GROUND WATER

Review Paper

Effects of Ground Water Exchange on the Hydrology and Ecology of Surface Water

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Abstract

Ground water exchange affects the ecology of surface water by sustaining stream base flow and moderating water-level fluctuations of ground water-fed lakes. It also provides stable-temperature habitats and supplies nutrients and inorganic ions. Ground water input of nutrients can even determine the trophic status of lakes and the distribution of macrophytes. In streams the mixing of ground water and surface water in shallow channel and bankside sediments creates a unique environment called the hyporheic zone, an important component of the lotic ecosystem. Localized areas of high ground water discharge in streams provide thermal refugia for fish. Ground water also provides moisture to riparian vegetation, which in turn supplies organic matter to streams and enhances bank resistance to erosion. As hydrologists and ecologists interact to understand the impact of ground water on aquatic ecology, a new research field called "ecohydrology" is emerging.

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Introduction

Surface water such as lakes, streams, and wetlands are ecologically important components of the landscape, providing habitat for diverse communities of plants and animals. Most surface water exchanges water and nutrients with the surrounding terrestrial environment, which has critical impact on the aquatic ecosystem. Surface water is almost always connected to ground water, and the exchange of water normally occurs at the sediment/water interface. Therefore, ground water/surface water interaction is often seen as a localized process at the interface. However, ground water flow has a much larger scale and is sensitive to the biogeographical conditions of the upland including geology, climate, vegetation, and land use.

This review paper is intended to (1) review fundamental ecological and hydrological concepts relevant to surface/ground water exchange, and (2) present some examples of the linkages between ground water and aquatic plants and animals.

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Fundamental Concepts: Ecology

Ecologists study the spatial and temporal distributions of combinations of species, called ecological communities, by examining three major processes: environmental conditions of the habitat, history or succession of the species, and interaction among the species such as competition or predation (Klijn and Witte 1999). Light, temperature, water quality, nutrient supply, and sediment type constitute important environmental factors, and ground water exchange affects all of them directly or indirectly. In the remainder of this section, we will briefly describe ecological characteristics of surface water environments, primarily condensing from Jeffries and Mills (1990).

Standing water bodies such as lakes (lentic systems) typically are characterized by stratification with depth. Light penetrating the water column is attenuated with depth until the energy harnessed by photosynthesis just equals the respiratory requirements of plants. This depth, called the compensation point, generally represents the lower limit of photosynthetic algae.

Besides light, plant growth relies on many chemical elements, of which nitrogen and phosphorus hold special importance. The availability of these nutrients determines the biological productivity of lentic bodies, often called the trophic status. A simple trophic classification system consists of the terms oligotrophic (nutrient poor), mesotrophic, and eutrophic (nutrient rich). The trophic status reflects nutrient loading, which is determined by watershed geology, climate, and vegetation. Anthropogenic input of nutrients, however, can significantly alter the trophic status.

The ecology of rivers and streams (lotic systems) is influenced by flow, and erosion and deposition of sediments. A river channel contains a mosaic of habitats. In the longitudinal direction, a typical pattern consists of alternating shallows, with coarse sediments and turbulent flow (riffles), and deeper, depositional areas

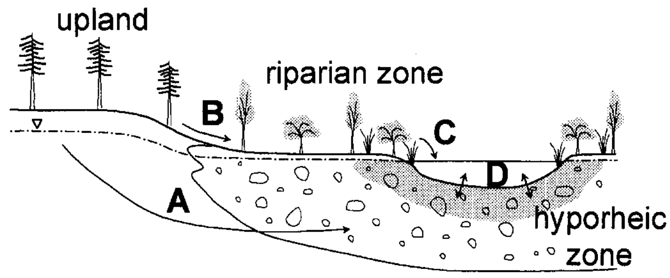

Figure 1. Schematic cross section of an active channel, hyporheic zone (shaded), riparian zone, and upland. Hatched area indicates fluvial sands and gravels, and the triangle indicates the water table. Major pathways of water and nutrient exchange are indicated: (a) ground water flow, (b) overland flow, (c) litter fall, (d) hyporheic exchange.

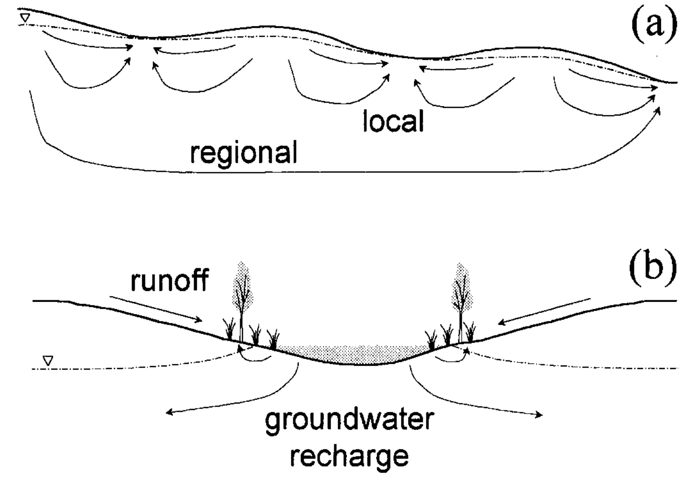

Figure 2. (a) Regional and local ground water flow systems. Arrows indicate ground water flow patterns. (b) Depression-focused ground water recharge.

of laminar flow (pools). In the lateral direction, a river often has incised channels within the floodplain, which is flanked by a transitional zone between the river and the surrounding uplands called the riparian zone (Figure 1). The riparian zone is an example of an ecotone, which is defined as a zone encompassing sharp gradients of environmental factors, ecological processes, and plant communities between two distinctively different environments, in this case aquatic and terrestrial (Gregory et al. 1991). The riparian zone has important ecological functions, such as shading the river surface and regulating the input of organic matter and nutrients. In this paper, the word riparian is used for both lotic and lentic systems despite its lotic origin, because hydrological and ecological functions at their terrestrial/aquatic interfaces are similar. In the vertical direction, continuous exchange of water and mass between the stream and underlying fluvial sediments creates a unique environment at and immediately beneath the sediment/water interface called the hyporheic zone (Figure 1).

