EVIDENCE OF GEOLOGIC PHOSPHORUS FROM GROUNDWATER SEEPAGE TO NEWNANS LAKE, FLORIDA

By

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A THESIS PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
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To my parents

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Eutrophication due to phosphorus (P) enrichment is viewed as a critical problem in Newnans Lake, a large (27 km^2) and shallow (mean depth = 1.6 m) lake, located east of Gainesville, Florida. While the watershed is underlain by the phosphatic clays of the Hawthorn Group, the role of this geologic source in surface water P enrichment is unknown; moreover, previous work suggests that surface water inputs fail to account for lake loading, suggesting that diffuse inputs or internal loading may play an important role in lake P enrichment. The objectives in this study were to enumerate a P budget for the lake, determine if groundwater seepage is a significant source of P to the lake, and determine if groundwater P loading is of geologic origin. Using a mass balance constructed from the gauged portions of the watershed for water and P revealed a water deficit of 1.48×10^8 m^3 (equivalent to the flow in Hatchet Creek) and a P deficit of 16,153 kg (equal to P inputs from Hatchet and Little Hatchet Creeks combined) from 1998 to 2008. The water deficit can plausibly be attributed to runoff from the ungauged portion of the basin, which comprises 37% of the total area. However, the estimate of P inputs from Prairie Creek Reach is not as certain, though I estimate an input of 21,516 kg, which more than accounts for the ungauged deficit. Several alternative P sources have been invoked to

explain the unaccounted for mass loading, including internal recycling from lake sediments and diffuse external loading from groundwater, which may be P enriched from interactions with the Hawthorn. Groundwater wells (n = 14) were installed around the lake perimeter and sampled monthly to estimate the magnitude and timing of groundwater flow and P loading. To ascertain if the P was geologic in origin, two natural tracers were used: fluoride (F), which is liberated along with phosphate during apatite weathering, and nitrate, which is simultaneously enriched when the P source is anthropogenic. The perimeter wells indicated net groundwater inflow of 140,375 m^3 from October 2007 to December 2008, which accounted for 0.3% of total water inflows. While P concentrations were high in most wells, and P concentrations across wells covaried strongly with F, indicating a geologic source, the net groundwater flux of P was actually out of the lake (50 kg over 15 months), indicating that lateral groundwater seepage is not an important source. Groundwater inputs to creeks via bank seepage were evaluated by longitudinal sampling of water in both Little Hatchet and Hatchet Creeks under various flow regimes; F and nitrate were used to infer the source of P. Longitudinal sampling in Hatchet Creek showed a significant positive relationship between F and P under baseflow conditions, and extremely high P concentrations in storm flow samples, both of which indicate the dominance of geologic P. A similar longitudinal pattern was not observed in Little Hatchet Creek despite extremely high concentrations of P and a strong association between P and F over time at a downstream location. The most likely explanation is that geologic P loading occurs upstream of the sampling locations. The apparent importance of geologic P loading presents a management challenge given recent adoption of regulatory thresholds that will necessitate watershed load reduction. Understanding ways in which human activities accelerate geologic P loading and managing in-lake processes (e.g. internal P recycling) may be important management strategies.

CHAPTER 1
INTRODUCTION

Background

As the need to regulate water quality has been recognized, the task of determining and attenuating sources of pollutants has become paramount. There are substantial uncertainties associated with source determination and resolving these is necessary for fair implementation of water quality restoration. Eutrophication, a biological response in aquatic systems caused by excessive nutrients, is the most common impairment of surface waters in the United States (US EPA, 1990). For most shallow lakes, excess phosphorus (P) inputs are the primary cause of eutrophication (Schindler, 1977; Smith, 1998). Eutrophication due to high P concentrations is the primary management challenge in Newnans Lake, a large (27 km^2) and shallow (mean depth of 1.6 m) lake located approximately 10 km east of Gainesville, Florida (Figure 1-1) (SJRWMD, 2002; Gao and Gilbert, 2003).

The reversal of eutrophication in freshwater lakes, such as Newnans Lake, is one of the most widespread challenges in water quality management (Spears et al., 2007). Internal P loading, the flux of P from enriched sediments to the water column, induces potentially long lag times in restoring freshwater lakes that have received, and thus stored, high P loads for long periods of time (Kroski-Vahala and Hartikainen, 2001). P that was accumulated in the sediment during a period of high P loading needs time to re-equilibrate with a new loading level, which causes a delay in restoration (Sondergaard et al., 2003).

The importance of internal P loading depends on the flushing rate, loading history, and the chemical characteristics of the sediment (Marsden, 1989). Flushing of sediment-released P until equilibration between the sediment and water column occurs is one solution (Spears et al., 2007). Other solutions to lake recovery include: establishing submerged aquatic vegetation so that turbidity declines, reducing nutrient loading to the lake, increasing water depth, increasing the rate of sedimentation (e.g. via addition of alum or by minimizing wind-induced resuspension), and removing organisms that recycle P quickly (Scheffer et al., 2001).

One short-term study of Newnans Lake found that the export of P from the lake via Prairie Creek was 280% larger than measured inputs from the two main surface water inflows, Hatchet and Little Hatchet Creeks (Nagid et al., 2001). They conclude that the P deficit is due to internal loading from P-rich lake sediments at low lake stage when resuspension by wind-induced turbulence is high, and they argue that this effect could be remediated by controlling lake stage. Ironically, a weir was in place on Prairie Creek between 1966 and 1991 (Gao and Gilbert, 2003) which was used to artificially stabilize water levels, possibly affecting P concentrations in the lake during that period. However, the Nagid et al. (2001) study did not quantify the water and P inflows from all gauged and ungauged surface water and diffuse groundwater inputs; the magnitude of these remains largely unknown. The observation that P export out of Prairie Creek greatly exceeds import through the tributaries (Nagid et al., 2001; Gao and Gilbert, 2003) could be a transient effect of depleting internal sedimentary stores, perhaps during periods of rapid flushing or low lake stage, or could suggest that there are other inputs of P in addition to measured surface water inflows.

A comparison between total lake water P concentration and lake stage in Newnans Lake (Figure 1-2) shows concentrations are strongly negatively correlated with lake stage. Several plausible explanations exist: one is increased P resuspension rates at low stage (i.e. internal loading); another is evapoconcentration of P during periods of low inflow and outflow; a third is that when lake levels are low, groundwater seepage into the lake increases in absolute and relative importance bringing in P due to the higher groundwater table (assuming the groundwater table rises and falls independently of the lake). Establishing which explanation holds in Newnans Lake is of clear relevance to water quality restoration.

Groundwater Seepage

Groundwater is increasingly recognized as a crucial linkage between terrestrial and aquatic habitats (Schneider et al., 2005) with the potential to substantially affect the elemental budgets of receiving water bodies. Groundwater flow into Newnans Lake is a potential source that could plausibly carry a high P load due to contact with the P-rich Hawthorn Group (Figure 1-3), a *Miocene clay/sand confining layer underlying the entire basin which is, incidentally, the principal source of mined P in North America. The exposure of this phosphatic material in some areas of the lake basin (KBN, 1993; Cohen et al., 2008) may lead to P enrichment that is* geologic in origin. Groundwater seepage is the most likely pathway for geologic P to get into the surface water both directly into the lake and indirectly via bank seepage into the creeks that drain to the lake.

In a previous study of groundwater seepage into a large, shallow lake, Schneider et al. (2005) observed no correlation between the soil composition and the groundwater flow rate. Schneider et al. (2005) also report rapid increases in groundwater flow rate with rainfall both onsite and within the watershed. This rapid response may hold for Newnans Lake as well because the soils are primarily sand, leading to a high hydraulic conductivity. On the other hand, the topographic gradients around Newnans Lake are relatively shallow, which may make the response rate slower.

A Florida Department of Environmental Protection study on Lake Lochloosa, located in the same physiographic province (Northern Highlands) as Newnans Lake found that naturally occurring P contributes a major load via groundwater pathways (FDEP, 2008). These findings were supported by a study that observed high radon levels in Lake Lochloosa, from which groundwater discharge was estimated to be 1.32×10^5 m^3/d (FDEP, 2008). More recently, greatly elevated, though highly variable, radon levels were observed in Newnans Lake (Burnett et al., 2007) suggesting a major groundwater contribution (1.46×10^5 m^3/d) to the lake; for reference, this represents a flow rate that is 5 times the gauged creek inflows, possibly implying substantial groundwater losses as well to ensure mass balance. Since Newnans Lake sits on top of the P-rich Hawthorn Group, verification of significant groundwater flow to the lake could also explain the observed discrepancies between P outputs and inputs.

Study Area

Newnans Lake is located in the Central Valley region where lakes tend to be large, shallow, and eutrophic (Griffith et al., 1997). The lake sits at approximately 20 m above sea level (SJRWMD, 2002) at Latitude 29°38’42” and Longitude 82°13’8” (Gao and Gilbert, 2003). The land area surrounding Newnans Lake is characterized by gentle slopes and soils derived from Pliocene phosphatic, clayey sands (SJRWMD, 2002). The lake has two main inflows, Hatchet Creek and Little Hatchet Creek, and one outflow, Prairie Creek. Hatchet Creek is the larger of the two main tributaries, and it flows through swamp and wet flatwood areas north of Gainesville, while its tributaries run through Austin Cary Memorial State Forest (SJRWMD, 2002). Little Hatchet Creek drains wet flatwoods and industrial areas north of Gainesville, and then flows into Gum Root Swamp before entering Newnans Lake (SJRWMD, 2002). The drainage basin of Newnans Lake is mostly forest (55%) with 17% wetlands, 10% urban, 8% water, 6% agriculture, 2% rangeland, and 2% transportation associated with the Gainesville Regional Airport (Lasi and Shuman, 1996).

The geology of the area is dominated by the Hawthorn Group, which acts as a confining layer separating the Floridan aquifer from the surface water (KBN, 1993). The Hawthorn Group, deposited during the Miocene period approximately 20 million years before present, underlies southeastern South Carolina, the lower coastal plain of Georgia, and most of Florida and consists of phosphatic sands and clays (FDEP, 2008). Several intermediate aquifers occur within the Hawthorn Group material; the intermediate aquifer is the source of water for some drinking wells (FDEP, 2008) and along with the surficial aquifer, a possible source of groundwater seepage in the Newnans Lake watershed.

Groundwater and surface water interactions are most likely dependent on the elevation of the groundwater relative to the surface water and the extent and thickness of the Hawthorn Group confining unit (FDEP, 2008). The Hawthorn has also been exposed by stream incision in several areas along Hatchet and Little Hatchet Creeks, principally in the lower reaches of the Newnans Lake basin (KBN, 1993); exposure of the phosphatic clays suggests that P inputs into Newnans Lake may be geologic in origin.

Much of the geologic P in the Hawthorn is stored as fluorapatite (Lazareva and Pichler, 2007); when this mineral is weathered, it releases phosphate and also fluoride (F) ions. As such, a positive correlation between soluble reactive P (SRP; the biologically available fraction of P), and F is presumed to be indicative of geologic P loading and can be used as a natural tracer of geologic P. This tracer is most useful when municipal water supplies, which are fluoridated, are not discharged to surface waters; the low-intensity land use and absence of major municipal water and wastewater treatment outfalls in the Newnans Lake watershed make application of this approach possible.

Management

Newnans Lake is a Class III Freshwater body, with a designated use for recreation and promotion of fish and wildlife (Gao and Gilbert, 2003). With the exception of Newnans Lake, most of the water bodies within the Orange Creek Basin are listed as fair to good quality, and several are even listed as Outstanding Florida Waters (SJRWMD, 2002). In 2003, Gao and

Gilbert (2003) conducted a study on Newnans Lake and its watershed in order to establish a total maximum daily load (TMDL) that has guided subsequent P reduction allocations. In order to set the TMDL, Gao and Gilbert (2003) first examined water quality data, particularly trophic state indices (TSI). The Florida specific TSI is based on chlorophyll-a, total nitrogen, and total phosphorus concentrations and is used to describe the trophic state of any given lake. Over the last 20 years, water quality data in Newnans Lake has shown accelerated eutrophication and TSI (SJRWMD, 2002), reaching a high TSI of 97 in 1999, possibly due to lowered water levels as a result of a major drought and the removal of the weir on Prairie Creek (Gao and Gilbert, 2003).

Past lake management most likely affected P concentrations within the lake, particularly through the water control structure on Prairie Creek. The weir on Prairie Creek was installed in 1966 by the Alachua County Recreation and Water Conservation and Control Authority to increase water levels in the lake (Gao and Gilbert, 2003). The weir was altered in 1976 by the Florida Game and Fresh Water Fish Commission to include removable flashboards, but these boards were removed in 1991 and the weir was completely removed in 1999 (Gao and Gilbert, 2003). Notably, most of the historical data that define the baseline conditions in the lake were obtained during the 1980s when water levels were artificially constrained by the flow control structure on Prairie Creek.

During the years the weir was in place, Newnans Lake ranged from being mesoeutrophic to eutrophic. Due to the relationship between total phosphorus (TP) in the water column and lake stage (Figure 1-3), while water levels were kept high by the weir, P concentrations were most likely lower than they were historically. Moreover, Kenney et al. (2002) showed increasing P concentrations with depth in the lake's sediment, which is highly uncommon. In general, P concentrations decrease with depth because recent loading is almost always higher than historical loading. These findings indicate that P concentrations in the lake were lowest within the long-term record just prior to sampling (between 1995 and 1997). Gao and Gilbert (2003) used baseline data during the time when the weir was in place to recommend a TMDL, which may complicate the successful implementation of strategies to meet water quality standards.

The TMDL for TP was recommended by Gao and Gilbert (2003) and later set by the Orange Creek Basin Working Group (OCBWG) in 2007 at 62 ppb, which would require a reduction in annual P loading from 11,672 kg to 4955 kg, or a 59% reduction. To achieve P reduction goals set by the TMDL, it is imperative that we develop a quantitative understanding of the pathways and magnitudes of P loading to the lake. Discriminating between internal loading, inputs from ungauged surface water pathways, and diffuse groundwater loading is important for selecting effective management strategies. The objectives of this study were to (1) quantify P and water budgets for the lake, and identify the temporal dynamics of P inputs and outputs and ; and (2) quantify the groundwater P load to the lake via both diffuse lateral seepage and bank seepage along the tributaries.

Research Objectives

Research Objective 1

Construct water and P budgets for Newnans Lake to identify significant sources of water and P entering and leaving the lake.

Hypothesis 1: Using only archival gauged creek data, both water and P deficits (more output then input) will be observed in Newnans Lake and the magnitude of the resulting P deficit is large in comparison with the water deficit.

Rationale: According to Nagid et al. (2001), the P export from Newnans Lake through Prairie Creek is 280% higher than the P import through Hatchet and Little Hatchet Creeks. They attribute the P deficit to internal loading (i.e. diffusive fluxes from the sediments into the water column). However, if a water deficit exists, this observed P deficit may be due in part to P loading from unmeasured sources of water; internal loading would not affect the water budget. Predictions of the water and P budgets include:

- Both surface water and P outputs will exceed inputs into the lake, creating water and P deficits that are large compared with prediction uncertainty.
- The magnitude of the P deficit will be large compared to the magnitude of the water deficit.
- The water and P deficits will be large during dry periods (when the lake stage is low and concentrations are high), and small during wet periods.

Research Objective 2

Quantify the magnitude and timing of groundwater and associated P fluxes into the lake, and determine whether the P is from the Hawthorn Group or anthropogenic sources.

Hypothesis 2a: Groundwater is an important source of water to Newnans Lake, particularly during periods of low lake stage.

Rationale: Previous research on Newnans Lake suggests that there is a large water deficit in the lake when only surface flows are taken into account (Nagid et al., 2001; Burnett et al., 2007). During low lake stage, the groundwater table is expected to be higher than the lake, driving groundwater flow into the lake. At high lake stage, the lake is expected to be higher than the groundwater table, and the lake water will recharge the groundwater. Predictions of groundwater input to the lake include:

- Groundwater flow will be into the lake (i.e. a source of water, not a sink), and of sufficient magnitude to impact the overall water budget of the lake.
- During low lake stage conditions, the groundwater will be higher than the lake level, and water will flow towards the lake (Figure 1-4).
- At high lake stage, groundwater flow will reverse and water will flow out of the lake (Figure 1-4).

Hypothesis 2b: The P in groundwater seeping into the lake is of geologic origin.

Rationale: Determining the source of P in groundwater is required to set management strategies; a principally geologic P source would necessitate strategies that are quite different than from an anthropogenic source. Since the Hawthorn Group is expected to be at or near the land surface around the entire perimeter of Newnans Lake and because land use intensity in the vicinity of the lake is comparatively low (principally plantation forest land), anthropogenic sources are expected to be small compared to geologic sources. Predictions of geologic P include:

- A positive correlation between P and F concentrations (due to fluorapatite weathering from the Hawthorn Group yielding both phosphate and dissolved fluoride) will indicate a geologic source of P.
- The absence of a covariation between P concentrations and nitrate, elevated levels of which are principally of anthropogenic origin, will further indicate the importance of geologic loading.

Hypothesis 3: Groundwater seepage along stream banks is an important pathway for geologic P delivery to Newnans Lake via Hatchet Creek and Little Hatchet Creek, the main tributaries. The loading of P will vary along different flow regimes.

Rationale: The Hawthorn Group is near the surface or exposed in both Hatchet and Little Hatchet Creeks. This relationship suggests that groundwater seepage may be contributing a significant amount of water into the creeks. Predictions of bank seepage include:

- During baseflow, the streams accumulate water and P longitudinally, particularly in areas where the streams are actively incising into the Hawthorn Group (Figure 1-5).
- During storm flows, both absolute concentration and longitudinal increases in P will be reduced due to dilution from precipitation or reduced groundwater/bank seepage (Figure 1-5).
- High P concentrations in the tributaries during storm flows will be observed, suggesting that groundwater may be seeping into the tributaries instead of directly into the creeks.
- A longitudinal positive correlation between P and F concentrations in the creeks will exist during low and moderate baseflow conditions.

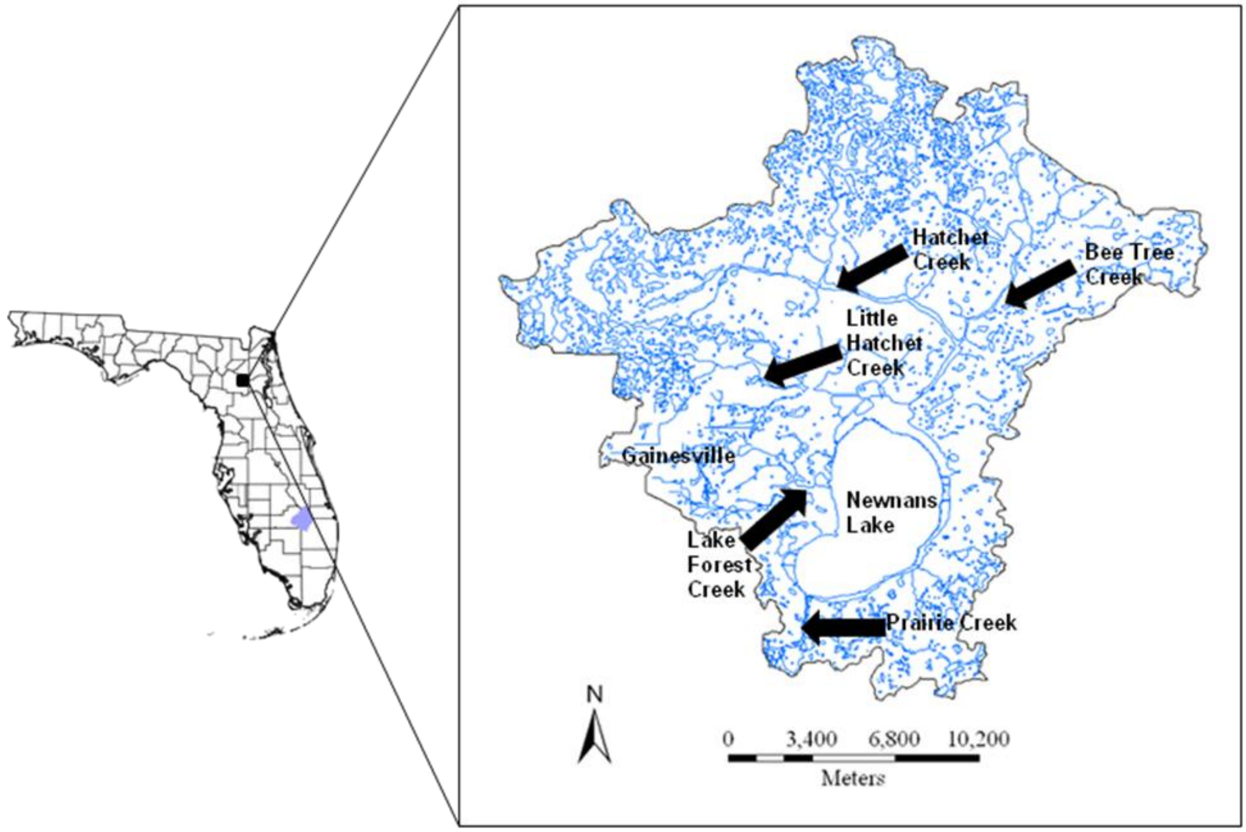


Figure 1-1. The location of Newnans Lake in Alachua County, Florida.

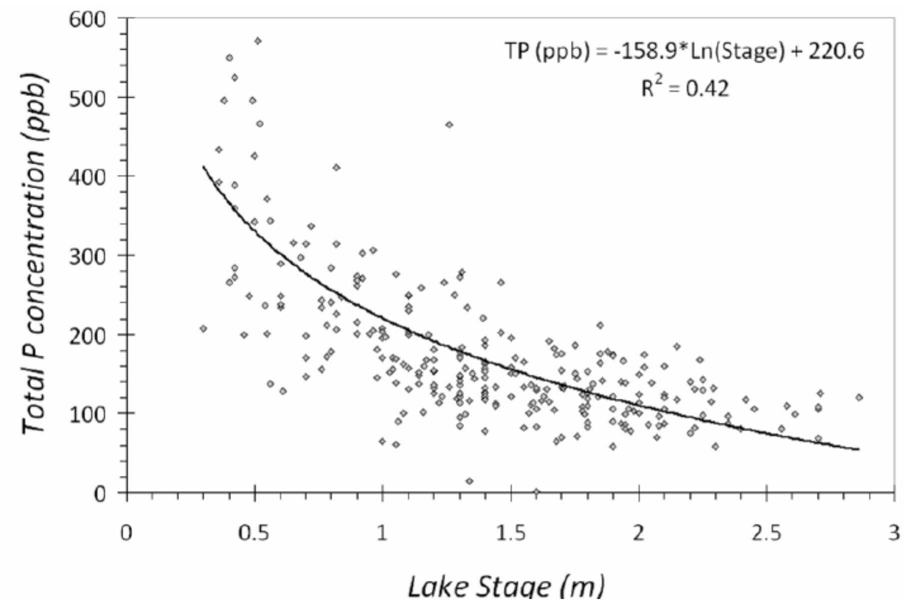


Figure 1-2. A comparison between lake water elevation and total P concentration in Newnans Lake (Cohen et al., 2007).

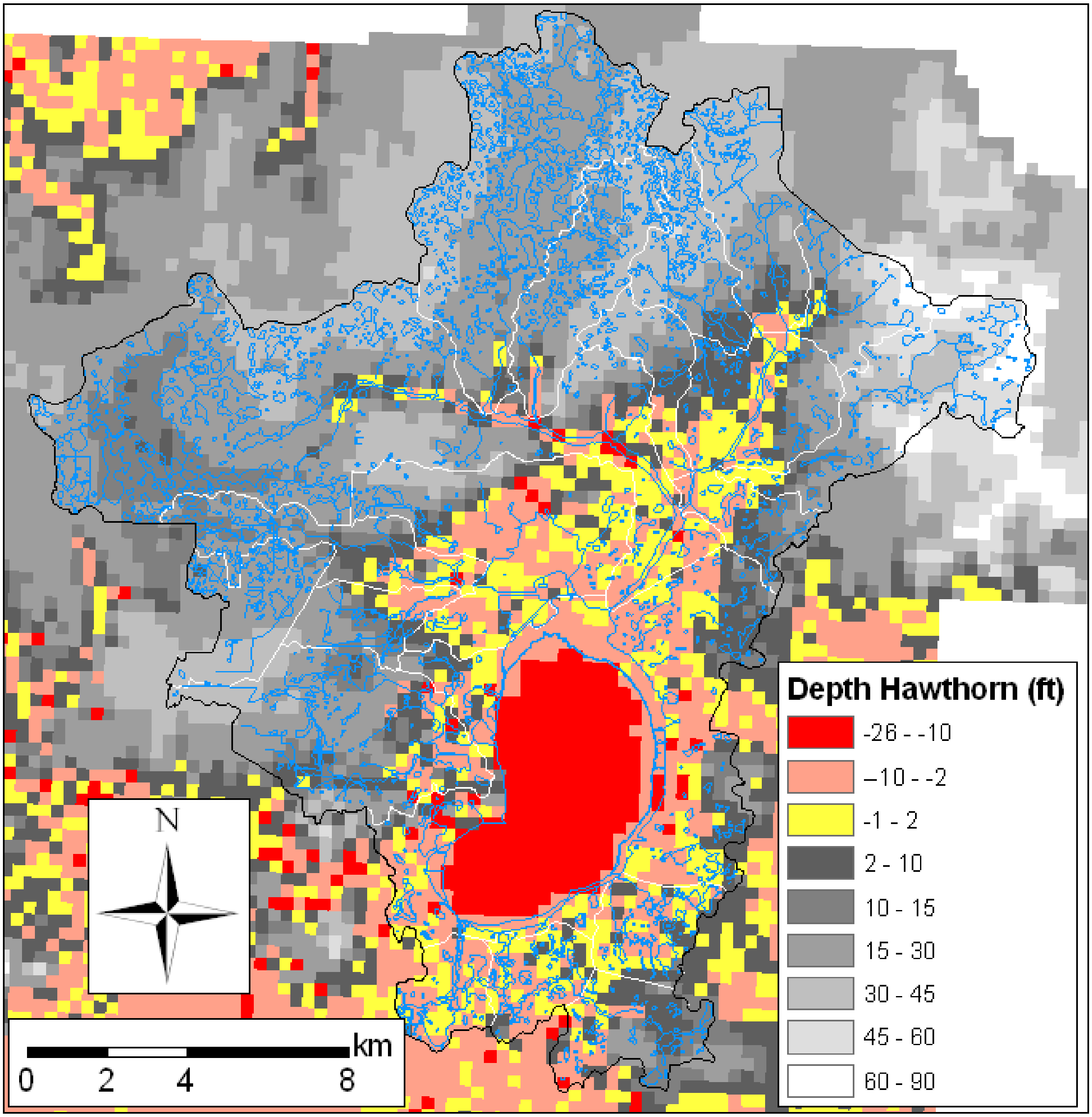


Figure 1-3. Depth to Hawthorn layer in the Newnans Lake watershed. Pixels are 100m x 100m (Cohen et al., 2008). The negative numbers indicate areas where the Hawthorn is exposed at the soil surface.

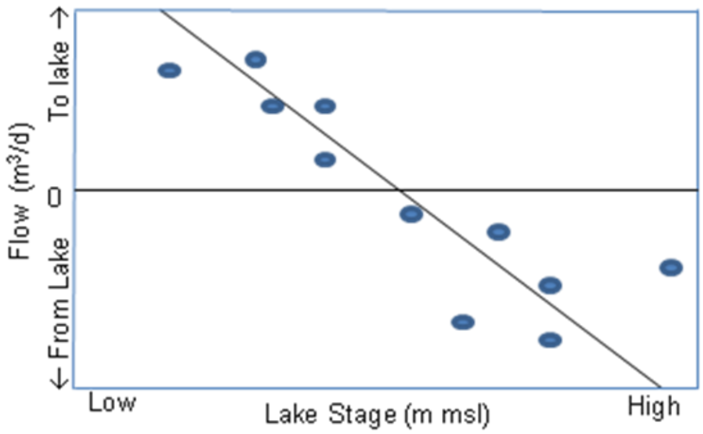


Figure 1-4. Expected trend in groundwater seepage to Newnans Lake, assuming that lake levels fluctuate independently of groundwater levels.

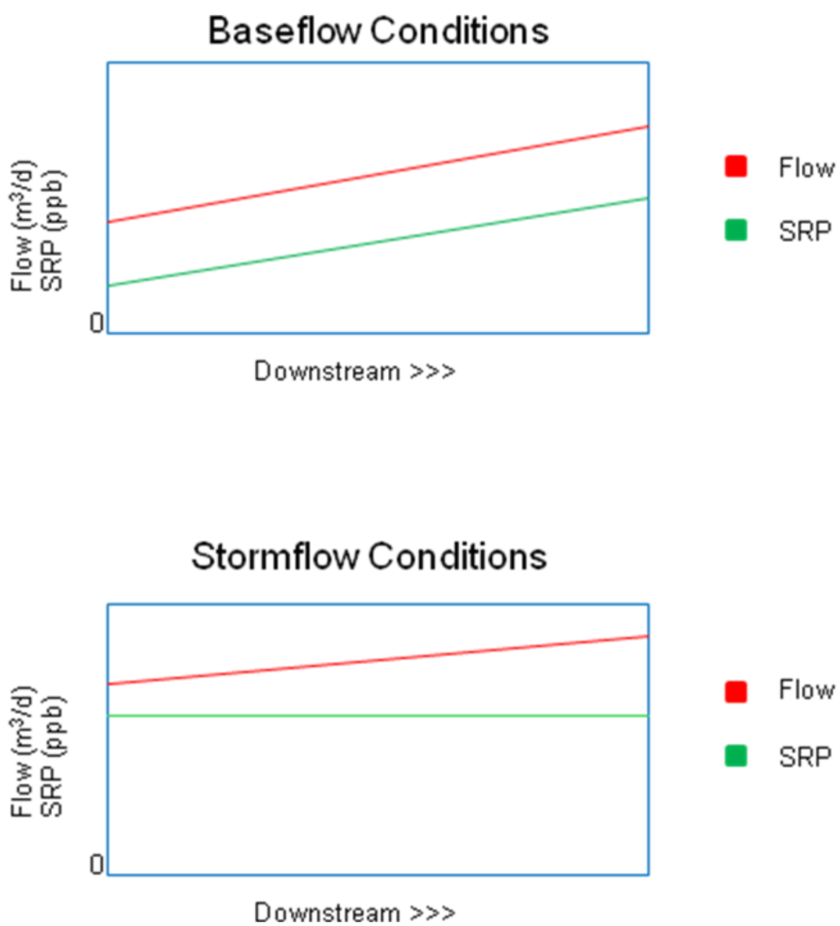


Figure 1-5. Diagram showing the expected trend in surface water flow and SRP concentrations in both Hatchet and Little Hatchet Creeks. Longitudinal increases are expected during baseflow conditions, while during storm flows, surface water flow and SRP concentrations will be high, but there will no longer be a longitudinal trend.

CHAPTER 2
METHODS

Water and Phosphorus Budgets

Data on inputs and outputs of water and P for Newnans Lake (Figure 2-1), including lake stage, creek surface water flow and P import and export from October 6, 1998 through August 31, 2008 were used to create water and P budgets for the lake. Only years 2000, 2001, 2004, 2005, 2006, and 2007 had continuous data; 1999 had missing Bee Tree Creek flows and 2002–2003 had missing lake stage data due to a drought. Daily archival data for rainfall, ET, and surface water flows in the tributaries were obtained from St. John's River Water Management District (SJRWMD) (http://arcimspub.sjrwmd.com/website/dahds/design/index.html). Monthly archival P data (also obtained from SJRWMD) were used to parameterize a P loading model to estimate daily P inputs and outputs from the lake through surface waters. Data do not exist for the ungauged portion of the watershed, but daily flows can be inferred from flow concordance with runoff generated in other basins. Internal loading and groundwater seepage estimates are poorly constrained and remain largely unknown.

