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TEMPORAL AND SPATIAL TRENDS IN NUTRIENT AND SEDIMENT LOADING TO LAKE TAHOE, CALIFORNIA-NEVADA, USA[1]

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ABSTRACT: Since 1980, the Lake Tahoe Interagency Monitoring Program (LTIMP) has provided stream-discharge and water quality data—nitrogen (N), phosphorus (P), and suspended sediment—at more than 20 stations in Lake Tahoe Basin streams. To characterize the temporal and spatial patterns in nutrient and sediment loading to the lake, and improve the usefulness of the program and the existing database, we have (1) identified and corrected for sources of bias in the water quality database; (2) generated synthetic datasets for sediments and nutrients, and resampled to compare the accuracy and precision of different load calculation models; (3) using the best models, recalculated total annual loads over the period of record; (4) regressed total loads against total annual and annual maximum daily discharge, and tested for time trends in the residuals; (5) compared loads for different forms of N and P; and (6) tested constituent loads against land use-land cover (LULC) variables using multiple regression. The results show (1) N and P loads are dominated by organic N and particulate P; (2) there are significant long-term downward trends in some constituent loads of some streams; and (3) anthropogenic impervious surface is the most important LULC variable influencing water quality in basin streams. Many of our recommendations for changes in water quality monitoring and load calculation methods have been adopted by the LTIMP.

(KEY TERMS: biogeochemistry; environmental impacts; lakes; rivers/streams; monitoring; statistics; environmental sampling; nutrients; sediment; eutrophication; total load.)

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INTRODUCTION

Lake Tahoe, a large ultra-oligotrophic lake in the central Sierra Nevada, is world renowned for its clarity and deep blue color. Over the last half-century, parts of the watershed have been developed for residential and commercial use, and the lake has undergone progressive eutrophication and loss in clarity. The growing water quality problems of the lake have been studied intensively since the early 1960s (Goldman, 1981), and have attracted considerable political attention. In spite of increased land use controls and export of treated sewage effluent from the basin, primary productivity of the lake (carbon fixation in g/m^2/yr) has increased fivefold since 1970 (Schladow, 2015). Since the early 1960s, its clarity declined at an average rate of 0.25 m/yr (Jassby *et al.*, 1999; Reuter *et al.*, 2003), but in recent years, the trend in clarity has leveled off, and

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even (during winter months) improved slightly (Schladow, 2015). Water quality problems in the basin are currently addressed through a "total maximum daily load" (TMDL) program administered by the Lahontan Regional Water Quality Control Board under the Clean Water Act (Lahontan and NDEP, 2010).

Since 1980, the Lake Tahoe Interagency Monitoring Program (LTIMP) has measured discharge and sampled water quality at over 20 stations on tributary streams in the Tahoe Basin (Alvarez *et al.*, 2006). Concentrations of suspended sediment (SS) and various forms of nitrogen (N), phosphorus (P), and iron have been measured (Rowe *et al.*, 2002). The purpose of the program has been to document long-term trends in water quality of both the major and minor tributaries to the lake, and thus provide a basis for public policy and management decisions that may affect clarity. A parallel program—the Regional Stormwater Monitoring Program—has shown that stormwater runoff from urbanized areas is a major contributor to water quality problems in the Tahoe Basin (Coats *et al.*, 2008; Tahoe Resource Conservation District *et al.*, 2014).

Several problems have limited the usefulness of LTIMP and over time some of these problems have become more apparent. Based on our current understanding of water quality and its effect on lake clarity, these problems include (1) changes in the chemical methods and chemical species analyzed, which complicate efforts to measure long-term trends; (2) changes in the time of sampling for some streams, which may have introduced bias in records for flow-driven constituents such as SS and total phosphorus (TP); and (3) use of statistical models that produce inaccurate and imprecise estimates of total constituent loads, without recognition or quantification of the uncertainty.

Two methods of load estimation have been used by LTIMP: the worked record and a simple rating curve method. The method of the worked record, used in the early days of LTIMP, may be thought of as an interpolating method. In this method, the time trace of discharge and concentration are plotted together, and the mean daily concentration is interpolated for days on which samples were not collected. This allows the technician to adjust concentrations up or down to take account of discharge variation. With a good database and relatively low intradaily variability in concentrations, the method is accurate in the hands of a skillful technician, but the results may not be reproducible, and it does not lend itself to an estimate of sampling error (Cohn, 1995). As mean daily concentration must be estimated from instantaneous concentration, errors may be introduced for constituents that vary widely over the course of a day.

Beginning in 1988, the simple rating curve method replaced the worked record method, and has been used since by the University of California-Davis Tahoe Research Group to calculate total nutrient loads for the Tahoe Basin streams (Byron and Goldman, 1989). Instead of regressing the log of concentration against that of discharge (log C_i *vs.* log Q_i), instantaneous load (L_i) is calculated as the product C_i Q_i, and regressed against log Q_i. The resulting relationship (with appropriate correction for retransformation bias) is used to estimate daily loads from mean daily discharge, and the estimates are summed over days for the water year. The load estimates by this variant are mathematically identical to those obtained by a regression of log C_i *vs.* log Q_i, but the apparent high correlation between log L_i and log Q_i is a "spurious self-correlation" (Galat, 1990) and can mislead hydrologists into using models that have no explanatory value for concentration.