Lotic systems are dynamic environments where channel locations and characteristics change frequently in response to erosion and deposition, and the ecological community continuously adapts to the changing environment. For example, younger stands of deciduous shrubs and trees grow on the floodplain close to the active channel, while older plant communities composed of upland species may characterize portions of the floodplain farther away (Gregory et al. 1991). Therefore, in addition to the three other dimensions (longitudinal, lateral, and vertical), time plays a major role in the ecology of lotic systems (Ward 1989).

Fundamental Concepts: Ground Water

Low river flow during periods of no rain or snowmelt input is called base flow, which represents the "normal" condition of rivers. Ground water provides base flow for essentially all rivers and has a major effect on the amount of water, chemical composition, and temperature of rivers. In smaller, low-order streams, ground water also provides much of the increased discharge during and immediately following storms (Sklash and Farvolden 1979). The shape of the water table often is a subdued replica of the land surface, in the sense that the water table is highest under the hills and slopes toward the lowest points in the landscape, such as depressions and valleys (Figure 2a). The water table is generally shallow under the riparian zone and becomes deeper under hill slopes (Figure 1). The horizontal flow direction of shallow ground water is determined by the slope of the water table. Therefore, rivers and lakes are commonly at the receiving end of the ground water flow that originates under uplands.

Figure 2a shows a commonly accepted paradigm of a ground water flow system (Toth 1963), wherein regional-scale (10^3 to 10^5 m) flow is driven by regional topography, and local-scale (10^0 to 10^2 m) flow is driven by local topography. From this diagram, it is clear that a change in the conditions of a recharge area, such as removal of vegetation, may significantly impact recharge and, therefore, stream base flow. Figure 2a represents a simple ground water flow system that has reasonably uniform input from precipitation. However, there are numerous other possible flow configurations where water flows from surface water to ground water. For example, in cold, semiarid regions, snowmelt runoff over frozen, impervious ground transfers a large amount of water from uplands to depressions (Woo and Winter 1993). As a result, ground water may temporarily flow from under the depressions to the surrounding uplands (Figure 2b). This phenomenon is called depression focused recharge (Lissey 1971). Additional discussion of ground water and surface water interactions in various physical settings can be found in Winter et al. (1998).

Figure 2a implies that ground water flow is fairly uniform over a large area. However, on a local scale, flow between ground water and surface water often is highly variable, and it is common to observe order-of-magnitude variations of flow rate over a short distance (Shaw and Prepas 1990). This variability is primarily due to the heterogeneous nature of sediments, although other factors will be discussed later in relation to the hyporheic zone. The highest rates of ground water discharge commonly occur in localized areas referred to as "upwellings" or springs.

The temporal variability in the surface water level and the water-table depth greatly impacts the types and density of vegetation and benthic organisms living at or near the surface/ground water interface. Species assemblages may depend on depth to the water table, variability in the water table, degree of extremely high or low water levels, timing of maximum water level, incidence of inundation, and duration of high or low water levels (Goslee et al. 1997; Wetzel 1999). Species distribution and composition frequently will change in response to the timing and magnitude of water-level changes in near-shore riparian areas (Wheeler 1999).

Ground water exchange directly affects the ecology of surface water by (1) sustaining stream base flow and moderating water-level fluctuations of ground water-fed lakes, (2) providing stable-temperature habitats, and (3) supplying nutrients and inorganic ions. Ground water also indirectly affects surface water by (1) providing moisture for riparian vegetation, and (2) controlling the shear strength of bank materials, thereby affecting slope stability and erosion processes.

Examples

Macrophytes in Lentic Systems

Aquatic macrophytes include flowering plants, algae, mosses, and ferns and may be emergent, floating on the water surface, or submergent (Jeffries and Mills 1990). Macrophytes actively alter the physicochemical environment of water and underlying sediments by shading the water column, changing concentrations of dissolved carbon dioxide and oxygen during respiration and photosynthesis, taking up nutrients, and releasing oxygen through roots in otherwise oxygen-depleted zones (Sand-Jensen et al. 1982). Macrophytes also supply organic matter to the food web and provide cover for smaller invertebrates and fish avoiding larger predators.

Type and abundance of macrophytes in lotic systems are correlated with the nutrient level in water, which often is controlled by the discharge of nutrient-rich ground water (Eglin et al. 1997). Distribution and abundance of lake-bottom macrophytes are commonly determined by water chemistry, wave exposure, light penetration, and substrate slope and type. However, these factors sometimes fail to explain distribution of submersed macrophytes. For example, in Sparkling Lake, located in a sandy, glacial-outwash aquifer in northern Wisconsin, Lodge et al. (1989) found that areas of enhanced growth coincided with areas of high ground water discharge flux (> 1 cm/day). Conversely, in another study in Minnesota, macrophytes were absent in near-shore springs with rapidly discharging ground water but were present along adjacent shorelines where springs were absent (Rosenberry et al. 2000). Both studies found a clear correlation between ground water discharge and macrophyte growth, but both failed to identify the specific chemical or physical process determining macrophyte distribution. Such conflicting results point out a need for interdisciplinary studies of aquatic plant physiology in relation to ground water discharge. It is also important to understand the effects of complex biogeochemical processes within lake bottom sediments. For example, LaBaugh et al. (1996) suggested that the macrophytes growing on the margin of wetlands in North Dakota provided an anaerobic zone in the sediments that resulted in loss of sulfate from ground water.

Eutrophication

Eutrophication commonly refers to the change in trophic status resulting from nutrient input by sewage discharge or agricultural runoff. The most conspicuous symptom of eutrophication is a large increase in the standing crop of phytoplankton, known as an algal "bloom." The water becomes turbid, supersaturated with oxygen in the daytime, anoxic at night and during decay of blooms. Some species of algae produce toxins, notably blue-green algae that often become increasingly dominant as eutrophication proceeds. The loss of diversity occurs in both floral and faunal communities while a few species tolerant of anoxic, turbid water survive (Jeffries and Mills 1990).