Lake stage was measured on the west side of Newnans Lake at gauge 04831007 (Figure 2-2). Of the four measured surface water inflows, Hatchet Creek (gauge 01950193), Little Hatchet Creek (gauge 02840233), and Bee Tree Creek (gauge 02850235) flow into the north end of the lake while Lake Forest Creek (gauge 19244274) flows into the west side of the lake. The only surface water outflow, Prairie Creek (gauge 08631958), flows out of the south end of the lake. The gauge at Hatchet Creek receives surface water flow from an area of 91 km^2 of the basin; Little Hatchet Creek receives flow from 12.6 km^2 of the basin; Bee Tree Creek receives flow from 52.8 km^2 of the basin; and Lake Forest Creek receives flow from 7.9 km^2 of the basin. Prairie Creek Reach is the ungauged portion of the Newnans Lake watershed directly around the lake itself; it accounts for 114.6 km^2 of the basin, or 37% of the total contributing area to the lake (Figure 2-3).

Water Budget

Rainfall data were reported daily for the Gainesville Regional Airport. Figure 2-4 shows the yearly and monthly averages of rainfall during the period of record for the water budget. Rainfall amounts were multiplied by the lake area (27 km^2) to estimate the volume of direct rainfall inputs. Evaporation data from SJRWMD relies on one estimate ("Gainesville") for all of the lakes in the Gainesville area including Newnans Lake. Total evaporation for "Gainesville" was estimated by dividing the average daily evaporation measurements at Lake Lochloosa by the average daily evaporation measurements at Orange Lake, which gave a coefficient of 0.87. The coefficient was then used to estimate the evaporation for "Gainesville"; every day, the evaporation at Orange Lake is measured and then divided by 0.87. All of the inflow and outflow data at each of the four main inflow and one outflow stations were measured on a daily basis and assumed to be constant over that day.

Surface water flow for Lake Forest Creek was not measured from October 6, 1998 through October 3, 2005. In order to include flows for Lake Forest Creek in the water budget, measured flows from 2005 to 2007 for Lake Forest Creek and the other three inflow creeks (Hatchet, Little Hatchet, and Bee Tree Creeks) were plotted to establish a predictive relationship. A significant association ($R^2 = 0.89$; $p < 0.0001$) was found between the flow of Little Hatchet Creek and Lake Forest Creek and this relationship was used to fill in missing flow data for these two creeks (Figure 2-5). The equation was:

$$\text{Lake Forest Creek flow (cfs)} = 0.4968 * \text{Little Hatchet Creek flow (cfs)} - 0.3091$$

Any further missing flow data from all creeks were examined to determine whether the flows on those particular days were unaccounted for or were actually zero. When the days with missing data were surrounded by days of no flow, zero flows were assumed.

Phosphorus Budget

P fluxes from rainfall were assumed proportional to rainfall, with a mean P concentration in rainfall of 11.8 ppb (Ahn, 1999). Since archival water quality was only measured once a month, it was necessary to make assumptions about the P concentrations in flows between observations. While it would be ideal to use archival flow and chemistry data to develop predictive relationships, the archival chemistry data for the creeks are, with the exception of Prairie Creek, from different locations than the archival discharge data. Instead, models were developed for Hatchet Creek, Little Hatchet Creek, Lake Forest Creek, and Bee Tree Creek using statistical relationships between surface water flow and TP concentrations constructed based on monthly measurements from 2007 and 2008 where flow and water quality were measured in the same location. For Prairie Creek, archival data from 1998 to 2006 were used to develop the model. The methods detailing the collection and analysis of the water samples are described in the longitudinal sampling section of the methods chapter.

There is inherent error associated with using equations instead of actual P measurements and so a model validation was undertaken. The analysis for Hatchet Creek, Little Hatchet Creek, and Lake Forest Creek was conducted by regressing actual P measurements obtained from the archival data to P calculated using the model; both measurements were taken for the same day. Unfortunately, the data used to create the model were taken at a different location than the archival data, making the two measurements difficult to compare. Archival data do not exist for Bee Tree Creek and so no validation was undertaken for the model; since concentrations for that creek were uniformly low and remarkably constant, it seems reasonable to presume that the error associated with using this unvalidated model are small. For Prairie Creek, a model validation was not undertaken due to a lack of data with which to compare the model.

Ungauged Portion of the Watershed

An estimate of surface water inflows from the ungauged Prairie Creek Reach was determined based on the fraction of rainfall that becomes runoff in the Little Hatchet Creek basin. The Little Hatchet Creek basin was used because more archival data exist for Little Hatchet Creek than any of the other creeks. The equation was:

$$\text{Estimated Basin Runoff } (m^3/d) = \%\text{Runoff} * \text{Rainfall (m)} * \text{Area of Prairie Creek Reach } (m^2)$$

Where %Runoff is the percentage of rainfall that reaches Newnans Lake on any given day.

$$\%\text{Runoff} = \text{Little Hatchet flow } (m^3/d) \;/\; \text{Area of Little Hatchet basin } (m^2) \;/\; \text{Rainfall (m)}$$

In order to estimate P fluxes in Prairie Creek Reach, water samples were collected in various locations throughout the basin (Figure 2-6). Water samples were collected and analyzed using the same method described under the longitudinal sampling section.

Groundwater Wells

Large lakes, such as Newnans Lake, tend to exhibit variation in shoreline substrates, topography, aquifer characteristics, and other features that have been shown to influence groundwater seepage (Schneider et al., 2005). Schafran and Driscoll (1990) also found that flow may differ between sites only meters apart along lake shores. Changes in flow can affect the chemical composition of the discharging groundwater (Sebestyen and Schneider, 2004), and this may have an effect on the productivity and health of the shoreline plants (Hagerthey and Kerfoot, 1998). Assessing lateral groundwater inputs to Newnans Lake requires wells at numerous locations around the entire perimeter of the lake to account for varying topography, soils, and vegetation; moreover, because water levels in the lake and groundwater are expected to vary with rainfall inputs, regular monitoring of inputs is necessary.

Fourteen surficial groundwater well sites were installed around the perimeter of Newnans Lake (Figure 2-7). At each well site, one lakebed well was installed just off shore and at least one well was installed up-gradient on land. Sites were chosen mainly based on access by roads; all other wells were installed by boat using a Global Positioning System (GPS) in order to ensure that the entire perimeter of the lake was represented and wells were relatively evenly spaced. Wells were made from 1.8 m sections of 5.1 cm diameter PVC pipe. A well screen 5.1 cm in diameter and approximately 45.7 cm long was attached at the bottom of each well. Wells were installed using a post hole digger until the clay layer was reached (approximately 0.91 to 1.5 m); after well installation, holes were backfilled with sand to approximate ambient bulk density, and allowed to equilibrate.

To determine groundwater seepage to the lake, Darcy's equation was used:

$$Q = K * A * (H1-H2) / L$$

where Q is the groundwater flow rate, K is the hydraulic conductivity, A is the cross sectional area of the seepage face (Figure 2-8), H1-H2 is the potentiometric gradient between the upland well (H1) and lake stage (H2) (positive when flow is into the lake, negative when flow is outwards) (Figure 2-9), and L is the distance between the upland well and lake edge. Hydraulic conductivity was estimated using the Hvorslev method by performing a slug test; 3.79L of water was added to the well and the time response of the water returning to its original height was recorded. The time required to return to 37% of the original height was measured and then hydraulic conductivity was calculated using the equation:

$$K = r^2 * \mathrm{Ln}(L_e / R) / (2 * L_e * T_{37})$$

where K is the hydraulic conductivity, r is the radius of the well, R is the radius of the well screen, L_e is the length of the well screen, and T_{37} is the time required to reach the 37% point between the initial and baseline level. The cross sectional area was calculated as half the distance from one well to the next on either side of the well and multiplied by the average depth of the lake (1.5 m) assuming that groundwater seepage would not be occurring below that depth; since a dense clay layer was observed at 1–2 meters below the soil surface for the upland wells, this constraint on the depth of the seepage face is plausible.

Water elevations in the upland wells were determined using a laser level. A PVC well was installed into the lake bed just off shore and the laser level was used to determine the difference in height between the sampling well and the well off shore. The difference between the heights of the water in the two wells was calculated, thereby calculating the slope between the groundwater and the lake.

Wells were monitored monthly for 15 months (October 2007–December 2008). During each site visit, a water sample was pumped from the well into an acid washed 250 mL Nalgene plastic bottle using a small battery operated pump. One lake water grab sample from just off the shoreline was also taken at each point to compare to the well samples. One grab sample from the middle of Newnans Lake was obtained during each field sampling day. Samples were stored on ice during transfer back to the lab, and were filtered within 24 hours. The filtrate was placed in small 20 mL plastic Nalgene bottles, and frozen until analyzed for soluble reactive P (SRP), nitrate, fluoride (F), chloride (Cl), and sulfate. SRP was analyzed colorimetrically using EPA method 365.1 on an Aquamate spectrophotometer. Nitrate was measured using the standard cadmium reduction method at the Lakewatch Lab in the School of Forest Resources and Conservation at the University of Florida. Anion concentrations (F, Cl, and sulfate) were measured using a Dionex DX500 ion chromatograph in the Department of Geological Sciences at the University of Florida.

To predict flows of water, correlations between groundwater flow and lake stage, and groundwater flow and hydraulic conductivity were evaluated using ordinary least squares regression in Excel. Covariance between P and both nitrate and F concentrations to estimate the importance of the geologic and anthropogenic sources was evaluated using ordinary least squares regression in Excel. Correlations between SRP and groundwater flow were also evaluated. Finally, lake water chemistry (Cl and sulfate) was compared to well water chemistry to determine if the water at the lake edge was the same as water in the wells. The F statistic, degrees of freedom, R^2, and p values were noted; a relationship was determined to be significant if the p value was less than 0.05.

Longitudinal Creek Sampling

Transects along Little Hatchet and Hatchet Creeks (Figure 2-10) were sampled to determine if groundwater seepage was occurring directly into the creeks and whether the P in seepage water was of geologic origin. Additional points were sampled upstream and downstream of these transects in places that were easily accessible by road. Twenty-two samples were taken along the Hatchet Creek transect and 24 samples were taken along the Little Hatchet Creek transect. Sampling sites were situated at all locations where tributaries flowed into the creek, even if they were dry at the time of sampling; the remainder of samples was spaced out evenly. In addition to asking about the seepage into the creek longitudinally, I predicted differences in seepage between baseflow and storm flow conditions; specifically where longitudinal increases in P were expected from bank seepage during baseflow, similar patterns were expected to be overwhelmed during storm flows. To test these predictions, transects were sampled three separate times: during low baseflow (July 2007), moderate baseflow (February 2008), and storm flow (February 2008 for Hatchet Creek and April 2008 for Little Hatchet Creek).

Water samples were collected at 2/3 the water depth with an acid-washed pre-rinsed 100 mL Nalgene bottle at each sampling point. Samples were stored on ice and then taken to the lab to be filtered and then analyzed for SRP using the colorimetric method. Analysis for nitrate was conducted by the Lakewatch Lab in the School of Forest Resources and Conservation; F was analyzed in the Department of Geological Sciences. When water depths in the creek exceeded one meter, flow velocity was measured with a SonTek FlowTracker at a depth of 0.6 of the total water depth; all other times flow velocity was measured using the float method: for each sampling location, the cross sectional area was evaluated after dividing the stream in three equal width segments; velocity and water depth were measured at the center of each segment. The measured velocities were multiplied by 0.8 to account for vertical velocity differences.

Regression analysis in Excel was done to determine the significance ($p < 0.05$) of the association between F and SRP and nitrate and SRP with distance ultimately to determine whether P present is geologic in origin.

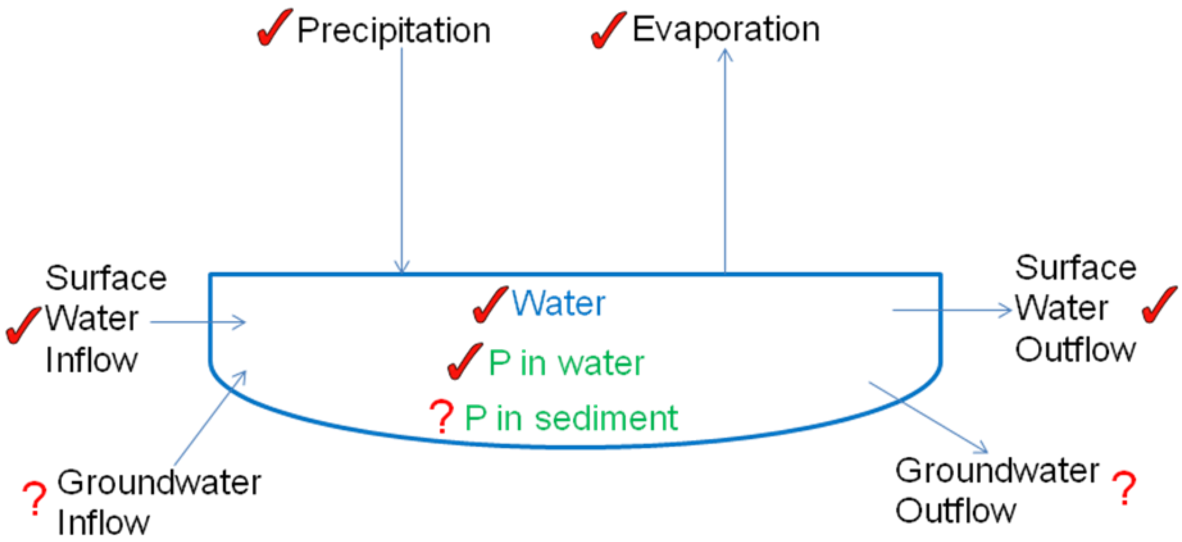


Figure 2-1. A diagram showing the inflows, outflows, and in lake stores of both water and P in Newnans Lake. The check marks indicate the parameters that are known based on data from SJRWMD; the question marks indicate unknowns.

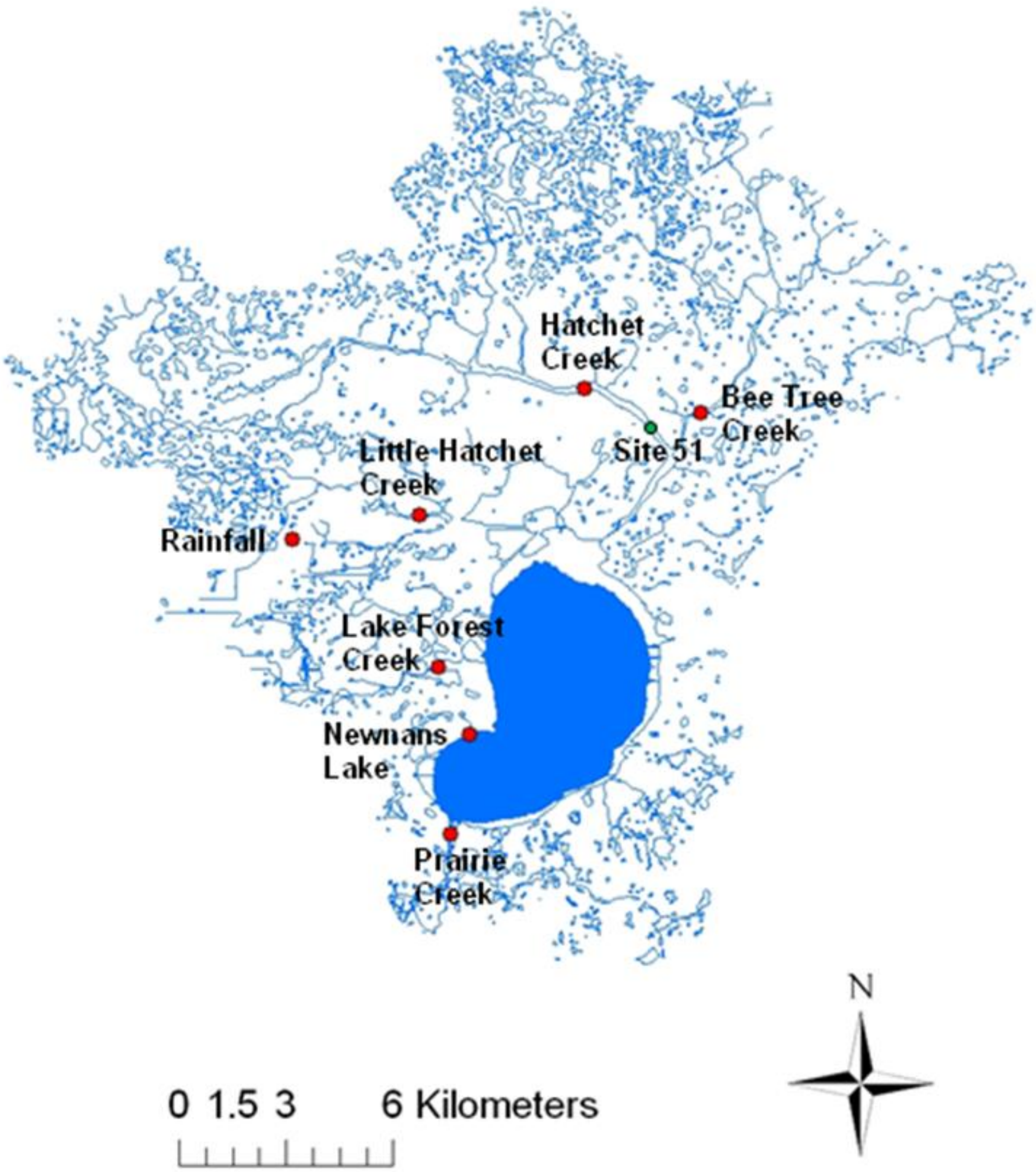


Figure 2-2. Location of the gauging stations used for the P and water budgets.

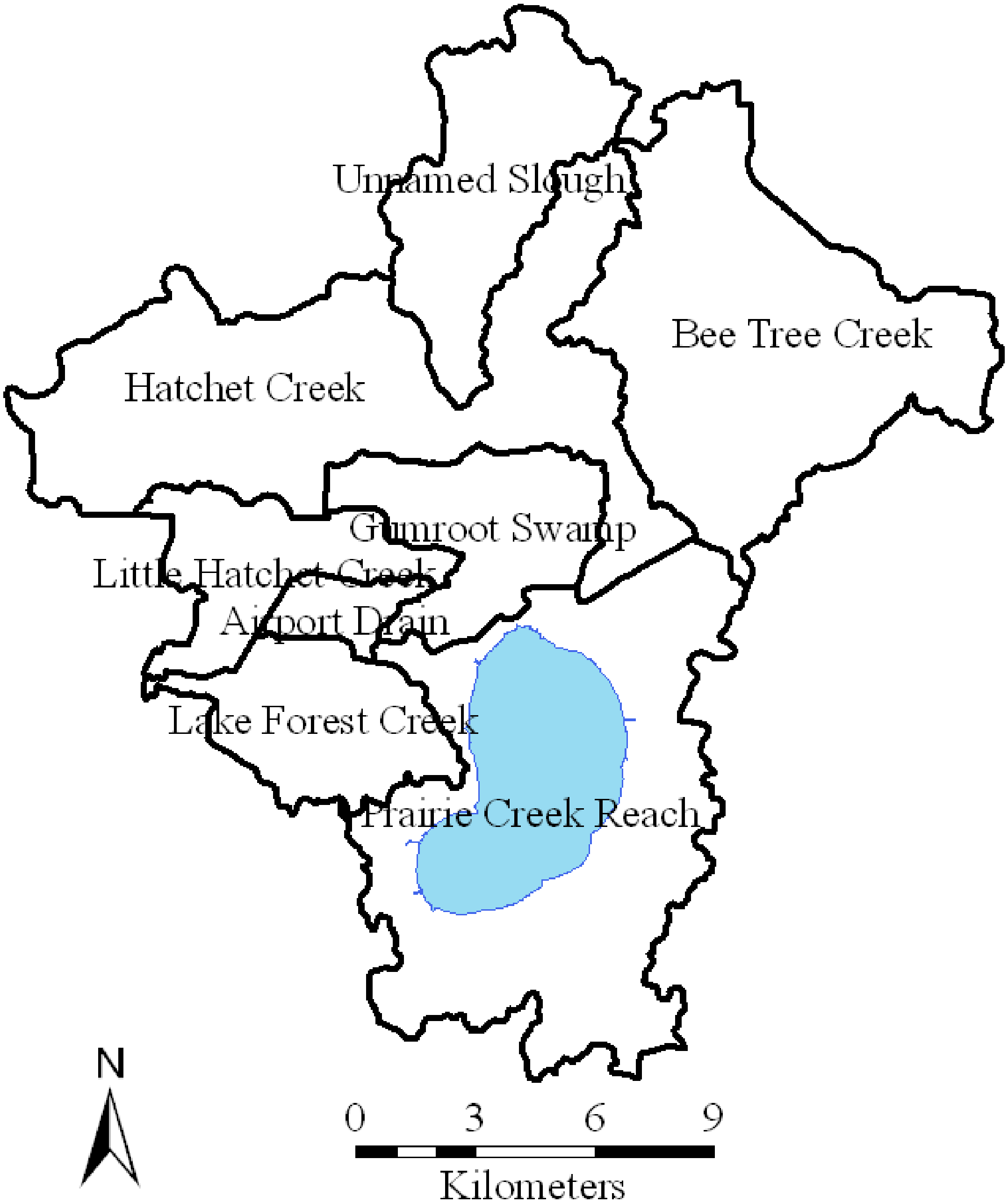
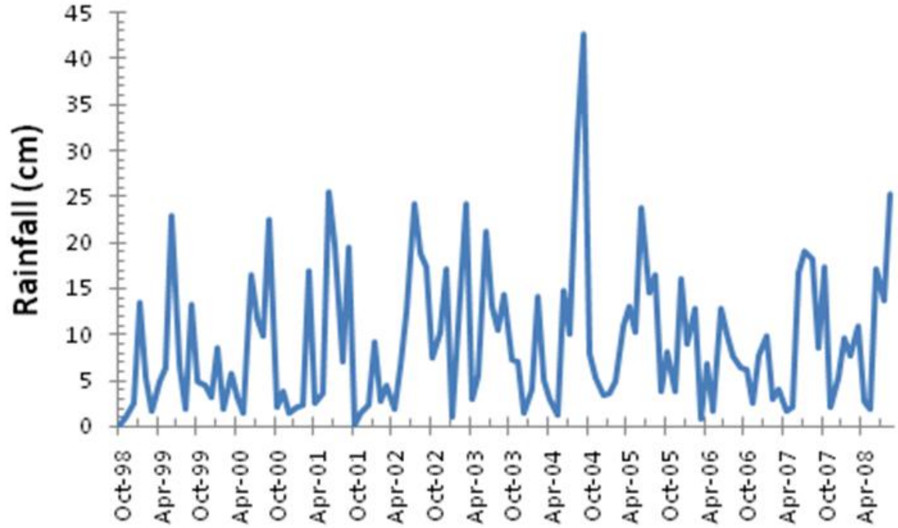


Figure 2-3. Major sub-basins in the Newnans Lake watershed.

A



B

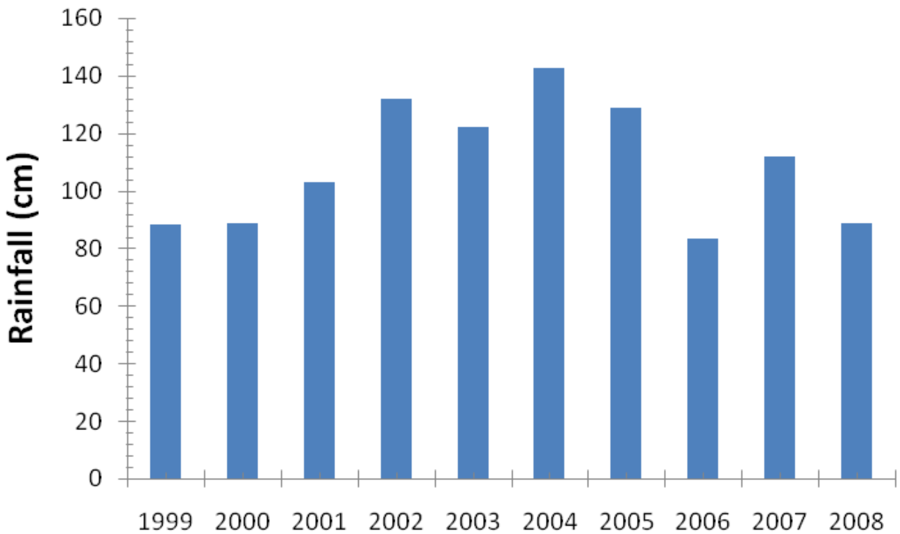


Figure 2-4. Rainfall in the Newnans Lake watershed for the period of record. A) Monthly rainfall over 10 years. B) Annual rainfall over 10 years.

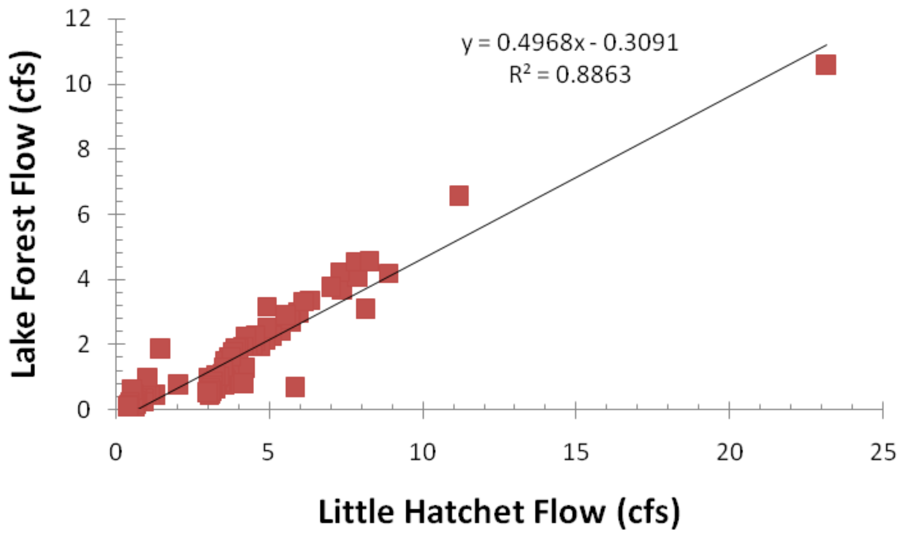


Figure 2-5. The relationship between the surface flows of Lake Forest Creek and Little Hatchet Creek.

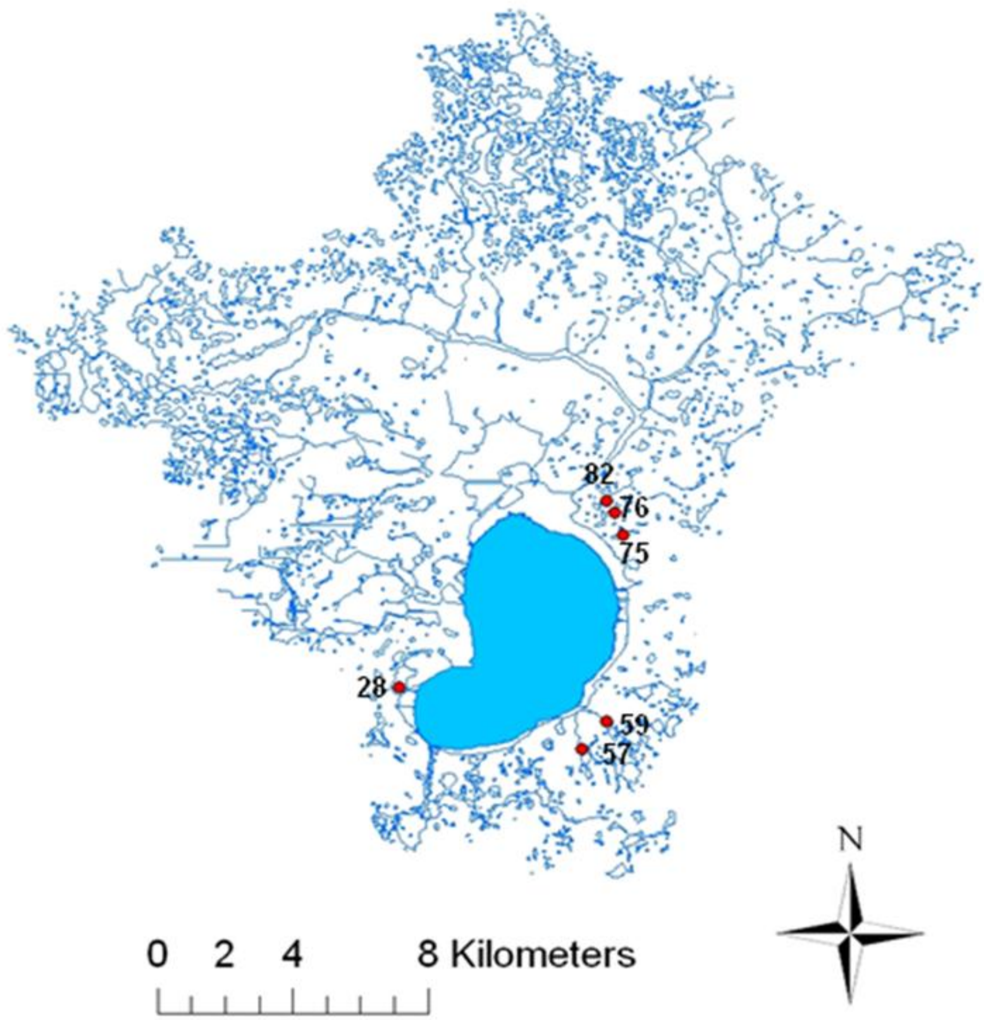


Figure 2-6. Sampling locations used for the estimated P fluxes in Prairie Creek Reach from March 12, 2008. *Sampling sites 82, 76, and 75 are located near a blueberry farm.

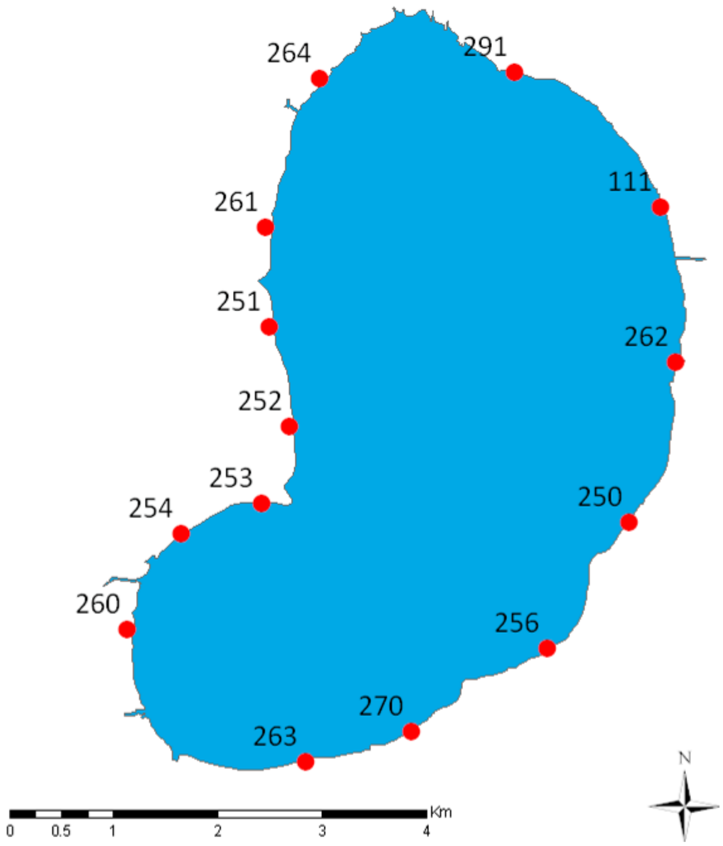


Figure 2-7. The locations of the 14 wells that were installed around Newnans Lake.

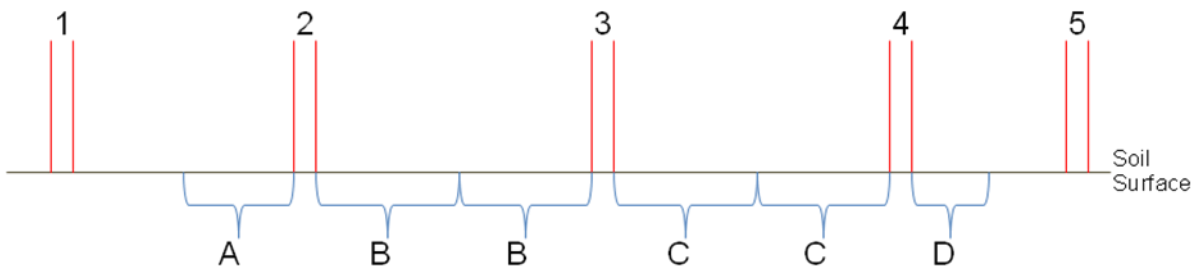


Figure 2-8. Diagram showing the calculation of the cross sectional area for each well. Well 2 is represented by the perimeter distance A + B so the seepage face is (A + B) * 1.5 m. Distances are in m and the average depth of Newnans Lake is 1.5m.

