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From submarine to lacustrine groundwater discharge

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Abstract Submarine groundwater discharge (SGD) and its role in marine nutrient cycling are well known since the last decade. The freshwater equivalent, lacustrine groundwater discharge (LGD), is often still disregarded, although first reports of LGD are more than 50 years old. We identify nine different reasons why groundwater has long been disregarded in both freshwater and marine environments such as invisibility of groundwater discharge, the size of the interface and its difficult accessibility. Although there are some fundamental differences in the hydrology of SGD and LGD, caused primarily by seawater recirculation that occurs only in cases of SGD, there are also a lot of similarities such as a focusing of discharge to near-shore areas. Nutrient concentrations in groundwater near the groundwater–surface water interface might be anthropogenically enriched. Due to spatial heterogeneity of aquifer characteristics and biogeochemical processes, the quantification of groundwater-borne nutrient loads is challenging. Both nitrogen and phosphorus might be mobile in near-shore aquifers and in a lot of case studies large groundwater-borne nutrient loads have been reported.

Key words surface water; groundwater; lake; nutrient budget; water budget; submarine groundwater discharge; lacustrine groundwater discharge

INTRODUCTION

Groundwater exfiltration into marine systems, is called submarine groundwater discharge (SGD) in the literature and defined as "any and all flow of water on continental margins from the seabed to the coastal ocean, regardless of the fluid composition or driving force" (Burnett *et al.* 2003). The freshwater equivalent is called lacustrine groundwater discharge (LGD) (Lewandowski *et al.* 2013). LGD includes any and all flow of groundwater from the lakebed to the lake.

Eutrophication – the over-enrichment of water by nutrients such as nitrogen (N) and phosphorus (P) – is a major cause of water quality impairment for both freshwaters and marine waters. Anthropogenic eutrophication has been attributed to the rapid increase of intensive agriculture (fertilizer and manure), industrial activities and population growth (Conley *et al.* 2009). Eutrophication of surface freshwater systems due to nutrient rich LGD is an emerging topic, since its relative impact on nutrient budgets of surface waters increases with decreasing inputs of nutrients from point sources.

As a basis to conduct effective management measures, water and nutrient budgets are required with all input paths quantified. Theoretically, a budget is quite simple: the sum of all inputs/sources minus the sum of all losses/sinks should equal changes in storage. Important input paths of water and nutrients include overland runoff, inflows of streams and rivers, precipitation onto the lake or ocean's water surface, dry atmospheric deposition (in the case of the nutrient budget), and groundwater discharge – LGD or SGD, respectively. Important losses are surface-water outflows (applies to lakes only), evaporation from the open water surface (predominantly water budget), and groundwater recharge through the lake or ocean bed. Although the latter might be ecologically relevant in some cases, the focus here is on groundwater discharge. However, reliable quantifications of most components of the water and nutrient budget are a challenge, especially with regard to temporal variations and spatial heterogeneities.

The aim here is to compare SGD and LGD, to present some reasons why groundwater discharge has long been overlooked, to give insights into the pertinent hydrological processes, to list available measurement techniques, and to describe the biogeochemical processes controlling nitrogen and phosphorus loads.

REASONS WHY GROUNDWATER HAS BEEN DISREGARDED IN WATER AND NUTRIENT BUDGETS

Groundwater has often been disregarded in water and nutrient budgets of lakes and oceans due to several different reasons:

(a) Groundwater and surface water have long been considered as separate entities.
(b) Groundwater discharge is invisible (except in the case of springs) and thus people are less aware of groundwater discharge compared to surface inflows, overland flow or precipitation.
(c) The interface is difficult to access, especially in deep lakes and coastal oceans.
(d) Due to the large extent of the interface, local discharge rates are usually small.
(e) Although a lot of different measurement techniques for LGD and SGD were developed in the last decades, there is still a lack of simple or convenient methodology.
(f) Spatial heterogeneity of discharge rates and groundwater composition result in large numbers of measurements required for reliable determinations of the LGD component in budgets.
(g) Temporal variability of discharge rates add to the tremendous challenge of reliable estimations of SGD and LGD fluxes.
(h) In lake settings the net groundwater component is often small leading to the assumption that it can be ignored – even if gross inflows and gross losses are large. Nutrient concentrations in inflowing groundwater might be orders of magnitude higher than those in outflowing water or in resident lake and ocean water. Thus, groundwater might be more important in nutrient budgets than in water budgets.
(i) The interface is often considered a reactive interface. Thus, concentrations determined close to the shore might differ from concentrations actually entering the lake or coastal ocean since chemical processing might occur in the last decimetre of sediment before entering the surface water body.