Of the various nutrients that can potentially affect the rate of phytoplankton productivity, phosphorus is the one most frequently limiting in fresh water (Schindler 1977). Phosphate has a high tendency to be adsorbed on soil particles and is commonly transported with suspended sediments in surface runoff. In ground water, phosphorus mobility is attenuated by its strong adsorption to soil particles. The next most frequently limiting nutrient is inorganic nitrogen in the form of nitrate or ammonium. Nutrient loading from agricultural watersheds may occur by surface runoff or discharge of shallow ground water, but nitrate is relatively mobile and its major pathway is subsurface flow (Hill 1996). Therefore, ground water exchange is a major factor controlling nitrate loading to surface water.

Jordan et al. (1997) studied the nitrate level in streams draining small (5 to 3000 ha) watersheds in Maryland. As expected, the nitrate level was higher in watersheds having a higher portion of fertilized cropland. The nitrate level was also higher in streams having high ratios of base flow to total flow, indicating the main pathway of nitrate is soil leaching and ground water flow. This study was significant because it examined ground water processes at landscape scales (Pringle and Triska 2000), going beyond the local scale exchange that operates at the stream/ground water interface.

Apart from agricultural sources, nitrate loading also occurs from septic systems, which are used by about one-third of the population of the United States for waste water disposal (Robertson et al. 1991). In particular, many cottages next to pristine, often oligotrophic, lakes use septic systems that may discharge nitrate-rich water in shallow aquifers connected to the lakes. Robertson et al. (1991) studied septic systems located on a shallow unconfined sand aquifer in Ontario, Canada. At one site (Muskoka), the plume of contaminated ground water migrated 20 m and reached an adjacent river. However, nitrate in the plume was almost completely removed before ground water discharged into the river, as indicated by samples from seepage meters. They attributed the attenuation to denitrification associated with organic matter decomposition in river-bed sediments. Transport of phosphorus to surface water via septic leachate plumes is less common because of its strong sorptive properties. However, phosphorus contamination of surface water is a problem for failing or old septic systems (Robertson et al. 1998).

Acidification

Acidification is another example of the anthropogenic alteration of aquatic ecosystems associated with ground water exchange. Since the early 1970s, most public attention and scientific research have addressed atmospheric deposition of sulfur and nitrogen, commonly referred to as acid rain (Schindler 1988). In the context of ground water, however, acidification most commonly occurs as discharge of acidic ground water from mines to lakes and streams. Waste rocks and tailings often contain high levels of sulfide minerals, such as pyrite, which release sulfuric acid upon oxidation (Blowes and Jambor 1990).

The impact of increased acid input is most strongly felt by lentic systems with little acid-neutralizing capacity (ANC). The ANC of surface water is provided primarily by dissolved silicates and carbonates, and is often controlled by the input of ANC-rich ground water. Webster et al. (1990) observed that the ANC of a small ground water-fed lake in Michigan decreased dramatically during a series of dry years due to reduced ground water input.

Some fresh water bodies are naturally acidic. For example, bogs, which are closed systems receiving only atmospheric water input, are extremely poor in nutrients and are naturally acidic (Mitsch and Gosselink 1986, p. 299). They are composed of peat formed by organic accumulation, notably *Sphagnum* moss species.

Organic acid released from peat binds cations in water and releases hydrogen ions, which enhances acidification. Many bogs are located on raised ground, resulting from massive peat accumulation. Therefore, the water level in bogs is often higher than the surrounding water table, which causes ground water to flow outward and flush the nutrients under bogs and transport them to fens at lower elevations (Siegel et al. 1995).

Acidification impacts aquatic ecosystems in many ways as described by Jeffries and Mills (1990). Most harmful impacts are not caused by acidity itself, but by dissolution of metals under low pH. Several very toxic metals such as mercury and zinc are present in increased concentrations in acid water, but aluminum often presents the largest problem. Dissolved aluminum exists in several forms and the total concentration of aluminum species increases as pH decreases (Appelo and Postma 1993, p. 208). Aluminum interferes with ion regulation and disrupts the important gas-exchange function of fish gills. Aluminum and other metals form complexes with phosphorus and disrupt the nutrient cycling of already oligotrophic systems. As a result, sensitive macrophytes and algae are often replaced by a few species of acid-tolerant algae and moss, and primary productivity is reduced. Metal complexes also remove organic debris and increase water clarity and light penetration. Acidified lakes have enchanting, crystal-clear water, but this represents the sterility and poor diversity of the ecosystem.

Hydrology of the Riparian Zone and Its Ecological Implication

The riparian corridor separating a terrestrial ecosystem from a riverine ecosystem is used as a migration pathway by waterfowl and other bird species while they simultaneously exploit the terrestrial system, riparian zone, and lotic system for food, cover, and nesting habitat (Triska et al. 1993). Riparian zones intercept sediment-laden surface runoff and nutrient-rich ground water before they enter surface water ecosystems (Figure 1). They also deliver organic matter to surface water in the form of litter, an important food source for aquatic invertebrates (Gregory et al. 1991). Streamside trees also modify the solar radiation input to the stream, thereby influencing stream temperature and primary production by photosynthesis.

Riparian vegetation usually requires a shallow water table to maintain high moisture content in the root zone. Common riparian trees, such as poplar and willow, are phreatophytes, which acquire water from the saturated zone below the water table (Robinson 1958). Therefore, riparian vegetation, and hence the riparian zone itself, is highly dependent on ground water. Using stable isotope tracers, Dawson and Ehleringer (1991) demonstrated that riparian trees (oak, maple) on a mountain stream in Utah selectively used ground water even when stream water was readily available. In eastern South Australia, Mensforth et al. (1994) also found that riparian *Eucalyptus* trees selectively used ground water even though the ground water salinity was much higher than stream water. Stromberg et al. (1996) conducted a detailed study of the riparian plant communities in relation to the water-table depth in Arizona. They showed that the abundance of wetland herbs declined sharply as the water-table depth dropped below 0.25 m. Water-table decline caused by ground water exploitation may result in alteration of the riparian ecosystem and loss of biodiversity.