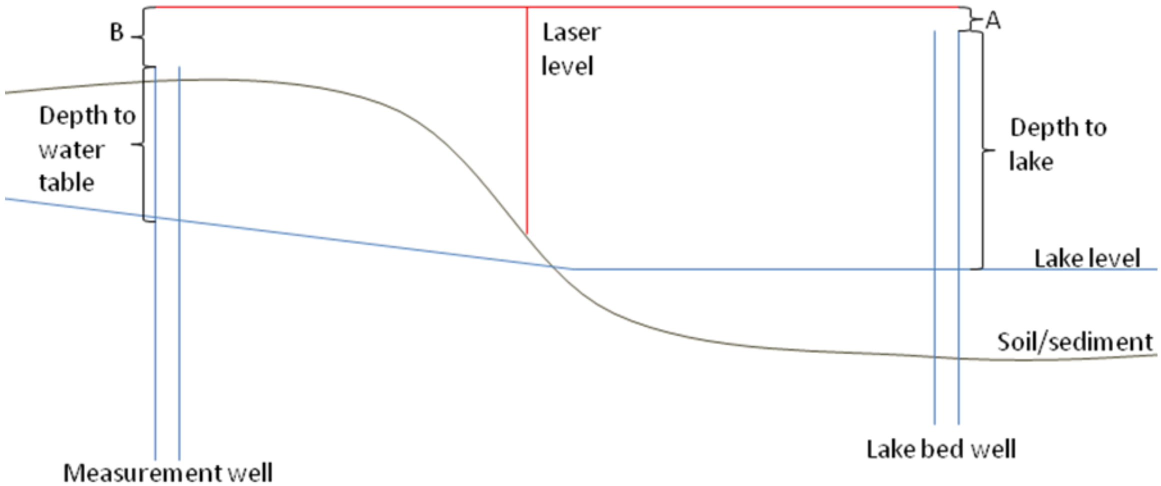


Figure 2-9. Diagram showing how to calculate piezometric water elevation (H1 – H2), which is the elevation of the groundwater compared to the lake level; positive when the water table is higher than the lake and negative when the water table is lower. The equation is: water elevation (H1 - H2) = [(A - B) + depth to lake] – depth to water table.

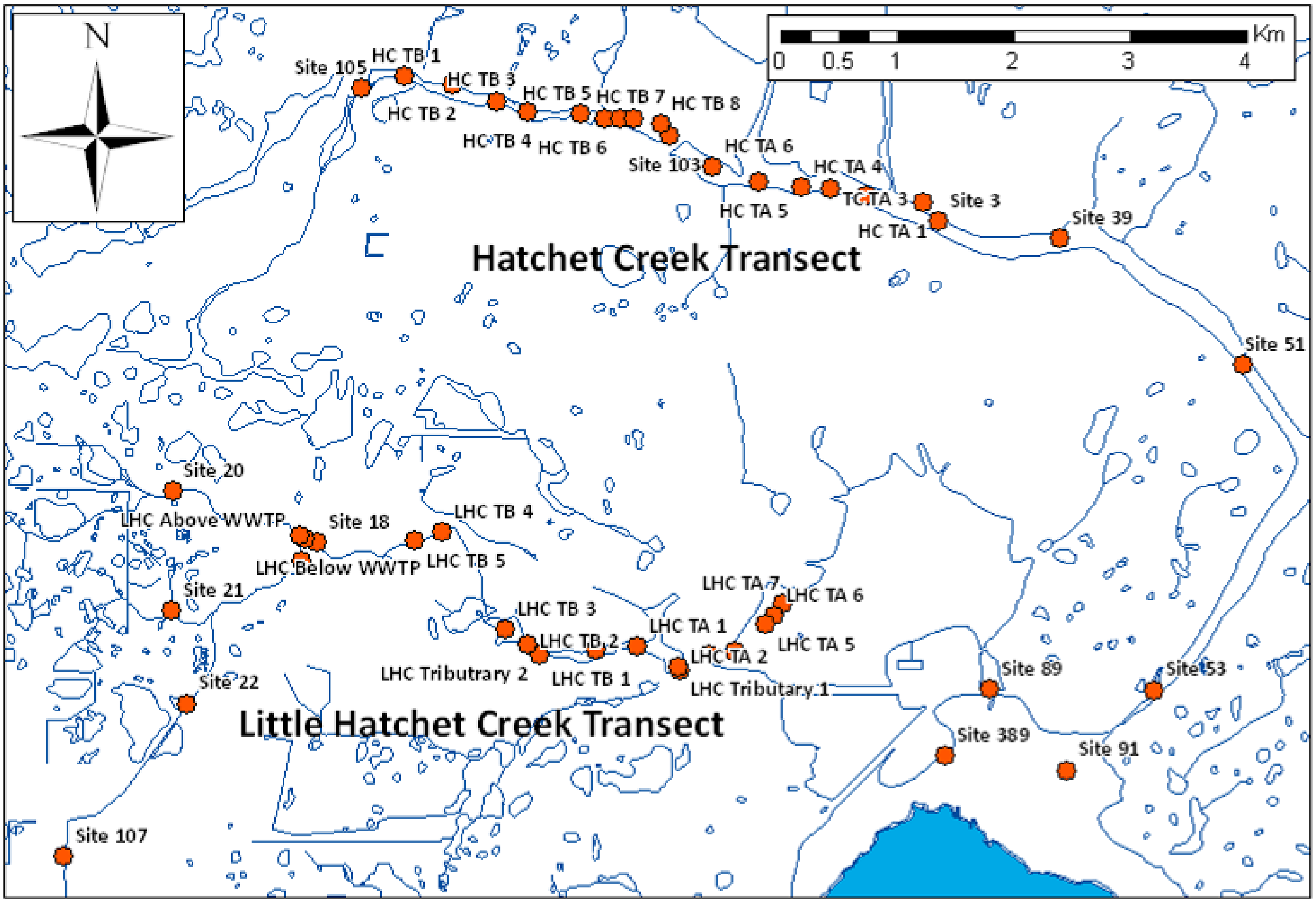


Figure 2-10. Locations of sampling points along Hatchet Creek and Little Hatchet Creek. *Site 201 where Bee Tree Creek enters Hatchet Creek is not located on the map.

CHAPTER 3
RESULTS

Phosphorus Budget Equations and Validation

The variables in the regression for the creeks include TP concentrations and flow rate at each individual point sampled (one site per creek). Due to a limited number of samples at the Hatchet Creek gauging station (5 samples), samples from both the gauging station and downstream site 51 (9 samples) were combined to create the equation (Figure 2-2). The equation for Hatchet Creek was ($R^2 = 0.55$, $p = 0.30$, Figure 3-1):

$$\text{TP (mg/L)} = (-)0.019 * \text{Ln(Flow cfs)} + 0.1575$$

The equation for Bee Tree Creek (based on 6 measurements) was ($R^2 = 0.62$, $p = 0.040$, Figure 3-2):

$$\text{TP (mg/L)} = 0.0673(\text{cfs})^{-0.11}$$

The equation for Little Hatchet Creek (based on 28 measurements) was ($R^2 = 0.50$, $p = 0.13$, Figure 3-3):

$$\text{TP (mg/L)} = 0.34 * \text{Flow(cfs)}^{-0.22}$$

The equation for Lake Forest Creek (based on 31 measurements) was ($R^2 = 0.02$, $p = 0.99$, Figure 3-4):

$$\text{TP (mg/L)} = 0.11 * \text{Flow(cfs)}^{-0.02}$$

The equation for Prairie Creek (based on 48 measurements) was ($R^2 = 0.30$, $p < 0.001$, Figure 3-5):

$$\text{TP (mg/L)} = 0.1684e^{-0.004 * \text{Flow(cfs)}}$$

Figure 3-6 shows the relationship between measured TP (archival data) and calculated TP (data developed from the regression model) for Hatchet, Little Hatchet, and Lake Forest Creeks. From the relationships, it appears that TP concentrations from Hatchet Creek and Little Hatchet

Creek are underestimated, but a comparison between archival flow and TP shows that the high TP concentrations occur at very low flows (Figure 3-7); under those conditions, P flux into the lake would be low. Comparing measured P flux and calculated P flux, it appears that calculated flux only slightly underestimates P flux for Little Hatchet Creek; flux in Hatchet Creek still shows an underestimation (Figure 3-8). Lake Forest Creek has relatively stable P concentrations (i.e. there is no relationship between P concentration and flow); although when calculated TP flux and measured TP flux are compared, it appears that P flux is underestimated. Although there is error associated with using the model for the creeks, it is only troublesome during low flow conditions, and it is the best option given the lack of archival data.

Water and Phosphorus Budgets

The archival data collected from SJRWMD (neglecting flows from the ungauged portion of the watershed) showed a cumulative water deficit (more water flowing out of the lake than in) over a period of 10 years of 1.48×10^8 m^3 (Figure 3-9). The deficit is approximately 1.2 times larger than the flow in Hatchet Creek, but half the flow of Prairie Creek over the same time period (Figure 3-10). The magnitude of the water deficit in Newnans Lake varied year to year (Figure 3-11); the budget seems to be driven by pulses. In 2004, an extremely large deficit was recorded compared to the other years, which may have been due to a decrease in lake stage and an increase in rainfall. In 2004, the highest annual rainfall out of 1998–2008 was recorded, but lake stage decreased from an average of 22.37 m msl in 2003 to 20.24 m msl in 2004. Since rainfall was increasing, more water was entering Newnans Lake, but lake stage was decreasing at the same time due to high outflows through Prairie Creek. The flow out of the lake through Prairie Creek nearly doubled from 2003 to 2004 (6.27×10^7 m^3 in 2003 to 1.04×10^8 m^3 in 2004). For the most part, Prairie Creek did not flow during the drought years from 2000 to early 2003; when rainfall started to increase in 2002 and 2003, there was a lag in Prairie Creek outflow and so lake stage rose rapidly. During 2003, Prairie Creek started to flow again and since the lake stage was so high, a lot of water flowed out of the lake in 2004 causing a large water deficit. The opposite trend appeared to happen in 2006–2007. There was a drought during these two years, and there was so little water inflow that average lake stage decreased from 2006 to 2007. Rainfall started to increase in 2007, but since the lake stage was low, the flow out of the lake through Prairie Creek was also low. The higher inflows due to the increased rainfall combined with the low lake stage caused the deficit to decrease and actually become a water surplus.

The flow for the unaccounted portion of the watershed, Prairie Creek Reach, was estimated to be 1.67×10^8 m^3 for 1998–2008. This estimate is only slightly greater than the water deficit. Prairie Creek Reach comprises a large portion of the watershed (37%) and so it would be expected that Prairie Creek Reach also contributes a proportional amount of water to the lake. In fact, Prairie Creek Reach is 21% larger than the Hatchet Creek sub-watershed, and the deficit is 20% larger than the flow in Hatchet Creek (Figure 3-12), indicating that it is likely that Prairie Creek Reach is the source of the unaccounted for water that comprises the deficit (Figure 3-13). Furthermore, the predicted Prairie Creek Reach flows loosely follow the trend in the water deficit. In 2004, there was the largest recorded deficit, and so I would expect Prairie Creek Reach to have the highest flows during that year in order to make up for the deficit. The highest predicted Prairie Creek Reach flows did in fact occur in 2004 (Figure 3-14). When the estimated daily flows from Prairie Creek Reach were added to the water budget (Figure 3-15), a cumulative water deficit of 2.09×10^7 m^3 over 10 years is predicted; this is approximately 48% of the volume of Newnans Lake. This is a small fraction of water considering that the deficit without Prairie Creek Reach is 3.4 times the volume of the lake. Since the water budget closes with the addition of the Prairie Creek Reach flows, all subsequent analysis includes Prairie Creek Reach.

The P budget for the lake between 1998 and 2008 indicates a cumulative P deficit of 16,153 kg (Figure 3-16). The P deficit is approximately equal to the estimated inputs from Hatchet Creek and Little Hatchet Creek combined (Figure 3-17). The P deficit (or surplus) varied greatly from year to year (Figure 3-18); five years (1999, 2000, 2001, 2002, and 2007) had P surpluses compared to one year that had a water surplus (2007). The five years that had P surpluses were all drought years, but 2006 was also a drought year and it had a P deficit. It appears that the driving factor in the P budget is whether or not Prairie Creek is flowing and not necessarily whether it is a wet or dry year. For the five years that showed P surpluses, Prairie Creek stopped flowing at the end of 1999 until the beginning of 2003, and in 2007 Prairie Creek had low flows in January and February and no flow during the summer months. The largest P deficits occurred in the years 2003–2006, which also corresponded to the highest Prairie Creek flows.

Prairie Creek Reach is also expected to contribute P into Newnans Lake, but it is not clear whether it can make up the entire P deficit. The flow-weighted concentration of the deficit is 109 ppb, which is approximately the same as the P concentration in Lake Forest Creek (Table 3-1). Prairie Creek Reach is 21% larger than the Hatchet Creek sub-watershed, but in order to account for the P deficit, Prairie Creek Reach would have to contribute 44% more P than Hatchet Creek (Figure 3-11). With the exception of a small area near a blueberry farm in the Northeast portion of Prairie Creek Reach (average P concentration = 362 ppb), the average P concentration in Prairie Creek Reach is 98 ppb. Water was sampled near the blueberry farm on one day at three sites. The summed flow of those sites on March 12, 2008 was 8808 m^3/d and the summed P flux was 2.83 kg; extrapolating this number over 10 years gives a flow of $3.2x10^7$ m^3, which is 22% of the water deficit and a P flux of 10,316 kg which is 64% of the P deficit. Assuming the other 78% ($1.12x10^8$ m^3) of the water deficit comprises the remainder of Prairie Creek Reach, the P flux would have to be 5837 kg which corresponds to a flow-weighted concentration of 52 ppb. The average concentration of the three nonblueberry sites sampled on March 12, 2008 was 100 ppb, and assuming a cumulative flow of $1.12x10^8$ m^3, the P flux over 10 years is 11,200 kg, which is much larger than the P that is unaccounted for (Figure 3-19). The estimate of the input from Prairie Creek Reach (21,516 kg) is larger than the observed P deficit of 16,153 kg possibly due to sedimentation in the lake. Since the estimation of P flux from Prairie Creek Reach is uncertain, all subsequent analyses do not include Prairie Creek Reach.

I predicted that water and P deficits would be large during the dry season or during drought conditions and reduced during the wet season or wet years. The relationships between rainfall and the magnitude of the water deficit as well as rainfall and the magnitude of the P deficit are not significant (Figures 3-20 and 3-21); it appears that rainfall does not have a direct effect on either the water or P budget. There appears to be no relationship between lake stage and the water deficit (Figure 3-22). The P budget in particular is driven by lake stage (R^2 = 0.67, p = 0.004, Figure 3-23), as lake stage increases, the P deficit increases, presumably due to higher Prairie Creek outflows. If the P deficit is accounted for by internal loading, this would suggest that internal loading occurs at high lake stage rather than when the lake is low, which is counter to internal loading at low lake stage due to wind-resuspension. If diffusion of P from the sediments was occurring, this relationship between the P deficit and lake stage would hold since at high lake stage the P concentrations in the water column decrease.

Groundwater Wells

The Ksat and cross sectional area for each well can be found in Table 3-2. Notably, the magnitude of variability in hydraulic slope within wells was much larger than variability in Ksat, suggesting that Ksat is not the principal control over perimeter seepage. The three wells where large and highly variable groundwater flows were observed had Ksat values of 7.0 m/d (well 291), 6.0 m/d (well 251), and 5.0 m/d (well 262); wells 260, 254, and 264, which had very high Ksat values (> 6.3 m/d), exhibited low seepage rates (Figure 3-24).

Groundwater flow for each of the 14 wells was calculated once a month for 15 months; overall, there was no clear trend of water flow into or out of the lake (Figure 3-25). With the exception of well 251, the wells along the western edge (252, 254, and 251) tended to exhibit relatively high groundwater fluxes, consistent with steeper mean land slopes on this side of the lake. However, the four eastern wells (wells 262, 250, 256, and 270) were located in relatively flat regions of the lake edge, but wells 250 and 270 had moderate flows while wells 262 and 256 had relatively large groundwater flows. The four northern wells (wells 261, 264, 291, and 111) and the two southern wells (wells 260 and 263) generally had the smallest groundwater flows, consistent again with expansive, low-relief littoral floodplains that exist in these areas.

All but three perimeter reaches had flows less than 275 m^3/d. Well 262 had very large groundwater inflows in February and March 2008 and well 291 had a large inflow in February 2008 as well. Well 251 had large groundwater outflows during July and October 2008.

Figure 3-26 shows the cumulative groundwater seepage over the 15-month period of record for each of the 14 wells. No obvious pattern in the spatial or temporal flows was apparent, but some wells regularly flowed towards the lake and others regularly flowed out. Notably, wells for which water generally flows toward the lake were interspersed in an apparently random way within wells where water flows away from the lake.

The cumulative perimeter groundwater seepage to Newnans Lake for the 15-month period was 140,375 m^3. This indicates an overall inflow from groundwater into the lake, mainly due to the large contributions of well 262 (147,661 m^3/d) and well 291 (64,808 m^3/d) and these are principally due to large rates observed in February 2008. In order to compare, Table 3-3 shows all of the Newnans Lake inflows during the period of October 2007–August 2008 (creek flow data were not available September–December 2008). The cumulative groundwater flow from October 2007 to August 2008 was 194,320 m^3, approximately 0.3% of total inflows for that period.

When the groundwater flow for each of the 14 wells was compared to the lake stage, the wells showed no consistent variation (Figure 3-27). Out of the 14 wells, the relationship between lake stage and groundwater flow was only significant for wells 253 (negative slope; $p < 0.001$) and 262 (positive slope; $p = 0.009$). When the relationship between lake stage and flow was compared across all wells, the relationship was still not significant. When rainfall was compared to groundwater flow across all wells, there was not a significant relationship, nor was there a significant relationship within wells and rainfall. It appears that lake stage and rainfall are not affecting groundwater flow, but in order to evaluate if the two factors are working together, rainfall divided by lake stage was regressed with groundwater flow; a significant relationship was not found. Even though significance was not found when comparing groundwater flow with rainfall or lake stage, all of the relationships were positive.

Average SRP concentrations in the groundwater ranged from 30 ppb to 3249 ppb (Figure 3-28). In general, the northern and southern wells had lower SRP concentrations while the eastern and especially western wells had much higher concentrations. The mean concentrations in the wells are extremely high, especially for wells 252 (1674 ppb), 253 (2621 ppb), and 254 (3249 ppb); these three wells are all located in the most developed portion of the lake and septic tanks may be a factor in the high P concentrations. Despite these enormously high concentrations, low and sometimes reversed water fluxes result in a cumulative SRP export from

Newnans Lake of -50 kg during the 15-month period of record (Figure 3-29); while this negative result was unexpected, the net mass outflow of SRP is negligible in the context of the P budget for the lake, which has a net P deficit of approximately 333 kg of P over the same period of record.

Comparisons of groundwater flow and SRP concentrations were equally variable in the 14 wells (Figure 3-30). None of the relationships were significant although the relationship between flow and SRP in well 291 is close ($p = 0.052$). Moreover, the sign of the association was unclear, with six wells exhibiting a positive relationship with flow, seven having a negative relationship, and one being neutral. When the relationship between mean flow and SRP was examined across all wells, a significant relationship was still not observed.

A significant ($p = 0.0007$) relationship was found between mean SRP and F concentrations across the groundwater wells surrounding Newnans Lake (Figure 3-31) consistent with a fluorapatite source. A significant relationship was not found between SRP and nitrate across all of the groundwater wells consistent with the hypothesis that the P is not anthropogenic (Figure 3-32). All three wells that had SRP greater than 1000 ppb (wells 252, 253, and 254), located in the most developed western portion of the lake perimeter had relatively low nitrate levels, especially in well 254. Well 254 had much higher concentrations of F than all of the other groundwater wells which may have suggested that the groundwater is Gainesville city water (which is fluoridated) that has gone through a septic tank; however, the low concentrations of nitrate do not support this explanation. Well 270 is the only well with elevated concentrations of nitrate in the groundwater; nitrate averaged 624 ppb and SRP averaged 524 ppb. Land use around well 270 is upland non-forest and a small amount of agriculture.

Samples were taken from the middle of the lake once a month where a significant relationship between SRP and F was not found. For water samples collected from the lake edge, a significant relationship was also not found between SRP and F. The SRP measured around the lake edge was also lower than the SRP measured in the groundwater with the exception of wells 262 and 263. Indeed, a detailed comparison between the Cl and sulfate concentrations at the lake edge and in the adjacent groundwater found no significant associations. This underscores the limited importance of groundwater for the lake's water and elemental budgets, even for near shore lake water chemistry.

Longitudinal Creek Sampling

Transects on both Hatchet Creek and Little Hatchet Creek were sampled during three different flow conditions (Figure 3-33): low baseflow (July 2007), moderate baseflow (February 2008), and storm flow (February 2008 for Hatchet Creek, April 2008 for Little Hatchet Creek). For each flow regime, the upper portion of the transect for both creeks was sampled on one day and the lower portion was sampled on another day; this likely caused artificial increases or decreases in flow and SRP.

Low Baseflow

During low baseflow conditions, SRP and TP in Little Hatchet Creek increases upstream of Waldo Road between sites 20 and 18 (7 ppb to 750 ppb) (Figure 3-34). After site 18, SRP remains steady until a tributary with low P concentrations enters near the Gainesville Regional Airport; at this point SRP drops and then remains steady until the end of the transect at site LHT1 with a concentration of 268 ppb. Flow also increases dramatically from upstream of Waldo Road to the airport between sites 20 and 100 (147 m^3/d to 1395 m^3/d), then fluctuates until the end of the transect when flow ceases altogether. A tributary with low P concentrations (9 ppb) enters Little Hatchet Creek at site LHt6M near the airport with a flow of 489 m^3/d; P decreases from 437 ppb to 274 ppb and flow increases from 1003 m^3/d to 1566 m^3/d at this point. The longitudinal effect is an increase in P export from site 20 to site 109 (1 g/d to 774 g/d) and then a gradual decrease to 177 g/d at the end of the transect. Variable flow and a longitudinal decrease in P were observed, which is the opposite of my prediction. On the other hand, P concentrations in Little Hatchet Creek are very high, indicating that groundwater seepage is occurring upstream of the transect.

During low baseflow in Hatchet Creek, SRP and TP steadily increases from 8 ppb at the most upstream site (site 105) to 314 ppb at the most downstream site (site 3); note that the reason site 3 represents the most downstream site during baseflow is the presence of a sinkhole in the stream that captured the entire flow of Hatchet Creek at the time this transect was sampled. Sites further downstream were not flowing and were not sampled. Flow is variable throughout the transect (Figure 3-35), but clearly increases upstream to downstream; no tributaries were flowing during sampling. The upper and lower part of the transect were sampled on different days, as such, the drop in flow between RTH8 and 103 is probably artificial. The overall effect is an increase in P export from 5 g/d at site 105 to 369 g/d at site 3. A longitudinal increase in flow and SRP (in other words, increasing flow rates and SRP concentrations with distance downstream) without the presence of tributaries was observed, which lends credence to the hypothesis that groundwater is seeping into Hatchet Creek and is acting as a conduit for P.

Moderate Baseflow

During moderate baseflow, SRP and TP initially increases in Little Hatchet Creek from 21 ppb at site 20 (upstream of Waldo Road) to 245 ppb at site LHCTA2 (just downstream of the airport) with no tributary inputs, and then decreases throughout the rest of the transect to 64 ppb at site 389, right at the edge of Newnans Lake (Figure 3-36). There is a dramatic decline in SRP concentration during passage through Gum Root Swamp (LHCTA7 to site 89), which may be a combination of dilution and P removal/conversion; this decline in SRP also occurred with TP. Both flow and P mass flux were variable along the transect. While no tributaries were flowing in the upper half of the transect, there were two tributaries flowing into the creek in the lower half. One tributary, entering Little Hatchet Creek between sites LHCTB3 and LHCTB2, had extremely low flow (actual flow was not measured) and high SRP concentrations (1608 ppb); this tributary does not change flow or SRP concentration in Little Hatchet Creek. A second tributary (14 ppb, 3425 m3/d) entering the creek between sites LHCTA2 and LHCTA4 increased flow (5970 m^3/d to 11249 m^3/d) and decreased SRP (245 ppb to 177 ppb).

In Hatchet Creek during moderate baseflow, SRP and TP gradually increases from 5 ppb to 93 ppb from site 105 to site HCTA1 and then falls between sites HCTA1 and 53 (93 ppb to 33 ppb) (Figure 3-37). Water flow gradually increases from 8710 m^3/d at site 105 to 28,698 m^3/d at site HCTA1. Between sites HCTA1 and 51, the flow decreases from 28,698 m^3/d to 18,105 m^3/d, and then increases between sites 51 and 53 (between which is the confluence with Bee Tree Creek) to 55,978 m^3/d, illustrating the magnitude of the Bee Tree Creek flow at the time of sampling. Overall, this creates a longitudinal increase in P export to Newnans Lake from 44 g/d at site 105 to 1517 g/d at site 91. There are four tributaries that flow into Hatchet Creek during moderate baseflow. From upstream to downstream, the first, second, and fourth tributary all have much higher SRP concentrations (221, 158, and 251 ppb, respectively) than does Hatchet Creek at their point of mixing. As such, SRP increases after each. The third tributary has low SRP (59 ppb) and the SRP concentration in the creek decreases after the inflow. The flow in all four tributaries is low compared to Hatchet Creek, with none contributing more than 1% of total stream flow. As such, P fluxes are not affected by tributary input.

Storm Flow

During storm flows, SRP and TP generally increased dramatically (68 ppb at site 18 to 251 ppb at site 389) with distance downstream in Little Hatchet Creek (Figure 3-38). Between sites 109 and LHCTA3, SRP was highly variable, but TP was not, perhaps due to the uptake of the readily available SRP at sites 109 and LHCTA1. Flow increases from 11,988 m^3/d at site 18 to 24,466 m^3/d at site LHCTB5 and then gradually decreases, with several unusual excursions, to 9052 m^3/d at site LHCTA6. The flow then increases sharply between LHCTA7 and 389 (11,499 m^3/d to 26,178 m^3/d). Discharge measurements at storm flow were far more logistically challenging than baseflow measurements, and some of the variability may arise from this uncertainty. However, increases in discharge at Gum Root Swamp (sties 89 and 389) may reflect multiple sources of inflow to that system and internal runoff generation. Notably, where SRP concentrations were observed to decline during passage through Gum Root Swamp at moderate baseflow, concentrations actually increase at storm flow. Overall, P mass export grew slowly until sampling point LHTCA7 where export increases 400%, from 1665 g/d to 6567 g/d during passage through Gum Root Swamp.

Two tributaries to Little Hatchet Creek were measured during storm flows. From upstream to downstream, the first tributary had low flow (244 m^3/d), which limited the effect of the tremendously high SRP concentrations (1750 ppb) on stream P fluxes. The second tributary had a moderate flow (3915 m^3/d), and low SRP concentrations of 18 ppb; the effect on P fluxes was substantial, increasing from 600 g/d to nearly 2500 g/d. The most dramatic tributary inputs are evidently associated with flows into Gum Root Swamp; P concentrations, flows of water, and fluxes of P all increase, the latter dramatically. TP is higher than SRP, although by a small amount, at the Gum Root Swamp sites, which may indicate that the wetland is releasing stored P; F concentrations are also lower at these sites than most of the other measured sites in Little

Hatchet Creek (Figure 3-39), which also suggests that the wetland is releasing P and that the increase in P from Gum Root Swamp into Newnans Lake is not due to tributary inputs alone. This indicates that during baseflow conditions, Gum Root Swamp is storing P, but when storm pulses flow through the wetland, it releases P, therefore acting as a source of P.

During storm flows in Hatchet Creek, SRP concentrations are widely variable with large decreases and increases between sampling points (Figure 3-40). Although SRP reaches 339 ppb at site 103, a dramatic increase from 19 ppb at the most upstream site (site 105), this appears anomalous because SRP decreases precipitously to 20 ppb at site 40, and ultimately discharges to Newnans Lake (site 91) at 71 ppb. The flow in Hatchet Creek is relatively steady between sites 105 and HCTB8; after this point, the flow increases rapidly to site 3 and then decreases rapidly to site 51, indicating that a lot of the water is being captured by the sinkhole at site 3. Between 51 and 53, Bee Tree Creek flows into Hatchet Creek at 159,762 m^3/d, increasing the flow rapidly. Overall, flow increases from 75,844 m^3/d at site 105 to 384,602 m^3/d at site 91. The combined effect of flow and SRP gives a widely variable P export with an overall positive trend to the lake of 1440 g/d at site 105 to 27,290 g/d at site 91. Four tributaries were noted during storm flows. From upstream to downstream, the first tributary has a very high SRP (814 ppb) and moderate flow (4257 m^3/d) and exerts a relatively significant effect on P fluxes. The second tributary also has high SRP (330 ppb) and moderate flow (2129 m^3/d), but P concentrations actually decline in that reach as do P fluxes. The third tributary has moderately enriched SRP (261 ppb), and fairly high flow (8318 m^3/d); however, while the P flux increases, the P concentration declines. The fourth tributary has a comparatively low SRP concentration (55 ppb) and high flow (15,903 m^3/d); the effects of this tributary on P fluxes are marked. Bee Tree

Creek (159,762 m^3/d, 11 ppb) is also a major tributary that flows into Hatchet Creek between sites 51 and 53; it has the effect of increasing flow, but decreasing SRP.

Source of Phosphorus

A significant positive relationship exists between F and SRP during low baseflow in Little Hatchet Creek ($R^2 = 0.76$, $p < 0.001$, Figure 3-41) suggesting that geologic P is present at low baseflow. There is no significant relationship between nitrate and SRP for Little Hatchet Creek, suggesting that there is no anthropogenic input. There is a significant ($R^2 = 0.64$, $p = 0.0002$) positive relationship between F and SRP for Hatchet Creek at low baseflow. There is a significant, but negative, relationship between SRP and nitrate for Hatchet Creek ($R^2 = 0.38$, $p = 0.008$); since nitrate is low when SRP is high, this suggests that the P present is not anthropogenic. The positive correlations between F and SRP, combined with the lack of a positive correlation between nitrate and SRP, indicates that there is a geologic source of P present in the creeks.

During moderate baseflow, the relationship between SRP and F for Little Hatchet Creek is not significant (Figure 3-42). A significant relationship was also not observed between SRP and nitrate for Little Hatchet Creek; with the removal of an outlier, the relationship becomes significant ($R^2 = 0.80$, $p < 0.001$) indicating that there may be anthropogenic influence during moderate baseflow, although unlikely since nitrate concentrations are low. A significant positive relationship ($R^2 = 0.38$, $p = 0.002$) exists between SRP and F during moderate baseflow for Hatchet Creek. As in low baseflow, a significant, yet negative, relationship is observed between nitrate and SRP ($R^2 = 0.65$, $p < 0.001$). Moderate baseflow conditions have the lowest SRP concentrations out of the three flow regimes, but it is important to note that the concentrations are still very high and that the P present in Hatchet Creek is geologic in origin; in Little Hatchet Creek, it appears that geologic and possibly anthropogenic sources of P are present.

During storm flow, a significant relationship was not observed between SRP and F for Little Hatchet Creek (Figure 3-43), nor between SRP and nitrate although it was close (R^2 = 0.23, p = 0.054). For Hatchet Creek, neither relationship was statistically significant.

I predicted that water and flow rate would longitudinally increase particularly where the flow was incising into the Hawthorn. Groundwater seepage (indicated by F vs. SRP) is more pronounced in Hatchet Creek due to the extent that the creek is in contact with the Hawthorn Group (Figure 1-4). The entire length of Hatchet Creek appears to be in contact with the Hawthorn, which is most likely why we see a longitudinal relationship between F and SRP. In Little Hatchet Creek only the lower portion of the creek is exposed to the Hawthorn. This is further supported by Cohen et al. (2007) who found a very strong relationship at sites 100, 109, and 25 (R^2 = 0.96) on the downstream portion of Little Hatchet Creek. If the relationship was compared over time on a site by site basis in Little Hatchet Creek, it is likely that certain sites would have strong correlations while others would not due to varying proximity to the Hawthorn Group.

