



RESEARCH ARTICLE

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Nutrient processes at the stream-lake interface for a channelized versus unmodified stream mouth

Richard Niswonger ¹, Ramon Naranjo ², David Smith², Jim Constantz ¹, Kip Allander ², Donald Rosenberry ³, Bethany Neilson ⁴, Michael R. Rosen ⁵, and David Stonestrom ¹

¹National Research Program, U.S Geological Survey, Menlo Park, California, USA, ²Nevada Water Science Center, U.S Geological Survey, Carson City, Nevada, USA, ³National Research Program, U.S. Geological Survey, Denver, Colorado, USA, ⁴Civil and Environmental Engineering, Utah State University, Logan, Utah, USA, ⁵Water Science Field Team, U.S. Geological Survey, Carson City, Nevada, USA

Key Points:

- The stream-lake interface (SLI) is a unique hydrologic environment for biogeochemical processing of nutrients
- The SLI is often channelized and this reduces nutrient concentrations in groundwater discharging to the nearshore lake region
- Inorganic nutrients are supplied to the nearshore benthic zone by mineralization of organic deposits in barrier beaches

Supporting Information:

- Supporting Information S1

Correspondence to:

R. Niswonger, rniswon@usgs.gov

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Abstract Inorganic forms of nitrogen and phosphorous impact freshwater lakes by stimulating primary production and affecting water quality and ecosystem health. Communities around the world are motivated to sustain and restore freshwater resources and are interested in processes controlling nutrient inputs. We studied the environment where streams flow into lakes, referred to as the stream-lake interface (SLI), for a channelized and unmodified stream outlet. Channelization is done to protect infrastructure or recreational beach areas. We collected hydraulic and nutrient data for surface water and shallow groundwater in two SLIs to develop conceptual models that describe characteristics that are representative of these hydrologic features. Water, heat, and solute transport models were used to evaluate hydrologic conceptualizations and estimate mean residence times of water in the sediment. A nutrient mass balance model is developed to estimate net rates of adsorption and desorption, mineralization, and nitrification along subsurface flow paths. Results indicate that SLIs are dynamic sources of nutrients to lakes and that the common practice of channelizing the stream at the SLI decreases nutrient concentrations in pore water discharging along the lakeshore. This is in contrast to the unmodified SLI that forms a barrier beach that disconnects the stream from the lake and results in higher nutrient concentrations in pore water discharging to the lake. These results are significant because nutrient delivery through pore water seepage at the lakebed from the natural SLI contributes to nearshore algal communities and produces elevated concentrations of inorganic nutrients in the benthic zone where attached algae grow.

1. Introduction

Lakes are an important component of the world’s freshwater supply that need to be protected and managed to avoid cultural eutrophication [Cooke *et al.*, 2016]. In nutrient limited lakes, excessive amounts of dissolved inorganic forms of nitrogen (DIN) and phosphorous (DIP) can stimulate primary production and accelerate eutrophication, and thus management of lakes is often focused on identifying and reducing sources of DIN and DIP [Schindler, 2006; Hale *et al.*, 2015]. Sources of nutrients from watersheds to lakes include agricultural, residential, and industrial development; atmospheric deposition; and natural biogeochemical processing within vegetated soils, riparian buffers, and hyporheic zones [Dillon and Kirchner, 1975; Schuster and Grismer, 2003; Seitzinger *et al.*, 2006; McMahon and Böhlke, 1996; McClain *et al.*, 2003; Gergans *et al.*, 2011; Galloway *et al.*, 2008; Michalak *et al.*, 2013; Naranjo *et al.*, 2015; Harvey and Gooseff, 2015]. Nutrient loads from surface water inflows to lakes typically are much greater than groundwater inflows, and more research has focused on nutrient inputs to lakes from surface water as compared to groundwater. However, there has been growing interest in the role groundwater plays in water and nutrient budgets in lakes [Wachniew, and Rózański, 1997; LaBaugh *et al.*, 1995; Rosenberry *et al.*, 2015; Lewandowski *et al.*, 2015, Prouty *et al.*, 2016].

Lakes in alpine and subalpine regions typically support nutrient-limited ecosystems in which primary production is sensitive to subtle changes in nutrient inputs, both spatially and temporally [Loeb, 1986]. Changes in primary production in lakes can negatively impact water quality, water clarity, and ecosystem function [Goldman, 1988]. The conditions and timing of nutrient delivery also can be an important factor, especially when the timing of DIN and DIP delivery corresponds to algal growth-limitation by these nutrients [Michalak *et al.*, 2013]. Nutrient-rich groundwater discharge can provide an efficient source of nutrients for benthic

algae as concentrations at the sediment interface will be higher relative to the more dilute adjoining water column [Hagerthey and Kerfoot, 1998; Rosen, 2015; Prouty et al., 2016; Anderson et al., 2002]. As with marine nearshore environments, the sediment-water interface near the lakeshore is known to host active microbial communities that generate reactive forms of nutrients [Correll, 1998; Gumbricht, 1993].

A recent study focusing on the immediate region where streams discharge to freshwater lakes reveals unique physical and hydrologic characteristics expected to affect nutrient cycling [Constantz et al., 2016]. These regions, referred to herein as the stream-lake interface (SLI), had not generally been distinguished from the broader stream or lake environment. The study presented results from two SLIs in the Lake Tahoe basin, Nevada, demonstrating distinctive differences in geomorphology, surface-water and groundwater (SW-GW) interactions, and other water and sediment properties for the SLI relative to both the upgradient stream and other lake shore areas. Given well-established linkages of SW-GW interactions and stream geomorphology, including dunes, riffles, pools, and parafluvial features that have been shown to control biogeochemical processes, there is good reason to hypothesize that an SLI's unique geomorphic features also would enhance biogeochemical processes that affect fluxes to lakes and their nearshore ecosystems [Holmes et al., 1994; Harvey and Fuller, 1998; Baxter and Hauer, 2000; Kasahara and Wondzell, 2003; Rahimi et al., 2015].

The present work embodies an ongoing effort to understand contributions of nutrient sources from groundwater and hyporheic exchange to the eutrophication of lakes [Meinikmann et al., 2015; Kidmose et al., 2015; Rosenberry et al., 2015; Périllon and Hilt, 2016]. In the past several decades, the impacts of cultural-climatic eutrophication have been increasingly apparent in significant changes to Lake Tahoe and other large lakes [Goldman, 1988; Kemp et al., 2005; Carpenter, 2008]. Hackley et al. [2013, 2016] reported increases in periphyton biomass blooms over the last decade at 0.5 m in nearshore regions of Lake Tahoe, despite successful management activities that have reduced anthropogenic sources of nutrients to the lake [Schuster and Grismer, 2004]. These increases in periphyton biomass suggest that other factors or sources of nutrients are affecting periphyton biomass in the nearshore regions of the lake, including possibly local sources of DIN and DIP. Alternative explanations of increased periphyton biomass include increased fluxes of nutrients in atmospheric deposition, and biogeochemical production of nutrients in soils that are delivered to the nearshore region in surface water and groundwater [Coats et al., 1976]. Identification of triggers for algal blooms is an important on-going research topic in Lake Tahoe and other lakes around the world [Bennett et al., 2001; Baker et al., 2014; Kleinman et al., 2015; Hackley et al., 2016].

Our research objective was to investigate the SLI as a unique hydrologic environment for biogeochemical processes affecting sources of nutrients to lakes. Establishing cause and effect relationships between SLIs and primary production in the nearshore area is beyond the scope of the present work. Instead we focus on nutrient processing in the SLI to provide observation-constrained modeling results on the importance of these near-shore environments as potential sources of DIN and DIP. Modeling results presented here will inform future studies of periphyton blooms along lakeshores.

Constantz et al. [2016] identified two distinct types of SLIs, one that has been channelized to confine and focus flow and one that naturally meanders and periodically forms a barrier beach and backwater pond adjacent to the lakeshore. Direct and indirect hydraulic and thermal estimates of seepage fluxes indicated profound impacts of channelization on SW-GW exchanges at the SLI. Here we extend their study and analyze organic and inorganic forms of phosphorus and nitrogen, dissolved organic carbon (DOC), pH, specific conductance (SPC), and dissolved oxygen (DO) concentrations collected in these two SLI environments to better understand biogeochemical processing and the impacts of SLI geomorphology on nutrient transport to the lake-nearshore region.

In the current study, heat and water flow simulations are used to estimate hydraulic properties for the two SLIs, using variably saturated two-dimensional stream profile models [Niswonger and Prudic, 2003; Stonestrom and Constantz, 2003; Constantz, 2012; Naranjo et al., 2012, 2013]. The calibrated models are then used to estimate spatially distributed mean residence times under steady flow conditions in the shallow subsurface at the SLI using the age-mass solute-transport approach [Goode, 1996; Naranjo et al., 2013].

Water quality parameters, including concentrations of DIN and DIP in surface water and shallow subsurface water, are analyzed with respect to mean residence times to quantify the conceptual hydraulic models of SLIs developed by Constantz et al. [2016] and explore their implications for water quality. Spatial

relationships of nutrient concentrations, DO, and DOC are compared to mean residence times in two types of SLIs, channelized and natural. A nutrient mass balance model is developed that considers net adsorption and desorption, mineralization, and nitrification along subsurface flow paths. This model is combined with simulated water flux rates and measured nutrient data to estimate mass fluxes in groundwater discharge to the lake for the natural SLI. Finally, nutrient mass fluxes are analyzed within the context of the two different SLI environments with respect to nutrient sources and relations between shallow subsurface flow patterns, residence times, and nutrient loading to Lake Tahoe.

2. Site Description and Methods

Lake Tahoe straddles the border of California and Nevada on the east side of the central Sierra Nevada. The lake's 1300 km² watershed ranges in altitude from 1300 m to 3290 m and receives most of its precipitation as snow. Lake Tahoe is a popular tourist destination and has been the focus of previous research, mostly focused on well documented decreases in the lake's exceptional clarity [e.g., Goldman, 1988; Jassby *et al.*, 1999; Swift *et al.*, 2006]. Recent efforts focus on changes in water quality and primary production in the nearshore region, including nutrient delivery by streams that receive snowmelt runoff [Heyvaert *et al.*, 2013]. There is evidence that biomass of attached algae has increased over the last decade [Hackley *et al.*, 2016]. During the winter season snowpack accumulates at the higher altitudes and drains through approximately 30 streams that flow to the lake during subsequent snowmelt. Incline and Marlette are relatively small creeks and their combined flows represent less than 5% of the total annual surface inflow to the lake [Coats and Goldman, 2001]. A survey has not been done of channelized versus nonchannelized streams entering Lake Tahoe and many of these streams are small and unnamed. Two of the largest streams entering Lake Tahoe, Upper Truckee River, and Trout Creek are not channelized.

