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SPATIAL NUTRIENT LOADING AND SOURCES OF PHOSPHORUS IN THE NEWNANS LAKE WATERSHED

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Spatial Nutrient Loading and Sources of Phosphorus in the Newnans Lake Watershed

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EXECUTIVE SUMMARY

Newnans Lake is a hyper-eutrophic shallow lake in North Florida. While natural eutrophic conditions have existed in the lake throughout the available paleolimnologic record, recent increases in total P concentrations in the lake, coupled with extensive and frequent exotic algal blooms have raised concern that anthropogenic loading is exacerbating lake conditions, leading to declines in lake ecological integrity. As part of the total maximum daily load (TMDL) developed for the lake, a target TP concentration of 50 ppb has been established; current concentrations vary between 75 and 200 ppb, and are strongly negatively correlated with stage.

To meet the TMDL will require external source reductions as well as in-lake management of stored P. This work focuses on the external loading of P (principally as soluble reactive P – SRP) and other constituents from the watershed. The central unexplained observation of this work is the discordance between lake water concentrations of P and the dominance of forests and wetlands as watershed land uses; over 75% of the total land surface is covered with these protective uses, and the expectation for a lake draining a watershed of this land use intensity is more than an order of magnitude lower TP concentrations. In addition to the comparative effect of different land cover types, this region has a rich source of geologic P in the Miocene clays of the Hawthorn Formation. The clay layer, which in this area provides an aquitard between surface water and the Floridan Aquifer, is among the richest sources of apatite ($CaPO_4$) in the world; where the Hawthorn intersects the surface, which occurs frequently in the region, significant potential for P weathering and subsequent export exists. While this geologic source does not immediately explain the recent upswing in lake water column P concentrations, determining the relative contribution of geologic P is important for load reduction planning. The overarching goal of this project was to *identify and quantify the source of P loaded from the watershed*. We partition this overarching goal into 4 objectives:

1) Determine the effect of land use on water quality
2) Contrast the land use effects with the effect of geologic sources of P on water quality
3) Quantify the load of P from diffuse groundwater sources
4) Parameterize a Newnans Lake elemental budget for P

This report summarizes a two year project that examined P loading from the basin, with over 580 samples collected from 15 core sampling sites (including the lake outflow at Prairie Creek), 60 synoptic sites, and 3 longitudinal transects along Hatchet and Little Hatchet Creeks. In addition, 14 perimeter wells were installed and monitored monthly for flow and water chemistry. The final aspect of the sampling involved collecting sediment and soil samples from which we addressed the question of land management effects (and particularly surface soil disturbance via pine plantation bedding) on load acceleration from the Hawthorn.

Based on the evidence from the samples collected, and put in the geologic, land cover, landscape position and hydrologic regime, we draw five conclusions about the sources of P from the Newnans Lake watershed:

Conclusion 1: Phosphorus loading to Newnans Lake is dominated by geologic phosphate

Lines of Evidence:

a) Land use does not appear to control P concentrations, whereas it does control N concentrations
b) NO_x and SRP concentrations are uncorrelated; we would expect a fertilizer source to load N and P together, yielding high N levels at P enriched locations.
c) P concentrations are well predicted by sampling site proximity to the Hawthorn Formation, the major source of geologic P.
d) P concentrations covary with concentrations of fluoride, a weathering by product from apatite minerals in the Hawthorn Formation.

Conclusion 2: P loading is a complex function of flow that depends strongly on the source basin.

Lines of Evidence:

e) Little Hatchet Creek exhibits strong dilution gradients with flow (i.e., flow and SRP and negatively correlated)
f) Lake Forest Creek shows no association between P and flow
g) Hatchet Creek exhibits decreasing concentrations with flow and then a reversal where concentrations increase with flow. Consequences for loads are tremendous. Sources are intermittent tributaries that drain pasture land, and are predicted to interact strongly with the Hawthorn. This line of evidence is confirmed by longitudinal transect sampling that shows maintenance of SRP concentrations with 100 fold changes in flow volume.
h) Loads of P from ungaged streams are modest (ca. 20% of load on average), but increases markedly during stormflows. Load fractions from ungaged portions of the basin may, therefore, be increasingly important during wetter climatic periods than the period of record.

Conclusion 3: Terrestrial soils do not appear to be a major source of P.

Lines of Evidence:

i) Soils (to 1 m deep) do not appear to be significantly enriched with P, particularly compared with P levels in the Hawthorn clays, even where the Hawthorn is predicted to be near the surface.
j) Soils are moderately enriched at depths of 80-100 cm, but still exhibit concentrations 2 orders of magnitude lower than Hawthorn clays.

Conclusion 4: Groundwater P appears to be of geologic origin, but groundwater appears to be a negligible source of water and P to the lake.

Lines of evidence:

k) SRP and F are strongly correlated across 14 perimeter wells.
l) Groundwater flow rates are constrained by moderate conductivities and low potentiometric gradients.
m) Flows are variably into the lake and out, both in time and space. Few obvious patterns of flow control (sign and magnitude) were evident.
n) While SRP concentrations are comparatively high in many of the wells, our estimate of annual load is negative to neutral; that is, P flow is away from the lake (0.09 kg/day).

Conclusion 5: P load reduction targeting can be optimized by accounting for landscape position and basin rather than land use alone. Successful load reduction strategies are likely to follow from hydrologic rather than P loading controls (e.g., stormwater management). Where load reduction (i.e., numerator of equilibrium P concentration equations) is not possible, efforts to regulate depth, sedimentation rates and export rates (i.e., denominator of equilibrium P concentration equations) may prove more useful.

Lines of evidence:

o) Headwater areas have significantly lower P concentrations across all basins.
p) Mid-reach areas have significantly elevated P concentrations in Little Hatchet and Hatchet Creeks; difference in Lake Forest Creek were in the same direction but not significant.
q) Terminal wetland areas (data were available only for Lake Forest and Little Hatchet creeks) exhibit lower concentrations than mid-reach sites in the same basin, which may be evidence of some attenuation potential for both SRP and NO_x.
r) Given strong associations between lake stage and TP, management of stage may provide valuable concentration reductions that both limit phytoplankton concentrations and also reduced downstream N loading (since a significant proportion of the N in the lake is internally fixed (Fig. 73).
s) Sink enhancements (shad harvest, alum treatments, sediment removal, etc.) may prove valuable in internal load reduction, and should be considered in light of the potential inability to reduce external loads via watershed management practices.

I. INTRODUCTION/BACKGROUND

Newnans Lake and creek systems that drain into the lake fail to meet state water quality criteria and are listed as impaired by FDEP (Florida Dept. of Environmental Protection). To restore water quality and ecological function in the lake, the St. Johns River Water Management District (SJRWMD) is charged with defining Pollutant Load Reduction Goals (PLRGs) which will guide watershed management. Nutrient loading to Newnans Lake is the result of complex interactions between watershed processes (land use, hydrologic response) and in-lake dynamics (water level, biological perturbation, wind-driven sediment entrainment, biological N fixation). Most previous research has focused on in-lake processes, and on burial/resuspension/lability of nutrients, effects of hydrologic fluctuation and on ecological consequences of both these drivers. To complement these previous efforts, this study focuses on watershed processes by examining the spatial and temporal distribution of nutrient loading to the lake. Our primary objective is to quantify exogenous sources of nutrients, characterize the spatial distribution of loads, delineate cultural from geological enrichment, and better understand pathways for nutrients fluxes to the lake. This objective will complement and refine existing efforts to quantity PLRGs for the watershed, and will develop a protocol for application to other lakes in the Orange Creek Basin (OCB) and other similar watersheds.

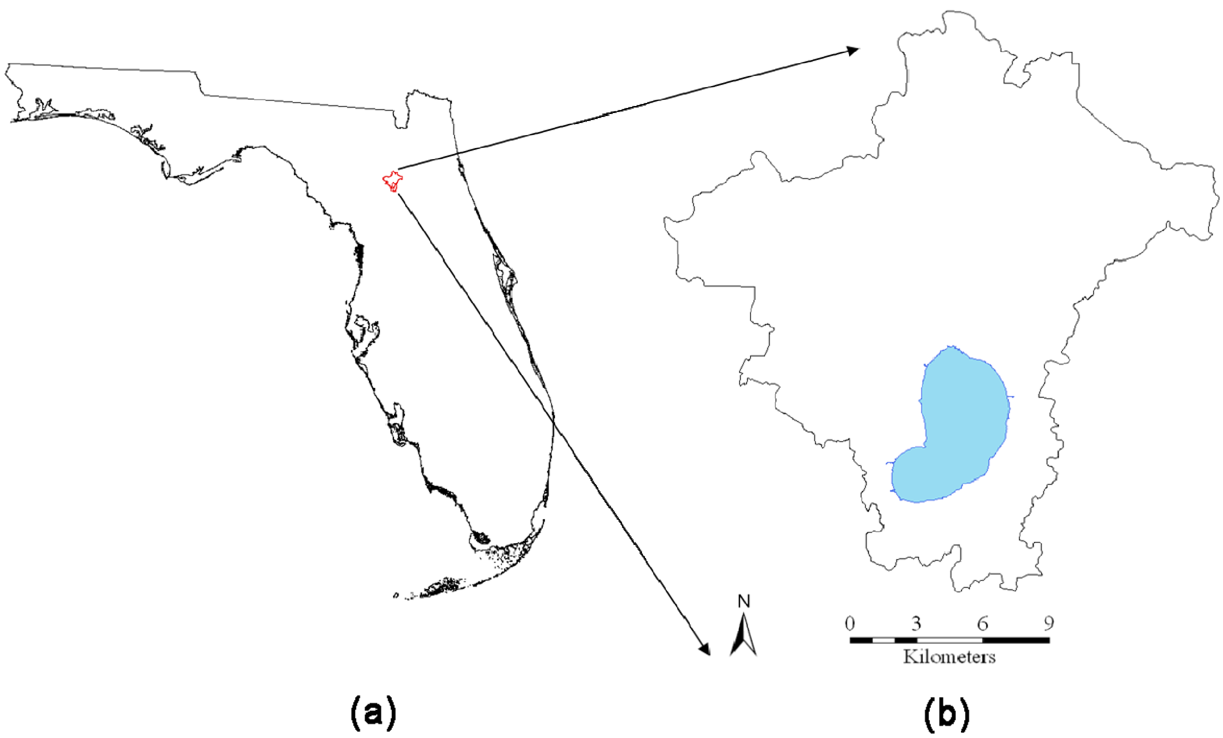

Fig 1. The Newnans Lake Watershed. Fig (a) location of the Newnans Lake watershed in north-central Florida, and (b) the Newnan's Lake and its watershed boundary.

Newnans Lake is a large (area - 27 km^2), shallow (mean depth – 1.6 m) hyper-eutrophic lake 8 km east of Gainesville in north-central Florida (Fig 1). It is located in the Northern Peninsula Plains of the Ocala Uplift District (Brooks 1981), and overlies phosphatic sands and siliciclastic clays of the Hawthorne Formation. Interaction between surface water and the Hawthorn is one largely unexplored potential cause of naturally high levels of phosphates. Paleolimnological evidence, summarized in Brenner and Whitmore (1998) suggests that the lake formed recently (5000-8000 ybp) and has had elevated levels of P throughout its history, and in particular prior to 1900. Despite evidence supporting significant natural sources of P in the lake, there is also evidence that recent (since 1900) changes in the biology and water quality of the lake are the result of development in the watershed. In particular sedimentation rates of C, N and refractory organic P have increased dramatically since 1900. Brenner and Whitmore (1998) conclude from diatom-based inference that the lake may be at least periodically N-limited, and observation consistent with the recent growth of *Cylindrospermopsis raciborskii*, a N-fixing cyanobacterium, as a dominant component of the phytoplankton community.

The Newnans Lake Watershed (NLW) has four major tributaries (Fig. 2): Hatchet Creek, Little Hatchet Creek, and Bee Tree Creek, located on the northern side of the lake, and Lake Forest Creek, located on the west. Hatchet has the largest drainage area (72.0 km^2) followed by Bee Tree (65.9 km^2), Lake Forest (20.3 km^2), and Little Hatchet (17.5 km^2). Bee Tree merges with Hatchet Creek, and they collectively drain into the lake as Hatchet Creek. Little Hatchet drains into Gumroot Swamp, which drains to the lake via two streams, both called Little Hatchet Creek. The Gainesville Regional airport catchment is a tributary of Little Hatchet, with a drainage area of 5.2 km^2. Lake Forest Creek is the most urban sub-basin, draining much of East Gainesville. Several intermittent streams drain the Prairie Creek Reach (Fig. 2). These drain a large blueberry farm to the northeast, a state prison to the northwest, and pine silviculture to the southeast. Newnans Lake has one surface outlet at Prairie Creek located at the south side.

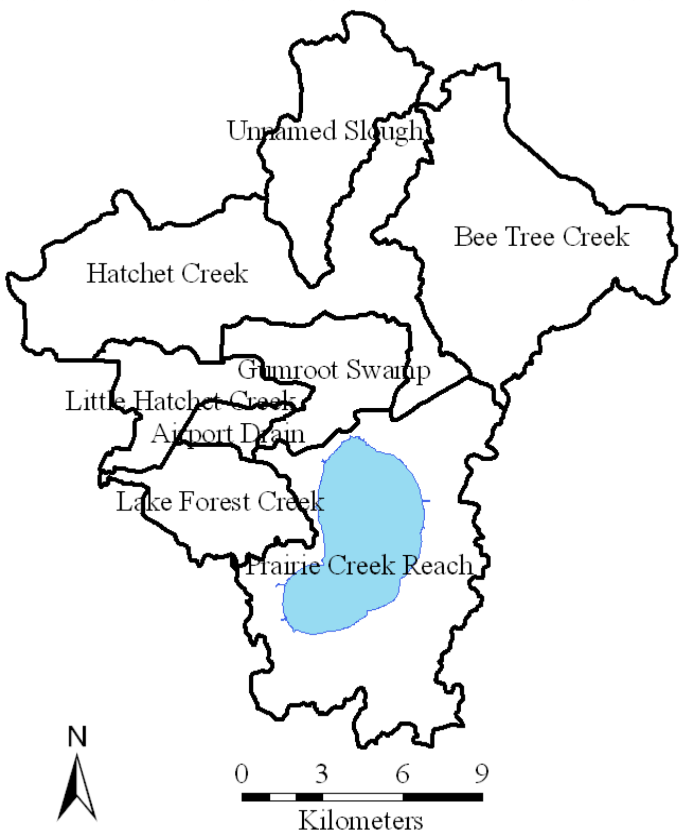

Fig 2. Major sub-basins in the Newnans Lake watershed.

While there is some evidence to suggest that internal load dominates modern nutrient dynamics (Nagid et al. 2001), recent specific sediment entrainment studies (Gowland and Mehta 2002) suggest that sediment entrainment may be lower than previously thought. Moreover, nutrient and water budgets suggest that the watershed continues to contribute significant load to the lake; spatial patterns and the role of storm events in loading are poorly understood. The objective of this work is to complement existing understanding of in lake processes with a more detailed examination of watershed controls on nutrient loading.

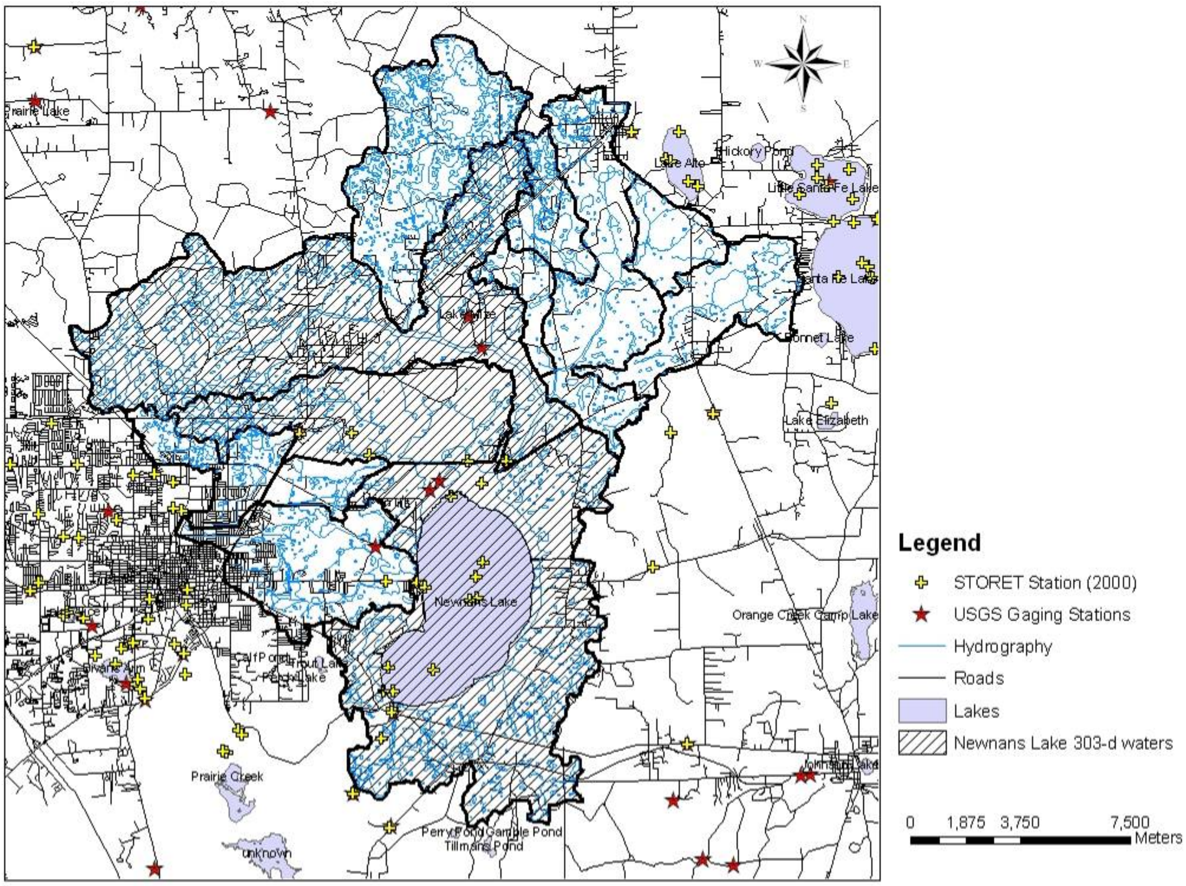

Fig. 3 – Newnans Lake watershed showing sub-basin boundaries, hydrographic network, regional road network, existing sampling stations (USGS sites for water quantity, STORET stations for water quality). Sub-basins failing to achieve water quality for their designated use under 303(d) of Clean Water Act are also shown.

Watershed load mapping is typically done using fixed stations at monthly frequency (e.g., STORET and USGS/SJRWMD stations shown in Fig. 3). A small but growing database of water quality data for the Newnans Lake watershed exists to complement data on hydrologic fluxes. This flow and water quality data along with a suite of ancillary thematic layers (elevation, land use, geology) allow construction of provisional water and nutrient budgets for the lake, and guide selection of sites for more detailed examination of water flow and chemistry over time and space. Moreover, they provide a benchmark for future assessment of progress towards pollutant load reduction goals on a basin-by-basin basis.

While this sparse station model for watershed assessment is widespread, there are several critical shortcomings when trying to understand watershed loading and ultimately target restoration efforts. First and foremost, the resolution of the observations are often insufficient for spatial targeting of interventions. As an example, NO_x fluxes in Florida's rivers vary substantially in time and space; the characteristic dynamics of this important water quality constituent are both more rapid in time (Cohen et al. 2007) and more spatially variable (Lamsal et al. 2007) than typical sampling regimes permit. Second, sparse stations frequently constrain construction of basin element budgets. A typical approach is to quantify loads from only principal drainage pathways (e.g., Little Hatchet and Hatchet creeks), neglecting both smaller, frequently intermittent loads (e.g., those draining the Prairie Creek Reach (Fig. 2) and diffuse inputs from groundwater. Nagid et al. (2001) adopt this approach, ultimately concluding that Newnans Lake exports some 280% more phosphorus (P) than is loaded by the watershed, and concluding that management of lake sediments rather than the watershed is of paramount importance. In some cases, models are used to up-scale data from a small number of stations for whole basin elemental budgeting. However, where the relationship between land use and nutrient loading is uncertain, and further where the hydrogeologic and runoff generating properties of the basin are unstudied, even these modeling efforts can be grievously inadequate for load estimation. As the TMDL process ensues, land uses in the watershed will be variably responsible for attenuating loading to meet TMDL goals. Establishing relationships between land use and loading is essential for that process. A distributed spatial approach that maximizes the diversity of contributing areas from which to draw inference about the role of land cover on pollutant loading is critically important in a watershed where over 75% of the area is under cover types typically considered protective of regional water quality (i.e., wetlands, conserved and planted forests). Identifying loading hot-spots so that the appropriate remedial action can be taken is an expected outcome of this work.

To accomplish this, there is a need for an orthogonal approach to watershed scale assessment. First, improve spatial resolution by synoptic sampling at locations pre-defined to help better understand the roles of land use, terrain and geology on water quality. This includes improving our understanding of longitudinal changes in water quality and quantity that can be used to focus attention on areas where loads or concentrations are changing with distance downstream. Second, maintain a regular sampling protocol at sites that help better delineate loads to the lake including in particular those locations not commonly part of sparse water quality networks. Both approaches, plus a comprehensive effort to quantify groundwater loading, were adopted in this two year project. Based on previous work in the watershed, information regarding the drivers of ecological change in the lake, and designation of impairment (303-d – Fig. 3), we focus our attention on phosphorus, nitrogen, iron and a suite of standard water and sediment quality analytes (pH, conductivity, total carbon, chloride) and those that actively interact with P in mineral form (Al, Ca, F).

Given an overarching objective of enumerating the sources and magnitudes of pollutant loading (mainly P) to Newnans Lake, this research was partitioned into 5 hypotheses and predictions that follow from them.

__Question 1__: That the NLW is predominantly forested makes the fact that the lake is hyper-eutrophic difficult to explain; nominal water column TP concentrations of between 75 and 225 ppb are enormously high for a watershed that is over 75% forested. The prevailing paradigm in water quality management is that changes in landscape loading are principally the result of changes in land use; more intensive land uses lead to increased water quality degradation, at least where remedial actions are not taken. As such, there remains significant uncertainty about the role of land use in P loading. Establishing local relationships is of paramount importance for watershed remedial planning.

H1 – Pollutant concentrations vary positively with land use intensity in the contributing area.

Prediction 1 – Positive correlation between pollutant concentrations and landscape development intensity (LDI) index (Brown and Vivas 2005) will be observed.
Prediction 2 – The fraction of organic nutrient species (dissolved organic P, dissolved organic N) will decrease with LDI. Humans increase the load principally of mineral species, and reduce the landscape residence times of water that facilitate export of organic species.
Prediction 3 – Export of DOC in baseflow will decline with LDI. As the nominal residence time of water in the landscape declines with increasing impervious surface, less DOC enrichment is expected
Prediction 4 – Specific conductance of baseflow water draining high LDI (intensive) regions will be higher than from low LDI regions due to the combined influence of fertilizers, diffuse surface discharge of groundwater (high conductance) from urban and agricultural irrigation uses, and increased evaporative salt concentration.

__Question 2:__ A major geologic source of P, the Hawthorn Formation, elsewhere mined for fertilizer, exists within the region. Weathering/erosion of this material could reasonably be invoked to explain at least part of the high P load to the lake. As such, a second question in this work is the extent to which geologic P (derived from the Hawthorn Formation) contributes to watershed-scale P loading. Several geologic and fertilizer covariates are used to trace sources.

H2 – SRP delivered to the lake is principally of geologic origin.

Prediction 1 – Relationships between SRP concentration and land use will be weak. If the source of P were anthropogenic activities, SRP would be strongly correlated with LDI.
Prediction 2 – Correlation between SRP and NO_x will be absent. If the source of P were fertilizer, these two nutrients would be expected to co-vary. N:P molar ratios will be lower than typical in areas with geologic P loads.
Prediction 3 –Hawthorn depth will be a strong predictor of SRP concentration. If most of the P were of anthropogenic origin, depth to geologic layers enriched with apatite would provide little predictive information regarding surface water SRP concentrations.
Prediction 4 – Simultaneous release of F and SRP during fluorapatite weathering will result in strong correlations between these solutes where geologic P is the principal source. Moreover, F concentrations (which are also loaded from municipal water supplies) will co-vary only where the Hawthorn is proximate to the land surface.

Question 3:, There is a need to quantify to role of aquifer loading to stream, and subsequently wetlands in ameliorating those loads. The intermediate aquifer in the Newnans Lake region is actually a set of intercalated aquifers residing within sand lenses that are part of the Hawthorn Formation. We predict that these aquifers are important for local P loading (e.g., along stream margins) and, as such, expect that longitudinal profiles of stream P concentrations will reflect the role of aquifer seepage. It is also well established that wetlands can provide valuable watershed-scale services such as hydrologic storage and P attenuation. By selecting sampling locations along streams we can begin to quantify areas of pronounced biogeochemical activity (e.g., dramatic changes in concentrations) in both directions.

H3A – Longitudinal P concentrations (and covariates) are controlled by the relative contribution of water from the intermediate aquifer (via lateral seepage), which has high P concentrations, and overland flow and headwater wetland drainage, which has low P.

Prediction 1 – P will increase markedly with distance downstream during baseflow, moderately during intermediate flow, and decrease during stormflow.
Prediction 2 – When P increases due to lateral groundwater seepage, F concentrations will covary, indicating that the P is of geologic origin.
Prediction 3 – During periods of higher flow, when non-seepage flowpaths dominate, longitudinal profiles in P and F will be muted (medium flow) or reversed (high flow).

H3B – Wetlands provide a significant sink for N and P in the Newnans Lake watershed.

Prediction 1 – Concentrations of mineral N (dissolved inorganic N, DIN = NO_x + NH_4) and P will decline consistently during passage through Gumroot Swamp.
Prediction 2 – Loads of total P will decline during passage through Gumroot Swamp during baseflow, but increase during stormflow, with large particulate organic P export.
Prediction 3 – P export from the swamp after drying events will exceed inputs as mineralized P is exported.

Question 4: One primary uncertainty that cannot be addressed by sparse spatial sampling networks is the role of surficial or intermediate groundwater fluxes directly to the lake and P loading that is associated with that flow. Rough element and water budgets suggest that measured inputs of both to the lake are lower than measured outputs; the unquantified source may be ungaged surface flows or groundwater. Direct groundwater inputs can be evaluated using perimeter wells; ungaged surface inputs are evaluated as part of regular monitoring.

H4 –Diffuse loading of P-enriched groundwater via surficial/intermediate aquifer flowpaths directly to the lake is responsible for a significant fraction of the lake P and water budget.

Prediction 1 –Wells near the lake fringe but distal from septic tanks will be enriched in SRP/TP when lake and groundwater levels are low. They will also be depleted in N, and enriched in Ca and F.

Prediction 2 – Wells near the lake fringe and proximate to septic tanks will be enriched in P, and also N.

Prediction 3 – Potentiometric gradients are positive and hydraulic conductivities are high in the sediments near the lake, so net inflow of water is large.

Prediction 4 – This process will be observed to be most important for water column TP concentrations during periods of low lake stage, when potentiometric gradients are positive towards the lake, and contact with P from geologic (Hawthorn) and anthropogenic (septic tanks) sources is maximized.

Question 5: Among the key unknowns is whether a) human activities are responsible for a significant fraction of the load (see H2) and b) if not, that is if most of the P load is of geologic origin, whether human activities have markedly increased the rate of geologic P mobilization. Clearly runoff patterns are influenced by human activities, with important implications on the interaction between surface water and the Hawthorn Formation. Moreover, this area has been used for industrial timber production for the last 75 years, and there is concern that bedding activities (which disturb the upper 20-30 cm of the soil profile to generate microtopographic relief that maximizes seedling survival) may have reworked the soil profile to entrain P-rich Hawthorn clays into regions of more active weathering. This soil profile inversion and subsequent accelerated weathering would increase geologic P loads; evidence should be observed in soil P concentrations, both over space (according to Hawthorn depth) and depth.

H5 –Surface soils in the NLW are not the source of P loading to creeks.

Prediction 1 –Surface soils (0-20 cm) are not enriched with P even in areas where the Hawthorn is close to the land surface.

Prediction 2 – Soils at depth (80-100 cm) will be variably enriched

Prediction 3 – Potentiometric gradients are positive and hydraulic conductivities are high in the sediments near the lake, so net inflow of water is large.

Prediction 4 – This process will be observed to be most important for water column TP concentrations during periods of low lake stage, when potentiometric gradients are positive towards the lake, and contact with P from geologic (Hawthorn) and anthropogenic (septic tanks) sources is maximized.

This report describes data collected during 2007 and 2008 targeting each of these predictions, and the inference that is drawn from evaluating evidence in support of each hypothesis.

II. METHODS

Site Selection

Site selection is a critical part of spatial survey design; a survey can only be effective if sites sampled adequately represent the range of sites that are found in the watershed. To meet this objective, we selected key variables that constitute ordinate axes for site selection. These included land use (our principal target), elevation, landscape position (headwater, mid reach, wetlands), soil type, proximity to wetlands, proximity to existing infrastructure/sampling locations, and areas of special interest (e.g., wastewater discharge sites, airport drains, sinkholes). In addition, the practical constraints of access were considered. Finally, we sought to maximize the information obtained from river confluences; specifically, a qualitative algorithm for site selection was developed wherein a given conveyance (e.g., Hatchet Creek) was sampled above and below major confluences.

A GIS database containing spatial layers (elevation, roads, hydrography, wetlands, soils, 2004 land cover, existing sample sites) was assembled. Of particular interest was land use; the entire watershed and each of the major tributaries are summarized by their respective land use in Tables 1 (whole watershed) and 2 (sub-basins). Figure 4 shows the spatial distribution of land cover types. Land use distribution across the NLW was derived from 2004 land use map developed by the St. Johns River Water Management District. As a small portion (2.4%) in the north-west part of the NLW lies in the Suwannee River Water Management District, a 2001 land use map of Suwannee River Water Management District was used to map land use in this portion of the watershed.

Upland forest (51.7%) and wetland (22.8%) are the major land use categories in the watershed. Upland forests are dominantly distributed across the watershed, while wetlands are restricted primarily to the area adjacent to the lake and along the creeks that drain to the lake. Urban areas are concentrated in the west side of the lake, while agricultural areas are found in patches across the watershed. Gainesville Regional Airport is the major utility facility.

Table 1: Extent of land use distribution in the Newnans Lake watershed.

Land use category	Extent (ha)	Extent (%)
Urban	2,454	7.7%
Agriculture	1,765	5.6%
Upland Non-Forest	1,018	3.2%
Upland Forest	16,453	51.7%
Water	2,125	6.7%
Wetland	7,259	22.8%
Barren Land	13	<1%
Utilities	707	2.2%

Table 2. *Total area (in sq km) under major sub-basins and the extent (%) of land use distribution in the sub-basins.*

	Hatchet Creek ($72\ km^2$)	Bee Tree Creek ($66\ km^2$)	Gum Root Swamp ($22\ km^2$)	Lake Forest Creek ($20\ km^2$)	Little Hatchet Creek ($18\ km^2$)	Airport Drain ($5\ km^2$)
Urban	5%	4%	6%	38%	27%	19%
Agriculture	6%	5%	2%	7%	-	-
Pasture	2%	7%	<1%	3%	4%	2%
Upland forest	62%	66%	60%	29%	40%	38%
Water	<1%	-	<1%	<1%	<1%	1%
Wetland	25%	17%	29%	21%	15%	7%
Barren land	-	-	-	<1%	-	-
Utilities	1%	1%	2%	2%	14%	33%

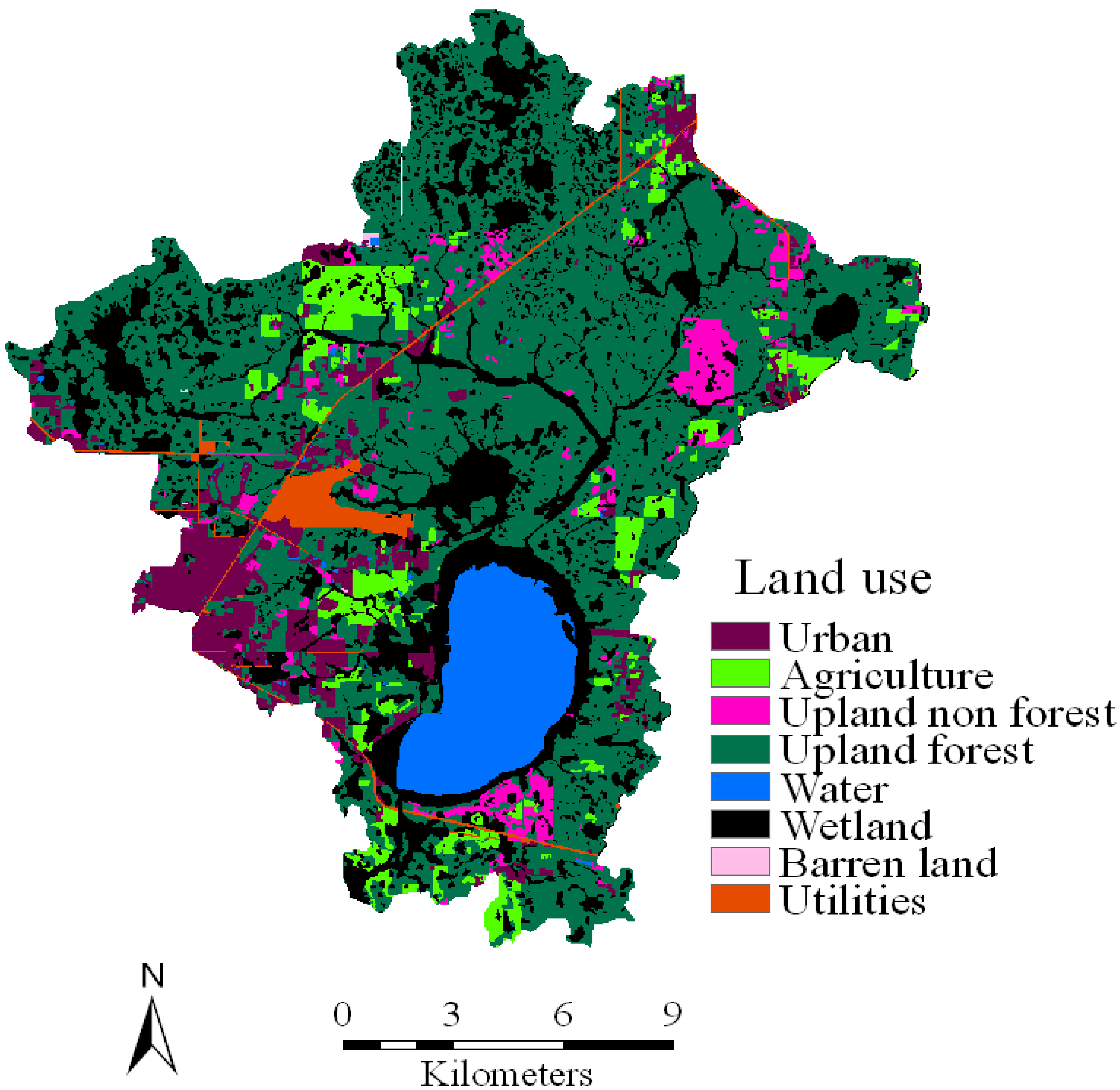

Fig. 4. Land cover in the Newnans Lake Watershed (ca. 2004).

Overall, the watershed is forested, and land use types that would be expected to degrade water quality are largely constrained to particular sub-watersheds (e.g., urban in Lake Forest creek, utilities in Little Hatchet/Airport Drain). The general expectation of a watershed with this level of forest cover is high water quality; Omernik (1977) reports the expected TN and TP concentrations in watersheds in the Eastern US varies as a function of forest coverage. The Newnans Lake watershed (with ~75% forested/natural land) is expected based on that model, to have mean baseflow concentrations of 15 ppb for TP and 719 ppb for TN. In general, export of N and P is expected to be in dissolved organic form; with increasing development, P and N are both expected to increase, and the fraction of nutrient delivered in organic form is expected to decrease (i.e., increased SRP, NO_x and NH_4 loads).

The algorithm for site selection includes more than just land use. We first selected over 150 sites at random based on the intensity of land use (using the Landscape Development Intensity index) in the contributing area (defined based on LIDAR digital elevation model for Alachua County), and then used field visits to ascertain access, ground-truth watershed development intensity and determine proximity to wetlands. Brown and Vivas (2005) describe the LDI; in short, LDI provides a score between 1 and 10, with higher numbers indicative of greater land use intensity. A typical table of LDI coefficients applied to each pixel in a raster land use map is given in Table 3. A weighted average LDI for each sampling site's contributing area makes an *a priori* prediction of the expected water quality.

Table 3: Summary of Landscape Development Coefficients (LDC) for calculation of the Landscape Development Intensity index (LDI).

Landscape Development Coefficient*	Landuse Category
1	Natural System/Open Water/Wetlands
1.5 - 2	Silviculture Operations
2 - 3	Rangeland
3 - 4	Low Intensity Pasture
4 - 5	Low Intensity Row Crops
5 - 6	High Intensity Pasture
5 - 6	High Intensity Row Crops
6 - 7	High Intensity Agriculture
7 - 8	Residential
8 - 9	Industrial and Transportation
9 - 10	High Intensity Commercial

Within each category, site-specific considerations (stocking rates, rotation times, recent changes in land-use) are used where scoring flexibility is inherent and where information is available.
* The LDC scores presented here are from Brown and Vivas (2005).

The particular land use distribution in the Newnans Lake watershed offers a unique test of the LDI concept for making predictions about water quality. Particular land uses are largely constrained by sub-watershed, and there is relatively little variability within particular sub-watersheds. Fig. 4 illustrates the land use distribution, and highlights how urban and agricultural impacts are localized within particular regions. Urban impacts are principally found in the western watersheds (Lake Forest and Little Hatchet), while industrial impacts (e.g., airport, municipal wastewater disposal) are concentrated in branches of Little Hatchet Creek. Agricultural impacts in the Hatchet Creek sub-watershed are principally pasture; the main crop land in the watershed is a blueberry farm drained by several small creeks discharging to the northeastern lake. These creeks were historically intermittent, but are more perennial now with pumping and runoff of irrigation water. In short, particular land uses are found in blocks with relatively independent drainage systems, so the role of land use on water quality can be effectively studied.

LDI values were assigned to each pixel in the raster land use map; the mean upstream land use intensity for each point was then computed based on lidar-derived contributing areas. The range of values for the 150 potential sites in the Newnans Lake watershed were between 1.05 (all upstream land conserved forest and wetland) and 6.83; while the highest weighted mean LDI was in the range of high intensity row crops, this site was in an urban setting, with a mixture of urban open space and medium density residential/commercial land uses. The LDI scores of

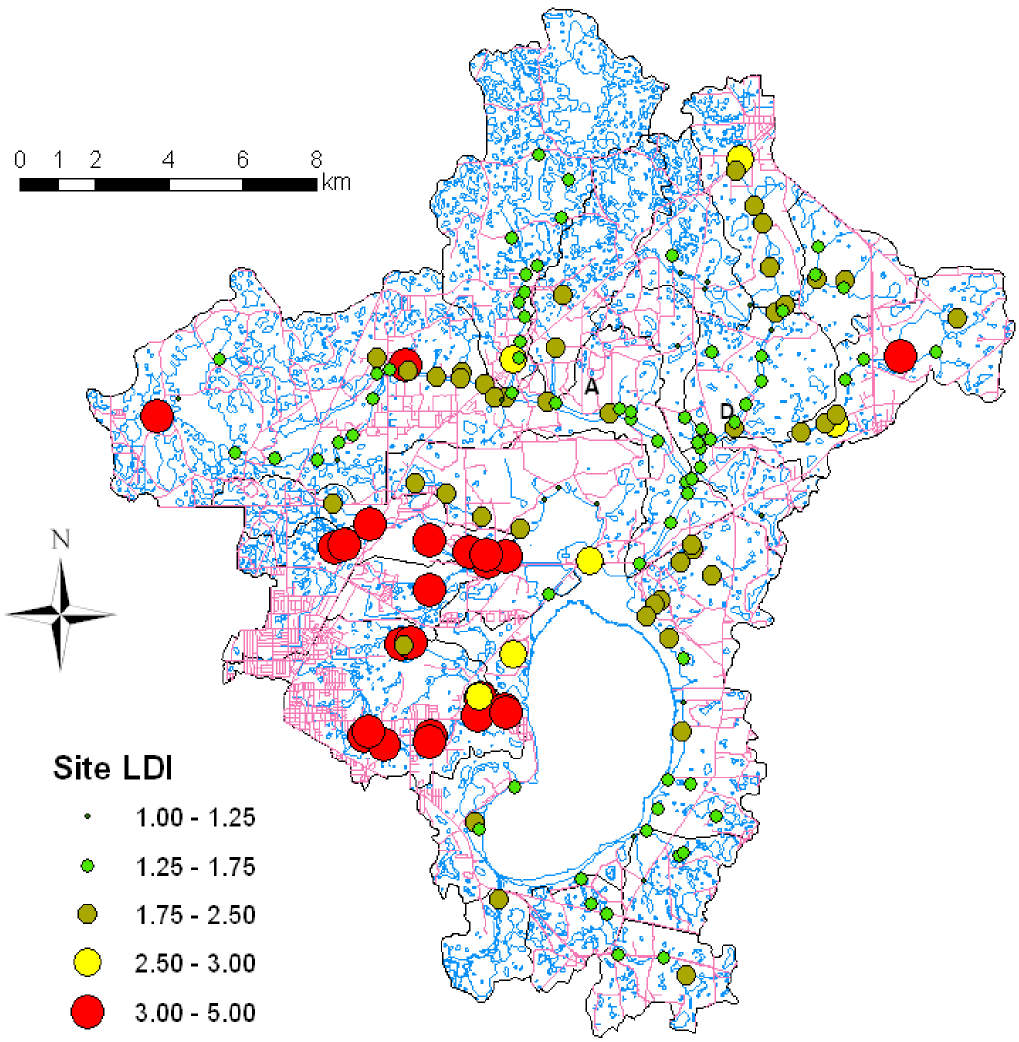

Fig. 5 – Site-level LDI for each of ~150 potential sites. High LDI indicates strong anthropogenic effects upstream; associated water quality is expected to be poor.

potential sites was strongly spatially patterned, as expected in this area (Fig. 5). In particular, the high LDI scores were observed in the western watershed, where the influence of east Gainesville (Lake Forest creek) and the airport drainage are most pronounced. Note that the sites presented in Fig. 5 were not all selected for further analysis.

One of the spatial layers used to select sites was proximity to existing water quality and quantity stations. All of the relevant time series data for water quality and flow were assembled from the existing monitoring sites (Fig. 6) for purposes of validation of our observations, and for construction of approximate water and nutrient budgets for the lake. Sites were selected to correspond with existing infrastructure to the maximum extent possible.