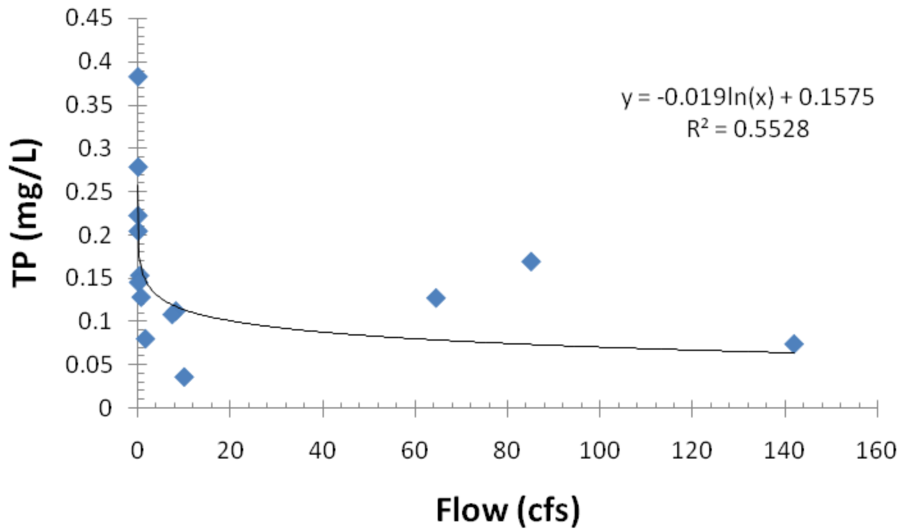


Figure 3-1. Flow vs. TP relationship for Hatchet Creek from 2007 – 2008 used to create the daily TP concentrations for the P budget.

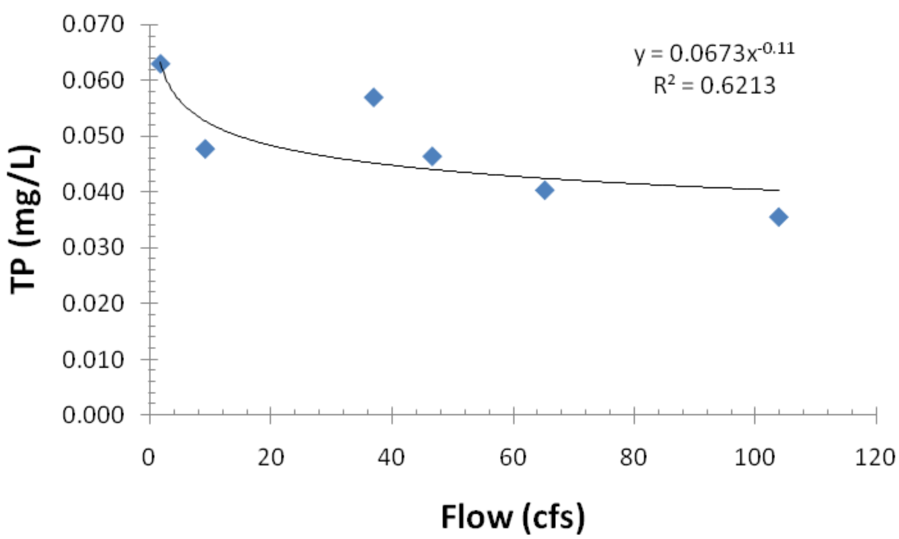


Figure 3-2. Flow vs. TP relationship for Bee Tree Creek from 2007 – 2008 used to create the daily TP concentrations for the P budget.

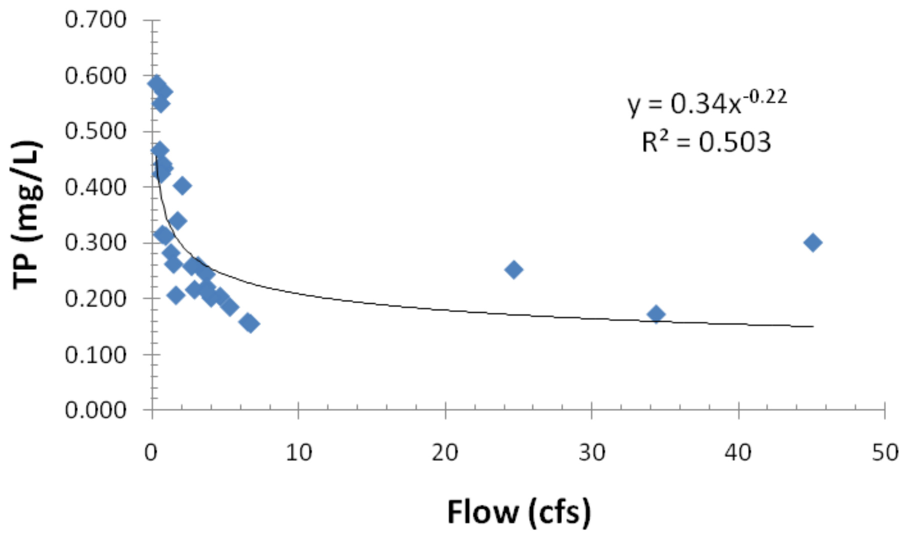


Figure 3-3. Flow vs. TP relationship for Little Hatchet Creek from 2007 - 2008 used to create the daily TP concentrations for the P budget.

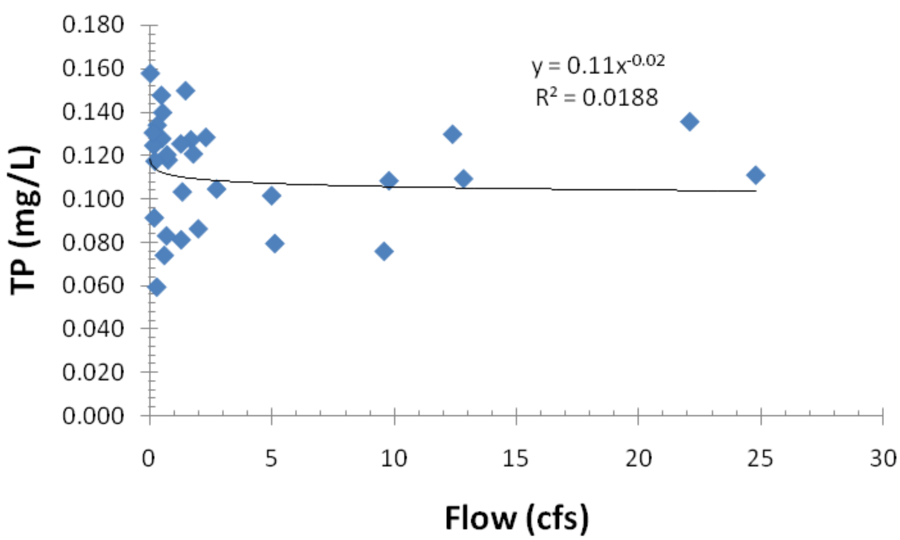


Figure 3-4. Flow vs. TP relationship for Lake Forest Creek from 2007 – 2008 used to create the daily TP concentrations for the P budget.

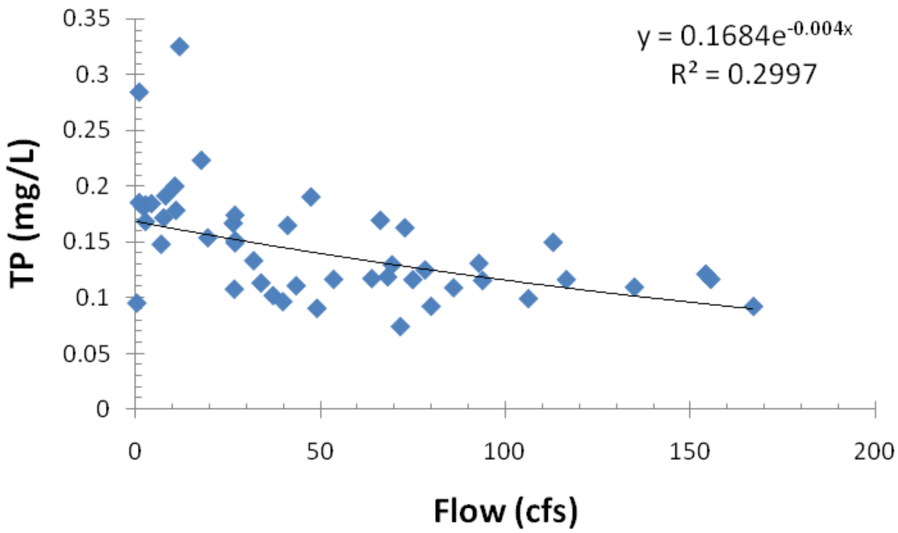


Figure 3-5. Flow vs. TP relationship for Prairie Creek from 1998 - 2006 used to create daily TP concentrations for the P budget.

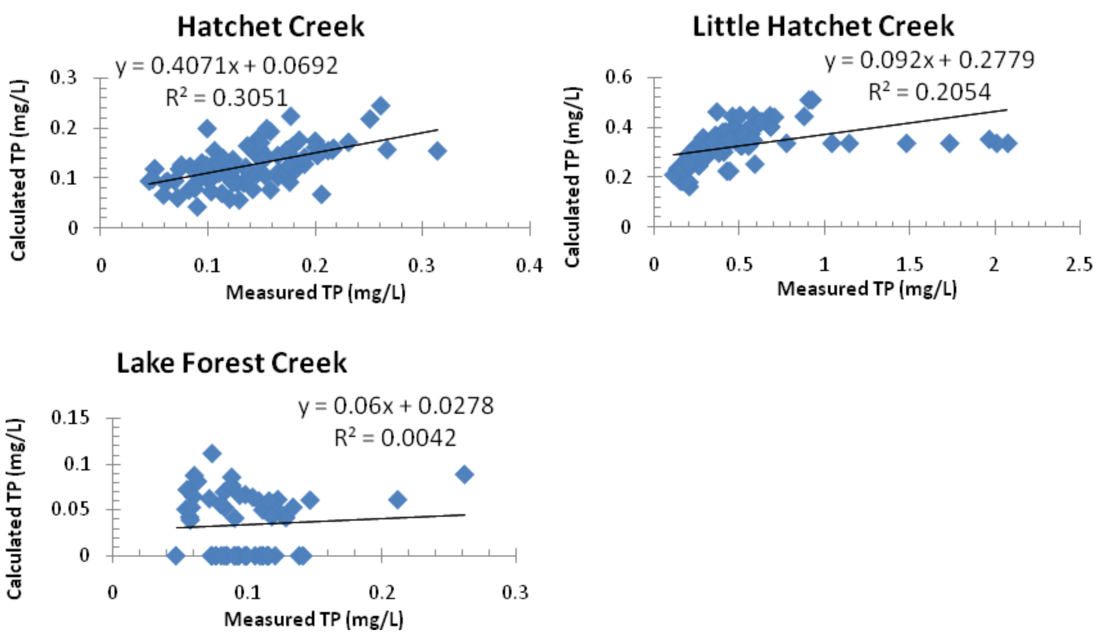


Figure 3-6. Calculated TP vs. measured TP for three of the input creeks.

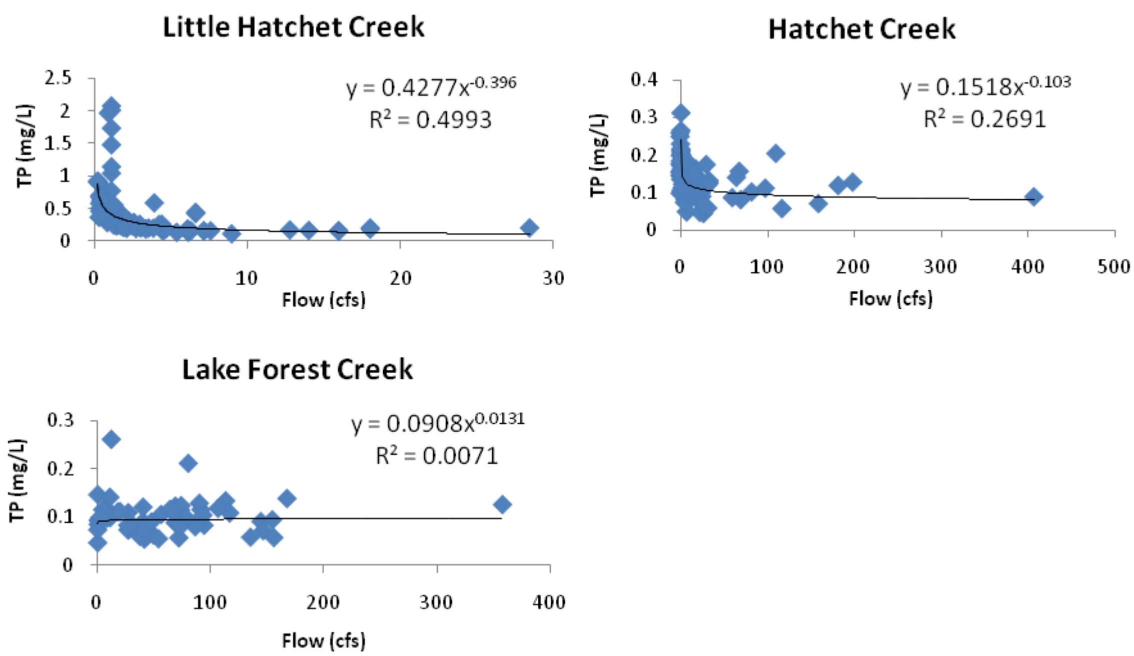


Figure 3-7. Archival flow vs. TP for three of the input creeks.

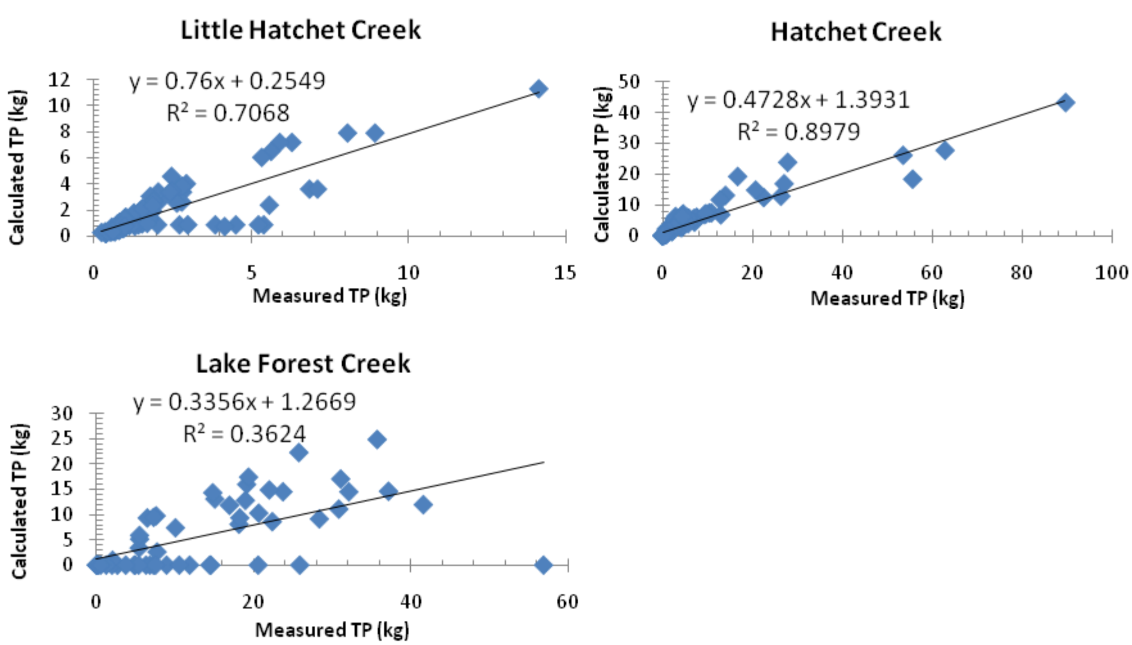


Figure 3-8. Measured TP flux vs. calculated TP flux for three of the input creeks.

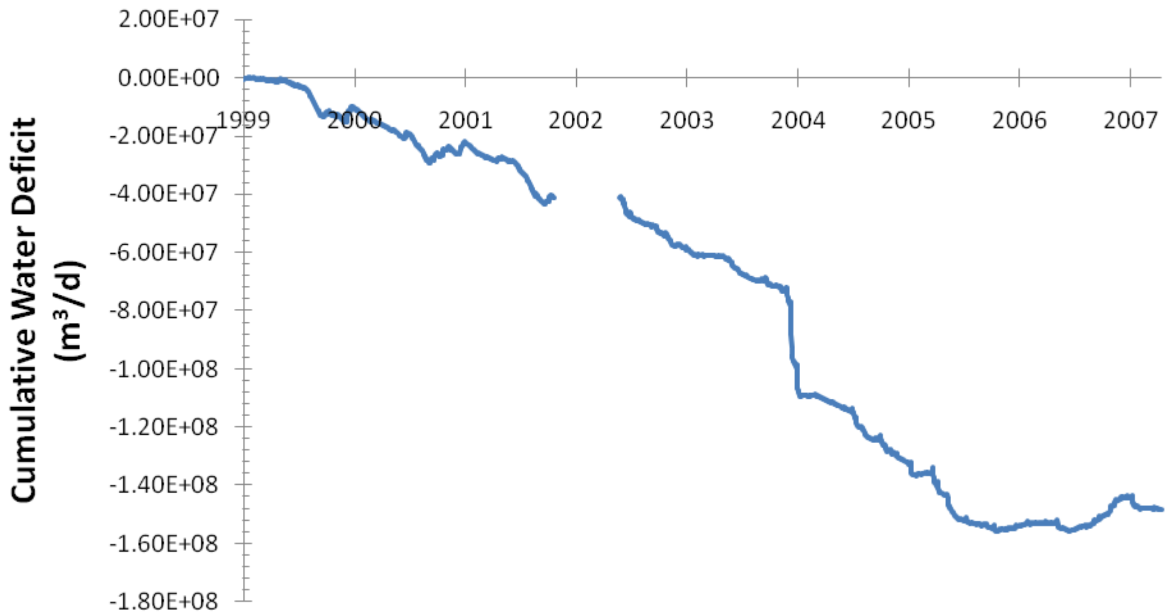


Figure 3-9. Cumulative change of the water budget from October 1998 through October 2007. *Prairie Creek and lake stage data are missing from July 20, 2002 through February 20, 2003.

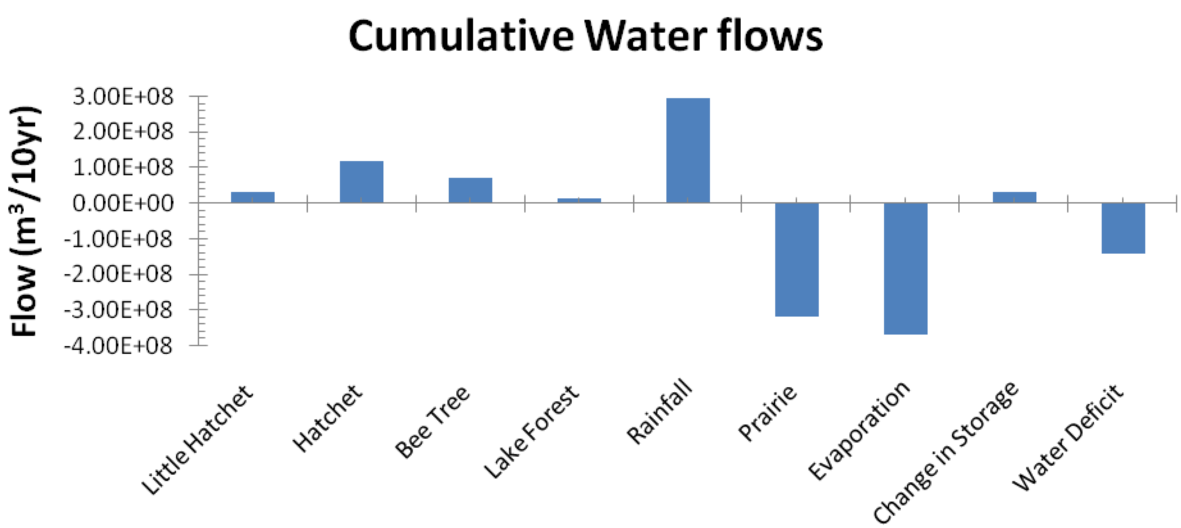


Figure 3-10. Cumulative flows from 1998-2008 of all measured inflows and outflows in Newnans Lake.

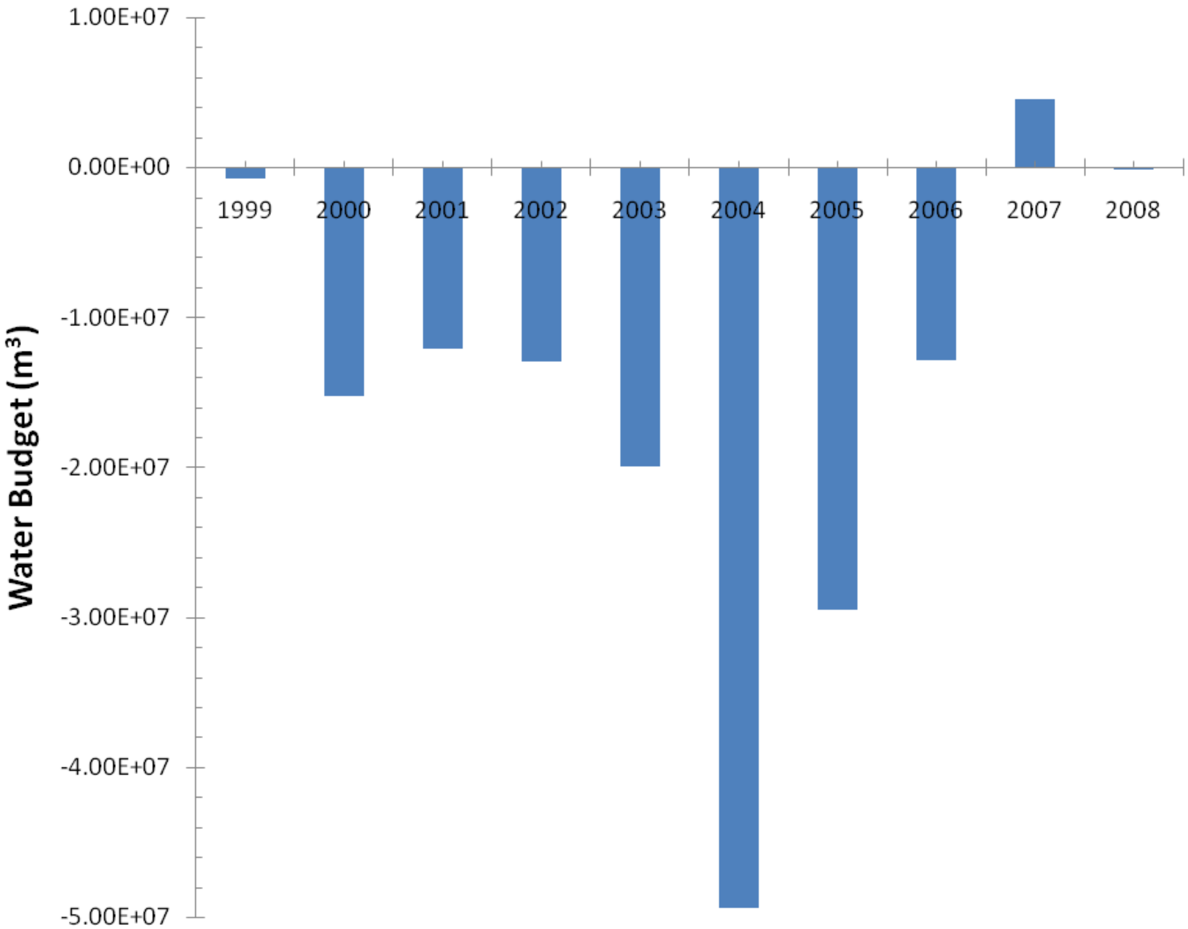


Figure 3-11. Cumulative change in the water budget each year. *1999, 2002, 2003, and 2008 are incomplete datasets.

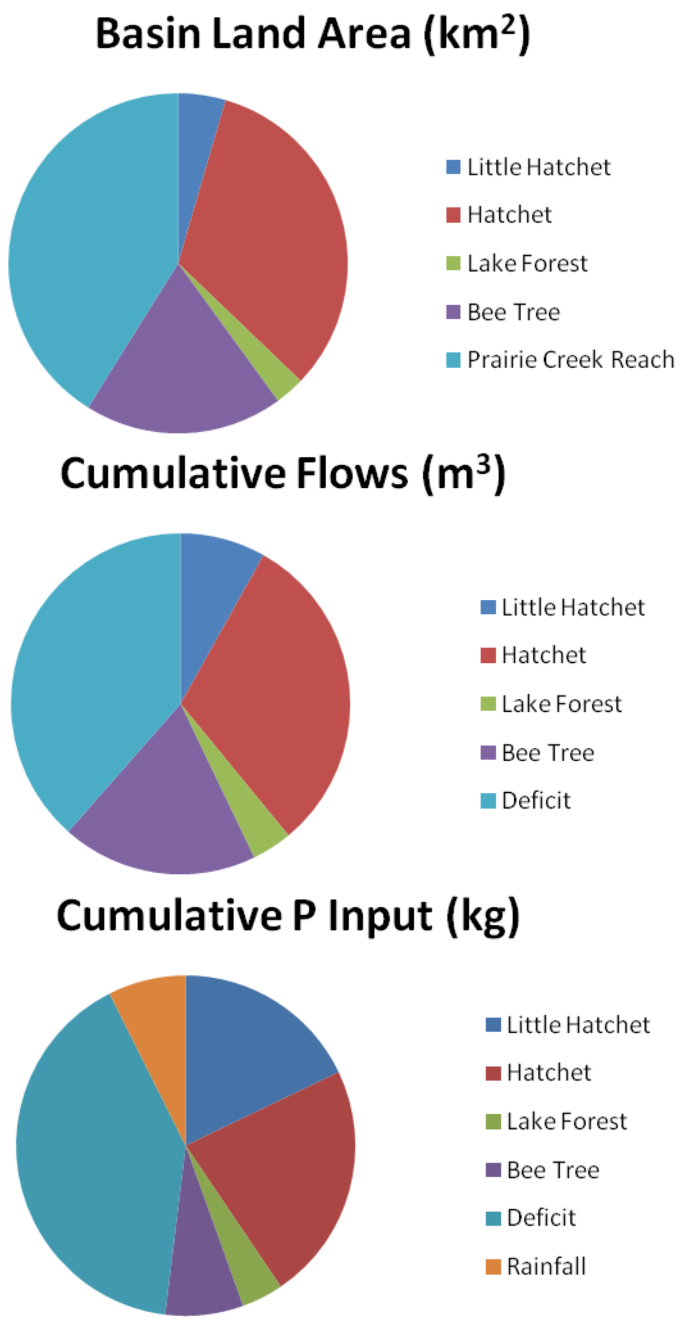


Figure 3-12. Pie graphs comparing basin land area, cumulative flows, and cumulative P input for each of the inflows into Newnans Lake. *For the purpose of comparing land area to the magnitude of flow, rainfall was not included in basin land area or cumulative flow.

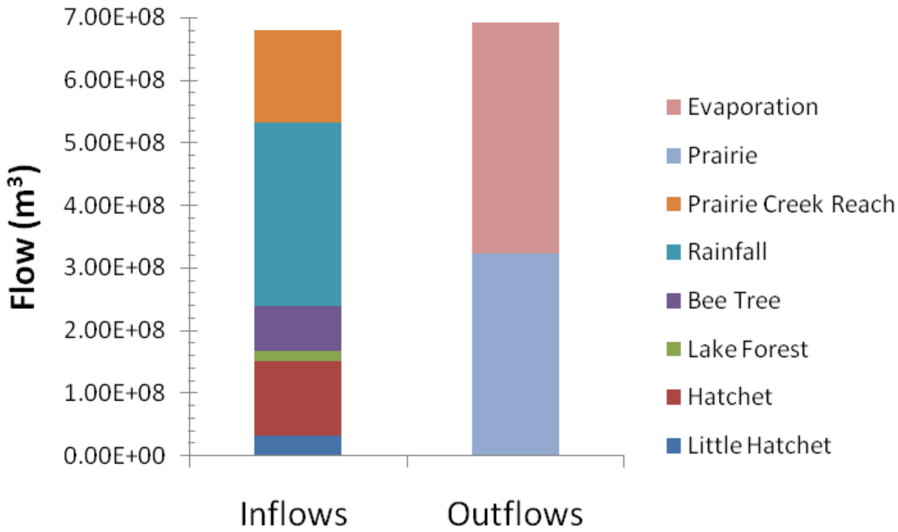


Figure 3-13. Inflows and outflows in Newnans Lake.

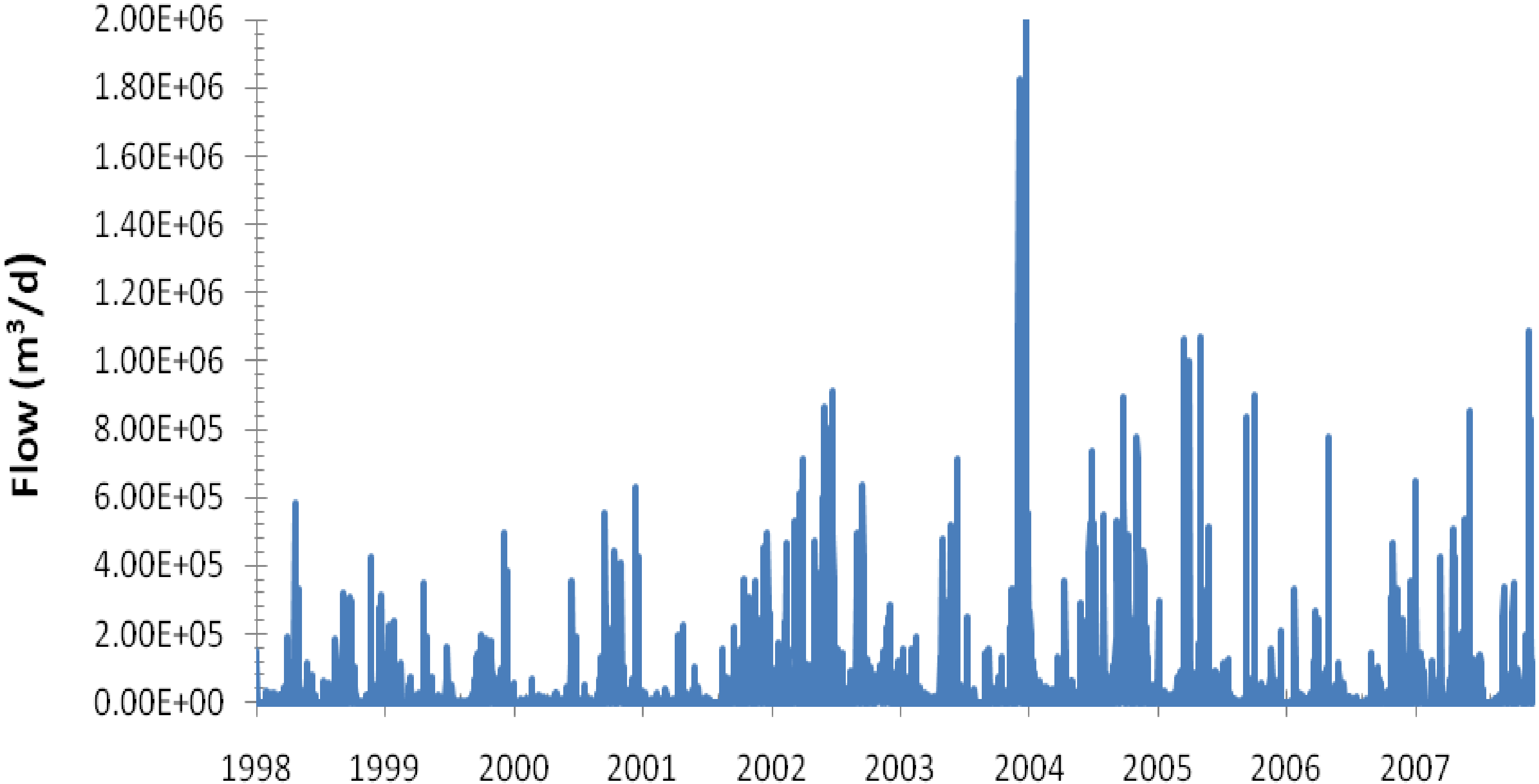


Figure 3-14. The predicted flow of Prairie Creek Reach over time.

