient storage is extremely basic. It assumes that the stream water and solute exchange repeatedly with a well-mixed storage zone. The reach-averaged parameters determined by fitting include an exchange coefficient, a storage zone size, and an average residence time in storage. Rearrangement of those quantities produces a reach-averaged hydrologic exchange flux per unit length of stream, which is equal to the storage zone cross-sectional area parameter divided by the average storage residence time parameter [*Harvey and Wagner*, 2000]. The mathematical form of the residence time distribution for a well-mixed storage reservoir is exponential, which often fits tracer data in real streams quite well, but does not mean that transient storage zones are actually well-mixed reservoirs, but rather that storage zones can be approximated by an exponentially distributed assemblage of many short exchange pathways and a few much longer exchange pathways. Solute tracers added to rivers at concentrations several orders of magnitude greater than background often reveal that residence times of hydrologic storage are more broadly distributed than exponential, exhibiting lognormal or power law distributions [*Wörman et al*., 2002; *Haggerty et al*., 2002], which has been confirmed by measuring residence time distributions directly in storage zones [*Gooseff et al*., 2008; *Harvey et al*., 2013].

Chemical reactions in stream corridors often are characterized by first-order uptake or production terms in the main channel and in the storage zone of transient storage models. However, a wide variety of chemical reactions can be simulated in addition to first-order decay, including gas exchange [*Choi et al*., 1998], sorption from main channel waters onto the streambed or in the storage zone [*Bencala*, 1983], and various other equilibrium speciation or kinetically controlled speciation reactions [*Runkel et al*., 1996a, 1996b, 1999; *Bencala*, 1983]. An easily used model of in-stream transport with dispersive mixing and first-order uptake terms and sorption terms is available from the USGS in its well-documented OTIS code [*Runkel*, 1998] that

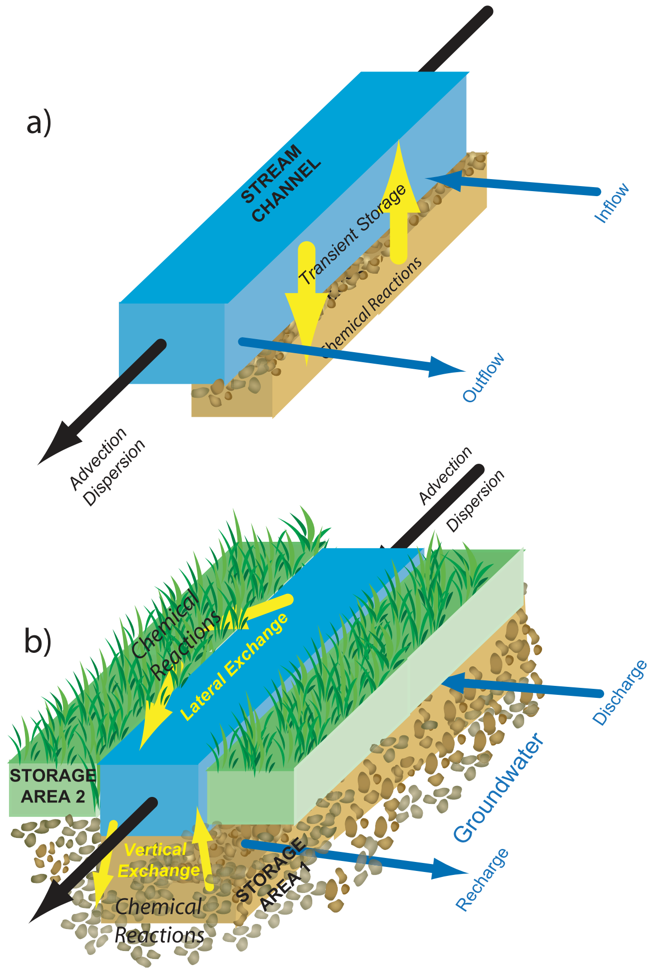

Figure 6. Dynamic analysis of solute transport: the transient storage model. (a) Mass transport by advection and longitudinal dispersion is modeled in a stream that exchanges mass with a transient storage zone, i.e., a well-mixed storage reservoir of a specified size and residence time beneath or alongside the main channel where more slowly moving surface or subsurface waters are present. The effect of transient storage is to delay downstream mass transport and to increase opportunities for chemical reactions in contact with geochemically and biologically active surface coatings on vegetation leaves and sediment grains. (b) Two-storage zone models have become popular to separate surface from subsurface transient storage.

provides a built in statistical code for inversely estimating parameters. The OTIS model expands on equations solved in *Bencala and Walters* [1983] and bears a close relation with similar models described by *Newbold et al.* [1982], *Stream Solute Workshop* [1990], *Mulholland et al.* [1997], *DeAngelis et al.* [1995], and *Jones and Mulholland* [2000]. *Mulholland et al.* [1997] used transient storage modeling to show that hyporheic zones increase important biogeochemical reactions such as heterotrophic metabolism and phosphorus uptake in streams. Several other detailed examples of transient storage modeling are given by *Kim et al.* [1990, 1992] who simulated transient storage and nitrate uptake kinetics of a periphyton community in a flume and then in a stream. *Kimball et al.* [1994] conducted tracer injections to measure metal loading from acidic mine drainage and to determine reaction rates that attenuate metals in the stream and in hyporheic zones. *Broshears et al.* [1996] manipulated stream pH and simulated the controls on aluminum and iron chemistry. The role of hyporheic zones in denitrification was investigated using transient storage modeling by *Gooseff et al.* [2004] and *Harvey et al.* [2013]. *Ensign and Doyle* [2006] examined results from many tracer experiments that quantified nutrient retention in streams and assessed the possible role of hyporheic flow. The reactive solute transport capabilities of OTIS were expanded in the One-Dimensional Transport with Equilibrium Chemistry model (OTEQ) that solves the equilibrium submodel MINTEQ in the context of transport with groundwater-surface water interactions [*Runkel*, 2010]. This model has been extensively tested in applications involving fate of metals released by acidic mine drainage.

5. Challenges Crossing Scales From Geomorphic Units to River Reaches and Basins

One of the valuable outcomes of transient storage modeling is the reach-scale estimates that link hydrologic transport, water quality, and stream ecology. There is a significant advantage in the spatial averaging of highly heterogeneous processes at the reach scale, but the approach is not without its challenges. For example, there are several limitations in the physical interpretation of storage processes. Stream tracer injections tend to be operationally intensive and prohibitively expensive in preparing the quantities needed for injections in large rivers [e.g., *Runkel*, 2015] or rivers with large and retentive storage zones in marginal channel areas [e.g., *Harvey and Wagner*, 2000]. These limitations place a limit on the size of rivers and the size (actually, the residence time) of storage zones that can be investigated with injected river tracers. Also, to simplify the measurements and analysis, the injections are usually restricted to times of steady flow, usually base flow, which negates their widespread use to characterize storm or flood-driven flows and concentrations. Finally, and perhaps most important, stream tracers typically only characterize a lumped hydrologic exchange process that combines effects of several types of exchange flows. Consequently there is the difficulty of specifying where transient storage occurs, i.e., in surface side zones of streams or in the subsurface, which potentially affects the amount of reaction that occurs, which is problematic for predicting water quality effects at the reach scale or drainage basin scale. For example, whether transient storage occurs in surface water at channel sides or in hyporheic flow paths is particularly important for understanding controls on chemical reactions such as metal uptake in mining areas or denitrification in hyporheic flow paths. Early on it was thought that because transient storage was the result of an assemblage of processes that are difficult to predict based on easily measured physical attributes, it might mean that results from one experiment could not be transferred to other streams or even within the same stream at a different discharge [*Wörman et al*., 2002; *Harvey et al*., 2003]. Fortunately, progress was made since then indicating that useful predictive models are not out of the question.

5.1. Discriminating Surface Water Exchange Flows From Subsurface Exchange Flows

A major challenge for river corridor scientists has been distinguishing between the contributions of surface and subsurface exchange flows to the overall, reach-averaged exchange. Simultaneously measuring tracer dynamics at the reach scale and at representative point locations led to key advancements over the past few decades. Measurements are made in representative storage zones of various types. For example, subsurface hyporheic flow exchange is often compared with surface water exchange with side cavities at the channel's margin. Such comparisons aid in parameterizing river corridor transport models with multiple types of storage zones. The earliest example of that approach was by *Bencala et al*. [1984] working in a boulder bed coastal stream in California, where the authors injected solute tracers in the stream to demonstrate how transport was affected by hydrologic exchange flows across the streambed. *Harvey and Bencala* [1993] directly observed hyporheic flow paths from start to finish and showed that only one class of hyporheic flow paths through gravel bars were observable with injected tracers. Larger and longer time scale hyporheic flow through alluvium up to several meters away from the stream was not detectable except by measuring head gradients, and was verified through measurement of the stream tracer in alluvial flow paths. *Harvey et al*. [1996] used small-scale measurements of hyporheic flow to specify parameters of a transient storage model. *Ensign and Doyle* [2005] investigated reach-scale effects on storage dynamics of creating surface water storage zones by adding wood baffles as surrogates for wood debris. The model representation of complex stream corridors was extended by adding additional storage zones with different exchange fluxes, storage-zone sizes, water residence times, and rates of chemical reaction [e.g., *Choi et al*., 2000]. *Harvey and Fuller* [1998] and *Wörman et al*. [2002] simultaneously measured reach-scale transport and hyporheic transient storage (HTS) and contrasted resulting storage areas and residence times with the reach-scale quantities. *Gooseff et al*. [2005] compared transient storage in neighboring stream reaches with and without substantial bedrock exposure to isolate HTS from surface water transient storage (STS). Similarly, *Harvey et al*. [2005] directly measured both STS and HTS in a flowing wetland and *Briggs et al*. [2009] directly measured the STS component of hydrologic exchange in a river and accounted for HTS as the remaining unexplained component. *O'Connor et al*. [2010] and *Jackson et al*. [2013b] added solute tracers directly to side cavity STS zones in streams and measured their flushing rate to estimate STS residence times and to develop hydraulic analysis of surface water exchange with side cavities. *Wörman et al*. [2002] and *Stonedahl* *et al*. [2012] evaluated physically based measurements in the subsurface and *O'Connor et al*. [2010] and *Jackson et al*. [2013b] evaluated hydraulic measurements in surface water, respectively, that can be used to estimate HTS and STS dynamics at the stream reach scale. *Gooseff et al*. [2011] used signal deconvolution techniques to separate main channel and STS tracer signals in order to estimate the STS residence time distribution. The results indicated substantial deviation from the ideal behavior of the simple systems depicted in Figure 6, which is indicative of the challenges that remain in discriminating surface and subsurface exchange flows.

5.2. Identifying Where and When Biogeochemical Reactions are Enhanced

The increasing capability of researchers to separate the effects of hydrologic exchange zones has led to improvements in understanding the zonation of key biogeochemical reactions. Enhanced biogeochemical reactions have been detected in streambed algal mats [*Gooseff et al*., 2004], shallow streambed hyporheic zones [*Argerich et al*., 2011; *Harvey et al*., 2013; *Briggs et al*., 2013a], gravel bars [*Pinay et al*., 2009; *Zarnetske et al*., 2011], bank storage exchange zones in river banks [*Squillace et al*., 1993; *Gu et al*., 2012], riparian zones [*Ensign et al*., 2008; *Wollheim et al*., 2014], and floodplains [*Richardson et al*., 2004; *Forshay and Stanley*, 2005; *Jones et al*., 2014; *Scott et al*., 2014], suggesting that sediment interfaces throughout the river corridor are capable of enhancing reactive uptake of nutrients and contaminants.

Rates of biogeochemical reaction rates have been compared between different subenvironments of the river corridor. Nitrogen removal, for example, has been compared within surface water exchange zones in side cavities and hyporheic zones [*O'Connor et al*., 2010], thalweg and bank margin hyporheic zones [*Harvey et al*., 2013], and near levee and backwater floodplain environments [*Richardson et al*., 2004]. A greater challenge is to understand the contribution of reactions in any one zone to river processing as a whole. *O'Connor et al*. [2010] and *Stewart et al*. [2011] assessed the relative importance of surface water transient storage and hyporheic transient storage on nitrogen removal in streams, although some simplifying assumptions were used to partition reactions rather than supporting conclusions with direct sampling of reactions in subenvironments. *Argerich et al*. [2011] found that conversion of resazurin to resorufin dye, a proxy for stream metabolism [*Haggerty et al*., 2008], occurred in a zone significantly smaller than the entire hyporheic zone. That result was supported by measurements of reaction rates as a function of depth in hyporheic zones that indicated that the dominant removal of nutrients and contaminants from rivers, enough to explain basin-scale outcomes for downstream water quality, may be isolated within the shallowest part of the hyporheic zone and not through its entire depth [*Harvey and Fuller*, 1998; *Harvey et al*., 2013; *Briggs et al*., 2013a]. The upshot is that hyporheic zones, riparian zones, and floodplains are important, however, the dominant reactions may only occur in a small part of the exchange zones. These findings could explain why large-scale studies comprised of many stream tracer experiments in contrasting streams (such as the LINX experiments, e.g., *Mulholland et al*. [2008]), were generally unable to explain differences in reach-scale reactions based purely on tracer estimated hydrologic exchange parameters such as average size and residence time of hydrologic exchange zones [*Ensign and Doyle*, 2006; *Webster et al*., 2003]. Instead it appears that better physically and chemically based measures of flow and reactions in different types of hydrologic exchange zones are going to be needed to accurately predict and understand the controls on river corridor reactions [see for example, *Marzadri et al*., 2014].