The variability of base flow, and associated changes in temperature and water quality, is a critical factor for the ecology of many fish and invertebrates (van der Kamp 1995). Transpiration by the riparian vegetation causes diurnal fluctuation of stream base flow as reported by numerous workers (Bren 1997), which may affect lotic faunal communities. Transpiration by emergent and floating-leaved aquatic plants also can greatly modify evaporative losses from surface water. For example, in a study of the wetland fringe of an open-water lake, evapotranspiration was 8% to 17% smaller than open-water evaporation depending on the emergent plant species (Burba et al. 1999). However, Wetzel (1999) cites other studies where evapotranspiration exceeded open-water evaporation.

On a larger scale, Carrere (1996) reported that the massive land-use conversion from native rain forests to high-water consumption *Eucalyptus* plantations in Brazil may have caused a severe decline of the water table, which resulted in disappearance of numerous stream ecosystems. In the western United States, phreatophytes have been removed from river floodplains in attempts to salvage water for downstream users. Culler et al. (1982) removed phreatophytes from a reach of the Gila River in Arizona, where phreatophyte consumption prior to eradication was as high as 1090 mm/year. Phreatophyte removal reduced the evapotranspiration on the flood plain by 360 to 480 mm/year. However, this reduction in transpiration produced a minimal increase in annual river flow and was only temporary because replacement vegetation quickly reestablished over most of the river floodplain. A similar study conducted on the Pecos River in New Mexico also resulted in minimal change in river flow (Welder 1988), probably because replacement vegetation transpired an approximately equal volume of water.

Aside from the obvious ecological functions just listed, riparian vegetation on floodplains increases hydraulic roughness during high flow and traps sediments (Tabacchi et al. 2000), thereby affecting fluvial processes that alter habitat distribution. The root network of riparian vegetation increases the mechanical strength of river banks against erosion. Keller et al. (1990) reported a case study of the Carmel River in California, which had a confined channel and lush riparian vegetation in the 1940s. Pumping from the alluvial aquifer underlying the river intensified during the early 1960s and caused a drawdown of the water table up to 10 m. Devegetation of the river bank occurred in response to the low water table, and intense bank erosion resulted in a channel widening from 25 to 65 m over 30 years.

Nitrate Removal by Riparian Vegetation

As mentioned earlier, ground water exchange often is the major pathway of nitrate loading to streams. As ground water flows from the upland to streams and lakes, it passes through the riparian zone. Many workers have noted significant nitrate removal prior to ground water discharge into surface water (Peterjohn and Correll 1984). The aforementioned study of Jordan et al. (1997) compared streams in coastal plains, where ground water is forced to flow above a low permeability clay layer passing through the root zone of riparian vegetation, with streams in the piedmont region, where ground water flows beneath the riparian vegetation before discharging to streams. They found that streams in coastal plains had lower nitrate levels than those in the piedmont region, and attributed this to nitrate removal by riparian vegetation. Hill (1996) suggested that riparian sites most effective in removing nitrate have hydrogeologic settings characterized by permeable surface soils underlain at 1 to 4 m depth by an impermeable layer.

Although many riparian zones effectively remove nitrate from subsurface water, there is considerable uncertainty about the relative importance of the two major removal mechanisms, vegetative uptake and denitrification (Hill 1996). Denitrification is carried out by facultative anaerobic bacteria that use nitrate as an electron acceptor in the absence of oxygen to oxidize organic carbon and obtain energy. Therefore, denitrification requires a continuous supply of organic carbon, anoxic conditions, and nitrate input. Riparian vegetation supplies organic carbon to denitrifying bacteria through litter decomposition and root exudates.

Natural removal of nitrate under riparian vegetation has prompted some workers to apply the same principle to the passive treatment of nitrate-contaminated ground water. Robertson et al. (2000) installed reactive barriers containing organic carbon at four sites in southern Ontario and observed the effective removal of nitrate as ground water flowed through the reactive barriers. Removal of nutrients and retention of suspended sediments by aquatic plants are also commonly used in artificially constructed wetlands to treat sewage from small communities and industrial plants (Kadlec and Knight 1996). This is an actively growing field of environmental engineering research, but little attention has been paid to hydrology, in particular the role of ground water (Choi and Harvey 2000).

Hydroperiod

In the regions where intense runoff occurs in a relatively short period of time, closed topographic depressions of varying sizes are filled by runoff water to form ephemeral ponds or wetlands. Common examples of ephemeral water are playas in arid regions (Scanlon and Goldsmith 1997) and prairie wetlands in semiarid cold regions of North America (Hayashi et al. 1998a). As the water level in a pond occupying a depression rises in response to input from overland flow and streamflow, water flows from the pond to ground water where the adjacent ground water head is lower than the pond (Figure 2b). The duration of standing water in the depression is called the hydroperiod.

The hydroperiod is an important parameter that affects the species richness of aquatic invertebrates, amphibians, and their predators. Snodgrass et al. (2000) studied 22 wetlands on the upper coastal plain of South Carolina, and found that amphibian species richness clearly increased with increased hydroperiod, but found no significant relationship between species richness and wetland size. This finding questions the validity of biased emphasis on large, permanent wetlands by environmental regulatory agencies. From an ecological perspective, small wetlands having intermediate hydroperiod are crucial for biodiversity (Semlitsch and Bodie 1998) because they maintain high productivity by periodic drying resulting in routine recycling of organic materials and nutrients. In cold environments, the surface water of these shallow wetlands warm early and provide food items at a time when the larger and deeper wetlands remain frozen. For this reason, small wetlands are extensively used by dabbling ducks during the early spring (Swanson et al. 1974).