HYDROLOGY

The rate of groundwater flow from the catchment to either a lake or coastal ocean depends on the hydraulic gradient and hydraulic conductivity in the porous medium adjacent to the lake or ocean. Since there are some fundamental differences in groundwater flow toward lake interfaces and ocean interfaces, we describe both interfaces here separately.

Lacustrine groundwater discharge (LGD)

In simple settings where geology is uniform, exchange may be focused along the near-shore margins. Several authors observed an exponential decrease of discharge with increasing distance to the shore line. This was found both in modelling studies and during *in situ* measurements (McBride and Pfannkuch 1975, Pfannkuch and Winter 1984). Flow lines of groundwater approaching the interface bend upwards (but will never cross each other) since groundwater will always flow in the direction of the largest gradient (McCobb *et al.* 2003). As a consequence, a vertical distribution of different groundwater composition occurring in the aquifer is projected more or less horizontally onto the lakebed, but is also commonly compressed close to the shore and extended with increasing distance from the shore. Focusing of LGD to near-shore zones is conducive to measurement of LGD since shallow near-shore areas are commonly more easily accessible (e.g. by wading). The focusing of LGD in near-shore areas is valid for more or less homogenous aquifers. Aquifers with zones of high hydraulic conductivity will exhibit high groundwater discharge rates where preferential flow paths enter the lake. Lakes in fractured-rock settings will often exhibit highly localized LGD focused to fractures. Lakes in contact with more than one aquifer may have large groundwater discharge rates; i.e. in addition to near-shore areas LGD can occur directly below the depth where the aquitard separating the two aquifers intersects the lake.

Besides modelling to quantify LGD, LGD can be identified and measured by several techniques that can be grouped into: (1) local point measurements, (2) integrating methods, and (3) methods of pattern identification (Rosenberry and LaBaugh 2008, Lewandowski *et al.* 2013).

Point methods:

(a) Seepage meters are basically cylinders with an attached plastic bag that collects the discharging groundwater.

(b) Temperature–depth profiles in the lake bed make use of the natural temperature difference between groundwater and lake water. LGD rates are calculated from the curvature of the temperature depth profile.

(c) Depth profiles of conservative ions in the lake bed can be used if concentrations in the groundwater approaching the interface are different from lake water concentrations. Ion–depth profiles are evaluated analogously to temperature–depth profiles.

(d) Piezometers installed at a single point but at variable depths in the lake bed (vertical flow), piezometers installed along a line perpendicular to the shoreline (horizontal flow), or installation of several groundwater observation wells in the catchment provide data allowing LGD rates to be calculated from hydraulic gradients based on Darcy's law. Estimates of hydraulic conductivity data also are required.

(e) LGD can be determined with injection of artificial tracers (e.g. salt, fluorescent dye).

Integrating methods:

(a) Calculations of the mean annual groundwater recharge in the lake's subsurface catchment should by definition equal the amount of mean annual LGD. In cases where groundwater discharges to springs or groundwater-fed streams are located in the catchment these volumes have to be subtracted from annual LGD.

(b) Quantifying all of the easier-to-measure components of a lake water budget, and solving for the groundwater component as a residual, is a relatively simple concept. However, the errors of all components are included in the groundwater component and the method determines only the net groundwater component.

(c) Mass balances of conservative ions such as chloride, stable isotopes in the water molecule, or radon can also be used to refine estimates of LGD in settings where all other inputs and losses are known.

Pattern identification might be the best approach for making point measurements where LGD rates are particularly large, since point measurements are usually very labour- and time-consuming. Temperature is a useful natural tracer for this purpose.

(a) Distributed temperature sensing (DTS) makes use of localized temperature anomalies at the sediment–water interface caused by LGD. Several hundred to thousand metre long fibre-optic cables are distributed across the sediment–water interface and temperatures are measured with a spatial resolution of 0.25 to 2 m along the cable. The broad areal coverage provided by the fibre-optic cable allows DTS to detect thermal anomalies associated with varying rates of LGD. Alternatively, if natural temperature differences between groundwater and surface water are insufficient, a heated DTS can be used and various rates of heat dissipation can be related to varying rates of groundwater discharge.

(b) Discharging groundwater will float on the lake surface under certain circumstances and can be measured as a temperature anomaly at the surface with airborne thermal infrared (TIR) imagery (Lewandowski *et al.* 2013).

The groundwater component dominates some lake water budgets. For example, LGD represents 74% of all inflows to Williams Lake, Minnesota (LaBaugh *et al.* 1997), 94% to Mary Lake, Minnesota (Stets *et al.* 2010), 90% to Cliff Lake, Montana (Gurrieri and Furniss 2004) and 85% to Lake Annie, Florida (Sacks *et al.* 1998). High LGD rates are reported for several lakes, for example 477 L m^{-2} d^{-1} for Ashumet Pond, Massachusetts (McCobb *et al.* 2009), 155 L m^{-2} d^{-1} for Dickson Lake, Ontario (Ridgeway and Blanchfield 1998), and 138 L m^{-2} d^{-1} Shingobee Lake, Minnesota (Rosenberry *et al.* 2000); all rates are point measurements by seepage meters.