5.3. Physically Based Versus Statistical-Empirical Models of Hydrologic Exchange

Substantial progress has been made in modeling the multiple types, and scales, of hydrologic exchange flows. *Stonedahl et al*. [2010, 2012, 2013] used relatively straightforward measurements of stream planform, streambed topography, water-surface slope, sediment hydraulic conductivity, etc., in order to construct a quasi-three-dimensional model with multiple scales of surface-subsurface water interactions. The authors tested their predictive model "head to head" against a posteriori modeling of tracer tests conducted in a stream in Indiana where tracer test results also had been modeled inversely using the transient storage model. While it can be said that the physically based model could not be implemented completely independently of the tracer analysis, the physically based model did provide more accurate and informative fits to field tracer test data. The model of *Stonedahl et al*. [2012] was particularly well suited to predicting delayed transport due to storage in deeper hyporheic flow paths that was not detectable by the transient storage model. However, *Stonedahl et al*.'s [2012] model was not at all good at predicting rapid hydraulic transport through the main channel and needed calibration with tracer data to fit a longitudinal dispersion coefficient.

In contrast to *Stonedahl et al.*'s [2012] multiscale model of river corridor transport, *O'Connor et al.*'s [2010] model was essentially a transient storage model that used a relatively simple and computationally inexpensive physical submodel to estimate surface water exchange with side cavities. *O'Connor et al*. [2010] contrasted the physically based modeling with another version of the transient storage model where exchange parameters were estimated using simple statistical relationships determined from previously published data and dimensionless groupings of simple to measure physical variables, e.g., Darcy-Weisbach friction factor [e.g., *Harvey and Wagner*, 2000; *Zarnetske et al*., 2007]. The use of statistical relationships or simple scaling laws to infer parameter values is straightforward and much less expensive and time consuming compared to undertaking a tracer test from scratch. Physical modeling of hyporheic flow produced estimates with better than order of magnitude accuracy compared to the expensive alternative of conducting a tracer test and fitting with the transient storage model. The scaled down statistical estimation alternatives only require basic field measurements of stream velocity, average depth and streambed slope, dimensions of major classes of topographic features, and estimates of bed hydraulic conductivity [*O'Connor et al*., 2010]. The authors concluded that the predictions using a physically based model were marginally better than those obtained using the published statistical relationships noted above or simple scaling laws [e.g., *Wörman et al*., 2002; *O'Connor and Harvey*, 2008; *Grant et al*., 2012].

Statistically based scaling approaches are the most efficient means of estimating hydrologic exchange flows, and are likely to see more use in the future. Physically based modeling is conducted at greater cost, but produces results that potentially can be transferred to other streams and flow conditions. There may be a middle ground in which a quasi-physically based model can be developed that avoids two-dimensional structure by acquiring some simple field tracer data (dye releases) and basic topographic analysis and slug tests in streambed sediments. Such measurements can be made in half a day at relatively small expense compared with tracer testing which often takes several days at least to complete the analysis. The above discussion of modeling hydrologic exchange flows only encompassed steady flow in channels without addressing the complications of rising and lowering stream stage and the expanding wetted river width as the bank overtops. Floodplain inundation is inherently a more demanding subject for physically based modeling than is steady flow modeling, and the approaches used to span the gamut between highly detailed physically based modeling informed by on the ground data collection during a few intensively studied floods [e.g., *Bates et al*., 1997, 1998] to much more empirical approximations of flooding based on widely collected airborne and remote sensing data [*Alsdorf et al*., 2007]. Using increasingly available syntheses of large data sets and new sources of airborne lidar and satellite data, modelers are responding by developing approaches to simulate river corridor transport in large basins and at regional and continental scales [e.g., *Kiel and Cardenas*, 2014; *Gao et al*., 2014; *Marzadri et al*., 2014; *Cohen et al*., 2014; *Gomez-Velez and Harvey*, 2014].

6. Hydrologic Exchange Measurements Have Inherently Limited Sensitivity

Hydrologic exchange fluxes are rarely measured directly in complex river corridors. More often, fluxes are indirectly estimated, e.g., as the unmeasurable residual term in a water balance, or Darcy flux calculations based on an estimated sediment permeability, or by using decay of radioisotopes along flow paths to estimate fluxes, or using heat as a tracer. All such methods have various levels of parameter uncertainty, including relatively certain rates of decay of radioisotopes such as tritium and relatively uncertain parameters such as sediment hydraulic conductivity and other parameters with uncertainties that are difficult to estimate because of the substantial heterogeneity of river systems. These uncertainties also influence modeling errors that arise from factors such as inadequate spatial resolution or incorrect identification of end member sources in chemical mixing analyses.

A less well-recognized uncertainty in measuring hydrologic exchange fluxes is the inherently low sensitivity to detect fluxes outside a limited range. The range of detection is set, at least in part, by experimental design and by characteristics of the method being used. For example, the accuracy of a hydrologic exchange flux measured using a radioisotopic tracer will depend on the decay time scale of the tracer relative the transport time scale in the system where it is measured (Figure 7b). Another example is the accuracy of water balance modeling and how it is influenced by the resolution at which hydraulic heads are

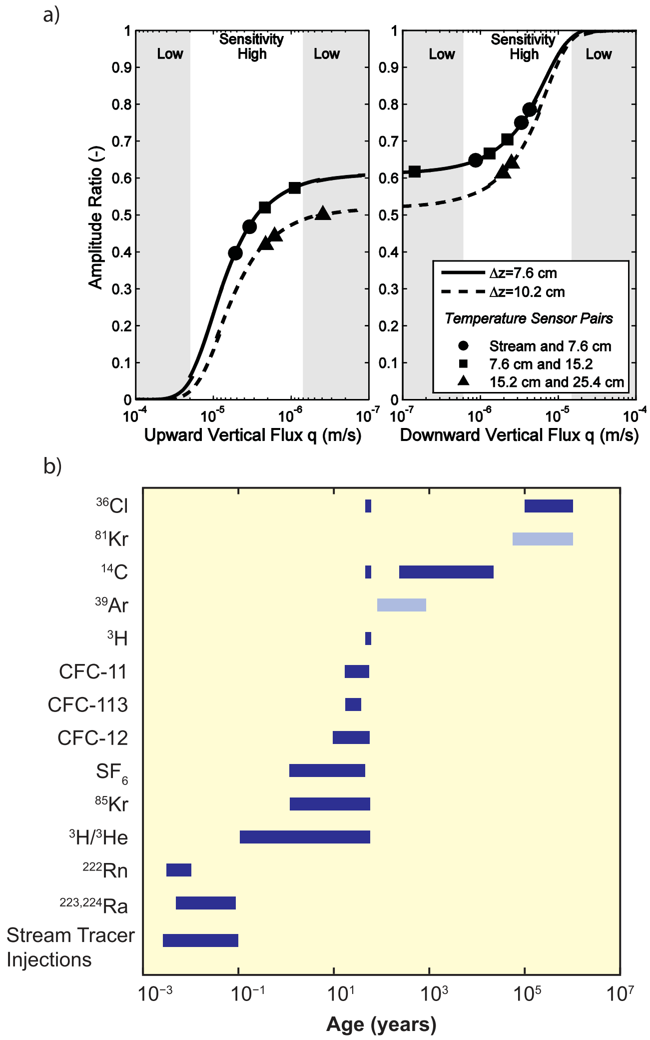

Figure 7. The window of detection for hydrologic transport as illustrated by approximate sensitivity time scales for various measurement approaches, (a) temperature tracing in streambed (from *Bhaskar et al*. [2012] used with permission) and (b) environmental solute tracing (modified from *Cook and Herczeg* [2000]).

mapped. *Stoertz and Bradbury* [1989] showed that apparent recharge and discharge through shallow groundwater into streams is affected by the resolution at which hydraulic head is measured. Higher fluxes and greater accuracy are associated with greater resolution of head measurements. There are practical limits, however, to the number of wells that can be emplaced in a given area, especially for larger-scale investigations. Another example comes from measuring vertical fluxes of water across streambeds by tracing diel fluctuations of heat into the bed. Heat tracing only has sensitivity to detect fluxes within a certain range, with fluxes too low or too high being undetectable (Figure 7a). The range of detection is determined by an interaction between the transport time scale of water though the bed, the depth and spacing of sensors, and the time scale of the tracer signal (often set by diel fluctuations of river temperature).

Following from above, the sensitivity of solute tracer experiments in rivers is also limited. The range of detection is determined by the time scale of the tracer injection and observation time of tracer movement, both of which affect the observable time scales of exchange and storage processes. Referred to as the "window of detection" [*Harvey and Wagner*, 2000], the range and sensitivity of exchange fluxes estimated by stream tracers is influenced by river size, average velocity, as well as the investigator's choice of experimental reach length (which affects observation time and opportunity for tracers to experience all possible storage environments). The influence of those factors is expressed in an experimental Damköhler number [*Wagner and Harvey*, 1997], which becomes a useful metric for tracer test sensitivity. There are additional practical factors that may influence tracer experimental outcomes, including practical limitations of labor costs and tracer costs which are influenced by the desired duration of the tracer injection and the desired dynamic range of the injected tracer [*Drummond et al*., 2012].

The upshot for stream tracer tests is that seemingly arbitrary choices of tracer injection time and the length of the experimental reach can interact with the actual exchange characteristics in a stream reach in a way that limits the spatial and temporal scale of exchange flows that can be detected [*Ward et al*., 2013; *Harvey and Wagner*, 2000]. It has been estimated that the range of detection of exchange flows is constrained between the tracer transport time scale through the experimental reach and the time scale of the injection [*Schmadel et al*., 2014]. As a consequence, exchange zones with relatively long residence times generally will not be observable and instead will be hidden in the noise in low-sensitivity parts of the tracer data where uncertainty in model fit is high [*Harvey et al*., 1996].

To some extent, the problems with tracer test bias can be overcome using principles of experimental design [*Harvey and Wagner*, 2000]. This means that experiments can be refined to answer important questions, e.g., separating surface and subsurface contributions to total exchange flows [*Gooseff et al*., 2005], or testing the effectiveness of constructing surface water storage zones to increase exchange flows [*Ensign and Doyle*, 2005]. However, a negative outcome of limited sensitivity of stream tracers is that there can be little confidence that model parameters determined for river reaches at the kilometer scale can easily be upscaled for use in larger basins [*Gooseff et al*., 2013].

In summary, measurement biases are often not considered in field studies of hydrologic exchange flows. When an investigator selects a field method, no matter whether it is a hydrometric or tracer-based technique, the sensitivity to detect hydrologic exchange is limited to only a small part of the full continuum of fluxes that may be occurring. As a result many investigators are attempting to broaden detection by combining several approaches, each better suited for a particular time scale or spatial scale. This approach adds expense and logistical complications, but pays back in the greater understanding of relative contributions from different types of hydrologic exchange fluxes and how they contribute to ecologically meaningful outcomes such as stream metabolism [*González-Pinzón et al*., 2014].

New methods are evolving to meet the challenges of locating and measuring stream-groundwater exchanges using quick and innovative assessments that discern spatial and temporal patterns of exchange along stream networks, such as *Lowry et al*.'s [2007] use of distributed temperature sensing, *Ward et al*.'s [2010, 2012] and *Briggs et al*.'s [2014] use of geophysical resistivity, and *González-Pinzón et al*.'s [2014] use of multiple scales of measurements, including solute tracers and temperature sensing at both reach and geomorphic unit scales. *Ward et al*.'s [2010, 2012] work applied electrical resistivity methods to tracing salt-tracer-labeled stream water through hyporheic zones. Results provided the opportunity to visualize in 2-D and 3 dimensions the extent and intensity of stream water penetration into the valley corridor sediments. *Covino et al*. [2011] combined models to predict inflows to streams and stream tracer injections over reaches >1000 m in length provided an opportunity to assess the distributed downstream legacies of water that enters the channel along any portion of the river network. Leveraging these and other new techniques with novel biogeochemical methods (e.g., smart tracers for stream metabolism [*Haggerty et al*., 2008, 2009] and eDNA tracers [*Deiner and Altermatt*, 2014]) will provide new understanding of the integral linkage between stream channels and associated aquatic processes and their surrounding valley's in large basins.

7. Connecting Controlling Processes With Cumulative Effects

Understanding hydrologic connectivity requires attention to both process and scale, with a principal challenge being to link fine-scale causal factors with large-scale consequences. Whereas the controls must be understood at fine scale of the hydro and geomorphic drivers, the consequences of hydrologic exchange for downstream water quality and ecology manifest themselves at much larger scales of entire drainage basins and ecoregions. Some scales of inquiry are best suited toward identifying the fundamental processes controlling material sources, biogeochemical reactions, habitat availability, etc., whereas others are best suited for assessing the cumulative effects on water quality and ecological function in large watersheds.