The hydroperiod of ephemeral water is determined by climatic factors (precipitation and evaporation), amount of surface runoff input, and ground water exchange. Trees and shrubs that grow in riparian areas can transpire large amounts of surface and ground water and significantly affect the hydroperiod of ephemeral ponds (Hayashi et al. 1998a). Rosenberry and Winter (1997) found the formation of water-table "troughs" around wetlands in places where ground water normally would discharge to the wetlands. These water-table troughs allowed water to flow out of the wetland, possibly shortening the wetland's hydroperiod. Removal of water by evapotranspiration also impacts the interaction between ground water and surface water in humid settings. In a study of the movement of road salt to a small lake in New Hampshire, ground water normally discharged into a stream that carried salty water from a road to a lake (Rosenberry et al. 1999). During summer, however, evapotranspiration reversed the gradient between stream and ground water, allowing salty stream water to contaminate the ground water beneath the stream.

The total water uptake by riparian vegetation is roughly proportional to the wetted perimeter of a pond and the storage capacity of a pond is roughly proportional to the area of water surface. Therefore, a pond having a high perimeter-to-area ratio has a high rate of water-level recession (Millar 1971) and tends to have a short hydroperiod. Hayashi et al. (1998b) estimated, using a chloride mass-balance technique, that as much as 70% to 80% of summer water loss from prairie wetlands in Saskatchewan, Canada, occurred as infiltration induced by riparian uptake.

In addition to riparian uptake, input and output of regional ground water (Figure 2a) also affects the hydroperiod. Numerous workers (Lissey 1971; LaBaugh et al. 1987) who examined prairie wetlands in relation to landscape setting noted that the wetlands at higher parts of the landscape tend to recharge ground water and have relatively short hydroperiods. In contrast, wetlands at lower parts of the landscape tend to receive ground water discharge and are more permanent. They also noted that recharge wetlands have relatively fresh water and discharge wetlands have saline water, sometimes exceeding the salinity of sea water. The plant community structure in the wetlands sensitively reflects this salinity gradient (Kantrud et al. 1989).

Hyporheic Zone

Active stream channels and underlying sediments frequently exchange water (D in Figure 1). Water in the "hyporheic zone," directly beneath the streambed, is a mixture of surface water and ground water, and is underlain by unmodified ground water with physical and chemical characteristics considerably different from stream water (Williams 1993). Therefore, the hyporheic zone is an ecotone between the surface environment characterized by light, high dissolved oxygen, and temperature fluctuation and the ground water environment characterized by darkness, less oxygen, and stable temperature (Gibert et al. 1994).

Invertebrates living in the hyporheic zone exploit the ground water environment to varying degrees. For example, some species spend their entire lives in ground water (permanent hyporheos), while other species use the hyporheic zone to seek protection from unfavorable situations. Some species spend their egg and larval stage in the hyporheic zone, then move to the surface environment (Gibert et al. 1994). For example, stoneflies in the Flathead River in Montana use both stream and hyporheic zones in their life cycle but need the riparian zone for mating (Stanford and Ward 1993). This study showed that water in the floodplain and active channel interacted frequently, and together served as a subsurface corridor within the landscape through which hyporheos moved up to 2 to 3 km away from the river channel.

The food web of the hyporheic zone is fueled by the heterotrophic microbial communities. These heterotrophs use dissolved oxygen provided by surface water exchange, particulate organic carbon occasionally reburied during floods, and dissolved organic carbon in nutrient-rich ground water (Findlay and Sobczak 2000). Microbes are typically associated with organic particles or biofilms that coat inorganic sediment particles. Microbes provide

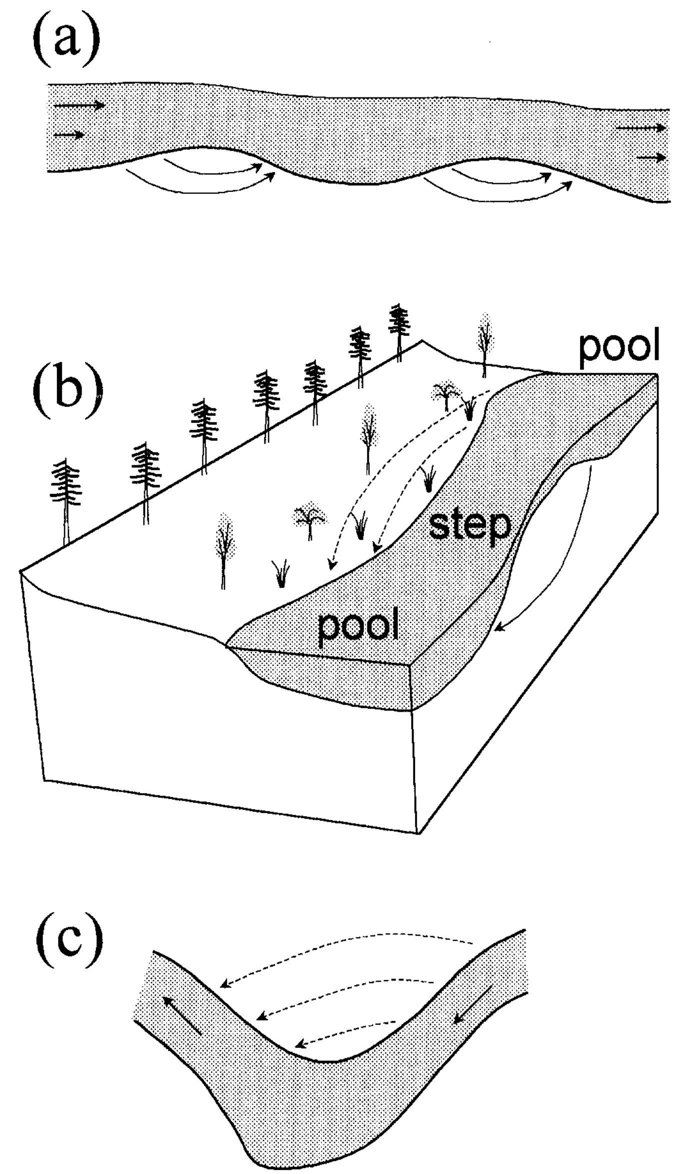

Figure 3. Schematic diagrams of ground water exchange mechanisms. Solid lines indicate ground water flowpaths on vertical cross section, and dashed lines indicate subsurface flow patterns projected on the surface: (a) a vertical cross section showing the flow through dune-like sediments; (b) flow induced by pool-step sequence; (c) lateral flow at an elbow.

food for grazers, which in turn provide food for invertebrate predators. Dissolved organic matter stored in the hyporheic zone can serve as a food resource when it is not readily available in surface water and, therefore, has a critical influence on the metabolism of the fluvial ecosystems (Brunke and Gonser 1997).