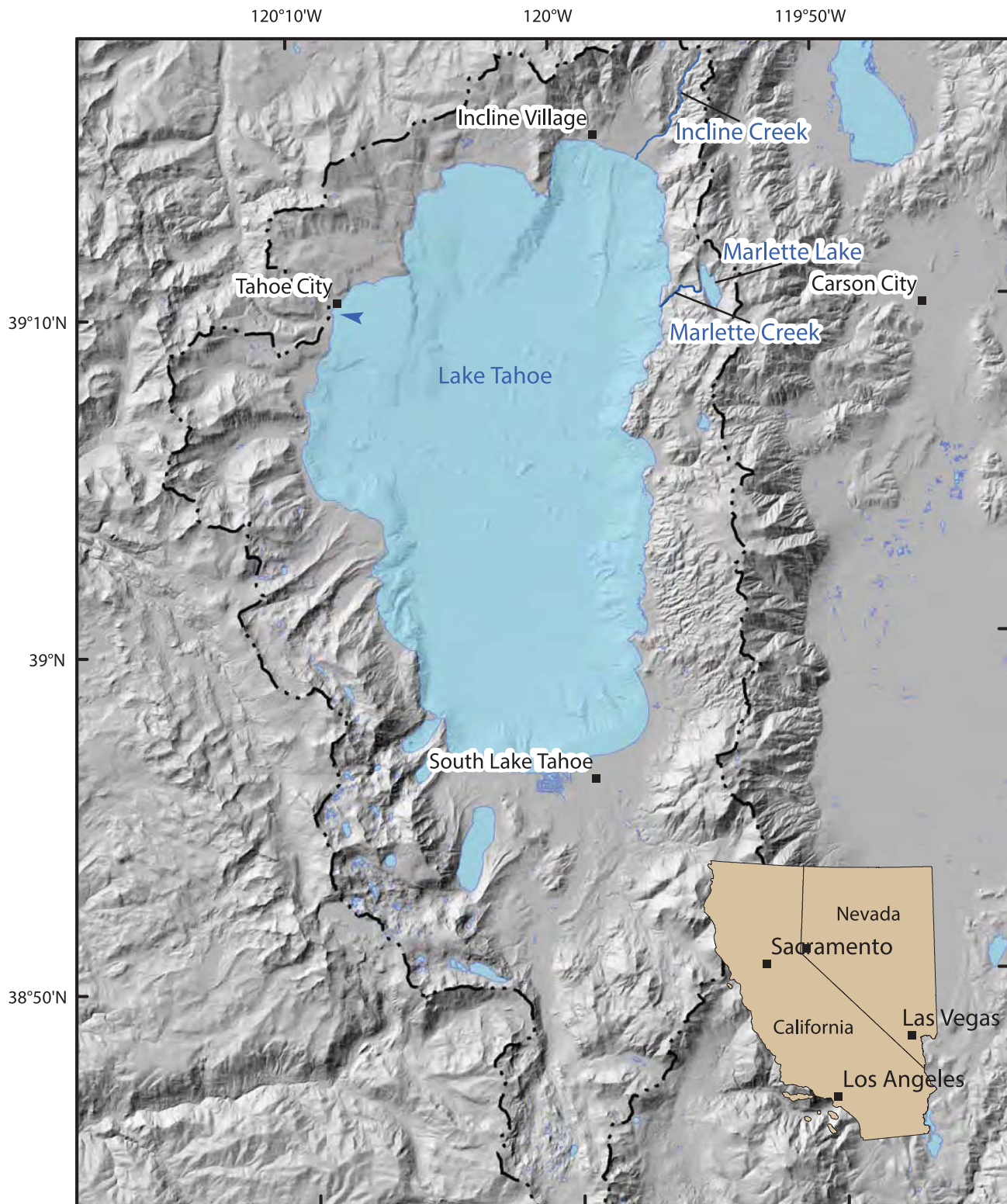
Constantz *et al.* [2016] provide a detailed description of two SLIs, located where Incline and Marlette Creeks enter the north side of Lake Tahoe, Nevada (Figure 1). The creek mouths are located approximately 10 km from each other, with Incline Creek to the north of Marlette Creek. Incline Creek drains a relatively developed watershed and enters the lake through a channelized mouth, whereas Marlette Creek drains a less developed watershed, and during low flows naturally meanders through the lakeshore beach region near its mouth. These two creeks were selected for study because they represent the two dominant conditions (manipulated vs. mostly natural) of SLIs in subalpine lake basins.

2.1. Instrumentation and Sampling

We surveyed land and stream topography and instrumented Incline Creek and Marlette Creeks with an array of automated transducers equipped with sensors for measuring water levels, and temperature, as described by Constantz *et al.* [2016], and using temperature probes described by Naranjo and Turcotte [2015]. Instruments were installed over approximate horizontal extents of 15 m and to a maximum depth of 2 m beneath the land or streambed or lakebed surface. Instruments were installed into the stream channel, beneath the barrier beach at Marlette, and along the shoreline area (Figures 2 and 3). Seepage meters were installed at locations described by Constantz *et al.* [2016] using methods described by Rosenberry [2008].

YSI model 6600 datasondes were used with a flow-through cell to measure time-sensitive water-quality parameters. A peristaltic pump was used to deliver water from sampling points in the stream, lake, and shallow groundwater to the sonde and sample bottles; 9.5-mm diameter stainless steel piezometers were used to sample groundwater at depths of 20, 40, and 60 cm beneath the water table or ground surface where the water table and ground surface coincided. Five piezometers were installed along the thalweg of Incline Creek at the SLI, and along a flow path through the barrier beach between a backwater pond and the shore of Lake Tahoe at Marlette Creek (Figure 2). Piezometers were spaced at approximately 3 m in transects.

In order to evaluate nearshore sources of nutrients to the lake, one-week hydrologic investigations were completed during mid-September of 2012 and 2013, and nutrient samples were taken during a 10-hr period at Incline and Marlette during September 18–19, 2013. Water quality samples were taken according to USGS water quality sampling protocol [Wilde, 2011; U.S. Geological Survey, variously dated]. Samples were taken using 125 mL bottles, stored in ice coolers and delivered to the laboratory for analysis within 48 hr.



Hydrography from 1:24,000-scale U.S. Geological Survey National Hydrography Dataset
 Universal Transverse Mercator Zone 11N, North American Datum of 1983

Figure 1. Lake Tahoe in the Central Sierra Nevada Mountain Range (CA/NV US). The study area is located where Incline Creek and Marlette Creek flow into Lake Tahoe on the northeast shore. Blue arrow indicates the location of the outlet to the Truckee River.

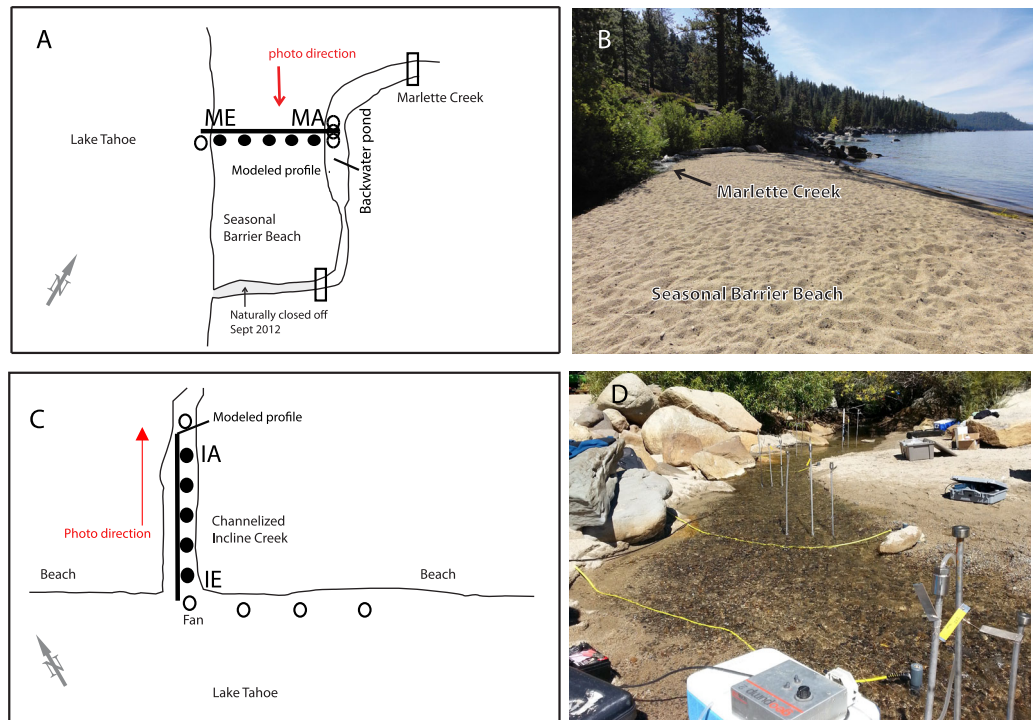


Figure 2. Plan views of Incline Creek, Marlette Creek, and adjoining shorelines. Red arrows give the direction from which the adjacent photograph was taken for each site. Thick black lines give the orientation of two-dimensional variably-saturated water flow, heat, and transport models. Top photo shows sampling and monitoring instrumentation. Black dots on left illustrations show sampling locations, temperature measurements, locations, open black circles show location of seepage meters, and rectangles show locations of flumes used for measuring streamflow.