The purpose of this project is to quantify the spatial and temporal patterns in the transport of N, P, and SS in streams of the Tahoe Basin. In the first stage, we reviewed the existing water quality monitoring program and recommended programmatic changes. This stage included (1) identifying and then removing or quantifying the sources of bias in the LTIMP dataset; (2) developing and comparing different models for calculating total constituent loads; and (3) quantifying the sampling error and required sample sizes for given levels of confidence in estimates of total load. Details of the first stage are described in Coats and Lewis (2014) and summarized in the Supporting Information. In the second stage, we used the best selected models to recalculate total loads of nitrate-N (NO_3-N), total Kjeldahl nitrogen (TKN), ammonium-N (NH_4-N), soluble reactive phosphorus (SRP), TP, and SS, for 20 stations and up to 39 years. We then tested for time trends in total load residuals (after removing the effects of total annual and annual maximum daily discharge), developed and tested hypotheses that might explain the observed trends, and examined patterns in the forms of N and P transport in basin streams. Although an explicit evaluation of the Tahoe Basin TMDL program was not the purpose of this project, the results are being applied to make the LTIMP a more useful and cost-effective program for water quality management in the Tahoe Basin, and may ultimately be used in future revisions of the TMDL program.

STUDY AREA

Lake Tahoe lies at an elevation of 1,898 m in the central Sierra Nevada, astride the California-Nevada border (Figure 1). Volume of the lake is 157 km^3, and its surface area is 501 km^2, 38% of the total basin area of 1,313 km^2. Mean annual precipitation ranges from over 140 cm/yr in watersheds on the west side of the basin to about 67 cm/yr near the lake on the east side of the basin (Daly *et al.*, 1994; see Figure 2). Most of the precipitation falls as snow between November and April, although rainstorms combined with rapid snowmelt account for the highest flows and occasional floods. There is a pronounced annual runoff of snowmelt in late spring and early summer, the timing of which varies from year to year and by location in the basin, but is shifting toward earlier dates in response to climate change (Coats, 2010). In some years, summertime monsoonal storms from the Great Basin bring intense rainfall, especially to high elevations on the east side of the basin. Winter storms often bring rain to the urbanized areas close to the lake with snowfall at higher elevations. Soils and vegetation of the Tahoe Basin are briefly described in the Supporting Information.

About 6% of the basin land area has been developed for residential and commercial uses, especially along the north, south, and west shores. The rate of development was especially intense during the 1960s and 1970s, but has since slowed due to land use controls.

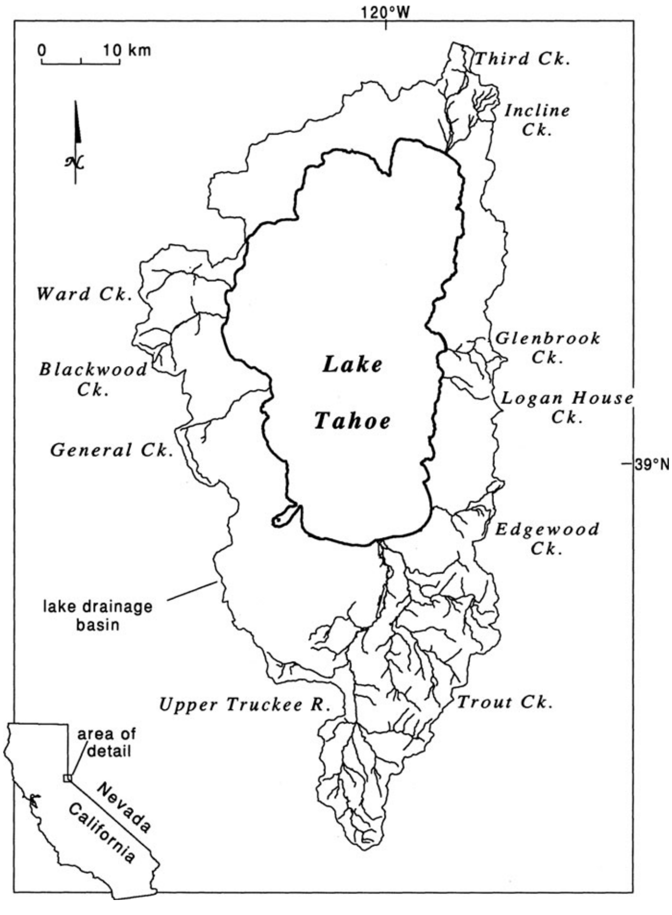

FIGURE 1. Map of the Tahoe Basin, Showing Locations of the Lake Tahoe Interagency Monitoring Program Watersheds.

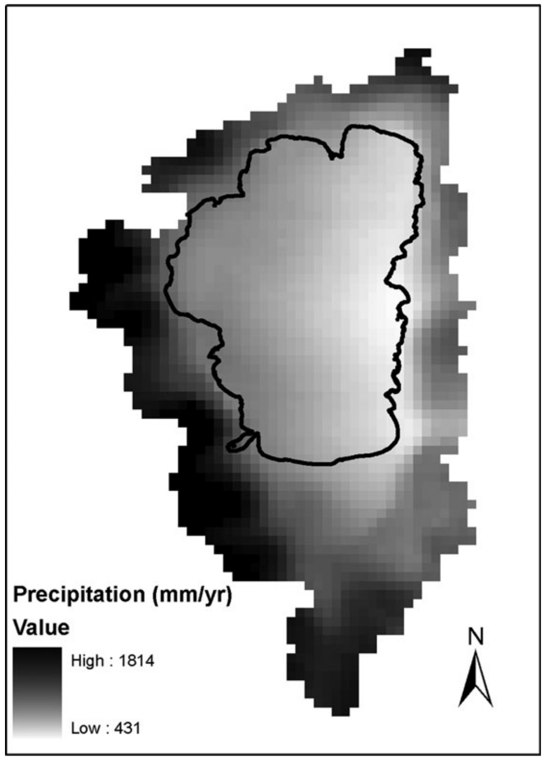

FIGURE 2. Mean Annual Precipitation in the Tahoe Basin, Based on the Parameter-elevation Relationships on Independent Slopes Model (PRISM) (Daly *et al.*, 1994).