Early attempts to describe the spatial extent of hyporheic zones were based on the vertical and lateral distribution of the surface water-derived fauna, which did not give a clear indication of the hyporheic/ground water boundary. This boundary is sometimes better defined by chemical parameters such as alkalinity, nitrate, and dissolved oxygen. For example, Fraser and Williams (1998) showed that the extent of the hyporheic zone under the Speed River in Ontario, Canada, was clearly indicated by alkalinity, which was lower in the hyporheic zone because of surface water mixing than in true ground water.

Surface/ground water exchange in the hyporheic zone is driven by several mechanisms ranging in scale from less than centimeters to several hundred meters. At the smallest scale, ground water downwelling occurs on the upstream face of dune-like sediment structures and upwelling occurs on the downstream face (Figure 3a) because of the pressure distribution across the dune (Packman et al. 2000). Harvey and Bencala (1993) studied ground water exchange along a section of a third-order stream in Colorado, where pools with gradual water surface slopes (< 1%) alternate with steeper channel units (steps) that have a slope of 20% or greater (a pool-step complex). Their tracer experiment showed that stream water flowed to the ground water system at the downstream end of a pool and ground water discharged to the stream at the upstream end of the next pool, bypassing the step between the two pools (Figure 3b). Wroblicky et al. (1998) found that a lateral hyporheic area can occur at the "elbows" of streams where ground water can take a shorter route (Figure 3c). Williams (1993) suggested that gravel bars, rocks, and debris (e.g., logs) that protrude above the general level of the streambed can strongly affect the intermixing of surface and subsurface water.

Ground Water and Fish

The temperature of shallow ground water is very stable relative to surface water and is approximately equal to the average temperature of the ground surface, which is similar to, or a few degrees higher, than the annual mean air temperature. Localized areas of ground water discharge have a stable temperature regime and provide thermal refugia for fish in both winter and summer.

Winter often is a critical period for fish when mortalities are high and stock densities are set by the volumes and suitabilities of winter refugia (Power et al. 1999). Fresh water fish have evolved no physiological mechanism against freezing, and they avoid being trapped in subsurface ice by moving into deeper lentic habitats or to areas influenced by ground water. Ground water discharge areas are often conspicuous in winter because they fail to freeze. In some sections, streamflow becomes subterranean during winter and only small pockets of upwelling ground water are available to fish. In these areas, ground water is essential for winter fish survival (Power et al. 1999). The ecological importance of upwelling was dramatically illustrated on the Chilkat River in Alaska by Keller et al. (1990), where a reach of the river, kept ice-free over the winter by upwelling ground water, attracted a large population of fall-spawning salmon. The weak, spawned-out salmon also provided a critical food source in winter to the world's largest population of American bald eagles.

During summer, ground water discharge areas also provide refuge from excessively warm stream temperatures that may slow growth, presumably because the optimum physiological temperature range has been exceeded (Power et al. 1999). High temperature lowers oxygen solubility and increases susceptibility of fish to bacterial infection (Dunne and Leopold 1978, p. 719). Fish often move long distances to seek the summer refugia offered by ground water or the riparian canopy (Barton et al. 1985).

Ground water also influences the spawning behavior of some fish. Curry and Noakes (1995) studied brook trout redds on the spawning areas on the Canadian Shield and showed that all spawning occurred in areas of ground water upwelling, whether in streams or lakes. They were unable to determine the specific mechanism of site selection, but the value of ground water for successful reproduction may be a stable temperature environment (Power et al. 1999).

Conclusion

The realization that ecological and hydrological settings are interrelated has prompted the coining of a new term to describe this interrelationship, called "ecohydrology" (Wassen and Grootjans 1996). A recent book provides many additional examples from a range of environments on how exchange between ground water and surface water affects interface ecology, and how the biological community affects ground water/surface water exchange (Baird and Wilby 1999). Numerous recent studies are investigating the advantages of this interrelationship. For example, invertebrate communities are used as an indicator of contaminated ground water discharge to surface water (Malard et al. 1996). Plants are being used to indicate areas of ground water discharge to surface water (Goslee et al. 1997; Rosenberry et al. 2000). The boundaries between ecological and hydrological research are gradually dissolving, but a need remains for closer collaboration between these traditionally distinct disciplines.

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References

Appelo, C.A.J., and R. Postma. 1993. *Geochemistry, Groundwater and Pollution*. Rotterdam, Netherlands: Balkema.

Baird, A.J., and R.L. Wilby. 1999. *Eco-Hydrology: Plants and Water in Terrestrial and Aquatic Environments*. New York: Routledge Press.

Barton, D.R., W.D. Taylor, and R.M. Biette. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. *North American Journal of Fisheries Management* 5, 364–378.

Blowes, D.W., and J.L. Jambor. 1990. The pore-water geochemistry and the mineralogy of the vadose zone of sulfide tailings, Waite Amulet, Quebec, Canada. *Applied Geochemistry* 5, 327–346.

Bren, L.J. Effects of slope vegetation removal on the diurnal variations of a small mountain stream. *Water Resources Research* 33, 321–331.

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

Burba, G.G., S.B. Verma, and J. Kim. 1999. A comparative study of surface energy fluxes of three communities (*Phragmites australis, Scirpus acutus*, and open water) in a prairie wetland ecosystem. *Wetlands* 19, 451–457.