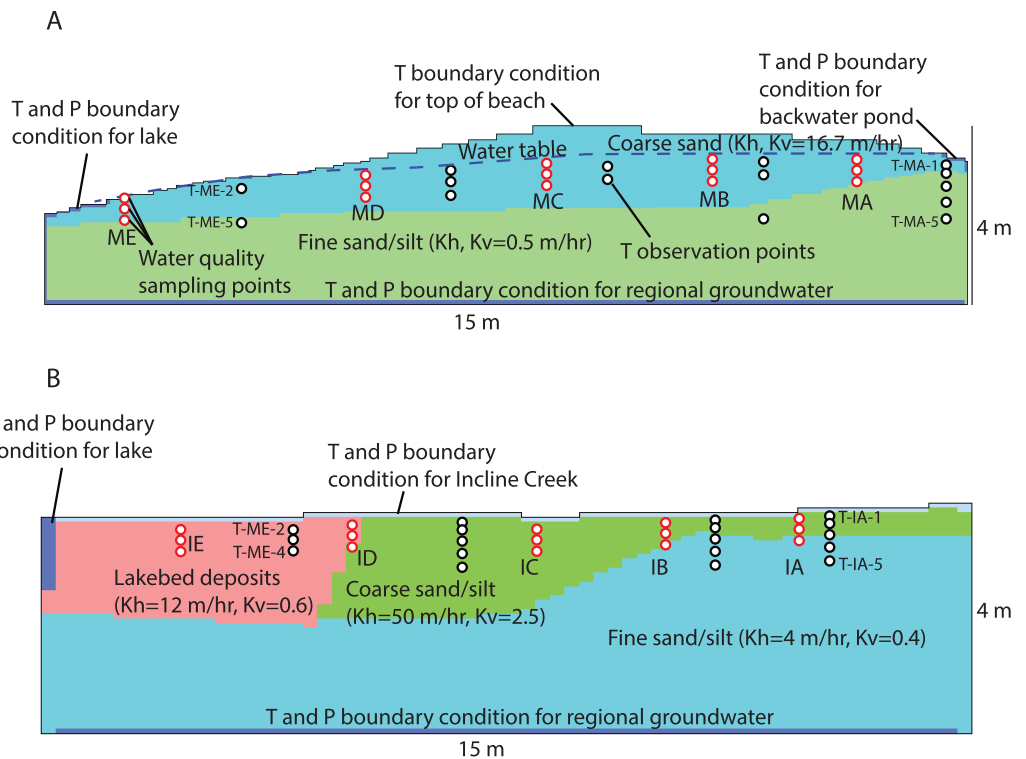


Figure 3. Illustration showing measurement locations and variably saturated heat and solute transport models used to determine hydraulic properties of sediment at the stream-lake interface, and mean residence times for (a) Marlette Creek barrier beach, and (b) Incline Creek.

All water quality samples were analyzed for dissolved nitrate plus nitrite using pyrophosphate (sampled through 0.45 μm membrane filters; method detection limit 2 $\mu\text{g/L}$); dissolved ammonium using indophenol (sampled through 0.45 μm membrane filters; method detection limit 3 $\mu\text{g/L}$); total Kjeldahl nitrogen (TKN; unfiltered; organic nitrogen plus ammonium; method detection limit 40 $\mu\text{g/L}$); soluble orthophosphate (SRP; sampled through 0.45 μm membrane filters; method detection limit 1 $\mu\text{g/L}$); dissolved phosphorus using persulfate digestion (DP; sampled through 0.45 μm membrane filters; method detection limit 2 $\mu\text{g/L}$); and total phosphorus using persulfate digestion (TP, unfiltered; method detection limit 2 $\mu\text{g/L}$). DOC was sampled in 300 mL amber baked glass bottles and analyzed using high temperature catalytic oxidation (Bird *et al.*, 2003; method detection limit 0.4 mg/L). N_2 gas samples were collected in 500 mL glass bottles with gas impermeable seals that were placed in a metal bucket filled with a constant stream of sample water to ensure that atmospheric contamination was minimized. For deep sampling wells, a peristaltic pump can create pressures less than atmospheric and result in degassing [Nielsen and Yeates, 1985]. However, this was not an issue during our study due to the very shallow sample depths (<1 m). Also, recharge temperatures (10–15°C) calculated from the measured dissolved gases are similar to measured groundwater temperatures and the upgradient stream water recharging the sandbar, indicating that temperature changes did not result in degassing of samples.

2.2. Heat and Mean Residence Time Modeling

Water flow, and heat and solute transport models were constructed using the variable saturated numerical models VS2DH and VS2DT [Healy, 1990; Healy and Ronan, 1996]. These models combine the heat tracing technique of Constantz [2012] for determining the hydraulic conductivity distribution in the hyporheic zone with the age-mass solute transport approach of Goode [1996] to estimate the mean residence times for steady flow conditions [Naranjo *et al.*, 2013, 2015]. Calibrated water- and heat-flow models provided the hydraulic properties and boundary conditions for subsequent solute transport modeling, after specifying the solute boundary conditions.

In order to simulate mean residence time under steady flow conditions, we followed the approach of Goode [1996] and Varni and Carrera [1998]. Our approach for simulating mean residence time in a shallow hyporheic zone beneath a stream is described by Naranjo *et al.* [2013] and the details are not repeated herein. Seepage beneath the SLI is unsteady at times due to wind waves on the lake and changes in streamflow. However, when winds are mild and during stream baseflow conditions, seepage beneath the SLI is relatively steady. For example, a constant flow during the sampling period of 0.045 m^3/s measured by the USGS streamflow gage at Crystal Bay, and the pond stage changed by less than 5 cm at Marlette during the sampling period. Wind was mild prior to and during the sampling at Marlette; however, wind picked up during the sampling at Incline.

The two-dimensional Marlette Creek VS2DH and VS2DT models represented a 15-m long vertical slice containing the flow path and instrumented section and extending to a depth of 4 m (Figure 3a). Marlette models were discretized into 125 rows and 53 columns, with an average cell width and thickness of 0.18 m and 0.09 m, respectively. Pressures measured on a 15-min interval using transducers located in the backwater pond, lake, and in a 2-m deep piezometer beneath the center of the barrier beach were used to specify hydraulic boundary conditions along the upgradient side, downgradient side, and bottom boundaries of the model, respectively.

The two-dimensional models of Incline Creek followed the stream thalweg starting 15 m upstream of the mouth and ended at the mouth, extending to a depth of 4 m beneath the surface of the sediment (Figure 3b). Incline models were discretized into 66 rows and 52 columns, with an average cell width and thickness of 0.25 m and 0.08 m, respectively. Pressure data collected from six transducers that measured stream head along the simulated transect were used to define hydraulic boundary conditions. Hydraulic boundary conditions varied in time for the heat transport simulations to represent wind waves following the water quality sampling period; however, hydraulic boundary conditions were constant for the mean residence time simulations to represent the steady flow conditions during the sampling period.

Temperature data collected every quarter hour in surface water, shallow soils, and at 1 m and 2 m beneath the sediment surface were used to define temperature boundary conditions in both the Marlette and Incline heat flow models (Figure 3). The model domain extended to 4 m below the top of the model to lessen the impact of uncertainty in groundwater age on the mean residence times modeling. Temperature at

4 m was extrapolated from measured temperatures at 1 m and 2 m using a temperature envelope approach [Bartolino and Niswonger, 1999].

VS2DH models were used to estimate hydraulic properties of sediment within the Marlette Barrier beach and Incline Creek transects (Figure 3). Temperature data at 10 cm, 20 cm, 40 cm, 60 cm, and 100 cm below ground surface were used as observations; however, temperatures for probes not located beneath the water table were excluded. In addition to measuring temperature with the probes of Naranjo and Turcotte [2015], individual temperature sensors were placed in piezometers at similar depths (Figure 3). Vertical and horizontal hydraulic conductivity (K_v and K_h), and the location of boundaries between sediment types with different values of K_v and K_h were adjusted using a hybrid automated/trial-and-error approach, such that the solution space over a range of values was sampled during repeated simulations until the error between the measured and simulated temperatures were minimized. Model hydraulic conductivity values for each sediment texture were estimated by calibrating the heat and water flow models using measured temperatures as observations. The Incline model was calibrated to temperature data measured during 2013. However, due to vandalism of temperature probes, the Marlette model was calibrated to temperature data measured during 2012. These calibrated flow models were then used to simulate mean residence times during the 2013 sampling period. Simulation times for Incline Creek and Marlette Creek heat modeling and calibration spanned the periods 9-17-2013 5 P.M. to 9-20-2013 1 P.M., and 9-19-2012 2 P.M. to 9-21-2012 12 P.M., respectively. Thermal properties used in the heat modeling were taken from values reported in the literature for representative sediment [Niswonger and Prudic, 2003]. Errors associated with assumed thermal parameters are likely small due to high advection in these permeable streambed and beach deposits.

2.3. Mass Balance Approach

Here we employ a mass balance approach to estimate nutrient mass transport, mineralization, nitrification, and net changes in total nutrient mass in pore water attributed to adsorption and desorption. Although it appears that denitrification rates were low, denitrification would result in errors in the other processes represented in our analysis. Further explanation for assuming negligible denitrification rates are provided in the discussion section. We assume constant solute flux into the barrier beach from the pond and that changes in nutrient concentrations in pore water are reasonably represented by adsorption-desorption and mineralization processes [Chapelle, 1995]. We assume increases in total Kjeldahl nitrogen (TKN) concentrations are attributable to: (1) net changes in adsorption and desorption of dissolved organic nitrogen (DON); and, (2) production of $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$. $\text{NO}_3\text{-N}$ and $\text{NO}_2\text{-N}$ are not distinguished in the sample analysis and thus results are presented strictly in terms of $\text{NO}_3\text{-N}$, as it typically constitutes the majority of a sample concentration. The nitrogen mass budget for sampled constituents can be written as:

$$M_{NTot} = M_{DIN} + M_{DON} \tag{1a}$$

$$M_{NTot} = (M_{\text{NH}_4} + M_{\text{NO}_3}) + (M_{\text{TKN}} - M_{\text{NH}_4}) \tag{1b}$$

$$M_{NTot} = M_{\text{TKN}} + M_{\text{NO}_3} \tag{1c}$$

Where M_{NTot} is mass of mobile N; M_{DIN} is mass of mobile DIN; M_{DON} is mass of mobile DON; M_{NH_4} is mass of ammonium; M_{NO_3} is mass of nitrate; M_{TKN} is mass of total Kjeldahl nitrogen, where all masses are per unit volume of pore water. All masses measured in samples are considered mobile and changes on total nitrogen are assumed to be caused by adsorption (loss) and desorption (gain). Considering nitrogen mass adsorption and desorption on soil particles assuming no denitrification, no loss of mass out of the control volume due to dispersion, and steady saturated flow, the total change of mass in a constant control volume (V) containing a flow path can be defined by:

$$\int_V \left[\frac{M_{NTot}}{\partial t} + \nabla \cdot (M_{NTot} u(x, z, t)) \right] dV = 0. \tag{2}$$