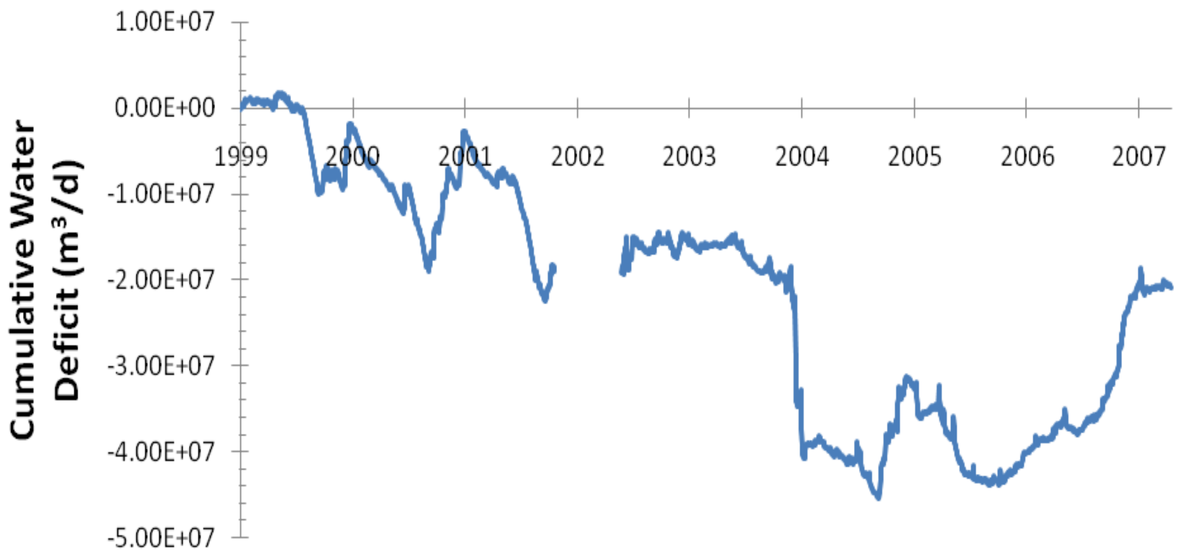


Figure 3-15. Cumulative Water deficit from 1998-2008 with Prairie Creek Reach added in as an inflow.

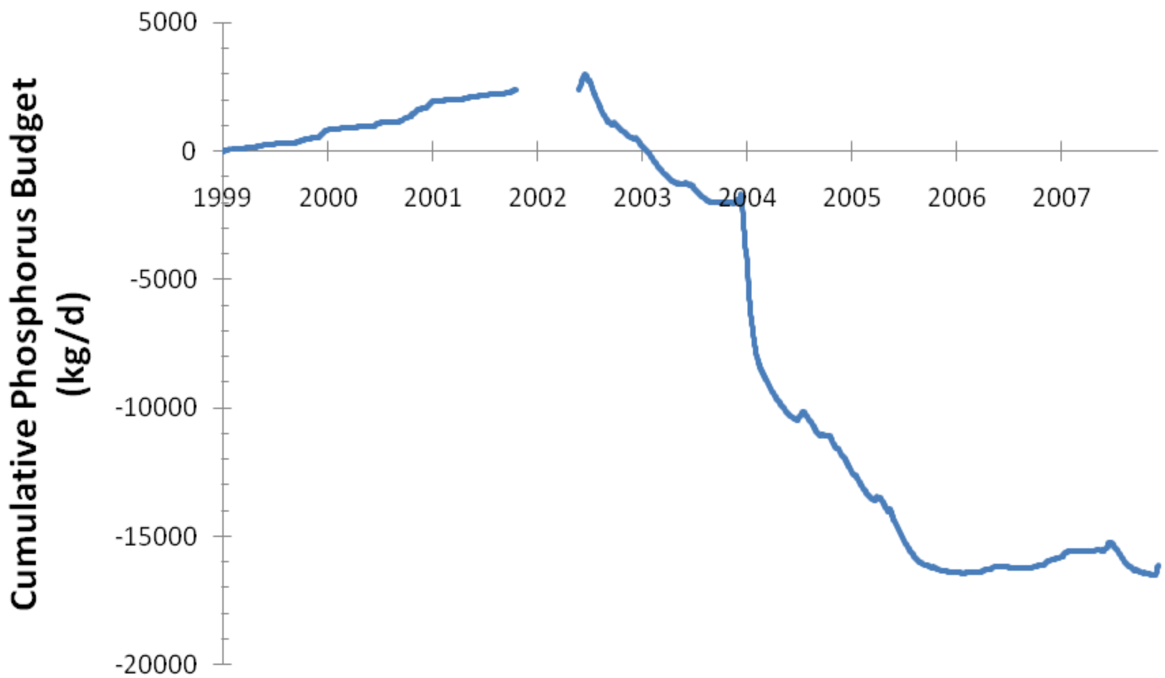


Figure 3-16. Cumulative change of the P budget from October 1998 through October 2007. *Prairie Creek and lake stage data are missing from July 20, 2002 through February 20, 2003.

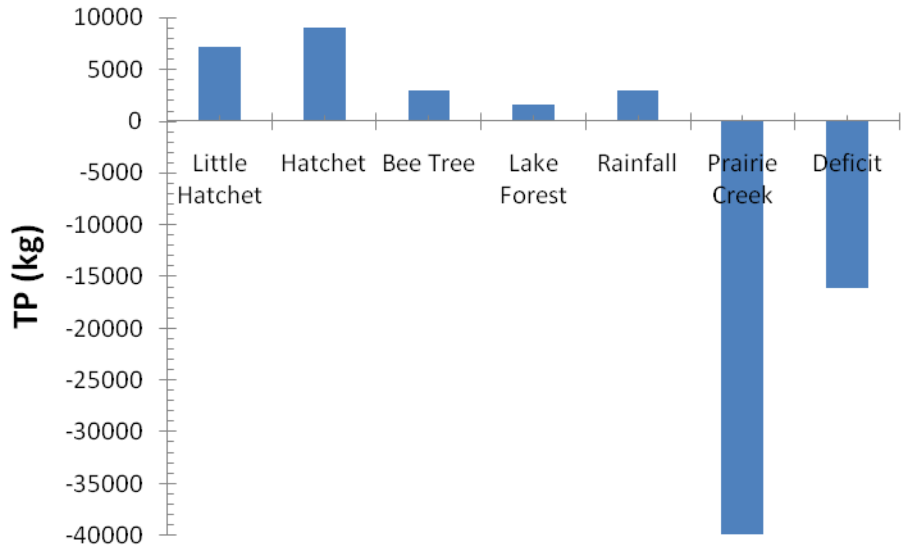


Figure 3-17. Cumulative inputs and outputs of TP for all measured sources from 1998-2008 for Newnans Lake.

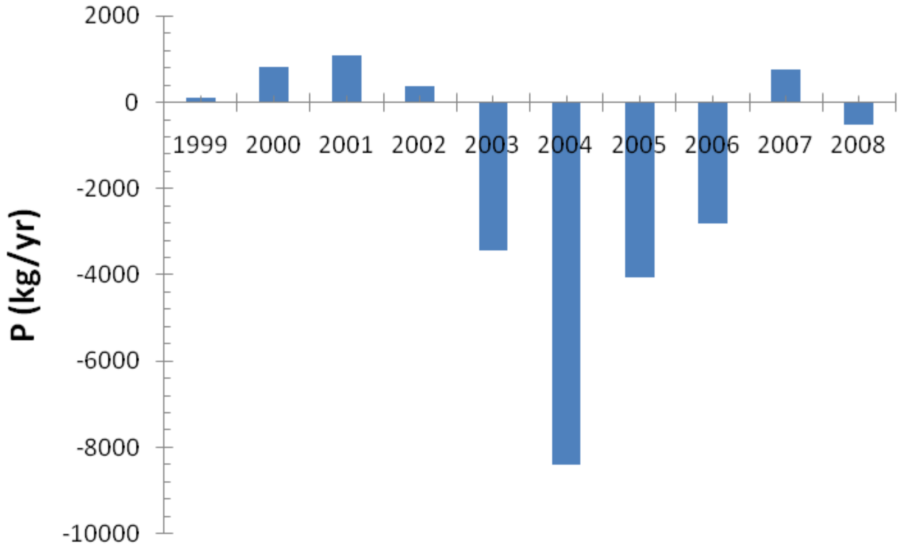


Figure 3-18. Annual P deficit. *1999, 2002, 2003, and 2008 are incomplete datasets.

A

B

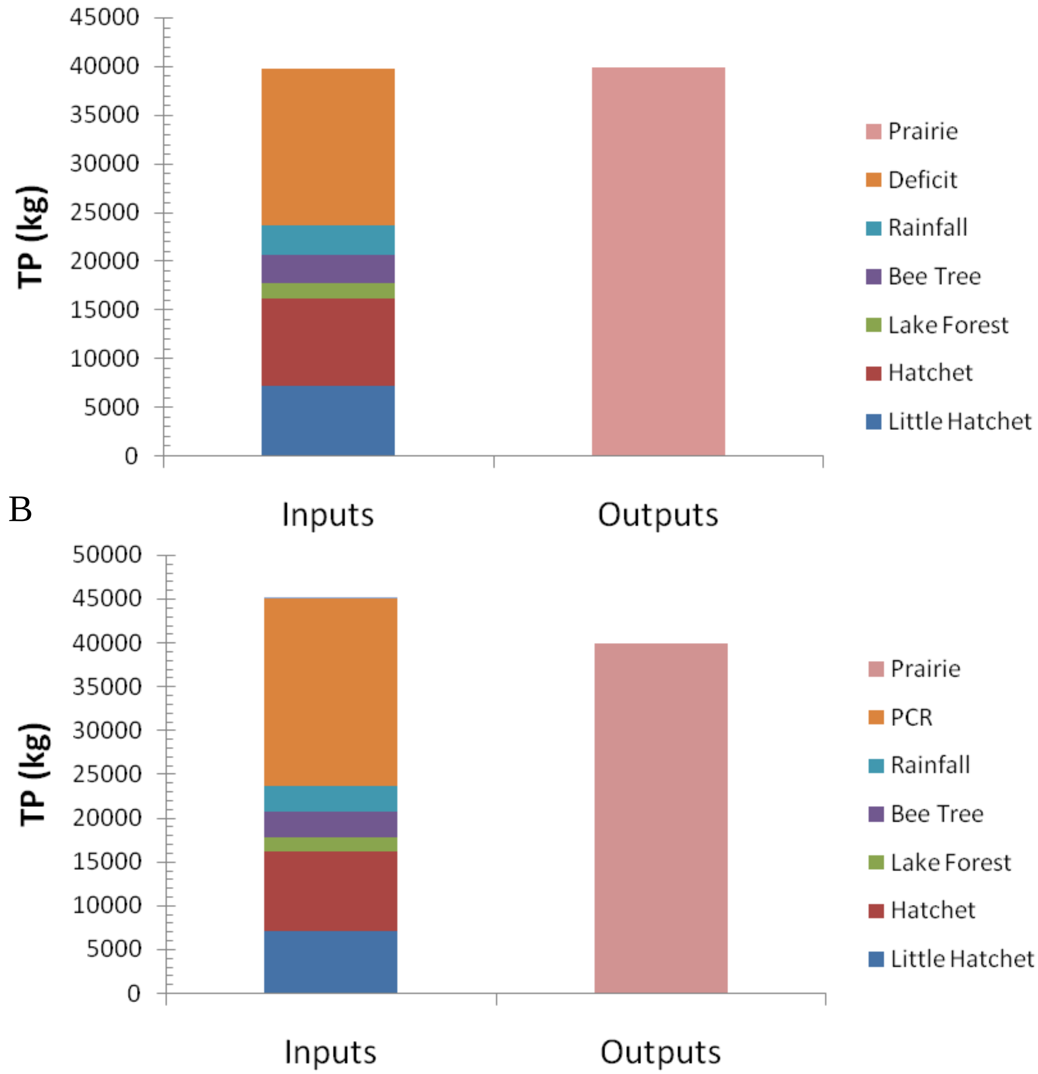


Figure 3-19. P inputs and outputs for Newnans Lake also showing the estimated P input for Prairie Creek Reach. A) The P budget includes the P deficit. B) The P budget includes the estimated input from Prairie Creek Reach (PCR). Note that with the estimated input of Prairie Creek Reach, the input of P is larger than the output. The estimation of inputs from Prairie Creek Reach is not a certain estimate and so it is difficult to determine if inputs truly exceed outputs. In any case, it is probable that Prairie Creek Reach accounts for most, or all, of the P deficit.

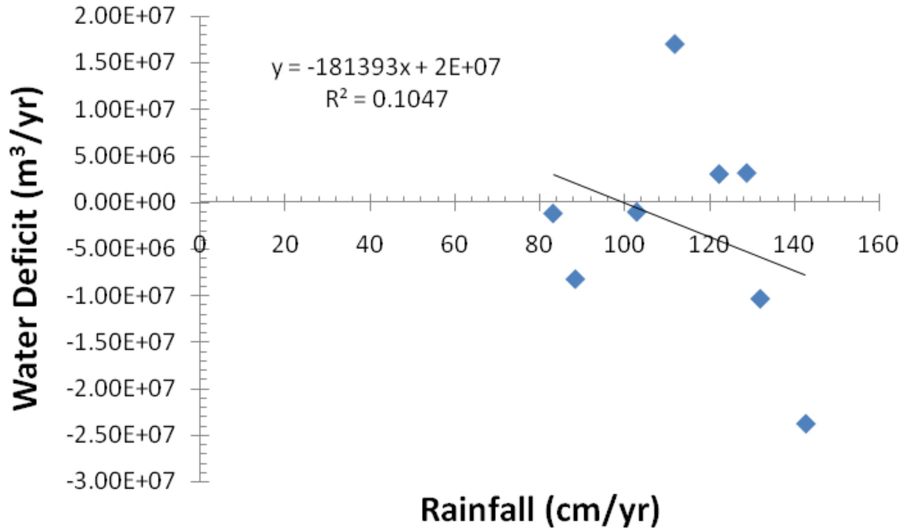


Figure 3-20. Rainfall and the magnitude of the water deficit plotted over the 10 year period of record. *This graph includes the estimated Prairie Creek Reach inflows.

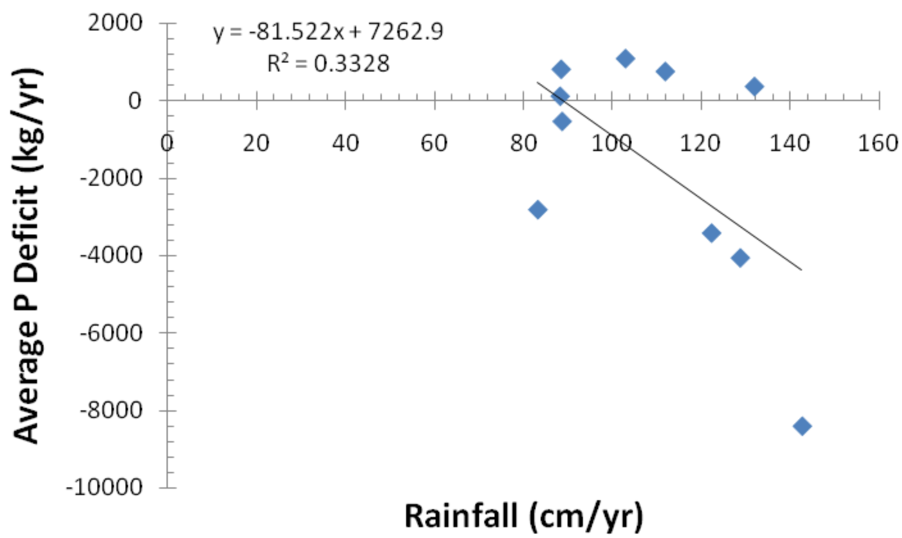


Figure 3-21. Rainfall and the magnitude of the P deficit plotted over the 10 year period of record.

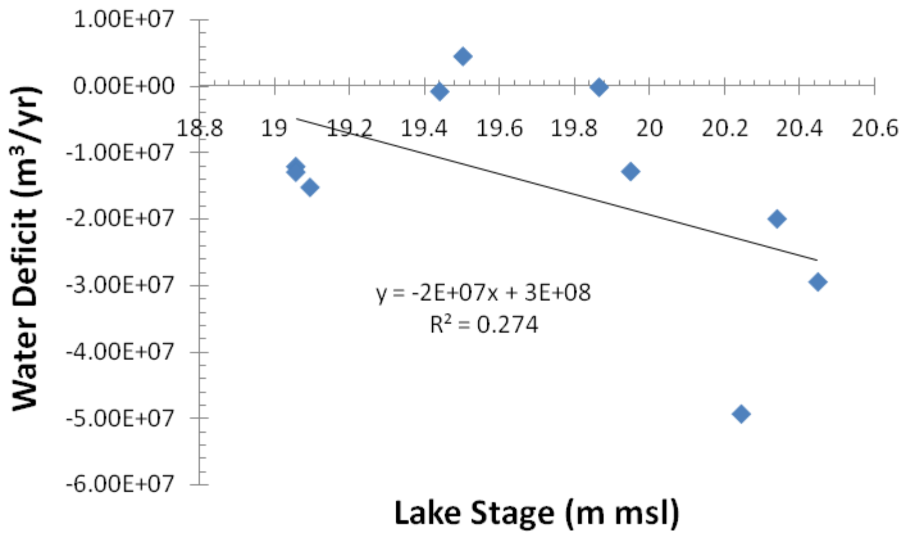


Figure 3-22. Annual mean lake stage and the magnitude of the water deficit plotted over the 10 year period of record. *This graph includes the estimated flows from Prairie Creek Reach.

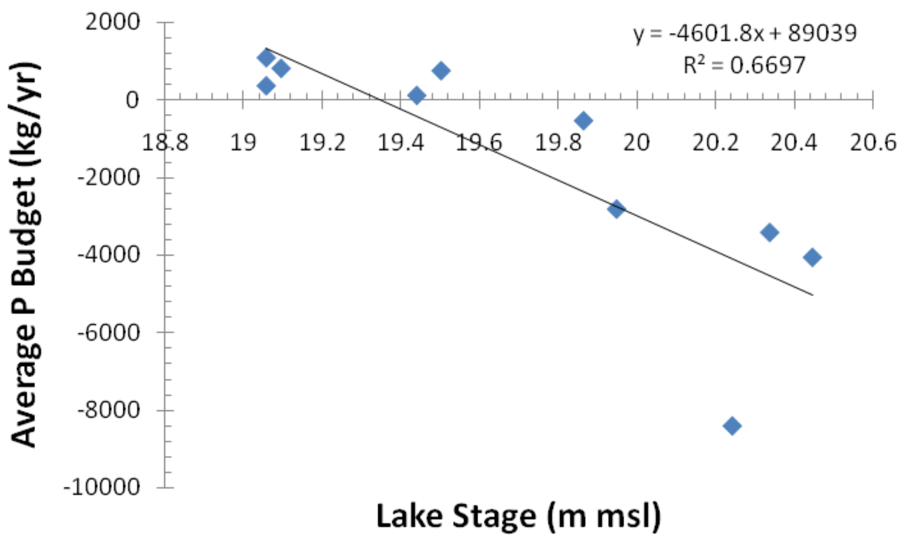


Figure 3-23. Annual mean lake stage and the magnitude of the P deficit plotted over the 10 year period of record.

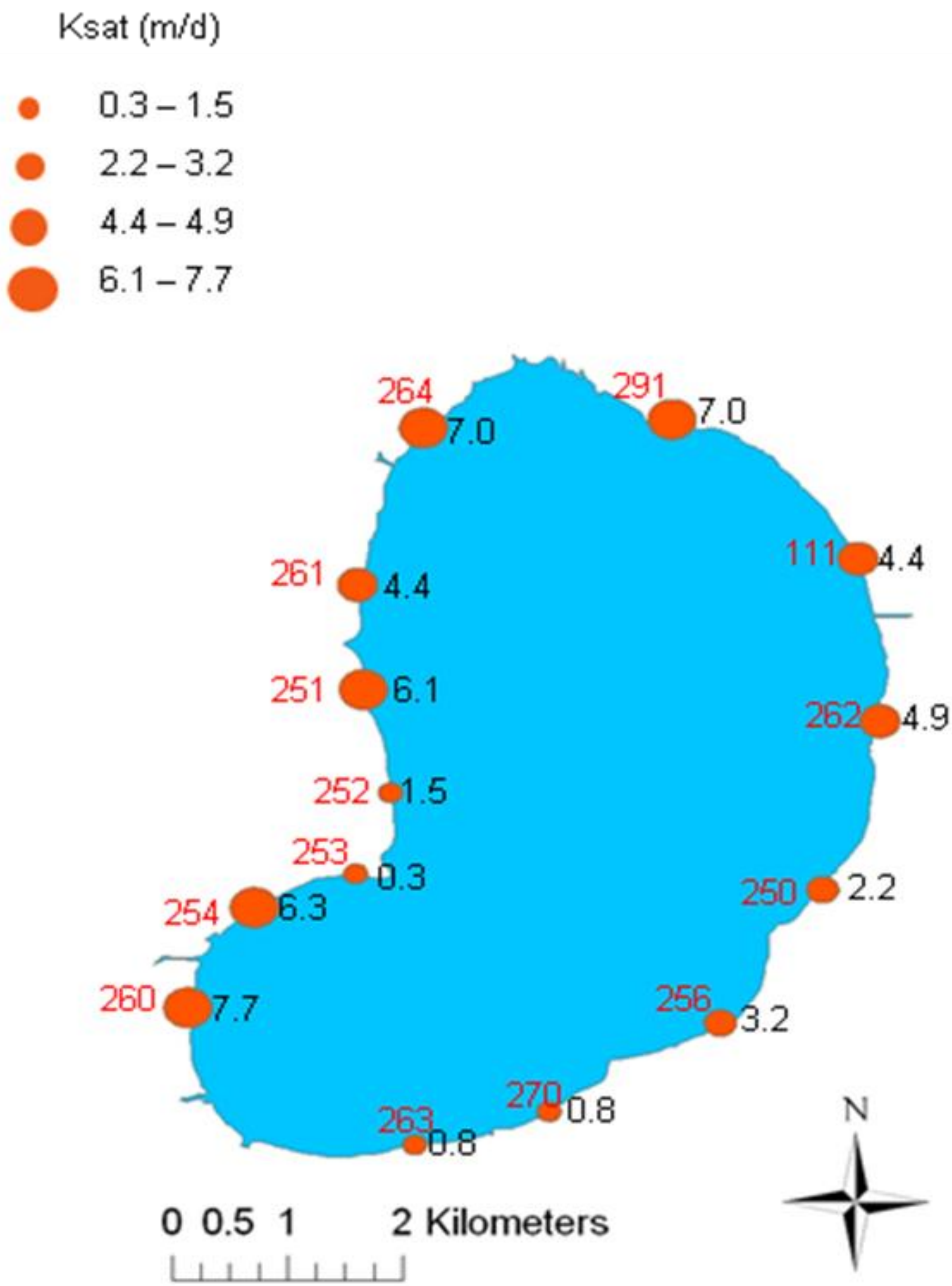


Figure 3-24. Hydraulic Conductivity for each well.

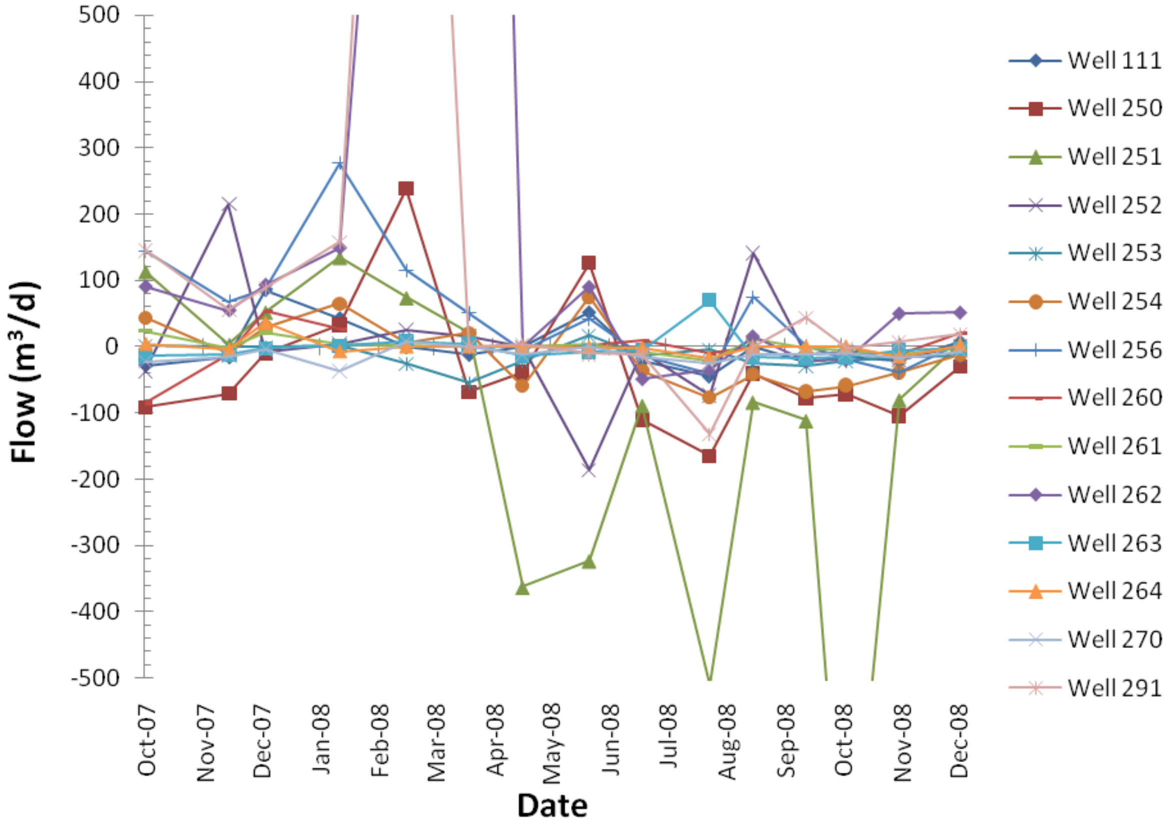


Figure 3-25. Groundwater flow for each of the 14 wells over the 15 month period of measurement. *February 2008 flows for well 291 and February and March 2008 flows for well 262 go over 500 m^3/d. July and October 2008 flows for well 251 go below -500 m^3/d.

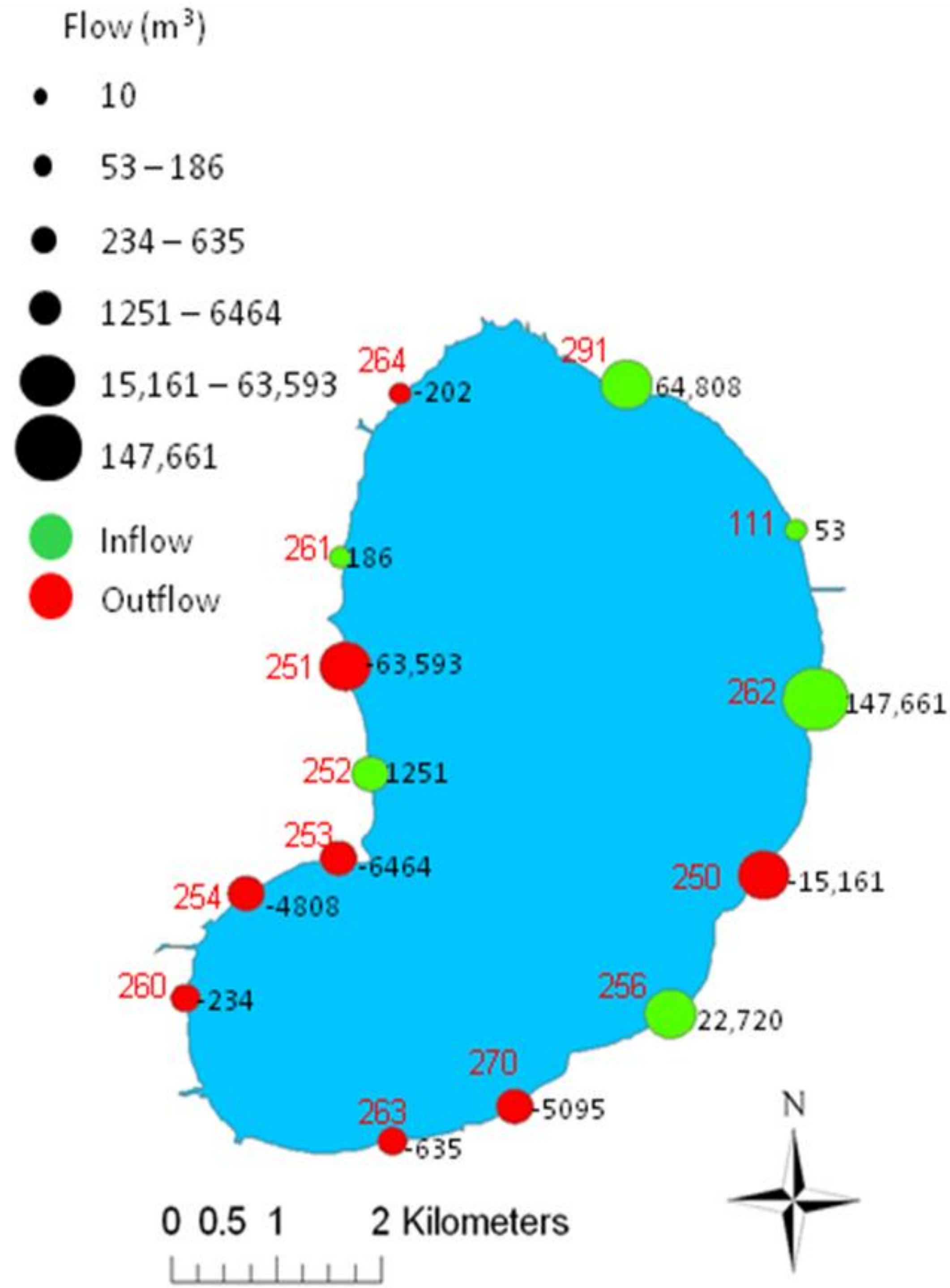


Figure 3-26. Cumulative flow for each well over the 15 month period of sampling.

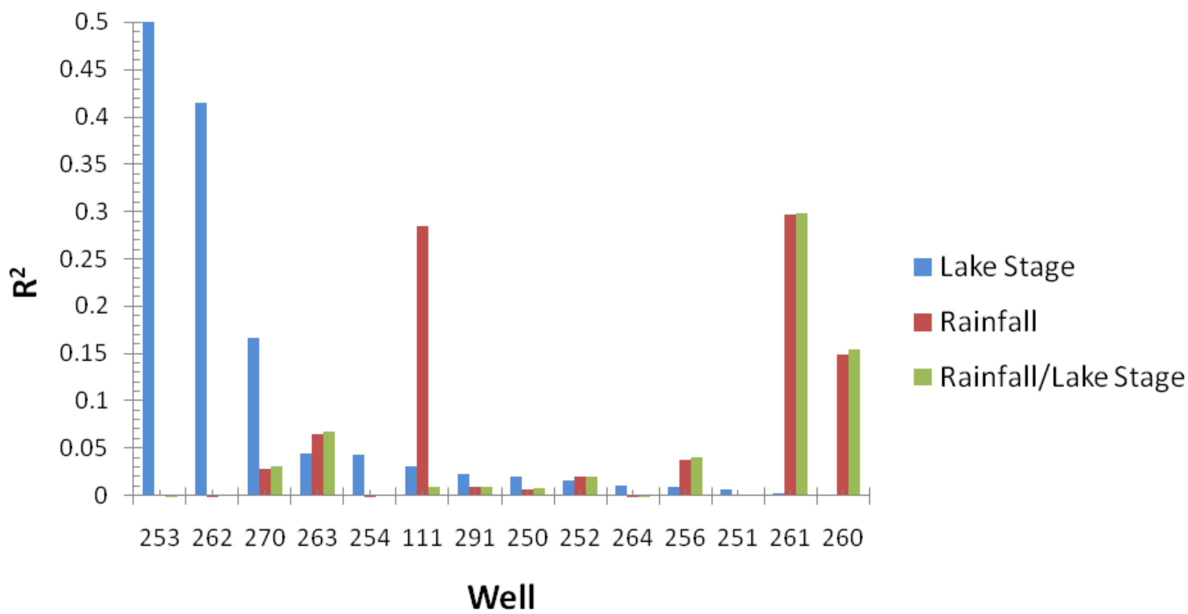


Figure 3-27. The R^2 values for the relationship between groundwater flow and lake stage, rainfall, and rainfall/lake stage. *The R^2 for lake stage in well 253 is 0.74.