Carrere, R. 1996. Pulping the South: Brazil's pulp and paper plantations. *Ecologist* 26, 206–214.

Choi, J., and J.W. Harvey. 2000. Quantifying time-varying ground-water discharge and recharge in wetlands of the northern Florida Everglades. *Wetlands* 20, 500–511.

Culler, R.C., R.L. Hanson, R.M. Myrick, R.M. Turner, and F.P. Kipple. 1982. Evapotranspiration before and after clearing phreatophytes, Gila River flood plain, Graham County, Arizona. U.S. Geological Survey Prof. Paper 655-P.

Curry, A., and D.L.G. Noaks. 1995. Groundwater and the selection of spawning sites by brook trout (*Salvelinus fontinalis*). *Canadian Journal of Fisheries and Aquatic Sciences* 52, 1733–1740.

Dawson, T.E., and J.R. Ehleringer. 1991. Streamside trees that do not use stream water. *Nature* 350, 335–337.

Dunne, T., and L.B. Leopold. 1978. *Water in Environmental Planning*. New York: Freeman.

Eglin, I., U. Roeck, F. Robach, and M. Tremolieres. 1997. Macrophyte biological methods used in the study of the exchange between the Rhine River and the groundwater. *Water Research* 31, 503–514.

Findlay, S., and W.V. Sobczak. 2000. Microbial communities in hyporheic sediments. In *Streams and Ground Waters*, ed. J.B. Jones and P.J. Mulholland, 287–306. San Diego, California: Academic Press.

Fraser, B.G., and D.D. Williams. 1998. Seasonal boundary dynamics of a groundwater/surface-water ecotone. *Ecology* 79, 2019–2031.

Gibert, J., J.A. Stanford, M.-J. Dole-Olivier, and J.V. Ward. 1994. Basic attributes of groundwater ecosystems and prospects for research. In *Groundwater Ecology*, ed. J. Gibert, D.L. Danielopol, and J.A. Stanford, 7–40. San Diego, California: Academic Press.

Goslee, S.C., R.P. Brooks., and C.A. Cole. 1997. Plants as indicators of wetland water source. *Plant Ecolology* 131, 199–206.

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

Harvey, J.W., and K.E. Bencala. 1993. The effect of streambed topography on surface-subsurface water exchange in mountain catchments. *Water Resources Research* 29, 89–98.

Hayashi, M., G. van der Kamp, and D.L. Rudolph. 1998a. Water and solute transfer between a prairie wetland and adjacent uplands, 1. Water balance. *Journal of Hydrology* 207, 42–55.

Hayashi, M., G. van der Kamp, and D.L. Rudolph. 1998b. Water and solute transfer between a prairie wetland and adjacent uplands, 2. Chloride cycle. *Journal of Hydrology* 207, 56–67.

Hill, A.R. 1996. Nitrate removal in stream riparian zones. *Journal of Environmental Quality* 25, 743–755.

Jeffries, M.J., and D.H. Mills. 1990. *Freshwater Ecology: Principles and Applications*. London: Belhaven Press.

Jordan, T.E., D.L. Correll, and D.E. Weller. 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. *Water Resources Research* 33, 2579-2590.

Kadlec, R.H., and R.L. Knight. 1996. *Treatment Wetlands*. Boca Raton, Florida: Lewis Publishers.

Kantrud, H.A., J.B. Millar, and A.G. van der Valk. 1989. Vegetation of wetlands of the prairie pothole region. In *Northern Prairie Wetlands*, ed. A.G. van der Valk. Ames, Iowa: Iowa State University Press.

Keller, E.A., G.M. Kondolf, and D.J. Hagerty. 1990. Groundwater and fluvial processes: Selected observations. Geological Society of America Special Paper 252, 319–340.

Klijn, F., and J.-P.M. Witte. 1999. Eco-hydrology: Groundwater flow and site factors in plant ecology. *Hydrogeology Journal* 7, 65–77.

LaBaugh, J.W., T.C. Winter, V.A. Adomaitis, and G.A. Swanson. 1987. Hydrology and chemistry of selected prairie wetlands in the Cottonwood Lake area, Stutsman County, North Dakota, 1979-82. U.S. Geological Survey Prof. Paper, 1431.

LaBaugh, J.W., T.C. Winter, G.A. Swanson, D.O. Rosenberry, R.D. Nelson, and N.H. Euliss Jr. 1996. Changes in atmospheric circulation patterns affect midcontinent wetlands sensitive to climate. *Limnology and Oceanography* 41, 864–870.

Lissey, A. 1971. Depression-focused transient groundwater flow patterns in Manitoba. Geological Association of Canada Special Paper 9, 333–341.

Lodge, D.M., D.P. Krabbenhoft, and R.G. Striegl. 1989. A positive relationship between groundwater velocity and submersed macrophyte biomass in Sparkling Lake, Wisconsin. *Limnology and Oceanography* 34, 235–239.

Malard, F., S. Plenet, and J. Gibert. 1996. The use of invertebrates in ground water monitoring: A rising research field. *Ground Water Monitoring & Remediation* 16, no. 1: 103–113.

Mensforth, L.J., P.J. Thorburn, S.D. Tyerman, and G.R. Walker. 1994. Sources of water used by riparian *Eucalyptus camaldulensis* overlying highly saline groundwater. *Oecologia* 100, 21–28.

Millar, J.B. 1971. Shoreline-area ratio as a factor in rate of water loss from small sloughs. *Journal of Hydrology* 14, 259–284.

Mitsch, W.J., and J.G. Gosselink. 1986. *Wetlands*. New York: Van Nostrand Reinhold.

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 bed forms. *Water Resources Research* 36, 2351–2361.

Peterjohn, W.T., and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology* 65, 1466–1475.

Power, G., R.S. Brown, and J.G. Imhof. 1999. Groundwater and fish—Insights from northern North America. *Hydrological Processes* 13, 401–422.