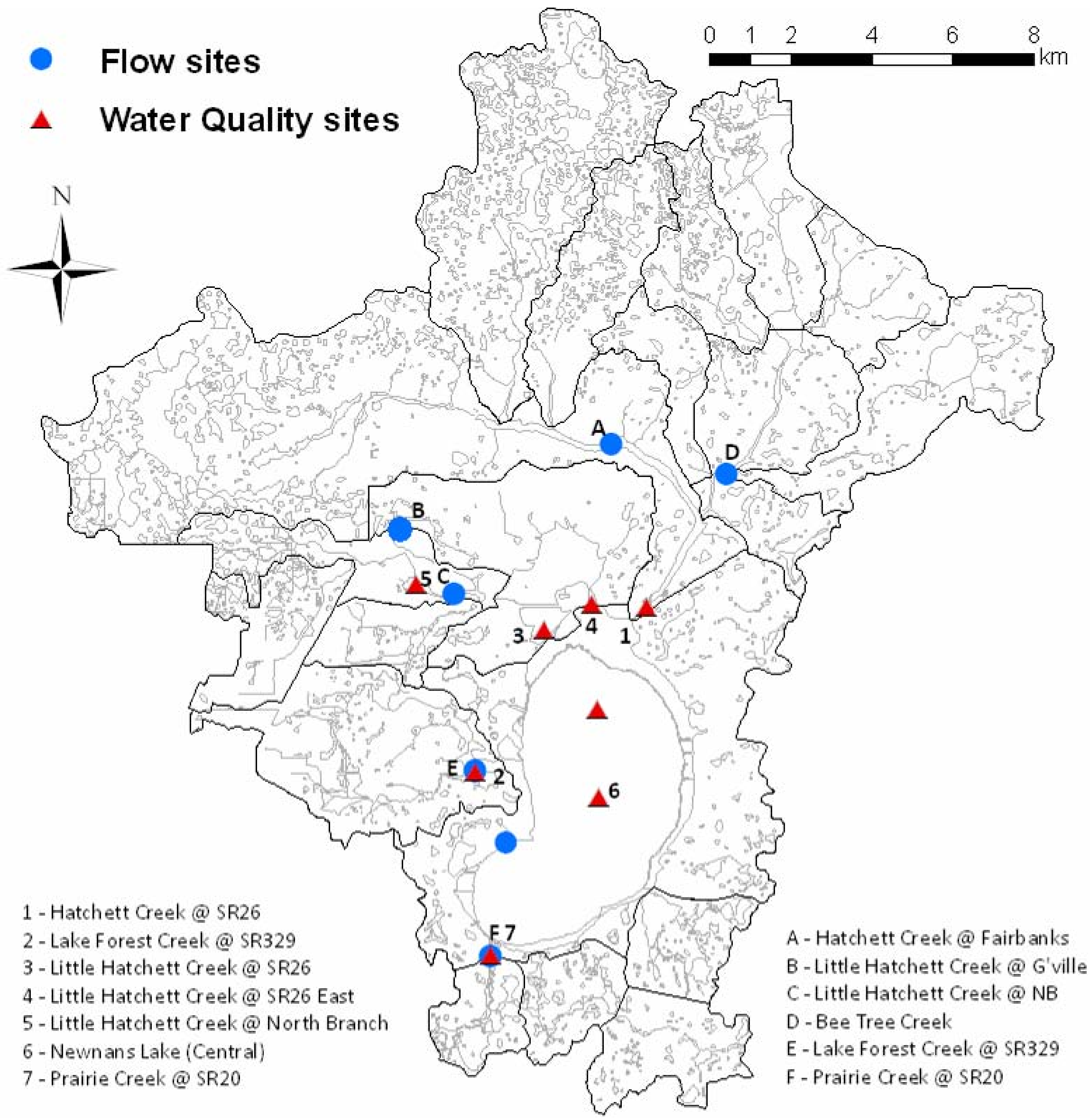

Fig. 6 – Locations of existing monitoring infrastructure in Newnans Lake watershed. Shown are long term gauges for flow and water quality. These data were used to validate the observations made in this work, and to provide a first-order water and nutrient budget for the lake.

Historical Data Analysis

Historical water quality analysis was done using water quality and flow data collected and assembled by the St. Johns River Water Management District. Data were collected from 1994 to present at Little Hatchet Creek at State Road 26 (LHAT26), Hatchet Creek at SR 26 (HAT26), Lake Forest Creek at State Road 329 (LFC329), Little Hatchet Creek at State Road 26 East (LHT26E), Little Hatchet Creek at North Branch (LHCNB), and Prairie Creek at State Road 20 (PC20). Water quality at these stations included physical (color, stage, turbidity, temperature) and dissolved/total chemical properties (hardness, metals, nutrients, salts, pigments, sediment). Our analysis focused on nutrients and salts.

Water quality measurements (concentrations) were coupled to daily flow rates where possible. In some cases, this necessitated concatenating observations at different stations. For example, water quality at station 5 (LHATNB – Fig. 6) was coupled to flows observed at station C. Similarly, flows are station A and B are linked to water quality at stations 1 and 4, respectively. While this introduces some potentially significant error (particularly in light of the spatial variability subsequently observed in observed water quality), it is a simple and reasonable way to develop watershed flow and nutrient budgets (process modeling modified to fit local hydrologic and geochemical conditions is another).

A watershed-scale water budget was constructed for two periods of record during which flows were available for all major tributary system (Feb. 03 – Apr. 06 and June 1995 – Aug. 1998). Daily inflows and outflows were tabulated, and direct rainfall inputs to and ET losses from the lake were computed based on information from Gainesville Regional Airport. Estimates of net direct rainfall are approximations principally to establish the extent to which discrepancies in inflows vs. outflows can be explained by direct rainfall. Cumulative differences between inflows and outflows were computed for both periods to determine the magnitude of water imbalance.

A similar process was undertaken to compute a P budget for the watershed. We used the period between June 1995 and April 2006 for comparison; flows and water quality availability were maximized for this period, but were not a complete record. In particular, inflow estimates were available for Lake Forest Creek only in the last 2 years of this period; during this period, this watershed was responsible for less than 5% of the total load. Hatchet Creek represents nearly 52% of the measured inflow load, followed by Bee Tree (18%) and Little Hatchet (13%).

Water quality (measured monthly) was assumed to apply to all flows during the relevant month, an assumption that introduces substantial error in light of temporal variability observed for particular stations. Total P inflows were partitioned by sub-watershed (Lake Forest, Bee Tree, Little Hatchet and Hatchet creeks) and compared to total P outflows at Prairie Creek. The cumulative difference (inflow – outflow) was computed over the period of record to determine the direction and magnitude of P budget uncertainties. For both water and P budgets, the magnitude of cumulative differences was compared with lake stores (water volume, P mass in the water column) to put those differences in a comparative perspective.

Site Sampling Protocols - Water

The preliminary set of 150 candidate sites were reduced to 75 sites after field visits during January 2007 to evaluate access, accuracy and representation. Sites were geo-registered using a differential GPS system (Earthmate BlueLogger, DeLorme). Hydrographic verification was performed in the field using the National Hydrography Dataset (NHD) and the GPS; concordance between observed flow conveyances and NHD "blue lines" was verified for all tributary samples. Only sites that were concordant with the NHD were selected to avoid confusion about contributing watershed characterization.

In addition to selecting 75 sites for quarterly sampling (Fig. 7) based on sub-catchments of general exploratory interest (Fig. 8, Table 4), we selected 15 "core" sites for regular (monthly) sampling. The original scope of work was to sample all sites quarterly; over the course of the 2-year project, most of the watershed has been dry. As such, we have focused sampling efforts where water was flowing, increasing the temporal density at some sites, and neglecting stagnant sites completely. On several occasions (June 2007, January 2008, March 2008, July 2008) we sampled the core sites on two or three consecutive days following large rain events for stormflow characterization.

When water samples were collected, two 500 mL aliquots of water were obtained from mid-depth in acid-washed Nalgene sample bottles. Samples were kept on ice at 4 °C until analysis. A multiparameter sonde (YSI600QS) was used to measure pH, temperature, dissolved oxygen, sediment ORP and specific conductance. Flow was measured at each site using a Sontek acoustic Doppler FlowTracker velocity meter; velocity was measured at 3 locations across each creek at 0.6*depth. Volumetric discharge was computed as the sum of the Depth*Velocity for each of the segments. The Sontek instrument was effective until water depths were below 20 cm. At shallower depths, we used timed floats to determine surface velocity at three locations across each stream and estimated the mean channel velocity as 0.8*surface velocity.

Water samples were partitioned into filtered and unfiltered fractions in the lab. Unfiltered fractions were analyzed for total Kjeldahl N, total P and total organic C. Filtered fractions were again partitioned, with one portion acidified (to pH 2.0 using 36N H_2SO_4), and the other portion unacidified. The acidified fraction was analyzed for soluble reactive P (within 48 hours of sampling), NO_x, dissolved organic C, dissolved ammonium, and total Al, Fe and Ca. The unacidified fraction was analyzed for major anion chemistry (F^-, Cl^-, SO_4^-, NO_3^-, PO_4^-) using a Dionex DX-500 ion chromatograph, and dissolved carbon (dissolved inorganic carbon is interpreted as the difference between DOC on acidified and unacidified samples). Analyses were run at the SFRC Forest Water Resources Laboratory, the UF/IFAS Analytical Research Lab and the Wetland Biogeochemistry Lab following standard protocols.

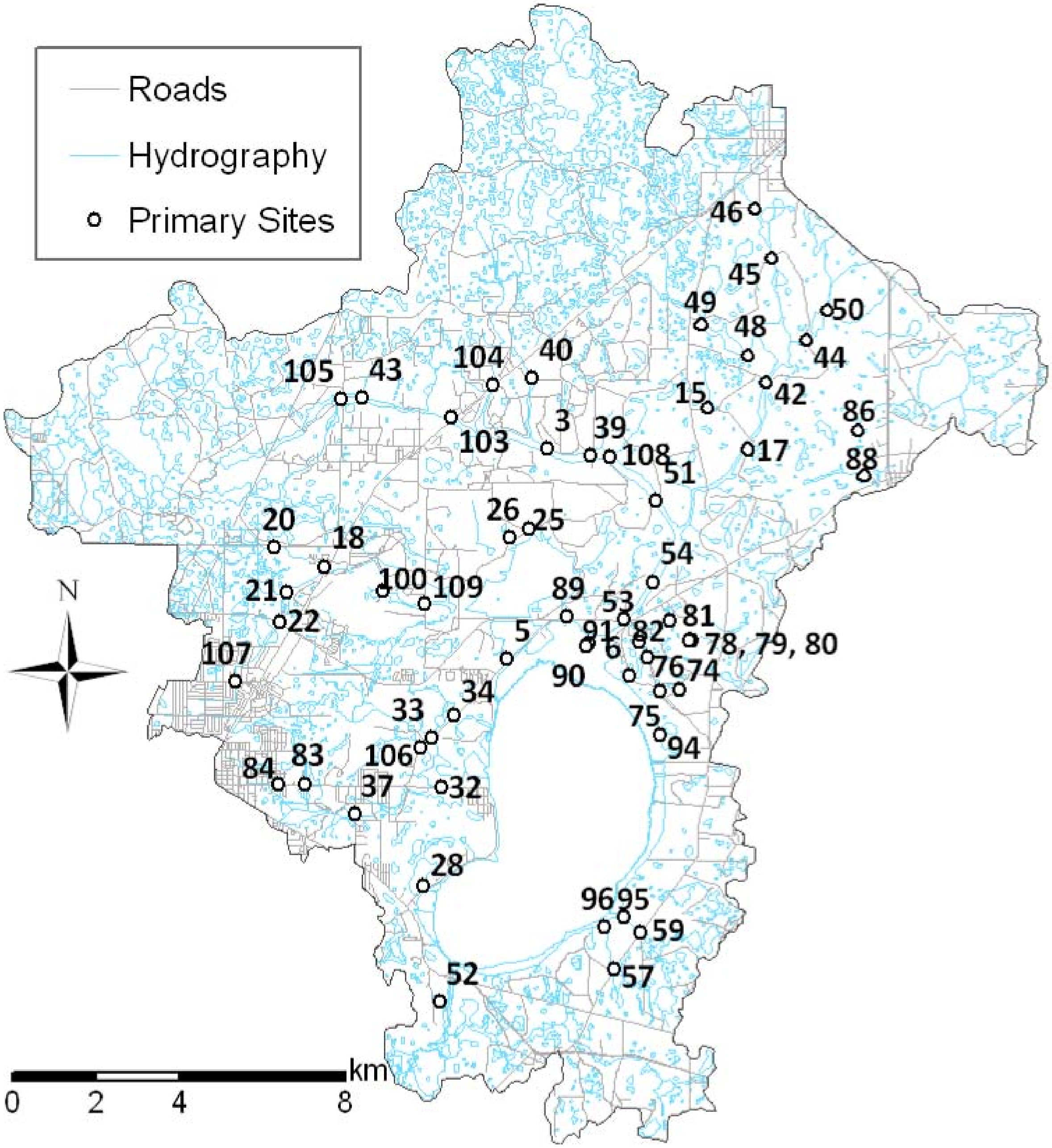

Fig. 7 – Location of sampling sites throughout the Newnans Lake Watershed.

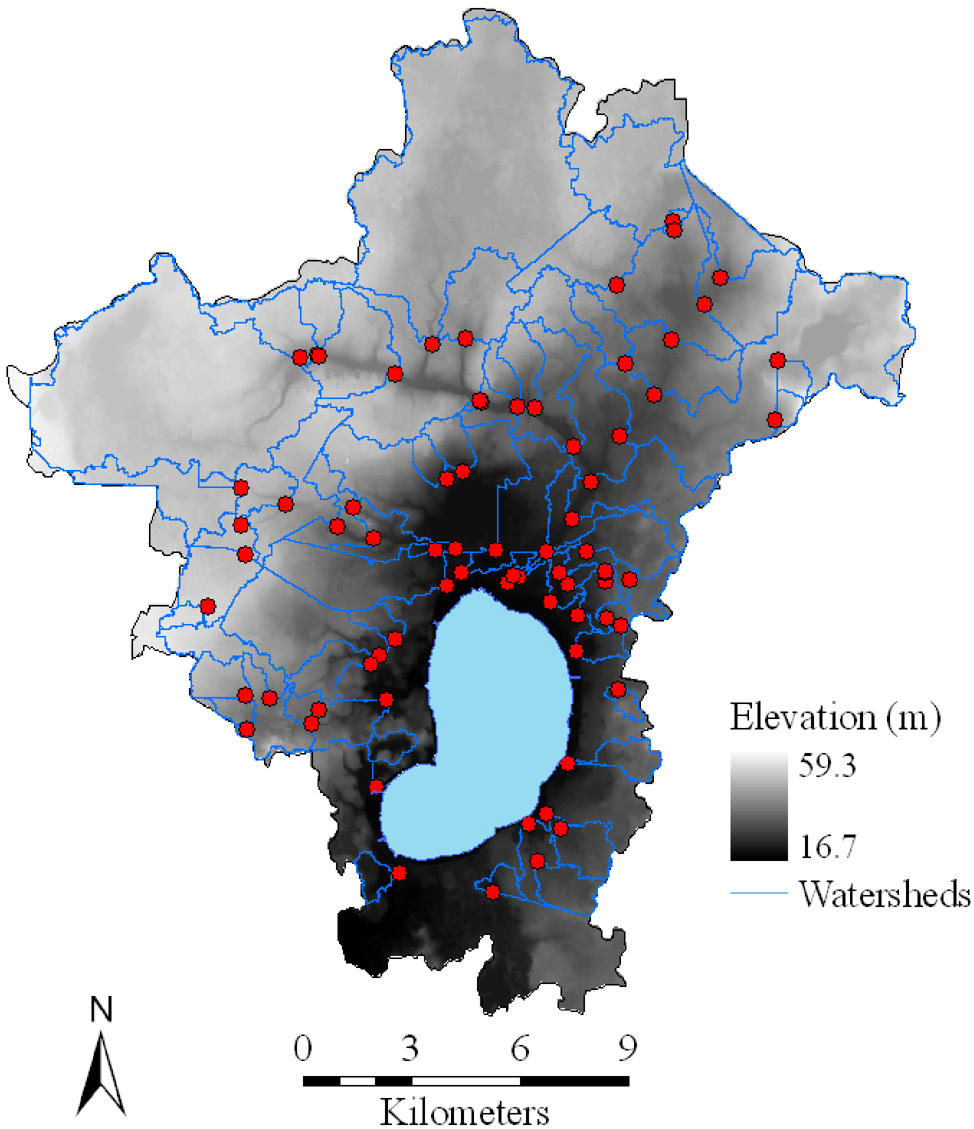

Fig. 8 - Sampling sites with derived sub-basins underlying lidar-derived digital elevation model (in meters, vertical datum is NGVD 1929).

Site Sampling Protocols – Soil and Sediment

During the first field visit, regardless of flow status, we collected sediment samples for chemical characterization. Our rationale is that water quality sampling is entirely predicated on flow, and variability in rainfall in space and time makes interpretation of particular water samples challenging. If relationships can be drawn between observed water quality and sediment quality, then the utility of synoptic sampling can be dramatically increased. The specific question was: is sediment is an effective predictor of water quality. If so, spatial loading measurements can be greatly expedited. To characterize the sediment, we collected three

Table 4 – Summary of site land use and intensity attributes

Site #	LDI	% Forest†	% Urban	% Utilities‡	% Agriculture
26	1.10	98%	2%	1%	0%
25	1.14	88%	12%	0%	0%
94	1.20	99%	0%	1%	0%
96	1.22	50%	0%	2%	4%
57	1.27	87%	1%	1%	1%
28	1.34	75%	1%	0%	24%
104	1.41	97%	0%	0%	1%
95	1.51	100%	0%	0%	0%
59	1.53	95%	0%	1%	5%
91	1.55	73%	0%	0%	14%
108	1.61	100%	0%	0%	0%
90	1.67	100%	0%	0%	0%
53	1.68	92%	0%	0%	1%
54	1.68	95%	0%	0%	3%
105	1.72	92%	4%	0%	3%
51	1.73	98%	1%	0%	0%
3	1.75	74%	12%	2%	10%
39	1.75	99%	0%	0%	0%
20	1.96	83%	10%	6%	0%
40	1.97	78%	5%	4%	0%
103	2.01	54%	5%	0%	38%
52	2.10	60%	8%	2%	30%
6	2.11	100%	0%	0%	0%
75	2.12	78%	0%	0%	22%
82	2.18	72%	4%	0%	22%
76	2.23	100%	0%	0%	0%
81	2.29	67%	5%	0%	23%
79	2.33	0%	0%	0%	0%
80	2.40	11%	0%	0%	89%
78	2.41	100%	0%	0%	0%
89	2.51	96%	3%	0%	0%
74	2.80	100%	0%	0%	0%
18	2.82	71%	17%	5%	0%
34	2.86	29%	3%	0%	66%
106	2.95	35%	34%	1%	26%
5	3.15	52%	19%	27%	3%
83	3.15	31%	65%	4%	0%
100	3.18	54%	11%	26%	7%
32	3.45	58%	31%	1%	8%
21	3.50	50%	27%	4%	0%
109	3.54	42%	4%	44%	0%
22	3.55	32%	61%	5%	0%
37	3.73	57%	35%	4%	0%
43	3.88	85%	4%	0%	11%
33	3.92	56%	27%	0%	10%
84	5.42	11%	84%	3%	0%
107	6.42	2%	98%	0%	0%

† - Forest includes conserved lands (LDI = 1.0), wetlands (LDI = 1.0) and production forests (LDI = 2.0).
‡ - Utilities includes rural paved roads, airport infrastructure and wastewater facility.

independent samples at each site, at the selected site and both 20 m upstream and downstream of the site (to quantify local variability in sediment quality). Samples were analyzed for TP, TN, TC, water extractable P, total Ca, total Fe, total Mg, water soluble ion concentrations and organic matter content. Analyses were performed at the SFRC Forest Water Resources Laboratory, the UF/IFAS Analytical Research Lab and the Wetland Biogeochemistry Lab following standard protocols.

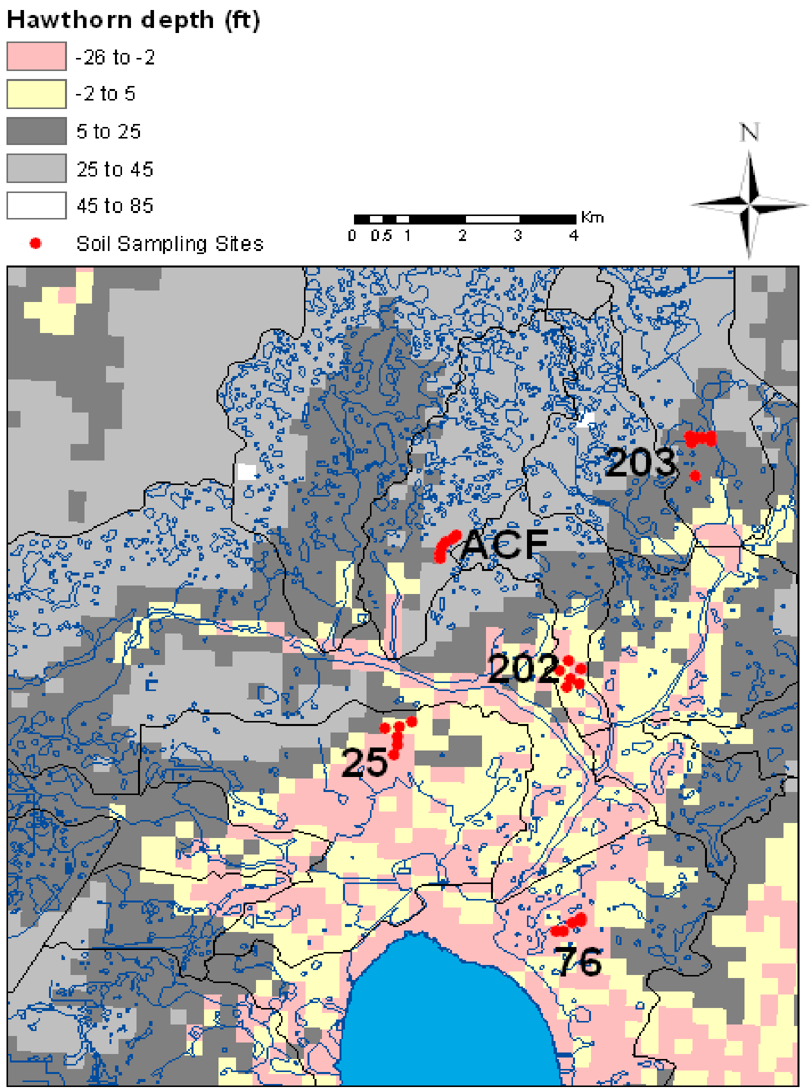

Fig. 9 – Locations of soil sampling locations (n = 5) at which soil cores were taken. Each core was 100 cm deep, and partitioned into 0-20, 20-80 and 80-100 cm sections. Estimated depth to the Hawthorn Formation guided selection of locations, as did presence of plantation bedding.

To test H5 (surface soil enrichment) we selected 5 locations in the basin spanning the gradient in depth to the Hawthorn Formation (< 2' to > 20'). We were particularly interested in sampling areas where silvicultural bedding had clearly occurred; as such, some sampling bias in siting of locations was necessary. At each location (Fig. 9) a 1 x 1 km block), 5 random sites were selected based on compass and pacing. At each site, two replicate soil cores were collected to

100 cm below the surface using a bucket auger. Soil cores were partitioned in 0-20 cm, 20-80 cm and 80-100 cm sections and analyzed for total P using the standard ashing and HCl digestion methods (Kuo 1986). In addition, during well installation along Hatchet Creek, we encountered clear evidence of Hawthorn Formation clays (grey-blue clays with dark apatite nodules). A sample of this material was obtained and similarly analyzed for TP concentrations.

Groundwater Sampling

In addition to sampling surface water flows and sediments, we sought to quantify the groundwater load of water and P to the lake. Fourteen (14) perimeter well arrays were installed at sites around the lake (Fig. 10) selected based on lidar estimates of land surface slopes in the vicinity. Each well array consisted of a well at the lake edge, and an interior well. Both wells, screened for 1.5', were installed to a depth of 3'. Differences in well cap elevations were determined using a laser level, and potentiometric head differences and direction based on tape-down measurements of free water surface depth below the well cap. Darcy's Law, where groundwater flow is computed from information about potentiometric gradients (vertical head), flow path length (distance between wells), profile area (discussed below) and hydraulic conductivity (K_{sat}) (Fig. 11). K_{sat} values were obtained using a slug test at the interior well; a slug of water was injected and water levels monitored over time until the water level had receded 63% of the difference between initial and base levels. The Hvorslev method (Millham and Howes 2005) was then used to determine K_{sat} (Fig. 11).

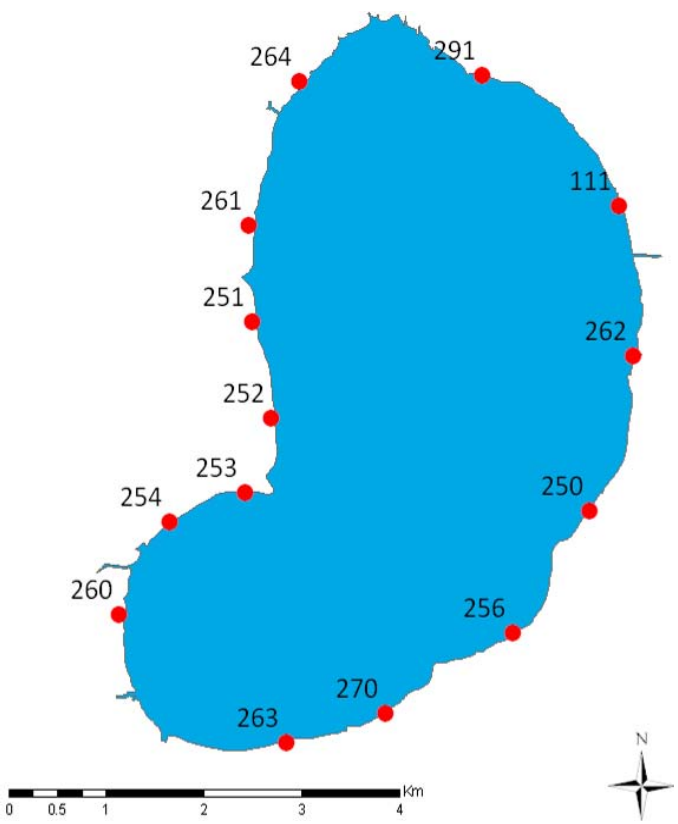

Fig. 10 – Locations of perimeter wells around Newnans Lake. Well ID values are shown.

The cross-sectional area over which flow occurs was determined based on the distance between wells, and the nominal depth of the lake. That is, the width of the cross-section was based on the distance between perimeter wells, with any given well assumed to represent a perimeter length halfway to the next well in each direction. The depth of the cross-section was assumed to be 1.6m (the nominal lake depth); this was assumed because the actual profile over which water flows is made complex by the interbedding of sand and clay layers. This makes estimates of groundwater flow comparatively generous, since it is likely that the depth of the flow field is vertically confined by the presence of low conductivity clay.

Wells were visited monthly between October 2007 and August 2008. At each well, potentiometric gradients were measured each time, and two water samples collected. The first sample was from the interior well (to characterize the chemistry of the groundwater); the second was from the lake perimeter to characterize the chemistry of the near shore lake. In addition, during each sampling day, a mid-lake water sample was collected against which near shore samples could be contrasted. Water samples were analyzed for SRP, TP, TN, NO_x, Cl, F, SO_4, specific conductance, pH and dissolved oxygen. We were principally interested in using chemical measurements to quantify the P load to the lake via diffuse groundwater input, and to trace the provenance of that P by the same covariances as used above to distinguish between fertilizer and geologic P.

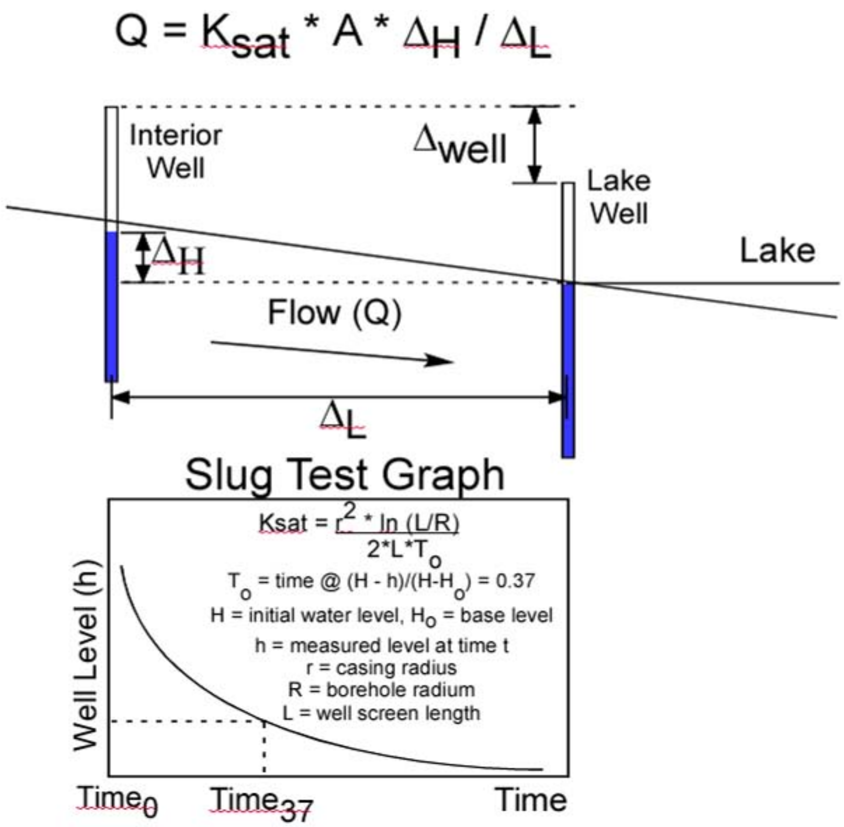

Fig. 11 – Schematic of well elevation measurements for inference of groundwater flow, including (lower panel) estimation of hydraulic conductivity from a slug test.

Water Quality Transects

Hatchet Creek and Little Hatchet Creek are the two main tributaries that flow into Newnans Lake, and historically the two major surface conveyances of P into the lake. Based on preliminary evidence of systematic spatial variability in SRP concentrations along these creeks, we chose to establish two transects to examine source/sink processes. The first is located on Hatchet Creek (Fig. 12 – red dots); it starts near the Gainesville Raceway and proceeds to Waldo Road and on to the sinkhole (located 50 meters downstream of Site #3) that, during the course of this work, was capturing the entire creek flow. The second, located on Little Hatchet Creek (Fig. 12 – red dots), starts below the airport and proceeds into Gumroot Swamp. Efforts to sample along the entire length of Little Hatchet Creek to State Road 26 were constrained by the sediments in Gumroot Swamp, which were extremely flocculent and nearly impossible to traverse on foot. Moreover, during most of the period of sampling, the flows declined to zero along the length of the selected transect.

Sampling along these transects occurred 3 times. During July 2007, extreme baseflow conditions were sampled; the sinkhole at Site 3 was capturing the entire flow in Hatchet Creek so all downstream locations were unsampled. In January 2008 moderate baseflow conditions were sampled, and in February 2008 stormflow conditions were sampled. Sampling logistics and site access limited flow measurements at some transect locations during the last sampling.

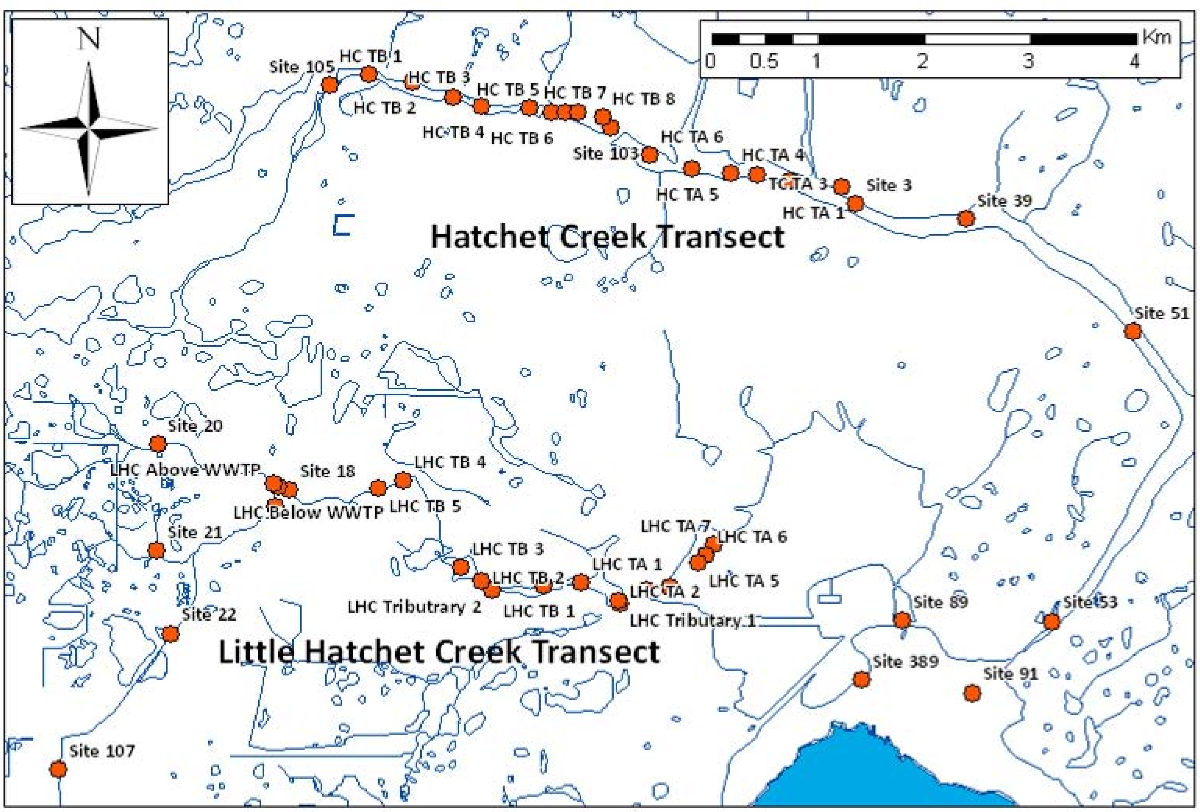

Fig. 12 – Locations of sampling points for water quality transects along Hatchet and Little Hatchet Creek.

At each location, a water sample was obtained and analyzed for the full suite of analytes (see above) and discharge measured using velocity profiles obtained with a Sontek acoustic Doppler velocity sensor.

<u>Hawthorn Formation</u>

The Hawthorn Formation is the principal control on groundwater-surface water interactions in North Florida, and a potential source of significant P loading to surface water systems. Despite the critical importance of this geologic feature on water systems, the extent, depth and thickness of the Hawthorn remains relatively uncertain. Fig. 13 shows a contour map for the elevation of the top of the Hawthorn Formation across the St Johns River Water Management District derived from stratigraphic data collected by the Florida Geological Survey and interpolated by the Groundwater Division at the St. Johns River Water Management District (J. Davis and D. Boniol, *unpublished data*). The horizontal and vertical resolution that can be obtained from this at the scale of the Newnans Lake Watershed (black line in Fig.

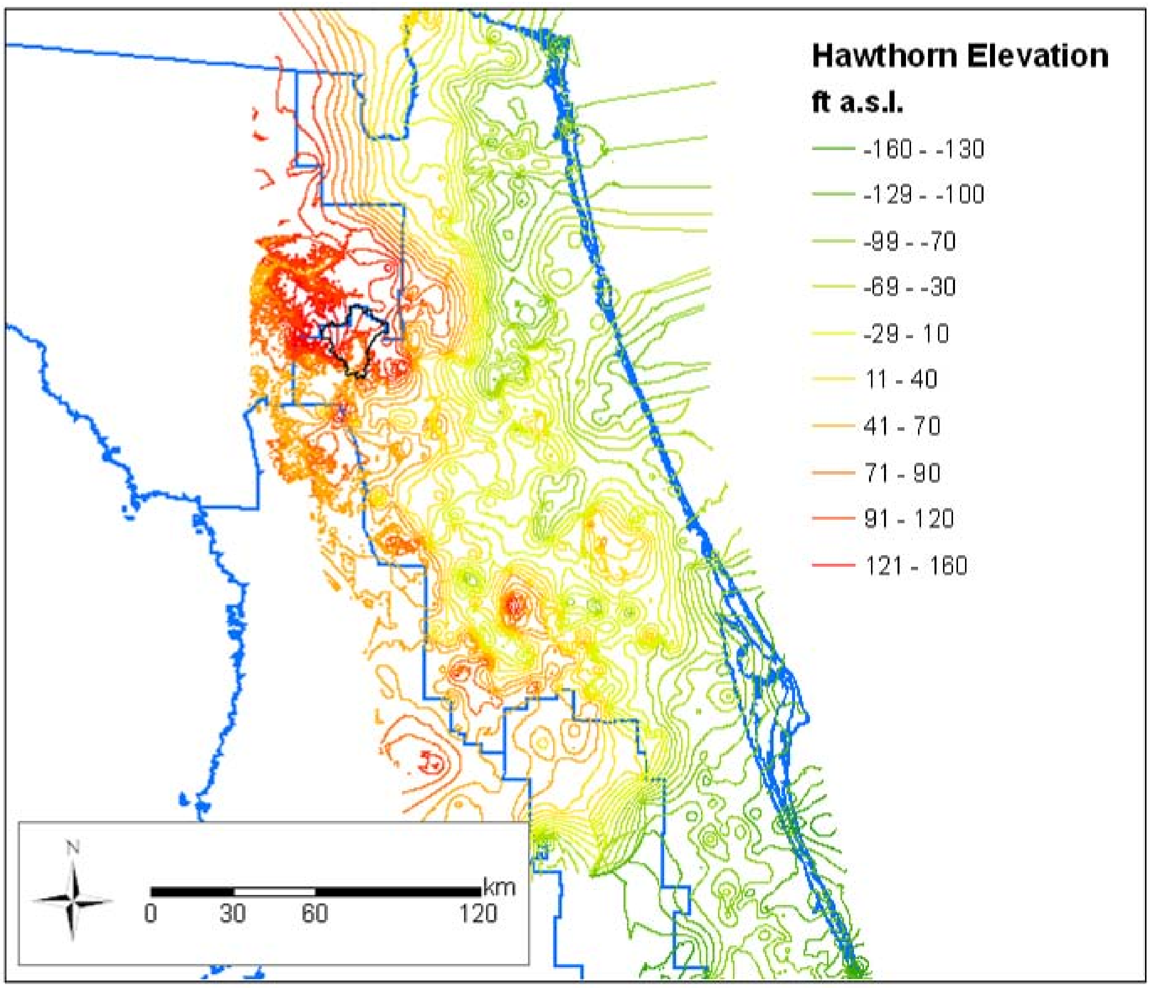

Fig. 13 – Contour map showing the elevation of the top of the Miocene Hawthorn Formation in feet above sea level (a.s.l.) throughout the St. Johns River Water Management District, as interpolated from stratigraphic cores. Black outline is the Newnans Lake watershed.

13) is rough, and absolute values should be considered approximations only. However, interpolating these contour lines reveals substantial variability within the basin with regard to the surface elevation. Using recent ground surface elevations estimated using lidar, the difference between the elevation of the land surface and the elevation of the top of the Hawthorn can be estimated (Fig. 14), revealing strong suggestive evidence of a spatially variable role of Hawthorn materials in the surface water systems of the Newnans watershed. In particular, there are areas of the basin where the Hawthorn is more than 50 feet below the land surface and others, particularly in the vicinity of the lake, where the land surface is at or even below the elevation of the top of the Hawthorn. Color coding in Fig. 14 corresponds to a priori estimated risk of Hawthorn effects of surface water processes; grey sites indicate locations where the Hawthorn is well below the land surface, yellow pixels areas where the land and Hawthorn surface are at the same elevation, and red where the land surface is below

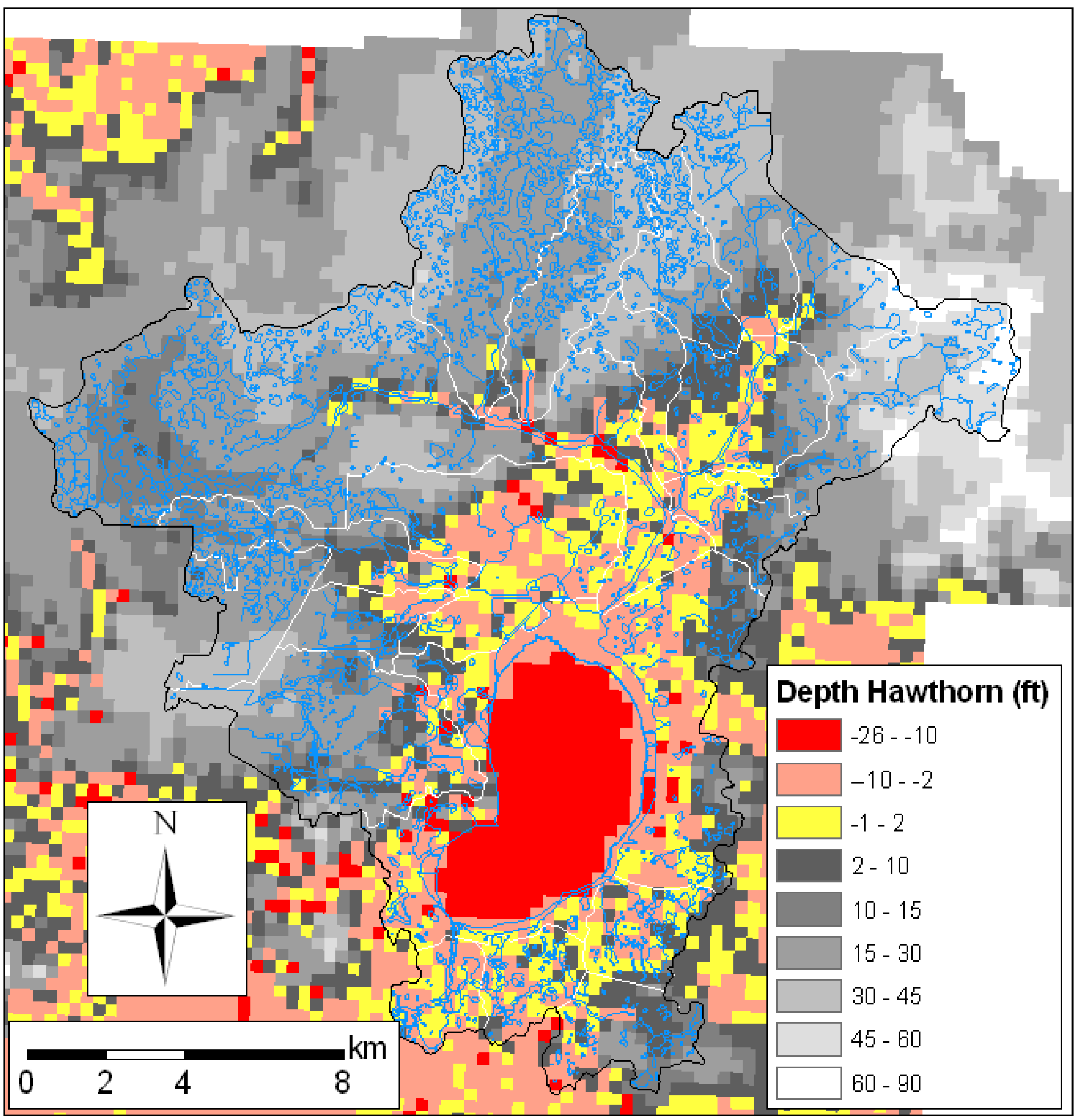

Fig 14. Depth to Hawthorn layer in the NLW. Pixels are 100 x 100 m.