Where t is time, and $u(x, t)$ is the flow velocity. Assuming the flow velocity is constant and restricted to the x direction, equation (2) can be rewritten as:

$$-\frac{M_{NTot}}{\partial t} = u_x \frac{M_{NTot}}{\partial x} \quad (3)$$

Here u_x is the steady Darcy velocity of water flowing laterally through the barrier beach, and x is the total distance traveled by a packet of water between two points along a flow path. Because denitrification is assumed negligible, the net change in nitrogen that can be sampled in a packet of flowing water is the sum of adsorption and desorption. Substituting equation (1c) into equation (3) provides:

$$\frac{\partial(M_{adsorb} - M_{desorb})}{\partial t} = u_x \left[\frac{\partial(M_{TKN})}{\partial x} + \frac{\partial(M_{NO_3})}{\partial x} \right]. \quad (4)$$

Where M_{adsorb} and M_{desorb} are the mass of DON and DIN adsorbed and desorbed during the time increment ∂t . Previous studies have noted both organic and inorganic forms of N added/removed to/from water flowing through porous media [Hedin et al., 1995; Fahey et al., 1985; Currie et al., 1996; Schimel and Bennett, 2004]. Note that some portion of M_{desorb} is mineralized into NH_4 -N and NO_3 -N [Hart et al., 1994]. Replacing terms using finite differences, writing the mass of each nutrient constituent in terms of the cell by cell water flux rate estimated by the heat and water flow model and the measured concentration interpolated to each model cell center, equation (4) can be approximated as:

$$\frac{M_{net}^{t_2} - M_{net}^{t_1}}{t_2 - t_1} = \left[\frac{\left(\sum_{i=1}^n C_{TKN}^i u_i A_i \right)_{x=2} - \left(\sum_{i=1}^n C_{TKN}^i u_i A_i \right)_{x=1}}{A_T(x_2 - x_1)} + \frac{\left(\sum_{i=1}^n C_{NO_3}^i u_i A_i \right)_{x=2} - \left(\sum_{i=1}^n C_{NO_3}^i u_i A_i \right)_{x=1}}{A_T(x_2 - x_1)} \right]. \quad (5)$$

Where $M_{net}^{t_2}$ is the net mass of desorbed nitrogen, equal to the difference between $M_{adsorb} - M_{desorb}$ at time t_2 , $M_{net}^{t_1}$ is the net mass of desorbed nitrogen, equal to the difference between $M_{adsorb} - M_{desorb}$ at time t_1 , and n is the total number of model cell faces representing lateral flow through the hyporheic flow path for each nutrient component, C_{TKN,NO_3}^i is the measured concentration of TKN or NO_3 interpolated to the model cell center, u_i is the simulated Darcy velocity through each cell face A_i , and A_T is the total area of the hyporheic flow path equal to the sum of A_i . The net change in DIN and DON due to adsorption and desorption, mineralization, and nitrification can be written as:

$$M_{net}^{x_1 \rightarrow x_2} = \left(\frac{\Delta t \sum_{i=1}^n C_{TKN}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=2} - \left(\frac{\Delta t \sum_{i=1}^n C_{TKN}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=1} + \left(\frac{\Delta t \sum_{i=1}^n C_{NO_3}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=2} - \left(\frac{\Delta t \sum_{i=1}^n C_{NO_3}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=1}. \quad (6)$$

Where $M_{net}^{x_1 \rightarrow x_2}$ is the net change in the mass of nitrogen per unit volume of pore water between two points along a flow path.

Similar to nitrogen, a mass balance equation can be developed for phosphorus. In this case, we consider mass fluxes of DIP, DOP, and particulate phosphorus (PP = TP-DP) to Lake Tahoe. As TP consists of dissolved and particulate inorganic and organic forms of phosphorus, the net change in TP in pore water between sample locations can be attributed to adsorption and desorption processes and mineralization of DOP to DIP, effectively represented as the measured value of SRP in the samples. DIP, DOP, and PP can be estimated from constituents measured in water samples as:

$$M_{DOP} = M_{DP} - M_{SRP}, \quad (7a)$$

$$M_{PP} = M_{TP} - M_{DP}, \quad (7b)$$

$$M_{DIP} = M_{SRP}, \quad (7c)$$

Net change in phosphorus in pore water between sample locations along the barrier beach can be calculated following the same derivation for equation (6) and results in:

$$M_{net}^{x_1 \rightarrow x_2} = \left(\frac{\Delta t \sum_{i=1}^n C_{DOP}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=2} - \left(\frac{\Delta t \sum_{i=1}^n C_{DOP}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=1} + \left(\frac{\Delta t \sum_{i=1}^n C_{DIP}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=2} - \left(\frac{\Delta t \sum_{i=1}^n C_{DIP}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=1} + \left(\frac{\Delta t \sum_{i=1}^n C_{PP}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=2} - \left(\frac{\Delta t \sum_{i=1}^n C_{PP}^i u_i A_i}{A_T(x_2 - x_1)} \right)_{x=1} \quad (8)$$

Where C_{DOP} is the concentration of DOP; C_{DIP} is the concentration of DIP; and C_{PP} is the concentration of particulate (solid) phosphorus in pore water.

3. Results

3.1. Heat Transport Simulations

Two sediment textures were used to represent the sediment hydraulic properties at Marlette, including a coarse sand ($K_v, K_h = 16.7$ m/hr) and an overlying fine sand and silt ($K_v, K_h = 0.5$ m/hr; Figure 3a) based on visual observations of sediment cores and best simulated fits to temperature data. Similarly, three sediment textures were used to represent hydraulic properties of Incline Creek streambed, including a coarse sand ($K_v = 2.5$ m/hr, $K_h = 50$ m/hr), an underlying fine sand and silt that contained meter-scale boulders ($K_v = 0.4$ m/hr, $K_h = 4$ m/hr), and sand and silt unit without boulders ($K_v = 0.6$ m/hr, $K_h = 12$ m/hr; Figure 3b). Root mean squared error (RMSE) for all temperature observations ($N = 3640$) at Marlette is 0.69 C, and the RMSE for Incline ($N = 3640$) is 0.6 C. Overall simulated temperatures exhibited good fits to the measured temperatures, both in terms of absolute values, and signal phase and amplitude. Comparisons between simulated and measured temperatures for the Incline and Marlette Creek SLs are available in the online supporting information.

The model predicted that water seeped laterally through Marlette barrier beach at a rate of 450 m³/d per meter of beach width between the water table and a depth of 40 cm, whereas lateral seepage beneath 40 cm was 4 orders of magnitude lower and vertical seepage was negligible, except at the lake-shore seepage face. At Incline, total seepage from the stream into the subsurface was 420 m³/d and total seepage from the streambed into the stream was 450 m³/d per square meter of streambed, owing to regional groundwater.

Comparison between seepage-meter spot measurements and heat-modeling estimates of groundwater discharge to the lake shore showed reasonably good correspondence. Modeled seepage rates were calculated from the simulated vertical velocity for cells located at the same distance offshore as the seepage meter, multiplied by the porosity and divided by the total area of cells approximately contained within the area of a seepage meter. At Marlette Creek, a seepage meter was placed approximately 5 m east of the temperature probe transect (Figure 2). The seepage meter was placed just offshore of the lake water line where there was a focused discharge zone 0.5 m wide that spanned the length of the barrier beach. Reconnaissance indicated that this zone had discharge rates that were orders of magnitude greater than lakeshore areas outside this zone. Multiple seepage estimates resulted in an average seepage of 54.6 cm/d; whereas the simulated seepage from the heat modeling was 41.4 cm/d. Similarly, a seepage meter was placed 1 m offshore at Incline Creek. The average seepage meter estimate at Incline was 5.7 cm/d, whereas the heat modeling simulated seepage was 4.61 cm/d. Similar agreement between thermal estimates and seepage measurements of upward groundwater fluxes in different hydrologic setting were found by *Rosenberry et al.* [2016]. An average seepage estimate was made by taking the difference in flows at the 2 flumes located on Marlette Creek. However, the seepage estimated from the flumes was a much larger value of 700 cm/d, likely reflecting other effects on flow between the two flumes, including seepage upstream of the pond, evapotranspiration losses by riparian vegetation, evaporation off of the pond surface, and variations in seepage through the beach relative to the modeled flow path.

3.2. Mean Residence Time Simulations

Boundaries representing old water entering the model domain, such as regional groundwater flow, require special consideration in the model. The model was run for varying lengths of time to evaluate changes in mean residence time in the upper 60 cm of the model domain for different simulation times to insure steady

Table 1. Simulated Mean Residence Times in Hours at Pore-Water Sampling Locations. 20 cm, 40 cm, and 60 cm Column Headings Indicate Depths Below Water Table

Distance From Lake Shore (m)	Site ID	20 cm	40 cm	60 cm
Marlette Creek				
3.7	ME	21.0	22.0	63.6
6.6	MD	16.7	17.8	51.2
9.8	MC	11.8	13.7	41.0
12.6	MB	8.1	9.6	34.9
15.5	MA	3.7	5.0	30.1
Incline Creek				
-0.7	IE	2.7	6.8	15.0
2.3	ID	47.1	81.4	178.0
5.1	IC	158.0	234.0	332.0
8.7	IB	2.3	4.4	7.7
11.0	IA	2.3	3.0	4.4

state conditions (i.e., steady mean residence times in the upper 60 m of the saturated zone). The age of regional groundwater entering through the model boundary is unknown at the SLI. *Huntington and Niswonger* [2012] sampled for chlorofluorocarbon-age from springs discharging to Incline Creek and estimated an apparent age of about 15 years, approximately 500 m upstream from the SLI; it is expected that shallow groundwater discharge in Marlette Creek is of similar age to that in Incline due to similar geology, climate, drainage area, and altitude. Simulated mean residence time became insensitive to the age of inflowing regional groundwater at the bottom of the model

domain before the end of the simulation period. Despite uncertainty in the mean residence times of regional groundwater upwelling into these SLIs, insensitivity of the simulated mean residence time within the region of interest to changes in regional groundwater mean residence time indicates this uncertainty is not important to the results presented herein.

Mean residence time at the Marlette and Incline water quality sampling locations ranged between 1.5 and 12.3 hr, and 0.9 to 719.9 hr, respectively, (Table 1). The mean residence time mean and standard deviation for Incline was 201 hr and 270 hr, respectively; and the mean residence time mean and standard deviation for Marlette was 4 hr and 7 hr, respectively for the whole model domains. As illustrated by the much lower mean residence time values at Marlette, flow through the barrier beach consisted primarily of pond water, whereas seepage through the Incline Creek streambed was much more influenced by deeper mixing of surface water and groundwater originating upstream in the watershed. Mean residence times at Incline Creek were significantly greater and more variable than at Marlette due to direct contact between the flowing stream and shallow groundwater upwelling and downwelling caused by topographic variations in the streambed and heterogeneity of streambed sediment (Figures 2 and 3).