METHODS

Discharge Measurement and Sampling

A network of U.S. Geological Survey (USGS) stream gaging stations forms the basis of the LTIMP stream monitoring (Rantz 1982; Sauer and Turnipseed, 2010; Turnipseed and Sauer, 2010). Table 1 lists the stations that have been included, and indicates a maximum record length of 43 years. Table SI-1 in the Supporting Information shows the full USGS site names and site numbers corresponding to the numbers in Table 1. Figure 1 shows the streams sampled in the LTIMP. Locations of the primary and secondary stations listed in Table 1 are shown on Figure SI-1, in the Supporting Information. Primary stations are closer to the lake near tributary mouths and secondary stations are higher in the watersheds. Note the significant cutback in the number of stations starting in 2011. Stream gaging and water quality sampling are described in the Supporting Information.

TABLE 1. List of Lake Tahoe Interagency Monitoring Program (LTIMP) Stations with USGS Code, LTIMP Code, and Range of Years Sampled. A few stations are missing some years, not shown. Primary station names are in italics. TC-1 is used for sampling; corresponding discharge measurements are from TC-4. For a list including temporary and miscellaneous stations, see Coats and Lewis (2014) or Rowe *et al.* (2002). Full USGS station names are available in the Supporting Information.

			Water Year		
Tributary Name	**LTIMP Sta. Name**	**USGS Sta. ID No.**	**Begin Record**	**End Record**	**Length of Record**
Blackwood	*BC-1*	10336660	1974	2012	39
Edgewood	*ED-1*	10336765	1984	2002	19
Edgewood	ED-3	103367585	1989	2002	14
Edgewood	ED-5	103367592	1990	2011	22
Edgewood	*ED-9*	10336760	1992	2011	20
General	*GC-1*	10336645	1980	2012	33
Glenbrook	*GL-1*	10336730	1972	2011	40
Incline	*IN-1*	10336700	1970	2012	43
Incline	IN-2	103366995	1989	2006	18
Incline	IN-3	103366993	1989	2011	23
Logan House	*LH-1*	10336740	1984	2011	28
Trout	*TC-1*	10336790	1972	2012	41
Trout	TC-2	10336775	1989	2011	23
Trout	TC-3	10336770	1990	2011	22
Trout	TC-4	10336780	1974	2002	29
Third	*TH-1*	10336698	1970	2012	43
Upper Truckee	*UT-1*	10336610	1970	2012	43
Upper Truckee	UT-3	103366092	1989	2011	23
Upper Truckee	UT-5	10336580	1989	2011	23
Ward	WC-3A	10336674	1991	2011	21
Ward	WC-7A	10336675	1989	2003	15
Ward	*WC-8*	10336676	1972	2012	41

Atmospheric Deposition

From 1983 through 2013, the Tahoe Environmental Research Center (TERC) operated a precipitation gage and Aerochem Metrics 301 Wet/Dry Collectors (Bushnell, Florida) for wet and dry atmospheric deposition at lake level, near the mouth of Ward Creek. Samples were usually collected within 24 h of a precipitation event, transported on ice to the laboratory at Tahoe City and later at Incline Village (Nevada), and analyzed for NO_3-N, NH_4-N, and SRP, using the same methods as for samples of Lake Tahoe water. Annual atmospheric loads of dissolved inorganic nitrogen (DIN) and SRP were calculated separately for wet and dry deposition, and provide an interesting comparison with stream loads. A second gage and bulk precipitation collector was operated in upper Ward Valley from 1980 through 2006. The DIN record from that collector was used to extend the lake-level record by regression back to 1981 ($R^2 = 0.62$; S.E. = 0.26 kg/ha/yr; or 13% of the average DIN deposition rate at the lake-level station).

Sample Analysis

Since its inception the LTIMP has made a number of changes in sampling methods, constituents sampled, and analytical methods used to measure them. Such changes are necessary to take advantage of changes in technology and in our understanding of the processes that influence lake clarity. The changes, however, create problems for analyzing long-term trends in water quality as detection limits, precision, and accuracy may change with a change in methods. The currently analyzed constituents (WY 2014) include suspended sediment concentration (SSC), NO_3-N, NH_4-N, TKN, SRP, dissolved P, TP, and fine sediment (particles less than 20 μm in size). (The analytical method for NO_3-N includes NO_2-N, although the latter rarely reaches detectible levels; hereafter both are called "NO_3-N.") From water years 1989-1993 and 2003-2011, Kjeldahl nitrogen (KN) was analyzed in both filtered and unfiltered samples, providing estimates of particulate organic nitrogen (PON) and dissolved organic nitrogen (DON). Table SI-2 in the Supporting Information shows the methods and references for constituents that are currently analyzed, plus total hydrolyzable phosphorus (THP). For more detail, see Liston *et al.* (2013).

Sources of Bias in Measurement of Total Load

Two sources of bias have been identified in LTIMP stream chemistry data. First, changes in analytic methods for NO_3-N and TP have affected the record for those constituents. From 1976 to April 2003, NO_3-N was analyzed by reducing it to nitrite, and developing a color for photometric analysis. In 2003, it was found that chemical interference (possibly from divalent cations) in streamwater samples resulted in inefficient recoveries, and that addition of a catalyst —pyrophosphate with copper—gave better results. This catalyst was used in subsequent analysis. To provide a basis for adjusting the old data, NO_3-N was measured by both the old and new methods in 2,370 pairs of samples from all LTIMP stations, between 2003 and 2008. A test for homogeneity of the regression coefficients showed significant differences between stations, possibly because the concentrations of interfering cations varied between stations. Separate regression equations for each station were thus used to adjust the old data to the value estimated for the new method. Details of the adjustment procedure and the history of the analytic methods are given in the Supporting Information.

From 1980 to 1988, THP rather than TP was measured. THP involved digestion of samples with sulfuric acid and spectrophotometric analysis of the resulting orthophosphate. The TP digestion uses acid persulfate, and breaks down compounds that are not dissolved in the sulfuric acid digestion. The LTIMP THP data have been adjusted upward to TP by linear regression (Hatch, 1997; see Supporting Information for details).