The disparity in scales at which the research is conducted creates a challenge, because without a process-based understanding at small scales there can be no predictions of cumulative effects at larger ones. Thus, it would be difficult to forecast changes in river water quality and ecological health in river networks responding to land use changes or climate change with confidence. Traditional hydrologic models cannot model both the processes and their cumulative effects in entire river networks. Therefore, developing new models that are faithful to the physics but that integrate hydrologic exchange flows and their effects on river transport, water quality, and aquatic ecology throughout river networks is a principal objective.

The contribution of river catchments to global nutrient cycles may has been largely underestimated, as pointed out by *Battin et al*. [2008] and *Ensign and Doyle* [2006] in their respective meta analyses of organic carbon and nutrient transformation in fluvial networks. After evaluating more than 300 studies, *Battin et al*. [2008] concluded that the high rates of organic C respiration in rivers is caused by close contact between river water and heterotrophic microorganisms on sediments, under conditions where even relatively recalcitrant carbon can be processed. *Ensign and Doyle* [2006] were more equivocal, leaving as an open question in which subenvironments of rivers nutrient transformations occur, but suggesting that headwater streams do not always exhibit the largest capacity to retain nutrients, and suggesting that larger rivers may be equally important.

Specifying which hydrologic exchange zones are most important for carbon, nutrient, and contaminant transformations is a significant challenge. The hydrologic exchange fluxes, storage-zone sizes, and residence times vary enormously from 1 m to the next and are not directly measureable at large spatial scales. Biogeochemical reactions in shallow hyporheic flow paths through streambeds are generally found to be more significant than in deeper hyporheic flow paths [*Briggs et al*., 2013a; *Harvey et al*., 2013]. However, the relative importance of microbial reactions occurring in hyporheic zones compared with near-stream riparian areas [*Wollheim et al*., 2014; *Roley et al*., 2012a], and lateral hyporheic flow paths through gravel bars and meanders [*Pinay et al*., 2009; *Zarnetske et al*., 2011], or in deeper mixing zones where groundwater discharge first encounters hyporheic flow paths [*Kennedy et al*., 2009] is not well understood.

7.1. Including Hydrologic Exchange in River Network Water Quality Models

Many models of water quality in river networks track the fate of N-based fertilizers and evaluate in-stream removal processes affecting nitrogen delivery to coastal waters [*Alexander et al*., 2000]. Typically an empirical approach is used that assumes a proportionality between the rate of removal and the river volume, which is supported by data showing an inverse relationship between a reaction rate constant k and the river depth h. Usually, the universe relationship is interpreted to mean that, despite their size, small streams account for most removal of nitrogen in the stream network. The reasoning is that the shallower depth of small streams keeps surface water in closer contact with reactive bed sediments compared to larger streams and rivers. Small streams have an efficient ratio of streambed surface area to stream volume, and they dominate in stream length within watersheds [*Alexander et al*., 2000], but nonetheless there is evidence that larger rivers are also efficient processors of nitrogen [*Wollheim et al*., 2006; *Ensign and Doyle*, 2006]. Seasonally varying flow conditions are also important, as shown by *Basu et al*. [2011] and *Böhlke et al*. [2009]. However, there are also complex and unexplained seasonal influences on nitrogen removal in river networks. In summary, patterns of regional water quality in river networks have been identified but not fully explained. More understanding is needed of the contribution made by hydrologic exchange flows in storing waters in contact with biogeochemical reaction hot spots [*Raymond et al.,* 2015; *McClain et al*., 2003].

The role of hydrologic exchange fluxes is not explicitly considered in most river network models of water quality. How do the effects of these small-scale reaction zones accumulate in river networks? *Findlay* [1995] hypothesized that hydrogeomorphic factors that drive hydrologic exchange flows, and that control the amount of river water contact with reactive sediments, vary more than the intrinsic rates of the biogeochemical reactions. If true, different basins with contrasting types of hydrologic exchange flows would be expected to produce vastly different outcomes for water quality if all other factors were equal. *Harvey et al*. [2013] and *Harvey and Fuller* [1998] formalized the relationship between hydrologic and biogeochemical factors that determine downstream water quality in a Reaction Significance Factor (*RSF)*. *RSF* is dimensionless index that quantifies the cumulative trade-offs between flux, residence time, and intrinsic reaction rate in a given type of hydrologic exchange zone and its influence on downstream water quality. The index

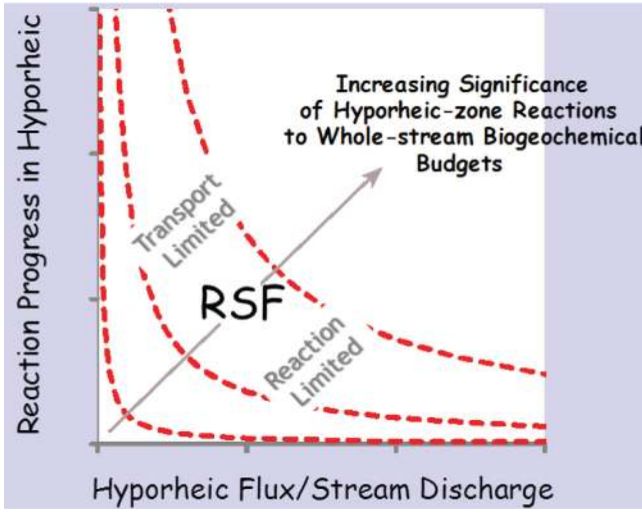

Figure 8. The cumulative effects of biogeochemical reactions in river corridors may be reaction limited, if the exchange is rapid relative to the reaction time scale and turnover of river water through exchange flow paths is fast. Alternatively, if the exchange time scale is slow relative to the reaction rate, the reaction is completed long before the water returns from the exchange zone to the river, for which reaction is transport limited and processing of river water through exchange zones is inefficient. These combined effects are expressed by a dimensionless reaction significance factor, RSF, where isolines have increasing values of RSF toward the upper right denoting the fraction of the reactant removed per characteristic (dimensionless) distance traveled in the stream (modified from *Harvey et al*. [2013]).

works by combining the effects of hydrology and biogeochemistry at both the local scale and also cumulatively at the river basin scale. The local factors are represented by the residence time and intrinsic reaction rate in an individual pathway of exchange (*y* axis in Figure 8), and large-scale controls are represented by the extent of river water turnover through all the similar exchange zones over a given distance of river transport (*x* axis in Figure 8). The resulting isolines in Figure 8 have increasing values of RSF toward the upper right, with values denoting the fraction of the reactant removed per characteristic (dimensionless) distance traveled in the stream.

RSF is a useful metric for specifying why some types of hydrologic exchange zones are more effective biogeochemical reactors than others in terms of their cumulative effects on downstream water quality. For example, *Gomez-Velez and Harvey* [2014] used *RSF* to conclude that, in most large river networks, vertical hyporheic exchange flow beneath river bedforms should be more efficient in enhancing denitrification compared with lateral hyporheic flow paths through gravel bars and meanders. The reason is that vertical exchange through river beds has a more favorable balance between the typical intrinsic time scale of denitrification and the residence time of hyporheic flow, which makes leads those exchange zones efficient processors for removing N from river water compared with less efficient processing in lateral hyporheic flow paths where too little river water is exchanged. Similar reasoning could help quantify hot spots for biogeochemical reactions [e.g., *McClain et al*., 2003] in a variety of river environments including aquatic respiration of organic matter and denitrification within algal layers on and just beneath the streambed surface [*Haggerty et al*., 2009; *Gooseff et al*., 2004] or in shallow streambed hyporheic zones where denitrification occurs [*Harvey et al*., 2013], and where metal contaminants from mining sites are oxidized and sorbed [*Harvey and Fuller*, 1998]. *Roley et al*. [2012a] and *Wollheim et al*. [2014] and *Jones et al*. [2014] and *Scott et al*. [2014] explored similar concepts controlling N removal in riparian zones and floodplains, respectively, and those studies are beginning to reveal the cumulative effects of reactive transformations where rising waters expand hydrologic exchange and increase contact with the highly reactive sediments along the margins of river corridors.

8. Relation to Hydrologic Management, Aquatic Ecosystem Function, and Societal Value

The form and function of river corridors, and their future state, will be determined by many combined interacting processes. Management of water resources is accelerating changes in hydrologic connectivity of river corridors through the dramatic effects imposed by agriculture, animal grazing in riparian areas, and urbanization effects that alter carbon and nutrient inputs and erosion patterns and sediment supply, as well as impose flow restrictions through dam regulation that move away from naturally varying flows. Predicting future states for water quality or ecology in river corridors, and how to target improvements through prioritizing management actions or restoration is still in its infancy. A simple example emphasizes interactions between stream overfertilization with nutrients that supports overproduction of periphyton and algae and movement of fine decaying organic particles that increases oxygen uptake and aerobic decomposition of organic matter in sediments. Such a simple process has far reaching effects on both sides of the surface-subsurface interface, causing anaerobic conditions in the near-river subsurface that leads to reductive dissolution of manganese and iron oxyhydroxides that release sorbed trace metals such as cadmium that can enter the drinking water supply when pumping occurs in alluvial aquifers. An unexpected consequence of removing phosphorus from detergent nearly forty years ago was a reversal of overfertilization in some rivers that reversed anaerobic conditions, returning the geochemical power of oxyhydroxide coatings on alluvial sediments to sorb trace metals and improve water quality [*Von Gunten and Lienert*, 1993]. Another example is the sealing of streambeds, termed streambed colmation, which can become a major problem where the positive feedbacks described above increasingly isolate the wetted channel from its streambed sediments [*Brunke*, 1999; *Velickovic*, 2005]. This and other geo-bio-morphodynamic processes may ultimately determine the trajectory of form and function in river corridors as they respond to land use and climate changes.

These of course are only the briefest of examples. Nevertheless, the picture that emerges is one of a dynamic ecosystem in which physical and biological components interact, adjust, and coevolve with one another. There is practical importance of the research in helping a reevaluation of what may be the single most important piece of legislation ever for the environment. The Clean Water Act (CWA) of the United States is applicable to Waters of the United States, which has had a range of definitions since the law's inception in 1972. Water bodies covered under the CWA are generally known to be navigable waters and their tributaries. Given the greater understanding of the connections among water bodies, mostly facilitated via groundwater flow paths, the U.S. Environmental Protection Agency is embracing a greater appreciation of connectivity in determining if a water body is indeed covered by the CWA [*Department of Defense and Environmental Protection Agency*, 2014].

This refinement of CWA jurisdiction is largely driven by the recent Rapanos v. United States [*Rapanos v. United States*, 2006] decision, after which the U.S. Army Corps of Engineers and EPA issued new guidance for the regulation of the waters of the United States [*Department of Defense and Environmental Protection Agency*, 2014]. In 2006, Supreme Court Justice Kennedy, in one of three plurality opinions, described a significant nexus requirement for Clean Water Act jurisdiction, whereby, a nexus exists if a connection with a wetland or other type of marginal water, either by itself or with other similar connections, significantly affects the physical, biological, and chemical integrity of the downstream navigable waterway. However, the divided opinion of the Supreme Court created a situation without a standard and practical test of significant nexus, which has proven to be a burden to landowners and regulators alike. Meanwhile there is greater recognition that the physical, chemical, and biological integrities of rivers, in particular, are indeed dependent upon processes that occur beyond the channel margins; that the river corridor functions as an integrated system including both terrestrial and aquatic domains. In particular, the near-stream environment, including hyporheic zones, riparian zones, and wetlands, may come to be regulated under the Clean Water Act due to their clear relationship with the physical, chemical, and biological integrity of traditionally regulated waters. There remain challenges to explicit measures that define connectivity among water bodies, though it is reasonable to consider that most water bodies are all connected, whether by biological movement (e.g., waterfowl, amphibian use of different water bodies for different life stages), or hydrologic connection (e.g., groundwater flow paths), that could be described by a gradient approach with a relative scale of the strength of the connection until improved methods are available [e.g., *Larsen et al*., 2012].

A fully functioning river corridor that has intact riparian habitats, river and floodplain morphologies, with enough complexity to host an array of river exchanges with marginal surface and subsurface waters is important to providing ecosystem services, especially good water quality. We tend to focus on the aesthetics of river corridors to judge whether it is impaired or not. However, even an aesthetically pleasing river corridor may not host all of the hydrologic exchange flows that underpin critical ecosystem functions and services. The most important exchanges are not always happening, and understanding what controls their intermittency is important. Other exchanges may be going on all of the time but may be invisible to casual inspection and therefore require targeted study. However, the converse statement is also true – river exchanges with marginal surface and subsurface waters alone are not likely to be the sole determinant or salvation of river corridor function. However, their underpinning across these ecosystems and ecotones is critical to many of the important ecosystem and water quality functions and services described above.

9. Summary and Prospectus

Rivers are a small fraction of the landscape, yet because of the connections beyond the main channel, they transform substantial amounts of solutes and energy-rich materials at rates that are disproportionately high relative to their aerial extent. The science of river corridors has advanced significantly over the past 50 years and has seen a broadening of views beyond the channel margins. Streams and rivers no longer are viewed as drains and transport is no longer modeled as if it occurred simply in rectangular or trapezoidal channels with boundaries that are impermeable and that are never overtopped. There have been tremendous strides in measuring and modeling hydrologic exchange flows of all types, including surface exchange with deep pools and marginal side cavities and hyporheic exchange, as well as the biogeochemical implications of expanding contact with hyporheic, riparian, and floodplain sediments during floods. Most studies have focused on small scales revealing the processes controlling hydrologic exchange flows at the scale of individual geomorphic units, e.g., sand ripples, gravel bar, pool and riffle, wood debris features, crevasse splays, and floodplain flow paths, etc. Simple and useful physically based equations have been advanced within the pages of *WRR* and verified against detailed field measurements at all of those scales, often showing order of magnitude accuracy or better in predictions based on a few key physical measurements.