Submarine groundwater discharge (SGD)

The total flux of SGD to the Atlantic Ocean is similar in volume to the amount of riverine discharge into the ocean, but since nutrient concentrations in groundwater are often higher than in stream water, SGD is probably even more important in oceanic budgets than surface water inflows (Moore 1996, Moore and Church 1996, Burnett *et al.* 2006, Moore *et al.* 2006, 2008).

A major difference between SGD and LGD is that some deep seawater is recirculated upwards in coastal zones (up to 90%) and thus included in SGD. Topography-driven discharge of fresh groundwater essentially “drags” saline water from the underlying saline groundwater body (Slomp and Van Cappellen 2004). Furthermore, tides and waves cause additional seawater recirculation in the aquifer. Similar to LGD there is a focusing of discharge in near-shore areas and anomalous rates of groundwater discharge still depend on aquifer heterogeneity.

In principal the same measurement methods as described for LGD can be used to quantify SGD. For example, the point-measurement methods can be conducted in marine settings, just as in limnetic settings. However, some of the integrating approaches cannot work. The water-budget approach is not practical for marine settings, with the possible exception of determining groundwater discharge on a global scale. One popular integrating approach that works well in marine settings, but that is not appropriate in limnetic settings, is a radium mass balance; radium is only mobile in the presence of saline water. Thus, this method is useful to distinguish between the freshwater component and the saline component of SGD. For pattern identification the methods described for LGD can be applied for SGD determination as well. The airborne study of SGD with TIR imaging is much easier in marine settings since the fresher, less dense groundwater will always float on top of the sea water, while a complicated set of prerequisites is required in limnetic settings (Lewandowski *et al.* 2013).

BIOGEOCHEMISTRY

Methods used to quantify nutrient fluxes

LGD-derived nutrient loads are commonly calculated by multiplying the volume of groundwater discharge by the concentration of the discharging groundwater. Since there is much spatial heterogeneity of both discharge volumes and nutrient concentrations, approaches for segmenting the area of interest are required (Meinikmann *et al.* 2013). Options for determining volumes of groundwater discharge are described above. For the determination of nutrient concentrations a range of groundwater sampling strategies can be applied:

(a) Near-shore groundwater wells are used to characterize the groundwater approaching the aquifer–surface water interface.

(b) Since installation of groundwater wells with a wheel-mounted drilling rig can be expensive, and near-shore groundwater levels are usually close to the land surface, hand-drilled near-shore piezometers are also used to sample the uppermost groundwater.

(c) Multi-level samplers are additionally used to determine the groundwater composition where high vertical resolution is required.

(d) Groundwater samples can be collected very close to the point of entry with piezometers and multi-level samplers installed in the sea- or lakebed. Due to the reactivity of the interface this approach provides data that more closely approximate the water that actually moves across the sediment–water interface to enter the surface-water body.

(e) Finally, samples from seepage meters can be used to collect the discharging groundwater and determine its composition provided that efforts are made to minimize water residence time within the seepage cylinder and seepage-collection bag (Rosenberry and LaBaugh 2008).

Nitrogen

Natural systems have become artificially enriched in N (Ibanhez *et al.* 2011) and nitrate contamination is a common problem world-wide (Wakida and Lerner 2005). Nitrogen in the aquifer often originates from fertilizers (agriculture, urban lawns and golf courses), atmospheric deposition, livestock and sewage. Excess applications of fertilizer and manure have resulted in very large N concentrations in some aquifers world-wide. Acid rain and dry atmospheric N deposition result in an additional N contamination of soils and subsequently of aquifers. Techniques for sewage disposal vary widely by region and culture; in many cases, treated sewage with substantial N loads reaches the aquifer either accidentally or intentionally (e.g. de-centralized sewage pits, sewage infiltration beds, on-site sewage treatment systems with drain fields, and leaky sewers; Wakida and Lerner 2005).

Nitrogen in the aquifer occurs in the forms of nitrate, nitrite, ammonium and dissolved organic nitrogen. Nitrate is highly mobile in oxic aquifers but is denitrified and converted to gaseous N_2 under anoxic conditions. Ammonium is less mobile and occurs under anoxic conditions. Under oxic conditions, ammonium is nitrified to nitrate (Slomp and Van Cappellen 2004). Dissolved organic nitrogen is less well studied, although it occurs in high concentrations in some aquifers.