The second source of bias is related to a change in the times of sampling throughout the day and night. Due to (very real) safety concerns, sampling in the dark was cut back or discontinued for streams in Nevada, as well as Trout Creek and the Upper Truckee River (UTR) in about 1989. The TERC, however, has continued nighttime sampling in Ward (WC-8), Blackwood (BC-1), and General (GC-1) Creeks, and the USGS has continued to sample as late as 21:00 around the summer solstice. In the larger watersheds, especially in the latter days of the snowmelt season, the daily snowmelt pulse (which carries most of the daily water volume with higher constituent concentrations and loads) begins in late afternoon and may peak after midnight. Daily hysteresis has been observed in these basins for SS and P concentrations (Stubblefield, 2007). In the presence of hysteresis, discharge-concentration rating curves are sensitive to the relative abundance of samples collected on rising and falling limbs of the hydrograph. A change in frequency of nighttime sampling could alter the likelihood of sampling at peak flows but, more importantly, it could systematically shift the rating curves, introducing (or masking) time trends in total load estimates. To test the possible effect of discontinuing nighttime sampling on estimates of total load, we calculated loads for the five constituents at WC-8, BC-1, and GC-1, with and without the inclusion of samples collected between 18:00 and 09:00, from 1989 onward. The time-of-sampling bias problem is discussed in detail in Coats and Lewis (2014). The bias cannot be removed from the data, but the problem in time trend analysis can be avoided by working with only the daytime samples.

Tests of Load Calculation Methods

For several reasons, the available and commonly used methods for calculating total annual constituent loads are not satisfactory for use in the Tahoe Basin. First, it has long been known that loads computed using simple rating curves, even with bias corrections for retransformation, are prone to very large errors (*e.g.,* Walling, 1977; Walling and Webb, 1988). Inclusion of covariates that characterize hysteresis at daily and seasonal time scales has the potential to improve simple regression estimates. Second, the most accurate and precise regression method for particulate constituents (TP and SS) may give imprecise and biased estimates for the dissolved constituents (SRP and NO_3-N) (Coats *et al.*, 2002). Third, the simple rating curve and its variants do not provide explicit measurements of error or confidence intervals. The agency staffs have been seeking methods that will measure the degree of uncertainty in load estimates, and that will help them improve the water quality sampling program (LTIMP). Fourth, off-the-shelf programs do not provide an explicit way of dealing with sources of bias in the datasets, especially the possible bias introduced by cessation of nighttime sampling.

With sources of bias due to changes in analytical methods identified and removed to the extent possible, we undertook to find and apply the most precise, unbiased, and cost-effective method for calculating total constituent loads and watershed yields (load per unit area). This included (1) generating synthetic datasets using turbidity, discharge, and time of year as explanatory variables for the concentration of different forms of N, P, and suspended and fine sediment; (2) resampling the synthetic concentration datasets (to which random error was added) and part of the historic record in experiments to compare the accuracy of different load calculation models; (3) identifying the best load calculation models for estimating each constituent load; (4) using the best models to recalculate total annual loads for all constituents and stations over the period of record; and (5) dividing total annual loads by watershed areas to allow comparison of yields.

The simulations of concentration and calculations of total load allowed us to examine the relationships between sample size and error in new load estimates for each constituent. We found, for example, that with 25 samples per year, one can be 90% sure that the true annual load of TP is within $\pm$20% of the value estimated using the multiple regression. We also found that measurement of continuous turbidity offers the opportunity to improve the efficiency of measuring TP and SS loads. For SSC at the 90/20 level, the required sample size drops from 67 to 20. Table SI-7 in the Supporting Information shows the relationship among confidence limits, error about an estimated mean, and sample size, for the constituents and alternative load calculation models.

Selecting a Load Calculation Model

Based on the tests discussed in Coats and Lewis (2014) and the Supporting Information, two methods of calculating total annual loads were selected and applied. For dissolved constituents (NO_3-N, NH_4-N, and SRP) and TKN, the Period Weighted Sampling (PWS) method (Dann *et al.*, 1986; Coats *et al.*, 2002) was chosen. In this method, each two successive concentrations are averaged, multiplied by the cumulative discharge between sampling times, and the resulting load increments summed over the water year. For the particulate constituents (TP and SSC), five alternative regression models for the log of concentration were fit to each combination of station and water year, and the best model was chosen automatically on the basis of Gilroy's mean square error (GRMSE; Gilroy *et al.*, 1990). GRMSE is computationally intensive but utilizes information in the prediction dataset as well as goodness of fit to the sample data to assess the error of the estimated flux. The five regression models that we compared are as follows:

1. $\log(c) \sim \log(q)$
2. $\log(c) \sim \log(q) + \log(\mathrm{MDQ}/\mathrm{MDQ}_1)$
3. $\log(c) \sim \log(q) + \log(\mathrm{MDQ}/\mathrm{MDQ}_1) + D$
4. $\log(c) \sim \log(q) + \log(\mathrm{MDQ}_1)$
5. $\log(c) \sim \log(q) + D$

where c is concentration, q is instantaneous discharge, MDQ is mean daily discharge, MDQ_1 is mean daily discharge on the day before the measurement, and D is day number since start of the water year (October 1). Instantaneous discharge data were available only at sampling times, so MDQ had to be substituted for q during the prediction step for estimating loads. MDQ_1 helps characterize a sample as belonging to the rising or falling side of a snowmelt cycle or rainfall event. D was included to index seasonal depletion of available sediment.