3.3. Stream-Lake Interface Water-Quality Conditions

3.3.1. Marlette Creek

Temperature patterns in the barrier beach indicate a shallow lateral seepage flow path that extends to about 40 cm beneath the water table (Figure 4a); the water table ranges between 0 and 0.5 m beneath the top of the sandbar. pH decreased and DO increased between the pond and lake (Figures 4b and 4c). The somewhat surprising increase in DO along the flow path will be addressed in the discussion section. SPC distribution was consistent with the temperature distribution in indicating a clear demarcation between rapid lateral seepage of pond water through the beach to a depth of 40 cm, and higher SPC below 40 cm indicative of groundwater not originating from the pond (Figure 4d). DOC increases from 3 to 4.5 mg/L through the barrier beach (Figure 4e). As indicated by the mean residence time simulations, water flowing through the barrier beach at Marlette Creek originates from a backwater pond that seeps through the beach and discharges to Lake Tahoe (Figure 4f). DIN in pond water is low, 6.9 µg/L, and increasing to 47.8 µg/L in pore water averaged over depths of 20 cm, 40 cm, and 60 cm, approximately 3.7 m upgradient of the lakeshore (Figure 3 and Table 2). These increases in DIN, and corresponding decreases in DON, indicate that mineralization of DON occurs followed by nitrification that increases NO₃-N concentrations from 1.6 µg/L to 38.6 µg/L between 6.6 m and 3.7 m upgradient of the lakeshore. Consistent acidification along the flow path also supports mineralization and nitrification, and also could be due to decomposition of organic matter [Chapelle, 1995]. Depth-average DOC concentrations of 2 mg/L at 15.5 m increases to 4 mg/L at 3.7 m upgradient from the lakeshore. High ratios of C:NO₃-N, low concentrations of NH₄-N, and lower DO concentrations likely explain the lack of nitrification over the upgradient portion of the barrier beach [Strauss et al., 2002]. Low concentrations of DIN (6.7 µg/L) and DON (77 µg/L) in pond water, and subsequent increases along the flow path, indicate that more than half of nitrogen in groundwater flowing to the lake comes from organic matter stored within the barrier beach. No excess N₂ gas was measured in pore-water samples indicating that denitrification was likely insignificant.

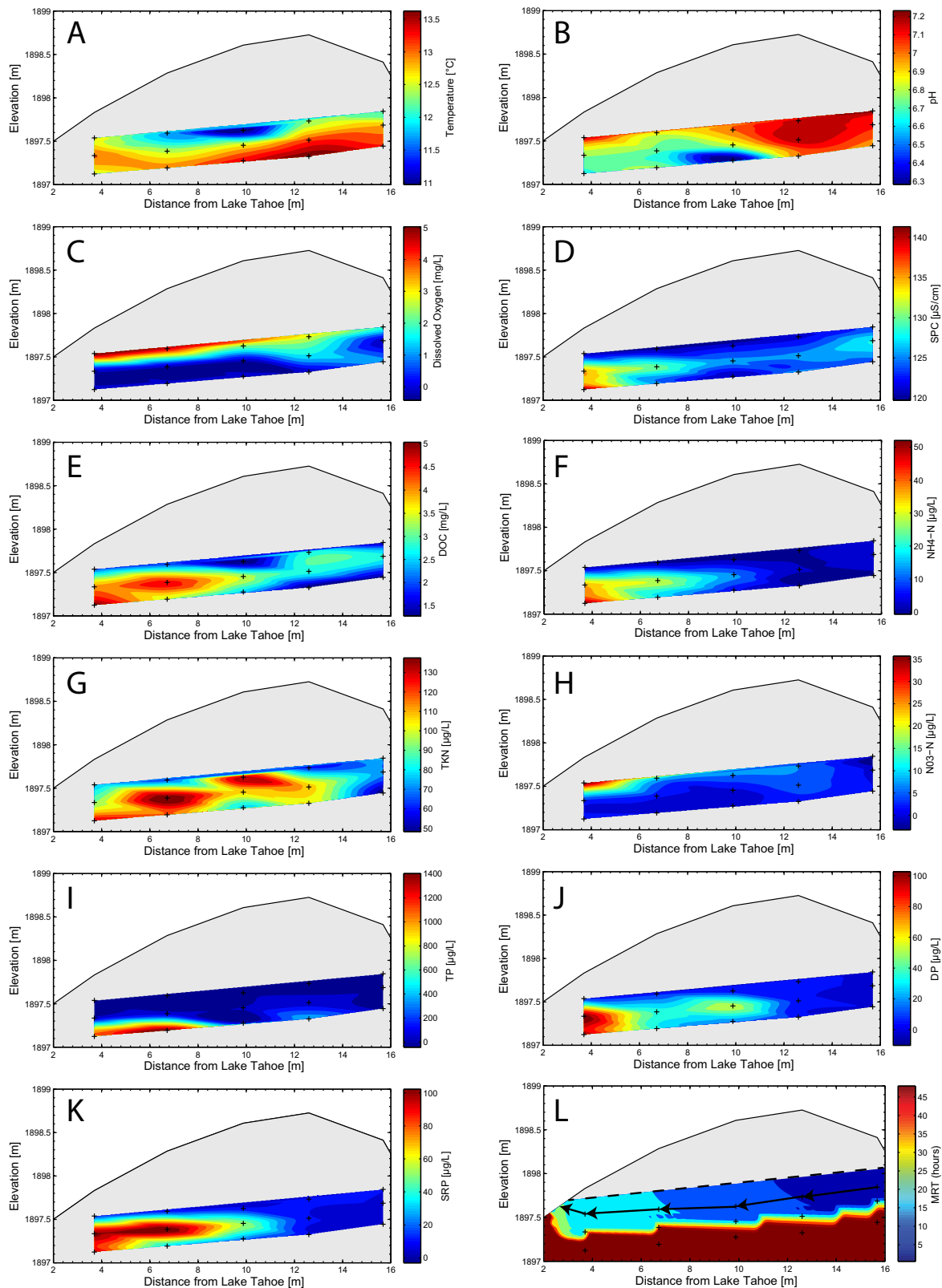


Figure 4. Contour plots of measured: (a) temperature, (b) pH, (c) dissolved oxygen, (d) specific conductance, (e) dissolved organic carbon, (f) ammonium, (g) total Kjeldahl nitrogen, (h) nitrate, (i) total phosphorus, (j) dissolved phosphorus, (k) soluble reactive phosphorus, and (l) simulated mean residence time of pore water beneath Marlette Creek barrier beach. Dashed line is water table and dashed rectangle is the contoured regions shown in (a–k). Lines with arrows in (l) show the dominant flow path. Sampling was done on 9/18/2013.

Table 2. Nutrient Concentrations and Other Water Quality Parameters Measured at the Stream-Lake Interface at Two Streams in the Lake Tahoe Basin

Site	Sample Date and Time	Distance From Lake Shore (m)	Depth (cm)	NO ₃ -N (μg/L)	NH ₄ -N (μg/L)	TKN (μg/L)	SRP (μg/L)	TP (μg/L)	DP (μg/L)	DO (mg/L)	SPC (uS/cm)	pH	Temp (°C)	DOC (mg/L)
Marlette Creek														
ME1	9/18/13 3:00 P.M.	3.7	20	38.6	3.6	77.0	9.9	21.0	12.4	5.4	121.5	7.2	12.8	2.2
ME2	9/18/13 3:30 P.M.	3.7	40	4.1	37.5	95.5	104.5	106.0	108.6	0.1	135.5	6.8	13.1	4.1
ME3	9/18/13 4:00 P.M.	3.7	60	3.7	56.0	134.4	103.0	1059.2	107.2	0.0	143.0	6.7	12.6	5.3
MD1	9/18/13 1:30 P.M.	6.6	20	11.2	2.2	68.8	8.3	10.9	10.2	4.7	120.9	6.9	11.8	2.1
MD2	9/18/13 2:00 P.M.	6.6	40	3.0	29.4	142.9	36.2	108.5	39.8	0.0	130.5	6.8	12.5	4.7
MD3	9/18/13 2:30 P.M.	6.6	60	1.8	21.0	121.1	36.0	1481.1	39.0	0.0	126.5	6.8	13.0	3.6
MC1	9/18/13 12:00 P.M.	9.8	20	7.2	2.7	133.4	5.9	35.3	8.1	2.5	122.5	7.0	11.0	1.3
MC2	9/18/13 1:30 P.M.	9.8	40	1.6	13.8	108.7	51.1	58.7	53.8	0.0	125.9	6.9	12.7	3.4
MC3	9/18/13 1:00 P.M.	9.8	60	0.9	7.4	82.1	20.5	107.2	22.3	0.0	121.2	6.3	13.4	2.6
MB1	9/18/13 10:30 A.M.	12.6	20	7.2	2.7	64.9	5.9	15.5	8.1	2.9	122.1	7.2	12.4	2.9
MB2	9/18/13 11:00 A.M.	12.6	40	6.0	3.1	111.2	5.2	96.4	8.7	1.7	125.9	7.2	13.2	2.9
MB3	9/18/13 11:30 A.M.	12.6	60	4.4	3.6	108.3	7.0	406.4	9.9	0.0	126.6	7.0	13.8	1.3
MA1	9/18/13 9:00 A.M.	15.6	20	1.6	3.3	81.2	4.0	19.2	1.9	2.1	124.7	7.3	12.3	1.4
MA2	9/18/13 9:30 A.M.	15.6	40	1.3	4.5	73.3	3.4	38.7	1.9	0.0	128.5	7.1	12.9	2.9
MA3	9/18/13 10:00 A.M.	15.6	60	2.0	2.5	48.2	2.9	27.8	4.1	1.1	124.9	7.0	13.3	1.4
Marlette pond	9/18/13 9:30 A.M.	16.5	surface water	1.8	4.9	82	11.0	16.5	14.6	9.1	123.0	7.9	15.2	*
Incline Creek														
Lake Tahoe	9/18/13 11:30 A.M.	-0.7	0	nd	nd	62	nd	25	13	6.79	94.0	7.56	18.3	*
IE1	9/19/13 3:00 P.M.	-0.7	20	7.2	2.5	61.9	12.2	29.4	15.2	7.7	94.0	7.4	10.7	1.9
IE2	9/19/13 3:30 P.M.	-0.7	40	6.0	2.9	132.0	16.4	23.5	18.6	7.4	94.8	7.5	10.0	2.5
IE3	9/18/13 4:00 P.M.	-0.7	60	6.5	11.2	74.2	16.9	35.9	18.6	6.1	91.6	7.5	9.6	1.7
ID1	9/18/13 1:30 P.M.	2.4	20	4.4	3.8	41.9	13.3	20.4	16.4	8.1	93.5	7.6	9.3	2.0
ID2	9/19/13 2:00 P.M.	2.4	40	5.8	2.5	38.6	16.0	22.0	18.3	7.9	94.2	7.5	9.6	1.9
ID3	9/19/13 2:30 P.M.	2.4	60	3.9	2.5	28.7	20.3	23.8	22.6	7.2	93.2	7.4	10.1	1.9
IC1	9/19/13 12:00 P.M.	5.1	20	7.4	2.7	38.6	14.6	19.2	17.0	7.2	90.5	7.3	10.1	1.9
IC2	9/19/13 12:30 P.M.	5.1	40	8.4	2.7	32.0	17.3	23.5	20.4	7.0	91.0	7.2	10.1	1.8
IC3	9/19/13 1:00 P.M.	5.1	60	10.9	3.1	59.5	13.7	18.6	13.9	6.0	89.2	7.1	10.6	1.0
IB1	9/19/13 10:30 A.M.	8.7	20	5.3	2.9	35.3	20.5	29.7	23.8	7.9	91.8	7.4	9.4	2.0
IB2	9/19/13 11:00 A.M.	8.7	40	5.1	2.7	61.9	23.4	29.4	26.6	8.0	92.5	7.4	9.3	2.5
IB3	9/19/13 11:30 A.M.	8.7	60	7.2	2.7	26.8	26.4	31.5	28.8	7.5	90.1	7.3	9.8	1.9
IA1	9/19/13 9:00 A.M.	11.0	20	5.3	2.7	41.0	16.9	22.6	21.0	7.1	93.5	7.5	9.7	1.8
IA2	9/19/13 9:30 A.M.	11.0	40	7.7	3.1	49.1	22.1	27.2	26.0	7.1	91.9	7.4	9.8	1.9
IA3	9/19/13 10:00 A.M.	11.0	60	9.8	2.2	42.9	21.6	26.0	24.8	7.0	91.2	7.3	10.3	1.0
Incline Creek	9/19/13 9:30 A.M.	11.0	Surface water	2.7	5.1	39	11.5	34.1	15.2	6.8	93.8	7.6	18.3	*