Table 5. Extent of depths to Hawthorn Formation in the Newnans Lake Watershed	
Depth (ft)	Extents (%)
< 0	28.6
0 - 5	9.2
5 -10	5.9
10 - 15	8.3
15 - 20	8.7
20 - 30	18.6
30 - 40	14.5
40 - 50	4.0
50 - 73	2.2

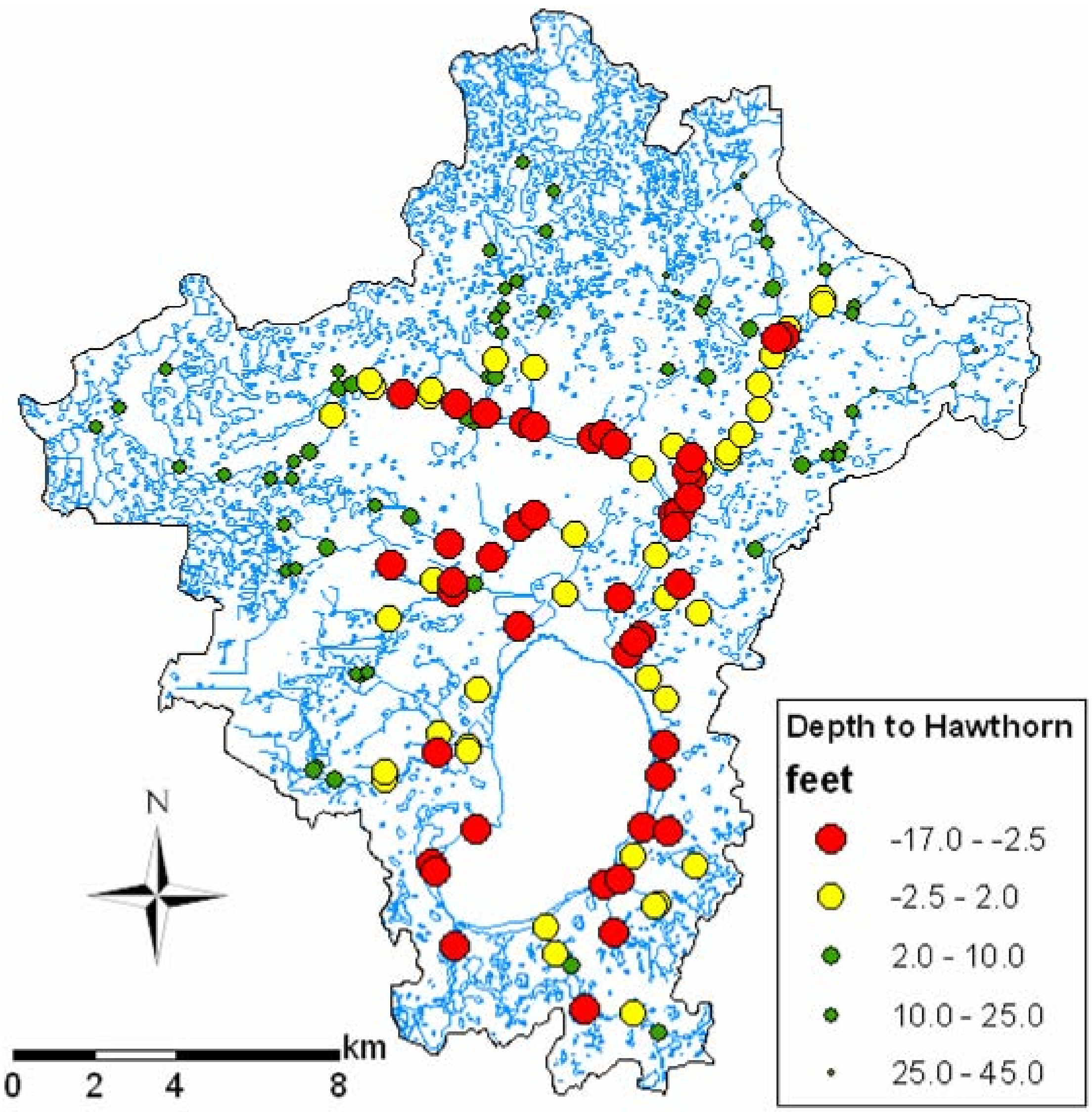

Fig. 15 – Estimated depth to Hawthorn layer at sampling sites (feet)

the estimated top of the Hawthorn. We use this information as a predictor of stream P concentrations; in particular the transects where water quality is sampled along Hatchet and Little Hatchet Creek. Table 5 summarizes the spatial representation of different depths, indicating that the land surface over one third of the watershed is either in or very near to the top of the Hawthorn. The depth to the Hawthorn at each of the locations sampled for water quality is depicted in Fig. 15.

III. RESULTS

Water Sampling Results – Exploratory Analysis

This two-year project allowed semi-continuous sampling of numerous locations in the Newnans Lake watershed. Overall, we collected over 580 stream water samples, 160 stream sediment samples, 150 soil samples, and 12 monthly samples from 14 lake perimeter wells. The samples by site and flow regime (Table 6) depicts the sampling effort for stream water.

Table 6. Summary of site observations for part of Year 1 (additional samples [~150] have subsequently been collected).

Site Number	Water Samples	Stormflow Samples
3	13	4
5	3	3
18	15	5
20	10	3
21	4	
22	5	5
25	4	3
32	25	11
33	9	4
34	4	1
37	21	7
39	7	3
40	1	3
43	2	1
51	8	4
52	9	2
53	3	5
57	2	3
59	2	2
75	2	3
76	3	3
82	2	2
83	9	4
84	7	2
89	21	8
91	3	2
100	28	9
103	16	7
104	3	2
105	11	4
106	4	2
107	3	5
109	10	3
201	1	5
Miscellaneous	24	19
Transect Samples	75	36
TOTAL	**369**	**185**

Each sample was analyzed for a suite of chemical analytes that were used to draw inference about the magnitude and source of nutrient loading to the lake. A summary of the full suite of observations across all sites, flow regimes and analytes is shown in Fig. 16. Note that the absence of global correlations does not imply the absence of a relationship; this report delves into covariance patterns among analytes in detail to extract relationships that are useful for understanding and predicting watershed dynamics. Of note in Fig. 16 is the observation that nearly all water quality parameters are non-normal, following instead a log-normal underlying distribution; all correlations are after natural log transformation. However, subsequent analyses, where appropriate have retained untransformed data to aid in interpretation of results.

Among the key elements of preliminary analyses of these data is to understand the relationships between concentrations of key solutes and flow. A series of figures (Fig. 17, 18, 19) shows flow relationships for three key stations: Site 103 (Hatchet Creek), Site 100 (Little Hatchet Creek) and Site 32 (Lake Forest Creek). These three sites were selected because they were visited most frequently, had the most consistent flow (e.g., Hatchet Creek sites below the sinkhole at Site 3 were not flowing for much of 2007 and early 2008). The figures summarize a the relationships between flow and solute concentrations.

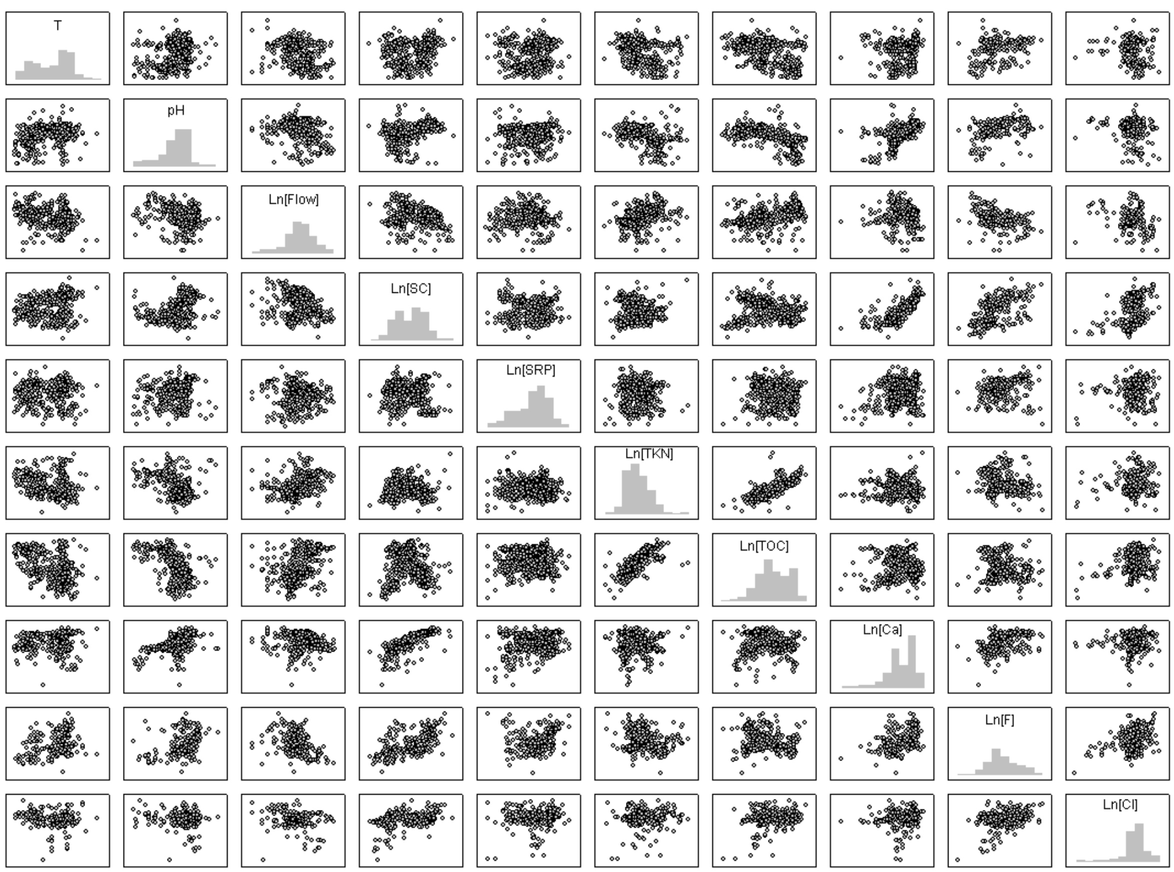

Fig. 16 – Matrix plot of analyte correlations across all sites and times. T = temperature, SC = specific conductance, SRP = soluble reactive P, TKN= total Kjeldahl N, TOC = total organic C.

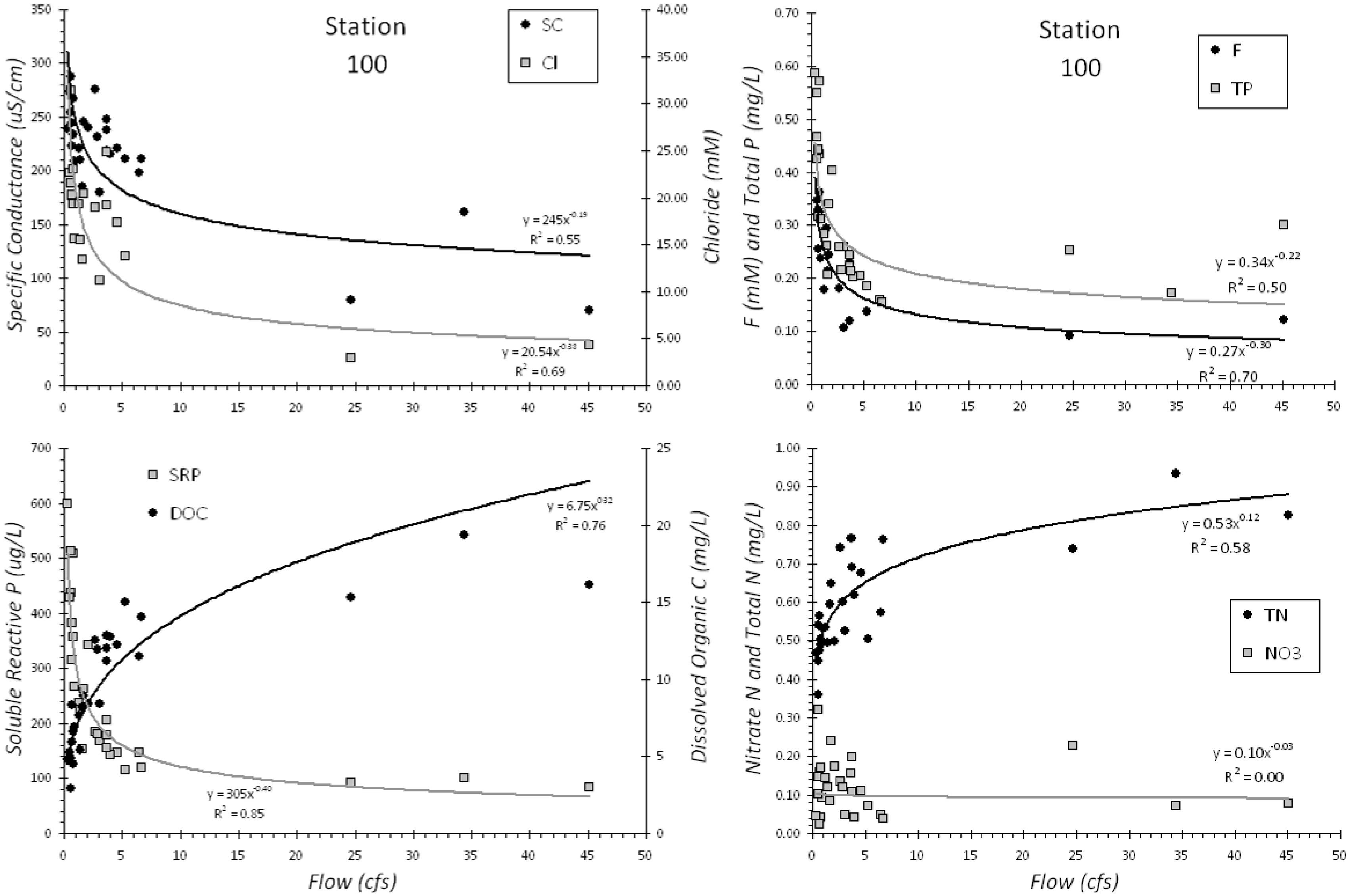

Fig. 17 – Covariance between flow and various attributes of water chemistry for station 100 (Little Hatchet Creek)

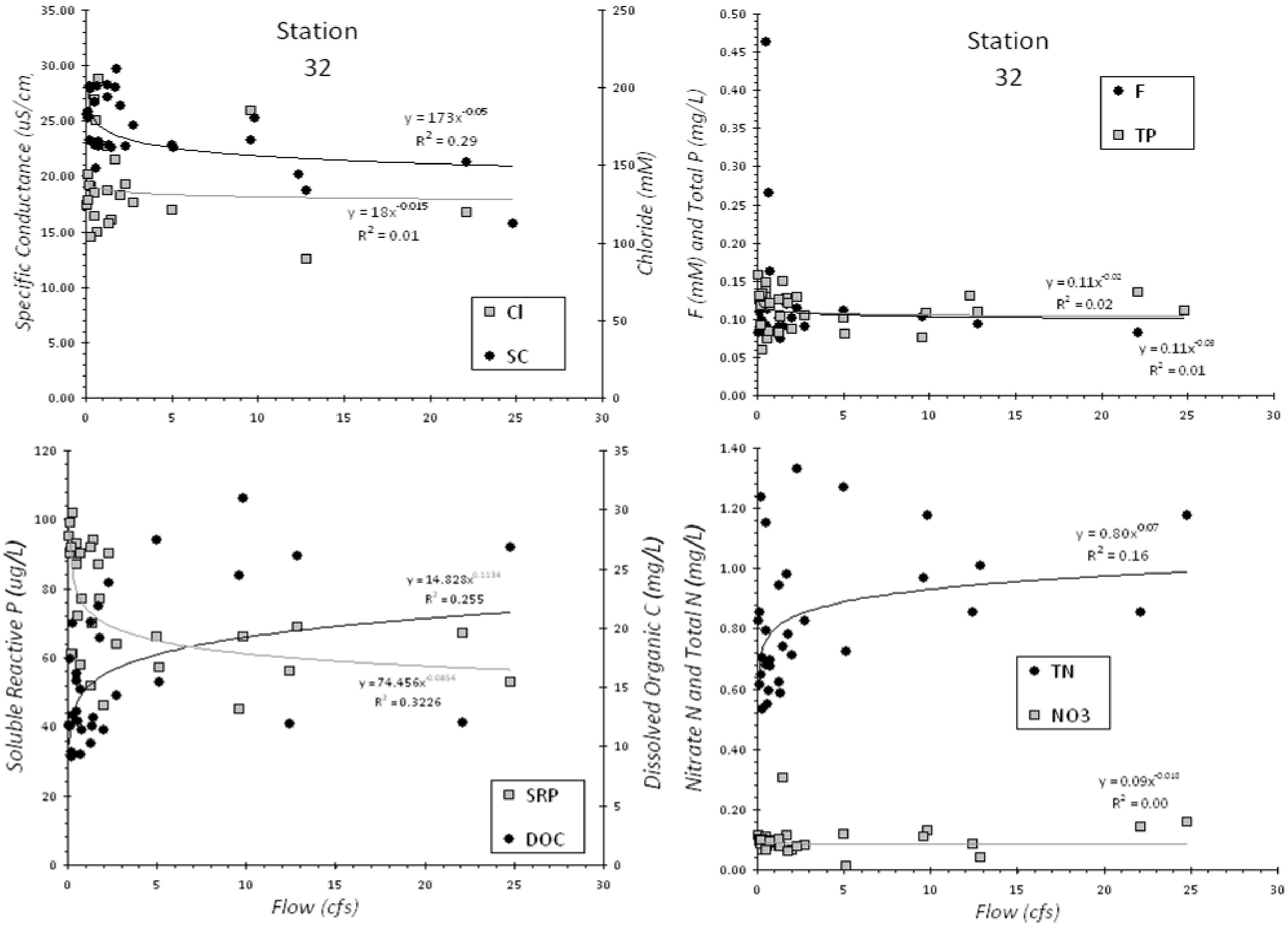

Fig. 18 – Covariance between flow and various attributes of water chemistry for station 32 (Lake Forest Creek)

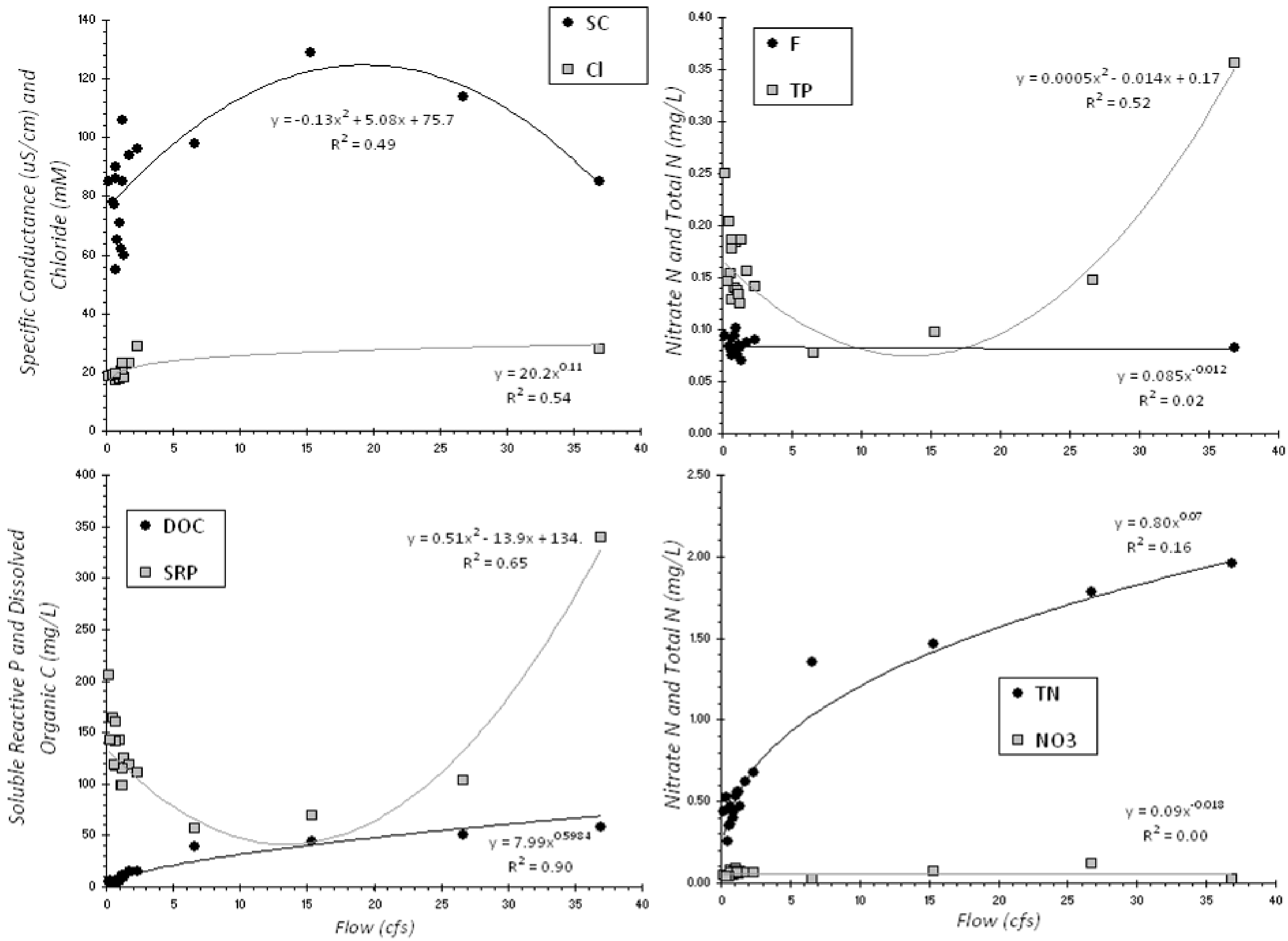

Fig. 19 – Covariance between flow and various attributes of water chemistry for station 103 (Hatchet Creek).

Fig. 17 shows the flow vs. solute relationship for site 100 near the discharge point of Little Hatchet Creek into Gumroot Swamp. This site is co-located with a SJRWMD flow gage, and was selected for regular sampling because of access and location. The relationships of note at this site are strong dilution gradients with flow. Both chloride (Cl) and specific conductance (SC) decline with flow as would be expected if baseflow is the principal source of ionic activity, diluted following storm events. The magnitude of the dilution effect is comparative small, with asymptotic levels of SC nearly 100 uS/cm (rainfall is closer to 10 uS/cm). There are similar, though more pronounced flow effects on total P and F (the association between these solutes will be explored in detail below). The effect of flow on TP concentrations would, with a longer term data set, reveal an interesting reversal of the dilution trend at high flow. On Fig. 17B there appears to be a rising trend of TP with flow at the highest flow rates. Notably, this peak flow trend reversal is not observed for F, an observation that could suggest that geologic flows of P (as traced by covariance with F) decline with flow, and other sources of P dominate at peak flow. It is essential to note that the SRP relationship with flow (Fig. 17C) does not show the same trend reversal, suggesting further that the additional P being entrained in flow is not mineral, but organic, perhaps mobilized from various landscape stores of organic material.

The covariance between F and flow is extremely strong, which suggests a constant flow of solute to the hydrologic system that is diluted variably depending on the mass of water that carries it. Given the multiple lines of evidence for geologic P that are discussed later, this supports a conceptual model of slow, constant weathering that liberates mineral P from the geologic system to the hydrologic system, at least within the Little Hatchet Creek watershed.

Other relationships of note in Little Hatchet Creek are a strong positive relationship between flow and DOC load; this is consistent with differences in TP and SRP vs. flow, suggesting that dissolved organic forms of both N and P would dominate at peak flows (see Fig. 17D for TN relationship). Notably, there is no association between NO_x and flow (Fig. 17D).

For Lake Forest Creek (Fig. 18), the relationships are slightly different. There is a far weaker dilution effect, with specific conductance values falling only ca. 20% from a high of 190 uS/cm at low flow to approximately 150 uS/cm at peak flow. The cause for this weak effect is unknown since the ratio of peak flow to base flow is roughly the same for both Little Hatchet and Lake Forest Creeks (~15:1). The flow effects on TP, SRP, F and DOC are also highly muted compared with Little Hatchet Creek, though still significant and following the same sign (increases in DOC, decreases in TP, SRP and F with flow). As before, no association with NO_x was observed, and the relationship with TN followed the pattern of DOC, suggesting that most of the N load is in organic form.

For Hatchet Creek (Fig. 19), the relationships are strong, but in some ways more complex. Most importantly, there does not appear to be a strong dilution effect evident from the data. Indeed, the covariance between flow and specific conductance is positive suggesting that a source of ionic activity is actually mobilized by higher flows, rather than diluted by additional runoff. There is essentially no relationship between flow and F, while the relationship with TP is sufficiently modal to warrant fitting of a quadratic function rather than the power law function that has been used for other watersheds. In particular, TP increases dramatically at high flow rates. Where this effect was observed in Little Hatchet Creek, it was clear that the additional P was organic in origin because SRP concentrations followed a typical dilution curve. In Hatchet Creek, however, the increase in concentration with flow appears to occur because of SRP concentrations; indeed, the correlation between SRP and TP in Hatchet Creek is nearly perfect ($r = +0.98$) whereas the same correlation for Little Hatchet Creek is slightly weaker ($r = 0.88$) with the main residuals at low concentrations when TP is dominated by organic P.

That the relationship of SRP with flow in Hatchet Creek (Fig. 19) is reversed to other creeks is not due to changes in the way that organic P (DOC is a surrogate) is delivered. In fact, the highest DOC and DON levels are observed for Hatchet Creek, with DOC levels approaching 60 mg/L at the highest flows, and DON concentrations nearing 2 mg/L. As with other creek, NO_x shows no association with flow, but concentrations are markedly lower in Hatchet Creek (mean NO_x = 0.05 mg/L) than in Little Hatchet (mean NO_x = 0.11 mg/L) and Lake Forest Creek (mean NO_x = 0.12 mg/L).

The most likely explanation for this unusual behavior between flow and SRP, which defies expectations and is tremendously important for loading models if it is sustained, is that the small feeder creeks that drain to Hatchet Creek in the vicinity of Site 103 (sampling locations 104 and 40 are examples) are actively incising into the Hawthorn. Since these creeks are intermittent, the weathering process leads to high yields of mineral P over time that is mobilized during storm events. During transect sampling efforts under stormwater conditions (described in detail below), we observed SRP concentrations arriving from these tributaries of 330 and 814 ppb. During moderate flow conditions, the tributaries were 221 and 158 ppb, respectively, which though enriched are not as large. Further work is necessary to determine the dynamics of this loading. One extremely important insight of these data is that increased flow may actually massively increase load; ordinarily concentrations vary negatively with flow so that while loads increase with discharge, the effect is muted. In this case, where flow and concentration are, at high flows positively correlated, loads will be greatly exaggerated at high flow. The implications are that a) understanding storm flow SRP concentrations is tremendously important (this work is a start towards that understanding, but more is needed) and b) that efforts to manage flow, therefore, may play and important role in load reduction.

Also relevant to understanding patterns of loading is the geography of mean concentrations for key analytes. While this visualization does not necessarily allow inference of drivers, it does offer some insight into patterns. Moreover, it helps underscore the intrinsic spatial variability that confounds straightforward inference regarding spatial load detection. For example, Fig. 20, which shows mean SRP values at all stations for which multiple observations through time were available indicates that along creek flowpaths, the SRP concentrations change markedly, from near zero at the headwaters of Little Hatchet Creek, to over 250 ppb below. Similarly, in the northeastern Prairie Creek catchment there were 8-10 sites within 2 km of each other that vary in mean SRP concentrations by an order of magnitude (40 to 400 ppb).

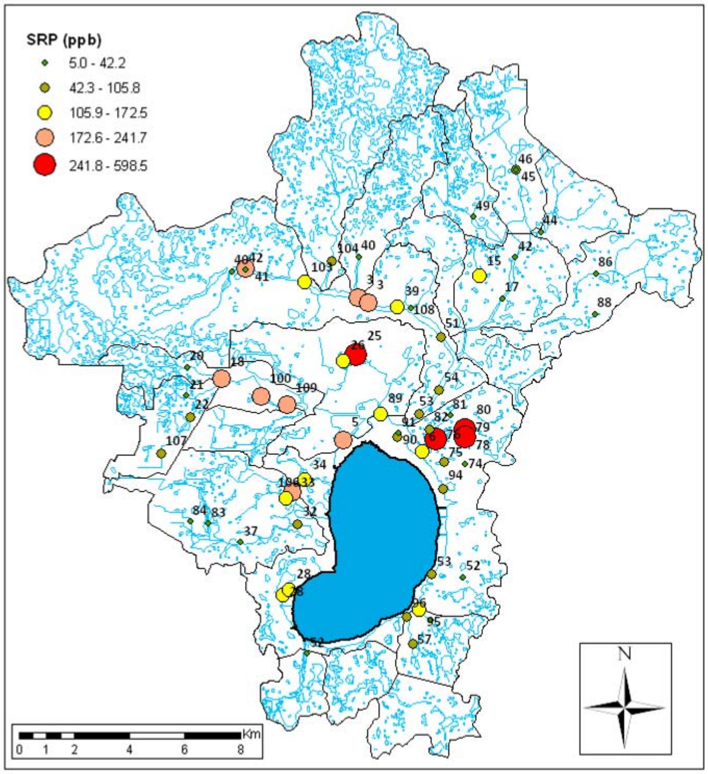

Fig. 20 – Geographic distribution of mean stream water SRP concentrations.

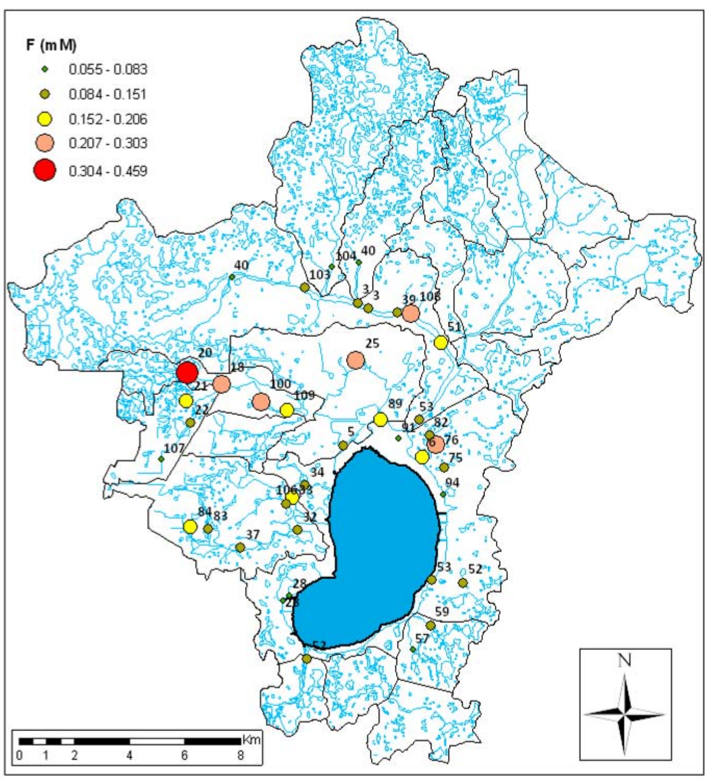

Fig. 21 – Geographic distribution of mean stream water fluoride concentrations.

Despite the substantial variability, several important inferences can be obtained from this visualization. First, with regard to SRP (Fig. 20), Bee Tree Creek loads very little. The same can be said, somewhat unexpectedly, of Lake Forest Creek and the feeder creeks that drain the intensive silviculture to the southeast of the lake. Peak concentrations appear in the mid reaches of Hatchet and Little Hatchet Creeks, and in the small creeks that drain the blueberry farm from the northeast. It also appears clear that there is a systematic attenuation or dilution gradient from the upper Hatchet Creek sites through to the sites where the creek discharges to the lake. This attenuation is almost certainly a consequence of the fact that flows at these lower stations are rare (typically only during stormflow) because of the influence of the sinkhole (at Site 3) that captures most of the accumulated baseflow (up to 2 cfs) from the upper and middle parts of that basin.

Similar patterns were observed for fluoride (Fig. 21), a covariate of P from geologic sources. However, the spatial pattern in this analyte is clearly confounded to a significant extent by flows of F from municipal water supply in upper Little Hatchet Creek (Site 20, which had the highest recorded F concentrations is a ditch that drains 53rd Avenue and the Murphree water treatment plant). High levels of F were also observed in Lake Forest Creek despite low P concentrations possibly due to leaking pipes in East Gainesville (the water in Lake Forest Creek at baseflow is surprisingly similar to tap water in conductance and elemental profile).

Patterns of Ca concentrations typically parallel those of F (Fig. 22), with some important differences, particularly in Hatchet Creek. The general pattern of low Ca (and low F) in the southeastern lake drainage, and throughout much of the northeastern feeder creeks suggests that this is dominated by surface water drainage, and not water that interacts extensively with Hawthorn clays. The exception is site 76 (largest Ca concentration in northwestern lake) which also has the highest P concentrations but comparatively low F concentrations. We surmise that this water is irrigation runoff from deep aquifer sources (high Ca, low F) that has entrained fertilizer (high P). High levels of Ca (and F and P) at site 25 (Gumroot seepage face) suggest a geologic P source in that area, and similar qualitative inference can be applied for Little Hatchet Creek flows below Waldo Road.

There were very few sites that had high NO_x concentrations (Fig. 23). There appears to be a pattern where higher concentrations are observed in Lake Forest Creek and the northwestern drainage paths as well as the northeastern creeks that drain the blueberry farm. Extremely low levels in almost the entire basin underscore the anomalous behavior of P, and the need for establishing evidence for a non-fertilizer source. Further attention is paid to this line of evidence below.

Finally, there are strong spatial patterns to specific conductance (SC) (Fig. 24) and dissolved organic C (Fig. 25). SC values can increase due to a) longitudinal changes in source from surface water to intermediate aquifer water, b) from mixing between surface water and groundwater due to human activities (leaking municipal supply pipes, irrigation water), and c) from long residence flowpaths that result in evapoconcentration of ionic activity. There is case to be made for all three mechanisms in the Newnans Basin.

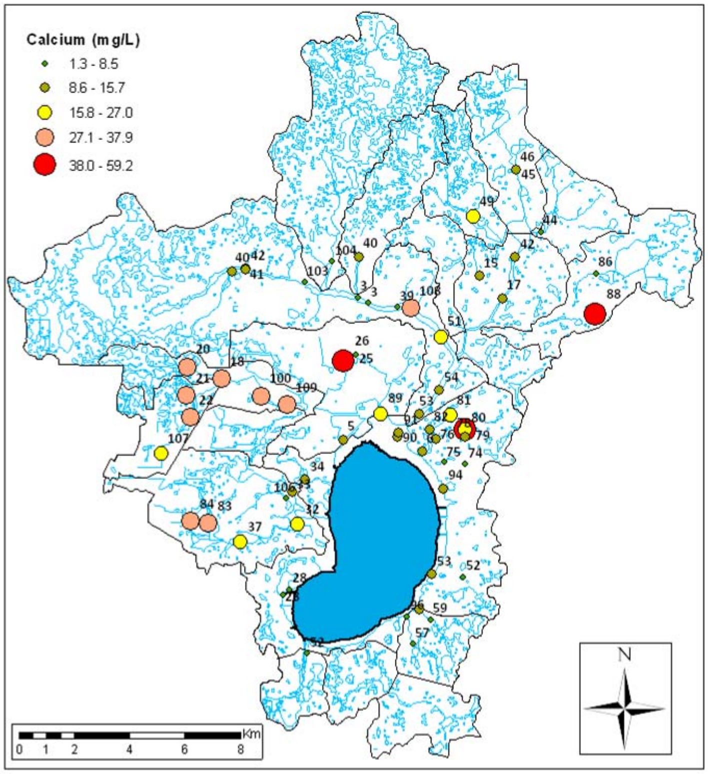

Fig. 22 – Geographic distribution of mean stream water calcium concentrations.

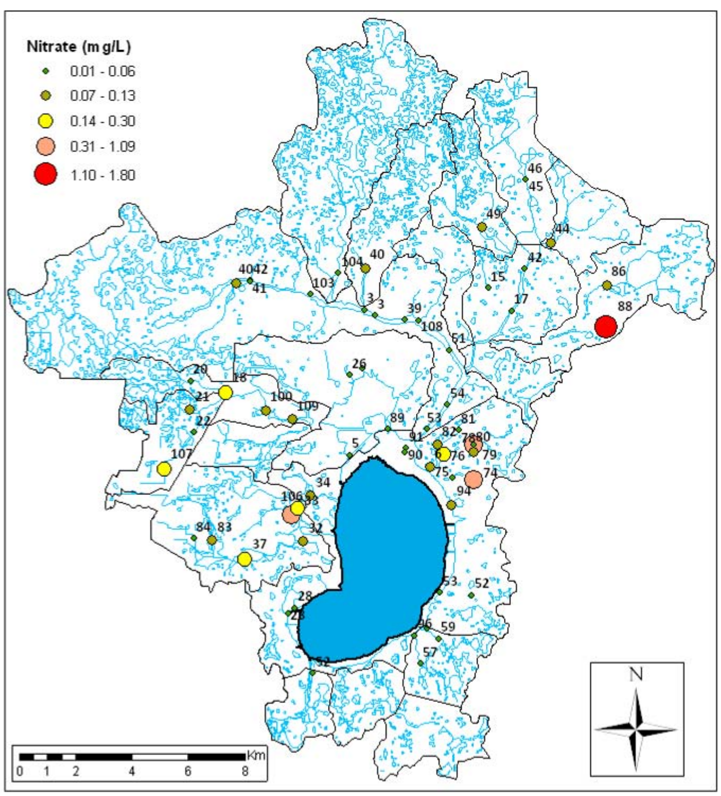

Fig. 23 – Geographic distribution of mean stream water NO_x *concentrations.*

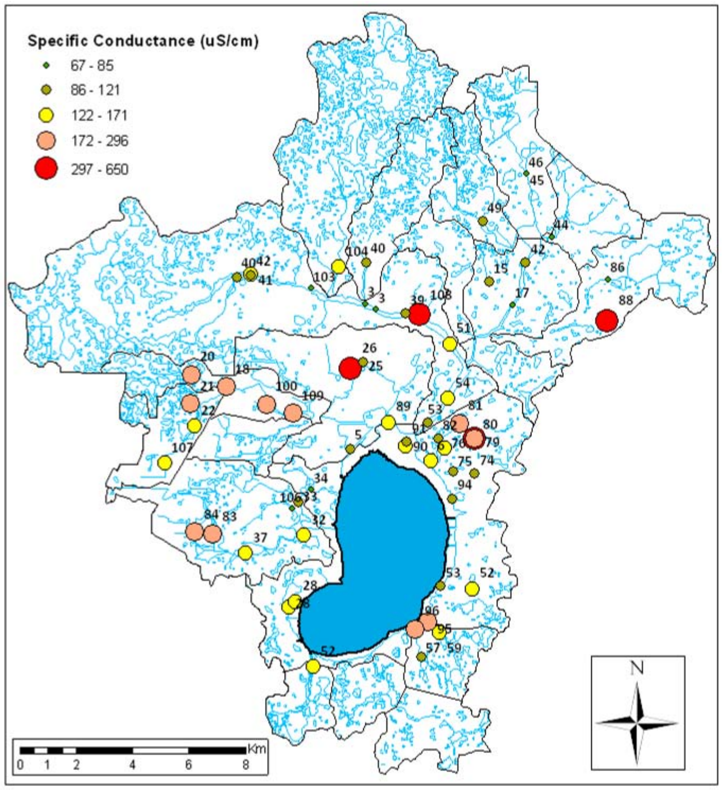

Fig. 24 – Geographic distribution of mean stream water specific conductance values.

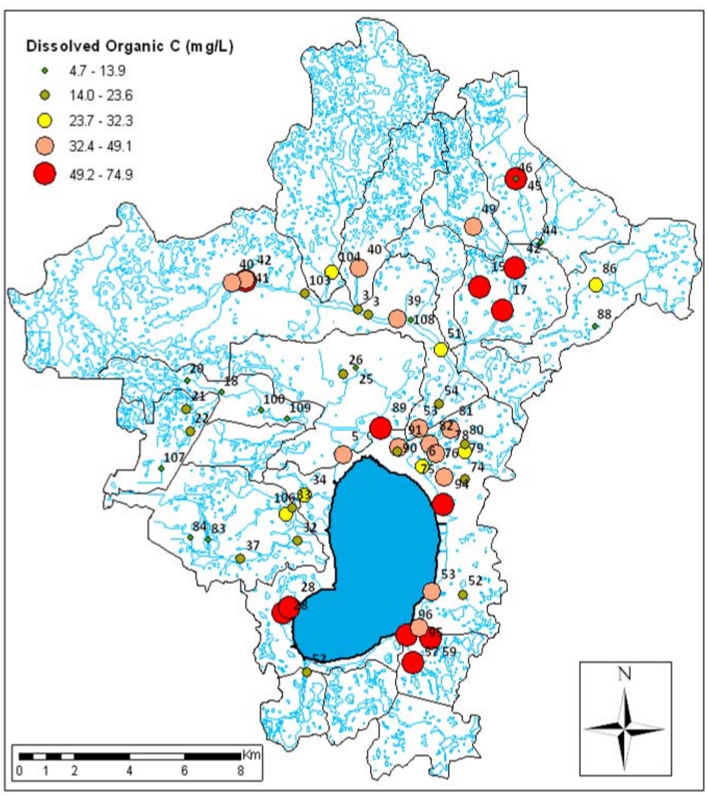

Fig. 25 – Geographic distribution of mean stream water dissolved organic C concentrations.

Evidence for seepage from intermediate aquifer water can be seen along Hatchet Creek, where SC values go from very low to moderate over the pathway from the headwaters to the lake. Bee Tree creek is uniformly low SC, which is principally due to the fact that flow was only observed in that system following major storms when dilution effects were most pronounced. There is also evidence of anthropogenic mixing gradients in Lake Forest Creek (where SC levels are unusually high in the headwaters) and in the northeastern creeks where irrigation runoff raises the SC of the water in some of the channels. Finally, there is evidence of long-flowpath evapoconcentration increases in the southeastern creeks where lack of channels and shallow gradients presumably lead to long landscape residence times. If water provenance becomes an area of active need, deployable specific conductance meters at key locations is likely to yield information that will allow quantitative deconvolution of water flows into sources (e.g., surface and groundwater).

Patterns of DOC are as expected (Fig. 25). High levels were observed in areas that drain large wetland complexes (e.g., site 105 which drains the Buck Bay area, and the southeastern creeks that are actually cypress sloughs with poorly defined surface channels). Also notable is the quantity of DOC picked up in Little Hatchet Creek during passage through Gumroot Swamp (to Site 89) and in Hatchet Creek along the same low gradient flowpath (to site 90). High levels of DOC in Bee Tree creek, again, are likely due to the fact that sampling was only possible in those locations during stormflows when DOC levels are typically high (see Figs. 17-19).

Question 1: Land Use and Pollutant Loading

The first hypothesis addressed the expected relationship between land use/cover and environmental loads of priority pollutants. TMDLs have been for this watershed for P and N, but relationships between land use and P are of principal concern because much of the N is fixed internally. Fig. 26 summarizes the relationship between site mean water quality parameter concentrations and the mean LDI in the contributing area. Our predictions were that SRP, NO_x, and specific conductance would increase with LDI, and DOC would decrease. Fig. 26 suggests that the expected positive relationship was observed for both NO_x and specific conductance ($p < 0.001$ and $p = 0.002$, respectively), but not for SRP; indeed the relationship is reversed with low intensity land uses generally supporting higher SRP concentrations, though that relationship is not significant. The relationship for NO_x was strong; further, the relatively low model efficiency ($r^2 = 0.30$) increases dramatically ($r^2 = 0.62$) when two extreme locations are removed from the regression. Those locations are high intensity agricultural sites in the northeastern lake draining a blueberry farm. It has been argued (Reiss 2004) that agricultural impacts have been underestimated by LDI, a contention reinforced by these observations.

The DOC relationship with LDI is strongly negative, as expected. The rationale for this expectation was that the presence of wetlands and riparian ecosystems along with a dominance by subsurface flowpaths in low intensity landscapes leads to greater DOC production. Moreover, impervious surfaces associated with high intensity landscapes tend to produce large stormflows that dilute dissolved organic matter.

Discordance between N and P responses to land use intensity suggests that other factors are strong confounders of the P vs. LDI relationship. Previous work (Brown and Vivas 2005, Reiss 2006) suggest similar expected trends for both nutrients.

A second prediction made regarding the relationship between LDI and water quality is that the fraction of mineral nutrient species would increase with LDI. Figure 26-E and 26-F summarize our observations of mineral fractions for both N and P, and suggests extremely strong evidence for this effect when considering N concentrations and no evidence for P concentrations. That is, the N fraction delivered in mineral form (dissolved inorganic N, DIN = NO_x+NH_4) increases markedly with land use intensity, as would be expected by the increased use of mineral fertilizers and short circuiting of landscape locations where transformation of inorganic N to organic forms can occur. That process was not observed for P despite significant evidence that the same mechanisms that control DIN delivery hold for P. As before, the sites that depress the model efficiency between LDI and DIN are sites that drain an intensive agricultural portion of the basin where DIN concentrations are extremely high. It also worth noting that the scales on the mineral axes are markedly different, with DIN fractions peaking at 30% with a mean of less than 10% while mineral P fractions are as high as 97%, with a mean value of 62%. This suggests a large source of mineral P that operates independently of N loading processes.