The science of watershed flux estimation is advancing rapidly with three new software tools in development while our analysis was underway. The Weighted Regressions in Time, Discharge, and Season (WRTDS) is a complex procedure that uses neighborhood weightings to fit a three-dimensional nonlinear model for concentration (Hirsch *et al.*, 2010). The composite method (Aulenbach, 2013) improves upon regression with autocorrelated residuals by augmenting the predictions with time-interpolated regression residuals. Both of these methods are promising and need to be evaluated for future applications in the Tahoe Basin. We are collaborating with USGS to compare WRTDS results with ours. The LOADEST method (Runkel *et al.*, 2004), implemented only in Fortran until very recently, employs multiple regression methods and incorporates provisions for censored data, nonnormal data, and for retransformation bias correction. Because of its Fortran implementation, it was impractical to utilize LOADEST in our resampling experiments or in estimation of ~3,200 constituent loads (20 stations by 5 constituents by 32 years). However, the only censoring in our data was for SSC; omitting censored values less than 1 mg/L produced more accurate load estimates (Coats and Lewis, 2014) than coding them to an arbitrary value. We made no parametric assumptions in computing our load estimates using least squares regression and the smearing (Duan, 1983) correction for retransformation bias. Our use of GRMSE in the model selection phase is a computationally intensive innovation developed for this project.

Table 2 shows the comparison of total loads averaged across stations and years (1990-2011) by the simple rating curve with average loads by the new methods. For all constituents and water years (1990-2011), the simple rating curve estimates higher total loads than the new method, in some years by more than 150%. Fortunately, however, nutrient loads for the TMDL were calculated using the "Load Simulation Program in C" (LSPC), a distributed hydrologic

TABLE 2. Comparison of Average Estimated Loads (averaged across stations and years) in the 10 Primary LTIMP Streams (1990-2011) by the Simple Rating Curve (SRC) with the New Modified Methods.

Constituent	Percent Difference 100 × (SRC − New)/New
TKN	65.6
NO_3-N	70.1
NH_4-N	55.3
SRP	45.5
TP	24.8
SS	44.5

Note: TKN, total Kjeldahl nitrogen; NO_3-N, nitrate-N; NH_4-N, ammonium-N; SRP, soluble reactive phosphorus; TP, total phosphorus; SS, suspended sediment.

model calibrated with runoff and concentration data (Lahontan and NDEP 2010; Riverson *et al.*, 2013). The percent differences in the TMDL estimates from the new load estimates (calculated as 100 × (LSPC−New)/New) for DIN, total nitrogen (TN), total organic nitrogen (TON), SRP, and TP were, respectively, −8, 30, 24, 106, and 13%. On average, the LSPC overestimates SRP, probably because its concentration does not vary much with discharge, but the LSPC uses a rating curve approach for surface and subsurface flow. Our SRP load estimates are based on the more appropriate PWS method.

Simon (2008), in the TMDL study on fine sediment loads to Lake Tahoe, used different rating curves for different levels of discharge, with different curves for the periods before and after the large flood of January 1997. SS loads are not included in the TMDL, and direct comparison of the new SS load estimates with Simon's estimates is complicated by his use of calendar year rather than water year.

Testing for Spatial and Temporal Patterns in Nutrient and Sediment Transport

The recalculated total loads allow us to examine spatial and temporal patterns in SS and in the different forms of N and P carried in Tahoe Basin streams. The ratio of DIN load (NO_3-N plus NH_4-N) to total nitrogen load (TKN plus NO_3-N) influences the availability of N to stimulate algae growth in the stream and lake, and may be quite different in the Tahoe Basin compared with streams from other regions. Likewise, SRP is more readily available to algae than the particulate phosphorus (PP = TP-SRP)). The DIN:TN and SRP:TP ratios were calculated from total loads for each station and water year.

As constituent ratios are unlikely to be normally distributed, analysis of variance (ANOVA) would be inappropriate for testing for differences between watersheds. Instead, we used the nonparametric Friedman test, which ranks the watershed averages across years, sums the ranks by watershed, and tests for differences among rank sums (Helsel and Hirsch, 2002; Gwet, 2011). Watersheds with rank sums that were not significantly different from each other in *post hoc* tests were then grouped together, and the groupings used as the basis for explanatory hypotheses.

The fraction of TON that is dissolved rather than particulate may also affect the availability of N for algae growth. Although there were not enough data from filtered KN samples to calculate DON loads, the discharge-weighted annual mean concentrations of DON and TON were calculated for each station and water year (1989-1993 and 2003-2011; n = 2,206) from the instantaneous discharge and concentration data. The discharge-weighted concentrations were then averaged across the 14 water years and across the 10 primary stations. The stations were sorted into two groups on the basis of the ratios of the averages of DON and TON, and the means of discharge of these two groups compared with a t-test. It is assumed that the particulate fraction (PON/TON) is 1 − DON/TON.

The ratio of total PP load (in kg) to suspended sediment load (in metric tons) is a measure of the average concentration of P in the suspended sediment. This calculation is possible because P (SRP and TP) and SSC were measured in samples collected within minutes of each other, and the loads of both were calculated from the same discharge values. For each station and water year (1989-2011), SRP load was subtracted from the TP load, and divided by the SS load to estimate the concentration (in parts per thousand) of PP in the suspended sediment.

To test relationships between land use/land cover (LULC) variables and average annual loads of nutrients and sediment, we first created a set of 19 independent subwatersheds by subtracting the average annual loads and flow at upstream stations (where they existed) from those at downstream stations. Next, we regressed total annual load (1989-2011) for each subwatershed and constituent against total annual flow, and tested the residuals from this regression for normality with the Kolmogorov-Smirnov test (S-Plus 6.1, Venables and Ripley, 1994). Residuals for all but SRP were not significantly different from normal. We then screened a suite of LULC explanatory variables using step-wise multiple regression. The LULC variables screened were developed for each of 19 subwatersheds from GIS datasets for the Tahoe Basin (Cartier *et al.*, 1994), and are defined in Coats *et al.* (2008). To reduce collinearity among explanatory variables, the final candidate models were chosen to include only explanatory variables with correlation coefficients (r) < 0.25.