The end users of river corridor science ultimately need predictions at watershed scales rather than at the scale of individual geomorphic units or short river reaches. Yet most of the empirical observations are from reach-scale tracer tests or studies on the scale of individual geomorphic units. As a consequence, little is known about how hydrologic exchange fluxes scale in river networks. Some theoretical and experimental analyses indicate power law scaling [*Wörman et al*., 2007; *Haggerty et al*., 2002], while a broad analysis of tracer experimental observations [*González-Pinzón et al*., 2013] suggests that the extent of delayed transport caused by hydrologic exchange fluxes is persistent through river reaches, and does not grow or shrink appreciably with river size or transport time. The empirical tracer results may be consistent with physically based modeling of transport through river networks [*Bellin et al*., 2015; *Gomez-Velez and Harvey*, 2014], suggesting dominance of hydrologic exchange flows by smaller geomorphic features. Predictive modeling of hydrologic exchange flows at the regional scale is only just beginning however. Scaling hydrologic exchange fluxes for use in predictive water quality models in large basins is sure to be an important forefront for research.

At the river network scale the key physical measurements are often sparse or lacking, including channel geometry, grain size, bank height, and bedform types and sizes. How to build scaling relationships from sparse data for analyzing transport in large basins is increasingly a subject of debate. A starting point is available in decades of work to specify channel hydraulic geometry relations. To be sure, data sets for hydraulic geometry analysis are expanding and being integrated with data sets on channel sinuosity, grain size, and dye tracer tests, and analysis is underway to integrate estimates across base flow to bankfull conditions, and flooding, in large basins. Of course prediction is more difficult at larger scales because exchanges with multiple scales of geomorphic units interact to affect transport and water quality, and on the ground-based observations are limited in large basins. *Merck et al*. [2012], *McKean et al*. [2014], and *Marzadri et al*. [2014] offer important examples of large-scale estimation of exchange flows using combinations of ground-based and airborne observations. Hydrologists will increasingly need to rely on airborne thermal imaging and satellite data to answer river network-scale questions involving exchange flows [*Dugdale et al*., 2015; *Alsdorf et al*., 2007]. Airborne and satellite hydrology offers abundant opportunities to not only estimate discharge from space, but also better constrain the channel conditions that trigger hydrologic exchanges with off-channel environments.

Both aquatic and terrestrial scientists have come to appreciate how lateral exchanges of water affect movement of material and energy between rivers and floodplains. Many of the hydrologic advances in the understanding of exchange flows and related implications for solute transport and fate have been extensively published in *Water Resources Research*. We fully expect that existing conceptual models of river corridors will continue to evolve, and thereby provide new opportunities for developing insightful numerical models and predictions of the controls on locations and times of rapid biogeochemical cycling—i.e., hot spots and hot moments. These advances will continue to better inform management and policy decisions regarding the dynamic equilibrium of river corridors, for the benefit of resident ecosystems and ecosystem services provided. The need for better hydrogeomorphic information and modeling of hydrologic exchange processes is keenly felt, from evaluating the effectiveness of river and watershed management practices all the way to clarifying regulatory authority under the Clean Water Act.

Acknowledgment
The authors thank editor Scott Mackay and reviewers Erich Hester, Robert Runkel, and an anonymous reviewer for their helpful comments. Support is acknowledged from the USGS NAWQA program and John Wesley Powell Center for Analysis and Synthesis (JH) and NSF grant EAR 0911435 (MG).

References

Alexander, R. B., R. A. Smith, and G. E. Schwarz (2000), Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico, *Nature*, *403*, 758–761.

Alexander, R. B., E. W. Boyer, R. A. Smith, G. E. Schwarz, and R. B. Moore (2007), The role of headwater streams in downstream water quality, *J. Am. Water Resour. Assoc.*, *43*, 41–59, doi:10.1111/j.1752-1688.2007.00005.x.

Alsdorf, D. E., E. Rodríguez, and D. P. Lettenmaier (2007), Measuring surface water from space, *Rev. Geophys.*, *45*, RG2002, doi:10.1029/2006RG000197.

Anderson, M. P. (2008), *Groundwater: Selection, Introduction, and Commentary by Mary P. Anderson, Benchmark Pap. Hydrol.*, Int. Assoc. of Hydrol. Sci., Oxfordshire.

Argerich, A., R. Haggerty, E. Martí, F. Sabater, and J. Zarnetske (2011), Quantification of metabolically active transient storage (MATS) in two reaches with contrasting transient storage and ecosystem respiration, *J. Geophys. Res.*, *116*, G03034, doi:10.1029/2010JG001379.

Aubeneau, A. F., R. L. Martin, D. Bolster, R. Schumer, D. Jerolmack, and A. Packman (2015), Fractal patterns in riverbed morphology produce fractal scaling of water storage times, *Geophys. Res. Lett.*, *42*, 5309–5315, doi:10.1002/2015GL064155.

Basu, N. B., S. E. Thompson, and P. S. C. Rao (2011), Hydrologic and biogeochemical functioning of intensively managed catchments: A synthesis of top-down analyses, *Water Resour. Res.*, *47*, W00J15, doi:10.1029/2011WR010800.

Bates, P. D., M. S. Horritt, C. N. Smith, and D. Mason (1997), Integrating remote sensing observations of flood hydrology and hydraulic modelling, *Hydrol. Processes*, *11*, 1777–1795, doi:10.1002/(SICI)1099-1085(199711)11:14<1777::AID-HYP543>3.0.CO;2-E.

Bates, P. D., M. Horritt, and J.-M. Hervouet (1998), Investigating two-dimensional, finite element predictions of floodplain inundation using fractal generated topography, *Hydrol. Processes*, *12*, 1257–1277, doi:10.1002/(SICI)1099-1085(19980630)12:8<1257::AID-HYP672>3.0.CO;2-P.

Battin, T. J. (2000), Hydrodynamics is a major determinant of streambed biofilm activity: From the sediment to the reach scale, *Limnol. Oceanogr.*, *45*(6), 1308–1319.

Battin, T. J., and D. Sengschmitt (1999), Linking sediment biofilms, hydrodynamics and river bed clogging: Evidence from a larger river, *Microbial Ecol.*, *37*, 185–196.

Battin, T. J., L. A. Kaplan, S. Findlay, C. S. Hopkinson, E. Marti, A. I. Packman, J. D. Newbold, and F. Sabater (2008), Biophysical controls on organic carbon fluxes in fluvial networks, *Nat. Geosci.*, *1*(2), 95–100.

Baxter, C. V., and F. R. Hauer (2000), Geomorphology, hyporheic exchange, and selection of spawning habitat by bull trout (Salvelinus confluentus), *Can. J. Fish. Aquat. Sci.*, *57*, 1470–1481.

Beaulieu, J. J., et al. (2011), Nitrous oxide emission from denitrification in stream and river networks, *Proc. Natl. Acad. Sci. U. S. A.*, *108*(1), 214–219, doi:10.1073/pnas.1011464108.

Bellin, A., D. Tonina, and A. Marzadri (2015), Breakthrough curve moments scaling in hyporheic exchange, *Water Resour. Res.*, *51*, 1353–1358, doi:10.1002/2014WR016559.

Bencala, K. E. (1983), Simulation of solute transport in a mountain Poop-and-riffle stream with a kinetic mass transfer model for sorption, *Water Resour. Res.*, *19*(3), 732–738.

Bencala, K. E. (1993), A perspective on stream-catchment connections, *J. North Am. Benthol. Soc.*, *12*, 44–47.

Bencala, K. E., and R. Walters (1983), Simulation of solute transport in a mountain pool-and-riffe stream: A transient storage model, *Water Resour. Res.*, *19*(3), 718–724.

Bencala, K. E., G. W. Zellweger, A. P. Jackman, and R. J. Avanzino (1984), Interactions of solutes and streambed sediment: 1. An experimental analysis of cation and anion transport in a mountain stream, *Water Resour. Res.*, *20*(12), 1797–1803.

Bernhardt, E. S., et al. (2005), Synthesizing U. S. river restoration efforts, *Science*, *308*(5722), 636–637.

Bhaskar, A. S., J. W. Harvey, and E. J. Henry (2012), Resolving hyporheic and groundwater components of streambed water flux using heat as a tracer, *Water Resour. Res.*, *48*, W08524, doi:10.1029/2011WR011784.

Boano, F., R. Revelli, and L. Ridolfi (2013), Modeling hyporheic exchange with unsteady stream discharge and bedform dynamics, *Water Resour. Res.*, *49*, 4089–4099, doi:10.1002/wrcr.20322.

Boano, F., J. W. Harvey, A. Marion, A. I. Packman, R. Revelli, L. Ridolfi, and A. Wörman (2014), Hyporheic flow and transport processes: Mechanisms, models, and biogeochemical implications, *Rev. Geophys.*, *52*, 603–679, doi:10.1002/2012RG000417.

Böhlke, J. K., K. Révész, E. Busenberg, J. Deak, E. Deseo, and M. Stute (1997), A ground-water record of halocarbon transport by the Danube River, *Environ. Sci. Technol.*, *31*, 3293–3299.

Böhlke, J. K., R. C. Antweiler, J. W. Harvey, A. E. Laursen, L. K. Smith, R. L. Smith, and M. A. Voytek (2009), Multi-scale measurements and modeling of denitrification in streams with varying flow and nitrate concentration in the upper Mississippi River basin, USA, *Biogeochemistry*, *93*, 117–141, doi:10.1007/s10533-008-9282-8.

Bottacin-Busolin, A., A. Marion, T. Musner, M. Tregnaghi, and M. Zaramella (2011), Evidence of distinct contaminant transport patterns in rivers using tracer tests and a multiple domain retention model, *Adv. Water Resour.*, *34*, 737–746.

Bourg, A. C. M., and C. Bertin (1993), Biogeochemical processes during the infiltration of river water into an alluvial aquifer, *Environ. Sci. Technol.*, *27*(4), 661–666.

Bourg, A. C. M., and C. Bertin (1994), Seasonal and spatial trends in manganese solubility in an alluvial aquifer, *Environ. Sci. Technol.*, *28*, 868–876.

Bridge, J. S. (2009), *Rivers and Floodplains: Forms, Processes, and Sedimentary Record*, John Wiley, Oxford.

Briggs, M. A., M. N. Gooseff, C. D. Arp, and M. A. Baker (2009), A method for estimating surface transient storage parameters for streams with concurrent hyporheic storage, *Water Resour. Res.*, *45*, W00D27, doi:10.1029/2008WR006959.

Briggs, M. A., L. K. Lautz, J. M. McKenzie, R. P. Gordon, and D. Hare (2012), Using high-resolution distributed temperature sensing to quantify spatial and temporal variability in vertical hyporheic flux, *Water Resour. Res.*, *48*, W02527, doi:10.1029/2011WR011227.

Briggs, M. A., L. K. Lautz, D. H. Hare, and R. González-Pinzón (2013a), Relating hyporheic fluxes, residence times and redox-sensitive biogeochemical processes upstream of beaver dams, *Freshwater Sci.*, *32*(2), 622–641, doi:10.1899/12-110.1.

Briggs, M. A., E. B. Voytek, F. D. Day-Lewis, D. O. Rosenberry, and J. W. Lane (2013b), Understanding water column and streambed thermal refugia for endangered mussels in the Deleware River, *Environ. Sci. Technol.*, *47*, 11,423–11,431, doi:10.1021/es4018893.

Briggs, M. A., M. A. Walvoord, J. M. McKenzie, C. I. Voss, F. D. Day-Lewis, and J. W. Lane (2014), New permafrost is forming around shrinking Arctic lakes, but will it last?, *Geophys. Res. Lett.*, *41*, 1585–1592, doi:10.1002/2014GL059251.

Briggs, M. A., F. D. Day-Lewis, J. P. Zarnetske, and J. W. Harvey (2015), A physical explanation for the development of redox microzones in hyporheic flow, *Geophys. Res. Lett.*, *42*, 4402–4410, doi:10.1002/2015GL064200.

Broshears, R. E., R. L. Runkel, B. A. Kimball, D. M. McKnight, and K. E. Bencala (1996), Reactive solute transport in an acidic stream: Experimental pH increase and simulation of controls on pH, aluminum, and iron, *Environ. Sci. Technol.*, *30*(10), 3016–3024.

Brunke, M. (1999), Colmation and depth filtration within streambeds: Retention of particles in hyporheic interstices, *Int. Rev. Hydrogeol.*, *84*, 99–117.

Brunke, M., and T. Gonser (1997), The ecological significance of exchange processes between rivers and groundwater, *Freshwater Biol.*, *37*, 1–33.

Butman, D., and P. A. Raymond (2011), Significant efflux of carbon dioxide from streams and rivers in the United States, *Nat. Geosci.*, *4*(12), 839–842, doi:10.1038/ngeo1294.