^aSite names begin with M or I for Marlette or Incline, letters A-E designate sampling transects with A furthest and E closest to the lakeshore, numbers 1, 2, and 3, designate sampling depths of 20 cm, 40 cm, and 60 cm, respectively.

Mostly anoxic conditions exist between 40 cm to 60 cm over the entire length of the flow path, likely limiting nitrification at these depths. At a depth of 60 cm, DIN as NH₄-N increases from 7.4 μg/L at 9.8 m to 56 μg/L at 3.7 m upgradient from the lakeshore (Table 2 and Figure 4). DO concentrations at a depth of 20 cm increase from 2.1 mg/L at 15.5 m to 5.4 mg/L at 3.7 m. NO₃-N increased from 1.6 μg/L at 15.5 m to 38.6 μg/L at 3.7 m (Figure 2a), indicating that there is a sudden increase in mineralization and nitrification. These increases in DO along the flow path are difficult to explain, as theoretical DO enrichment from diffusion through the capillary fringe cannot explain these increases. Fluctuations in the water table due to variations in pond or lake surface levels could encapsulate and pressurize air, enhancing DO enrichment in the shallow saturated sediment. Fluctuations in pond, lake, and water table elevations were less than 5 cm throughout the sampling period. Precipitation did not occur within 24 hr of the sampling period, indicating that flow conditions and water quality of inflowing water were likely steady during the sampling period. Repeated DO measurements during the sampling period indicated that DO concentrations were steady. However, DO concentrations during the following day were 1-3 mg/L higher than during the sampling period, likely a result of increases in wind on the lake that generated 10-30 cm waves. These conditions indicate that fluctuations in the water table caused by waves on the lake, and subsequent air entrapment from the capillary zone are likely an important mechanism for enhancing DO concentrations in pore water flowing to the lake.

Total phosphorous (TP) and soluble reactive phosphorus (SRP) increase along the flow path through the barrier beach at Marlette; concentrations also increase with depth from 20 cm to 60 cm, indicating greater amounts of phosphorous desorbs in the deeper, more permanent beach sediment between 6.6 m and 3.7 m (Table 2 and

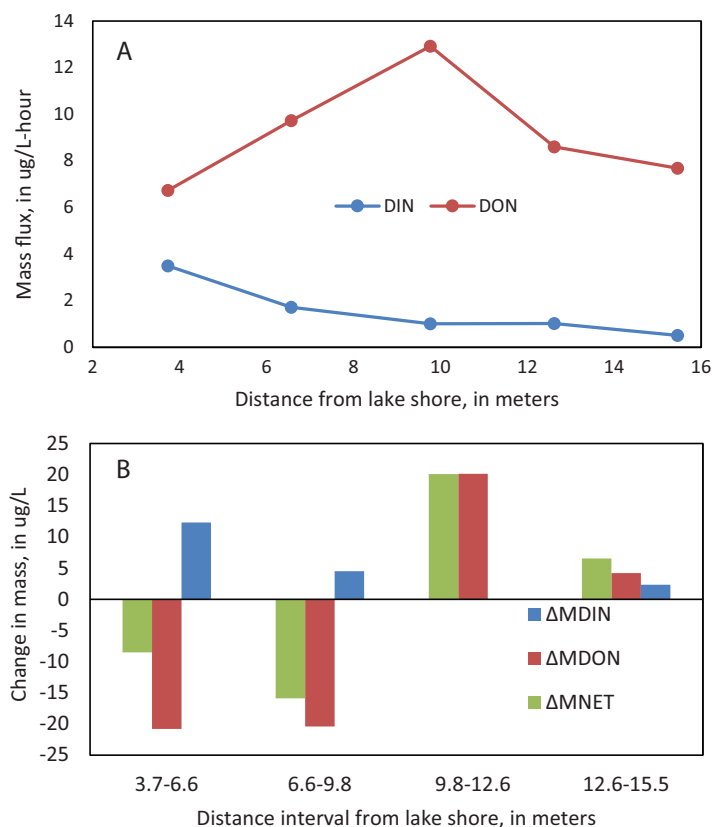


Figure 5. Plots of: (a) Lateral mass fluxes of dissolved inorganic nitrogen and dissolved organic nitrogen toward the lake, and (b) water flux weighted changes in nitrogen concentration vertically averaged over 60 cm at Marlette Creek barrier beach.

the lake shore, per meter width of barrier beach (Figure 5a). DIN flux remains relatively constant through the barrier beach until between 9.8 m and 3.7 m, where it increases from 1.01 $\mu\text{g/L-hr}$ to 3.48 $\mu\text{g/L-hr}$ per meter width of barrier beach before discharging to Lake Tahoe. Multiplying these mass flux values by the simulated depth-integrated change in mean residence time between sample sites (vertical integration of values shown in Figure 4l) provides a water flux weighted and depth integrated change in nutrient mass at points along the barrier beach (Figure 5b). The depth averaged total mass of DON increases by 20.14 $\mu\text{g/L}$ between 15.5 m and 9.8 m upgradient from the lakeshore during the 22 hr average residence time in this interval. A decrease in the mass of DON of 24.4 $\mu\text{g/L}$ occurs between 9.8 m and 3.7 m upgradient of the lakeshore, and DIN increased by 12.3 $\mu\text{g/L}$ due to mineralization (Figure 5b). Significant positive and negative changes in the total dissolved mass of nitrogen in the barrier beach indicates adsorption/desorption processes are important in these sediments.

Substituting the measured phosphorus concentrations and simulated water velocities into equation (8) results in an increase in mass flux of DIP from 0.15 $\mu\text{g/L-hr}$ to 0.59 $\mu\text{g/L-hr}$ between 15.5 m and 6.6 m upgradient of the lake shore, per meter width of barrier beach (Figure 6a). DOP increases at a relatively constant rate over the length of the barrier beach from 0.0 $\mu\text{g/L-hr}$ to 0.59 $\mu\text{g/L-hr}$. Most of the total phosphorus sampled in the barrier beach consisted of particulate phosphorus ($\text{PP} = \text{TP} - \text{DP}$) and the mass flux increased from 1.6 $\mu\text{g/L-hr}$ to 2.5 $\mu\text{g/L-hr}$ between 15.5 m and 9.8 m, and decreased to 0.8 $\mu\text{g/L-hr}$ at 3.7 m. As with nitrogen, multiplying these mass flux values of phosphorus by the depth integrated change in mean residence time provides a water flux weighted and depth integrated change in nutrient mass along the barrier beach (Figure 6b). DIP increases by 8.6 $\mu\text{g/L}$ between 15.5 m and 6.6 m during the 81 hr average residence time in this interval. DIP decreases by 0.7 $\mu\text{g/L}$ of between 6.6 m and 3.7 m (Figure 6b).

3.3.2. Incline Creek

A wide distribution of flow paths at Incline result in relatively large exchanges between the stream and groundwater, including vertical and lateral seepage. During the monitoring period (September 17–20, 2013)

Figures 4i–4k). Lower DO at 40 cm and 60 cm likely causes increases in TP due to an inverse relationship between DO concentrations and P mobilization [Chapelle, 1995]. Concentrations exceed 1000 $\mu\text{g/L}$ and 100 $\mu\text{g/L}$ for TP and SRP, respectively, at 3.7 m upgradient of the lakeshore (Table 2). Relatively large changes in TP and SRP occurring over a flow path and mean residence times less than 36 hr indicate that nutrient-rich organic matter is present in the beach deposits at the Marlette SLI, and that these nutrients are mineralized and mobilized into shallow pore water flowing to the lake.