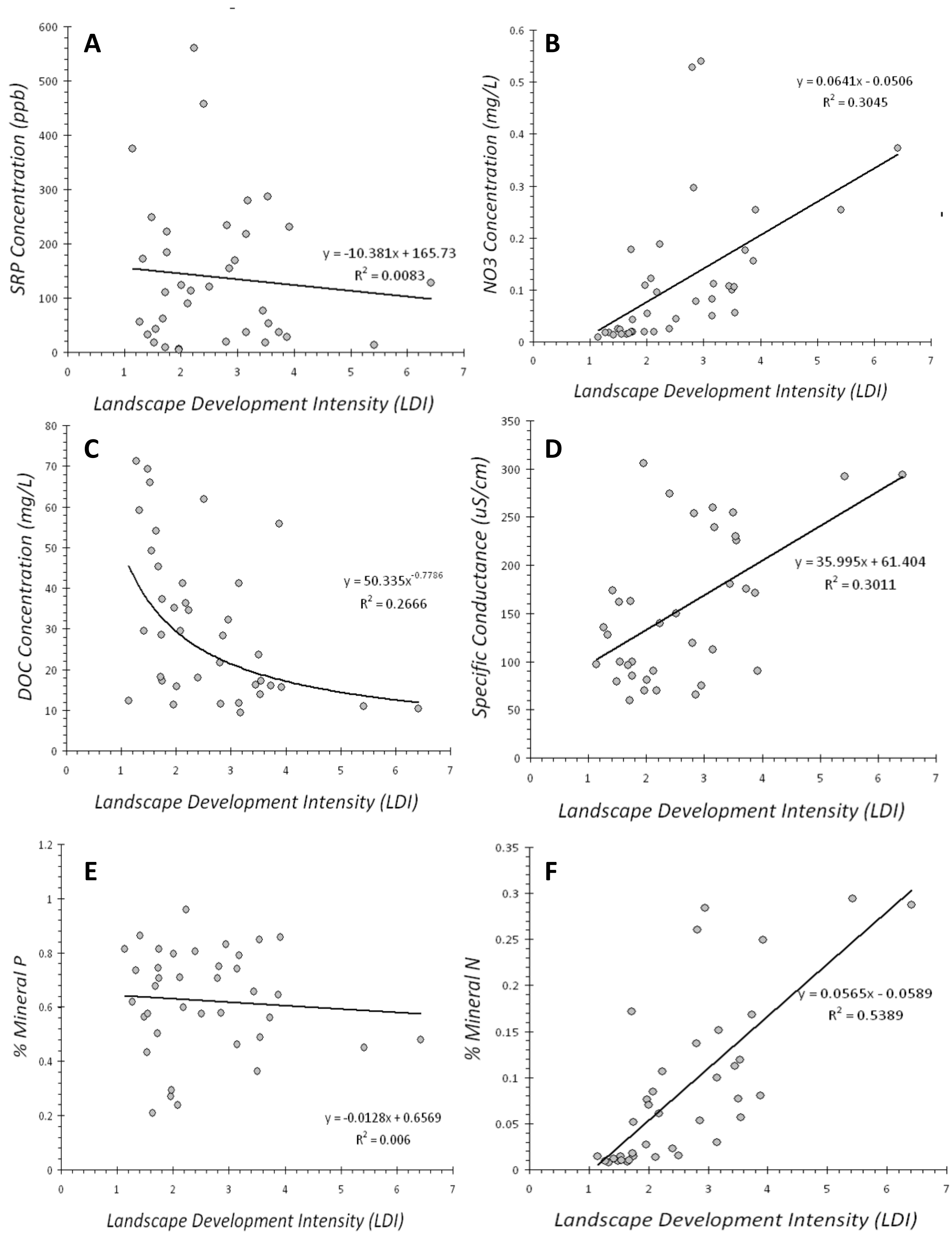

Fig. 26 – Relationships between key water quality analytes and landscape development intensity. In each case except DOC, the expected association is positive.

A more detailed view of relationships between SRP and land cover (Fig. 27 shows observed SRP concentrations as a function of the fraction of contributing area under specific land cover types. Note that samples from different times at the same site are treated as independent. No association exists for % forest. Relationships with % agriculture and % industrial are both moderately significant ($p = 0.02$ and $p = 0.04$, respectively). While the effect is not significant, the direction of association is reversed to that expected for urban sites, suggesting that other factors dominate the loading signal from urban areas. The relationship of SRP with %agriculture is strongly influenced by two extreme observations; when these are omitted from the analysis, the relationship disappears ($r^2 = 0.0015$). The link between industrial land cover (almost exclusively Gainesville Regional Airport) is significant and positive. Further examination of the mechanism via which P is enriched in water from those land uses is explored later. Note also that the explanatory power of land use for P concentrations is low, despite statistical significance of trends. We also note that the projected concentrations given 100% cover of each land use are 90 ppb for forests, 540 ppb for agriculture, 25 ppb for urban and 450 ppb for industrial. This inference is runs counter to conventional wisdom about P loading from human activities, and is accompanied by significant residual uncertainty that needs to be explained.

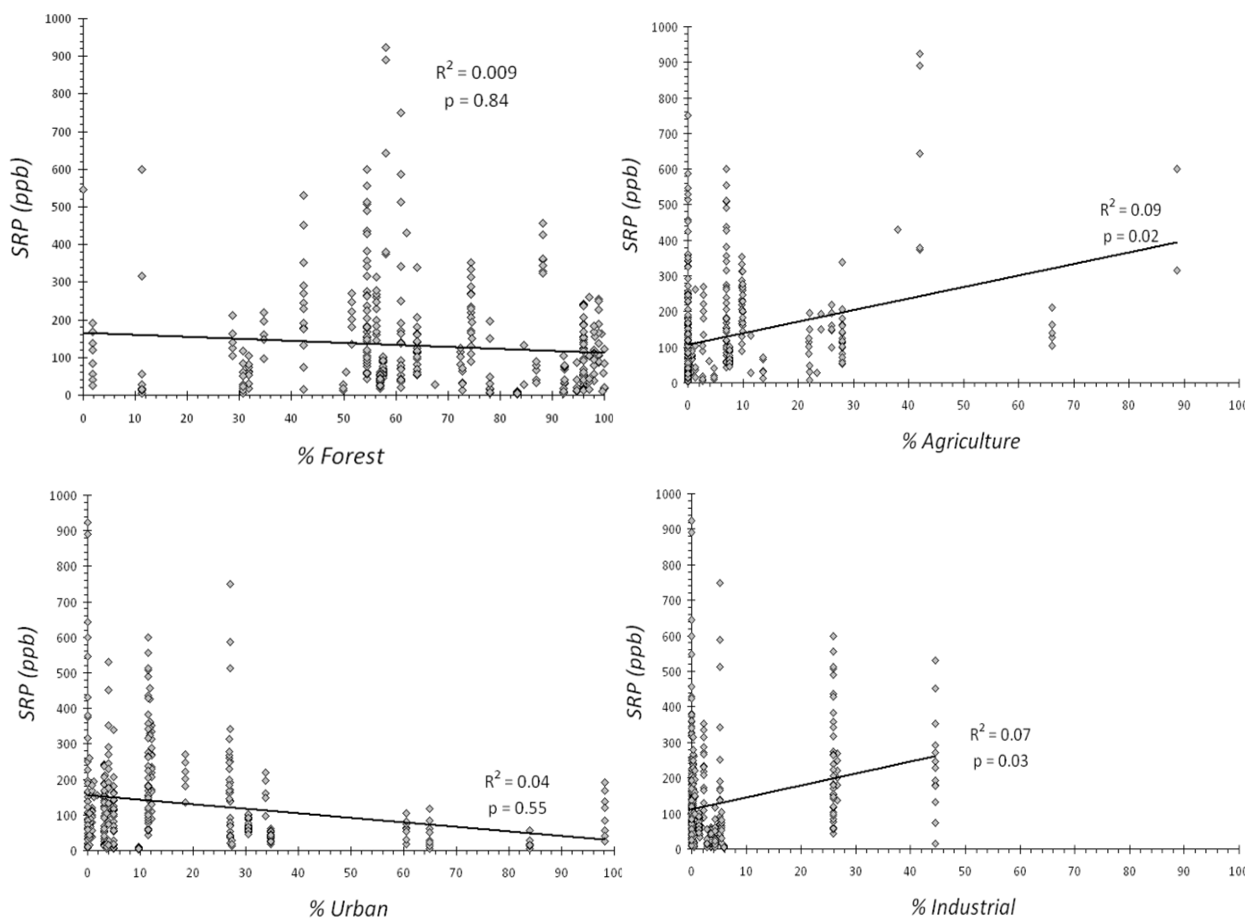

Fig. 27 – Association between % cover of a particular land use in an upstream accumulation area and values of SRP.

We conclude that land use does affect water quality, but the particular effects of land use on P in this watershed appear to be weaker than expected, and inconsistent with expected patterns. Control of confounding factors may be more important in efforts to attain P reduction targets. The reason for a weak land use vs. P load relationship is unlikely the be that land use practices are markedly better here than in other Florida watersheds; it is more likely that P loading from land uses is occurring as normal (i.e., higher levels from higher intensity uses) but that that signal is being overwhelmed by an anomalous source of P.

We also conclude that N is behaving as expected in this watershed; concentrations of DIN (NO_x + NH_4) are generally very low (25% - 75% quartiles for NO_x = 17 – 130 ppb; for NH_4 = 30 to 50 ppb), and most of the N is in dissolved organic form, particularly from forested systems. Low concentrations and principally organic forms are consistent with the relatively low intensity of land use in the watershed. The story for P is not well described by variability in land use. None of the land uses, with the exception of industrial are significantly correlated with SRP concentrations, and while LDI is correlated, the direction of association is reverse to expected. Therefore, we conclude that using land use specific loading information is unlikely to fairly attribute responsibility for meeting the P TMDL, and by extension (since internal N fixation is driven by excess P), potentially alleviating N concentrations and downstream loading as well.

Question 2: Assessing the Geologic Source of P

The second question, strongly related to Question 1, deals with identifying the source of the P load, and in particular assembling the evidence for or against a primary role for geologic P as a source. There are multiple overlapping lines of evidence that can be used to address this question, including the failure of land use to provide effective prediction, reasoning that LDI has been a useful proxy for water quality elsewhere, and its inability to predict P dynamics in the Newnans Basin suggest some other control of P loading. We explore three lines of evidence in this section.

Evidence Line 1: N vs. P

The first deals principally with covariance between N and P, previously mentioned but not fully explored. The rationale for this line of evidence is similar to the one for LDI. Were the P of anthropogenic origin, there is an expectation of covariance with N, regardless of whether the source is fertilizer runoff or wastewater. Anomalies in the covariance between N and P, and further analysis of the N:P ratio for water samples, offers insight into the magnitude of anthropogenic controls on P, and by association, the potential for other sources to dominate loading.

The first effort is to ascertain the correlation structure between NO_x and SRP. We focus on the mineral forms of the nutrients because a) they have more significant ecological

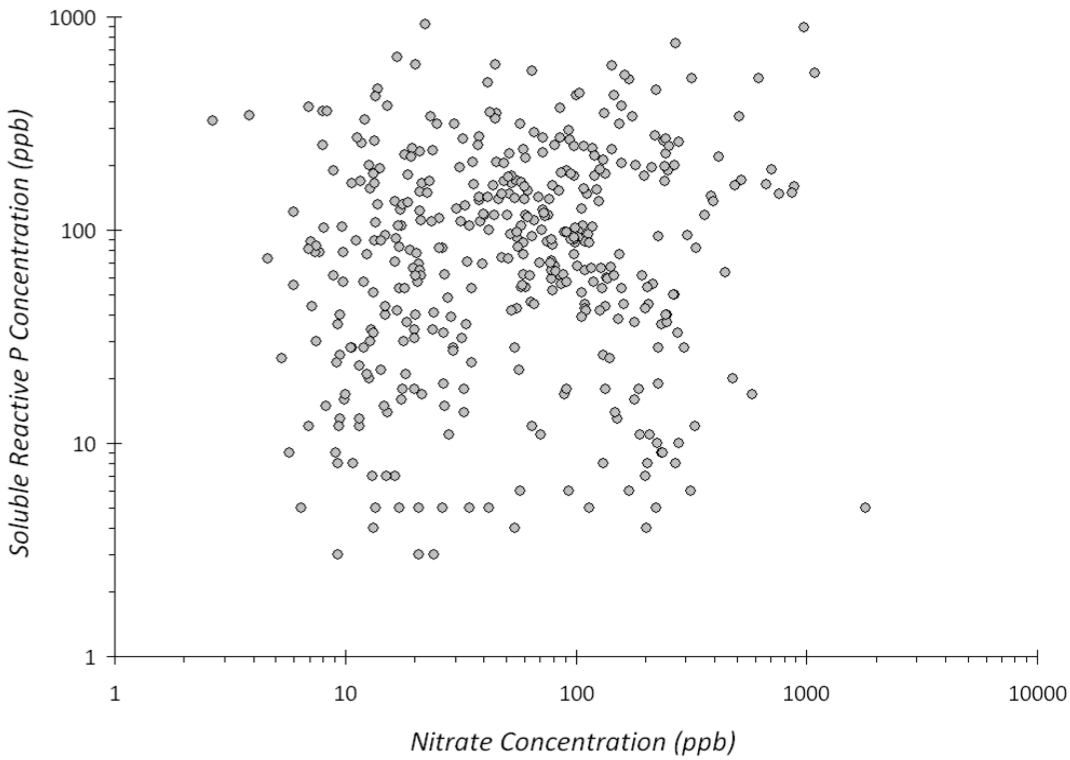

Fig. 28 – Covariance between NO_x and SRP concentrations across the entire lake basin. The Pearson correlation coefficient was r = +0.03 (p = 0.24).

impacts and b) they are the more likely forms of anthropogenic loading. It is important to note that the TMDL is written for total P and total N, and that organic forms can, over time, become available to fuel algal growth. Note the relatively small fraction of total N in mineral form (Fig. 26F) and the extreme fraction of total P in mineral form (Fig. 26E). Fig. 28 summarizes the global relationship across all samples collected in the basin, indicating that, at that coarse scale, the association is entirely absent.

Notably, the mean concentrations in the cloud of data points in Fig. 28 suggest nearly equivalent concentrations of both, a condition that is atypical in Florida. However, while global correlations are useful indicators, individual sites in different geologic and land use regimes would naturally be expected to exhibit different relationships. Fig. 29 presents the NO_x vs. SRP relationship for particular land uses.

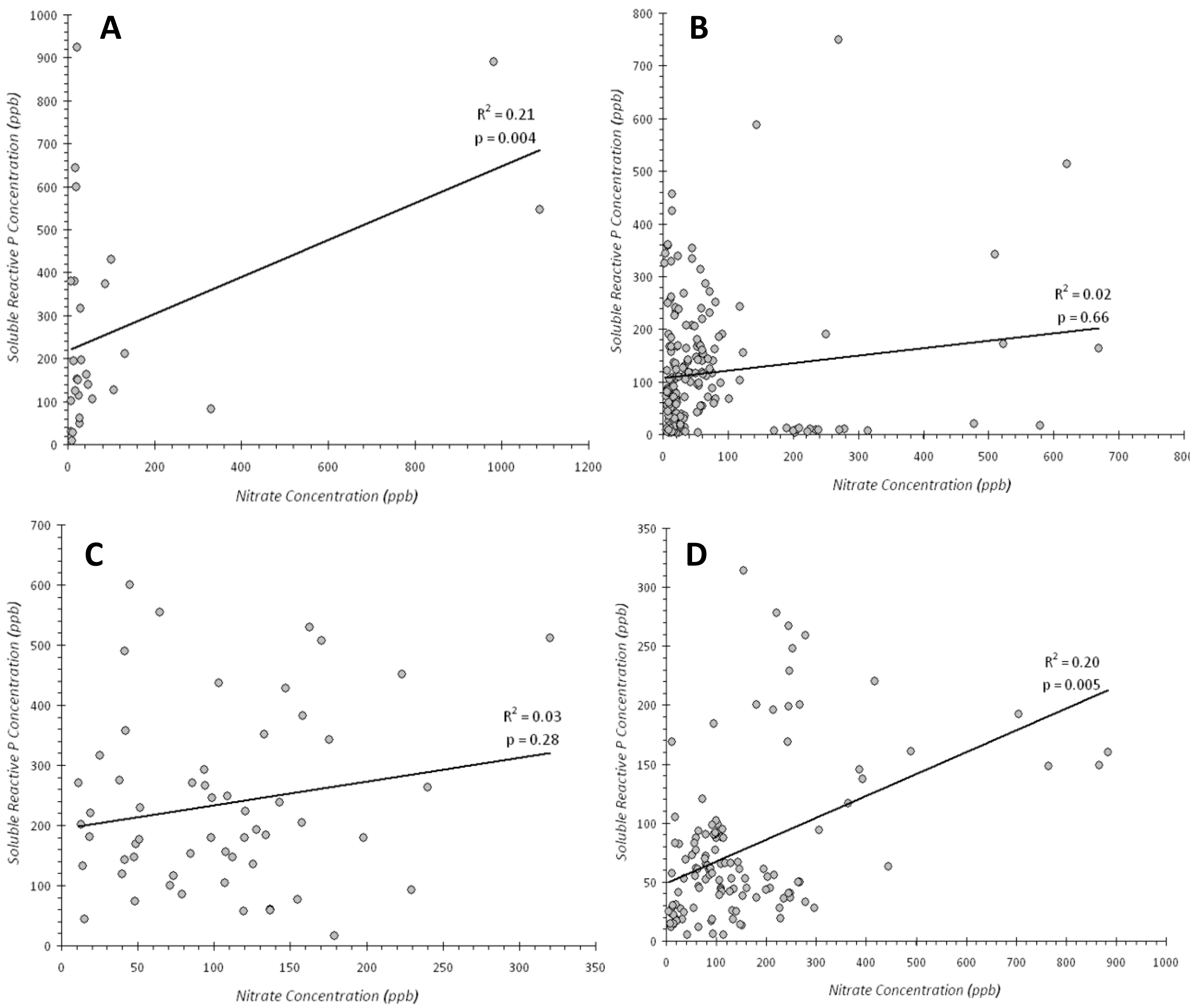

Fig. 29 – Covariance analysis between NO_x and SRP for stream sampling locations with differing upstream dominant land use: A) agricultural sites, B) forest sites, C) industrial sites, and D) urban sites. Mean values are 118, 66, 149 and 103 ppb for NO_x, respectively, and 268, 117, 235 and 76 for SRP, respectively. Note the different scales for both x and y variables.

We observe strong positive associations between NO_x and SRP for agricultural sites (though two points in the regression are highly leveraged) and for urban sites. For industrial and forested sites, despite comparable axis scales, the covariance is not significant, and knowledge of NO_x concentrations predicts a vanishingly small fraction of variance in SRP.

In contrast, when the data are partitioned by landscape position (and particularly within subbasin), the relationships become much clearer. The association between NO_x and SRP in headwater areas is strong in Little Hatchet Creek (an area dominate by urban land cover); this association disappears in the mid-reach sampling locations (Fig. 30). Similarly, in Hatchet Creek there is a weak though significant positive association between NO_x and SRP in the headwater sampling sites, but a non-significant trend reversal in the mid-reach sites (that is, SRP and NO_x covary negatively) (Fig. 31). Clearly, where a sample is collected within the landscape matters for water quality. It is also important to note that the concentrations of NO_x are significantly higher in headwater sites, and P concentrations are significantly higher in mid-reach sites suggesting dramatic changes in both relationships and magnitudes. This is particularly pronounced for Hatchet Creek (Fig. 31). Also notable, and the subject of further discussion below, is that the concentrations of both SRP and NO_x are attenuated within the terminal wetlands at the bottom of Little Hatchet Creek (Gumroot Swamp).

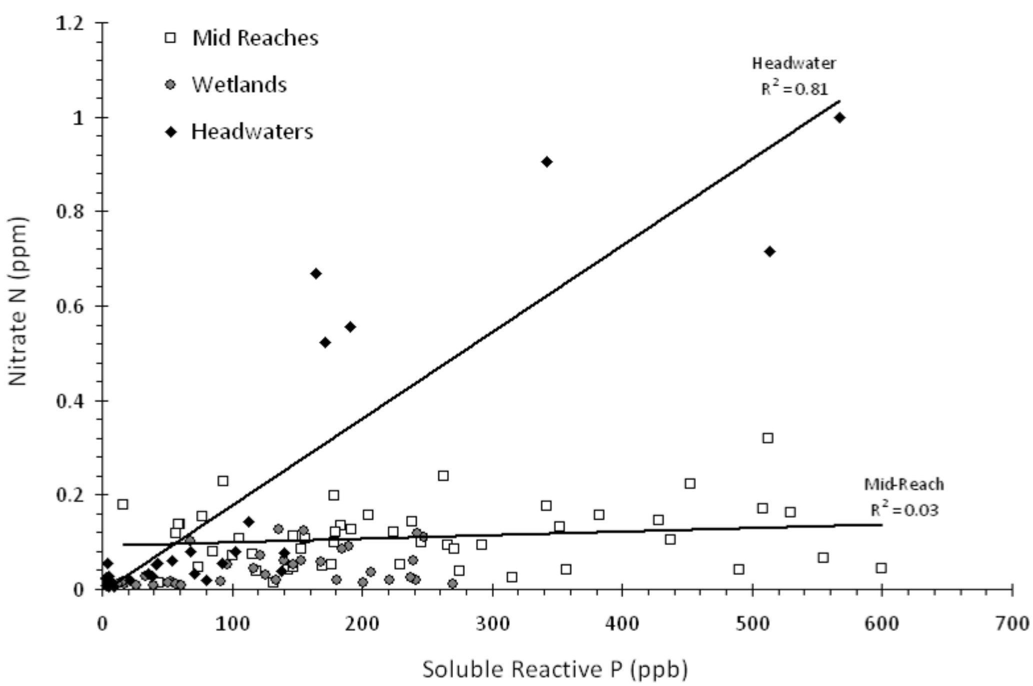

Fig. 30 – Covariance between NO_x and SRP for stations in Little Hatchet Creek, partitioned by landscape position.

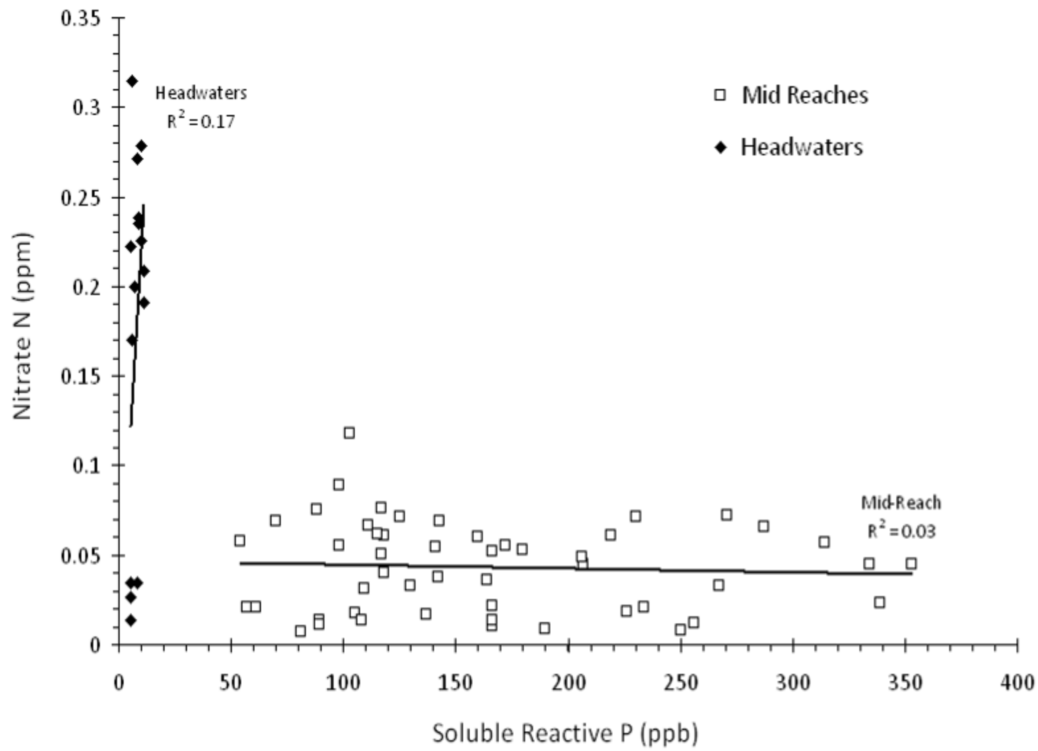

Fig. 31 – Covariance between NO_x-N and SRP for stations in Hatchet Creek, partitioned by landscape position.

Finally, we examine the nominal concentrations of NO_x and SRP in the different portions of each main sub-basin. Fig. 32 summarizes the concentrations NO_x as a function of basin and landscape position, and illustrates the dramatic decline in NO_x concentrations observed between headwater sites and mid-reach sites in Hatchet Creek. A similar though less striking trend was observed for Little Hatchet Creek. For Lake Forest Creek, in contrast, NO_x concentrations increase with passage from headwater to midreach sites, a phenomenon possibly due to changes in land use and channel morphology (ditches to open channels). Also notable are the dramatic and statistically significant declines in NO_x between the mid-reach sites (upstream of terminal wetlands) and samples collected within or below those terminal wetlands. This is particularly pronounced for Little Hatchet Creek after passage through Gumroot Swamp.

The same information for SRP (Fig. 33) illustrates a uniform pattern of P enrichment between sites in headwater locations and sites in mid-reach positions. This effect is most pronounced for Hatchet Creek but is statistically significant for all three sub-basins. It is also notable that concentrations in most developed basin (Lake Forest) are generally the lowest and that the concentrations in Little Hatchet increase in the region of the watershed where land use intensity actually decreases, suggesting that whatever controls P concentrations is operating independently of land use. Finally, it is important to note declines in SRP between mid reaches and wetlands (significant in the case of Little Hatchet Creek, not so in the case of Lake Forest). The durability of this apparent treatment effect is a subject that requires additional study.

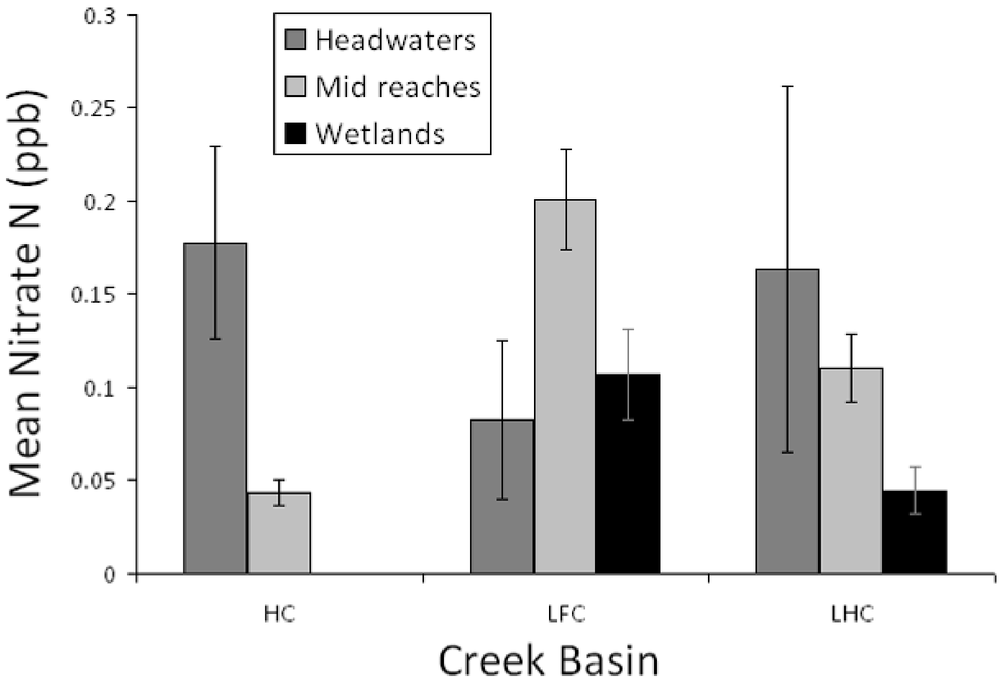

Fig.32 – Breakdown of NO_x-N concentrations by basin and landscape position. Note the dramatic attenuation of concentrations after passage through terminal wetlands in Lake Forest and Little Hatchet Creeks.

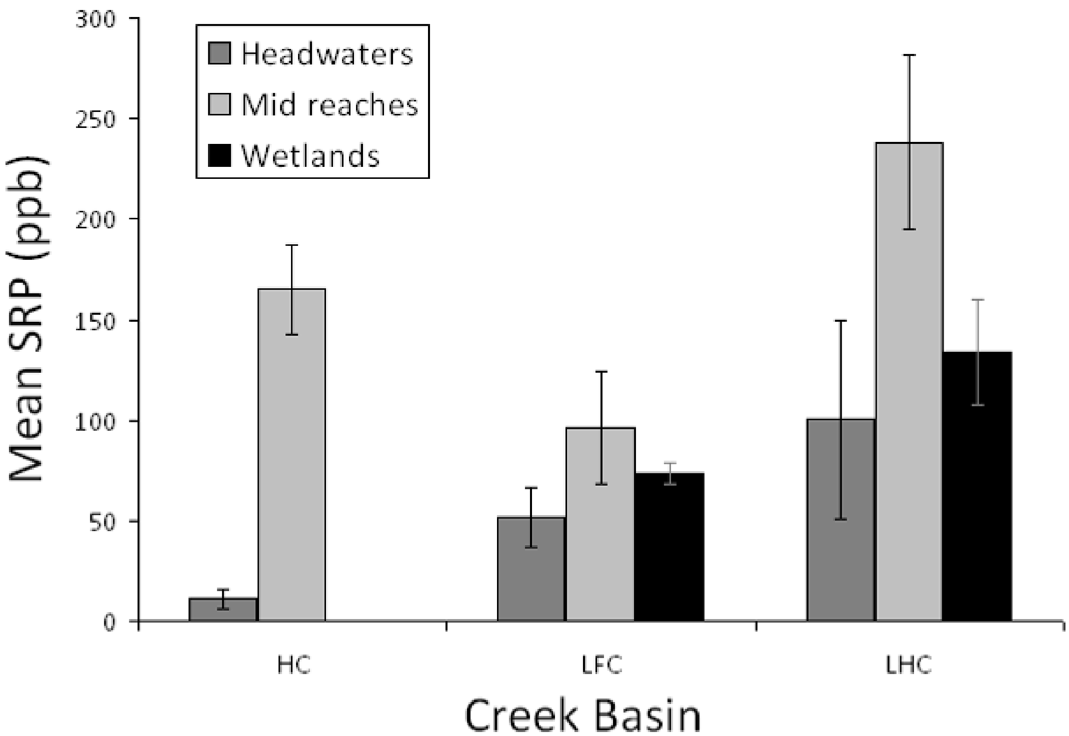

Fig. 33 – Breakdown of SRP concentrations by basin and landscape position. Note the lowered concentrations following passage of creek water through the terminal wetland systems in Lake Forest and Little Hatchet Creeks.

In considering the N and P dynamics in the watershed, it is fruitful to consider the N-to-P ratio as a measure of independent vs. covarying enrichment of nutrients. It is clear from event mean concentration data compiled from numerous land uses in Central Florida (Harper 1994) that the N:P ratio varies strongly with land use type and, to a lesser extent, intensity (Fig. 34). Typical values range from nearly 25:1 for open space to 5:1 for row crops. This is for total nutrients. However, in black water systems like the creeks in the Newnans Lake watershed, N and P are both controlled to a large extent by dissolved organic nutrient fractions that are poor predictors of eutrophication effects. For the Newnans watershed, N:P ratios for total nutrient range from a high of 150:1 at station 20 to a low of 3.1 at station 25 (Fig. 35). However, these values are strongly dominated by the organic fraction, and may not be the most illustrative measure of N vs. P enrichment because they are basically predicated on DOC loads, which have been shown to vary widely across the basin.

Instead, we focus on the molar ratio of mineral species (SRP vs. NO_x+NH_4) as a metric of enrichment, reasoning that these are the active component likely to drive ecological processes in the lake. Fig. 36 shows these values along with mean values at each site for DON and DIN. We assume that values less than 5 are low for the mineral nutrient ratio. Note that the fraction of DIN declines along with the N:P ratio, and that most of the sites with low N:P ratios have extremely high P concentrations, high F concentrations, and are situated in areas expected to be proximate to the Hawthorn Formation.

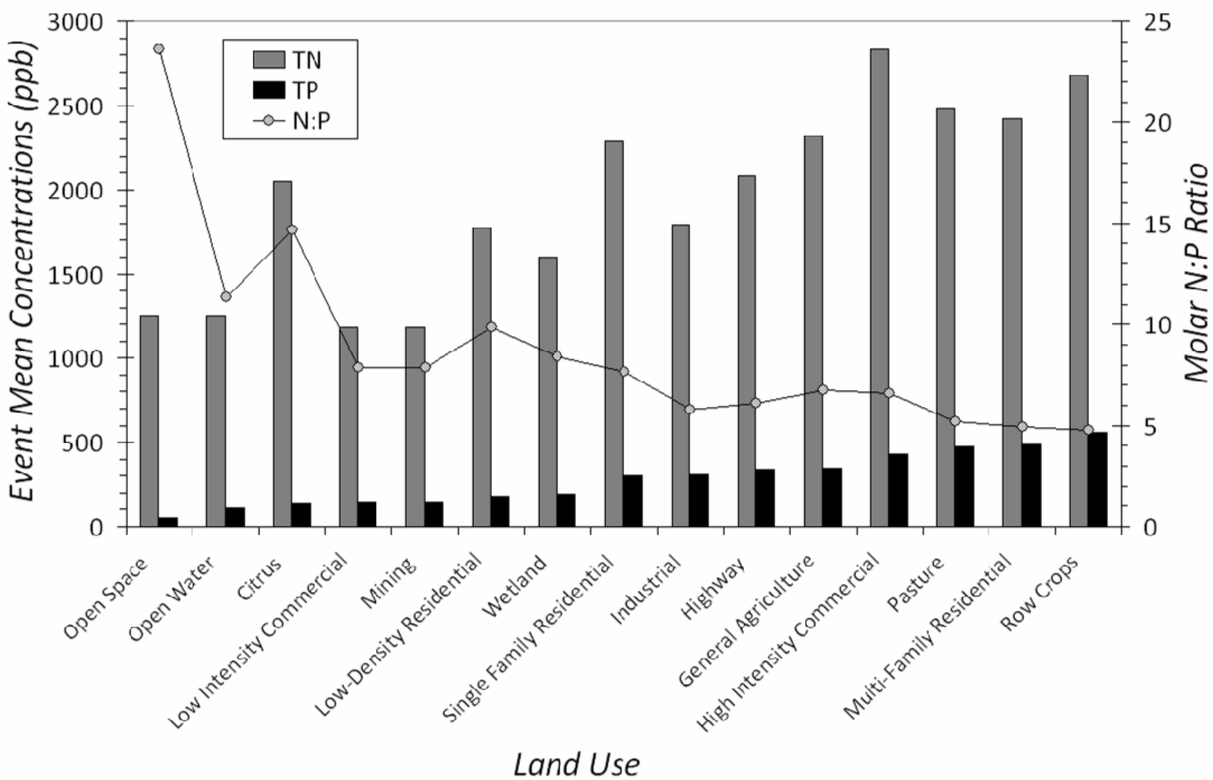

Fig. 34 – Expected N:P molar ratios for different land uses based on Event Mean Concentrations (EMC) for Florida stormwater runoff (Harper 1994).

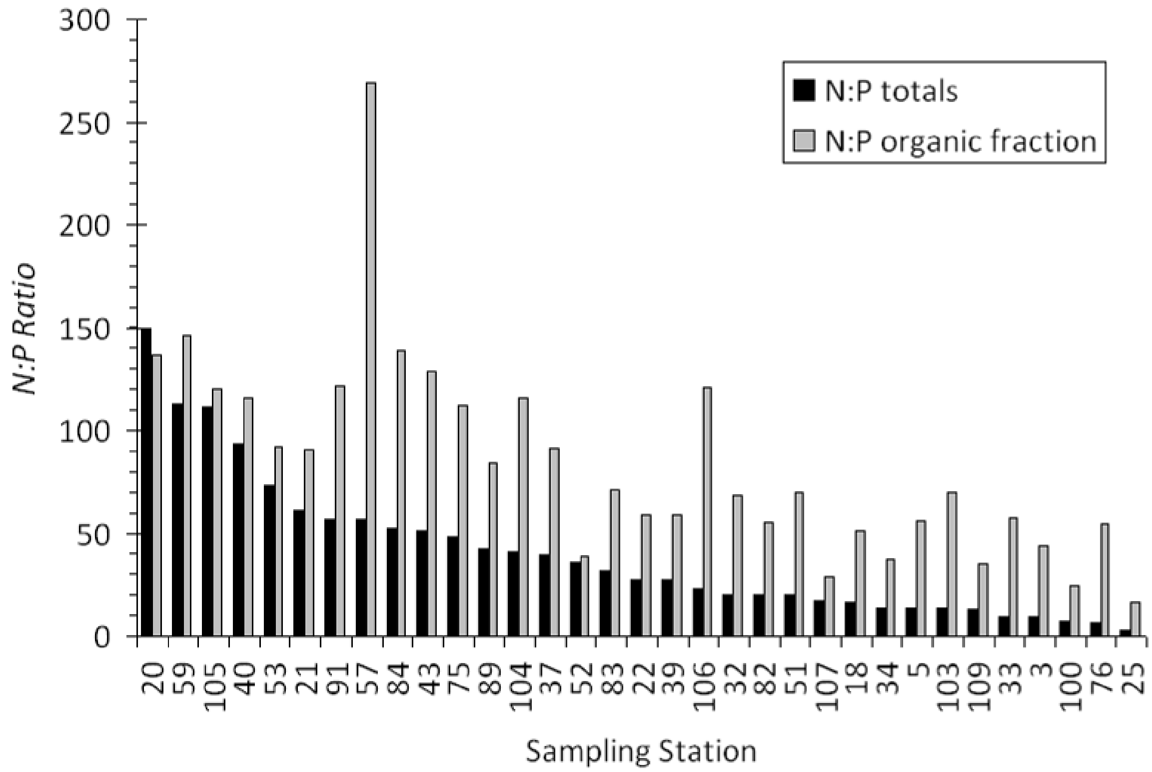

Fig. 35 – Mean N:P ratios by station for total nutrients (mineral + organic fractions) and for organic fractions alone. Note that the magnitude of the total N:P ratio is strongly influenced by the organic fraction (r = +0.63) which is less ecologically reactive.

We conclude that N:P ratios in the lake are atypically low when considered on a mineral species basis, and moderately low when considered on a total basis. Values for over 65% of the sites are below 5, and the extreme sites (3, 39, 76 and 25) have values less than 1 (i.e., more mineral P than N). Notably, N:P values for dissolved organic fractions are both highly variable (ranging from 20 to 270) and strongly associated with landscape position, with lower organic N:organic P ratios in sites with strong evidence of geologic P loading. Moreover, total N:P and organic N:P ratios are strongly correlated (r = +0.63) while the mineral fraction appears to be independent of the organic fraction. This all suggests that landscape sinks and conversion sites within the landscape are operating to varying degrees, but that mineral sources of nutrients are operating somewhat independently of that process. Since mineral P constitutes more than 65% of P for most sites in the basin, processes that regulate mineral P are of paramount importance. It is clear from the N:P ratio based on mineral species that mineral P loading is anomalous, and that it occurs in areas proximate to the Hawthorn Formation.

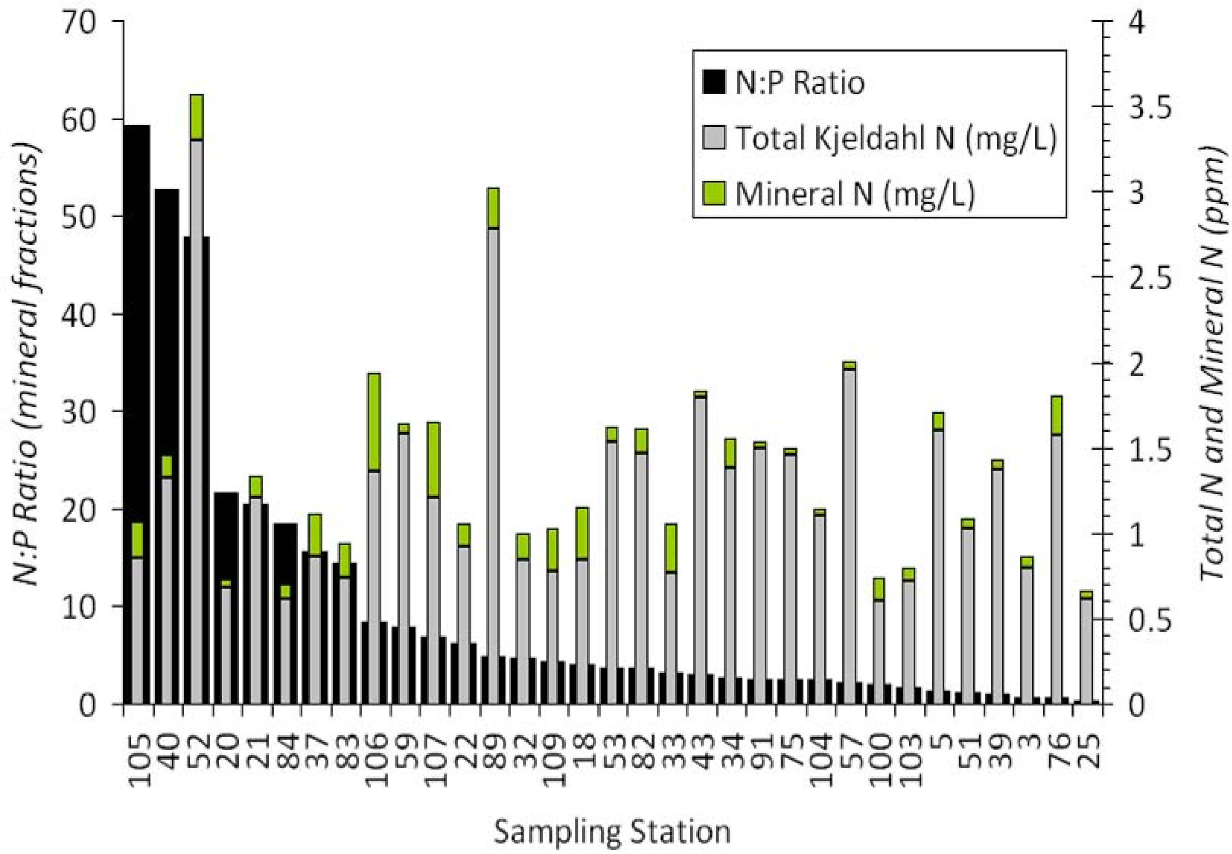

Fig. 36 – Molar N:P ratios for mineral nutrients (i.e., neglecting organic fractions associated with recalcitrant dissolved organic matter) and mean TN and NO_x values for a subset of sites with multiple observations. N:P ratios less than 5:1 are considered low.

<u>*Evidence Line 2: Proximity to the Hawthorn Formation*</u>

The Hawthorn Formation exerts critical control over the surface water quantity in Florida by regulating the connectivity between surface and groundwater. The effects of the Hawthorn on regional water quality are not nearly as well studied, but in lake regions like the northern highlands where Newnans Lake (also Orange and Lochloosa) is located, proximity to the Hawthorn and the general abundance of P are likely to be linked.

To test that effect, we explored the sampled data in several ways. First, evaluated the relationship between the depth to the Hawthorn (quantized into shallow – less than 2 ft below the land surface, and deep – more than 2 feet below the land surface) and various analyte observations in a global sense, that is not controlling for land use, basin etc. Fig. 37 illustrates the dramatic and strongly significant ($p < 0.001$) effects of Hawthorn depth on SRP concentrations (76 ppb vs. 161 ppb), NO_x concentrations (132 ppb vs. 85 ppb) and DIN:SRP ratios (11.4 vs. 2.8). Effects on TKN are, we surmise, principally due to inclusion of terminal wetland sites that are in positions in the landscape where they are proximate to the Hawthorn, and that are responsible for large quantities of TKN generation (and NO_x removal).