It is well known that p-values obtained from stepwise regression are severely downward biased. Therefore the best model for each response variable was tested for significance using the more conservative permutation method, which has been shown to be unbiased and consistent (Finos *et al.*, 2010). In this method, the response variable (*e.g.,* residual of the average annual TKN load for each of the 19 subwatersheds in the dataset) was randomly permuted and reassigned to the original predictor matrix 10,000 times. For each permuted dataset, an all-possible subsets regression procedure was iterated, and the fraction of regressions with p-value smaller than the value found for the original unpermuted dataset was taken to be the significance level for the candidate model. Intuitively, the permutation test significance level is the probability that a p-value as low or lower than that obtained for the original unpermuted dataset could have been obtained by chance, *i.e.,* from a dataset where no real relationship exists between the response variable and the predictor matrix.

We also tested for relationships between the ratio of SRP load to PP load and the LULC variables and average total annual discharge. The SRP:TP ratios were logit transformed (Warton and Hui, 2011). In a plot of SRP load *vs*. PP load, Logan House Creek (LH-1), the primary station with the smallest average total annual discharge, fell well below the trend line for the other stations, and was eliminated as an outlier.

Perhaps the greatest value in recalculating loads by a consistent method is the opportunity it affords to test for time trends. The agencies responsible for managing land and water resources in the Tahoe Basin are particularly interested in knowing if stream-borne loads of nutrients and sediment are trending up or down. We have chosen to address this question directly using annual loads rather than sample concentrations to eliminate the issue of autocorrelation without discarding data, while retaining respectable sample sizes due to the lengthy period of record. Aggregating greatly reduces variability, which offsets some of the loss in statistical power that comes with reduced sample sizes. To evaluate historic trends, we developed regression models for annual constituent loads to explain the natural variability due to weather as characterized by annual maximum daily flow (peak) and total annual discharge (flow). Subscripts j refer to individual gaging stations (*e.g.,* peak_j and flow_j). Significant interactions between location and annual peaks and flows were included in these models by permitting coefficients to vary by watershed (*i.e.*, $b_{3\text{j}}$ and $b_{4\text{j}}$). The model with all potential terms is represented as follows:

$$\begin{aligned}\log(\text{load}_j) = {} & b_{0j} + b_1 \log(\text{flow}_j) + b_2 \log(\text{peak}_j) \\ & + b_{3j} \log(\text{flow}_j) + b_{4j} \log(\text{peak}_j)\end{aligned} \tag{1}$$

This is a generalization of a simpler model based on the idea of an annual discharge-weighted mean concentration, $\text{QWM}_j = \text{load}_j/\text{flow}_j$. The simple model $\log(\text{QWM}_j) = b_{0\text{j}}$, which can be rewritten as $\log(\text{load}_j) = b_{0j} + \log(\text{flow}_j)$, assumes that $b_1 = 1$ and $b_2 = b_{3j} = b_{4j} = 0$ for all j. Our model is able to explain far more variability in the loads by estimating all coefficients from the data. In the best model for each constituent one or more of these terms in Equation (1) are not significantly different from zero and were therefore eliminated (Table 3). The residuals from each of these models were tested by station for monotonic trend using the "adjusted variable" Mann-Kendall test (Alley, 1988) recommended by Helsel and Hirsch (2002). Alley's test is basically a Mann-Kendall test on the partial regression plot of log(load) *vs.* water year. The partial regression plot shows the effect of time on load_j after accounting for possible changes in the other predictors (flow_j and peak_j); it is created by regressing both log(load) and water year on the same set of predictors and plotting the two sets of residuals against one another. The relationship is tested for monotonic trend using the Mann-Kendall test. As there were 20 stations and 20 trend tests, the family-wise Type I error rate was kept to 0.05 using the Bonferroni correction (Miller, 1981), so the critical p-value was set to $\alpha = 0.05/20 = 0.0025$ for each test. Because of the possibility that time trends could be induced by reduction in nighttime sampling, trends were analyzed for day-only samples.

RESULTS AND DISCUSSION

Time-of-Sampling Bias

Table 4 shows the percent bias in average annual total load estimates for Ward (WC-8), Blackwood (BC-1), and General (GC-1) Creeks that would be introduced by eliminating sampling between 18:00 and 09:00. The introduced bias in SS is considerably

TABLE 3. Models for Constituent Total Annual Loads to Account for Hydrologic Variability. Using daytime only samples in calculating total annual loads does not degrade the ability of the models to account for variance in the annual loads. Subscripts j refer to individual gaging stations.

		R^2	
	Model	**All**	**Day Only**
SS	$\log(\text{load}_j) = b_{0j} + b_1 \log(\text{flow}_j) + b_2 \log(\text{peak}_j) + b_{4j} \log(\text{peak}_j)$	0.894	0.897
TP	$\log(\text{load}_j) = b_{0j} + b_1 \log(\text{flow}_j) + b_2 \log(\text{peak}_j) + b_{3j} \log(\text{flow}_j)$	0.921	0.928
TKN	$\log(\text{load}_j) = b_{0j} + b_1 \log(\text{flow}_j) + b_2 \log(\text{peak}_j)$	0.882	0.882
NO_3-N	$\log(\text{load}_j) = b_{0j} + b_1 \log(\text{flow}_j) + b_2 \log(\text{peak}_j) + b_{3j} \log(\text{flow}_j)$	0.861	0.861
SRP	$\log(\text{load}_j) = b_{0j} + b_1 \log(\text{flow}_j) + b_2 \log(\text{peak}_j) + b_{3j} \log(\text{flow}_j)$	0.952	0.954

TABLE 4. Percent Bias in Average Annual Load Estimates that Would Result from Elimination of Nighttime Sampling (18:00-09:00 h) for the Three Streams on the West Side of the Tahoe Basin with Data from Both Day and Night Sampling (1982-2012). Percent bias = 100 × [(loads from day-only samples) − (loads from all samples)]/(loads from all samples).