Cardenas, M. B. (2015), Hyporheic zone hydrologic science: A historical account of its emergence and a prospectus, *Water Resour. Res.*, *51*, 3601–3616, doi:10.1002/2015WR017028.

Cardenas, M. B., and J. L. Wilson (2007), Dunes, turbulent eddies, and interfacial exchange with permeable sediments, *Water Resour. Res.*, *43*, W08412, doi:10.1029/2006WR005787.

Cardenas, M. B., J. L. Wilson, and V. A. Zlotnik (2004), Impact of heterogeneity, bedforms, and stream curvature on subchannel hyporheic exchange, *Water Resour. Res.*, *40*, W08307, doi:10.1029/2004WR003008.

Castro, N. M., and G. M. Hornberger (1991), Surface-subsurface water interactions in an alluviated mountain stream channel, *Water Resour. Res.*, *27*(7), 1613–1621, doi:10.1029/91WR00764.

Chanson, H., N. Trevethan, and S. Aoki (2008), Acoustic Doppler Velocimetry (ADV) in small estuary: Field experience and signal post-processing, *Flow Meas. Instrum.*, *19*(5), 307–313, doi:10.1016/j.flowmeasinst.2008.03.003.

Chen, X. H. (2001), Migration of induced-infiltrated stream water into nearby aquifers due to seasonal ground water withdrawal, *Ground Water*, *39*(5), 721–728.

Choi, J., and J. W. Harvey (2000), Quantifying time-varying groundwater discharge and recharge in wetlands: A comparison of methods in the Florida Everglades, *Wetlands*, *20*(3), 500–511.

Choi, J., S. M. Hulseapple, M. H. Conklin, and J. W. Harvey (1998), Modeling CO_2 degassing and pH in a stream-aquifer system, *J. Hydrol.*, *209*, 297–310.

Choi, J., J. W. Harvey, and M. H. Conklin (2000), Characterizing multiple timescales of stream and storage zone interaction that affect solute fate and transport in streams, *Water Resour. Res.*, *36*(6), 1511–1518, doi:10.1029/2000WR900051.

Cirpka, O. A., M. N. Fienen, M. Hofer, E. Hoehn, A. Tessarini, R. Kipfer, and P. K. Kitanidis (2007), Analyzing bank filtration by deconvoluting time series of electric conductivity, *Ground Water*, *45*, 318–328, doi:10.1111/j.1745-6584.2006.00293.x.

Cohen, S., A. J. Kettner, and J. Syvitski (2014), Global suspended sediment and water discharge dynamics between 1960 and 2010: Continental trends and intra-basin sensitivity, *Global Planet. Change*, *115*, 44–58, doi:10.1016/j.gloplacha.2014.01.011.

Conant, B., Jr., J. A. Cherry, and R. W. Gillham (2004), A PCE groundwater plume discharging to a river: Influence of the streambed and near-river zone on contaminant distributions, *J. Contam. Hydrol.*, *73*(1–4), 249–279, doi:10.1016/j.jconhyd.2004.04.001.

Cook, P. G., and A. L. Herczeg (2000), *Environmental Tracers in Subsurface Hydrology*, Springer, N. Y.

Cook, P. G., S. Lamontagne, D. Berhane, and J. F. Clark (2006), Quantifying groundwater discharge to Cockburn River, southeastern Australia, using dissolved gas tracers 222Rn and SF6, *Water Resour. Res.*, *42*, W10411, doi:10.1029/2006WR004921.

Covino, T., B. McGlynn, and J. Mallard (2011), Stream-groundwater exchange and hydrologic turnover at the network scale, *Water Resour. Res.*, *47*, W12521, doi:10.1029/2011WR010942.

Cunge, J. A. (1969), On the subject of a flood propagation computational method (Muskingum method), *J. Hydraul. Res.*, *7*(2), 205–230.

Dahm, C. N., N. B. Grimm, P. Marmonier, H. M. Valett, and P. Vervier (1998), Nutrient dynamics at the interface between surface waters and groundwaters, *Freshwater Biol.*, *40*(3), 427–451, doi:10.1046/j.1365-2427.1998.00367.x.

DeAngelis, D. L., M. Loreau, D. Neegaard, P. J. Mulholland, and E. R. Marzolf (1995), Modeling nutrient-periphyton dynamics in streams—The importance of transient storage zones, *Ecol. Modell.*, *80*(2–3), 149–160.

Deiner, K., and F. Altermatt (2014), Transport distance of invertebrate environmental DNA in a natural river, *PLOS One*, *9*(2), e88786.

Department of Defense and Environmental Protection Agency (2014), Definition of "Waters of the United States" under the Clean Water Act, *Fed. Regist.*, *79*(76), 22,188–22,247.

Doyle, M. W., E. H. Stanley, D. L. Strayer, R. B. Jacobson, and J. C. Schmidt (2005), Effective discharge analysis of ecological processes in streams, *Water Resour. Res.*, *41*, W11411, doi:10.1029/2005WR004222.

Drummond, J. D., T. P. Covino, A. F. Aubeneau, D. Leong, S. Patil, R. Schumer, and A. I. Packman (2012), Effects of solute breakthrough curve tail truncation on residence time estimates: A synthesis of solute tracer injection studies, *J. Geophys. Res.*, *117*, G00N08, doi:10.1029/2012JG002019.

Dudley-Southern, M., and A. Binley (2015), Temporal responses of groundwater-surface water exchange to successive storm events, *Water Resour. Res.*, *51*, 1112–1126, doi:10.1002/2014WR016623.

Duff, J. H., F. Murphy, C. C. Fuller, F. J. Triska, J. W. Harvey, and A. P. Jackman (1998), A mini drivepoint sampler for measuring pore water solute concentrations in the hyporheic zone of sand-bottom streams, *Limnol. Oceanogr.*, *43*, 1378–1383.

Dugdale, S. J., N. E. Bergeron, and A. St-Hilaire (2015), Spatial distribution of thermal refuges analysed in relation to riverscape hydromorphology using airborne thermal infrared imagery, *Remote Sens. Environ.*, *160*, 43–55.

Dunne, T., L. A. K. Mertes, R. H. Meade, J. E. Richey, and B. R. Forsberg (1998), Exchanges of sediment between the flood plain and channel of the Amazon River in Brazil, *Geol. Soc. Am. Bull.*, *110*(4), 450–467.

Elliott, A. H., and N. H. Brooks (1997a), Transfer of nonsorbing solutes to a streambed with bedforms: Theory, *Water Resour. Res.*, *33*(1), 123–136.

Elliott, A. H., and N. H. Brooks (1997b), Transfer of nonsorbing solutes to a streambed with bedforms: Laboratory experiments, *Water Resour. Res.*, *33*(1), 137–151, doi:10.1029/96WR02783.

Endreny, T., L. Lautz, and D. Siegel (2011), Hyporheic flow path response to hydraulic jumps at river steps: Hydrostatic model simulations, *Water Resour. Res.*, *47*, W02518, doi:10.1029/2010WR010014.

Ensign, S. H., and M. W. Doyle (2005), In-channel transient storage and associated nutrient retention: Evidence from experimental manipulations, *Limnol. Oceanogr. Methods*, *50*(6), 1740–1751.

Ensign, S. H., and M. W. Doyle (2006), Nutrient spiraling in streams and river networks, *J. Geophys. Res.*, *111*, G04009, doi:10.1029/2005JG000114.

Ensign, S. H., M. F. Piehler, and M. W. Doyle (2008), Riparian zone denitrification affects nitrogen flux through a tidal freshwater river, *Biogeochemistry*, *91*(2-3), 133–115.

Fernald, A. G., P. J. Wigington, and D. H. Landers (2001), Transient storage and hyporheic flow along the Willamette River, Oregon: Field measurements and model estimates, *Water Resour. Res.*, *37*(6), 1681–1694, doi:10.1029/2000WR900338.

Fick, J., H. Soderstrom, R. H. Lindberg, C. Phan, M. Tysklind, and D. G. J. Larsson (2009), Contamination of surface, ground, and drinking water from pharmaceutical production, *Environ. Toxicol. Chem.*, *28*(12), 2522–2527.

Findlay, S. (1995), Importance of surface-subsurface exchange in stream ecosystems: The hyporheic zone, *Limnol. Oceanogr.*, *40*(1), 159–164.

Fischer, H., J. Imberger, E. List, R. Koh, and N. Brooks (1979), *Mixing in Inland and Coastal Waters*, Academic, N. Y.

Fischer, H., F. Kloep, S. Wilzcek, and M. Pusch (2005), A river's liver—Microbial processes within the hyporheic zone of a large lowland river, *Biogeochemistry*, *76*(2), 349–371.

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

Fleckenstein, J., M. Anderson, G. Fogg, and J. Mount (2004), Managing surface water-groundwater to restore fall flows in the Cosumnes River, *J. Water Resour. Plann. Manage.*, *130*(4), 301–310.

Forshay, K. J., and E. H. Stanley (2005), Rapid nitrate loss and denitrification in a temperate river floodplain, *Biogeochemistry*, *75*(1), 43–64.

Francis, B. A., L. K. Francis, and M. B. Cardenas (2010), Water table dynamics and groundwater–surface water interaction during filling and draining of a large fluvial island due to dam-induced river stage fluctuations, *Water Resour. Res.*, *46*, W07513, doi:10.1029/2009WR008694.

Gao, J. H., et al. (2014), Changes in water and sediment exchange between the Changjiang River and Poyang Lake under natural and anthropogenic conditions, China, *Sci. Total Environ.*, *481*, 542–553, doi:10.1016/j.scitotenv.2014.02.087.

Gariglio, F. P., D. Tonina, and C. H. Luce (2013), Spatiotemporal variability of hyporheic exchange through a pool-rifflepool sequence, *Water Resour. Res.*, *49*, 7185–7204, doi:10.1002/wrcr.20419.

Geist, D. R., M. C. Joy, D. R. Lee, and T. Gonser (1998), A method for installing piezometers in large cobble bed rivers, *Groundwater Monit. Remediation*, *18*, 78–82, doi:10.1111/j.1745-6592.1998.tb00604.x.

Gerecht, K. E., M. B. Cardenas, A. J. Guswa, A. H. Sawyer, J. D. Nowinski, and T. E. Swanson (2011), Dynamics of hyporheic flow and heat transport across a bed-to-bank continuum in a large regulated river, *Water Resour. Res.*, *47*, W03524, doi:10.1029/2010WR009794.

Glover, R. E., and G. G. Balmer (1954), River depletion resulting from pumping a well near a river, *Trans. AGU*, *35*(3), 468–470.

Gomez-Velez, J. D., and J. W. Harvey (2014), A hydrogeomorphic river network model predicts where and why hyporheic exchange is important in large basins, *Geophys. Res. Lett.*, *41*, 6403–6412, doi:10.1002/2014GL061099.

González-Pinzón, R., R. Haggerty, and M. Dentz (2013), Scaling and predicting solute transport processes in streams, *Water Resour. Res.*, *49*, 4071–4088, doi:10.1002/wrcr.20280.

González-Pinzón, R., A. S. Ward, C. E. Hatch, A. N. Wlostowski, K. Singha, M. N. Gooseff, R. Haggerty, J. W. Harvey, O. A. Cirpka, and J. T. Brock (2014), A field comparison of multiple techniques to quantify groundwater–surface-water interactions, *Freshwater Sci.*, *34*(1), 139–160.

Gooseff, M. N., D. M. McKnight, R. L. Runkel, and B. H. Vaughn (2003), Determining long time-scale hydrologic flow paths in Antarctic streams, *Hydrol. Processes*, *17*(9), 1691–1710.

Gooseff, M. N., D. M. McKnight, R. L. Runkel, and J. H. Duff (2004), Denitrification and hydrologic transient storage in a glacial meltwater stream, McMurdo Dry Valleys, Antarctica, *Limnol. Oceanogr. Methods*, *49*(5), 1884–1895.

Gooseff, M. N., J. LaNier, R. Haggerty, and K. Kokkeler (2005), Determining in-channel (dead zone) transient storage by comparing solute transport in a bedrock channel-alluvial channel sequence, Oregon, *Water Resour. Res.*, *41*, W06014, doi:10.1029/2004WR003513.

Gooseff, M. N., R. A. Payn, J. P. Zarnetske, W. B. Bowden, J. P. McNamara, and J. H. Bradford (2008), Comparison of in-channel mobile-immobile zone exchange during instantaneous and constant-rate stream tracer additions: Implications for design and interpretation of non-conservative tracer experiments, *J. Hydrol.*, *357*, 112–124, doi:10.1016/j.jhydrol.2008.05.006.

Gooseff, M. N., D. A. Benson, M. A. Briggs, M. Weaver, W. Wollheim, B. Peterson, and C. S. Hopkinson (2011), Residence time distributions in surface transient storage zones in streams: Estimation via signal deconvolution, *Water Resour. Res.*, *47*, W05509, doi:10.1029/2010WR009959.

Gooseff, M. N., M. A. Briggs, K. E. Bencala, B. L. McGlynn, and D. T. Scott (2013), Do transient storage parameters directly scale in longer, combined stream reaches? Reach length dependence of transient storage interpretations, *J. Hydrol.*, *483*, 16–25.

Grant, S. B., M. J. Stewardson, and I. Marusic (2012), Effective diffusivity and mass flux across the sediment-water interface in streams, *Water Resour. Res.*, *48*, W05548, doi:10.1029/2011WR011148.