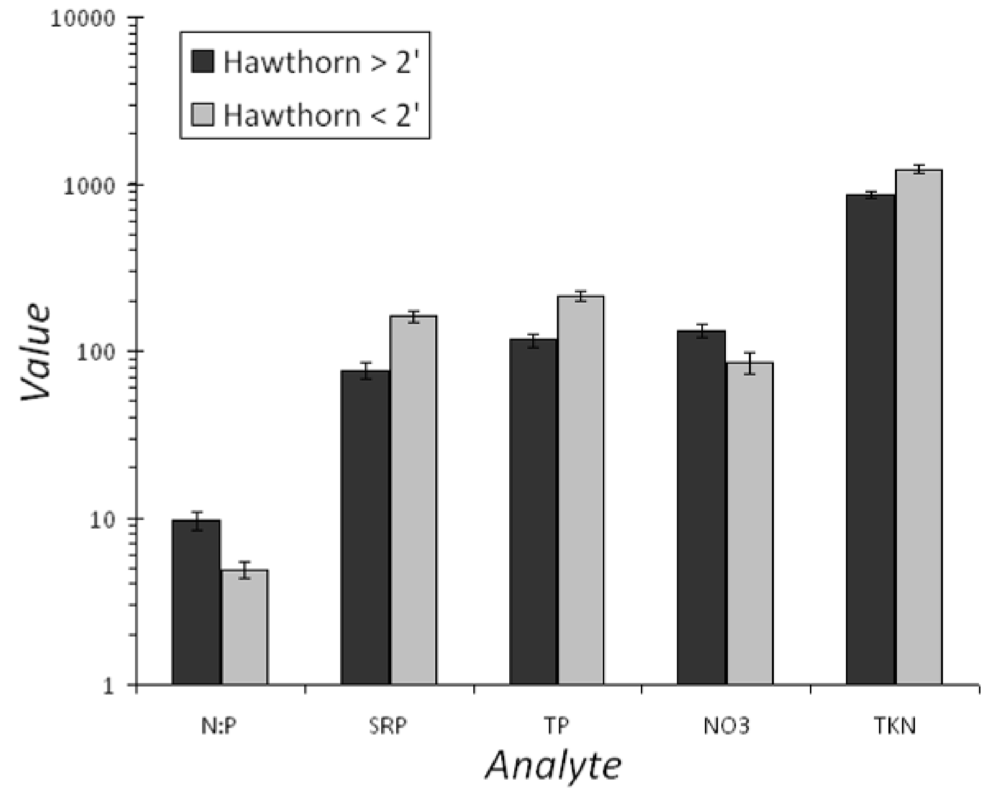

Fig. 37 – Summary of the effect of the Hawthorn Formation on surface water chemistry. All effects are significant at $p < 0.001$.

We also explored the effect of Hawthorn depth on SRP concentrations as a function of landuse. To start, Fig. 38 illustrates the dramatic effect of a continuous measure of Hawthorn proximity (i.e., not quantized as before) on SRP concentrations in that location. Indeed it's clear that any land use signal is swamped by the influence of geologic controls, with the exception that agricultural sites in proximity to the Hawthorn appear to have higher concentrations than other land uses. Moreover, the use of the 2 foot threshold for depth appears to be a viable option; the change point for the influence of Hawthorn depth appears to be between 0 and 5 feet.

Despite the clear effect of geology, there remain numerous sites in proximity to the Hawthorn that do not exhibit elevated P concentrations. There are several explanations for this, all of which cannot be answered with current data. First, it is highly likely that the Hawthorn Formation elevation map from which these depths are derived is not accurate at the fine scale. That is, there is some significant uncertainty that arises from the use of a coarse map for local scale assessment. Inquiries with the Florida Geological Survey (R. Copeland, personal communication) confirm that the map that was used for these analyses is the best information available. Further work on mapping the depth, thickness and elemental content of the Hawthorn Layer may prove useful for water quality management. Second, the Hawthorn Formation is a complex assemblage of interspersed layers of sand and clay. Given the spatial and vertical variability expected in the Hawthorn, it is not surprising that some areas where the land surface and the Hawthorn intersect are dominated by minerals that are not SRP rich.

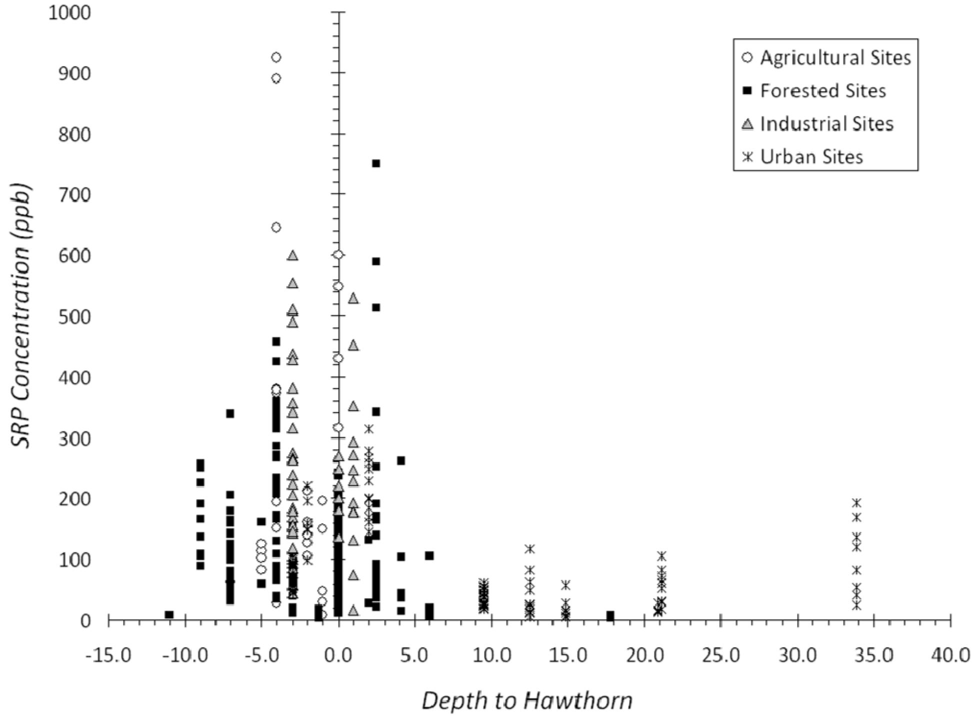

Fig. 38 – Plot of Hawthorn depth effect on surface SRP concentrations divided by land use.

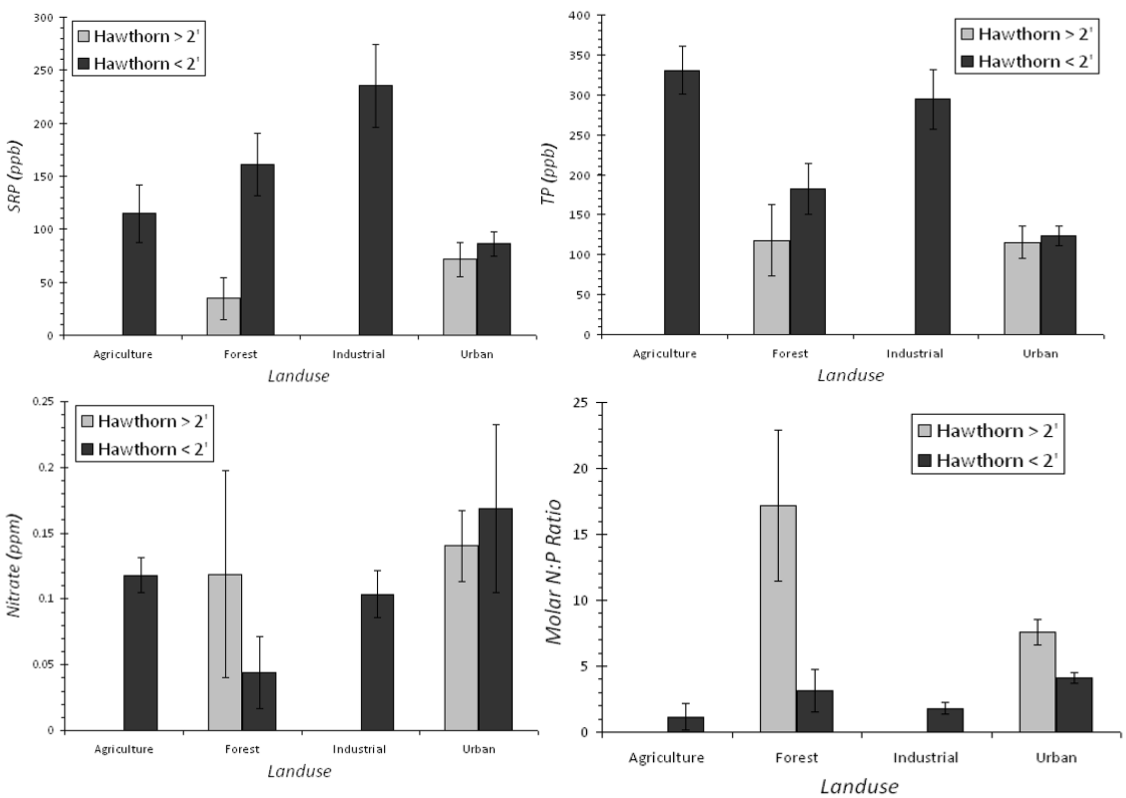

Fig. 39 – Summary of water quality observations stratified by land use and Hawthorn depth. Note that all land uses have molar N:P ratios considered unusually low when the Hawthorn is close to the surface.

Finally, we examined the interactive effects of land use and categorical Hawthorn depth on SRP, TP, NO_x and DIN:SRP ratios (Fig. 39). This reinforces previous observations that, while the effects of land use are significant, they are minor compared the variance reduction that can be obtained by knowing where a site lies with respect to the geologic store of P. SRP concentrations increase a factor of 5 in forested areas between sites far from and close to the Hawthorn. Notably, however, the effect of Hawthorn depth on urban lands is lower; we speculate based on anecdotal sampling observations that the map of the Hawthorn depth is particularly erroneous in the Lake Forest Creek basin where the Hawthorn appears to be far deeper than is predicted from the interpolated elevation map. Further inquiry from existing geological stratigraphy information for the county would help clarify this uncertainty. The effects of Hawthorn depth on DIN:SRP ratios are even more pronounced, with values in forested areas declining from 17.2 to 3.2 with this change in geologic proximity. Overall, we conclude that Hawthorn depth is a powerful predictor of P dynamics in the water, suggesting that any effort to ameliorate P loading to the lake will need to consider landscape position.

Evidence Line 3: Covariance with Hawthorn weathering by-products

Another similar line of evidence, but one that does not rely on ancillary site knowledge, is covariance with by-products of apatite weathering. Among the candidates are calcium (apatite is partially $CaPO_4$). Despite the plausibility of this option, Ca is both abundant in other non-phosphatic minerals (not least of which is limestone). Another option derives from the observation that much of apatite in the Hawthorn is fluorapatite, a mineral in which fluoride is abundant. Since fluoride is highly water soluble, as weathering degrades the mineral structure liberating soluble reactive P, it might also be expected to liberate soluble F. Moreover, F is comparatively easy to measure using ion chromatography.

Despite ease of measurement and plausibility of covariance with geologic P and not with fertilizer P, it remains unclear how much P in the Hawthorn is associated with fluorapatites, and how much is not (hydroxyl apatites and other P bearing minerals that do not contain F may also be dominant). It was beyond the scope of this work to determine the answer to the question of spatial variability in the F vs. P relationship in the Hawthorn, but we proceeded with testing this relationship assuming that any spatial variability would occur on a relatively small scale, and thus be integrated out when considered on the large scale that water samples represent.

In addition to uncertainty about the covariance of P and F in the Hawthorn materials, there is a strong potential confounder of F concentrations from municipal water supply, to which F is added for public health reasons. In urban areas, where leaking distribution pipes and lawn irrigation load F into the landscape, we expect to see weaker correlations between F and P because of anthropogenically elevated levels of F.

To analyze the data, we first summarized the F concentrations by subbasin and landscape position (Fig. 40). From this analysis it is clear that there is a modest increase in F from headwater to mid-reach sites in Hatchet (HC; statistically significant) and Lake Forest (LFC; non-

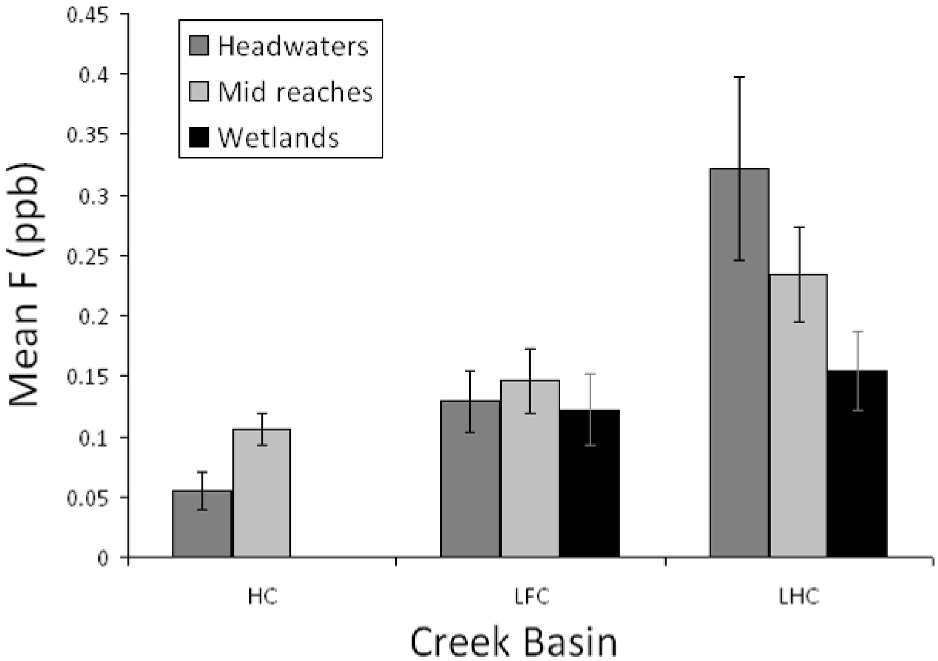

Fig. 40 – Fluorine (F) concentrations by basin and landscape position. The high levels of F in the headwaters of Little Hatchet Creek suggests that using F concentrations as a tracer of geologic P can be confounded by loading from municipal water supply.

significant) creeks, but a pattern that reverses the expected direction in Little Hatchet Creek (LHC). Indeed the highest overall F concentration (Fig. 41) is at Site 20 which is just downstream of the Murphree water treatment facility where F is added to the municipal water supply. Moreover, other mid-reach sites in Little Hatchet have ample opportunity for anthropogenic enrichment, including the Brittany Estates municipal wastewater plant effluent just above Site 18. It is also clear from Fig. 40 that if F is enriched as a result of Hawthorn weathering, the concentrations diagnostic of that enrichment are not markedly higher than concentrations observed with anthropogenic loading; as such, inference of geologic P source from F data alone needs to be done carefully.

We also examined F vs. SRP relationship as a function of sub-basin and station. Within each sub-basin we picked three stations (headwater, mid-reach, wetland/lower reach) for which sufficient samples were obtained to explore within site covariance. The results are summarized in Fig. 41 (Little Hatchet Creek), Fig. 42 (Lake Forest Creek) and Fig. 43 (Hatchet Creek).

In Little Hatchet Creek there is significant promise in the use of F as a tracer of geologic P (Fig. 41), subject to the cautionary requirements of dealing with a natural tracer that is, in some areas, substantially affected by human loading. What is clear is that the covariance between SRP and F is strong, particularly for site 100, which lies in an area predicted to be in close contact with Hawthorn clay material. The covariance is also strong for site 89 which is the discharge of Little Hatchet to the lake after passage through Gumroot Swamp. In contrast, however, the relationship between SRP and F is weak or non-existent at Site 20 (near the Murphree treatment facility) and at site 18 (near Brittany Estates).

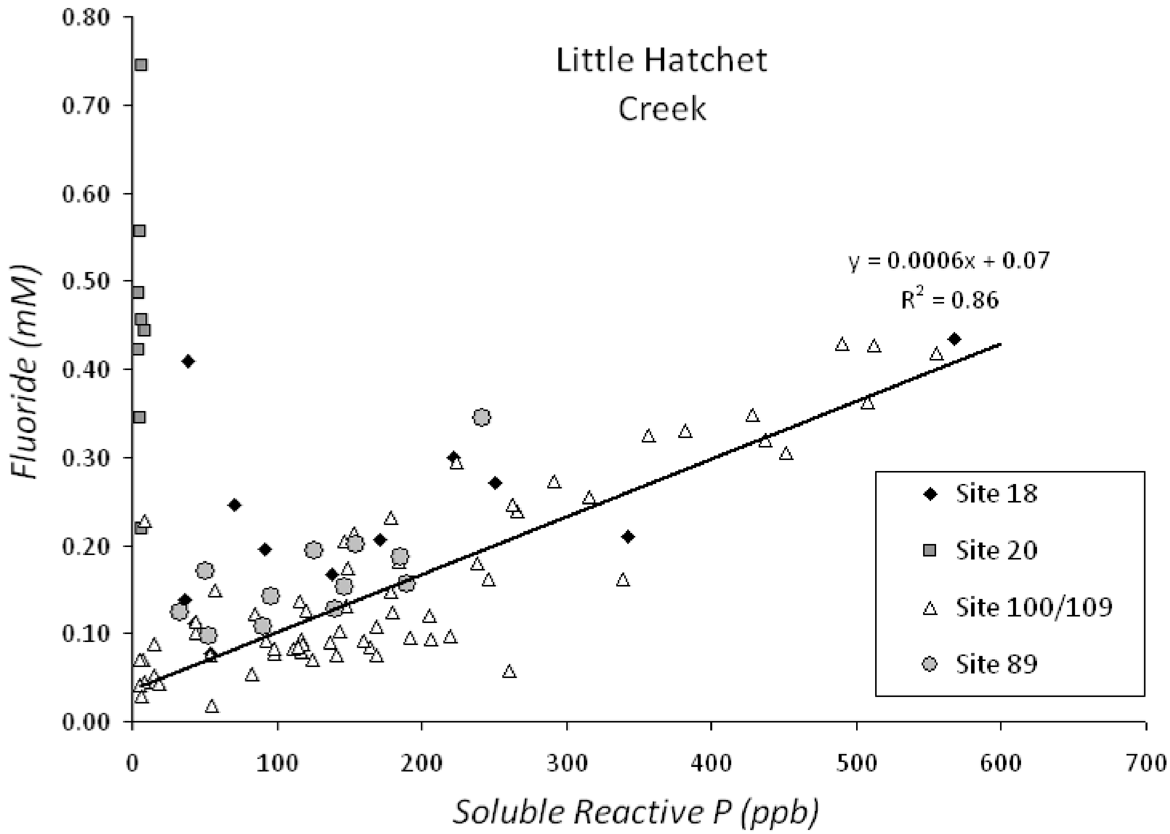

Fig. 41 – Evidence for geologic P as the dominant source in Little Hatchet Creek. Note that high values of F at Site 18 are likely due to proximity to the Murphree water treatment plant. The P-value for the overall regression is < 0.001.

The same patterns of covariance are not observed for Lake Forest Creek (Fig. 42). Indeed, if anything the global relationship between SRP and F is negative, possibly consistent with our contention that the headwater stations (Site 83 and 84) in Lake Forest Creek maintain steady baseflows due to leaking GRU water distribution pipes (this inference is based on the chemical properties of that baseflow water vis-à-vis tap water). In this case, the F concentrations are generally high and SRP concentrations generally low, reinforcing our contention that a) the P coming out of Lake Forest creek is not of geologic origin, and b) that maps of Hawthorn depth that depict close proximity of the Hawthorn to the land surface in this area of the basin are erroneous.

Finally we examined the same patterns in Hatchet Creek (Fig. 43), and observed, as in Little Hatchet Creek, strong positive covariance between F and SRP, with the added observation that the headwater site had low concentrations of both, meaning that the caveat about anthropogenic loading P may not be as problematic in this subbasin. Overall, for both Little Hatchet and Hatchet Creeks, F provides information that reduces over 80% of the variance in SRP, making it a highly useful tracer. Future analysis of surface and groundwater samples should be evaluated in accord with this geologic relationship.

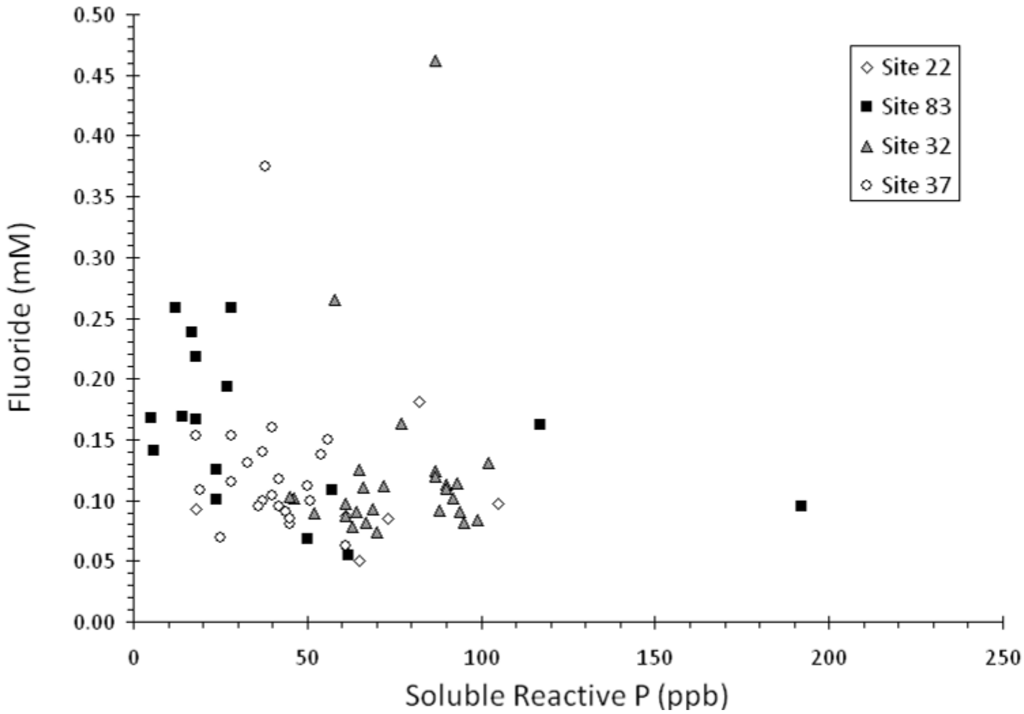

Fig. 42 – Covariance of SRP and F in the Lake Forest Creek watershed. Weak negative correlations (r = -0.06) suggest that a) the P in the basin is not of geologic origin and b) that high F levels may be from municipal supply pipe leaks. The correlation was not significant (p = 0.43).

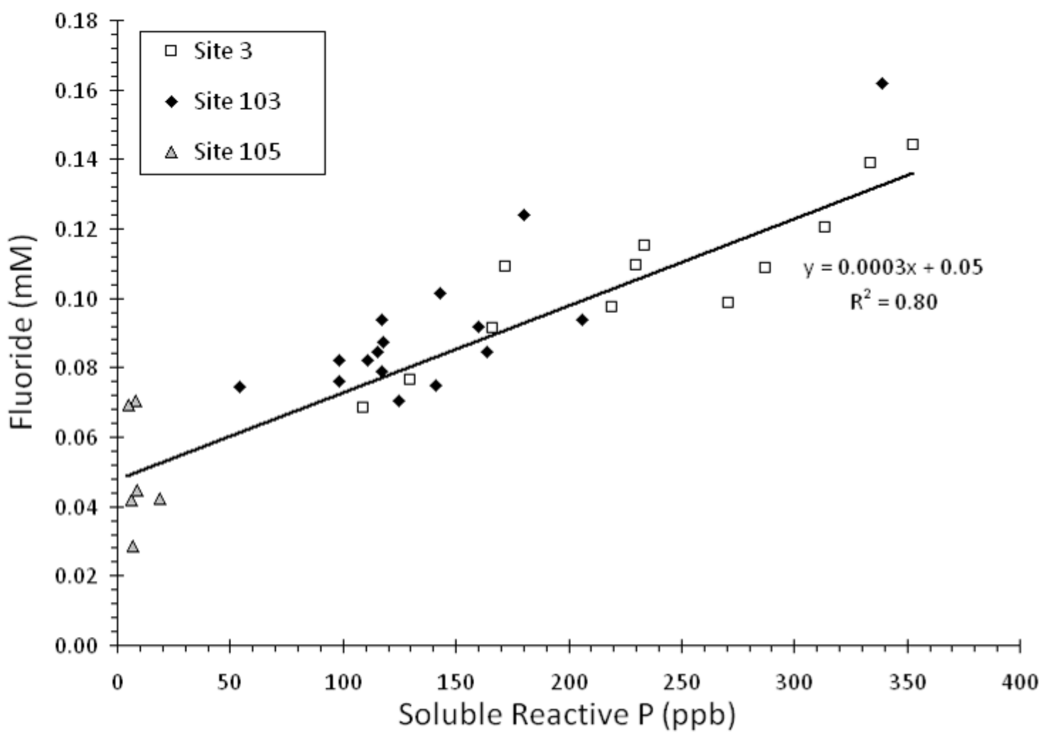

Fig. 43 – Covariance between SRP and F in Hatchet Creek (p < 0.001). Sites 103 and 3 both show strong positive covariance suggestive of geologic P loading, while site 105 shows weak correlation and low concentrations, suggestive of water not in close contact with the Hawthorn.

Question 3: Longitudinal Loading Patterns

The discussion of the sources and sinks of P has focused principally on the role of geologic P as a primary source in a basin that has insufficient land use intensity to maintain P concentrations like those regularly observed. Equally important from a management and amelioration perspective is likely to be the identification of loading hot-spots and hot-moments. While previous figures have shown that P concentrations vary strongly in space, and somewhat predictably with geologic location, and that there are strong effects of flow, the location of hot spots in the context of creek reaches still requires some explication.

The first effort is to show the temporal patterns in concentration for stations that span the entire length of the creeks. Fig. 44 summarizes the entire period of record for Hatchet Creek, with stations from the headwaters (Site 105) to the mid-reaches (Site 103). Insufficient observations below Site 39 were available because of the sinkhole effect (at Site 3) and the loss of water along the stream due to summertime ET. Indeed, there were only 3 observations of flow, all during storm events, at Site 53 (SR26 bridge) during the entire period of record.

Among the striking observations from the Hatchet Creek concentrations are the dramatic increases from headwater sites to downstream sites (particularly site 3 near the sinkhole which is generally the most concentrated). Moreover, there appears to be only a weak effect of flow on the concentrations; a nearly 30 fold increase in flow lowered the SRP concentration slightly, but far less than would be expected by dilution alone. This reinforced the previous observation that the flow dynamics of P in Hatchet Creek are strongly non-linear, with increases in

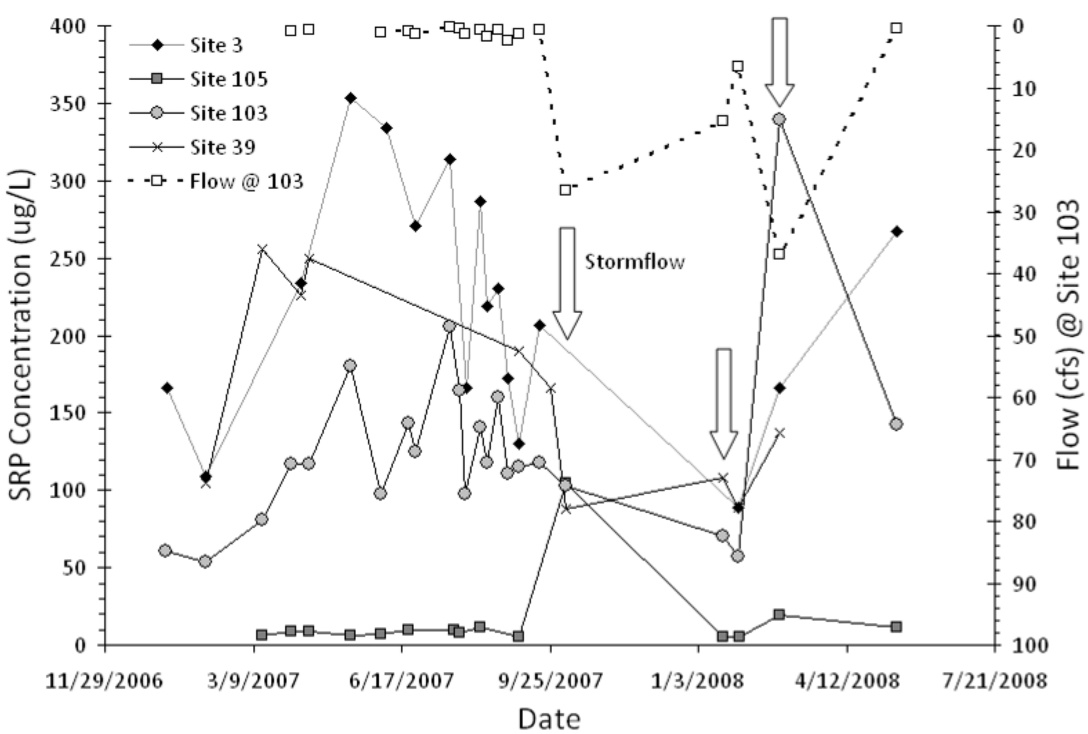

Fig. 44 – Flow and soluble reactive P over time at stations along Hatchet Creek.

concentration evident during stormflows. This is particularly pronounced at Site 103, which is proximate the tributary inputs that go from no flow during baseflow conditions, to small discharge with low P concentrations at moderate flow conditions, to comparatively large discharge with enormous concentrations during stormflow.

Patterns in Lake Forest Creek are similar in character for SRP concentrations (Fig. 45) but less dramatic. While there does appear to be a trend in SRP with distance downstream (headwater Site 83, mid-reach site 37, wetland site 32), the magnitude of the increase is ca. 40 ppb on average. There also appears to be a marked dilution effect; SRP declines with increasing flow, and appear to rebound to higher concentrations as flow returns to base conditions. Note again that the scale of the SRP axis is far smaller in this case than for Hatchet or Little Hatchet Creek.

Finally, the patterns observed in Little Hatchet Creek (Fig. 46) exhibit both enormous longitudinal increases in SRP from headwater sites (18) to mid reach sites (100 and 109), and then declines to wetland discharge sites (89), and also pronounced dilution effects, with the most dramatic effects of stormflow on concentration of any of the creek systems. For example, at Site 100 the storm that occurred in early June 2007 led to a decline SRP concentrations from over 550 ppb to less than 100 ppb, followed by a rapid recovery to over 450 ppb within 5 days. Moreover, an extended period of comparatively high flows in early 2008 yielded lower concentrations, and as flows declined during May and June of 2008, concentrations rose markedly in response. This suggests a comparatively small and invariant groundwater source of P that can be diluted by runoff.

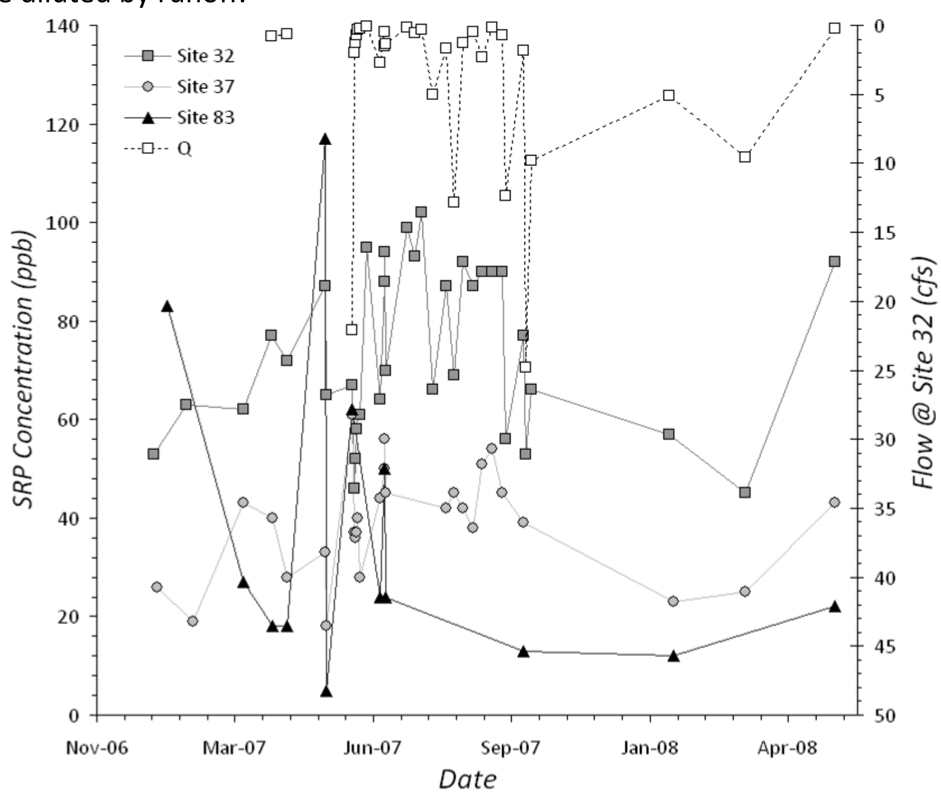

Fig. 45 – Flow and soluble reactive P over time at stations along Lake Forest Creek.

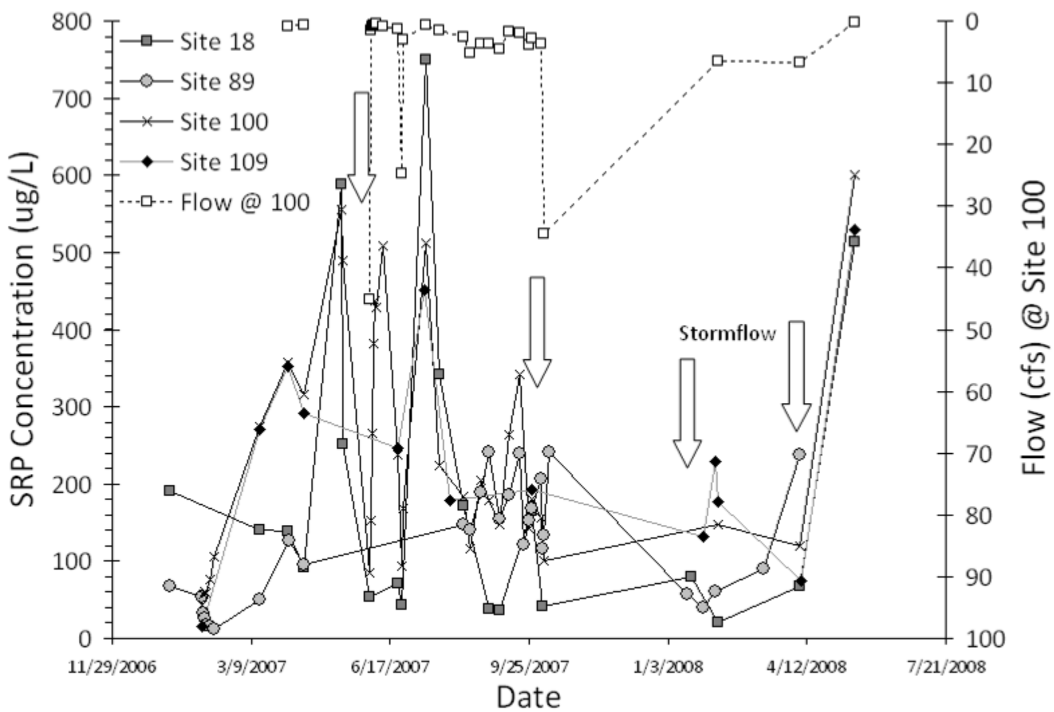

Fig. 46 – Flow and soluble reactive P over time at stations along Little Hatchet Creek.

This fixed and finite volume of P enriched water in Little Hatchet Creek is in contrast to Hatchet Creek where the pool of enriched groundwater appears to be larger, and mobilized both at baseflow (when concentrations are high), but also at peak flows (when concentrations are also high). This behavior, as has been mentioned before, has important implications for watershed flow management since increases in peak flows may mobilize enormous quantities of P.

A more straightforward view of the longitudinal patterns of P loading comes from sampling along two water quality transects (one on Hatchet Creek, the other on Little Hatchet Creek) during three hydrologic phases: extreme baseflow, moderate baseflow and stormflow. The results of the water quality analyses of these data are summarized in Figs. 47 – 52.

The first three figures pertain to Hatchet Creek. During low baseflow (Fig. 47), there was strong evidence of bank seepage into the creek; there were no flowing tributaries encountered during this transect despite the identification of channels that, under higher flow conditions, do appear to contribute flow. As such, the longitudinal increases in both concentration (from less than 10 ppb at Site 105 to more than 300 ppb at the sinkhole at Site 3) and P load are due to diffuse seepage from the surficial or intermediate aquifer into the stream. Since flow increases several fold over this reach, we infer that the seepage has a SRP concentration of in excess of 350 ppb. Moreover, we observe strong patterns in F concentrations that parallel the increases in SRP, confirming that this P is likely of geologic origin. The Hawthorn depth maps reinforce this contention since the Hatchet Creek channel is a clear area of shallow Hawthorn depths.

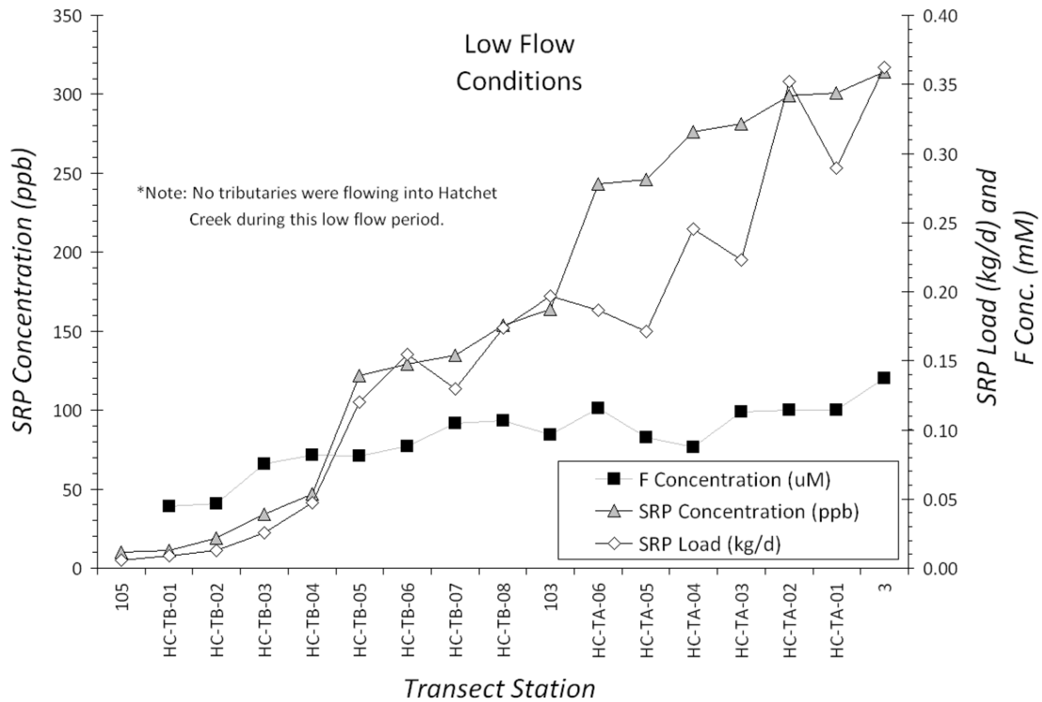

Fig. 47 – Transect water quality and loading profiles for Hatchet Creek during extreme baseflow.

The same transect during moderate flow conditions (Fig. 48), with the addition of several sites further downstream (Sites 39, 51 and 53) yields a similar picture, but more complex due both the effects of tributary inputs and wetland attenuation of both water and P concentration. Over the upper portion of that transect, the same pattern holds, with dramatic increases in both load and concentration. Tributary inputs are generally of low discharge and moderate P concentrations, and don't have a dramatic impact on the longitudinal pattern, with the exception of the inflow from Bee Tree Creek, which is both high flow and extremely low P. Notably, however, the specific conductance declines with distance downstream, a result that runs counter to the inference that the flow is from the intermediate aquifer. Unfortunately, F concentrations were not available. It is important to note that the total concentrations are comparatively low, peaking at less than 100 ppb, in stark contrast to the concentrations observed during extreme baseflow conditions. It is reasonable to invoke dilution to explain this since the flow rates are approximately 4 times larger.

Finally, we evaluated the same transect (as many sites as was feasible to sample) during peak flow conditions (Fig. 49). Interestingly, the dilution effect that was observed during moderate flow conditions was not observed under peak flows. Concentrations in the main channel were generally low, but the tributaries to the creek contributed larger quantities of flow, and enormous SRP concentrations (in one case as high as 814 ppb). As such, despite reduced effects of bank seepage along the creek, the concentrations of P were over 350 ppb at Waldo Road and declined principally in response to dilution from two creeks (one measured, the other inaccessible) that appear to load at very low concentrations. As such, there is a marked decline in both load and concentration between the mid-reaches near the sinkhole (Site 3) and the discharge at Site 53. Also notable is the distinct absence of a trend in F concentrations despite

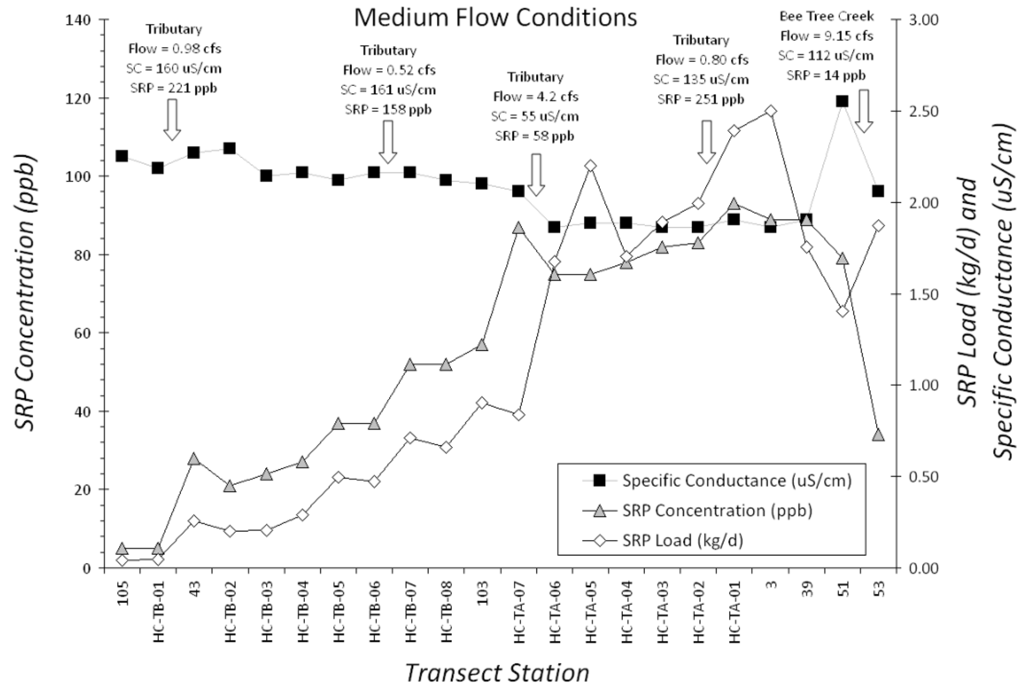

Fig. 48 – Transect water quality and loading profiles for Hatchet Creek during moderate baseflow.