	Percent Bias		
Constituent	**BC-1**	**GC-1**	**WC-8**
NO_3-N	−6.8	0.7	−19.2
SRP	1.0	2.0	−0.1
SS	−22.1	−21.2	−17.9
TKN	−8.5	−2.7	−4.0
TP	−11.0	−7.1	−5.9

greater than for the other constituents in Blackwood and General Creeks, but is slightly edged out by the high bias for NO_3-N in Ward Creek. The numbers may somewhat overstate the percent bias, however, since as daylight length increases during snowmelt, the USGS sampling crews sample into the early evening, sometimes as late as 21:00. On the other hand, nighttime sampling at these three stations has always been less frequent than daytime sampling, so some negative bias may be present in the load estimates even when all available samples are included.

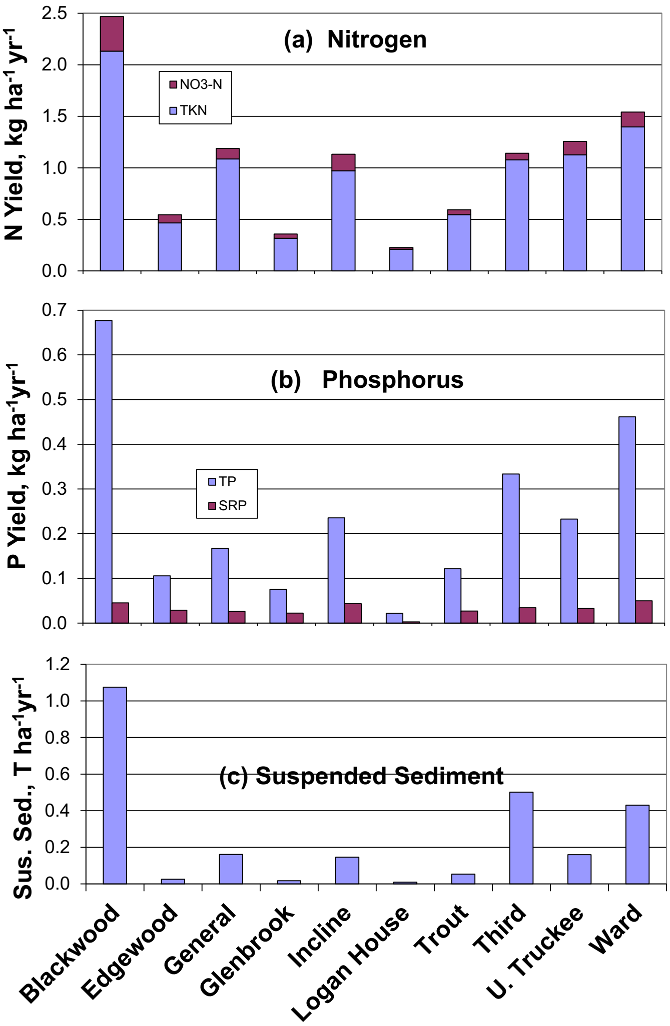

FIGURE 3. Average Annual Yields of Nitrogen, Phosphorus, and Suspended Sediment, 1989-2011, for Primary Lake Tahoe Interagency Monitoring Program Watersheds Stations.

Nitrogen Loads

Figure 3a shows the average annual yields (watershed contributions per unit area) for TKN and NO_3-N by watershed (primary stations), for the period 1989-2011. NH_4-N makes a negligible contribution to N load, and is not shown separately from TKN.

The importance of DIN yields in both absolute terms (kg/ha/yr) and relative to organic N (as TON or TKN) contrasts sharply with other regions of the U.S. In the Tahoe Basin, the total load of DIN (integrated over all LTIMP stations for 1989-2011) accounts for just 12% of the TN load, and DIN yield averages 0.12 kg/ha/yr. In the Northeast, DIN accounts for about 55% of the TN load in forest streams (Binkley *et al.*, 2004), with DIN yields as high as 5.7 kg/ha/yr (Campbell *et al.*, 2004). In the Rocky Mountains, Baron and Campbell (1997) reported NO_3-N yields of 1.7 kg/ha/yr and TN yields of 2.0 kg/ha/yr.

Differences in both atmospheric deposition and vegetation may account for these regional differences. Figure 4 shows the annual wet and dry DIN deposition, 1981-2013, measured near the mouth of Ward Creek. Annual precipitation at the lower-elevation deposition collector is shown for comparison. Wet DIN deposition averaged 1.16 kg/ha/yr and dry deposition averaged 0.77 kg/ha/yr (Hackley *et al.*, 2013; TERC, 2014). Note that the DIN deposition peaked in 1989 and 1990, and then declined. In the northeastern U.S., atmospheric DIN wet deposition ranges from 2.7 to 8.1 kg/ha/yr in some areas, according to data from the National Atmospheric Deposition Program (Campbell *et al.*, 2004). And in the Rocky Mountains, wet deposition of DIN is typically around 4 kg/ha/yr, and has led to conditions of N saturation in alpine and subalpine ecosystems (Williams *et al.*, 1996).