Substituting the measured nitrogen concentrations and simulated water velocities into equation (6) results in a mass flux of DON that increases through the barrier beach from 8.6 $\mu\text{g/L-hr}$ to 12.9 $\mu\text{g/L-hr}$ between 15.5 m and 9.8 m, and decreases from 12.2 $\mu\text{g/L-hr}$ to 6.7 $\mu\text{g/L-hr}$ between 9.8 m and 3.7 m upgradient from

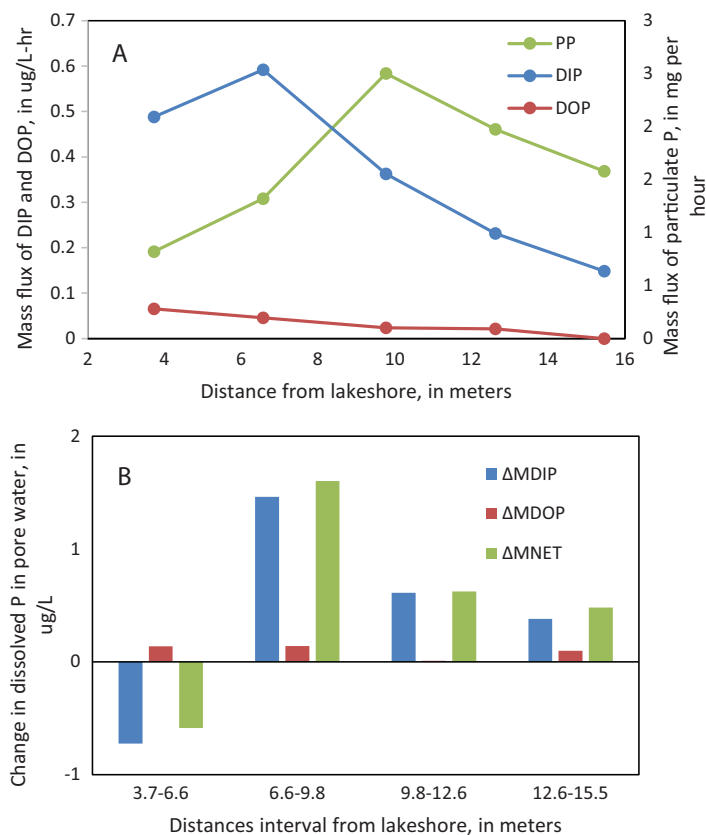


Figure 6. Plots of: (a) Lateral mass fluxes of dissolved inorganic phosphorus, dissolved organic phosphorus, and particulate phosphorus, and (b) water flux weighted changes in phosphorus concentration vertically averaged over 60 cm at Marlette Creek barrier beach.

the depth of the flowing stream ranged between 0.04 m in the riffle to 0.30 m in the pool. Groundwater levels were lower than the stream water surface, and the average hydraulic gradient measured between the stream and groundwater at a depth of 2 m was 0.005 m/m. Temperature, pH, DO, SPC, and DOC all have a similar spatial distribution with a clear demarcation of downwelling zones at the upstream and downstream ends of the study area, and an upwelling zone at the small pool in the middle of the study area, where mean residence time values are the greatest (Figure 7). Lake and stream water pH (7.56 and 7.6) and specific conductance (SPC = 94 uS/cm and 93.8 uS/cm) were greater than the average of samples at 60 cm (pH = 7.3 and SPC = 91.1 uS/cm), indicating that the stream mouth is a downwelling zone, likely due to draining of regional groundwater upstream of the SLI that lowers

groundwater levels beneath the mouth and wind waves on the lake that pushes water into the subsurface [Huntington and Niswonger, 2012]. Lateral seepage in the shallow subsurface also is important for enhancing mixing and the distribution of relatively high concentrations of DO in the hyporheic zone, as indicated by elevated DO concentrations in the shallow upwelling zone at the middle of the study reach. Direct contact between the stream and groundwater at Incline, and a distribution of long and short mean residence times at Incline likely explain greater DO concentrations relative to Marlette.

One clear distinction in the water quality at Incline Creek relative to Marlette is the significantly lower NH₄-N, TKN, and NO₃-N concentrations (Figures 7f–7h). The maximum DON concentration at Incline was 132 μg/L beneath the mouth, only modestly different than a maximum of 143 μg/L at Marlette (Table 2). However, the average for all sample location at Incline was 51 μg/L, relative to an average of 97 μg/L at Marlette. DIN in Incline stream water was 8 μg/L, and although low relative to pore water, nearly twice that of pond and stream water at Marlette.

DIN concentrations in pore water at Incline ranged between 6 ug/L and 18 ug/L, and were greatest in the upwelling zone in a small pool at the base of a riffle (Figure 7). Greater DIN in the groundwater upwelling zone is an indication that the majority of DIN is coming from regional groundwater that originates from snowmelt, from DON that is mineralized higher up in the watershed, or anthropogenic sources in the lower developed portion of the watershed. Snow in the Tahoe basin is enriched in DIN by atmospheric deposition, although a portion is taken up by trees in forested watersheds [Coats et al., 1976]. Incline Creek watershed is more developed and has greater impervious areas as compared to Marlette Creek watershed, which may explain higher DIN concentrations in the stream water. Nitrification does not appear to occur along shallow upwelling and downwelling flow paths beneath the streambed in the study area. Additionally, low NO₃-N concentrations and no measurable N₂ gas indicate that denitrification is insignificant as well, consistent with high DO concentrations (> 6.8 mg/L) throughout the Incline streambed.

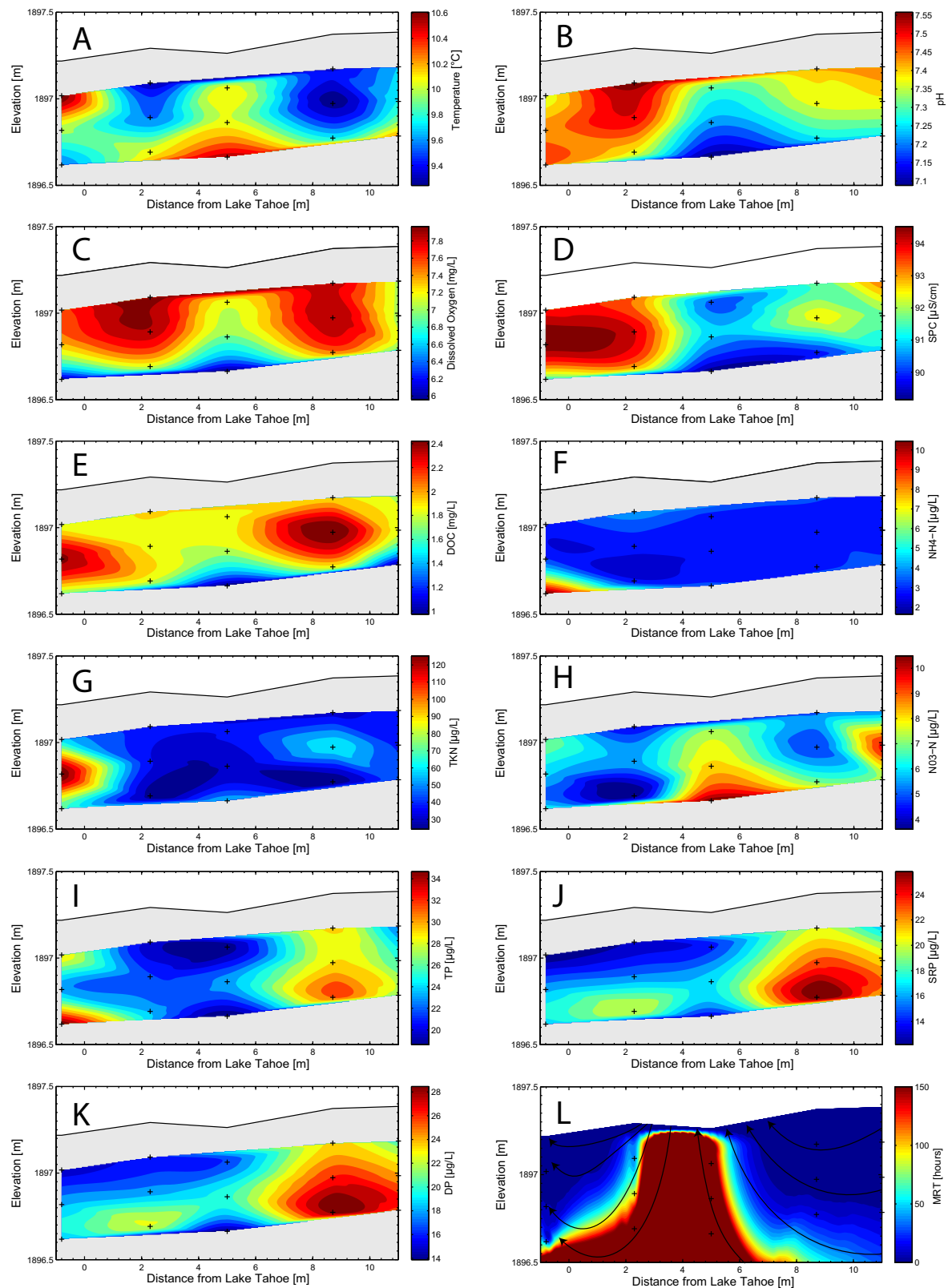


Figure 7. Contour plots of measured: (a) temperature, (b) pH, (c) dissolved oxygen, (d) specific conductance, (e) dissolved organic carbon, (f) ammonium, (g) total Kjeldahl nitrogen, (h) nitrate, (i) total phosphorus, (j) dissolved phosphorus, (k) soluble reactive phosphorus, and (l) simulated mean residence time of pore water beneath Incline Creek. Dashed is the contoured regions shown in (a–l). Lines with arrows in (l) show dominant flow paths. Sampling was done on 9/19/2013.

Unlike Marlette, seepage through the subsurface to the lake at Incline does not follow a single dominant flow path with monotonically increasing mean residence time approaching the lake (Figure 7l). Accordingly, the distribution of DON and DIN does not indicate a clear evolution of mineralization and/or nitrification pathway. Rather, the nutrient distribution is most correlated with areas of groundwater upwelling and downwelling of lake water from the lakebed and stream mouth. Effects of variations in lake levels has a greater impact on water quality beneath Incline Creek because lake waves can move up Incline Creek, likely enhancing DO concentrations in the hyporheic zone due to downwelling. This is in contrast to the Marlette SLI, where the barrier beach serves as a dam blocking lake water from moving up the channel. At Incline, DON is contributed from the lake (62 $\mu\text{g/L}$), as evidenced by increases in TKN relative to the stream (55 $\mu\text{g/L}$), where water downwells at the stream mouth (Figure 7g).