High concentrations of nitrogen in groundwater approaching the interface with surface water can result in high N loads, e.g. LGD: 641 g m^{-2} $year^{-1}$ in Colgada Lake, Spain (whole lake, Pina-Ochoa and Alvarez-Cobelas 2009), 453 and 456 g m^{-2} $year^{-1}$ in two different gravel pit lakes, Austria (whole lake, Weilhartner *et al.* 2012, Muellegger *et al.* 2013); SGD: 25 580 g m^{-2} $year^{-1}$ at Badum site, Peniscola, Castello, Spain (area of SGD plume, about 20 km^2, Garcia-Solsona *et al.* 2010), 2685 g m^{-2} $year^{-1}$ West Coast Mauritius (25 m broad near-shore zone, Paytan *et al.* 2006), 2880 g m^{-2} $year^{-1}$ in Loxahatchee River estuary, Florida (whole estuary, Swarzenski *et al.* 2006), and 767 g m^{-2} $year^{-1}$ in Waquoit Bay, Massachusetts (0.8 m broad beach face, Spiteri *et al.* 2008).

Phosphorus

Phosphorus concentrations in pristine groundwater are low (<50 µg P L^{-1}). Analogous to N, groundwater-P might be increased due to fertilizers, atmospheric deposition, manure and sewage (500 to 5000 µg P L^{-1}; Wendland *et al.* 2005; Kunkel *et al.* 2005). The mineralization of naturally present organic matter might also increase P concentrations. Phosphorus itself occurs only in the oxidation state P(+V). However, its binding partners are redox sensitive. Thus, reducing conditions favour the transport of P in the aquifer, but even under reducing conditions there is some retardation compared to groundwater flow velocities.

In the last century P in an aquifer was considered as immobile, consequently P concentrations in groundwater were assumed negligible and P transfer via LGD received little attention (Vanek 1987, Cherkauer *et al.* 1992, Kilroy and Coxon 2005). However, some researchers have disproved this statement and coined the statement "Phosphate does not migrate in groundwater – Better think again" (http://toxics.usgs.gov/highlights/phosphorous_migration.html, access: 25 September 2013). A major difference between marine and lake systems is that N is commonly the limiting nutrient in marine systems while P is commonly considered the limiting nutrient in freshwater settings.

Phosphorus concentrations in groundwater approaching the interface might be high resulting in high P loads, e.g. LGD: 2.3 g m^{-2} $year^{-1}$ in Lake Bysjön, Sweden (whole lake, Vanek 1993), 1.1 g m^{-2} $year^{-1}$ in Sparkling Lake, Wisconsin (mean for two sites with high LGD, Hargerthey and Kerfoot 1998), and 0.4 g m^{-2} $year^{-1}$ in Fishermans Cove of Ashumet Pond, Cape Cod, Massachusetts (whole lake, McCobb *et al.* 2003); SGD: 333.7 g m^{-2} $year^{-1}$ at Badum site, Peniscola, Castello, Spain (area of SGD plume, about 20 km^2, Garcia-Solsona *et al.* 2010), 58.7 in Okatee River Estuary, South Carolina (whole estuary, Moore *et al.* 2006), and 20.4 g m^{-2} $year^{-1}$ in Waquoit Bay, Massachusetts (0.8 m broad beach face, Spiteri *et al.* 2008). Note that P loads imported by SGD and LGD are about two orders of magnitude lower than N loads.

CONCLUSION AND OUTLOOK

Groundwater has long been disregarded in water and nutrient budgets of lakes and oceans. Following some key publications about SGD at the beginning of the new millennium there has been a dramatic increase in publications about SGD and groundwater-borne nutrient loads reaching the coastal areas of the world's oceans (Moore 2010). With this, there has also been a significant shift in the awareness of the role of groundwater-borne nutrients for the ocean's nutrient budgets. It is nowadays well accepted that the role of submarine groundwater discharge is at least in the same order of magnitude as discharge to the oceans from rivers and streams. In contrast, there is still less awareness of the role of LGD, particularly as it applies to lake nutrient budgets, even though the first studies of LGD date back to 1940 (Hubbert 1940, Harvey *et al.* 2000, Bowen *et al.* 2007). Groundwater can be a relevant component in the water and nutrient budgets, even in some large lakes (e.g. Lake Tahoe, Loeb and Goldman 1979), and can be a very large component in lakes with small surface-water inflows.

The quantification of volumes and loads remains a challenge both in LGD and SGD. Further development of measurement methods is required. Other open questions address the reactivity of the aquifer–surface water interface. The importance of the processing of nutrients and whether that results in a delayed transport of nutrients to the receiving water bodies, or perhaps permanent removal, has largely yet to be determined.

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