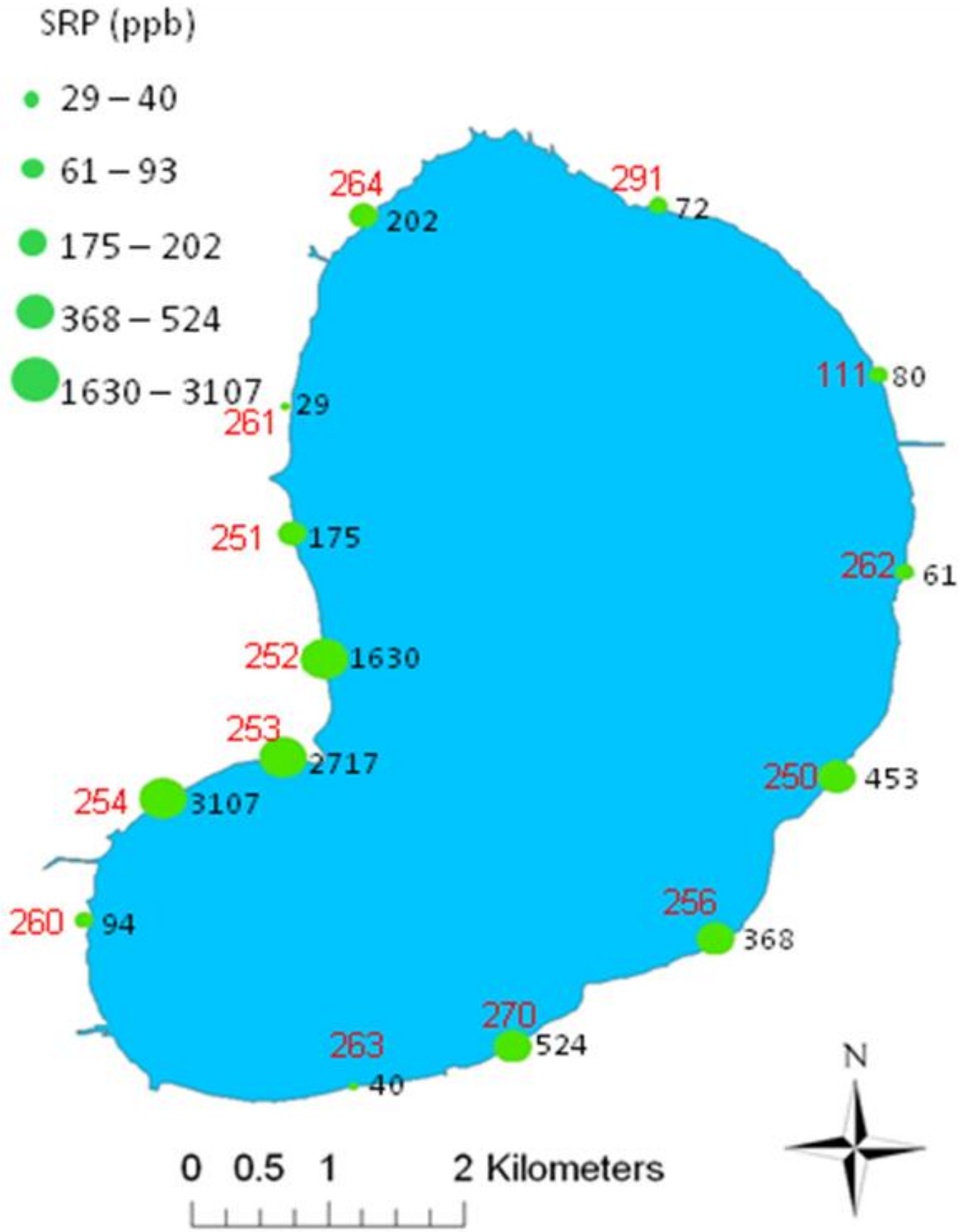


Figure 3-28. Average SRP concentrations measured in the groundwater wells.

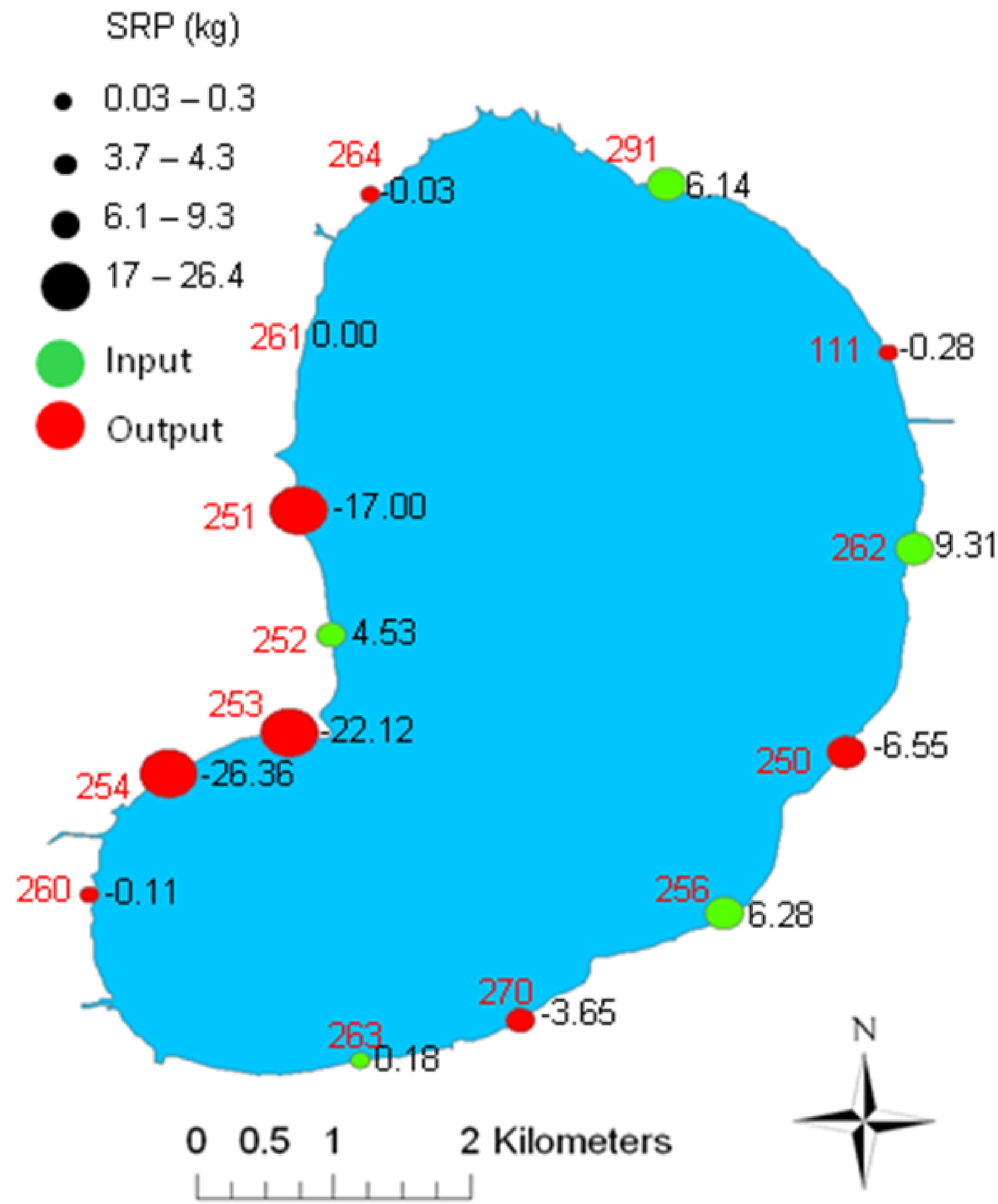


Figure 3-29. Cumulative SRP import or export over the 15 month period of sampling.

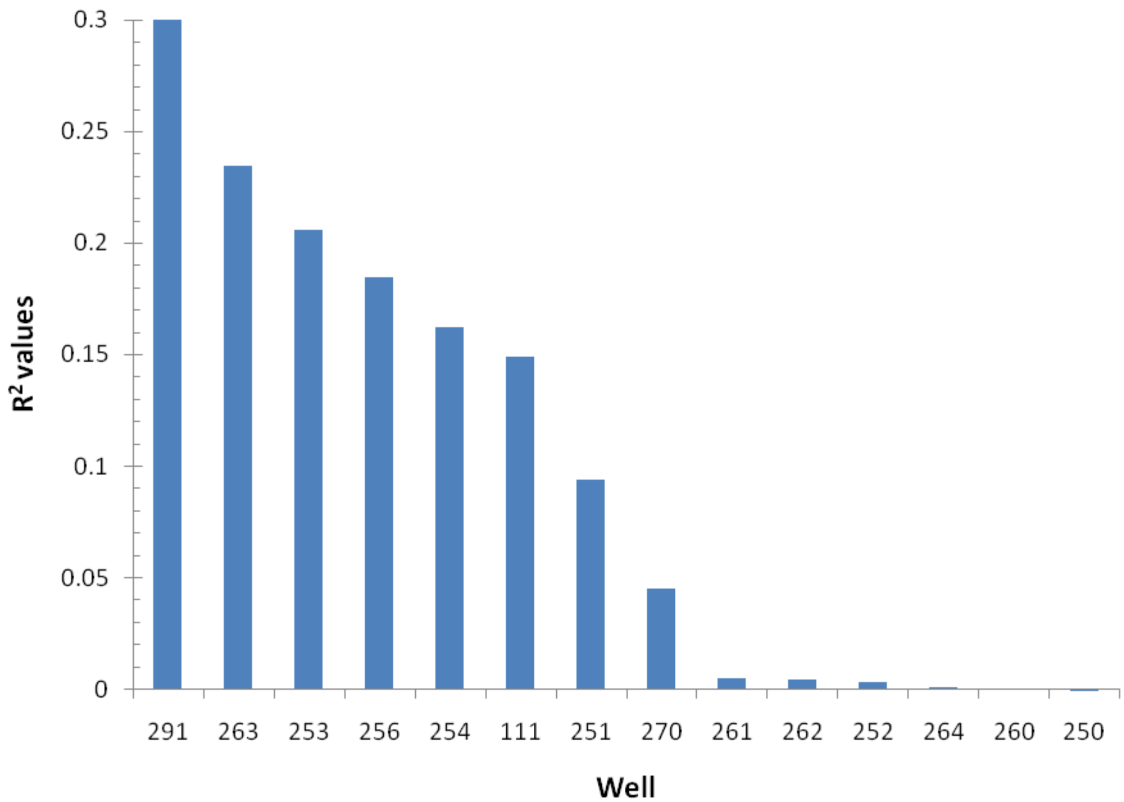


Figure 3-30. R^2 values for a comparison of within well SRP concentrations and groundwater flow.

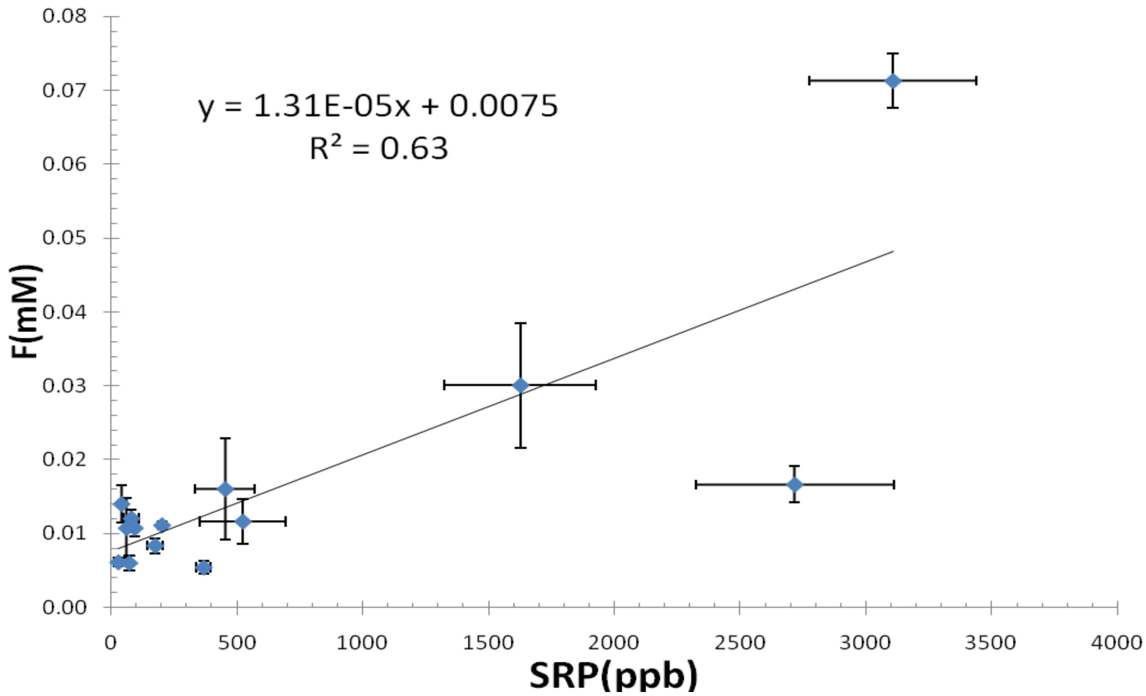


Figure 3-31. The relationship between SRP and F in the water samples taken from the groundwater wells around Newnans Lake. *Concentrations of both analytes are very high compared to surface water samples. Note also that the point that has 2700 ppb P, but low F, is in an area where septic tank loading is high.

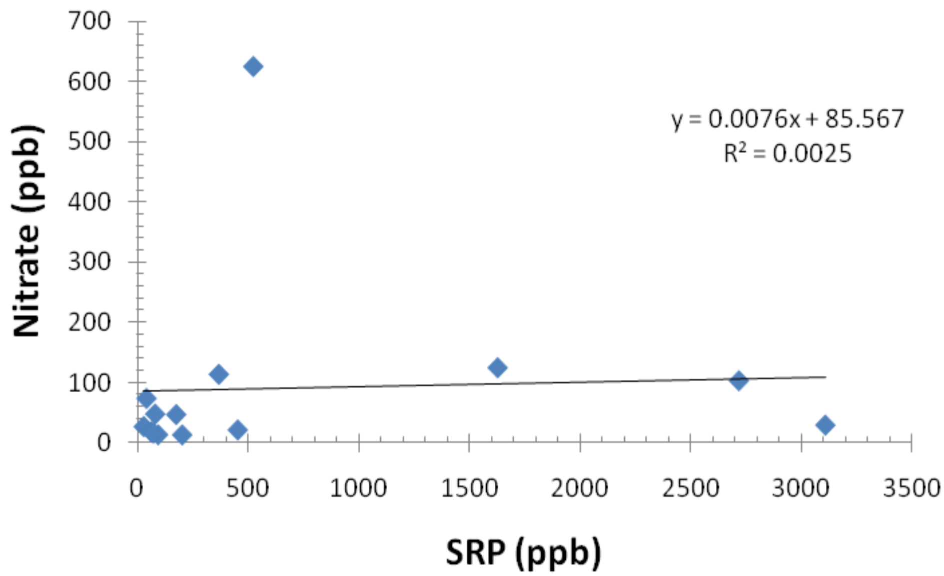


Figure 3-32. The relationship between SRP and nitrate in groundwater taken from the wells around Newnans Lake. *The outlier is well 270 located in the southeast portion of the lake.

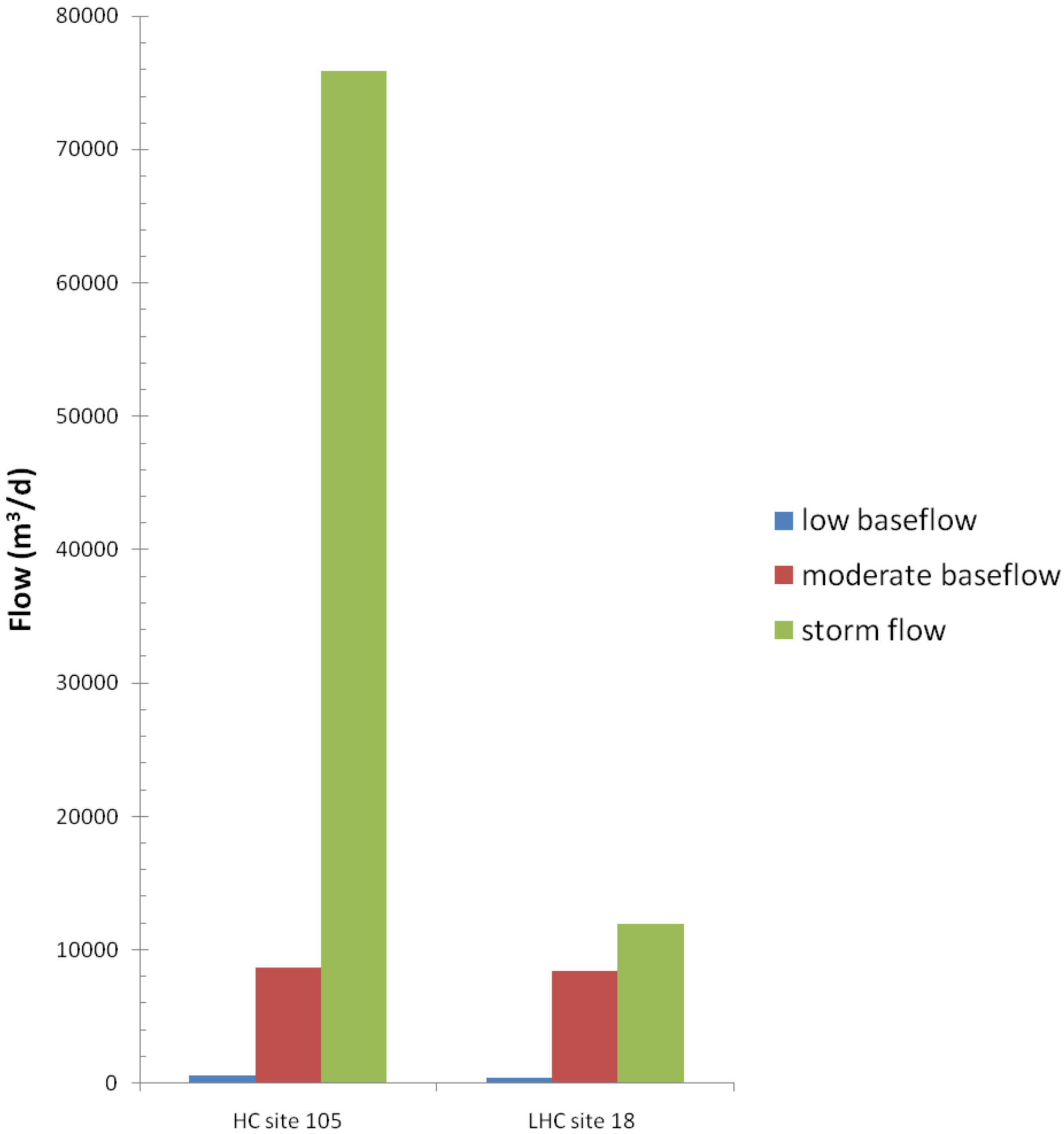


Figure 3-33. Flow rate comparison for each of the three flow regimes for both Hatchet and Little Hatchet Creeks.

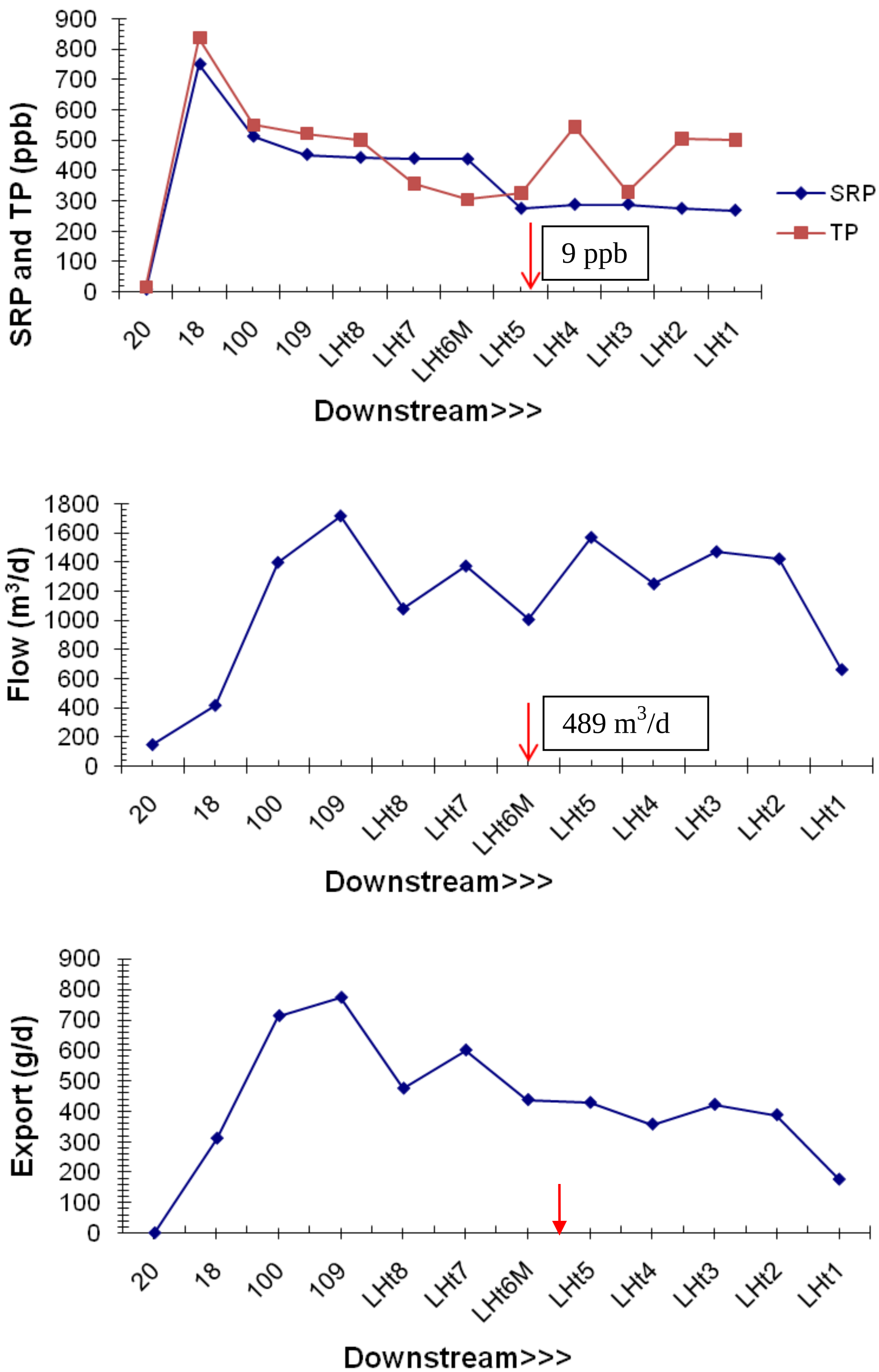


Figure 3-34. Little Hatchet Creek transect during low baseflow. The red arrow indicates a tributary and the flow rate and P concentration are listed in text boxes.

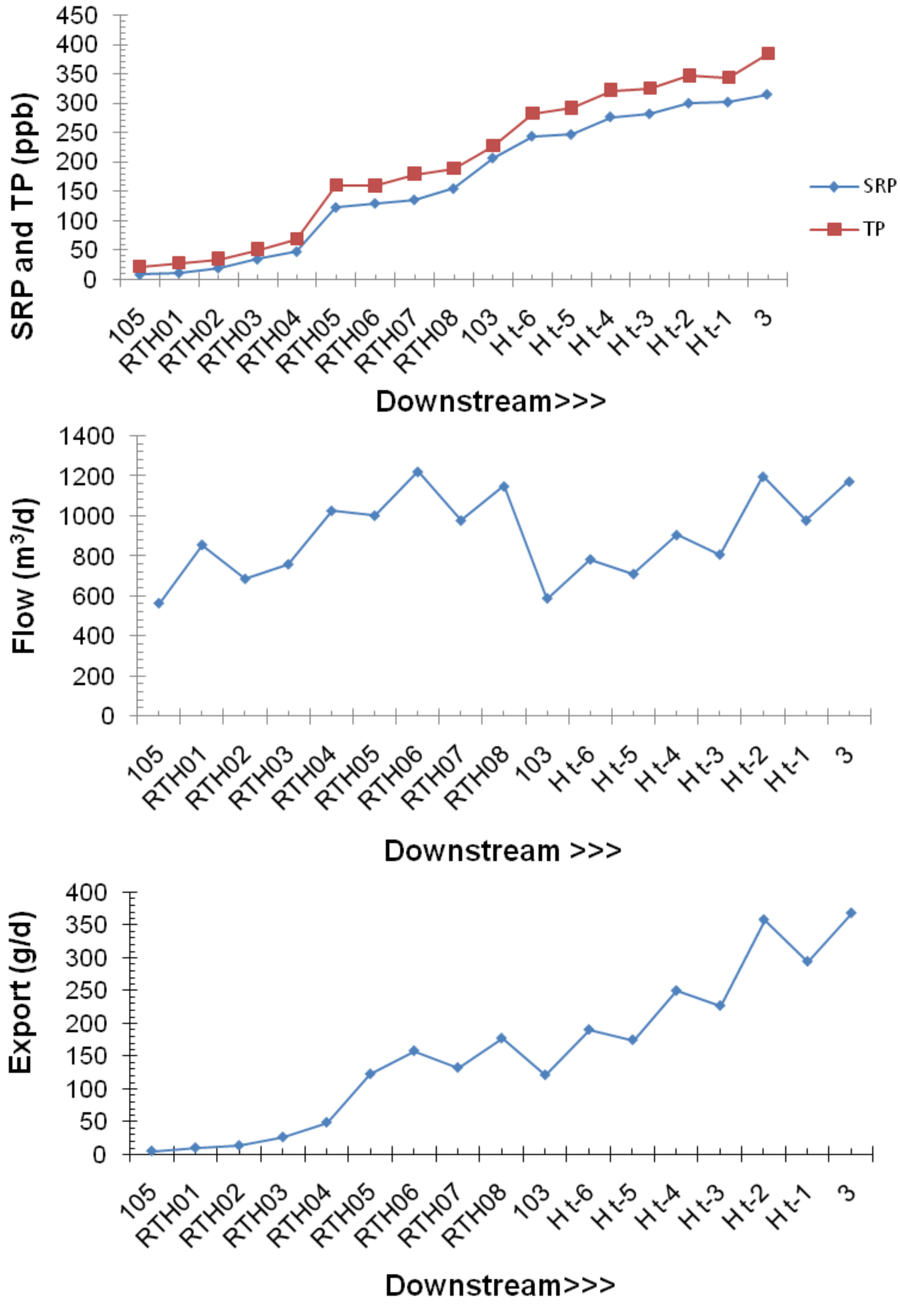


Figure 3-35. Hatchet Creek transect during low baseflow.

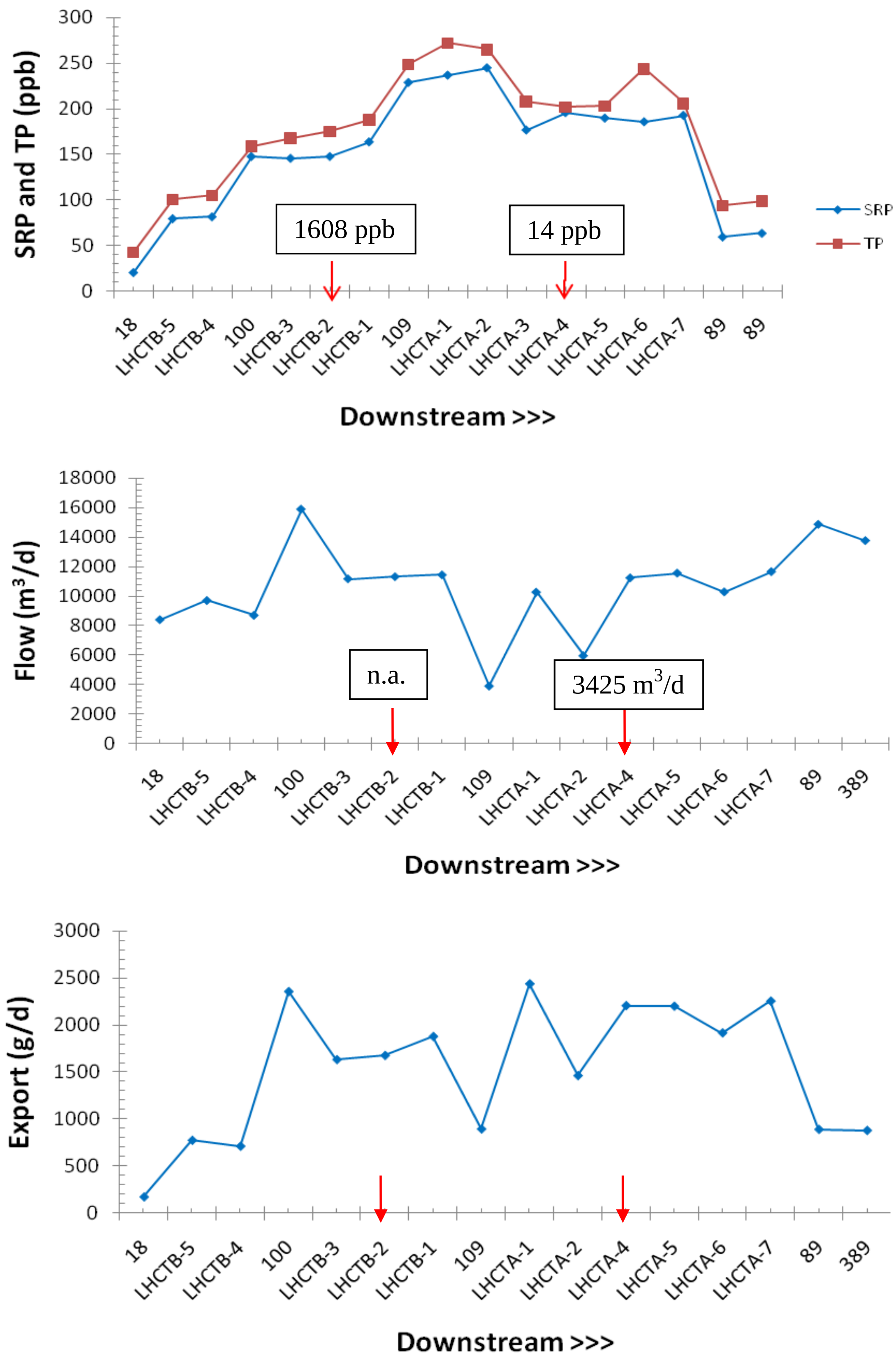


Figure 3-36. Little Hatchet Creek transect at moderate baseflow. Red arrows idicate tributaries and the flow rate and P concentrations are listed in text boxes.