DIP and TP concentrations in pore water beneath Incline Creek are low. Maximum and average DIP concentrations are 16 $\mu\text{g/L}$ and 18 $\mu\text{g/L}$ at Incline, as compared to 105 $\mu\text{g/L}$ and 26.9 $\mu\text{g/L}$ at Marlette (Table 2 and Figure 7). Maximum and average TP concentrations are 36 $\mu\text{g/L}$ and 26 $\mu\text{g/L}$ at Incline, and 1481 $\mu\text{g/L}$ and 239 $\mu\text{g/L}$ at Marlette. These differences could be due to much greater DO concentrations in pore water at Incline that tends to reduce mobilization of phosphorus, or alternatively that the sedimentary organic matter content and reactivity is lower at Incline than at Marlette [Chapelle, 1995]. TP and SRP concentrations also are more homogeneous within the Incline pore water relative to Marlette. However, evidence that phosphorus is desorbing and mineralizing in the streambed is indicated by local areas of modestly higher TP, SRP, and DP concentrations in the upper portion of the study area where the stream is downwelling (Figure 7i–7k). Generally, the Incline streambed appears to be a smaller source of inorganic nutrients relative to Marlette.

4. Discussion

4.1. Characterizing Nutrient Flow Paths Using Numerical Models

We calibrated water, heat, and transport models to characterize seepage rates, mean residence time, and fate and transport of DIN and DIP at two SLIs in Lake Tahoe. Calibrating flow models using heat as a tracer provides a means to constrain the flux rates and volume over which subsurface seepage occurred and this provided estimates the loading of DIN and DIP from groundwater seepage beneath the SLI. This approach differs from previous hyporheic studies in that we did not monitor an injected conservative tracer along a characterized flow path to infer differences between fate and transport [e.g., Triska *et al.*, 1989; Holmes *et al.*, 1994; Zarnetske *et al.*, 2011]. Due to regulatory constraints, we were unable to input external chemicals to these systems and relied on heat as a natural tracer to characterize the relevant hydrologic domain [Constantz, 2012]. We defined a control volume per unit width by calibrating a 2D flow model to estimate vertically distributed seepage rates, mean residence time, and fate and transport of DON, DOP, DIN, and DOP. Our approach has advantages and disadvantages relative to injected tracer approaches. Advantages include consideration of vertically varying water and nutrient flux rates and representation of spatially distributed mean residence times within a 2D heterogeneous model domain. A disadvantage is that our approach is indirect, and relies on a nonconservative tracer (i.e., heat). Thus, imperfect calibration and model uncertainty influences our estimates of nutrient fluxes.

4.2. Nutrient Mineralization at the Stream-Lake Interface

Both of these SLI systems had 2–3 times the amount of organic nutrients relative to inorganic nutrients, indicating that mineralization likely is limiting fate and transport of DIN and DIP (Figures 5 and 6; Aber and Melillo, 2001). Beneath the channelized SLI (Incline Creek) concentrations of DIN and DIP in groundwater entering the lake were much lower than concentrations in groundwater entering the lake beneath the non-channelized SLI (Marlette Creek). It is important to note that higher concentrations of DIN and DIP in Incline Creek surface water relative to Marlette Creek surface water results in greater total loading at Incline Creek; however, higher concentrations in focused groundwater discharge to the lake shore at Marlette likely is more efficient for delivering DIN and DIP for uptake by attached algae [Hagerthey and Kerfoot, 1998; Rosen, 2015; Prouty *et al.*, 2016; Anderson *et al.*, 2002].

Net production of DIN in this study through the length of the barrier beach of 2.2 $\mu\text{g/L-hr}$ is less than other studies sampling water with similar travel distances and times. For example, Jones *et al.* [1995] reported 23 $\mu\text{g/L-hr}$, Zarnetske *et al.* [2011] reported 36.7 $\mu\text{g/L-hr}$; however, Triska *et al.* reported a much lower rate of 0.002 $\mu\text{g/L-hr}$. Denitrification was documented in each of these studies, with the exception of Jones *et al.*

[1995] that reported no denitrification, consistent with our study of the SLIs. A unique aspect of DIN production at the Marlette SLI is the short distance over which most of the mineralization and nitrification occurred. Very little DIN production occurred between 15.5 m and 8.7 m upgradient of the lakeshore, and the bulk production occurred 3.7 m upgradient of the lakeshore, where the depth averaged travel time over this distance was 7 hr. Other studies have reported localized regions of greater nitrification relative to other flow path locations, such as near the streambed/surface interface [e.g., *Holmes et al.*, 1984; *Jones et al.*, 1995; *Harvey and Gooseff*, 2015]. At Marlette, increased nitrification was not associated with the streambed interface. However, increased nitrification occurred in the zone where wave run up from the lake occurs periodically that results in infiltration of lake water into the barrier beach.

4.3. Negligible Denitrification

Denitrification did not appear to be an important process at either the Incline or Marlette SLIs. At Incline, DO concentrations were high enough to limit denitrification [*Hedin et al.*, 1998]; however, at Marlette DO concentrations were low below 40 cm, indicating denitrification could occur at these locations. Other studies have reported a lack of denitrification in the hyporheic zone where DO was generally greater than 1 mg/L [*Holmes et al.*, 1984; *Jones et al.*, 1995]. This is consistent with conditions at the Incline SLI. Marlette DO concentrations were near zero 40 cm beneath the water table, where lateral pore water velocities decreased four orders of magnitude. DO was greater than 2 mg/L above 40 cm where most of the lateral seepage occurred. Because the vertical component of seepage was very low below 40 cm, very little mixing occurred. Although denitrification likely occurred at the fringe between low and high DO (~40 cm) mass fluxes were very low and this likely limited denitrification rates, and likely explains why there was no N₂ gas detected. Another indication that denitrification was likely negligible at in the Marlette Creek barrier beach is the similar NO₃ concentration among shallow, high DO and deeper, low DO sampling locations (i.e., site MA-MC; Table 2), followed by modest increases in NO₃ and NH₄ at site MD and greater increases at site ME. However, greater decreases in DON-N relative to increases in DIN-N (Figure 5) indicate that either DON-N was adsorbing to sediment [*Schimel and Bennett*, 2004] or that denitrification occurred and we were not able to sample N₂ gas that was produced. If denitrification occurred, then our mineralization and nitrification rates are conservative.

4.4. Channelized Versus Nonchannelized Stream-Lake Interface

Our study distinguished two types of SLIs, one that seasonally forms a barrier beach and another that is purposely channelized for recreational purposes. A barrier beach is a geomorphic feature that is analogous to the parafluvial zone of a stream [e.g., *Holmes et al.*, 1994]. Similarities include seasonal erosion and depositional processes that could include import and export of organic matter. It may be that like the parafluvial zone, barrier beaches are unique in their biogeochemistry due to limited inundation by surface water, and differences in sediment and nutrient transport processes. Source and bioavailability of organic matter stored within the barrier beach likely play an important role in distinguishing this type of SLI [*Thurman*, 2012; *Shen et al.*, 2015]. Our data clearly indicate greater DON and DIN within the Marlette barrier beach relative to the channelized streambed at Incline. Other than channelization within the SLI, these mountain streams appear to be very similar.

At the Incline SLI, a permanently inundated channel that connects the lake to the stream creates a greater degree of unsteady seepage processes and mixing between surface water and groundwater, similar to previous hyporheic studies beneath the active channel [*Valett et al.*, 1994; *Campana*, 1996; *Briggs et al.*, 2014; *Naranjo et al.*, 2015]. The much lower concentrations of DOC, DON, and DOP in pore water at Incline relative to Marlette may be due to the stagnant backwater pond that forms at Marlette that acts as a sink for organic matter that is subsequently buried by sediment deposition. As channeled surface water has higher velocity at Incline, it could be that less organic matter collects at the SLI. Organic matter could also be deposited at the SLI from the lake; however, this does not explain the differences between these SLIs. Bulk sediment organic matter content was not measured at either SLI, thus differences in dissolved organic matter may be due to differences in biological communities that support desorption and mineralization process, differences in chemical makeup of sediment, or less organic matter beneath the channel [*Hart et al.*, 1994]. Slightly higher concentrations of DIN and DIP in Incline Creek surface water relative to Marlette Creek surface water indicate that the main differences in nutrient concentrations in shallow pore water discharge to the lake are likely due to conditions at the SLI.

The results from this work are limited to the study of only two SLIs in the Lake Tahoe basin. However, these SLIs are similar in physical characteristics to other locations around Lake Tahoe, as well as locations at many other subalpine lakes in general, suggesting transfer value of these findings. Strong contrasts in water quality and nutrient loading to the lake are apparent for these two SLIs and these differences are likely caused to a large degree by differences in geomorphology associated with engineered channelization of the stream at the SLI. Other factors are important, such as sediment heterogeneity that affects SW-GW interaction.

Another important finding is that there appears to be much more organic nutrients stored within the Marlette barrier beach than in the streambed at Incline, suggesting that channelization induces greater mobilization of sediment and corresponding removal of organic material. Further work is required to better understand erosional and depositional processes and sources of organic material at the SLI. This study clearly shows that the SLI is a location of focused nutrient processing that is important for impacting nearshore lake water quality and associated ecological conditions. Export of organic material to the lake bed may impact other processes within the lake, such as increasing the chances of internal loading of nutrients to the deep lake water. As this study is a first exploration of nutrient relations at modified and unmodified SLIs, additional work is needed to further elaborate on the findings presented herein and to determine how broadly these findings apply to other lake systems.

5. Conclusions

The stream-lake interface (SLI) is a unique environment that can have an important effect on near shore lake water quality and ecological systems that are sensitive to nutrient inputs. Synoptic nutrient sampling and hydrologic monitoring over a 2–3 day period provided a means of estimating pore water mean residence times and nutrient loading to a lake from pore water flowing through a barrier beach and channelized streambed. Subsurface nutrient loading from an SLI that is not channelized is significant and originated from biologically mediated reactions with organic deposits present in the beach sediment. At each of the SLIs, the concentrations of both dissolved phosphorus and nitrogen changed over the same portions of the subsurface flow paths, indicating that organic matter stored within the sediment was likely the principal source of nutrients in pore water. Channelization of the SLI increases SW-GW mixing and increases DO and appears to dilute nutrients in pore water upwelling into the lake, relative to the nonchannelized SLI. Channelization appears to increase sediment mobilization at the SLI due to greater surface water velocities, as indicated by less dissolved organic nutrients in the streambed. Channelization also appears to reduce nutrient concentrations in groundwater discharging to the lake; however, this is likely balanced by a greater amount of organic nutrients entering the lake during episodic erosion of the stream mouth during high flow events. Studies of other SLIs would help strengthen the results of this present study to determine the robustness of these findings in representing these systems more broadly around the world. This study highlights the SLIs influence on nutrient fluxes to the nearshore zone and possible impacts on primary production of algae and more broadly on cultural eutrophication of lakes.

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Erratum

The originally published version of this article misstated Dr. Rosen’s institutional affiliation. The error has now been corrected and this may be considered the official version of record.