Green, M. B., W. M. Wollheim, N. B. Basu, G. Gettel, P. Suresh Rao, N. Morse, and R. Stewart (2009), Effective denitrification scales predictably with water residence time across diverse systems, *Nat. Precedings*. [Available at http://hdl.handle.net/10101/npre.2009.3520.1.]

Griffin, E. R., J. W. Kean, K. R. Vincent, J. D. Smith, and J. M. Friedman (2005), Modeling effects of bank friction and woody bank vegetation on channel flow and boundary shear stress in the Rio Puerco, New Mexico, *J. Geophys. Res.*, *110*, F04023, doi:10.1029/2005JF000322.

Grimm, N., and S. Fisher (1984), Exchange between interstitial and surfacewater—Implications for stream metabolism and nutrient cycling, *Hydrobiologia*, *111*, 219–228.

Gu, C., W. Anderson, and F. Maggi (2012), Riparian biogeochemical hot moments induced by stream fluctuations, *Water Resour. Res.*, *48*, W09546, doi:10.1029/2011WR011720.

Haggerty, R., S. M. Wondzell, and M. A. Johnson (2002), Power-law residence time distribution in the hyporheic zone of a 2nd-order mountain stream, *Geophys. Res. Lett.*, *29*(13), 1640, doi:10.1029/2002GL014743.

Haggerty, R., A. Argerich, and E. Martí (2008), Development of a ''smart'' tracer for the assessment of microbiological activity and sediment-water interaction in natural waters: The resazurin-resorufin system, *Water Resour. Res.*, *44*, W00D01, doi:10.1029/2007WR006670.

Haggerty, R., E. Martì, A. Argerich, D. von Schiller, and N. B. Grimm (2009), Resazurin as a ''smart'' tracer for quantifying metabolically active transient storage in stream ecosystems, *J. Geophys. Res.*, *114*, G03014, doi:10.1029/2008JG000942.

Hall, R. O., Jr., M. A. Baker, E. J. Rosi-Marshall, J. L. Tank, and J. D. Newbold (2013), Solute-specific scaling of inorganic nitrogen and phosphorus uptake in streams, *Biogeosciences*, *10*, 7323–7331, doi:10.5194/bg-10-7323-2013.

Harvey, J. W. (1993), Measurement of variation in soil solute tracer concentration across a range of effective pore sizes, *Water Resour. Res.*, *29*(6), 1831–1837, doi:10.1029/93WR00529.

Harvey, J. W., and K. E. Bencala (1993), The Effect of streambed topography on surface-subsurface water exchange in mountain catchments, *Water Resour. Res.*, *29*(1), 89–98, doi:10.1029/92WR01960.

Harvey, J. W., and C. C. Fuller (1998), Effect of enhanced manganese oxidation in the hyporheic zone on basin-scale geochemical mass balance, *Water Resour. Res.*, *34*(4), 623–636.

Harvey, J. W., and B. L. Wagner (2000), Quantifying hydrologic interactions between streams and their subsurface hyporheic zones, in *Streams and Ground Waters*, edited by J. B. Jones and P. J. Mulholland, pp. 3–44, Academic Press, San Diego.

Harvey, J. W., R. M. Chambers, and J. R. Hoelscher (1995), Preferential flow and segregation of porewater solutes in wetland sediment, *Estuaries*, *18*(4), 568–578.

Harvey, J. W., B. J. Wagner, and K. E. Bencala (1996), Evaluating the reliability of the stream tracer approach to characterize stream-subsurface water exchange, *Water Resour. Res.*, *32*(8), 2441–2451.

Harvey, J. W., M. H. Conklin, and R. Koelsch (2003), Predicting changes in hydrologic retention in an evolving semi-arid alluvial, *Adv. Water Resour.*, *26*, 939–950.

Harvey, J. W., J. T. Newlin, and J. E. Saiers (2005), Solute transport and storage mechanisms in wetlands of the Everglades, South Florida, *Water Resour. Res.*, *41*, W05009, doi:10.1029/2004WR003507.

Harvey, J. W., J. T. Newlin, and S. L. Krupa (2006), Modeling decadel timescale interactions between surface water and ground water in the central Everglades, Florida, *J. Hydrol.*, *320*(3-4), 400–420, doi:10.1016/j.jhydrol.2005.07.024.

Harvey, J. W., R. W. Schaffranek, G. B. Noe, L. G. Larsen, D. J. Nowacki, and B. L. O'Connor (2009), Hydroecological factors governing surface water flow on a low-gradient floodplain, *Water Resour. Res.*, *45*, W03421, doi:10.1029/2008WR007129.

Harvey, J. W., et al. (2012), Hydrogeomorphology of the hyporheic zone: Stream solute and fine particle interactions with a dynamic streambed, *J. Geophys. Res.*, *117*, G00N11, doi:10.1029/2012JG002043.

Harvey, J. W., J. K. Böhlke, M. A. Voytek, D. Scott, and C. R. Tobias (2013), Hyporheic zone denitrification: Controls on effective reaction depth and contribution to whole-stream mass balance, *Water Resour. Res.*, *49*, 6298–6316, doi:10.1002/wrcr.20492.

Hatch, C. E., A. T. Fisher, J. S. Revenaugh, J. Constantz, and C. Ruehl (2006), Quantifying surface water–groundwater interactions using time series analysis of streambed thermal records: Method development, *Water Resour. Res.*, *42*, W10410, doi:10.1029/2005WR004787.

Heeren, D. M., R. B. Miller, G. A. Fox, D. E. Storm, T. Halihan, and C. J. Penn (2010), Preferential flow effects on subsurface contaminant transport in alluvial floodplains, *Trans. Am. Assoc. Agric. Biol. Eng.*, *53*(1), 127–136.

Heeren, D. M., G. A. Fox, A. K. Fox, D. E. Storm, R. B. Miller, and A. R. Mittelstet (2014), Divergence and flow direction as indicators of subsurface heterogeneity and stage-dependent storage in alluvial floodplains, *Hydrol. Processes*, *28*, 1307–1317.

Helton, A. M., et al. (2011), Thinking outside the channel: Modeling nitrogen cycling in networked river ecosystems, *Frontiers Ecol. Environ.*, *9*(4), 229–238, doi:10.1890/080211.

Hess, L., A. G. Affonso, E. de Moraes Arraut, E. M. L. de Moraes Novo, R. Gielow, V. F. Renó, B. Barbarisi, and B. Marioni (2011), Evaluation of low-cost, tree-mounted temperature loggers for validation of satellite-based flood mapping on the Amazon floodplain, in *Anais XV Simpósio Brasileiro de Sensoriamento Remoto—SBSR*, p. 5278, Inst. Naci. de Pesqui. Espaciais, Curitiba.

Hester, E. T., and M. N. Gooseff (2010), Moving beyond the banks: Hyporheic restoration is fundamental to restoring ecological services and functions of streams, *Environ. Sci. Technol.*, *44*, 1521–1525.

Hester, E. T., and M. N. Gooseff (2011), Hyporheic restoration in streams and rivers, in *Stream Restoration in Dynamic Fluvial Systems: Scientific Approaches, Analyses, and Tools, Geophys. Monogr. Ser.*, vol. 194, edited by A. Simon, S. J. Bennett, and J. M. Castro, pp. 167–187, AGU, Washington, D. C.

Hoehn, E., and O. A. Cirpka (2006), Assessing residence times of hyporheic ground water in two alluvial flood plains of the Southern Alps using water temperature and tracers, *Hydrol. Earth Syst. Sci.*, *10*(4), 553–563.

Hopkinson, L. C., and T. M. Wynn (2009), Vegetation impacts on near bank flow, *Ecohydrology*, *2*, 404–418.

Hunt, B. (1999), Unsteady stream depletion from ground water pumping, *Ground Water*, *37*(1), 98–102.

Hunt, R. J., D. P. Krabbenhoft, and M. P. Anderson (1996), Groundwater inflow measurements in wetland systems, *Water Resour. Res.*, *32*(3), 495–507.

Hunt, R. J., T. B. Coplen, N. L. Haas, D. A. Saad, and M. A. Borchardt (2005), Investigating surface water-well interaction using stable isotope ratios of water, *J. Hydrol.*, *302*(1–4), 154–172, doi:10.1016/j.jhydrol.2004.07.010.

Jackman, A. P., R. A. Walters, and V. C. Kennedy (1984), Transport and concentration controls for chloride, strontium, potassium and lead in Uvas Creek, a small cobble-bed stream in Santa Clara County, California, U.S.A. Part 2: Mathematical modeling, *J. Hydrol.*, *75*, 111–141.

Jackson, T. R., R. Haggerty, and S. V. Apte (2013a), A fluid-mechanics based classification scheme for surface transient storage in riverine environments: Quantitatively separating surface from hyporheic transient storage, *Hydrol. Earth Syst. Sci.*, *17*, 2747–2779, doi:10.5194/hess-17-2747-2013.

Jackson, T. R., R. Haggerty, S. V. Apte, and B. L. O'Connor (2013b), A mean residence time relationship for lateral cavitiesin gravel-bed rivers and streams: Incorporating streambed roughness and cavity shape, *Water Resour. Res.*, *49*, 3642–3650, doi:10.1002/wrcr.20272.

Jencso, K. G., B. L. McGlynn, M. N. Gooseff, K. E. Bencala, and S. M. Wondzell (2010), Hillslope hydrologic connectivity controls riparian groundwater turnover: Implications of catchment structure for riparian buffering and stream water sources, *Water Resour. Res.*, *46*, W10524, doi:10.1029/2009WR008818.

Johnson, A. N., B. R. Boer, W. W. Woessner, J. A. Stanford, GC. Poole, S. A. Thomas, and S. J. O'Daniel (2005), Evaluation of an inexpensive small-diameter temperature logger for documenting ground water-river interactions, *Ground Water Monit. Remediation*, *25*(4), 68–74.

Jones, C. N., D. T. Scott, B. L. Edwards, and R. F. Keim (2014), Perirheic mixing and biogeochemical processing in flow-through and backwater floodplain wetlands, *Water Resour. Res.*, *50*, 7394–7405, doi:10.1002/2014WR015647.

Jones, J., and P. J. Mulholland (2000), *Streams and Ground Waters*, Academic, San Diego, Calif.

Jung, M., T. P. Burt, and P. D. Bates (2004), Toward a conceptual model of floodplain water table response, *Water Resour. Res.*, *40*, W12409, doi:10.1029/2003WR002619.

Junk, W. J., P. B. Bayley, and R. E. Sparks (1989), The food pulse concept in river floodplain systems, *Can. Spec. Publ. Fish. Aquat. Sci.*, *106*, 110–127.

Kasahara, T., and S. M. Wondzell (2003), Geomorphic controls on hyporheic exchange flow in mountain streams, *Water Resour. Res.*, *39*(1), 1005, doi:10.1029/2002WR001386.

Käser, D. H., A. Binley, A. L. Heathwaite, and S. Krause (2009), Spatio-temporal variations of hyporheic flow in a riffle-step-pool sequence, *Hydrol. Processes*, *23*(15), 2138–2149.

Kennedy, C. D., D. P. Genereux, D. R. Corbett, and H. Mitasova (2009), Spatial and temporal dynamics of coupled groundwater and nitrogen fluxes through a streambed in an agricultural watershed, *Water Resour. Res.*, *45*, W09401, doi:10.1029/2008WR007397.

Kiel, B. A., and M. B. Cardenas (2014), Lateral hyporheic exchange throughout the Mississippi River network, *Nat. Geosci.*, *7*, 413–417, doi:10.1038/ngeo2157.

Kilpatrick, F. A., and E. D. Cobb (1985), *Measurement of Discharge Using Tracers, TWRI Book 3*, chap. A-16, U.S. Geol. Surv., Wash.

Kim, B. K. A., A. P. Jackman, and F. J. Triska (1990), Modeling transient storage and nitrate uptake kinetics in a flume containing a natural periphyton community, *Water Res. Resour.*, *26*(3), 505–515.

Kim, B. K. A., A. P. Jackman, and F. J. Triska (1992), Modeling biotic uptake by periphyton and transient hyporheic storage of nitrate in a natural stream, *Water Resour. Res.*, *28*(10), 2743–2752.

Kim, H., H. F. Hemond, L. R. Krumholz, and B. A. Cohen (1995), In-situ biodegradation of toluene in a contaminated stream. Part 1. Field studies, *Environ. Sci. Technol.*, *29*(1), 108–116, doi:10.1021/es00001a014.

Kimball, B. A., R. E. Broshears, K. E. Bencala, and D. M. McKnight (1994), Coupling of hydrologic transport and chemical-reactions in a stream affected by acid-mine drainage, *Environ. Sci. Technol.*, *28*(12), 2065–2073.

Knighton, D. (1998), *Fluvial Forms and Processes: A New Perspective*, Oxford Univ. Press, N. Y.

Kondolf, G. M., and H. Piègay (2003), *Tools in Fluvial Geomorphology*, John Wiley, Chichester, U. K.

Krest, J. M., and J. W. Harvey (2003), Using natural distributions of short-lived radium isotopes to quantify groundwater discharge and recharge, *Limnol. Oceanogr. Methods*, *48*, 290–298.