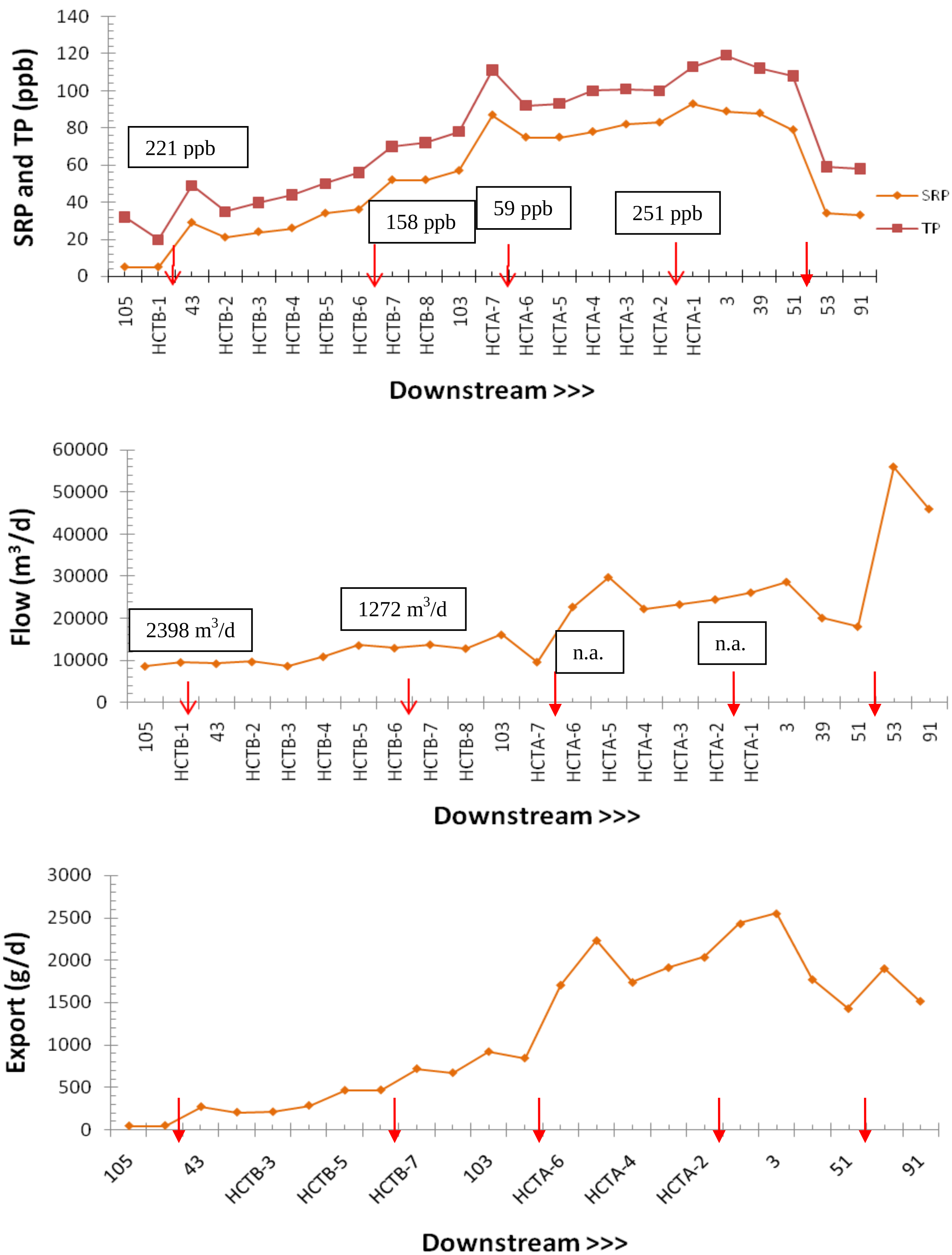


Figure 3-37. Hatchet Creek transect at moderate baseflow. Red arrows indicate tributaries and the flow rate and P concentrations are listed in text boxes.

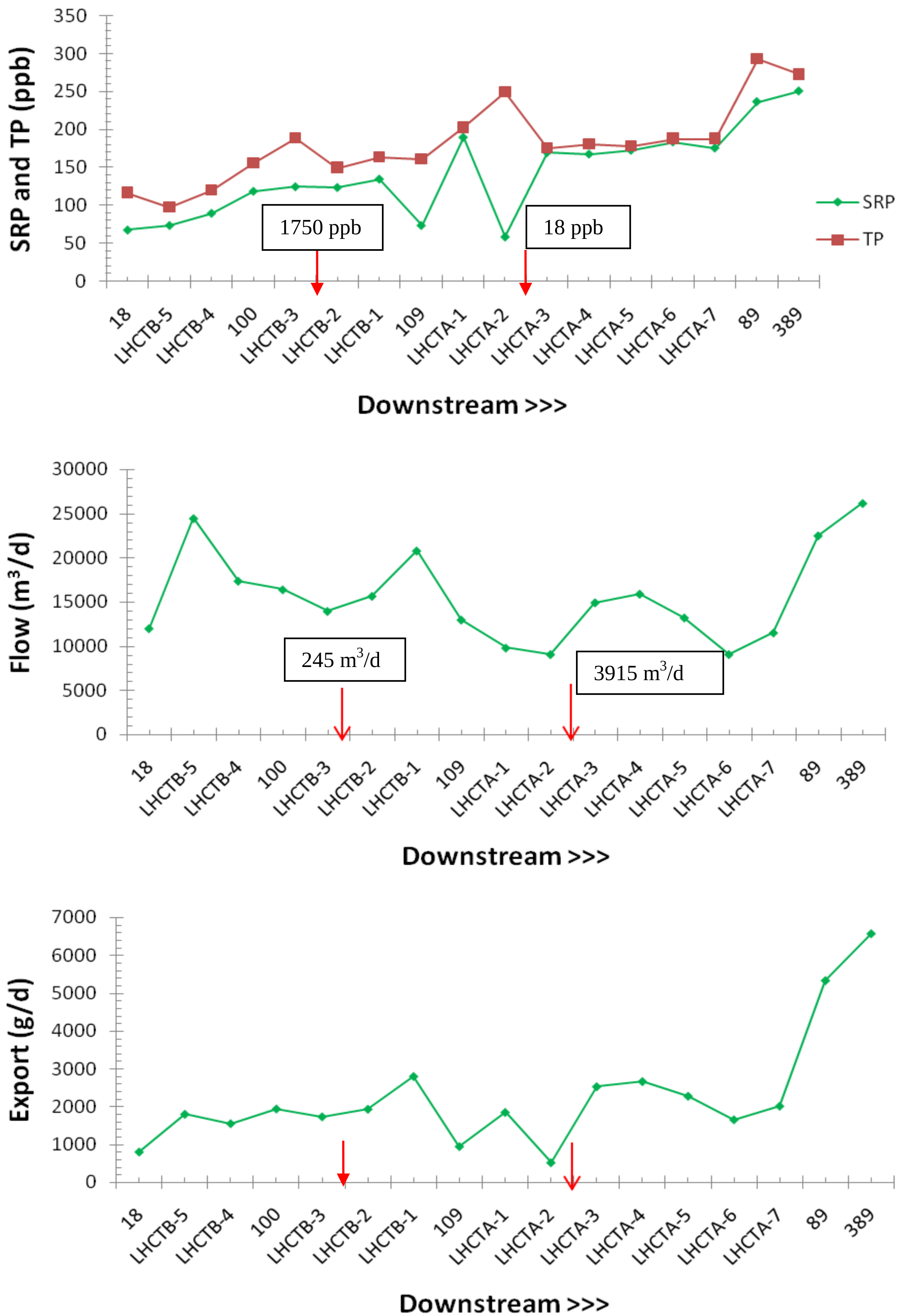


Figure 3-38. Little Hatchet Creek transect during storm flow conditions. Red arrows indicate tributaries and the flow rate and P concentrations are listed in text boxes.

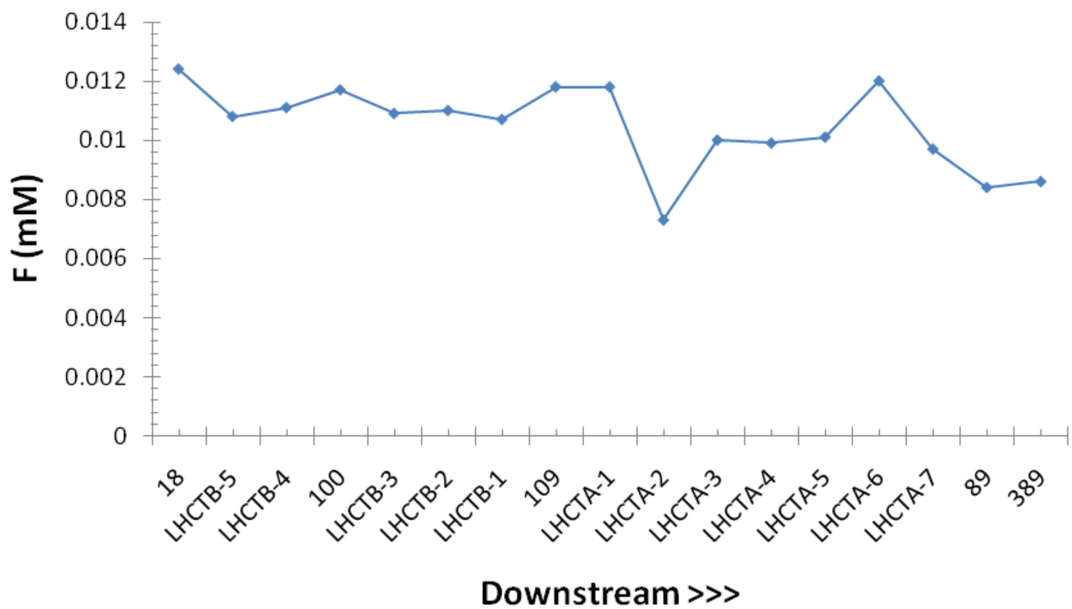


Figure 3-39. F concentrations in Little Hatchet Creek during storm flows.

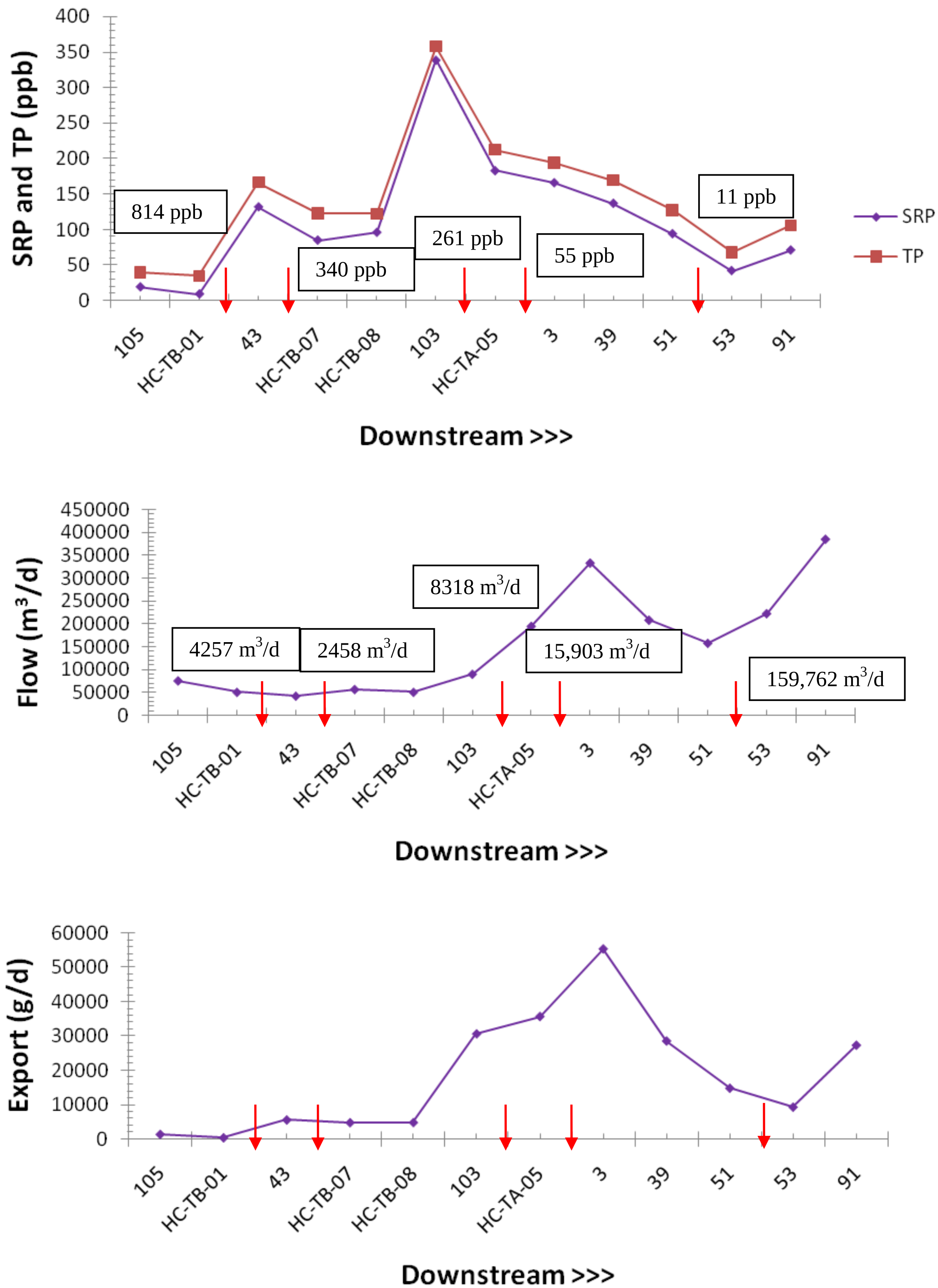


Figure 3-40. Hatchet Creek transect during storm flow conditions. Red arrows indicate tributaries and the flow rate and P concentrations are listed in text boxes.

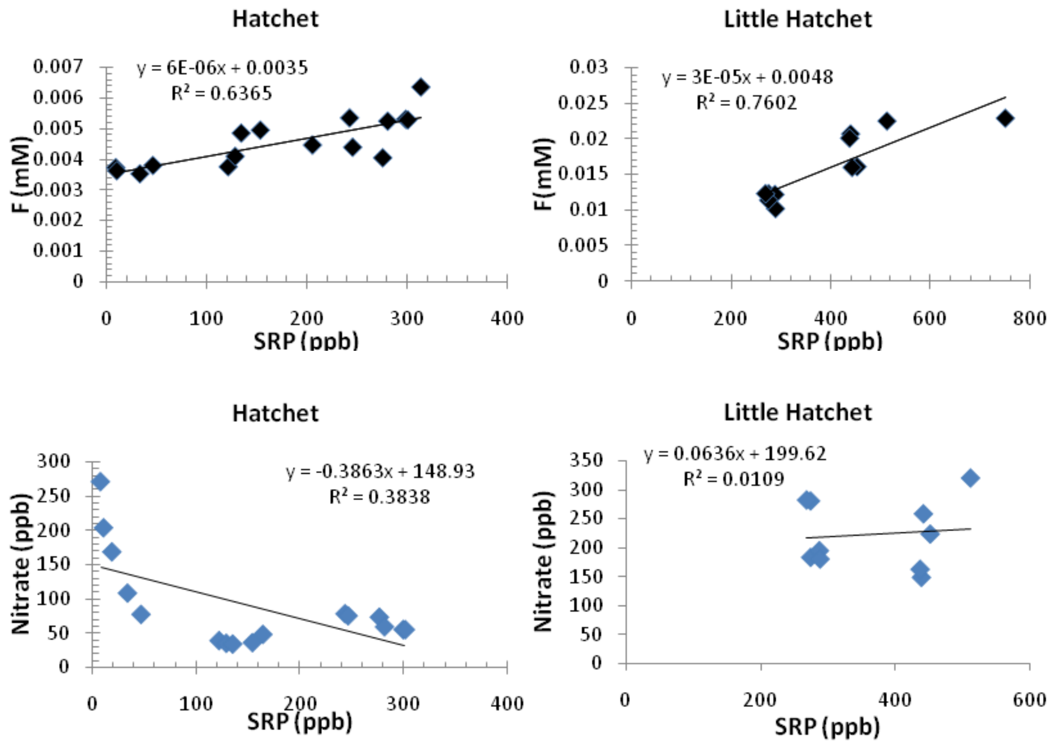


Figure 3-41. The relationships between SRP and F and SRP and nitrate for both Hatchet and Little Hatchet Creeks during low baseflow. *Site 20 is not included for Little Hatchet Creek in the SRP vs. F graph since it is municipal supply water that is F enriched. Sites 20 and 18 were not included in the SRP vs. nitrate graph for Little Hatchet Creek since they were outliers.

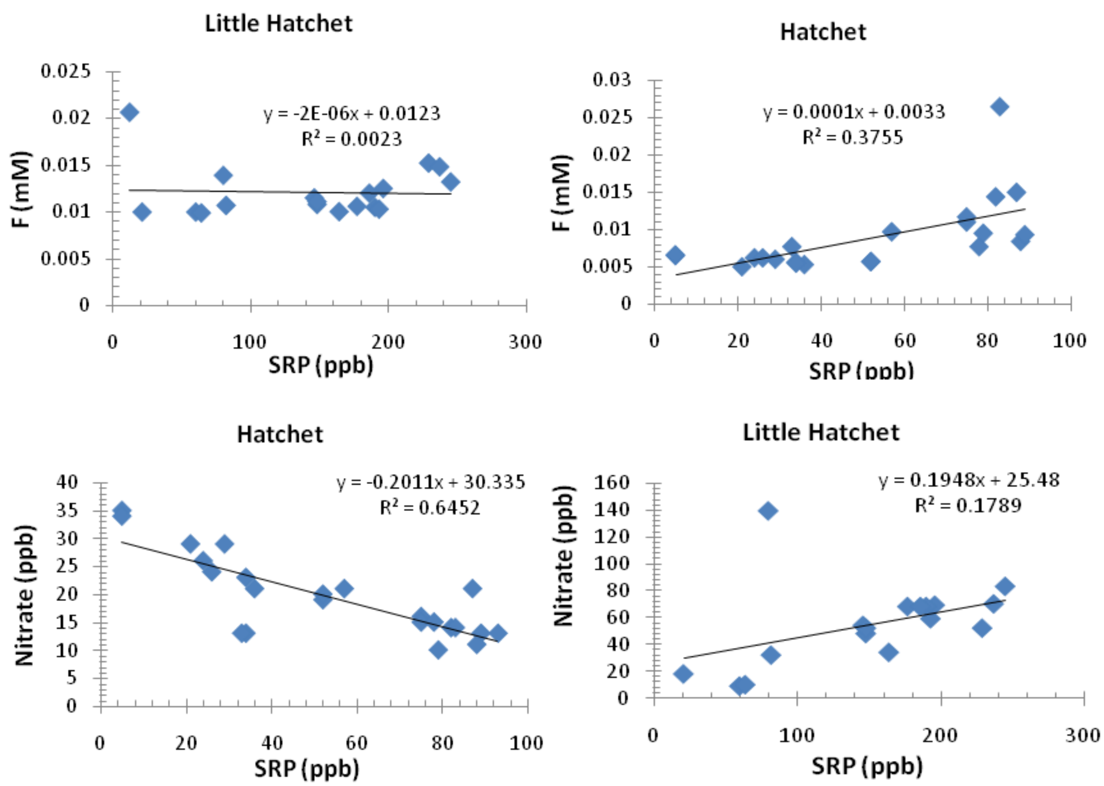


Figure 3-42. The relationships between SRP and F and SRP and nitrate for both Hatchet and Little Hatchet Creeks during moderate baseflow. *F concentrations are enriched substantially compared to typical surface waters in Little Hatchet Creek.

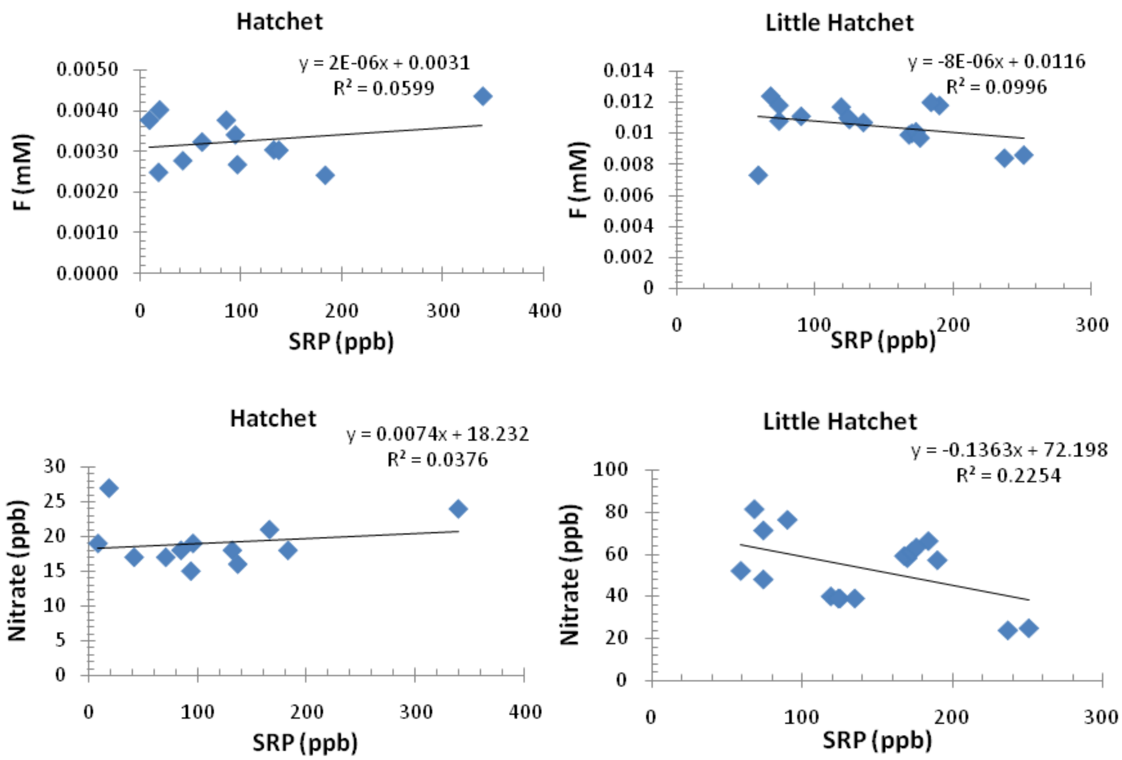


Figure 3-43. The relationships between SRP and F and SRP and nitrate for both Hatchet and Little Hatchet Creeks during storm flows. *F concentrations are still markedly enriched in Little Hatchet Creek compared with the surface water elsewhere in the watershed. Note the magnitude of dilution in F that occurs with storm flow in comparison with baseflow in Hatchet Creek.

Table 3-1. Flow-weighted P concentrations for each of the inflows and outflows in Newnans Lake.

Inflow/Outflow	Flow-weighted P concentration (ppb)
Hatchet Creek	76
Little Hatchet Creek	227
Lake Forest Creek	107
Bee Tree Creek	41
Rainfall	10
Prairie Creek	162
Deficit	109

Table 3-2. Hydraulic conductivity and cross sectional area for each well.

Well	Ksat (m/d)	Cross Sectional Area (m^2)
111	4.4	2747
250	2.2	2433
251	6.1	1538
252	1.5	1400
253	0.3	1288
254	6.3	1495
256	3.2	2373
260	7.7	2534
261	4.4	1981
262	5.0	2479
263	0.8	2546
264	7.0	2725
270	0.8	2066
291	7.0	3058

Table 3-3. Cumulative inflows for October 2007- August 2008.

Inflow	Cumulative Flow (m^3)
Hatchet Creek	9,498,847
Little Hatchet Creek	3,205,460
Lake Forest Creek	1,851,840
Bee Tree Creek	9,708,861
Rainfall	30,607,254
Groundwater	194,320

CHAPTER 4
DISCUSSION

P concentrations are tremendously high in Newnans Lake as well as the entire watershed, which has led to undesirable ecological conditions in the lake. While regulatory efforts have been started that have the goal of reversing the consequences of high P concentrations in the lake, the question of where the P is coming from remains largely unanswered. In order to determine where the P in Newnans Lake is coming from, water and P budgets were created to assess whether there are sources other than surface runoff that are not accounted for (e.g. groundwater seepage and internal recycling).

Water and Phosphorus Budgets

I hypothesized that when only accounting for gauged inputs and outputs both water and P deficits will be observed in Newnans Lake and that the P deficit will be large relative to the water deficit. Neglecting fluxes from the ungauged Prairie Creek Reach, I estimated a cumulative water deficit of 1.48×10^8 m^3 and a P deficit of 16,153 kg over a 10 year period between 1998 and 2008. With the additional estimates of fluxes from Prairie Creek Reach, both water and P budgets appear to close, although with less certainty in the P budget.

During the 10 year period of record, a significant regional drought occurred between 2000 and 2002, and in 2004, numerous tropical storms passed through or near Gainesville creating very wet conditions; the 10-year period of record used to create the water and P budgets cover a wide variety of hydrologic conditions.

Water Budget

The cumulative water deficit over 10 years is approximately equal to the flows in Hatchet Creek. The water deficit can be accounted for by either runoff from the ungauged portion of the lake (Prairie Creek Reach, Figure 2-3) or groundwater seepage. The cumulative water deficit over the 10-year period is roughly equal in proportion to the area of the basin that is ungauged which suggests the deficit could be explained entirely by the runoff from Prairie Creek Reach. In fact, the estimated cumulative surface water flow to the lake from the ungauged portion of the watershed is 1.67×10^8 m^3 which is slightly larger than the calculated water deficit of 1.48×10^8 m^3. On an event basis, the addition of Prairie Creek Reach into the water budget creates a slight water deficit of 2.09×10^7 m^3; to put this number in perspective, Hatchet Creek alone accounts for 1.19×10^8 m^3 over 10 years, more than five times larger than the deficit.

Since the estimated flows from Prairie Creek Reach are fairly certain, these data were added into the final water budget. The annual average water input into the lake is 6.04×10^7 m^3, while the average output is 6.47×10^7 m^3 (Table 4-1 shows annual cumulative and average water input as well as standard deviation); output is approximately 7% larger than input each year. Allowing for 10% error, I conclude that the water budget indeed closes.

Phosphorus Budget

The P deficit is roughly equal to the P inputs from Hatchet and Little Hatchet Creeks combined; this is notably less than the 280% increase in P export reported by Nagid et al. (2001). Although the P deficit is much less than predicted, it still is large relative to the water deficit. The P deficit could be due to several factors: the calculations used to estimate P from surface water flow do not accurately reflect P concentrations; runoff from Prairie Creek Reach is contributing P; groundwater seepage is contributing P; or there is internal P loading in the lake releasing P into the water column.

When I compared the TP I calculated based on relationships between archival P concentrations and gauged surface water flow to the actual measured TP, the data showed that the two methods were comparable even though the validation showed that there appears to be some underestimation of Little Hatchet and Hatchet Creeks. Even with the uncertainty, the P deficit is still large. By completing this check, it is possible that I may have underestimated the inputs, therefore overestimating the P deficit, but with a lack of archival data, the model is still the best option.

The water deficit that results from using only gauged surface water flows is accounted for by Prairie Creek Reach, but it is unclear whether P in runoff from this sub-basin can account for the entire P deficit. In order for the runoff from Prairie Creek Reach to make up for the P deficit, the P input would be approximately as large as the input of Hatchet Creek and Little Hatchet Creek, with a flow-weighted concentration of 109 ppb. A very rough estimate using measured P concentrations and flow on March 12, 2008 suggests that Prairie Creek Reach may contribute 21,516 kg of P to Newnans Lake over 10 years, which is much larger than the P deficit. Input of P may be larger than output of P due to sedimentation; if there is net sedimentation, the P budget may still close even if P inputs are larger than outputs through Prairie Creek. Since the P flux from Prairie Creek Reach is not as certain as surface water flow, these data were not included in the final P budget. However, I conclude that fluxes from the ungauged portion of the watershed constitute the majority of the missing P, although the uncertainty may leave room for minimal amounts of internal recycling or groundwater seepage.

I hypothesized that during dry periods the deficit would be larger due to the concentration of P when lake levels are low; I did not find this to be true. There were P surpluses during 1999 to 2002, most likely influenced by the drought and the lack of flow in Prairie Creek, where the annual average input was 593 kg and the average output was 4 kg (Table 4-2 shows the cumulative and average inputs and outputs and also the standard deviation). From 2003 to 2008, there were P deficits each year with an annual average input of 2987 kg and an output of 6074 kg.

Groundwater Wells

Groundwater Flow

I hypothesized that both the water and P deficit will be accounted for by P-rich groundwater contributions to the lake, mainly during low lake stage. The water budget closes when accounting for water from Prairie Creek Reach, suggesting that groundwater inflows cannot be large without also requiring a significant unaccounted for water expense. I found a cumulative inflow of groundwater of 140,375 m^3 from October 2007 to August 2008, which is less than 1% of total inflows (creek flow and rainfall) over that same period. I can therefore conclude that lateral groundwater seepage is not a significant source of water into Newnans Lake.

Groundwater seepage into Newnans Lake was expected to occur when the water table is higher than the water level in the lake. This prediction was not observed in the perimeter wells during the period of observations; I found no significant relationship between lake stage and groundwater flow. During the period of sampling, lake stage ranged from 19.6 m msl to 20.27 m msl, and since the lake has been historically much lower, it is possible that during dryer years when the lake elevation drops below what we measured, that all of the wells would show groundwater seeping into the lake. On the other hand, when lake levels are higher than were observed during this study, we might see increased groundwater outflows (during the last 10 years, lake stage ranged from less than 19 m msl to 21.6 m msl, the higher end being larger than what was observed during the groundwater study).

In order to determine if significant lateral groundwater seepage is possible under other weather conditions than those observed in this study, I computed the vertical gradient that would be needed in order to generate 10% of the water deficit. Assuming that each of the 14 wells would contribute an equal amount of groundwater to the lake, each well would need to contribute

290 m^3/d in order to contribute $1.48x10^7$ m^3 over 10 years (10% of the deficit). Using well 111, the vertical gradient required to yield 290 m^3/d of groundwater seepage to the lake would be 68 cm. In contrast, the largest vertical gradient for well 111 measured during my study was 25 cm, which corresponded to a lake stage of 19.6 m msl, the lowest lake stage observed. At this low lake stage, 9 of the 14 wells showed groundwater inputs to the lake, and of those 9 wells, only 4 wells showed the largest vertical gradient at 19.6 m msl. The erratic groundwater flow, even at the lowest lake stage observed, combined with the hydraulic gradient of well 111 only being 37% of what is needed to make up for 10% of the water deficit, lends support to the conclusion that direct groundwater seepage is not a significant contributor to the water budget in Newnans Lake under any hydrologic condition.

The variation in saturated hydraulic conductivity (Ksat) of the soils surrounding Newnans Lake does not appear to exert much control over the magnitude of the flows. Our findings parallel the conclusions by Schneider et al. (2005), who found that the substrate surrounding a large, shallow lake in New York had no effect on the rate of groundwater seepage into the lake. Their study also found that rainfall causes large increases in groundwater flow. This was not apparently the case for Newnans Lake; a significant relationship was not observed between antecedent rainfall (the sum of 30 days prior to the groundwater measurement) and groundwater flow. There was a slight negative trend in the relationship between groundwater flow and rainfall, which is the opposite of what would be expected. I predicted that lake stage would drive groundwater flow and Schneider et al. (2005) suggested that rainfall may drive groundwater flow in large, shallow lakes; neither of these predictions held true for Newnans Lake. When the effects of lake stage and rainfall were combined (rainfall/lake stage) and regressed with groundwater flow, a significant relationship was still not observed suggesting that neither antecedent rainfall nor lake stage control groundwater seepage (unless there is a more complex relationship between the two).

No physical relationships appear to drive flow and so another possibility is that there may be error associated with the groundwater measurements. Hydraulic gradients were calculated by measuring groundwater and lake elevations in relation to the fixed top of the wells using a tape measure (Table 4-3). The accuracy of the measurements was approximately 0.5 cm, so using the tape measure may cause an error when measurements were below 0.5 cm; 25 out of 180 hydraulic gradient measurements were 0.5 cm or less suggesting that 14% of the groundwater seepage rates may have been calculated in error. A 0.5 cm error on a gradient of 0.5 cm is a very small hydraulic gradient and would only contribute very little flow, and since 86% of measurements were larger than my presumed uncertainty with measurement, I can discount measurement error.