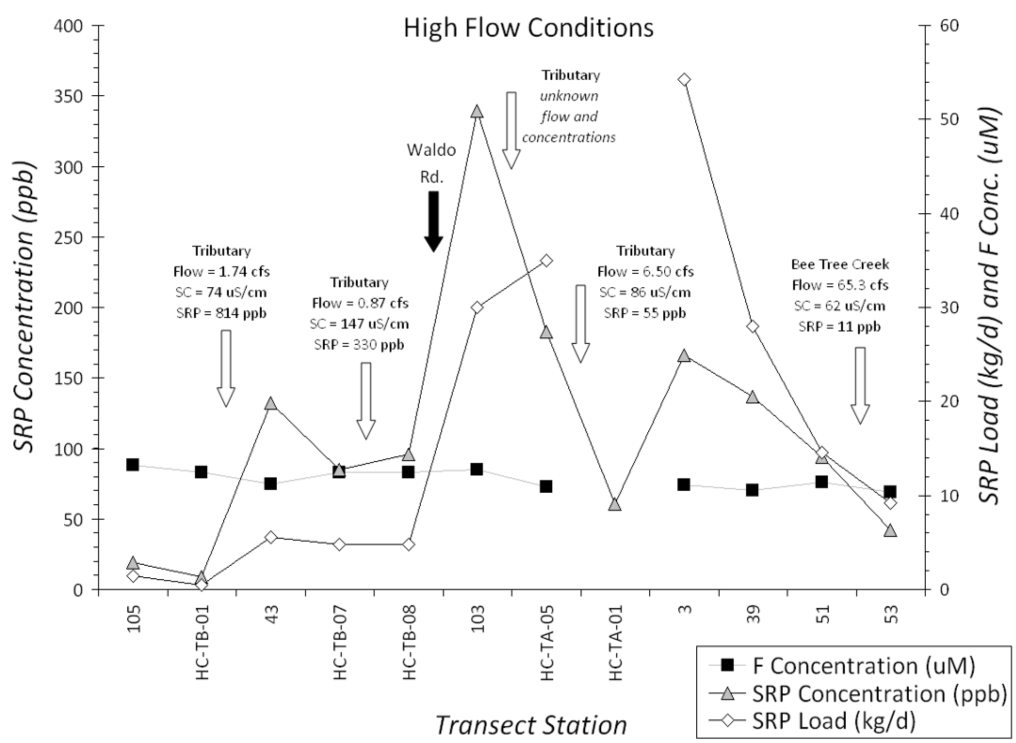

Fig. 49 – Transect water quality and loading profiles for Hatchet Creek during stormflow. Lines are discontinuous when samples were not obtained for intermediate stations.

comparatively dramatic variability in SRP concentrations and loads. It is possible that the P source into which stormflows tap is not geologic P (there is a large pasture at the headwater of the two tributaries that load heavily during stormflow) but rather anthropogenic P that is only mobilized when the hydrologic source area includes parts of the landscape where loading is high. Further work would be necessary to determine if that is indeed the process that is occurring. It is equally plausible that the creeks in the vicinity of Waldo Road are where active incision into the Hawthorn is occurring, but that this process is only happening during stormflows. Discriminating between these two alternatives is potentially critically important for the regional control of development and stormwater.

Another transect was executed in the Little Hatchet Creek basin, spanning a reach from the disparate headwaters of Sites 20 and 18 down to the creek discharge to the lake at Site 89. This transect was also sampled at low, moderate and peak flow. Fig. 50, which shows the longitudinal profile during baseflow conditions, indicates that there is a zone between Site 18 and Site 109 within which enormous loading occurs. This reach includes Gainesville Regional Airport. After a peak concentration at Site 18, level actually decline; note, however, that the concentrations are extreme (above 500 ppb). Below the Site 109, the loading declines systematically despite comparatively little change in concentration; this occurs principally because the stream is losing, with water seeping into the riparian forest (note that this transect was run during July when transpiration rates are at an annual maximum).

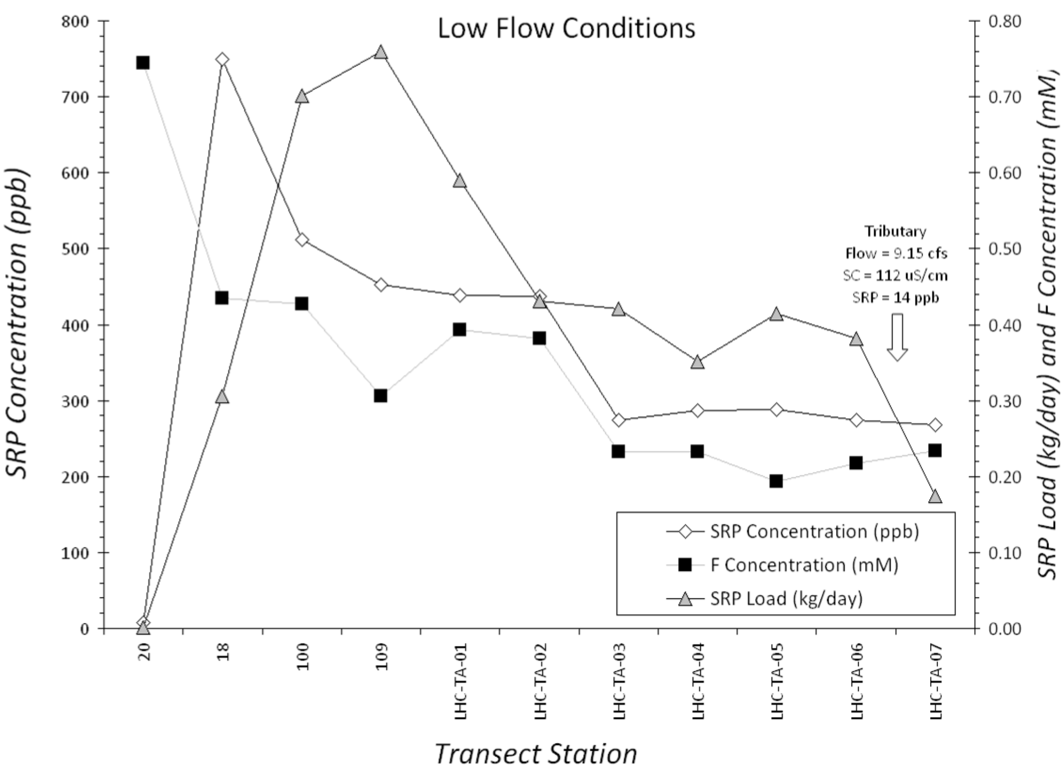

Fig. 50 – Transect water quality and loading profiles for Little Hatchet Creek during extreme baseflow.

The same transect during moderate flow conditions (Fig. 51) suggests that the pattern of P concentrations is sharply upwards between Site 18 and Site 109, then static before passage through Gumroot Swamp. Moreover, the role of tributaries is generally to dilute flow, with the exception of a very small seepage flow just upstream of Site 109. We infer that this region, at and below Gainesville Regional Airport is the basin hotspot for geologic P loading, possibly exacerbated by the volume of stormwater generated by significant impervious surface in that area. It is also notable that specific conductance values fall with distance downstream, a surprising observation given the already well established relationship between SRP and F concentrations.

Finally, the same transect, sampled during stormflows exhibits a remarkably consistent upward trend in SRP concentration and therefore load (Fig. 52). As before, the hot spot for loading appears to be between Site 18 and Site 109, which is the reach during which the creek traverses Waldo Road and passes under the Regional Airport runway. To illustrate the dramatic sediment movement that was observed along the Little Hatchet Creek transect, Fig. 53 shows a recent sediment plume emanating into Gumroot Swamp from the creek where gradients slow and channel form starts to become less defined.

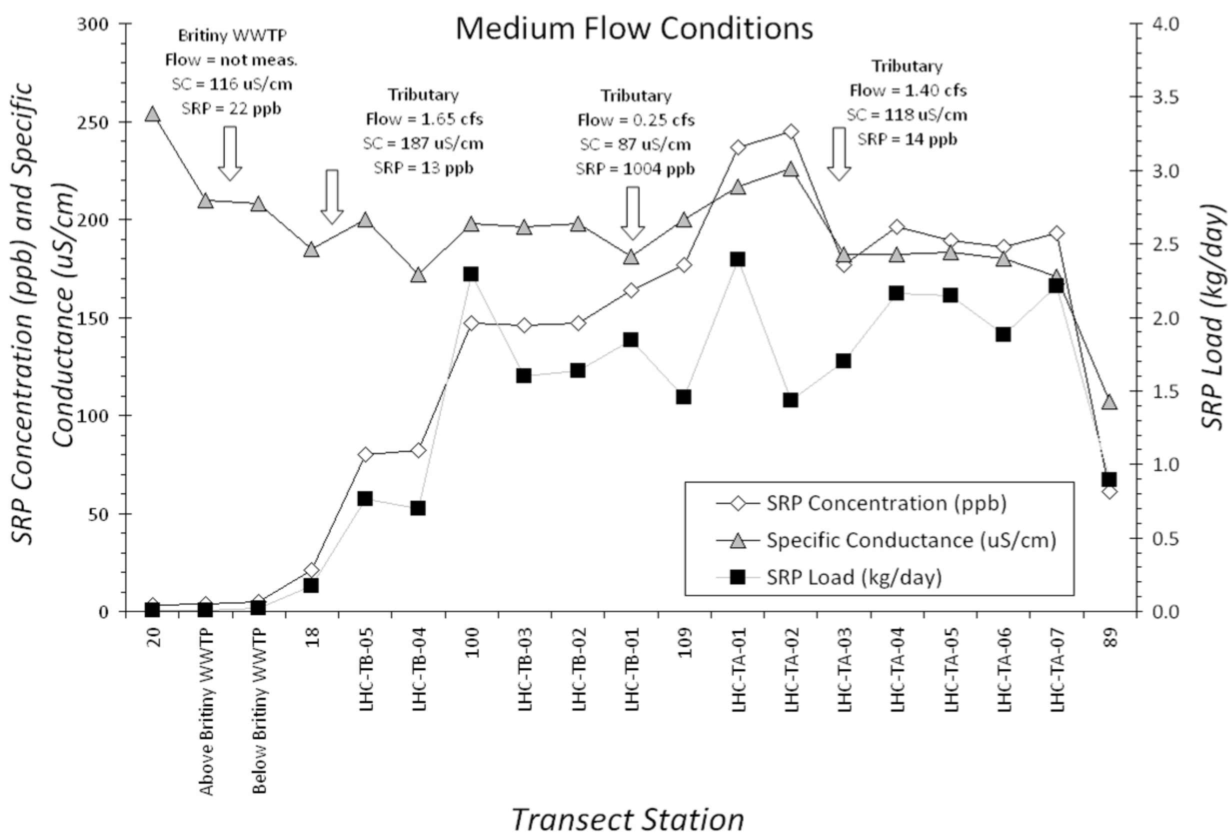

Fig. 51 – Transect water quality and loading profiles for Little Hatchet Creek during moderate baseflow.

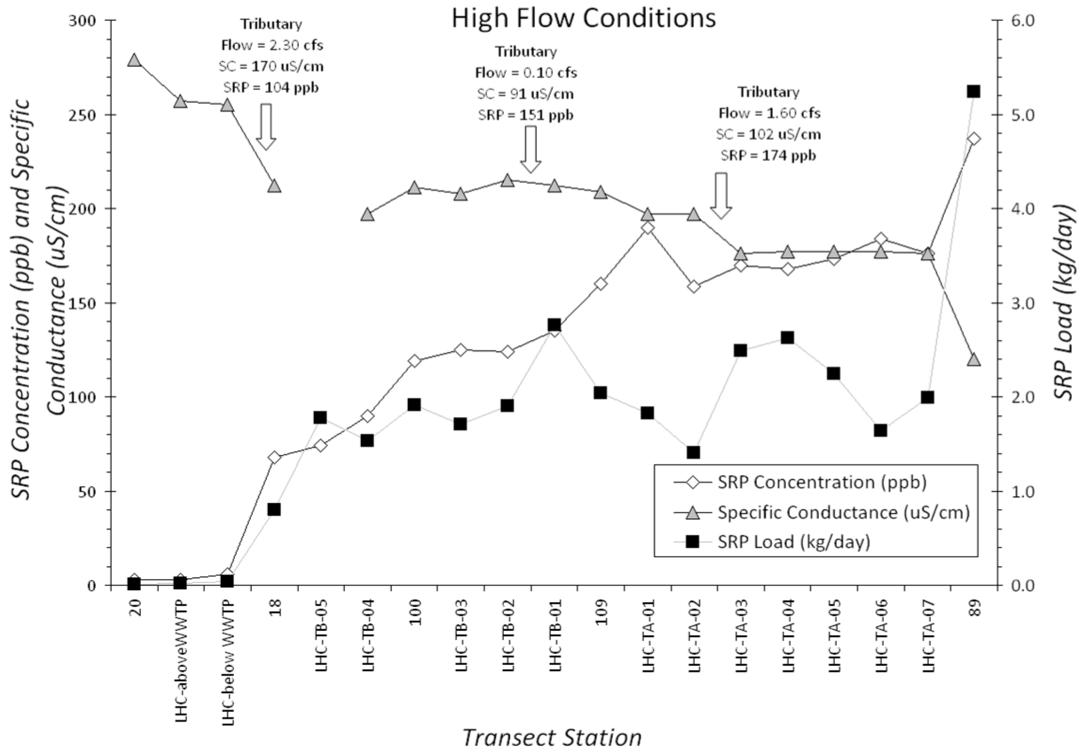

Fig. 52 – Transect water quality and loading profiles for Little Hatchet Creek during storm flow.

Fig. 53 – Example of sediment plume that develops as Little Hatchet Creek discharges into the northern lobe of Gumroot Swamp. The magnitude of particulate P removal is unknown; given evidence for water interactions with phosphatic sands and clays in the region, sedimentation of eroded Hawthorn Formation sands may be substantial.

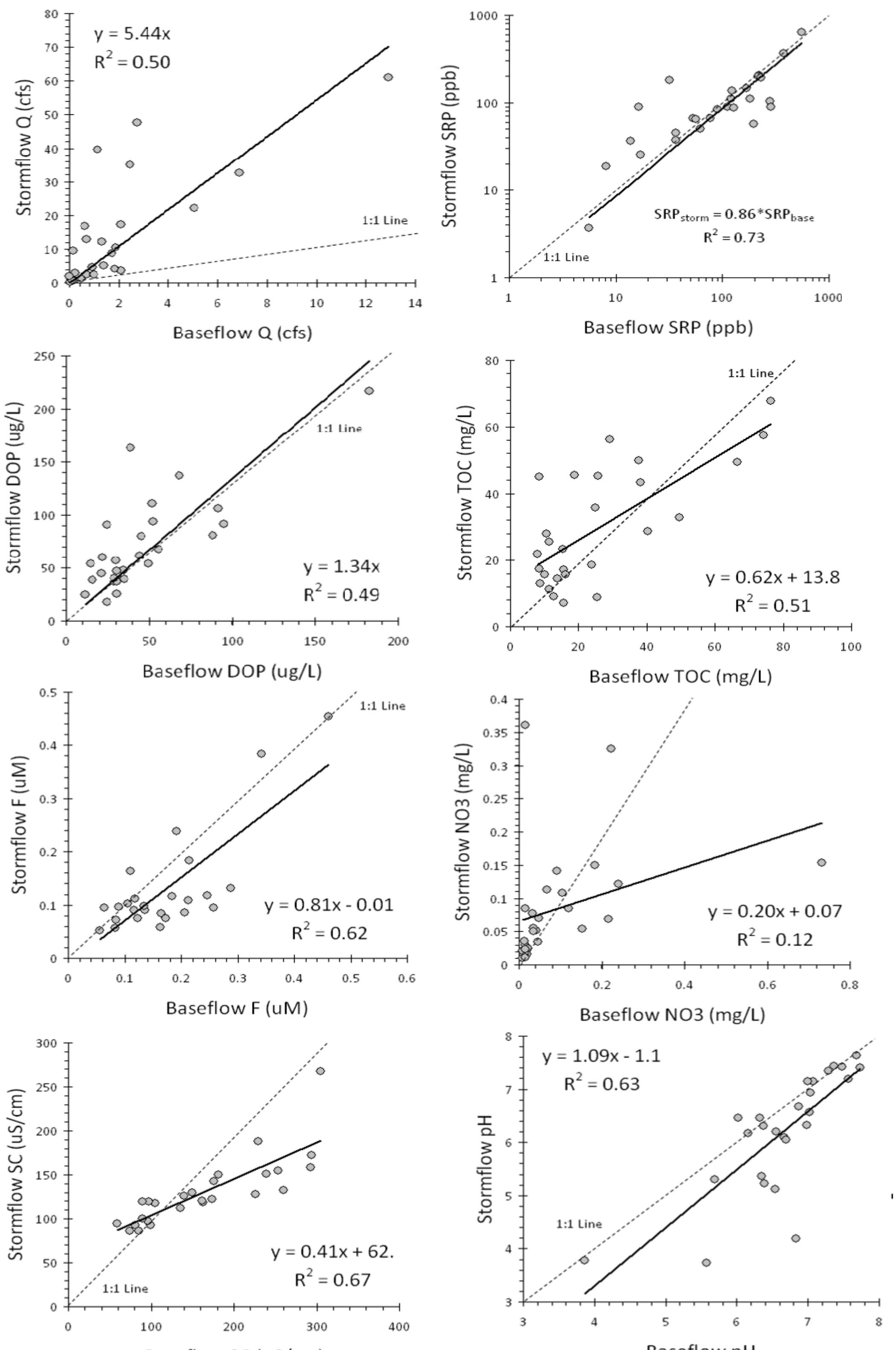

Fig. 54 – Mean baseflow vs. stormflow flows and concentrations for sites with multiple observations within each regime. Best-fit regression line (solid) and 1:1 line (dashed) indicate the correspondence between hydrographic regimes for each site (grey circles).

Finally, it is instructive to compare stormflows to baseflows to understand the role of extreme events on loading patterns. Fig. 54 plots the mean values for a particular station for baseflows (x-axis) vs. mean values during storm flow at the same station. The first panel shows the relationship for flow; the slope of the fitted line shows that the mean difference was 540% between baseflow and stormflow, the 1:1 line provides a visual reference for equality between hydrologic regimes for a particular station.

The subsequent panels show various water quality analytes. Most striking are the relationships for P, where there is no evidence, as a general rule, of a dilution effect with increasing flow. That dilution effect is clear for specific conductance and NO_x. For F and pH the effect appears to be more binary, with slopes similar to 1, but with clear offsets (in both cases with higher values at baseflow). The patterns for DOC are that stations with low baseflow concentrations saw increases, but stations with high levels at baseflow were diluted during stormflow; the former are principally headwater and mid-reach sites, while the latter are wetland sites.

The fact that SRP and DOP are basically constant across flow regimes, despite evidence for dilution effects at some stations (e.g., Site 100/109 – residuals below the line near baseflow concentrations of 300 ppb), suggests that hot-moments for loading are peak flow events, particularly for the few stations where stormflow concentrations were actually higher (principally on Hatchet Creek). The mechanisms via which high concentrations can be maintained at high flow, particularly if the source is a geologic layer with high clay fractions and therefore low transmissivity, are unknown.

Question 4: Groundwater Loading

Water Flows

Monitoring well sites were selected based on access, perimeter representation and slope. At each well, the hydraulic and terrain properties and given in Fig. 55. Some wells exhibited extremely low conductivity (< 0.3 m/day) while others were embedded in moderately transmissive sediments (K_{sat} values between 4 and 7 m/d). The land surface slopes variably, as derived from high resolution lidar digital elevation data; wells along the western perimeter (Lakeshore Drive) were in steeper terrain, while wells to the east and north were within sites with extremely shallow slopes, indicating that landform may contribute to significant variability in potentiometric gradients.

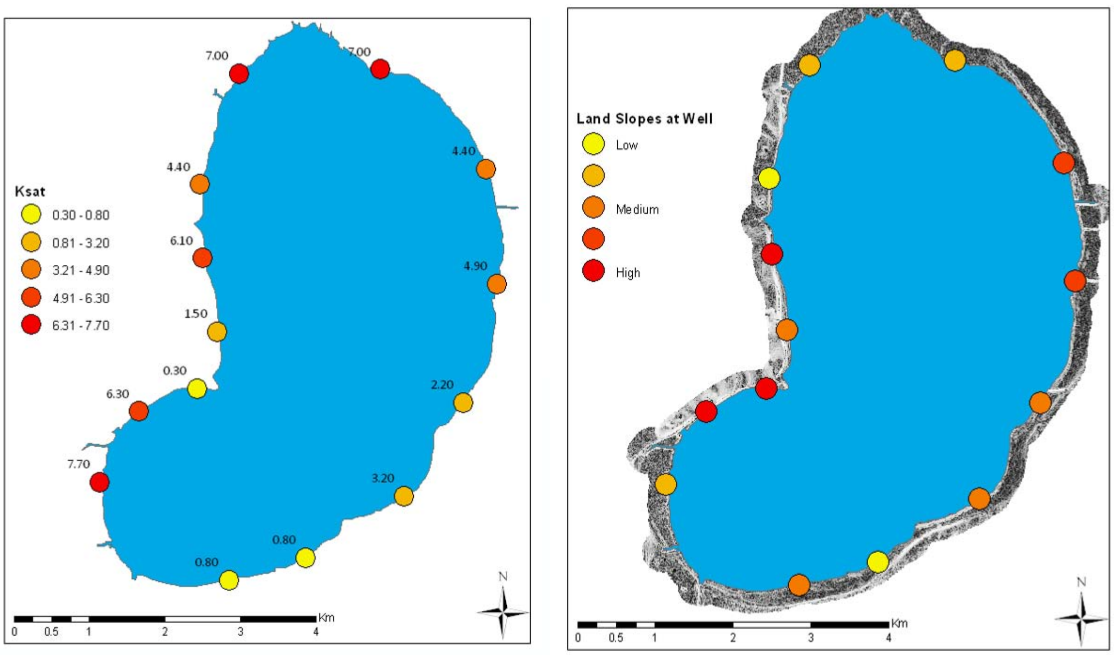

Fig. 55 – A) Measured saturated hydraulic conductivity (K_{sat} in m/day) at lake perimeter wells. K_{sat} values were derived from slug tests in shallow wells. B) Land slope at well location derived from regional lidar digital elevation model.

Monthly sampling of flow at most well arrays (some wells were inaccessible when lake levels were very low) yield a mixed picture of flow directions and magnitudes (Fig. 56) over the period of record. Correlations between flow magnitudes and even directions are not strong, a qualitative observation confirmed by a correlation matrix among flows (Table 6) where some wells (e.g., 291 and 250) are strongly negatively correlated. Note that positive flows are towards the lake in all cases, and flow volumes (m^3/day) are for a perimeter distance defined by the distance to the nearest well in each direction (an average of 2.3 km).

Overall, nearly 35% of correlations were negative, and moreover, there was no apparent pattern to the correlations geographically. Either, well flow rates are highly locally variable (and 14 wells is an inadequate sample size to capture lake fluxes) or flows were small enough to be sensitive to measurement uncertainty.

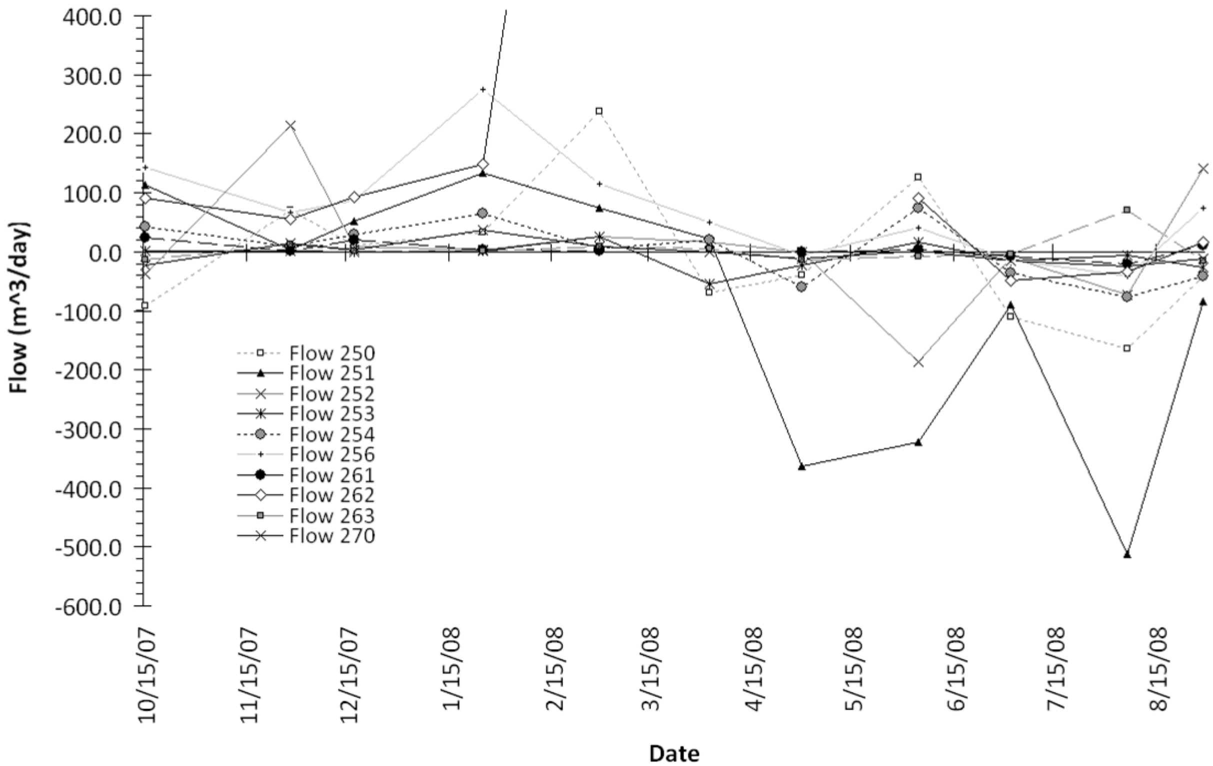

Fig. 56 – Flows over time at each of 11 wells with continuous monthly measurements. Flows at well 262 go to over 1500 m^3/day.

Table 6. Summary of well data for 11 month period (10/07-8/08) for 14 perimeter wells. Shown are mean flow (positive indicates flow towards the lake) and variance information, and also correlation coefficients among wells.

Well ID	Means	Std.Dev.	Flow 111	Flow 250	Flow 251	Flow 252	Flow 253	Flow 254	Flow 256	Flow 260	Flow 261	Flow 262	Flow 263	Flow 264	Flow 270
111	9.76	43.0	-												
250	-3.55	116.0	0.79	-											
251	-88.5	216.0	0.35	0.24	-										
252	9.25	103.5	0.01	0.09	0.51	-									
253	-7.04	22.2	0.52	0.75	-0.09	-0.36	-								
254	3.05	50.4	0.68	0.77	0.54	-0.15	0.74	-							
256	72.0	87.5	0.41	0.37	0.76	0.16	0.27	0.72	-						
260	1.05	43.3	0.69	0.41	0.02	0.21	-0.06	0.02	0.01	-					
261	3.41	12.9	0.50	0.36	0.76	0.10	0.36	0.72	0.57	-0.15	-				
262	479.7	969.5	0.68	0.67	0.60	0.01	0.66	0.91	0.87	0.08	0.71	-			
263	4.14	23.8	-0.43	-0.49	-0.70	0.00	-0.26	-0.75	-0.47	0.07	-0.75	-0.52	-		
264	4.52	15.9	0.79	0.39	0.66	0.25	0.13	0.48	0.41	0.47	0.77	0.55	-0.51	-	
270	-1.81	17.8	0.68	0.72	0.49	0.36	0.30	0.59	0.71	0.59	0.20	0.69	-0.30	0.40	-
291	395.3	849.9	-0.63	-0.85	-0.02	0.30	-0.98	-0.81	-0.28	-0.03	-0.44	-0.68	0.42	-0.24	-0.37

Another striking aspect of the well summary information (Table 6) is that the mean flows, reported in m^3/day are comparatively low, and some wells, over the period of record, were net sinks for water rather than sources. The relative magnitude of flows is a result to which we will return below. The spatial pattern of mean groundwater contributions (Fig. 57) suggests that the wells for which flow to the lake was both large and consistent are on the west side of the lake, but not necessarily proximate. Moreover, the well with the largest input (on the east side of the lake) is proximate to two wells where water flow is away from the lake. The reason for this strong spatial variability is unknown, but is unlikely to be a function of measurement error since the laser leveling technique is capable of capturing the potentiometric gradients (and therefore the sign of flow though not the magnitude which is also dependent on K_{sat} measurements) with a precision of ca. 1 cm.

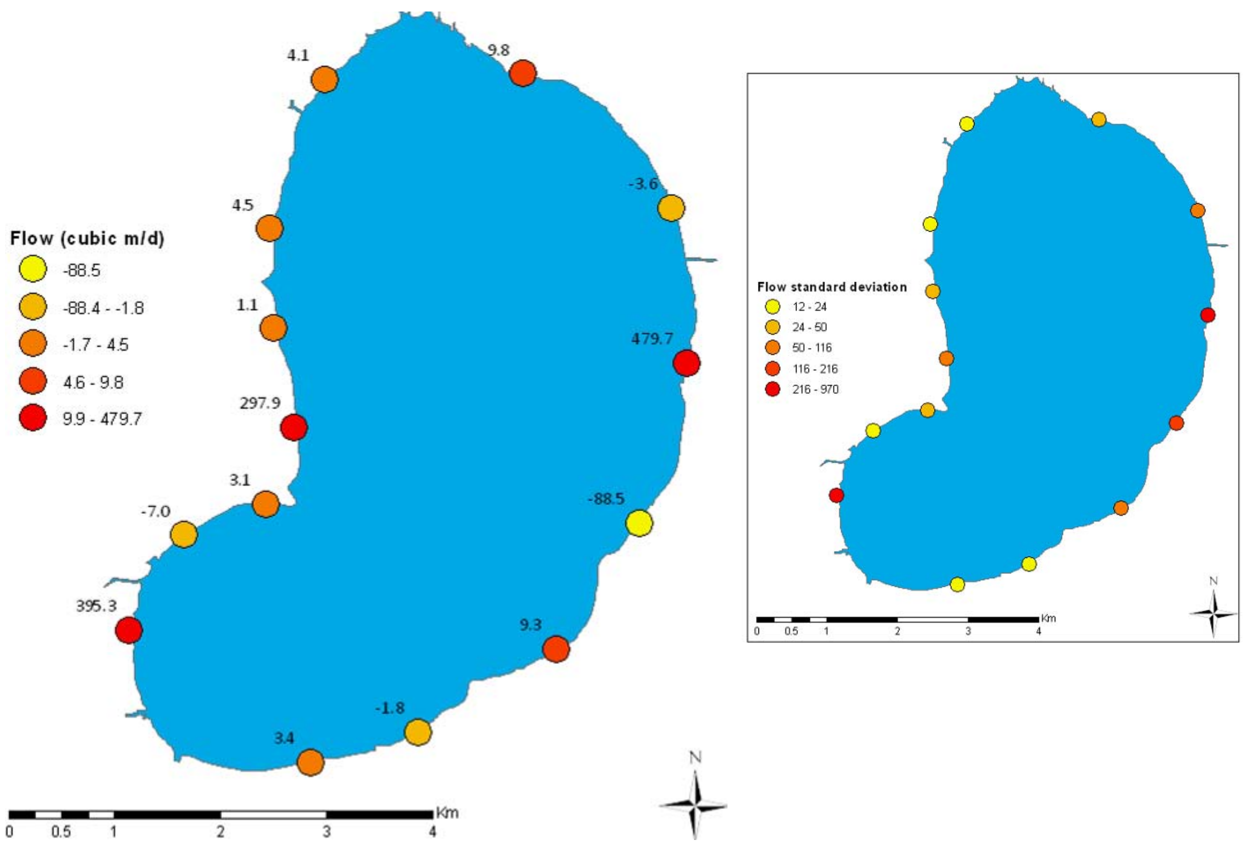

Fig. 57 – Well locations and estimated mean flow rates (to lake = positive, out of lake = negative). Flow estimates are in m^3/day across a seepage face that extends from each sampling well halfway to the next well in each direction. Inset is flow standard deviation, also in m^3/day.

To understand the confusing well data, it is worth noting first that each well flowed in and out of the lake at least one month during the period of sampling. Fig. 58 shows the lake stage (in meters asl) at which flows were into and out of each well (bars), and also measures the frequency of positive flows (black diamonds). Flows out of the lake occurred at higher lake stage this was particularly evident at sites 251, 253, 254, 256 and 260. However, the stage at which flows reversed were widely different. That is, some wells were sources of water while, at the same lake stage, other wells were sinks. This suggests significant spatial variability in intermediate aquifer groundwater levels (the presumed flow source and sink). Moreover, some wells (270, 253) discharge to the lake at stages higher than those when water flows away from the lake. What is clear from this work is that improving our understanding of groundwater flow dynamics will require more frequent measurements (potentially continuous water level recorders) and possibly estimates of regional surficial aquifer levels.

No obvious association between the proportion of observations with flow to the lake and lake stage reversal was evident. That is, sites that exhibited consistent flows towards the lake were not necessarily those where lake stage was strongly different between flow directions. This suggests that groundwater flows to the lake are co-regulated by groundwater levels and lake levels, with unique conditional relationships for each well location. This makes generalization difficult, but, given the magnitude of flows overall, perhaps of little consequence.

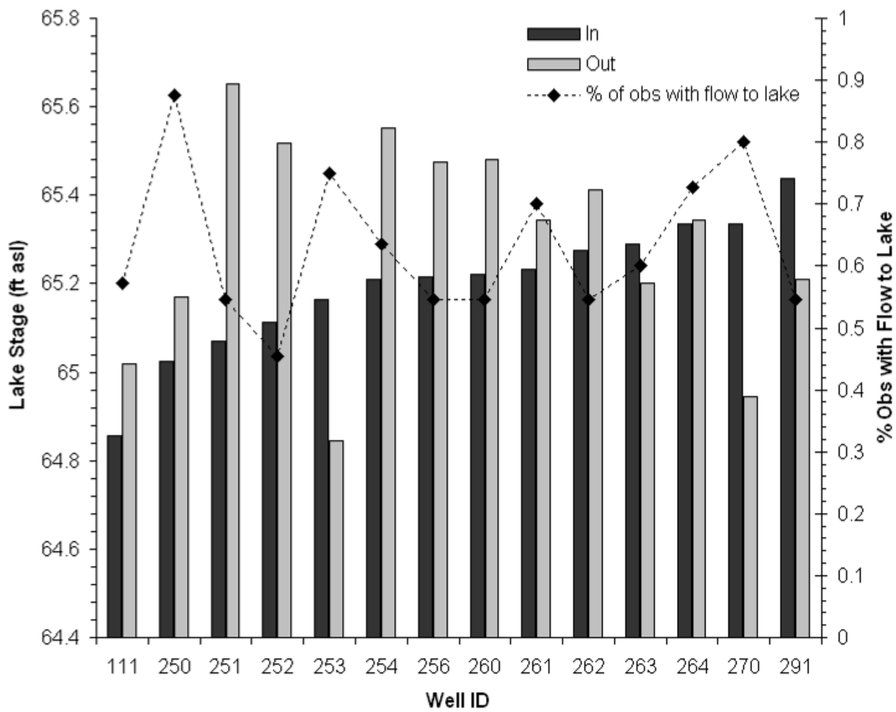

Fig. 58 – Mean lake stage for flows into and out of the lake from each well. Flows out of the lake occur at higher lake stage for most wells, but flow reversal does not occur at the same lake stage. Note that not all wells were visited all months due to access constraints.

Fig. 59 summarizes the volumetric flow to the lake across all wells for the period of record, plotted with lake stage for comparison. There is modest evidence of a correlation between total flow and lake stage, with evidence of time lags as well, but the more pressing observation is the magnitude of the total flow. Our hypothesis was that groundwater represents an important flowpath for water delivery to the lake. These data suggest that this is not the case. Estimated water fluxes to the lake total nearly 240,000 m^3 (2.4E5 m^3) over the year from October 2007 to August 2008 (including periods when groundwater flow was in sum leaving the lake). In contrast, net direct rainfall delivered nearly 20 times more water (ca. 5.7E6 m^3) over the same period. Hatchet Creek, the principal input, has delivered an average of 3.4E6 m^3 annually (range 1.2 – 7.3E6 m^3), over an order of magnitude more flow. Despite groundwater flows that are small in comparison, the potential for these flows to influence lake chemistry are still unknown since high P concentrations might be expected for water flowing through Hawthorn Formation materials.

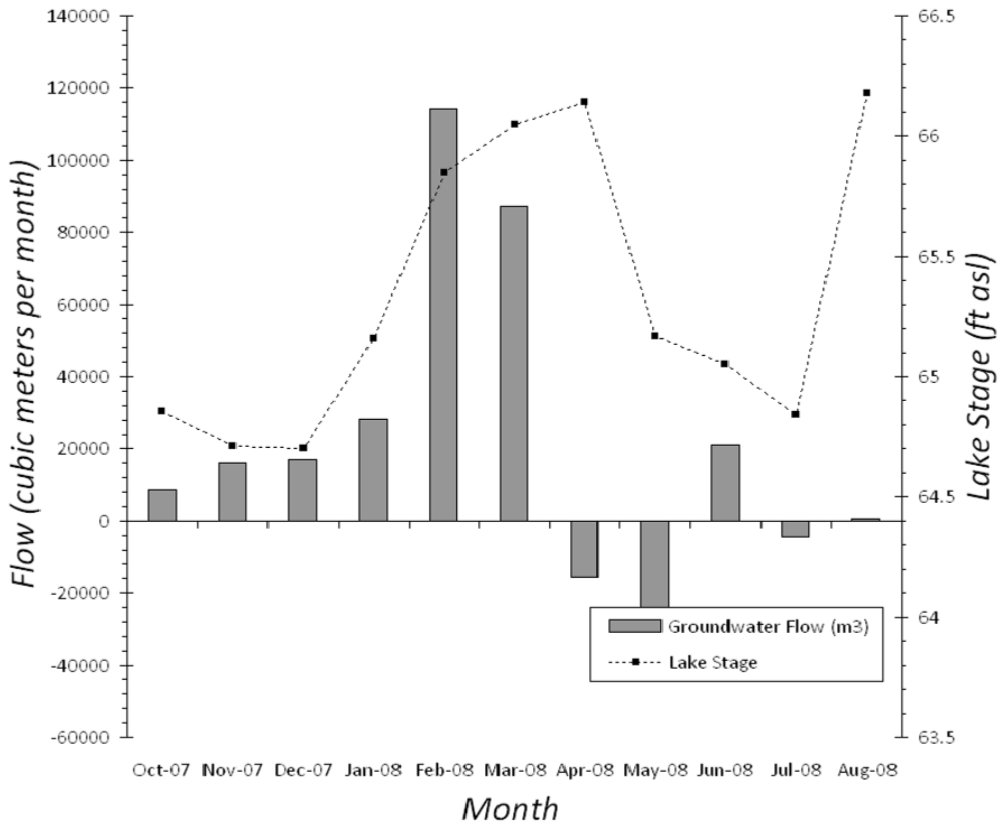

Fig. 59 – Estimated groundwater flow (m^3 per month) and lake stage over period of observation.

Well Water Chemistry

Despite the comparatively minor flow contribution from groundwater inferred from the well observations, the chemistry of the samples collected was still needed to determine if loads were important. Fig. 60 summarizes the mean concentrations of SRP and F in the wells over the period of record. We report SRP and F because the former is the main component of P load to the lake from subsurface sources, and because the latter is a covariate with P when P is from the Hawthorn. SRP concentrations are tremendously high in some areas, with persistent levels over 2 mg/L (2000 ppb) in wells along the western lake edge. Notably, despite the presence of septic tanks in that area, F concentrations are also high, which may suggest that the P is more likely to be of geologic origin; since some of the groundwater flow may be from septic tanks that are on municipal water supply, there is also a possibility that the F and P are of anthropognic origin. However, NO_x concentrations in those samples are comparatively low, suggesting that septic effluent to a relatively well oxygenated aquifer may not be the main reason for significant P enrichment. Further work on well samples is needed to determine the ultimate source.

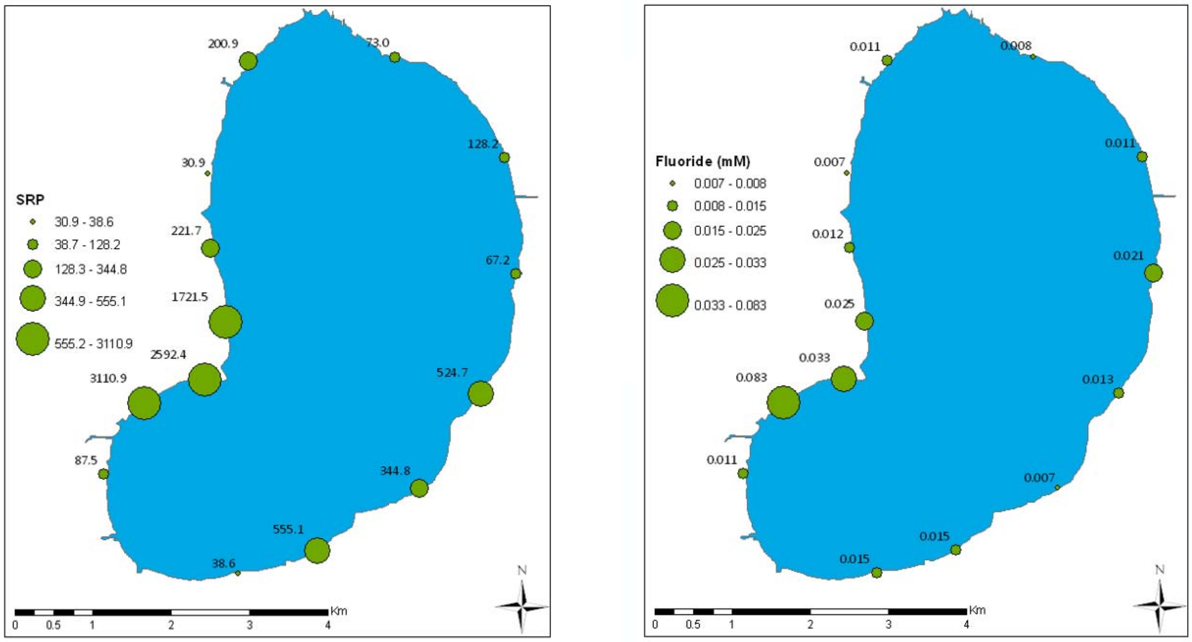

Fig. 60 – Patterns of SRP (ppb) and F concentrations in perimeter wells around Newnans Lake.

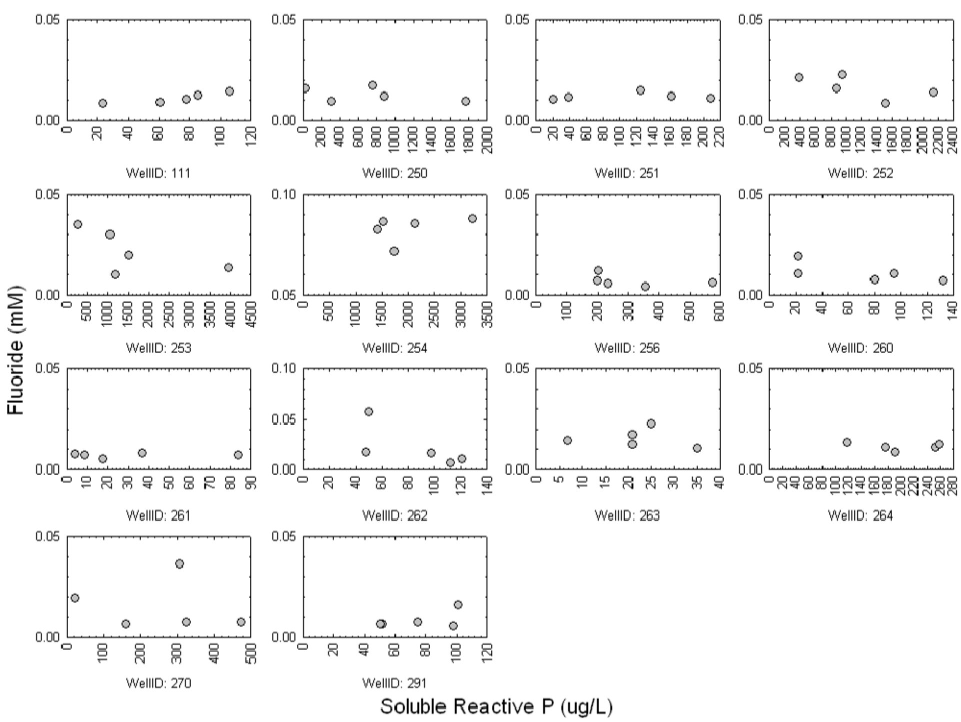

Fig. 61 – Covariance between F and SRP for each well over the period of record. Note that F concentrations were not available for all months for which wells were sampled.