Lamontagne, S., and P. G. Cook (2007), Estimation of hyporheic water residence time in situ using 222Rn disequilibrium, *Limnol. Oceanogr. Methods*, *5*, 407–416.

Larkin, R. G., and J. M. Sharp (1992), On the relationship between river-basin geomorphology, aquifer hydraulics, and ground-water flow direction in alluvial aquifers, *Bull. Geol. Soc. Am.*, *104*(12), 1608–1620.

Larsen, L. G., J. Choi, M. K. Nungesser, and J. W. Harvey (2012), Directional connectivity in hydrology and ecology, *Ecol. Appl.*, *22*, 2204–2220, doi:10.1890/11-1948.1.

Larsen, L. G., J. W. Harvey, and M. M. Maglio (2014), Dynamic hyporheic exchange at intermediate time scales: Testing the relative importance of evapotranspiration and flood pulses, *Water Resour. Res.*, *50*, 318–335, doi:10.1002/2013WR014195.

Lautz, L. K., and R. M. Fanelli (2008), Seasonal biogeochemical hotspots in the streambed around restoration structures, *Biogeochemistry*, *91*(1), 85–104.

Lautz, L. K., D. I. Siegel, and R. L. Bauer (2006), Impact of debris dams on hyporheic interaction along a semi-arid stream, *Hydrol. Processes*, *20*(1), 183–196, doi:10.1002/hyp.5910.

Lee, D. R. (1977), A device for measuring seepage flux in lakes and estuaries, *Limnol. Oceanogr.*, *22*(1), 140–147.

Loague, K. (2010), *Rainfall-Runoff Modeling: Selection, Introduction, and Commentary by Keith Loague, Benchmark Pap. Hydrol.*, 506 p., Int. Assoc. of Hydrol. Sci., Oxfordshire.

Lowry, C. S., J. F. Walker, R. J. Hunt, and M. P. Anderson (2007), Identifying spatial variability of groundwater discharge in a wetland stream using a distributed temperature sensor, *Water Resour. Res.*, *43*, W10408, doi:10.1029/2007WR006145.

Martin, R. L., and D. J. Jerolmack (2013), Origin of hysteresis in bedform response to unsteady flows, *Water Resour. Res.*, *49*, 1314–1333, doi:10.1002/wrcr.20093.

Marzadri, A., D. Tonina, A. Bellin, and J. L. Tank (2014), A hydrologic model demonstrates nitrous oxide emissions depend on streambed morphology, *Geophys. Res. Lett.*, *41*, 5484–5491, doi:10.1002/2014GL060732.

Menichino, G. T., A. S. Ward, and E. T. Hester (2014), Macropores as preferential flow paths in meander bends, *Hydrol. Processes*, *28*, 482–495.

McClain, M. E., et al. (2003), Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems, *Ecosystems*, *6*(4), 301–312, doi:10.1007/s10021-003-0161-9.

McDonnell, J. J., and K. Beven (2014), Debates—The future of hydrological sciences: A (common) path forward? A call to action aimed at understanding velocities, celerities and residence time distributions of the headwater hydrograph, *Water Resour. Res.*, *50*, 5342–5350, doi:10.1002/2013WR015141.

McKean, J. D. Tonina, C. Bohn, and C. W. Wright (2014), Effects of bathymetric lidar errors on flow properties predicted with a multi-dimensional hydraulic model, *J. Geophys. Res. Earth Surf.*, *119*, 644–664, doi:10.1002/2013JF002897.

McKnight, D. M., B. A. Kimball, and K. E. Bencala (1988), Iron photoreduction and oxidation in an acidic mountain stream, *Science*, *240*, 637–640, doi:10.1126/science.240.4852.637.

Merck, M. F., B. T. Neilson, R. Cory, and G. W. Kling (2012), Variability of instream and riparian storage in a beaded arctic stream, *Hydrol. Processes*, *26*, 2938–2950, doi:10.1002/hyp.8323.

Mertes, L. A. K. (1997), Documentation and significance of the perirheic zone on inundated floodplains, *Water Resour. Res.*, *33*(7), 1749–1762, doi:10.1029/97WR00658.

Minshall, G. W., K. W. Cummins, R. C. Petersen, C. E. Cushing, D. A. Bruns, J. R. Sedell, and R. L. Vannote (1985), Developments in strewam ecosystem theory, *Can. J. Fish. Aquat. Sci.*, *42*, 1045–1055.

Moench, A. F., V. B. Sauer, and M. E. Jennings (1974), Modification of routed streamflow by channel loss and base flow, *Water Resour. Res.*, *10*(5), 963–968, doi:10.1029/WR010i005p00963.

Montgomery, D. R. (1999), Process domains and the river continuum, *J. Am. Water Resour. Assoc.*, *35*, 397–410.

Mulholland, P. J. (1992), Regulation of nutrient concentrations in a temperate forest stream: Roles of upland, riparian, and instream processes, *Limnol. Oceanogr.*, *37*(7), 1512–1526.

Mulholland, P. J., E. R. Marzolf, J. R. Webster, D. R. Hart, and S. P. Hendricks (1997), Evidence that hyporheic zones increase heterotrophic metabolism and phosphorus uptake in forest streams, *Limnol. Oceanogr.*, *42*(3), 443–451.

Mulholland, P. J., et al. (2008), Stream denitrification across biomes and its response to anthropogenic nitrate loading, *Nature*, *452*, 202–206, doi:10.1038/nature06686.

Mwakanyamale, K., L. Slater, F. D. Day-Lewis, M. Elwaseif, D. Ntarlagiannis, and C. D. Johnson (2012), Spatially variable stage-driven groundwater-surface water interaction inferred from time-frequency analysis of distributed temperature sensing data, *Geophys. Res. Lett.*, *39*, L06401, doi:10.1029/2011GL050824.

Nadeau, T. L., and M. C. Rains (2007), Hydrological connectivity between headwater streams and downstream waters: How science can inform policy, *J. Am. Water Resour. Assoc.*, *43*(1), 118–133, doi:10.1111/j.1752-1688.2007.00010.x.

National Research Council (2002), *Riparian Areas: Functions and Strategies for Management*, Natl. Acad. Press, Washington, D. C.

Neilson, B. T., D. K. Stevens, S. C. Chapra, and C. Bandaragoda (2010), Two-zone transient storage modeling using temperature and solute data with multiobjective calibration: 2. Temperature and solute, *Water Resour. Res.*, *46*, W12521, doi:10.1029/2009WR008759.

Newbold, J. D., J. W. Elwood, R. V. O'Neill, and W. Van Winkle (1981), Measuring nutrient spiralling in streams, *Can. J. Fish. Aquat. Sci.*, *38*, 860–863.

Newbold, J. D., R. V. O'Neill, J. W. Elwood, and W. Van Winkle (1982), Nutrient spiralling in streams: Implications for nutrient limitation and invertebrate activity, *Am. Nat.*, *120*(5), 628–652.

Nikora, V. I., and D. G. Goring (1998), ADV measurements of turbulence: Can we improve their interpretation?, *J. Hydrol. Eng.*, *124*(6), 630–634.

Nowinski, J. D., M. B. Cardenas, and A. F. Lightbody (2011), Evolution of hydraulic conductivity in the floodplain of a meandering river due to hyporheic transport of fine materials, *Geophys. Res. Lett.*, *38*, L01401, doi:10.1029/2010GL045819.

O'Connor, B. L., and J. W. Harvey (2008), Scaling hyporheic exchange and its influence on biogeochemical reactions in aquatic ecosystems, *Water Resour. Res.*, *44*, W12423, doi:10.1029/2008WR007160.

O'Connor, B. L., M. Hondzo, and J. W. Harvey (2010), Predictive modeling of transient storage and nutrient uptake, *J. Hydraul. Eng.*, *136*(12), 1018–1032.

Packman, A. I., N. H. Brooks, and J. J. Morgan (2000), A physicochemical model for colloid exchange between a stream and a sand streambed with bedforms, *Water Resour. Res.*, *36*(8), 2351–2361, doi:10.1029/2000WR900059.

Payn, R. A., M. N. Gooseff, B. L. McGlynn, K. E. Bencala, and S. M. Wondzell (2009), Channel water balance and exchange with subsurface flow along a mountain headwater stream in Montana, United States, *Water Resour. Res.*, *45*, W11427, doi:10.1029/2008WR007644.

Payn, R. A., M. N. Gooseff, B. L. McGlynn, S. M. Wondzell, and K. E. Bencala (2012), Exploring changes in the spatial distribution of stream baseflow generation during a seasonal recession, *Water Resour. Res.*, *48*, W04519, doi:10.1029/2011WR011552.

Peterson, B. P., et al. (2001), Control of nitrogen export from watersheds by headwater streams, *Science*, *292*(5514), 86–90, doi:10.1126/science.1056874.

Pinay, G., T. C. O'Keefe, R. T. Edwards, and R. J. Naiman (2009), Nitrate removal in the hyporheic zone of a salmon river in Alaska, *River Res. Appl.*, *25*(4), 367–375.

Pinder, G. F., and S. P. Sauer (1971), Numerical simulation of flood wave modification due to bank storage effects, *Water Resour. Res.*, *7*(1), 63–70, doi:10.1029/WR007i001p00063.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg (1997), The natural flow regime source, *BioScience*, *47*(11), 769–784.

Poole, G. C. (2010), Stream hydrogeomorphology as a physical science basis for advances in stream ecology, *J. North Am. Benthol. Soc.*, *29*(1), 12–25.

Poole, G. C., J. A. Stanford, S. W. Running, and C. A. Frissell (2006), Multiscale geomorphic drivers of groundwater flow paths: Subsurface hydrologic dynamics and hyporheic habitat diversity, *J. North Am. Benthol. Soc.*, *25*(2), 288–303.

Prestegaard, K. L. (1983), Bar resistance in gravel bed streams at bankfull stage, *Water Resour. Res.*, *19*(2), 472–476.

Price, R. M., Z. Top, J. D. Happell, and P. K. Swart (2003), Use of tritium and helium to define groundwater flow conditions in Everglades National Park, *Water Resour. Res.*, *39*(9), 1267, doi:10.1029/2002WR001929.

Rapanos v. United States (2006), Docket No. 04-1034.

Raymond, P. A., J. E. Saiers, and W. V. Sobczak (2015), Hydrological and biogeochemical controls on watershed dissolved organic matter transport: Pulse-shunt concept, Ecology. [Available at http://dx.doi.org/10.1890/14-1684.1.]

Reddy, M. M., M. B. Reddy, K. L. Kipp, A. Burman, P. Schuster, and P. S. Rawlik Jr. (2008), Peat porewater chloride concentration profiles in the Everglades during wet/dry cycles from January 1996 to June 1998: Field measurements and theoretical analysis, *Hydrol. Processes*, *22*, 1713–1724, doi:10.1002/hyp.6739.

Richardson, W., E. A. Strauss, L. A. Bartsch, E. M. Monroe, J. C. Cavanaugh, L. Vingum, and D. M. Soballe (2004), Denitrification in the Upper Mississippi River: Rates, controls, and contribution to nitrate flux, *Can. J. Fish. Aquat. Sci.*, *61*, 1102–1112, doi:10.1139/f04-062.

Richey, J. E., L. A. K. Mertes, T. Dunne, R. L. Victoria, B. R. Forsberg, A. C. N. S. Tancredi, and E. Oliveira (1989), Sources and routing of the Amazon River Flood Wave, *Global Biogeochem. Cycles*, *3*(3), 191–204, doi:10.1029/GB003i003p00191.

Roley, S. S., J. L. Tank, M. L. Stephen, L. T. Johnson, J. J. Beaulieu, and J. D. Witter (2012a), Floodplain restoration enhances denitrification and reach-scale nitrogen removal in an agricultural stream, *Ecol. Appl.*, *22*, 281–297.

Roley, S. S., J. L. Tank, and M. A. Williams (2012b), Hydrologic connectivity increases denitrification in the hyporheic zone and restored floodplains of an agricultural stream, *J. Geophys. Res.*, *117*, G00N04, doi:10.1029/2012JG001950.

Rosenberry, D. O., R. W. Sheibley, S. E. Cox, F. W. Simonds, and D. L. Naftz (2013), Temporal variability of exchange between groundwater and surface water based on high-frequency direct measurements of seepage at the sediment-water interface, *Water Resour. Res.*, *49*, 2975–2986, doi:10.1002/wrcr.20198.

Ruehl, C., A. T. Fisher, C. Hatch, M. Los Huertos, G. Stemler, and C. Shennan (2006), Differential gauging and tracer tests resolve seepage fluxes in a strongly losing stream, *J. Hydrol.*, *330*(1), 235–248.

Runkel, R. L. (1998), One-dimensional transport with inflow and storage (OTIS): A solute transport model for streams and rivers, *U.S. Geol. Surv. Water Resour. Invest. Rep., 98–4018*, Denver.

Runkel, R. L. (2010), *One-Dimensional Transport With Equilibrium Chemistry (OTEQ)—A Reactive Transport Model for Streams and Rivers, Techniques and Methods, Book 6, Chap. B6*, U.S. Geol. Surv., Reston.

Runkel, R. L. (2015), On the use of rhodamine WT for the characterization of stream hydrodynamics and transient storage, *Water Resour. Res.*, doi:10.1002/2015WR017201, in press.