Pringle, C.M., and F.J. Triska. 2000. Emergent biological patterns and surface-subsurface interactions at landscape scales. In *Streams and Ground Waters*, ed. J.B. Jones and P.J. Mulholland, 167–193. San Diego, California: Academic Press.

Robertson, W.D., J.A. Cherry, and E.A. Sudicky. 1991. Ground-water contamination from two small septic systems on sand aquifers. *Ground Water* 29, no. 1: 82–92.

Robertson, W.D., S.L. Schiff, and C.J. Ptacek. 1998. Review of phosphate mobility and persistence in 10 septic system plumes. *Ground Water* 36, no. 6: 1000–1010.

Robertson, W.D., D.W. Blowes, C.J. Ptacek, and J.A. Cherry. 2000. Long-term performance of in situ reactive barriers for nitrate remediation. *Ground Water* 38, no. 5: 689–695.

Robinson, T.W. 1958. Phreatophytes. U.S. Geological Survey Water-Supply Paper 1423.

Rosenberry, D.O., P.A. Bukaveckas, D.C. Buso, G.E. Likens, A.M. Shapiro, and T.C. Winter. 1999. Migration of road salt to a small New Hampshire lake. *Water, Air, and Soil Pollution* 109, 179–206.

Rosenberry, D.O., and T.C. Winter. 1997. Dynamics of water-table fluctuations in an upland between two prairie-pothole wetlands in North Dakota. *Journal of Hydrology* 191, 266–289.

Rosenberry, D.O, R.G. Striegl, and D.C. Hudson. 2000. Plants as indicators of focused ground water discharge to a northern Minnesota lake. *Ground Water* 38, no. 2: 296–303.

Sand-Jensen, K., C. Prahl, and H. Stokholm. 1982. Oxygen release from roots of submerged aquatic macrophytes. *Oikos* 38, 349–354.

Scanlon, B.R., and R.S. Goldsmith. 1997. Field study of spatial variability in unsaturated flow beneath and adjacent to playas. *Water Resources Research* 33, 2239–2252.

Schindler, D.W. 1977. Evolution of phosphorus limitation in lakes. *Science* 195, 260–262.

Schindler, D.W. 1988. Effects of acid rain on freshwater ecosystems. *Science* 239, 149–157.

Semlitsch, R., and J.R. Bodie. 1998. Are small, isolated wetlands expendable? *Conservation Biology* 12, 1129–1133.

Shaw, R.D., and E.E. Prepas. 1990. Groundwater-lake interactions: I. Accuracy of seepage meter estimates of lake seepage. *Journal of Hydrology* 119, 105–120.

Siegel, D.I., A.S. Reeve, P.H. Glaser, and E.A. Romanowicz. 1995. Climate-driven flushing of pore water in peatlands. *Nature* 374, 531–533.

Sklash, M.G., and R.N. Farvolden. 1979. The role of groundwater in storm runoff. *Journal of Hydrology* 43, 45–65.

Snodgrass, J.W., M.J. Komoroski, A.L.J. Bryan, and J. Burger. 2000. Relationships among isolated wetland size, hydroperiod, and amphibian species richness: Implication for wetland regulations. *Conservation Biology* 14, 414–419.

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

Stromberg, J.C., R. Tiller, and B. Richter. 1996. Effects of groundwater decline on riparian vegetation of semiarid regions: The San Pedro, Arizona. *Ecological Applications* 6, 113–131.

Swanson, G.A., M.I. Meyer, and J.R. Serie. 1974. Feeding ecology of breeding blue-winged teals. *Journal of Wildlife Management* 38, 396–407.

Tabacchi, E., L. Lambs, H. Guilloy, A.-M. Planty-Tabacchi, E. Muller, and H. Decamps. 2000. Impacts of riparian vegetation on hydrological processes. *Hydrological Processes* 14, 2959–2976.

Toth, J. 1963. A theoretical analysis of groundwater flow in small drainage basins. *Journal of Geophysical Research* 68, 4795–4812.

Triska, F.J., J.H. Duff, and R.J. Avanzino. 1993. The role of water exchange between a stream channel and its hyporheic zone in nitrogen cycling at the terrestrial-aquatic interface. *Hydrobiologia* 251, 167–184.

van der Kamp, G. 1995. The hydrology of springs in relation to the biodiversity of spring fauna: A review. *Journal of the Kansas Entomological Society* 68, 4–17.

Ward, J.V. 1989. The four-dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society* 8, 2–8.

Wassen, M.J., and A.P. Grootjans. 1996. Ecohydrology: An interdisciplinary approach for wetland management and restoration. *Vegetatio* 126, 1–4.

Webster, K.E., A.D. Newell, L.A. Baker, and P.L. Brezonik. 1990. Climatically induced rapid acidification of a softwater seepage lake. *Nature* 347, 374–376.

Welder, G.E. 1988. Hydrologic effects of phreatophyte control, Acme-Artesia reach of the Pecos River, New Mexico, 1967-82. U.S. Geological Survey Water-Resources Investigations Report 87-4148.

Wetzel, R.G. 1999. Plants and water in and adjacent to lakes. In *Eco-Hydrology: Plants and Water in Terrestrial and Aquatic Environments*, ed. A.J. Baird and R.L. Wilby, 269–299. New York: Routledge.

Wheeler, B.D. 1999. Water and plants in freshwater wetlands. In *Eco-Hydrology: Plants and Water in Terrestrial and Aquatic Environments*, ed. A.J. Baird and R.L. Wilby, 127–180. New York: Routledge.

Williams, D.D. 1993. Nutrient and flow vector dynamics at the hyporheic/groundwater interface and their effects on the interstitial fauna. *Hydrobiologia* 251, 185–198.

Winter, T.C., J.W. Harvey, O.L. Franke, and W.M. Alley. 1998. Ground water and surface water a single resource. U.S. Geological Survey Circular 1139.

Woo, M.-K., and T.C. Winter. 1993. The role of permafrost and seasonal frost in the hydrology of northern wetlands in North America. *Journal of Hydrology* 141, 5–31.

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 Resources Research* 34, 317–328.