On a well-by-well basis, the covariance between P and F is less clear, as indicated by the categorized scatter plot shown in Fig. 61. For some sites (111, 251, 291) the correlation is positive, but for others, the trend is reversed or absent. Significant within well variability makes generalization about wells problematic, but Fig. 62 summarizes the mean well concentrations of P vs. F and NO_x. What can be inferred from this plot is first that P and F are strongly correlated at the well level despite the failure to observe within well covariance of these analytes, and that P and NO_x are negatively correlated (though the model fit is not statistically significant. Overall, we conclude that the P is likely at least partially of geologic origin. More important than the source, however, is whether the load is of significant importance in the context of the entire lake basin, warranting management and source identification attention.

To address the total load question, we estimate perimeter P loading based on monthly flow observations (Fig. 63) and concentrations of P. A summary of estimated loads (kg P per day) based on the sum of fluxes estimated from each well (Fig. 63) suggests that the total load from groundwater is vanishingly small, despite enormous concentrations of P in the groundwater. Indeed, the annual P load to the lake from October 2007 to August 2008 is -0.09 kg/day; that is, the lake appears to be exporting P to the groundwater. This finding is subject to significant error due to flow estimates (and in particular assumptions about saturated hydraulic conductivity and flow cross-sectional area), but even with significant revisions to the hydrologic estimates, it is unlikely that groundwater P loading is a significant source to the lake, even at low lake stage. The magnitude of the daily flow (-0.09 kg/day) can best be understood in comparison to surface water flows. Lake Forest Creek loads at an average of 1.08 kg P per day, Little Hatchet Creek at an average of 3.45 kg P per day, and Hatchet Creek at an average of 2.71 kg P per day over the period of record. Even the largest net input month (Nov. 2007; Fig. 63) had a P loading rate of 0.6 kg P per day, which is less than 10% of the total load.

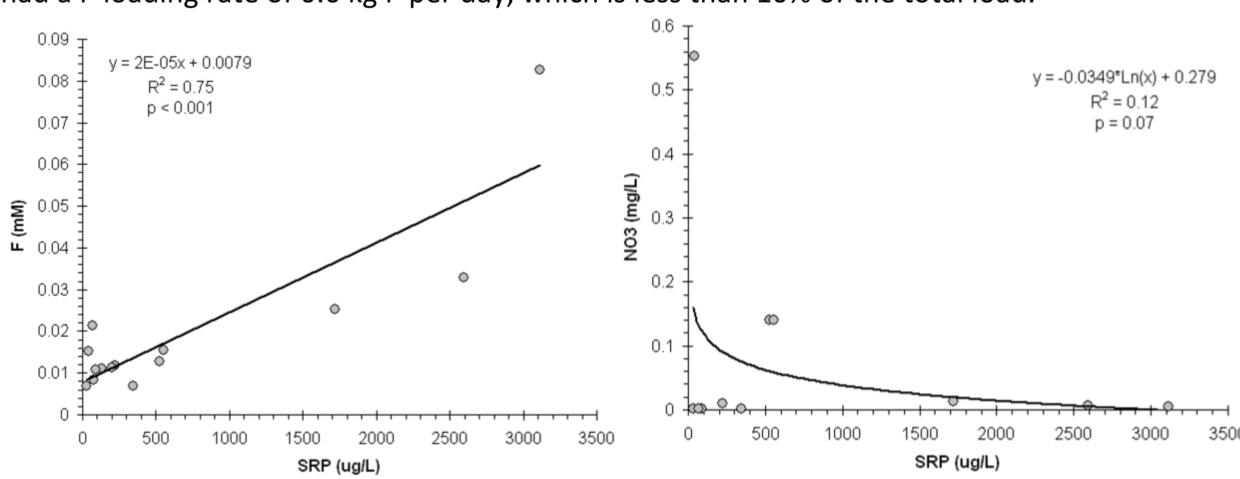

Fig. 62 – Mean well concentrations of SRP vs. covariates suggestive of source. A) Covariance with F, a tracer of geologic P, and B) NO_x, a tracer of fertilizer or wastewater sources. Positive covariance with F suggests that most of the P arriving to the lake from groundwater is of geologic origin (though extreme values of F at one site may be due to F loading from municipal water supply). Correlations with NO_x are negative but not statistically significant.

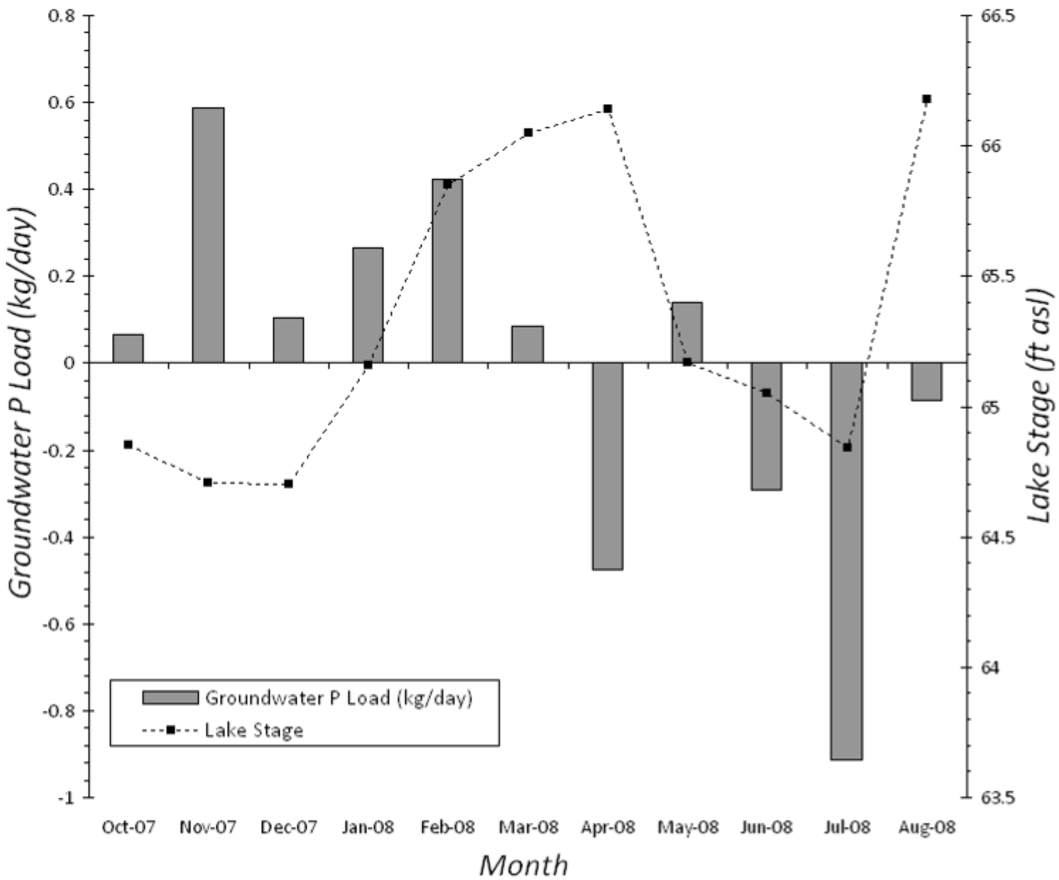

Fig. 63 – Monthly load of P to Newnans Lake from groundwater as inferred from perimeter wells. Note that groundwater is a sink for P (i.e., high TP concentrations leaving the lake) during April and July 2008.

Based on these data, we conclude that while further study of groundwater inputs is needed, the relative importance of diffuse groundwater loading to the lake is low. Further monitoring of the perimeter wells is an important part of ongoing research in the basin, particularly in light of estimates of groundwater loading to Lake Lochloosa (132,000 m^3 per day; Hicks et al. 2008), but even order of magnitude errors in loading rates will not dramatically impact the lake P budget. The relative importance of groundwater load would also be expected to decline with increasingly wet climatic conditions because the magnitude of surface water loading will increase, while no obvious trend with lake stage or antecedent rainfall was evident for wells.

Question 5: Analysis of Soils and Sediments as a P source

Analysis of Surface Soils

The hyper-eutrophic character of Newnans Lake is difficult to reconcile with the low intensity state of the land cover in the watershed. The evidence for a geologic source of P in the surface water systems of the lake strongly supports consideration of different modes for eutrophication than are typical in other watersheds. That is, mechanisms to minimize the mobilization of geologic P may take management precedence over efforts to reduce fertilization. Two particular attributes of the watershed warrant consideration as management responsive activities. The first is control of stormwater runoff flows; increased flow of storm water due to increased impervious surface cover may exacerbate the contact between stream water and Hawthorn materials massively enriched in P. Further consideration of this evidence is discussed in the section on longitudinal water quality sampling. Another proposed mechanism for anthropogenic entrainment of geologic P into the lake basin P budget is the disturbance of the surface soils in areas where the Hawthorn is proximate to the land surface. Specifically tillage and bedding practices invert soil profiles, placing P enriched Hawthorn sediments at the land surface where accelerated weathering can mobilize the P in leachate and surface runoff.

To test this hypothesis, we examined soil P profiles in 5 sites with varying degrees of entrainment risk based on the depth of the Hawthorn (Fig. 9). Results of this sampling effort (Fig. 64) immediately discount the potential for this mechanism to be important in P mobilization in the lake basin. Nearly all of the soil sampled, with the exception of one site near a seep with consistently high SRP concentrations, were extremely low (mean TP concentration in surface soils was 40 mg/kg). Site 25, selected because the Hawthorn is evident very near the surface in the vicinity of the seep that is regularly sampled for water quality, is the exception, with high P concentrations vis-à-vis agronomic requirements. However, even here the soil is massively depleted of P compared with concentrations within the Hawthorn Formation clays which have concentrations of 30,000 mg/kg or higher.

In addition to observing low concentrations of soil P, we also observe evidence of stratigraphy, with profiles at depth (80-100 cm) exhibiting significantly higher P levels. This suggests that what little stratification of P concentrations in the soil exists has not been dramatically disturbed by multiple decades of human management of the surface (0-50 cm) soil profile. While it is not possible to rule out an effect of soil management on P enrichment fro deeper layers, principally because there are no upland areas that could act as an unbedded control, we are able to conclude that the concentrations of P in the surface soils are highly unlikely to be a principal source of P to the lake because absolute magnitudes are comparatively small, even in areas where the Hawthorn is proximate to the land surface (sites 25 and 76 in particular).

Among the notable observations from Fig. 64 are a) remarkably low P levels that persist at almost all locations, even where the Hawthorn is predicted to be comparatively near the land surface (e.g., site 76) and b) the increase in TP with depth. With regard to the first point, while

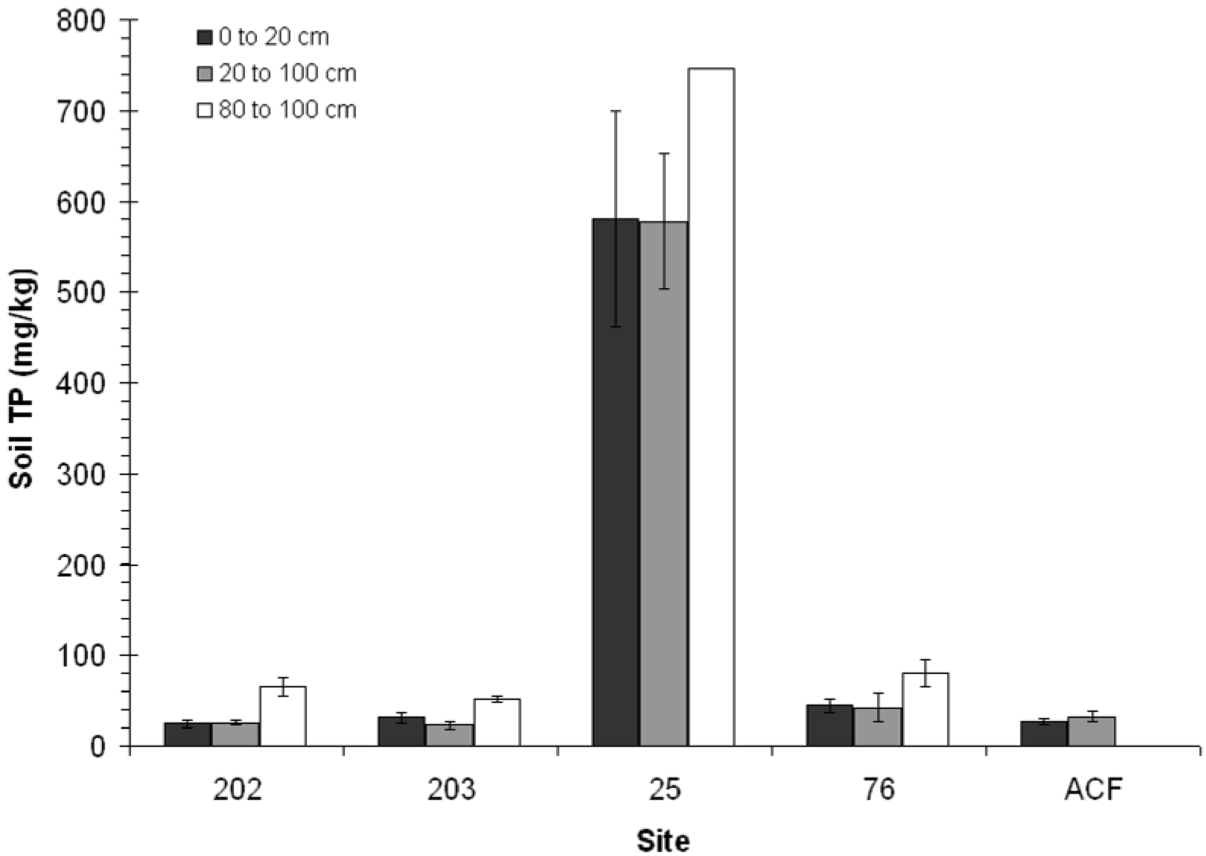

Fig. 64 – Soil TP in each of the 5 landscape blocks selected to examine concentrations in space and with depth. Note that no 80-100 cm profiles were obtained at ACF because there were no stratigraphic discontinuities in those soils. The block at site 25 corresponds to an area where high SRP concentrations are routinely observed in seepage water. The block at site 76 corresponds to an area downstream of a large blueberry farm where high SRP concentrations are also regularly observed.

higher levels were observed at site 25, where the Hawthorn was also expected to be near the land surface, these levels are vanishingly small compared with TP levels in clay sediments obtained from groundwater wells along Hatchet Creek (near site 39), where TP averaged 32,500 mg/kg (i.e., 3% P by mass). Using that concentration as one end-member of a soil mixture that contains clays and mineral soil (with a P concentration of 50 mg/kg), the relative enrichment in soil P at site 25 suggests that those soils are less than 2% Hawthorn clay, by mass. As such, it is reasonable to conclude that significant mixing of phosphatic clays into the upper soil profiles has not occurred, and where the modest enrichment has occurred, it is highly localized.

With regard to the second point (increase in P with depth), we note that downward migration of P is likely in the low anion exchange capacity soils, and that the differences in concentration are statistically but perhaps not biologically significant. We expect that low P concentrations in the surface soils persist throughout the basin, making it highly unlikely that contemporary or historic soil management practices are responsible for high levels of SRP in baseflows.

To further examine this, we constructed an approximate mass balance between surface soil P throughout the basin, and the mass of P loaded to the lake each year. Given observed soil P concentrations (mean = 130 mg/kg) across the entire basin (275 km^2), and given an average annual P load to the lake of 40,000 kg, the entire P mass in all surface soils would be mined in less than 500 years if those soils were the principal source of P. This strongly reinforces the conclusion that the source of P is from deeper layers that are not subject to surface disturbance effects except highly locally (e.g., where clay material is already at the land surface).

Analysis of Stream Sediments

Stream sediment samples collected to examine covariance between sediment and water quality properties were analyzed for a suite of properties. Overall, we observed strong covariance among stream sediment measurements (Fig. 65). Among the notable correlations were that TP was almost entirely comprised of mineral P (i.e., almost no organic store of P except when P

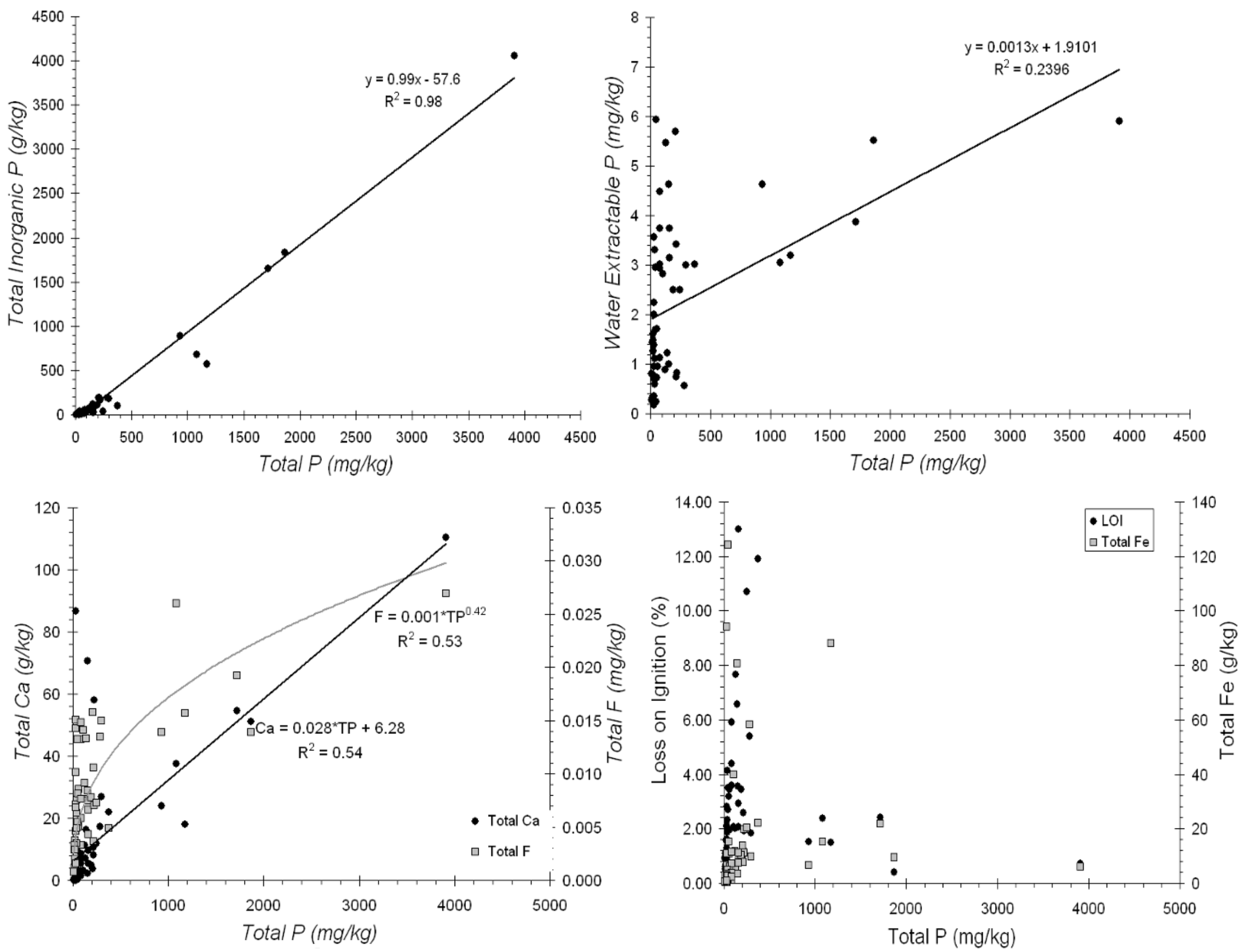

Fig. 65 – Patterns of covariance within creek sediments. A) Between total P and total inorganic P; B) Between total P and water extractable P, C) Between total P and Ca and F, D) between total P and LOI and Fe. Together, these suggest P is principally in mineral form in creek sediments, and remains closely associated with fluoride bearing apatite minerals

concentrations were quite low). We also observed that most of the P is in relatively recalcitrant form; the fraction of P that is water soluble is very low (note axis scales in Fig. 65B). This suggests that while there are tremendously high levels of P in some sediments, and generally abundant P overall (the median concentration was 180 mg/kg), the fraction that is soluble at any given time is low. This is consistent with the fact that apatite, the expected principal form of phosphate mineral in the Hawthorn, is very resistant with weathering. As such, the mobilization of geologic P is probably a diffuse, slow process rather than one that operates under the influence of episodic drivers.

We also observed comparatively strong associations between TP and Ca and F (Fig. 65 C). This reinforces the conclusion that much of the P stored in the sediments is apatite (and a significant fraction is fluorapatite where F would be a by-product of the weathering process). Moreover, there is no evidence that much of the P is associated with organic materials or Fe (Fig. 65 D) except at low concentrations. It is also clear that P is principally found in sediments collected from stream segments that are in or near the Hawthorn Formation (Fig. 66). Proximity to the Hawthorn is also an excellent predictor of other stream sediment properties, including total N, total F and organic matter content (the latter presumably due to the additional organic storage in the bottomland forests that persist at the topographic bottom of the basin landscape).

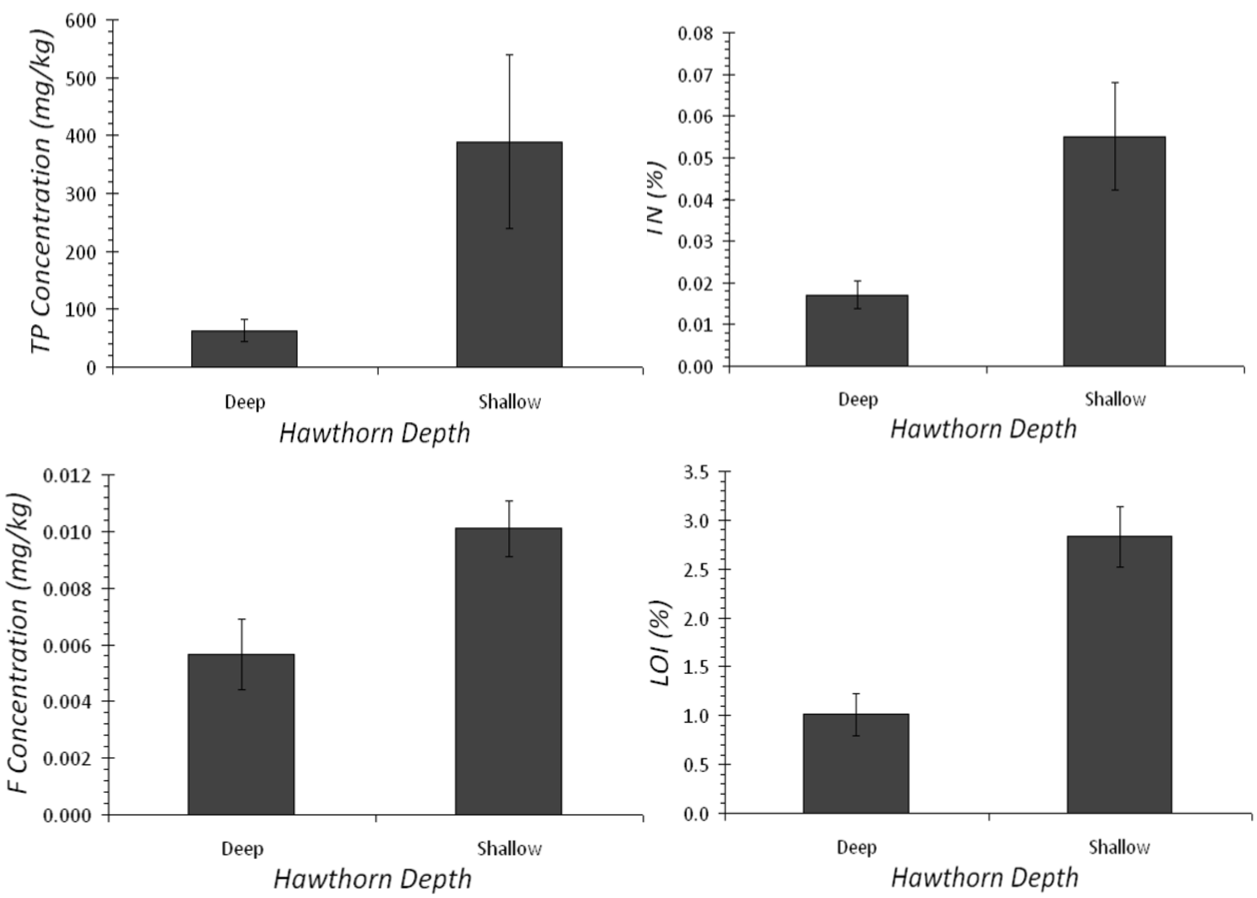

Fig. 66 – Mean stream sediment properties stratified by depth to Hawthorn; error bars denote 95% confidence intervals for the mean. Contrasts were all significantly different ($p < 0.01$).

Covariance with water quality attributes was generally weak. Indeed, the association between sediment TP or water extractable P and water SRP was not significant (r = +0.17 and +0.17, respectively, $p > 0.05$ in both cases). However, there was a significant positive association between stream water SRP and sediment F (p = 0.01) and also a negative correlation with LOI (organic matter content) (p = 0.04) (Fig. 67). While the latter is likely associative and not causative, it does suggest that stream water P concentrations are dominated by mineral interactions. The association with sediment F is compelling because it reinforces the potential use of that fluorapatite weathering breakdown product as a tracer of geologic P, at least where significant anthropogenic loading of F from municipal water supplies is not present.

Overall, we conclude that sediment properties are likely important in regulating stream water chemistry, but that the strength of the association makes their measurement and use as surrogates of limited utility. Future work on the manner and magnitude of P sorption to stream sediments at the reach scale (e.g., via measurements of P spiraling) would require more detailed analysis of stream sediment properties to determine if P sorption is possible, and if so, how permanent the sorption is. We also conclude that the sediment-water associations that do exist support the contention that much of the P in stream water samples is of apatite origin. To summarize, the three reasons that we conclude this latter point are 1) covariance between stream water SRP and F, 2) extreme concentrations of TP in some sediments with a comparatively small fraction that is water soluble, and 3) covariance between sediment P and F.

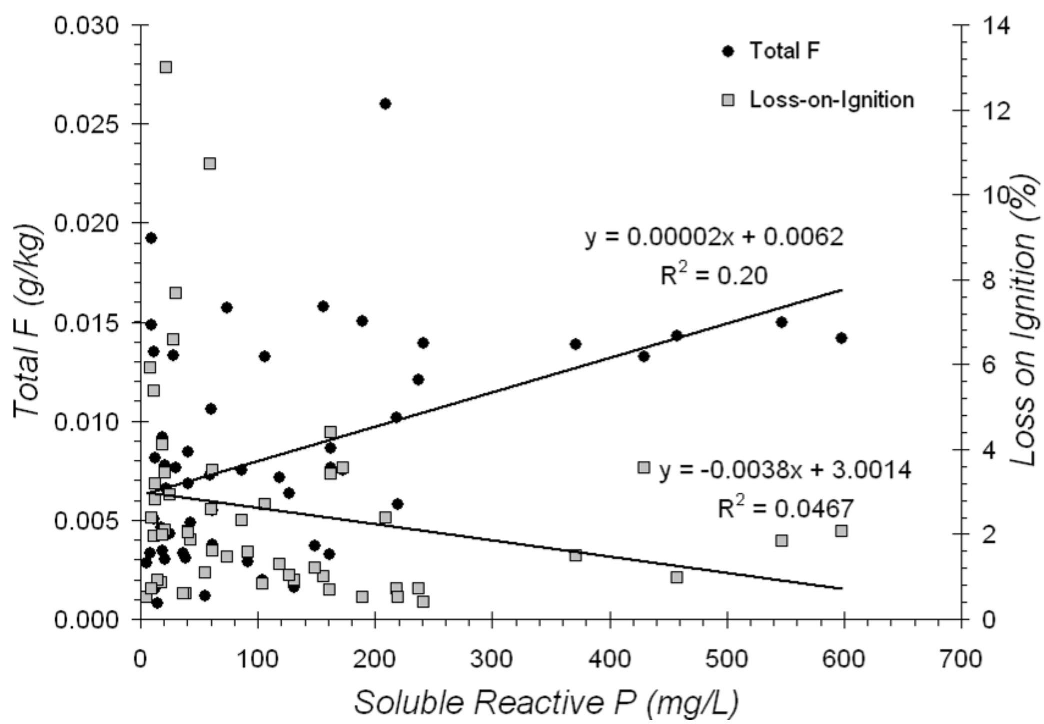

Fig. 67 – Relationships between sediment properties and mean SRP in the water at each site. The only variables that showed a significant covariance with SRP concentrations were fluoride concentration (r = +0.45) and LOI (r = -0.21). Covariance with total P, total inorganic P and total Ca were marginally significant ($p < 0.10$; r = +0.17, +0.18, and +0.16, respectively).

Analysis of Archival Data

Newnans Lake watershed has received significant monitoring attention; the archival data that has been assembled from this effort allows water quality observations made during this work to be put in a longer term context for interpretation and validation. This section reports analysis of archival data for correlations among water quality constituents, comparative concentrations and loads by site, and preliminary water and P budget calculations for the lake watershed.

First, the flows observed during the current period of study (2007/08) are placed in historical context. Table 7 shows the mean flows observed at each station at or near existing long term monitoring sites. To place these flows, observed during an historic drought, in context, Figure 16 shows frequency histograms for each of the long term stations; Table 7 shows the summary metrics for each, and compares those values with observed data in this work. Notably, Bee Tree Creek has been dry for the entire duration of this work despite median flows of nearly 1 cubic foot per second. Similarly, the lake discharge (Prairie Creek) flows at a median rate of nearly 10 cfs, but was well below that level when flow was observed; no flow was observed between June 2007 and February 2008. The only stations that showed flow rates comparable with historical levels are Lake Forest Creek and Little Hatchet Creek (North Branch). While it is perilous to compare synoptic observations with continuous data collection, it is clear that the current study period represents a significant dry period. Observations of concentrations and correlations should be interpreted in light of these extenuating circumstances.

Table 7. Observed flows at (or near) long term flow monitoring stations

	Historical Flows			Contemporary Flows (2007/2008)				
Long Term Location	25^{th}% (cfs)	Median (cfs)	75^{th}% (cfs)	Site	# Obs.	Min. (cfs)	Mean (cfs)	Max. (cfs)
Prairie Creek	9.8	36	81.0	52	2	4.2	5.1	6.0
Hatchet (Fairbanks)	1.4	5.9	20.3	103	19	0.2	5.3	36.9
Little Hatchet (G'ville)	0.9	2.1	5.3	18	16	0.2	13.2	144
Bee Tree Creek	0.6	3.5	12.0	101	7	1.8	44.0	104
Little Hatchet (NB)	0.7	1.6	3.7	100	28	0.3	5.9	45.1
Lake Forest Creek	0.3	1.1	2.8	32	31	0.0	4.0	24.8

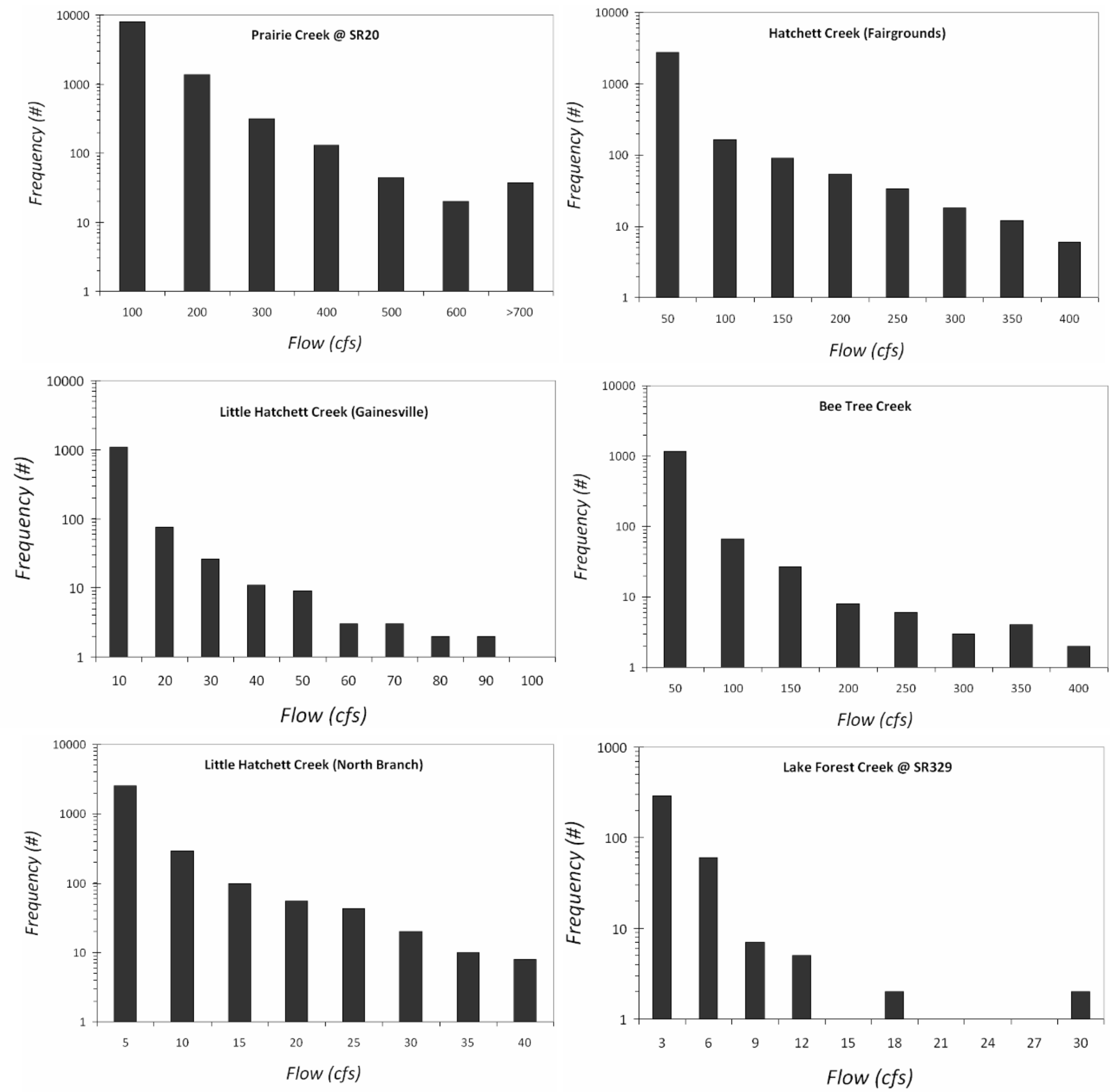

Fig. 68 – Summary of historical flows in Newnans Lake watershed principal inflows and outflow. See text for contemporary flows during the current research in comparison with historical flows.

Using the archival flow observations at the 6 stations (Fig. 68), an approximate lake water budget was computed (Fig. 69). The period of record was selected for when the principal inflow and outflow creeks had maximum data coverage. A roughly 7 year period was selected, during which significant rainfall events (1998 El Nino and 2004/05 hurricane activity) and drought periods occurred. The cumulative deficit in water (inflows minus outflows summed over the period) was large (2.2E+08 m^3 over the 11 years period) and could not be accounted for by tabulating the net rainfall (precipitation – evaporation ~ 5.8E+06 m^3/yr). The total input unaccounted for is nearly 4 times the lake volume (~4.3E+07 m^3, assuming 1.6 m average depth), averaging 20E+06 m^3/yr though with substantial inter-annual variability, suggesting that unmeasured inputs are large.

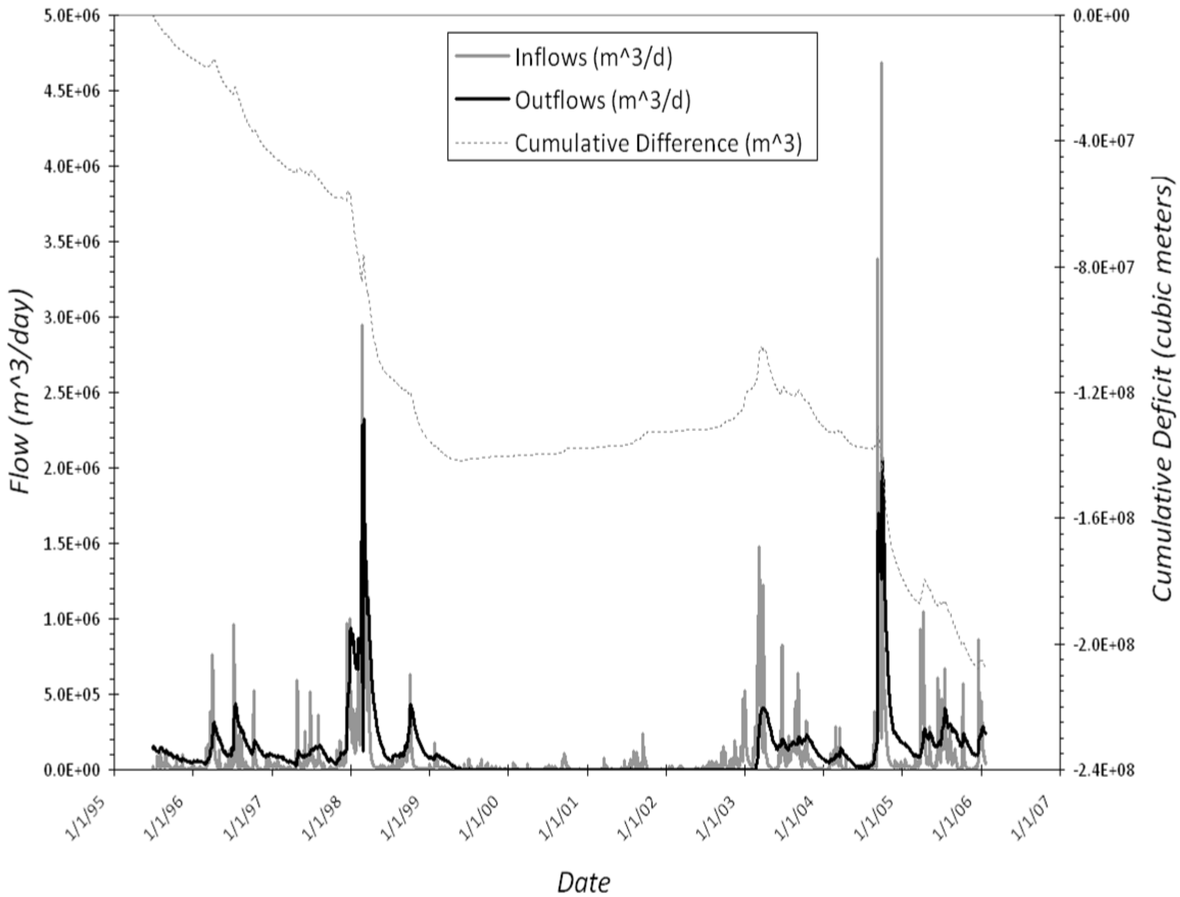

Fig. 69 – Water budget for Newnans Lake for June 1995 though April 2006. Inflows (sum of measured creek flows) minus outflows (Prairie Creek) are summed over the period of record to determine the period deficit. Net rainfall inputs (precipitation – evaporation) average 5.8E+06 m^3 per year over the lake area (6.1E7.

Historical analysis of water quality was principally focused on determining nominal observed concentrations for each station for comparison with contemporary data. Figures 70 through 74 show historical mean concentrations of the principal water quality analytes of interest, including indicators of surface vs. groundwater sources (Fig. 70), tracers of substrate-water interactions and residence times (Fig. 71), total and labile C concentrations, (Fig. 72), NO_x/TKN concentrations (Fig. 73) and total and labile P concentrations (Fig. 74). In each figure, error bars represent 95% confidence intervals for the mean. Post-hoc comparisons of means across creeks was not done because the periods of record do not overlap and observation counts differ substantially. Figure 6 shows the location of each of the water quality stations compared.

Figure 70 illustrates that both Lake Forest Creek and the north branch of Little Hatchet Creek are not typical surface water systems for North Florida. Conductivity values are typical of intermediate or Floridan Aquifer water, and inconsistent with flows in Hatchet Creek. This signal is reflected in both alkalinity (showing extremely high buffering capacity in Little Hatchet Creek) and conductance, which leads us to conclude that groundwater sources to both sites are large. This is particularly interesting because the only creeks that maintained flow during this historic drought are these two. If they are fed by water stores with much longer residence times (i.e., groundwater not surface water), this behavior would be expected. For Lake Forest Creek in particular, where sources of deep aquifer water are absent (our inference that this is Floridan Aquifer water, not intermediate aquifer water), this is suggestive of leaking municipal water pipes.

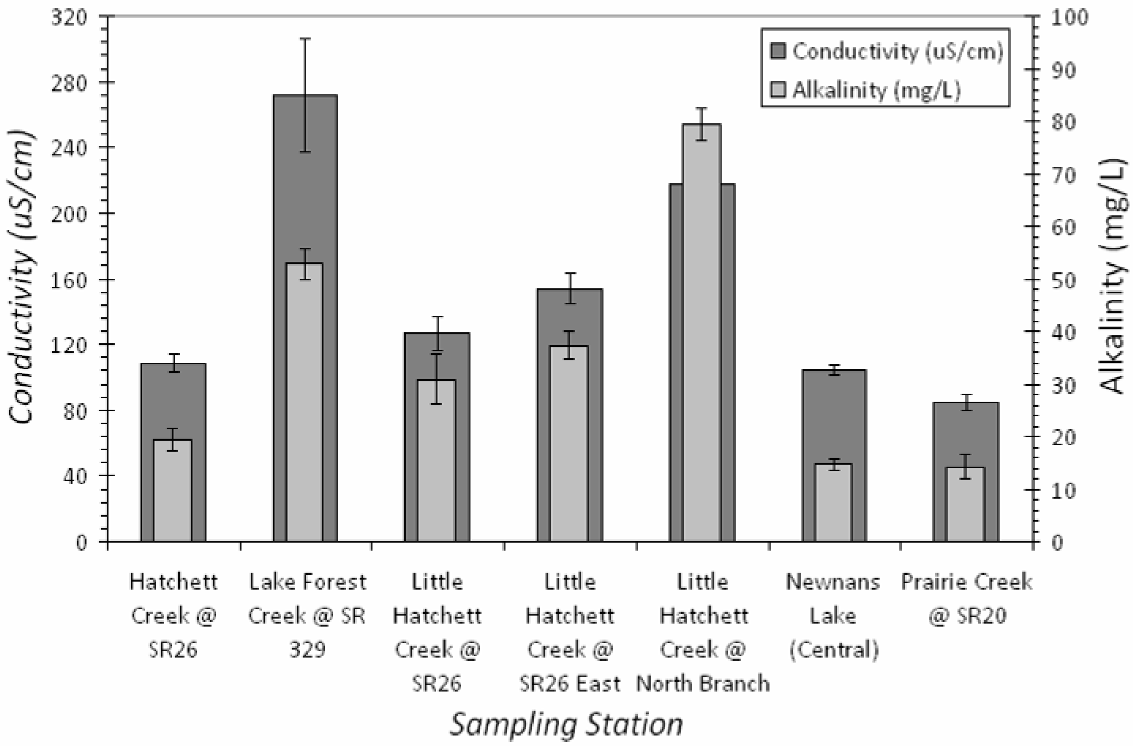

Fig. 70 – Comparison of specific conductance and alkalinity observations (archival data) among the creeks draining to and from Newnans Lake.