Runkel, R. L., K. E. Bencala, R. E. Broshears, and S. C. Chapra (1996a), Reactive solute transport in streams: 1. Development of an equilibrium-based model, *Water Resour. Res.*, *32*(2), 409–418, doi:10.1029/95WR03106.

Runkel, R. L., D. M. McKnight, K. E. Bencala, and S. C. Chapra (1996b), Reactive solute transport in streams: 2. Simulation of a *p*H Modification Experiment, *Water Resour. Res.*, *32*(2), 419–430, doi:10.1029/95WR03107.

Runkel, R. L., B. A. Kimball, D. M. McKnight, and K. E. Bencala (1999), Reactive solute transport in streams: A surface complexation approach for trace metal sorption, *Water Resour. Res.*, *35*(12), 3829–3840.

Savant, S. A., D. D. Reible, and L. J. Thibodeaux (1987), Convective transport within stable river sediments, *Water Resour. Res.*, *23*(9), 1763–1768.

Sawyer, A. H., M. B. Cardenas, A. Bomar, and M. Mackey (2009), Impact of dam operations on hyporheic exchange in the riparian zone of a regulated river, *Hydrol. Processes*, *23*, 2129–2137, doi:10.1002/hyp.7324.

Sawyer, A. H., M. B. Cardenas, and J. Buttles (2011), Hyporheic exchange due to channel-spanning logs, *Water Resour. Res.*, *47*, W08502, doi:10.1029/2011WR010484.

Schmadel, N. M., B. T. Neilson, and T. Kasahara (2014), Deducing the spatial variability of exchange within a longitudinal channel water balance, *Hydrol. Processes*, *28*, 3088–3103, doi:10.1002/hyp.9854.

Scott, D. T., R. F. Keim, B. L. Edwards, C. N. Jones, and D. E. Kroes (2014), Floodplain biogeochemical processing of floodwaters in the Atchafalaya River Basin during the Mississippi River flood of 2011, *J. Geophys. Res. Biogeosci.*, *119*, 537–546, doi:10.1002/2013JG002477.

Selker, J. S., et al. (2006), Distributed fiber-optic temperature sensing for hydrologic systems, *Water Resour. Res.*, *42*, W12202, doi:10.1029/2006WR005326.

Shope, C. L., J. Constantz, C. A. Cooper, M. D. Reeves, G. Pohll, and W. A. McKay (2012), Influence of a large fluvial island, streambed, and streambank on surface water-groundwater fluxes and water table dynamics, *Water Resour. Res.*, *48*, W06512, doi:10.1029/2011WR011564.

Squillace, P. J., E. M. Thurman, and E. T. Furlong (1993), Groundwater as a nonpoint source of atrazine and deethylatrazine in a river during base flow conditions, *Water Resour. Res.*, *29*(6), 1719–1729, doi:10.1029/93WR00290.

Stanford, J. A., and T. Gonser (Eds.) (1998), Rivers in the landscape: Special issue on riparian and groundwater ecology, *Freshwater Biol.*, *40*(3), 402–585.

Stanford, J. A., and J. V. Ward (1988), The hyporheic habitat of river ecosystems, *Nature*, *335*(6185), 64–66.

Stanford, J. A., and J. V. Ward (1993), An ecosystem perspective of alluvial rivers: Connectivity and the hyporheic corridor, *J. North Am. Benthol. Soc.*, *12*(1), 48–60.

Stewart, R. J., W. M. Wollheim, M. N. Gooseff, M. A. Briggs, J. M. Jacobs, B. J. Peterson, and C. S. Hopkinson (2011), Separation of river scale nitrogen removal among main channel and two transient storage compartments, *Water Resour. Res.*, *47*, W00J10, doi:10.1029/2010WR009896.

Stoertz, M. W., and K. R. Bradbury (1989), Mapping recharge areas using a ground-water flow model—A case study, *Ground Water*, *27*, 220–228, doi:10.1111/j.1745-6584.1989.tb00443.x.

Stonedahl, S. H., J. W. Harvey, A. Wörman, M. Salehin, and A. I. Packman (2010), A multiscale model for integrating hyporheic exchange from ripples to meanders, *Water Resour. Res.*, *46*, W12539, doi:10.1029/2009WR008865.

Stonedahl, S. H., J. W. Harvey, J. Detty, A. Aubeneau, and A. I. Packman (2012), Physical controls and predictability of stream hyporheic flow evaluated with a multi-scale model, *Water Resour. Res.*, *48*, W10513, doi:10.1029/2011WR011582.

Stonedahl, S. H., J. W. Harvey, and A. I. Packman (2013), Interactions between hyporheic flow produced by streammeanders, bars, and dunes, *Water Resour. Res.*, *9*, 5450–5461, doi:10.1002/wrcr.20400.

Storey, R. G., K. W. F. Howard, and D. D. Williams (2003), Factors controlling riffle-scale hyporheic exchange flows and their seasonal changes in a gaining stream: A three-dimensional groundwater flow model, *Water Resour. Res.*, *39*(2), 1034, doi:10.1029/2002WR001367.

Stream Solute Workshop (1990), Concepts and methods for assessing solute dynamics in streamecosystems, *J. North Am. Benthol. Soc.*, *9*, 95–119.

Streeter, H. W., and E. B. Phelps (1925), *A Study of the Pollution and Natural Purification of the Ohio River. III. Factors Concerned in the Phenomena of Oxidation and Reaeration, Public Health Bull. 146*. [Reprinted by U.S. Dep. of Health, Educ. and Welfare, Public Health Serv., Washington.]

Swanson, T. E., and M. B. Cardenas (2011), Ex-Stream: A MATLAB® program for calculating fluid flux through sediment-water interfaces based on steady and transient temperature profiles, *Comput. Geosci.*, *37*, 1664–1669, doi:10.1016/j.cageo.2010.12.001.

Thackston, E. L., and K. B. Schnelle (1970), Predicting effects of dead zones on stream mixing, *J. Sanit. Eng. Div.*, *96*, 319–331.

Thibodeaux, L. J., and J. D. Boyle (1987), Bedform-generated convective transport in bottom sediment, *Nature*, *325*, 341–343.

Townsend, P. A., and J. R. Foster (2002), A synthetic aperture radar–based model to assess historical changes in lowland floodplain hydroperiod, *Water Resour. Res.*, *38*(7), doi:10.1029/2001WR001046.

Vannote, R. J., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing (1980), The river continuum concept, *Can. J. Fish. Aquat. Sci.*, *37*, 130–137.

Valentine, E., and I. Wood (1979), Experiments in longitudinal dispersion with dead zones, *J. Hydraul. Div. Am. Soc. Civ. Eng.*, *105*, 999–1016.

Velickovic, B. (2005), Colmation as one of the processes in interaction between the groundwater and surface water, *Facta Univ.*, *3*, 165–172.

Von Gunten, H., and C. Lienert (1993), Decreased metal concentrations in ground-water caused by controls of phosphate emissions, *Nature*, *364*, 220–222.

Voytek, E. B., A. Drenkelfuss, F. D. Day-Lewis, R. Healy, D. Werkema, and J. W. Lane Jr. (2014), 1DTempPro—Analyzing temperature profiles for groundwater/surface-water exchange, *Ground Water*, *52*(2), 298–302, doi:10.1111/gwat.12051.

Wagner, B. J., and J. W. Harvey (1997), Experimental design for estimating parameters of rate-limited mass transfer: Analysis of stream tracer studies, *Water Resour. Res.*, *33*(7), 1731–1741.

Ward, A. S., M. N. Gooseff, and K. Singha (2010), Characterizing hyporheic transport processes—Interpretation of electrical geophysical data in coupled stream-hyporheic zone systems during solute tracer studies, *Adv. Water Resour.*, *33*(11), 1320–1330.

Ward, A. S., M. N. Gooseff, and P. A. Johnson (2011), How can subsurface modifications to hydraulic conductivity be designed as stream restoration structures? Analysis of Vaux's conceptual models to enhance hyporheic exchange, *Water Resour. Res.*, *47*, W08512, doi:10.1029/2010WR010028.

Ward, A. S., M. Fitzgerald, M. N. Gooseff, T. J. Voltz, A. M. Binley, and K. Singha (2012), Hydrologic and geomorphic controls on hyporheic exchange during base flow recession in a headwater mountain stream, *Water Resour. Res.*, *48*, W04513, doi:10.1029/2011WR011461.

Ward, A. S., R. A. Payn, M. N. Gooseff, B. L. McGlynn, K. E. Bencala, C. A. Kelleher, S. M. Wondzell, and T. Wagener (2013), Variations in surface water-ground water interactions along a headwater mountain stream: Comparisons between transientstorage and water balance analyses, *Water Resour. Res.*, *49*, 3359–3374, doi:10.1002/wrcr.20148.

Ward, A. S., M. N. Gooseff, M. Fitzgerald, T. J. Voltz, and K. Singha (2014), Spatially distributed characterization of hyporheic solute transport during baseflow recession in a headwater mountain stream using electrical geophysical imaging, *J. Hydrol.*, *517*, 362–377.

Ward, J. V. (1989), The four-dimensional nature of lotic ecosystems the four-dimensional nature of lotic ecosystems, *J. North Am. Benthol. Soc.*, *8*(1), 2–8.

Warner, N. R., C. A. Christie, R. B. Jackson, and A. Vengosh (2013), Impacts of shale gas wastewater disposal on water quality in western Pennsylvania, *Environ. Sci. Technol.*, *47*(20), 11,849–11,857, doi:10.1021/es402165b.

Webster, J. R., et al. (2003), Factors affecting ammonium uptake instreams—An inter-biome perspective, *Freshwater Biol.*, *48*, 1329–1352.

Winter, T. C., J. W. Harvey, O. L. Franke, and W. M. Alley (1998), Groundwater and surface water: A single resource, *USGS Circ. 1139*, U.S. Geol. Surv, Denver.

Woessner, W. W. (2000), Stream and fluvial plain ground water interactions: Rescaling hydrogeologic thought, *Ground Water*, *38*(3), 423–429.

Wohl, E., B. P. Bledsoe, R. B. Jacobsen, N. L. Poff, S. L. Rathbun, D. M. Waters, and A. C. Wilcox (2015), The natural sediment regime in rivers: Broadening the foundation for ecosystem management, *BioScience*, *65*, 358–371.

Wollheim, M. N., T. K.Harms, B. J. Peterson, K. Morkeski, C. S. Hopkinson, R. J. Stewart, M. N. Gooseff, and M. A. Briggs (2014), Nitrate uptake dynamics of surface transient storage in stream channels and fluvial wetlands, *Biogeochemistry*, *120*, 239–257, doi:10.1007/s10533-014-9993-y.

Wollheim, W. M., C. J. Voosmarty, B. J. Peterson, S. P. Seitzinger, and C. S. Hopkinson (2006), Relationship between river size and nutrient removal, *Geophys. Res. Lett.*, *33*, L06410, doi:10.1029/2006GL025845.

Wondzell, S. M., and F. J. Swanson (1996), Seasonal and storm dynamics of the hyporheic zone of a 4 th-order mountain stream. 2: Nitrogen cycling. *J. North Am. Benthol. Soc.*, *15*(1), 3–19.

Wondzell, S. M., and F. J. Swanson (1999), Floods, channel change, and the hyporheic zone, *Water Resour. Res.*, *35*(2), 555–567.

Wondzell, S. M., M. N. Gooseff, and B. L. McGlynn (2010), An analysis of alternative conceptual models relating hyporheic exchange flow to diel fluctuations in discharge during baseflow recession, *Hydrol. Processes*, *24*, 686–694.

Wörman, A. (2000), Comparison of models for transient storage of solutes in small streams, *Water Resour. Res.*, *36*(2), 455–468.

Wörman, A., A. Packman, and K. Jonsson (2002), Effect of flow-induced exchange in hyporheic zones on longitudinal transport of solutes in stream and rivers, *Water Resour. Res.*, *38*(1), 1001, doi:10.1029/2001WR000769.

Wörman, A., A. I. Packman, L. Marklund, J. Harvey, and S. H. Stone (2007), Fractal topography and subsurface water flows from fluvial bedforms to the continental shield, *Geophys. Res. Lett.*, *34*, L07402, doi:10.1029/2007GL029426.

Wroblicky, G. J., M. E. Campana, H. M. Valett, and C. N. Dahm (1998), Seasonal variation in surface-subsurface water exchange and lateral hyporheic area of two stream-aquifer systems, *Water Resour. Res.*, *34*(3), 317–328.

Zarnetske, J. P., M. N. Gooseff, T. R. Brosten, J. H. Bradford, J. P. McNamara, and W. B. Bowden (2007), Transient storage as a function of geomorphology, discharge, and permafrost active layer conditions in Arctic tundra streams, *Water Resour. Res.*, *43*, W07410, doi:10.1029/2005WR004816.

Zarnetske, J. P., R. Haggerty, S. M. Wondzell, and M. A. Baker (2011), Dynamics of nitrate production and removal as a function of residence time in the hyporheic zone, *J. Geophys. Res.*, *116*, G01025, doi:10.1029/2010JG001356.

Zimmer, M. A., and L. K. Lautz (2014), Temporal and spatial response of hyporheic zone geochemistry to a storm event, *Hydrol. Processes*, *28*, 2324–2337, doi:10.1002/hyp.9778.