I would also expect location of the wells to affect groundwater seepage rates. Location with regard to elevation of the land area surrounding Newnans Lake only appears to affect groundwater flows minimally. The steeper western side of the lake generally had higher groundwater flows with the exception of well 253, and the flat northern and southern edges of the lake had lower groundwater seepage with the exception of well 291, which had flows two orders of magnitude larger than the other wells. Although local steepness does not appear to substantially affect groundwater seepage, the location around the lake compared to the natural levee that exists around some of the perimeter may have an effect. In some areas around the lake, a natural levee exists, behind which is a backwater wetland. This levee was presumably formed when hurricanes or other large storms deposited sediment from the lake onto the shore. Natural sorting of sediment material could allow compaction, thereby affecting groundwater flow rates. During wet periods, there are areas in the northern part of the lake where these back water swamps discharge to the lake via surface connections; in those areas, it is possible that groundwater flows to the swamps, but discharges to the lake as surface water through microchannels.

Groundwater Phosphorus

Next, I hypothesized that P from groundwater seeping into the lake is of geologic origin. A significant positive relationship was found between SRP and F across all of the groundwater wells surrounding Newnans Lake, consistent with the presence of geologic P in the groundwater. Despite high groundwater concentrations (e.g. well 254 averaged 3249 ppb), the net P flux was actually out of the lake (50 kg over the 15-month period of record). As such, even though geologic P is present, it is evidently not getting into the lake through groundwater seepage. I can therefore conclude that lateral groundwater seepage is not a significant source of P to the lake.

Many of the groundwater wells had very high SRP concentrations, especially in the most developed western edge of the lake. The western edge of the lake where wells 251, 252, 253, and 254 are located appears to have less contact with the Hawthorn Group than the rest of the lake perimeter (Figure 1-4). Even though wells 252, 253, and 254 do not appear to be in direct contact with the Hawthorn, they have the highest F concentrations of the 14 wells; this could indicate that P in the groundwater is due to septic tanks, but a lack of an association between SRP and nitrate suggests otherwise. Septic tank effluent is generally composed of ammonia and dissolved organic N and not nitrate, and it is likely that any N present has not been nitrified due to reducing conditions, which could be why I did not find high concentrations of nitrate. Ammonia and total N were not evaluated for the groundwater samples and so it is not certain whether there is septic tank influence in wells 252, 253, and 254.

While evidence from perimeter wells suggest that groundwater seepage from the perimeter of the lake is negligible, Burnett et al. (2007) report large groundwater fluxes to the lake through the sediments. Specifically, Burnett et al. (2007) observed high levels of radon and methane, natural gases used as groundwater tracers, in Newnans Lake. Using a diffusion model that estimates the groundwater seepage necessary to maintain the observed radon concentrations (based on observed porewater concentrations and known gas diffusion rates and the assumption that the principal mechanism to deliver excess radon to the water is advection), Burnett et al. (2007) estimated groundwater discharge of 1.46×10^5 m^3/d. Assuming constant groundwater seepage of this magnitude, the ungauged water input to the lake would be 5.33×10^8 m^3 over a 10-year period, substantially larger than the water deficit of 1.48×10^8 m^3 (not including flows from Prairie Creek Reach) predicted in this study over that same period. Moreover, extrapolating the groundwater discharge that Burnett et al. (2007) report over 10 years creates the reverse problem wherein there is more water entering the system than leaving, by a factor of 1.6. Although the model implemented by Burnett et al. (2007) is widely used, there is no known flow path via which such a large magnitude of water could be lost since both ET and Prairie Creek outflows (the only two outflows of note given low perimeter groundwater seepage) are well constrained. In examining the methods that led to their flow rate, it is clear that their sampling trail was a small subset of the lake, mostly in the southern portion, and that the concentrations of radon were highly heterogeneous across the lake, with several observations that were an order of magnitude higher than the mean concentration. This yields estimates that are strongly skewed, possibly leading to an overestimation of groundwater discharge.

Burnett et al. (2007) found strongly heterogeneous radon concentrations, but this study observed low perimeter seepage rates; to reconcile these observations, one explanation is a point source of groundwater (a subaqueous spring). One way to examine lake bottom groundwater seepage is through aerial thermal maps. Newnans Lake is fairly well mixed, so if areas of different temperature are present on the thermal map, these could reasonably be attributed to groundwater inputs, particularly if consistent patterns are observed across seasons and years. There is strong precedence for intermediate aquifer springs in this area (Scott et al., 2004), and the location would most likely be just off Palm Point on the west side of the lake, the steepest sloping area of the lake edge. During my study, I found that temperature in Newnans Lake reached 13.9 °C in January 2008 and 34.3°C in July 2008. Thus, there is a good chance that if a subaqueous spring exists, it would be apparent on a thermal map for either month (assuming that groundwater is constant at 21°C); given the magnitude of the proposed groundwater flux, the spatial extent of any thermal anomaly would be relatively large, and detectable using coarse-scale thermal imagery available on the Landsat platform (approximately 60 m pixel resolution). Upon examining the thermal maps, there is no consistent evidence suggesting a point source indicating a possible spring (Figure 4-1). Given the observed trends, it is unlikely that lateral or lake bottom groundwater seepage is contributing a significant amount of water to the lake.

Assuming a P deficit of 16,153 kg and groundwater seepage rate of 5.33×10^8 m^3 over a 10 year period (Burnett et al., 2007), the groundwater P concentration would be 30 ppb. If the groundwater contribution predicted by Burnett et al. (2007) is correct, then the P concentration is very low for groundwater, particularly since porewater in the lake is typically over 2 mg P/L (L. Long, unpublished data). If groundwater is seeping through the bottom of the lake, it is also likely P from the Hawthorn Group is using the groundwater as a conduit into the lake, which may account for the P deficit. Since Burnett et al. (2007) found that radon is present in the lake water, this indicates that P is present as well. The fact that radon is found in the lake water implies that there is advection of pore water into the lake and not diffusion since diffusion across the air-water interface is much faster than diffusion across the water-sediment interface. It is possible that large amounts of P are being contributed to the lake through lake bottom groundwater flows, but since the water budget closes without the input of groundwater, it is unlikely that P is entering the lake in this manner.

Alternatively, P may be released from porewater along with radon due to turbulence during storm events (Figure 4-2). At low stage in Newnans Lake, organic sediment compacts and/or mineralizes; following rewetting, the mineral soil containing radon is nearer the sediment-water interface. Wind-induced turbulence may entrain mineral sediments into the water column, along with the porewater that had equilibrated with those sediments, thereby mixing radon into the water column, though without advection of groundwater. This effect would be transient, but sampling the water column after a sediment entrainment event could yield radon concentrations that would generally be interpreted as advection; only continuous measurements of radon and/or other mineral porewater constituents would inform whether this alternative mechanism for radon and P enrichment, which requires no net advection of water into the lake, is plausible. Notably, when the lake level rises and the sediment entrainment is reduced because of weaker mixing gradients, radon associated with the porewater will not be mixed into the water column, and P concentrations in the water column would also be reduced, consistent with observed negative relationship between stage and water column P concentration.

If the P is not being advected or entrained into the water, it may be diffusing in from the lake sediments. Numerous studies have noted greater P outputs than inputs at the whole lake scale indicating some manner of internal loading (Nagid et al., 2001; Gao and Gilbert, 2003), but models from the SJRWMD indicate that this is not the case (Di et al., 2009). Internal recycling has been suggested to be the major contributor of P to the water column for many years, but it has never been directly quantified. It is important to note that this is a temporary phase; if net internal loading is happening in Newnans Lake, then over time, the P will be mined out of the sediment or buried as it moves below the mixing zone.

A study by Bachmann et al. (2000) suggested that 100% of the Newnans lakebed is subject to resuspension and that 21% of the lake area is disturbed 50% of the time. Assuming that the P deficit computed from the elemental budget work herein is solely due to internal loading (i.e. there is no loading from Prairie Creek Reach), the sediment P flux rate would be 598 $mg/m^2/10yr$ or approximately 60 $mg/m^2/yr$. In order to put this number in perspective, internal loading rates for Lake Apopka and Lake Okeechobee, both large, shallow lakes located in Florida, were compared to this estimate for Newnans Lake. According to a P mass balance model for Lake Okeechobee by James et al. (1997), 1.92 $mg/m^2/d$ or 701 $mg/m^2/yr$ SRP is released to the water column by resuspension. In that study, the P deficit was found to increase with increasing lake stage. This is contrary to the expectation that internal loading in Newnans Lake due to wind-induced turbulence would be accelerated at low lake stage. It is possible that diffusion of P from the sediment into the water column increases at high lake stage due to diluted P concentrations in the lake. A laboratory study conducted by Reddy et al. (1996) found that diffusion alone in Lake Apopka releases 1 mg $P/m^2/d$ (365 $mg/m^2/yr$), which is larger than the estimated rate for Newnans Lake that would be necessary to account for the P deficit observed in this study. The internal loading and diffusion rates for both Lake Apopka and Lake Okeechobee exceed the estimated rate for Newnans Lake, indicating that it is possible for the P deficit in Newnans Lake to be comprised of internal loading from diffusive fluxes alone. While diffusion cannot explain the high radon concentrations in the lake, the mass fluxes of P should be an area of further study.

Longitudinal Creek Sampling

Finally, I hypothesized that groundwater seepage along stream banks is an important pathway for geologic P delivery to Newnans Lake. Although groundwater seepage around the lake perimeter does not appear to contribute significantly to the lake, the longitudinal creek sampling on Hatchet and Little Hatchet Creeks showed strong evidence of substantial groundwater and P fluxes seeping into the creeks. Most notable was Hatchet Creek: during low baseflow conditions, Hatchet Creek accumulated water and P longitudinally without the input of tributaries. This longitudinal increase continued during moderate baseflow for SRP, but flow was much more variable. The longitudinal increase in SRP was not observed during storm flows, as expected, due to reduced groundwater seepage within the creek itself. I also predicted that SRP concentrations would be reduced due to a dilution effect from the increased rainfall; this does not appear to be the case since P concentrations are still very high. The increased rainfall has the effect of decreasing the pH in Hatchet Creek to an average of 4.9 (this is not observed in Little Hatchet Creek); apatite would dissolve under low pH therefore liberating large amounts of P.

It would appear that the longitudinal increases are lost due to pulses through the system; during storm flow, when flow is high, SRP generally tends to be high. Presumably most of the flow during those periods is either overland or via seepage at some upstream location. Furthermore, there was a significant positive relationship between SRP and F, along with decreasing nitrate as SRP increased, during low and moderate baseflow indicating that geologic sources dominate. During storm flows, this relationship is no longer observed, but the persistent high P concentrations during all flow regimes in the tributaries indicate that some P source is still contributing upstream of the transect.

The longitudinal SRP and F relationship was only significant for Little Hatchet Creek during the low baseflow regime; during low baseflow, there was no association between nitrate and SRP. When SRP and F were examined over time at downstream stations on Little Hatchet Creek (near site LHCTA1 in Figure 2-10), Cohen et al. (2007) showed a near perfect relationship ($R^2 = 0.97$, $p < 0.0001$) between SRP and F. They infer that P flux is of geologic origin, possibly from groundwater seepage, but perhaps only in the lower reaches of the creek, possibly below Gainesville Regional Airport. The creek also appears to be in contact with the Hawthorn Group in the lower reach, but not the upper reach (Figure 1-4). SRP concentrations observed during my longitudinal transects were consistently very high in Little Hatchet Creek suggesting that groundwater seepage and geologic P loading may be occurring upstream of the transect. The correlation between nitrate and SRP at moderate baseflow indicates that there may also be an anthropogenic P load in Little Hatchet Creek, though the concentrations of nitrate are still very low compared to the concentrations of phosphate (mineral N:P molar ratios of less than 1:1). During storm flows, the relationship is not significant, but it is positive suggesting possible anthropogenic influence; as with the moderate flow condition, the nitrate concentrations are markedly lower (approximately one order of magnitude) than P concentrations. It appears that while groundwater seepage occurs throughout Hatchet Creek, groundwater seepage in Little Hatchet Creek is more localized and there may be more of an anthropogenic influence than what is observed in Hatchet Creek.

During low and moderate baseflow, SRP concentrations in Little Hatchet Creek decrease once the creek flows into Gum Root Swamp, indicating that the swamp is, at least temporarily, sequestering P. During storm flows, flow increases dramatically through the swamp as do SRP and TP concentrations. TP concentrations are higher than SRP indicating that the wetland is releasing stored P in particulate or organic form that was sorbed onto organic matter; F concentrations were also low in Gum Root Swamp further indicating that the P is released from wetland stores, and not from groundwater fluxes. Mass flux of P to the lake is very low during baseflow conditions, but during storm flows, export increases from 800 g/d to nearly 7000 g/d indicating that the wetland acts as a source of P (though principally organic P) to Newnans Lake during storm pulses. The lability of this episodic P input is unknown.

The longitudinal creek sampling showed that bank seepage was occurring and also contributing geologic P, presumably from the Hawthorn Group, to Hatchet and Little Hatchet Creeks. Furthermore, when comparing the P:Cl ratio over time at site 3 in Hatchet Creek and site 109 in Little Hatchet Creek (Figure 4-3), I observe a significant relationship. As flow increases, P:Cl decreases suggesting that during baseflow, weathering is occurring since the water in the groundwater table is in contact with the Hawthorn Group for a long period of time releasing a large amount of P. The groundwater then slowly seeps into the creek acting as a conduit for P. Assuming that Cl concentrations are constant in the groundwater (with the exception of evapoconcentration), I would expect P:Cl to be large during baseflow, which is what was observed. After a period of little rainfall, Cl will be highly evaopconcentrated in the soil; when a storm pulse comes through, large amounts of Cl is pushed into the creek along with P which is most likely why a lower P:Cl is observed during high flows even though P concentrations are still high. These relationships suggest that during all three flow regimes, water reaches the creeks through groundwater flow, and not overland flow.

Bee Tree Creek and areas of Lake Forest Creek also appear to be in contact with the Hawthorn Group and I expect that if longitudinal samples were taken along those two creeks, there would also be evidence of geologic P, though the low concentrations observed in these creeks across a suite of flow regimes (Cohen et al., 2008) may suggest that they are not as impacted by interactions with the Hawthorn as Hatchet and Little Hatchet Creeks. These four creeks combined account for 78% of water inflow and approximately half of the input of P into Newnans Lake. Moreover, because proximity of the Hawthorn to the land surface is most pronounced near the lake, any flows arriving to the lake from the ungauged Prairie Creek Reach are likely to be influenced heavily by interactions with the Hawthorn. The spatially distributed sampling around Prairie Creek Reach (Cohen et al., 2008) suggests that numerous ephemeral creeks carry high concentrations of P and F, underscoring this interaction between surface water fluxes and the P-rich geology of the region. Overall, this suggests that even without direct evidence confirming the source of the P deficit, geologic P is important to the nutrient dynamics in Newnans Lake, prompting several important management considerations.

Management Implications

While Newnans Lake has likely always been eutrophic, suggested by the high P concentrations in the deeper sediments of Newnans Lake (Kenney et al., 2002), recent increases of P in the water column have created hypereutrophic conditions since 1999. There are multiple explanations that should be considered for this recent pattern of extreme phytoplankton productivity. One possibility is that anthropogenic activities coupled with a drier climate have increased the mobilization of geologic P to the lake. The impermeable surfaces associated with urban areas increase the rate of runoff, therefore incising into creek beds, and possibly exposing more of the Hawthorn Group to the soil surface within the creeks. The Gainesville Regional Airport, currently receiving a lot of scrutiny as a site of increased geologic P loading, was built in the 1940s; this suggests that significant eutrophication would have been observed prior to 1999 if the airport were the major contributing factor. In short, the timing of the airport construction is not consistent with a recent regime shift in the lake, indicating that something else is likely driving the recent increase in nutrients in Newnans Lake.

Another explanation is that the removal of the flow control structure on Prairie Creek altered the water column particulate P balance towards a net flux to the water column. The weir on Prairie Creek was installed in 1966 and was removed in 1999, and during that time period, lake stage, and therefore water leaving Newnans Lake through Prairie Creek was controlled (Gao and Gilbert, 2003). The lake was not permitted to reach its historic lows, during which P entrainment and remineralization would be expected to be at their maximum; as such, P concentrations may have been artificially low during this time. In 2003, the FDEP conducted a study on Newnans Lake to determine a TMDL for the lake and subsequent P load reductions. The data that guide the TMDL for Newnans Lake were from the period during which the weir was in place and the water levels were managed quite differently than today; this may lead lake managers to strive to reduce P concentrations below levels that are feasible given the geologic loading. In the absence of water quality data prior to the weir being installed, it is difficult to assess what target concentrations ought to be.

An additional problem is the uncertainty about where the P is coming from. This study suggests that the water budget closes when accounting for all of the surface water flows, and therefore, groundwater seepage is most likely not a significant factor in the elemental budgeting for Newnans Lake. Since estimating P concentrations is not as certain as estimating surface water flow from Prairie Creek Reach, a P deficit may still exist, indicating some of the P in the lake may be contributed by internal recycling. Nagid et al. (2001) assert that most of the P in the water column is internally recycled, and therefore management of lake levels, not exogenous loading, is likely to yield the desired water quality condition. There are many problems associated with management of lake levels; the first being that sediments can and do build up behind the water control structure since sediments would not be permitted to leave the lake through Prairie Creek. The flashboard removal from the weir in Newnans Lake in 1991, and the ultimate removal of the weir itself in 1999, was implemented primarily with the idea that cypress trees located along the perimeter of the lake need lake level fluctuation to reproduce (Nagid et al., 2001), and moreover that the accumulation of algal detritus on the lake bottom exacerbates undesirable ecological conditions (Gottgens and Crisman, 1992).

The surest way to reverse eutrophication in water bodies is by reducing the loading rate of P, though this process may be subject to significant response lags arising from the resilience of an algal dominated state (Scheffer et al., 2001). Since much of the P present in the watershed of Newnans Lake is evidently geologic in origin from the Hawthorn Group, point source controls of P, though still useful, are not likely to yield dramatic water quality improvements. In other words, regulatory agencies should not expect P load reductions from the predominantly low-intensity land use in the watershed to result in appreciable P reductions in the lake. Understanding how humans have accelerated geologic P loading and discerning ways to manage the loading should become important objectives to Newnans Lake managers; strong evidence that this is already occurring has begun to emerge (Hiers, 2009).

In the case of Newnans Lake, approximately 50% of P is known to enter the lake through gauged surface flow and rainfall, and assuming that the flow weighted mean P concentration of the ungauged portion is the same as the rest of the basin, Prairie Creek Reach contributes another 40% of the load. While surface water P inputs contribute the majority of the load, internal recycling may still be a factor, albeit small. Promoting sedimentation or raising water levels are two examples of how to decrease P release into the water column through sediments.

Sedimentation is the process of sediment settling out of the water column onto the bottom, thus ensuring P stays adsorbed to the soil particles. Raising water levels would theoretically decrease sediment resuspension since the lake would no longer be shallow enough to promote internal loading, because the sediment would not be disturbed by the mixing from the wind; if P is being released due to diffusion at high lake stage, this control would not be valid.

It is possible, and even likely, that the majority of the P deficit is loaded from Prairie Creek Reach. In this case, managing in lake processes may not be as effective with the exception of promoting sedimentation and settling of particulate P. With large inputs of geologic P being contributed by the creeks, it is even more important to understand how humans accelerate geologic P loading. Management strategies may include sequestering P in wetlands before it reaches Newnans Lake. Due to high geologic P inputs, it is important for management of the lake to shift from controlling anthropogenic sources to capturing P before it reaches the lake and promoting in lake processes such as sedimentation.

Conclusions

- Both water and P budgets nearly close with the inclusion of estimates from the ungauged portion of the watershed Prairie Creek Reach, although the uncertainty does leave room for small inputs from groundwater seepage or internal recycling.
- Groundwater loading has been shown to be low during dry climate conditions and it is unlikely that it is ever important.
- There are extremely high P concentrations in the groundwater.
- A positive significant relationship between SRP and F exists for the groundwater wells indicating the presence of geologic P in the groundwater.
- There is evidence of bank seepage contributing geologic P in both Hatchet and Little Hatchet Creeks during low and moderate baseflow. This is indicated by the longitudinal increases in flow and P and the significant positive relationship between SRP and F in Hatchet Creek. In Little Hatchet Creek, a significant relationship was also observed between SRP and F along with the results from Cohen et al. (2008) demonstrating a near perfect relationship between SRP and F over time at three downstream sites.

- During storm flows, these relationships in the creeks break down, but P concentrations are high during all flow regimes indicating that some other mechanism is still occurring upstream of the transect.

- There is no evidence of anthropogenic P in Hatchet Creek during any flow regime, but the weak associations between nitrate and SRP indicate the possibility of anthropogenic influence in Little Hatchet Creek during moderate baseflow and storm flows although nitrate concentrations are much lower than P concentrations.

- During low and moderate baseflow, Little Hatchet Creek stops flowing at Gum Root Swamp, but during storm flows, flow and export of P to the lake increase dramatically. The low concentrations of F present in the swamp during storm flows indicates that P is being released from the wetland, thereby acting as a source of P during storm flows and a sink during low and moderate baseflow.

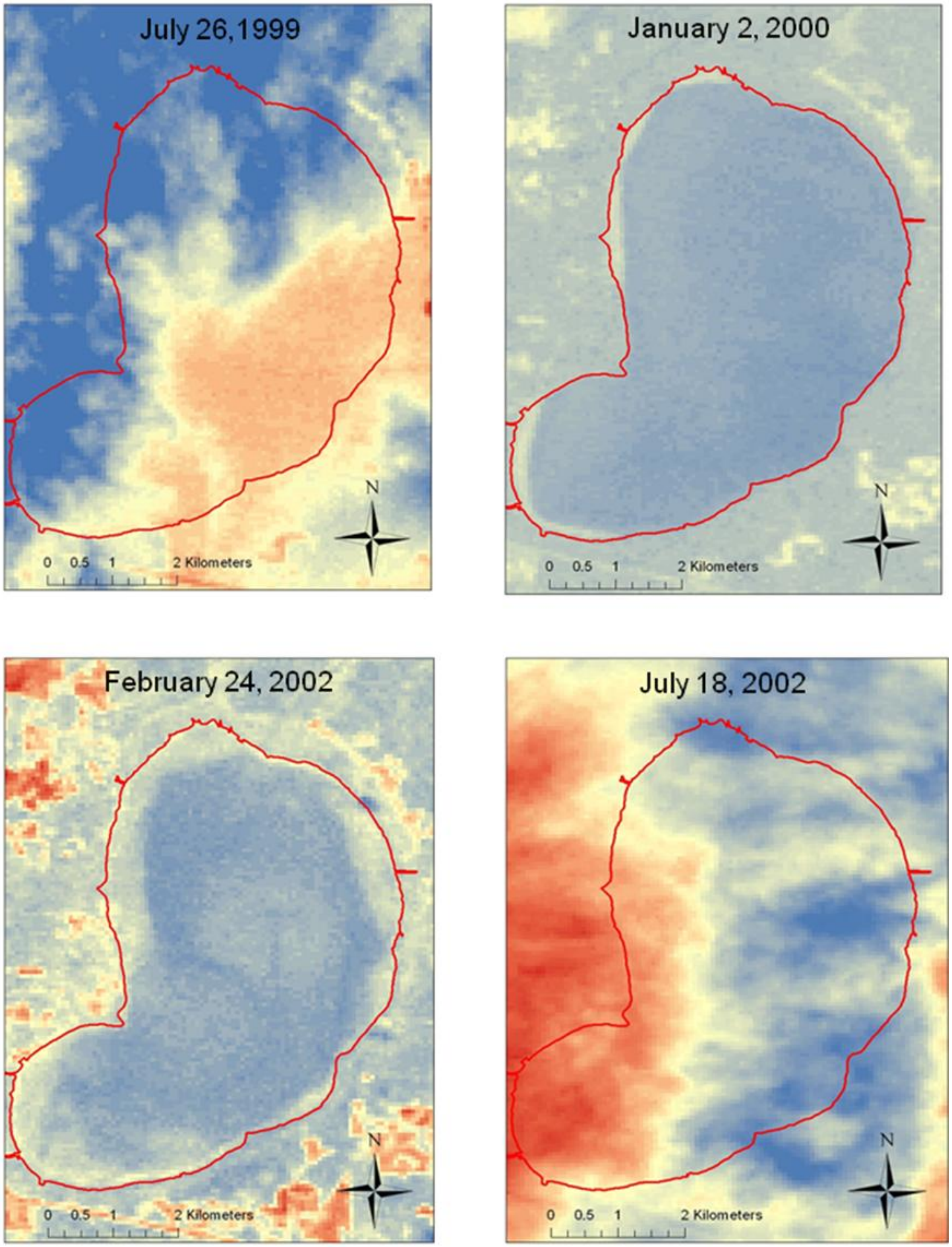


Figure 4-1. Thermal maps of Newnans Lake. Blue colors are cooler while red colors are warmer.

Figure 4-2. Diagram showing how radon, located in mineral sediments, can be advected into the lake during storm events. When lake stage is low, sediments are compacted, allowing radon to be near the sediment/water interface. As turbulence disrupts the sediment, radon is released into the water column. After the storm, the lake stage has risen and the sediment settles.

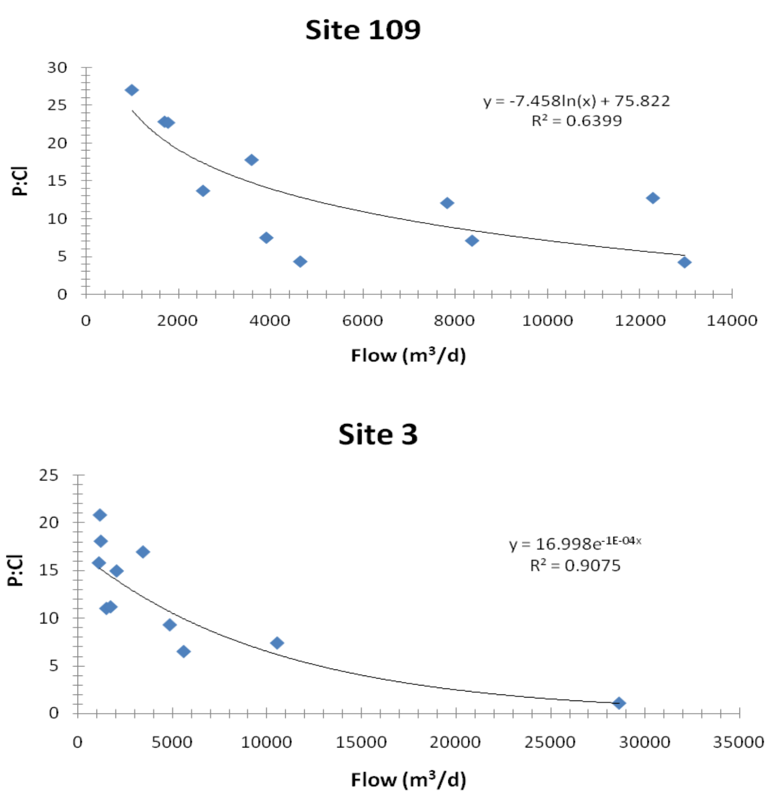


Figure 4-3. P:Cl ratio in relationship to flow over time at site 109 (Little Hatchet Creek) and site 3 (Hatchet Creek).

Table 4-1. Annual inputs and outputs for the water budget (including estimated flows from Prairie Creek Reach). *There are no evaporation data for 2008, and so it was not included.

Year	Cumulative input (m^3/yr)	Cumulative output (m^3/yr)	Average input (m^3/d)	Average output (m^3/d)	SD input (m^3/d)	SD output (m^3/d)
1999	$4.09x10^6$	$7.24x10^6$	$4.87x10^4$	$7.79x10^4$	$1.28x10^5$	$3.08x10^4$
2000	$3.41x10^7$	$4.57x10^7$	$9.31x10^4$	$1.25x10^5$	$2.66x10^5$	$5.99x10^4$
2001	$4.49x10^7$	$4.59x10^7$	$1.23x10^5$	$1.26x10^5$	$2.81x10^5$	$5.57x10^4$
2002	$1.69x10^7$	$2.72x10^7$	$8.49x10^4$	$1.37x10^5$	$2.38x10^5$	$6.84x10^4$
2003	$1.04x10^8$	$1.01x10^8$	$3.30x10^5$	$3.2x10^5$	$4.9x10^5$	$1.17x10^5$
2004	$1.28x10^8$	$1.43x10^8$	$3.51x10^5$	$3.91x10^5$	$9.57x10^5$	$4.75x10^5$
2005	$1.13x10^8$	$1.07x10^8$	$3.1x10^5$	$2.94x10^5$	$4.57x10^5$	$1.07x10^5$
2006	$5.03x10^7$	$7.66x10^7$	$1.38x10^5$	$2.10x10^5$	$3.43x10^5$	$1.34x10^5$
2007	$4.90x10^7$	$2.92x10^7$	$1.34x10^5$	$8.01x10^4$	$3.39x10^5$	$2.82x10^4$

Table 4-2. Annual inputs and outputs for the P budget (not including estimated fluxes from Prairie Creek Reach).

Year	Cumulative input (kg/yr)	Cumulative output (kg/yr)	Average input (kg/d)	Average output (kg/d)	SD input (kg/d)	SD output (kg/d)
1999	117	6	1	0.06	2	0.1
2000	813	3	2	0.007	4	0.04
2001	1093	9	3	0.02	4	0.01
2002	361	0	2	0	3	0
2003	4232	7654	13	24	14	10
2004	4390	12800	12	35	25	57
2005	4344	8428	12	23	13	7
2006	1819	4638	5	13	9	14
2007	1197	447	3	1	5	1
2008	1939	2477	8	10	13	6

Table 4-3. Hydraulic gradient (cm) for each well location during each sampling period.

Date	111	250	251	252	253	255	256	260	261	262	263	264	270	291
Oct-07	-8	-12	15	-11	3	4	8	-4	5	10	-23	0.5	-37	36
Nov-07	-5	-12	0.5	78	-2	-1	6	-0.6	-0.5	7	-23	-0.5	-27	13
Dec-07	25	-2	4	-3	-0.2	3	7	3	5	12	-3	4	-7	12
Jan-08	11	3	7	0.8	1	6	1	1	0.8	9	2	-0.5	-52	35
Feb-08	-0.2	26	2	9	-28	0.3	17	1	0.4	92	22	0.4	29	100
Mar-08	-2	-26	1	10	-23	0.3	5			69	5		0.6	
Apr-08		-4	-33	-1	-19	-2	-0.8		0.04		-25		-31	0.02
May-08	14	12	-20	-50	23	7	0.3	0.1	0.6	13	-11	0.08	2	-1
Jun-08	-6	-11	-6	-2	-23	-4	-3	0.5	-2	-9	-7	-0.2	-23	1
Jul-08	-13	-22	-40	-22	-2	-9	-2	-0.6	-5	-7	130	-2	-43	-13
Aug-08		-4	-2	36	-23	-2	10		0.4	0.8	-42		-37	
Sep-08	-4	-8	-3	-4	-27	-3	0.2		-0.1	-1	-40		-39	0.7
Oct-08	-3	-11	-36	-7	-25	-3	-3		-1	-2	-44		-38	-0.2
Nov-08	-5	-9	-3	-16	-22	-3	-8	-0.6	-3	7	-6	-2	-20	0.5
Dec-08	2	-3	0.4	1	-21	-1	1	1	-0.5	8	-3	0.3	-10	2

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BIOGRAPHICAL SKETCH

Lauren Long was born in 1985 in Fort Pierce, Florida. She resided in Stuart, Florida for the first 18 years of her life, where she spent much of her time at the beach and exploring local ecosystems. In high school, she took part in an environmental science project, investigating the cause of lesions on fish found in the St. Lucie River, and this cemented her interest in pursuing the environmental sciences. After graduating from Martin County High School in 2003, Lauren attended the University of Florida, as an undergraduate from 2003 to 2007, where she achieved a Bachelor of Science in environmental science with minors in soil and water sciences and business administration.

During the summer of 2005, Lauren completed an internship with the Arthur R. Marshall Foundation studying the Everglades ecosystem and the Comprehensive Everglades Restoration Plan. The internship solidified her interest in wetland ecosystems and her desire to attend graduate school. In 2007, she began her Master of Science program in interdisciplinary ecology, under the supervision of Matthew Cohen in the School of Forest Resources and Conservation.

Lauren received her M.S. degree from the University of Florida in the fall of 2009, and is currently pursuing a Master of Philosophy at the University of Waikato in Hamilton, New Zealand. She has received a Fulbright scholarship to fund her research on the long-term effectiveness of denitrification walls.