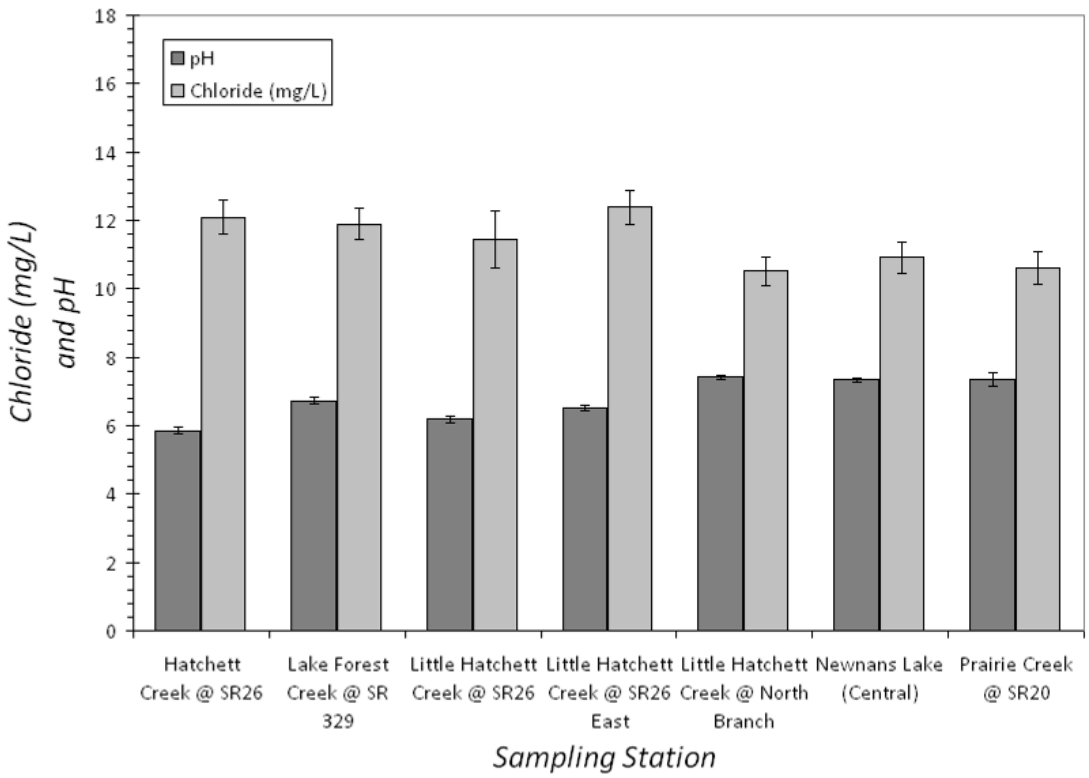

Fig. 71 – Comparison of chloride concentrations and pH observations (archival data) among the creeks draining to and from Newnans Lake.

Chloride, which is a useful parameter to track because it acts as a conservative tracer, documenting evaporative losses during hydrologic conveyance, is not indicative of any major differences among creeks (Fig. 71). Typical chloride concentrations in regional rainfall are 0.4 mg/L (NADP 2006; http://nadp.sws.uiuc.edu/ads/2006/FL03.pdf) suggestive of strong evaporative pressure on water traveling through the entire system. Observations of pH suggest greater acidity in Hatchet Creek and Little Hatchet Creek at SR26 than at other inflow sites, possibly illustrating the influence of wetlands (riparian forests for Hatchet, Gumroot Swamp for Little Hatchet) on water quality. Evaporative concentration and dilution appear minor in this basin based on concentrations in the Lake and Prairie Creek. Note that lake water quality observations during the period 5/2000 through 9/2003 are omitted due to a regional drought that ceased flow in both inflow and outflow creeks. Evaporative concentration in Cl^- and other solutes (reactive and otherwise) was observed during that period, during which much of the lake bed was exposed.

Export of carbon (total dissolved – DOC, and labile – BOD; Fig. 72) shows the influence of wetlands vs. groundwater as the flow source. Lake Forest and Little Hatchet Creek (North Branch) are very low in both C fractions, while Hatchet and Little Hatchet (below Gumroot Swamp at SR26) are high. Notably, most C export is in recalcitrant form, with the exception of Prairie Creek, for which ~35% of the C export is labile. Mean concentrations of DOC in Hatchet and Little Hatchet (at SR26) are high, though typical of blackwater creeks regionally.

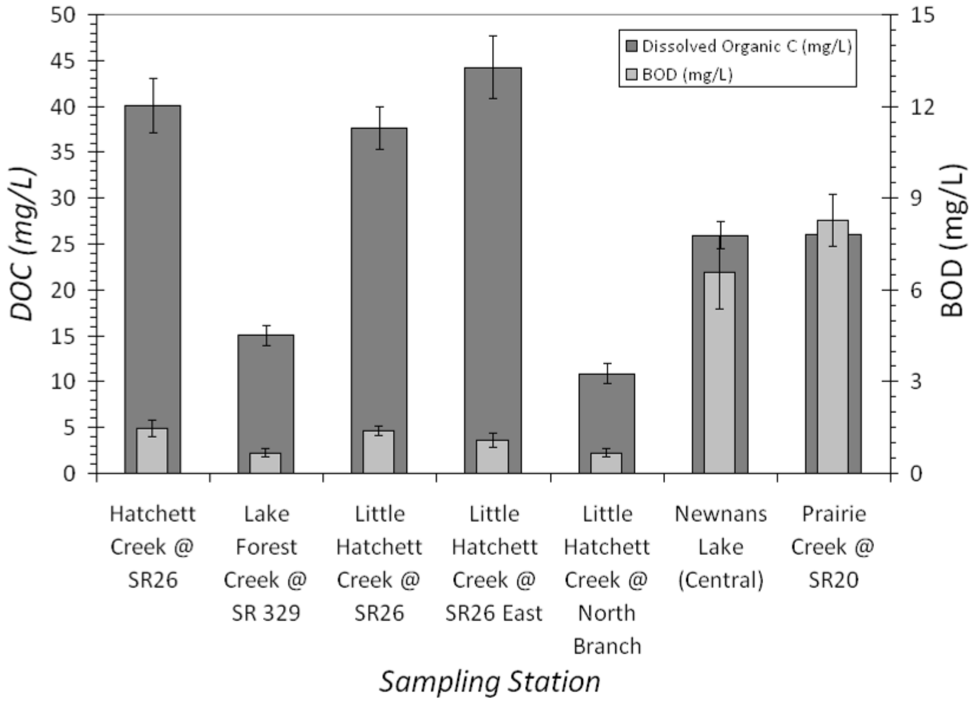

Fig. 72 – Comparison of DOC and BOD observations (archival data) among the creeks draining to and from Newnans Lake.

N loading to the lake is not viewed as a concern because 1) the lake is principally P limited, 2) historical concentrations are so low, and 3) N fixation is apparently the largest fraction of the lake N budget (Fig. 73). Concentrations of NO_x vary dramatically among creeks, but are generally very low (< 150 ppb). Levels are much higher in Lake Forest and Little Hatchet (North Branch) sites, possibly reinforcing the contention that these systems are more affected by groundwater sources. These two creeks are also embedded within the highest LDI regions of the watershed, and may be more heavily loaded by fertilizer runoff.

TKN levels are higher (0.5 – 2 ppm) in the inflow creeks, but almost entirely dissolved organic N that is not immediately bioavailable (Fig. 73). Nearly all the export from Newnans Lake is as organic N. The decrease in NO_x and contemporaneous increases in TKN and water flows from upper to lower Little Hatchet Creek is indicative of the role that wetlands play in landscape assimilation/transformation of N. Note, however, that TMDLs are set on the basis of total nutrient loads (rather than components) reflecting an appreciation for internal transformations in lakes, particularly those with long residence times.

A similar assessment of P concentrations (SRP and TP) in the watershed reveals some expected and some unexpected results (Fig. 74). It was expected that SRP would be largely absent from Prairie Creek flow as most soluble water column P is incorporated into algal biomass and exported as particulate P. We note that the concentrations of TP are comparable to the flow weighted mean of the creek inputs (~ 150 ppb). It was also expected that most of the P input would be in SRP form; little opportunity for algal incorporation occurs in the highly shaded creeks draining to the watershed.

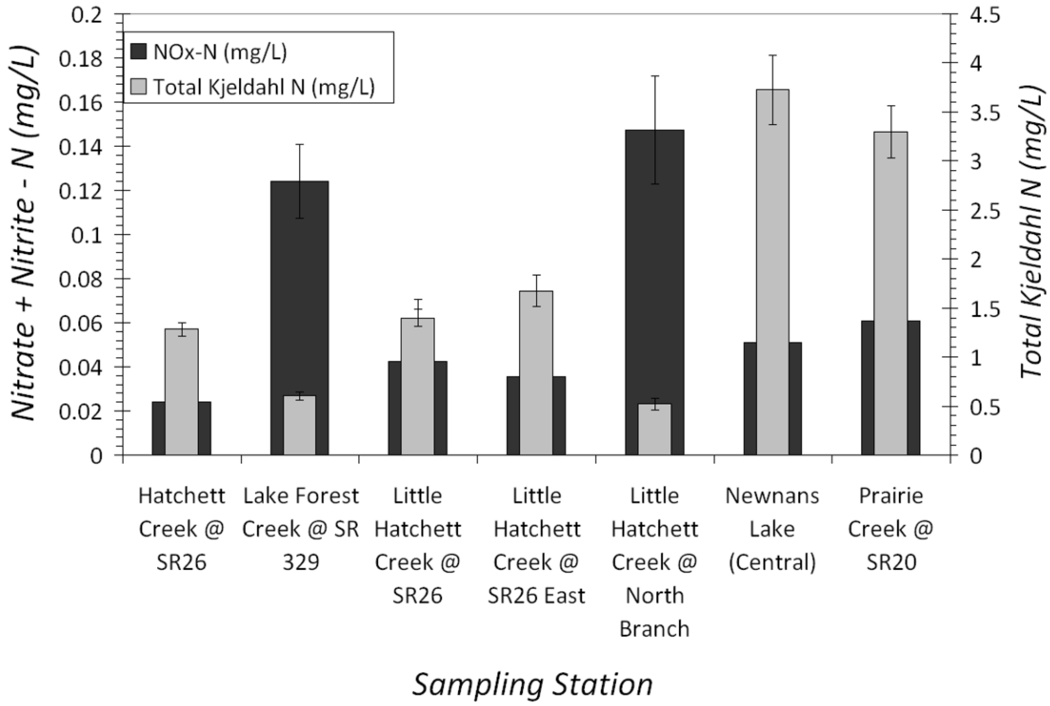

Fig. 73 – Comparison of NO_x and TKN observations (archival data) among the creeks draining to and from Newnans Lake.

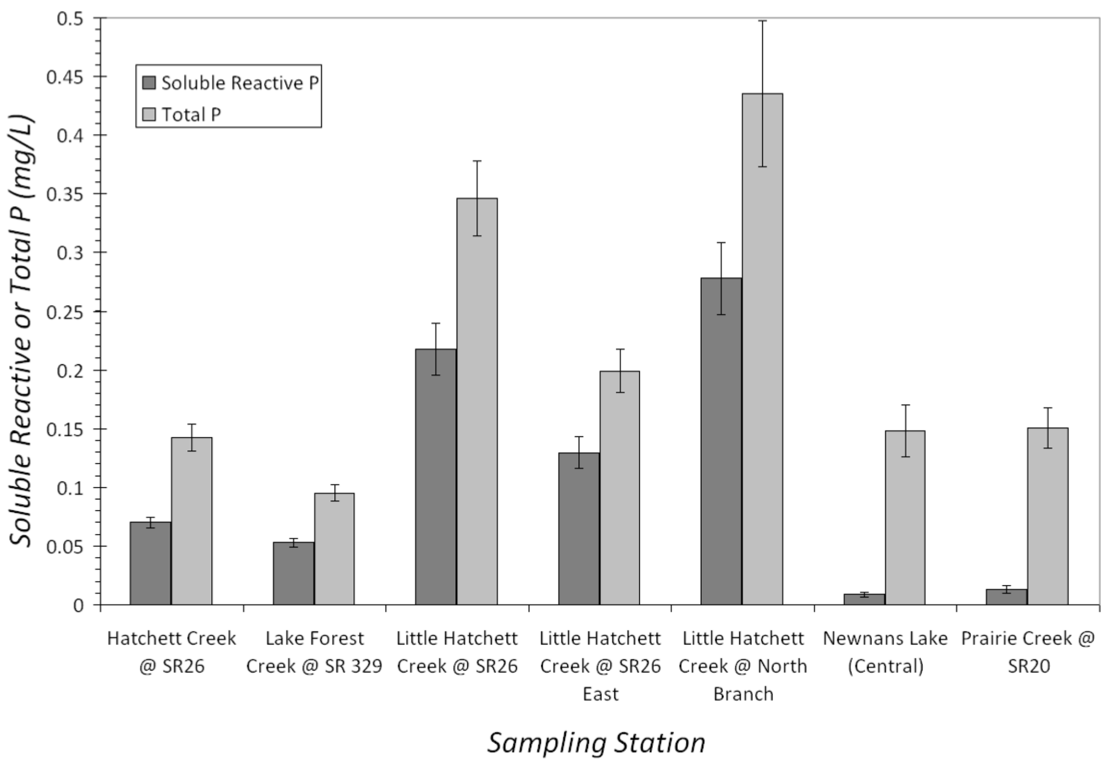

Fig. 74 – Comparison of SRP and TP observations (archival data) among the creeks draining to and from Newnans Lake.

What was not expected were the patterns of SRP (and TP; correlations between these two variables is 0.94), where Lake Forest Creek, which drains an urban area, has the lowest concentration. Moreover, in all cases, the concentrations of P are unusually high for forested drainage. Omernik (1977) predicts a TN concentration for a watershed with 75% cover of 700 ppb (Newnans Lake is markedly lower) and a TP concentration of 15 ppb (Newnans Lake creeks are an order of magnitude – or more – higher). The highest mean historical concentrations, which were observed at Little Hatchet Creek (North Branch), are nearly 300 ppb. We reiterate that the Little Hatchet (North Branch) site appears to derive from groundwater; concentrations of this magnitude suggest an intermediate aquifer source. In contrast, the Lake Forest Creek discharge is typical of Floridan Aquifer water, which is typically low in SRP.

The inferred effect of wetlands (principally Gumroot Swamp) on P removal is much weaker than for N, though still consistent. Additional observation is needed to determine the fate and sustainability of that sink.

Finally, we used archival data to construct a P balance for the watershed, using a period of record (June 1995 to April 2006 – Fig. 75) during which most of the inflow creeks had monthly data. Using estimated flow rates (from nearby or co-located flow stations) and assuming monthly observations applied to all flow for that month, we compared inflow and outflow fluxes of P. The cumulative deficit/surplus was computed as the inflow minus outflow over the entire period of record, and compared with stores of P in the lake water (~ 6.5 metric tons assuming 150 ppb nominal concentration and 4.3E+07 m^3 volume), and lake sediment (5,400 metric tons assuming 1 m sediment depth, 1000 mg/kg sediment concentration and 0.2 g/cm^3 sediment bulk density – Brenner and Whitmore 1998).

Over the 11 year period of record, P exports exceed imports by nearly 20 metric tons (19.6E+03 kg TP),ca. 3 times the lake water P mass. While that discrepancy is a small fraction of sediment P (~ 0.3%), the entire sediment profile is not interacting with the lake water. It may be that internal recycling is sufficient to explain this P discrepancy, but it is also possible that this load arrives from inputs that are not well characterized by the archival data record. The contention of Nagid et al. (2001) that the lake is a net exporter of P is unlikely to be true because of these uncharacterized sources.

The estimated water input deficit over this 11 year period is approximately 2.1E+08 m^3 (Fig. 69). The observed P deficit is approximately 19.6E+03 kg. If the unaccounted for water (e.g., from diffuse groundwater inflow and non-gauged surface sources) carried all of the unaccounted for P, the concentration of that water would be 95 ppb (0.095 mg/L), which is within the range of concentrations observed in the watershed. Continuing observations of intermediate aquifer water and ungauged surface water inputs is a clear priority, but we have determined from preliminary evidence that groundwater contributions to the lake are small. Notably, groundwater contributions to Lake Lochloosa are estimated at 2 orders of magnitude higher than our estimate for Newnans (Hicks et al. 2008)

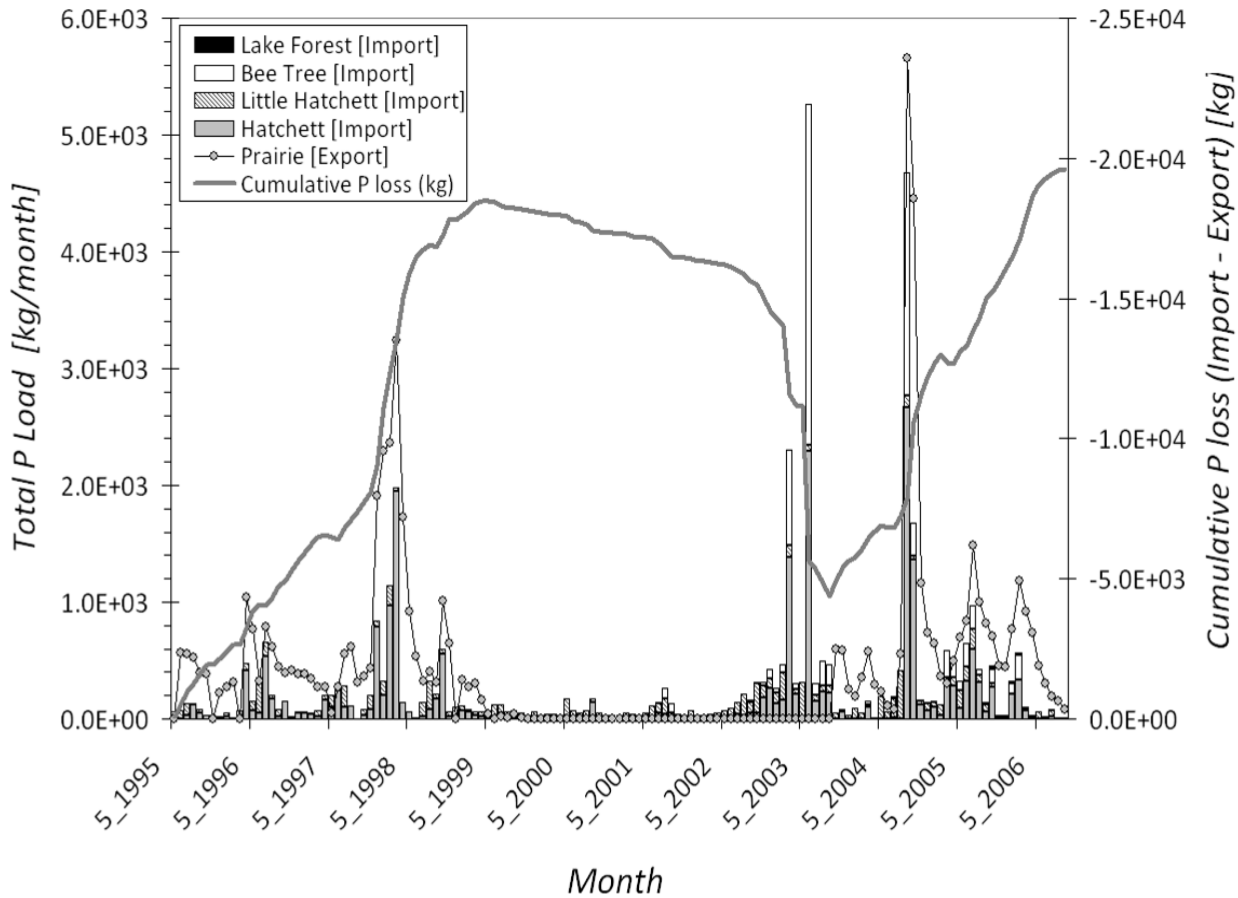

Fig. 75 – Estimated P balance (Imports = bars; Exports = black line with grey circles) between May 1995 and September 2006. The cumulative P deficit (grey line) is just under 20 metric tons.

Analysis of Surface Water Loading

While the outcome of this research was not overtly the construction of a water and P budget for the lake, the data collected to date permit some insight in that regard. The reason that this effort is relevant to the ongoing study of the lake is the contention articulated in Nagid et al. (2001) that the lake is a net exporter of P. The data used to construct that mass balance were from 1998, a period that can clearly be considered a hydrologic anomaly; February 1998 saw the delivery of nearly 16 inches of rainfall, and after April 1998 the region entered a severe drought. Moreover, the analysis considered only the flows from Hatchet and Little Hatchet Creeks, neglecting the flows from some of the smaller ungaged streams. Therefore, it remains unclear whether the contentions in that paper hold under more typical circumstances.

It is not possible, based on the data that we've collected, to fully explore the mass balance of water and P for the lake. The particular reason for this is that while we have more regular nutrient concentration data, we have only intermittent flow data. As gage station data become available for the period of record, we will be able to construct a more complete mass balance, but flow information on many of the ungaged streams will remain unknown. Our approach, instead, was to compile all water quality data that we do have for all stations that comprise the perimeter of the lake (or slightly upstream where necessary for data completeness). On each date for which we have observations at each location (including an observation of no flow at the sampling period) we compiled the fluxes. We report the data only for those dates; that is, we do not interpolate to unsampled days. For the entire period of record, we then sum all the days for which data were available and report the fractional contribution from each source. This was done for flow, total P, SRP and total N. The results are summarized in Fig. 76 through 78.

Fig. 76 summarizes 24 sampling periods (2-4 days) during which all or most of the sampling locations comprising the lake perimeter were sampled. Most of the dates in 2007 were comparatively dry (~ 1,000 to 50,000 m^3/day) while several storm flows in early 2008 yielded between 200,000 and 500,000 m^3/day. Each bar is comprised of multiple flow stations, color coded by site. Overall, the median flow was 12,000 m^3/day, or approximately 4,300,000 m^3/yr. More important that this inherently uncertain number is the fractional contribution from the three gaged stations (Lake Forest, Little Hatchet, Hatchet) vis-à-vis other synoptically measured stations. The inference from Fig. 76 is that these other stations represent a comparatively small fraction of the total flow (ca. 12%). Modeling analysis of these flows suggest a higher fraction (J. Di, SJRWMD, personal communication). This may be due to prevailing climatic conditions during the period of record. Reinforcing this potential is the observation that non-gaged flow paths are important on only 3 dates, all of which were following large storms. Following Tropical Storm Fay in July 2008, nearly all flowpaths to the lake were flowing, and with associated increases in levels in the surficial and intermediate aquifers, may lead to a new flow regime that supports higher diffuse flows. Additional monitoring of both chemistry and flow is required to establish long term flow fractions and thus to construct a credible lake water budget.

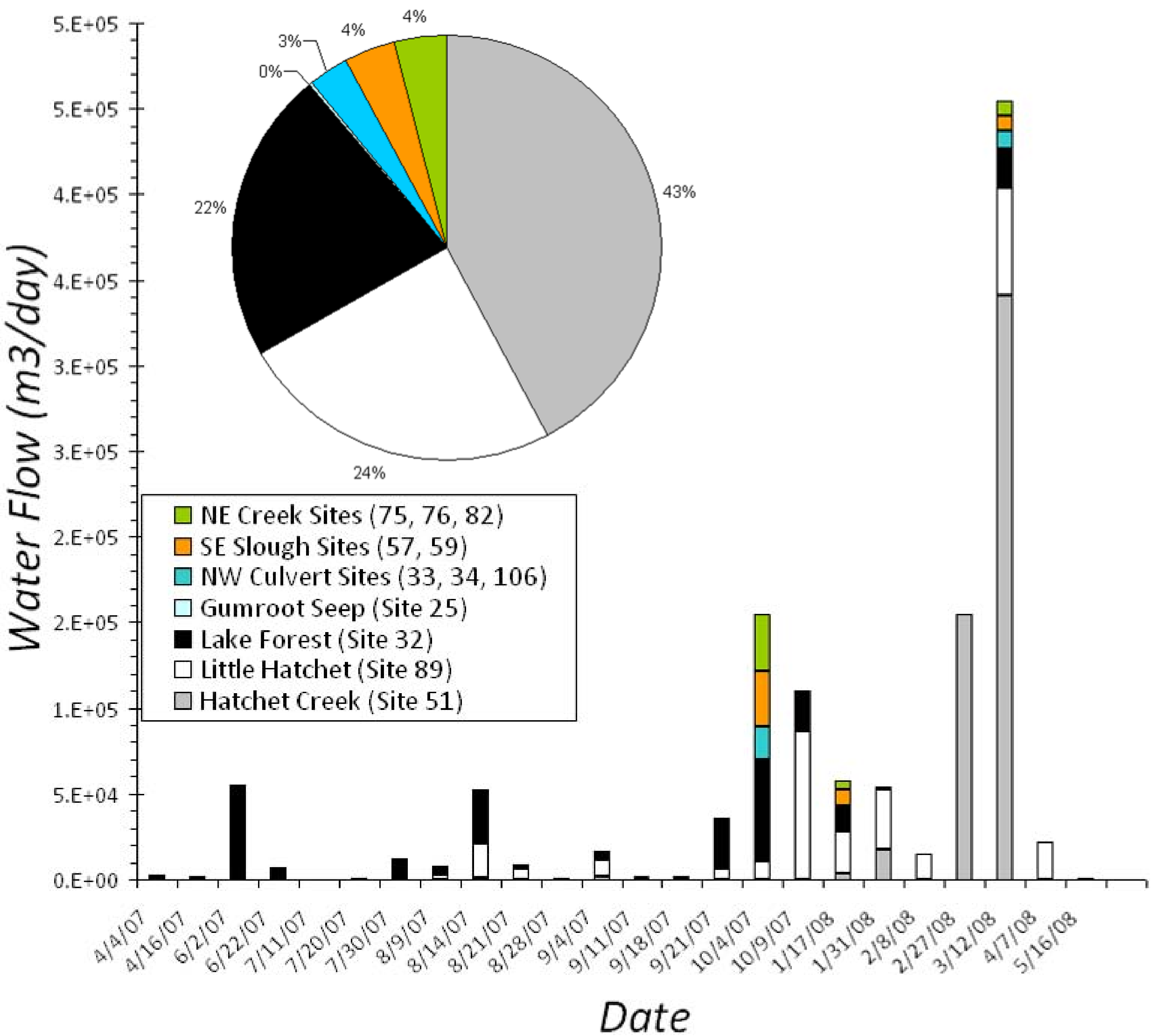

Fig. 76 – Loading of water flows to Newnans Lake from multiple sources for each sampling instance (± 2 days) for which data available. Inset is estimate of total fractional load for the period of record.

A similar analysis of SRP and TP loads to the lake (Fig. 77 A and B, respectively) suggests that while the ungaged creeks comprise a small fraction of total flow, they are markedly more important for elemental budgeting. Overall these systems supply over 20% of the total mass. Moreover, as with flow, this mass fraction is maximized during periods following storms; in one instance (10/4/07) the loading from ungaged sources exceeds those registered from the three main creeks. As such, periods of high flow (such as those during which Nagid et al. (2001) collected samples) might reasonably be inferred to have been stark underestimates of total P loading. It is also important to note that the total load of P is approximately 50% higher than the load of SRP, and that differences are most pronounced following storms, when DOC mobilization is expected.

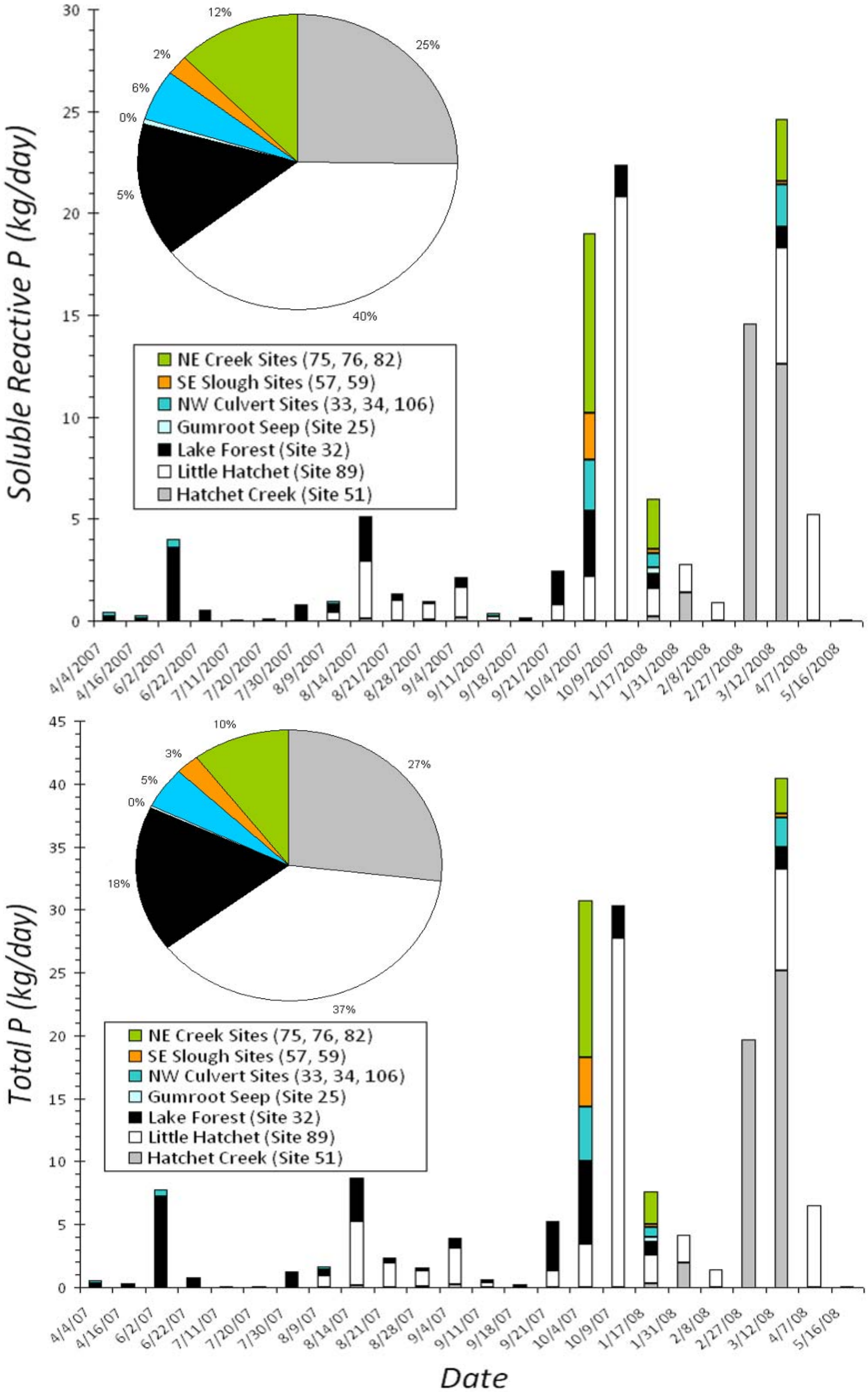

Fig. 77 – Loading of P loads (soluble and total) to Newnans Lake from multiple sources for each sampling instance (± 2 days) for which data available. Inset is estimate of total fractional load for the period of record.

Similar patterns were observed for total N (Fig. 78). The mass loading was dominated by the three main creeks (91% of total mass), but stormflow induced a far higher fractional level of ungaged creek contribution, suggesting that these regions would be generally more important during wetter climatic conditions.

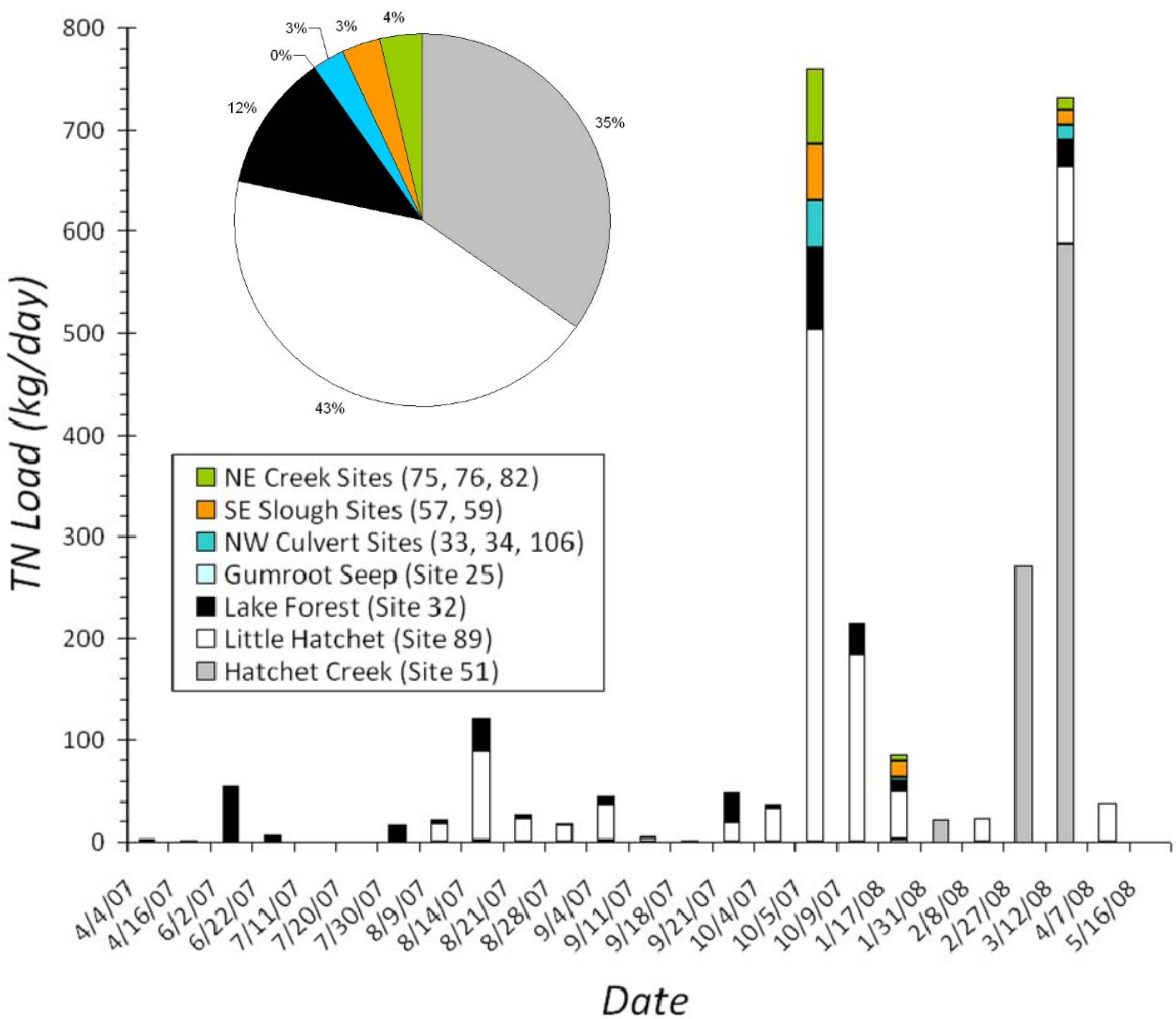

Fig. 79 – Loading of N to Newnans Lake from multiple sources for each sampling instance (± 2 days) for which data available. Inset is estimate of total fractional load for the period of record.

IV. DISCUSSION

The results and noteworthy inferences presented in the previous section lead to the conclusion that a large fraction of the P entering the lake is not of anthropogenic origin, instead coming from weathering of apatite in the Hawthorn Formation. While it is also true that lake water column concentrations have recently increased (Di et al. 2006), the cause for that increase remains uncertain. Strong negative covariance between lake stage and water column concentration may suggest internal recycling processes that maintain higher concentrations at low stage. Structural changes in lake management occurred in 1991 with removal of a control dam at the Prairie Creek outflow. While direct evidence of an effect of dam removal is ambiguous, it is plausible that allowing lake levels to fluctuate naturally coinciding with a period of extreme drought in the late 1990's and early 2000's led to exaggerated contemporary concentrations independent of changes in the lake basin.

Taking as a given the observation that geologic P is a large component of the watershed load, it remains unclear what management actions can most effectively ameliorate loading. One possibility is the use of water flow control management practices that would serve to reduce the velocity and thus stream power of storm flows. There is clear evidence of channel incision in the upper portions of Little Hatchet Creek, and it is possible that accelerated incision increases the magnitude of geologic flowpaths via two mechanisms. First, exposure of apatite to weathering by removing overlying material is likely to accelerate the natural process. Second, physical entrainment of sediment and downstream deposition could be acting. Since our data for dissolved P indicate that this was the vast majority of total P except during storm flows when dissolved organic P was significant, we conclude that the active transport of phosphatic sediments is not well supported. Management of storm flows to reduce the delivery of weathered phosphates and to limit the exposure of unweathered materials may play a role in source load reduction. We note, however, that there is strong evidence for P enrichment in Hatchet Creek also, where impervious surface cover is low so stormwater controls, while important, are unlikely to dramatically change the creek hydrograph.

What can be said unequivocally is that the current state of the Newnans Lake watershed, which is tremendously protective of water quality, should be carefully managed for future anthropogenic load. Even if geologic loads cannot be reduced, increased loading that would be expected with additional development pressure is ill-advised. The rationale for this can come from both the desire to protect the ecological quality of Newnans Lake and control N loading to Alachua Sink, a process that is controlled to a large degree by the P concentrations in Newnans Lake which regulate lake N fixation.

While we cannot establish from these data the efficacy of flow control efforts on reducing overall lake P loading, it is reasonable to use our findings to consider management practices most likely to improve lake water quality. The Vollenweider model for predicting lake water column concentrations of constituents loaded from the watershed (Vollenweider 1968) yields some insight in this regard:

$$TP = \frac{L}{\overline{Z} * (\rho + \sigma)}$$

Where TP is the nominal water column concentration of P in mg/L, L is the area loading rate (mg/m^2/yr), $\overline{Z}$ is the mean lake depth (m), ρ is the lake flushing rate (yr^{-1}) and σ is the P sedimentation rate (yr^{-1}). While we cannot fully populate the elements of this model with the current data, and acknowledging that more sophisticated models exist, we observe that there are two ways to reduce water column P concentrations: reducing the loading rate and accelerating the removal rate. If geologic sources cannot be ameliorated sufficiently to yield a marked reduction of loading and a meaningful decrease in TP concentrations, efforts to increase the denominator of the equation are necessary. These options include increasing lake depth (perhaps by judicious management of lake levels to ensure regular drawdowns), increasing the flushing rate (i.e., by shortening the residence time) or by increasing sedimentation via a) reducing resuspension (e.g., shad harvest, lake level management), b) manual P removal (e.g., shad or sediment extraction) or c) addition of chemical treatments that accelerate flocculation (e.g., alum treatment). Given the well documented relationship between lake stage and TP concentrations, there is some evidence that regulating lake levels higher could help meet water quality goals.

Finally, the evidence from our work and from what limited paleolimnological research has been done suggest that Newnans receives enormous amounts of P naturally. Recent declines in the ecological condition of the lake (i.e., chlorophyll a concentrations in excess of 180 µg/L) may have as much to do with prolonged drought as with human stressors, and, as such, may not respond to management.

Ongoing research in this watershed should focus on two principal unknowns. First, the water and P budget for the lake needs to be refined with better data. Among the key challenges are the placement of current continuous gaging stations (at Sites 39 and 100), both of which are sufficiently distant from the lake to be of suspect value for lake budgeting. Further, increasing the density of measurements to parameterize the relationship between discharge and P concentrations (Figs. 17-19) will allow these to be used as surrogate for what is clearly inadequate monthly sampling. Regular (i.e., monthly) baseflow and stormflow sampling at the main discharge locations around the lake will also allow insight into the climatic controls on P delivery; the current work was performed during a period of significant drought, and evidence from several storms suggests that when a larger portion of the watershed begins contributing runoff, P concentrations actually go up. This means that lake P loading is dramatically increased during periods when the source area for creek flow is larger than has been the case over the period of record for the current work. Examining P loading as a function of antecedent rainfall and surficial aquifer conditions will be useful in determining the role of decadal climate cycles in lake water chemistry. Our contention that drought conditions that have prevailed over the region since 1998 may be substantially relevant to lake water quality declines could be tested with revised ongoing monitoring effort.

V. CONCLUSIONS

Based on the evidence from the samples collected, and put in the geologic, land cover, landscape position and hydrologic regime, we draw five conclusions about the sources of P from the Newnans Lake watershed:

Conclusion 1: Phosphorus loading to Newnans Lake is dominated by geologic phosphate

Lines of Evidence:

a) Land use does not appear to control P concentrations, whereas it does control N concentrations
b) NO_x and SRP concentrations are uncorrelated, as would be expected with non-fertilizer sources of P
c) P concentrations are well predicted by sampling site proximity to the Hawthorn Formation, a major source of geologic P.
d) P concentrations covary with concentrations of fluoride, a weathering by product from apatite minerals in the Hawthorn Formation.

Conclusion 2: P loading is a complex function of flow that depends strongly on the source basin.

Lines of Evidence:

e) Little Hatchet Creek exhibits strong dilution gradients with flow (i.e., flow and SRP and negatively correlated)
f) Lake Forest Creek shows no association between P and flow
g) Hatchet Creek exhibits decreasing concentrations with flow and then a reversal where concentrations increase with flow. Consequences for loads are tremendous. Sources are intermittent tributaries that drain pasture land, and are predicted to interact strongly with the Hawthorn. This line of evidence is confirmed by longitudinal transect sampling that shows maintenance of SRP concentrations with 100-fold changes in flow volume.
h) Loads of P from streams not routinely monitored are significant (ca. 20% of load on average), but increases markedly during stormflows. Load fractions from ungaged portions of the basin may, therefore, be increasingly important during wetter climatic periods that the period of record.

Conclusion 3: Terrestrial soils do not appear to be a major source of P.

Lines of Evidence:

i) Soils (to 1 m deep) do not appear to be significantly enriched with P, particularly compared with P levels in the Hawthorn clays, even where the Hawthorn is predicted to be near the surface.
j) Soils are moderately enriched at depths of 80-100 cm, but still exhibit concentrations 2 orders of magnitude lower than Hawthorn clays.

Conclusion 4: Groundwater P appears to be of geologic origin, but groundwater appears to be a negligible source of water and P to the lake.

Lines of evidence:

k) SRP and F are strongly correlated across 14 perimeter wells.
l) Groundwater flow rates are constrained by moderate hydraulic conductivities and low potentiometric gradients.
m) Flows are variably into and out of the lake, both in time and space. Few obvious patterns of flow control (sign and magnitude) were evident.
n) While SRP concentrations are comparatively high in many of the wells, our estimate of annual load is negative to neutral; that is, P flow is away from the lake (33 kg/yr).

Conclusion 5: Load reduction targeting can be optimized by accounting for landscape position and basin rather than land use alone. Successful load reduction strategies are likely to follow from hydrologic rather than P loading controls (e.g., stormwater management). Where load reduction (i.e., numerator of equilibrium P concentration equations) is not possible, efforts to regulate depth, sedimentation rates and export rates (i.e., denominator of equilibrium P concentration equations) may prove more useful.

Lines of evidence:

o) Headwater areas have significantly lower P concentrations across all basins.
p) Mid-reach areas have significantly elevated P concentrations in Little Hatchet and Hatchet Creeks; difference in Lake Forest Creek were in the same direction but not significant.
q) Terminal wetland areas (data were available only for Lake Forest and Little Hatchet creeks) exhibit lower concentrations than mid-reach sites in the same basin, which may be evidence of some attenuation potential for both SRP and NO_x.
r) Given strong associations between lake stage and SRP, management of stage may provide valuable concentration reductions that both limit phytoplankton concentrations and also reduced downstream N loading (since internal N fixation is an important component of the lake's N budget - Fig. 73).
s) Sink enhancements (shad harvest, alum treatments, sediment removal, etc.) may prove valuable in internal recycling reduction, but need to considered in light of the potential inability to reduce external loads via typical management practices.

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