Vegetation, however, may also play a role in controlling the form of N loads in forest streams. In the Tahoe Basin, mountain alder is known to be a significant contributor to NO_3-N loads, especially in small tributaries (Coats *et al.*, 1976; Leonard *et al.*, 1979). Binkley *et al.* (2004) in a survey of nutrient concentrations in forest streams found that in conifer

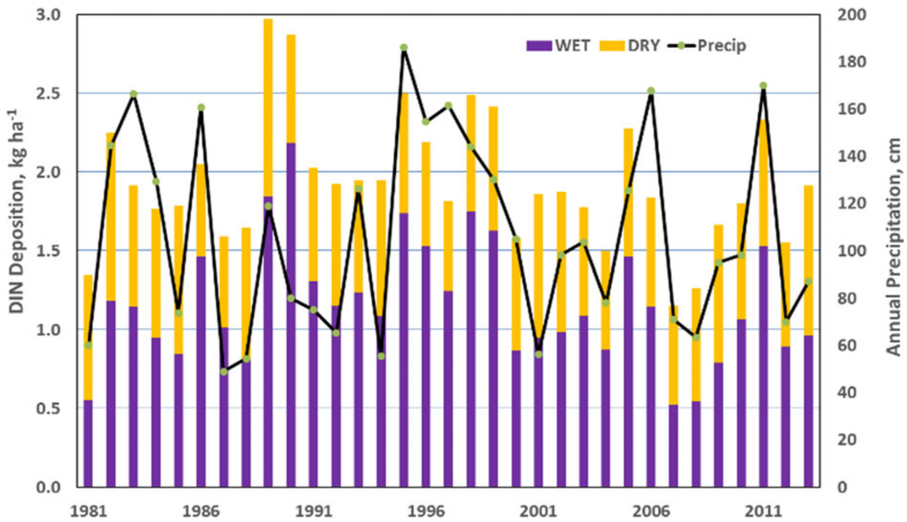

FIGURE 4. Wet and Dry Dissolved Inorganic Nitrogen (DIN) Deposition from a Collector near the Mouth of Ward Creek, 1981-2013. Values for 1981-1988 were estimated by regression with the Ward bench precipitation gage record at the head of Ward Valley, 1989-2006.

forests, DON on average accounted for 80% of dissolved N, whereas in hardwood forests (such as those of the Northeast) NO_3-N accounted for 60% of the dissolved N. Although the effects of atmospheric deposition and vegetation are thus somewhat confounded, the chemistry of litter decomposition in coniferous forest soils may provide a mechanism that reduces the DIN:DON ratio. On low-nutrient sites, the formation of insoluble complexes of proteins with tannin and polyphenols may limit mineralization and nitrification, and help conserve scarce N resources onsite (Rice and Pancholy, 1973; Northrup *et al.*, 1995).

The fraction of TN loads accounted for by DIN loads varies significantly among watersheds of the Tahoe Basin. Table 5 shows the results of the Friedman rank test on the ratio of DIN:TN. There are significant differences in the DIN:TN ratio between watersheds, but the mean annual runoff (1989-2011) does not explain the variance. Considered by water year, however, (averaging over all watersheds) annual runoff has a highly significant effect on DIN: TN ratio; the higher the runoff, the lower the DIN load as a fraction of TN ($R^2 = 0.26$, $p < 0.01$). In wet years, the load of organic N increases more than the load of inorganic N. This is consistent with the hypothesis of a biogeochemical control on DIN production, but could also reflect a greater contribution of organic debris and differences in flow paths and source areas in wet compared to dry years.

Averaging the discharge-weighted annual mean concentrations (Q-wtd means) of DON and PON across the 10 primary stations and the 14 years for which we had TKN measurements in both filtered and unfiltered samples, we found that on average 43% of the TON is dissolved. For individual station-

TABLE 5. Dissolved Inorganic Nitrogen (DIN) Load as Fraction of Total Nitrogen (TN) Load, by Watershed, Averaged over Water Years 1989-2011. Shaded bars indicate groupings of stations that are not significantly different at the 0.05 level according to the Friedman test.

Station	Mean DIN:TN Ratio	Rank
IN-1	0.21	8.39
BC-1	0.27	8.17
ED-9	0.23	7.83
WC-8	0.16	5.70
UT-1	0.14	5.61
GL-1	0.15	4.83
TC-1	0.12	4.39
GC-1	0.14	4.35
LH-1	0.11	3.74
TH-1	0.07	2.00

year Q-wtd means (from instantaneous concentration and discharge), the percent DON ranged from 6 to 91% of TON. There are significant differences between watersheds in the percent of DON. Table 6 shows the results from the Friedman Rank Test on the ratio of DON:TON, and Figure 5 shows the discharge-weighted mean DON and PON concentrations by watershed, averaged over the 14 years. The watersheds with higher DON seem to have lower annual discharge (mean discharge = 26 cm) compared with the watersheds with higher average PON (mean discharge = 62 cm). Grouping the 10 watersheds into two groups on the basis of the DON:TON ratio and comparing the mean discharge of the two groups using a *t*-test supports this hypothesis ($p < 0.04$). The relationship between watershed discharge and the form of organic N seems plausible, as in watersheds with higher discharge, one might expect a shorter

TABLE 6. Results from Friedman Rank Test on DON:TON Ratios of Discharge-Weighted Means Averaged across Water Years by Watershed. Shaded cells indicate groups with ratios that are not significantly different at the 0.05 level according to the Friedman test. The effect of watershed (station) is significant at $p < 10^{-12}$.

Station	Mean DON:TON Ratio	Rank Average
ED-9	0.66	8.07
GC-1	0.65	7.86
GL-1	0.63	7.29
LH-1	0.63	7.07
TC-1	0.61	7.07
UT-1	0.53	5.79
WC-8	0.44	4.14
BC-1	0.37	2.93
IN-1	0.36	2.71
TH-1	0.30	2.07

residence time for organic debris and more energy available for its transport.

Using discharge and chemistry data from USGS Hydrologic Benchmark Network (HBN) stations and loads calculated by the PWS method, Lewis (2002) developed regression relationships relating N yields (kg/ha/yr) to runoff (mm/yr). Stations from regions with high rates of atmospheric wet deposition of DIN (>10 kg/ha/yr) were excluded. Relating yields to runoff guarantees high significance levels and R^2 values due to the problem of spurious self-correlation (discussed above). Nevertheless, we compared average discharge-weighted mean NO_3-N and TN concentrations and yields for our 19 subwatersheds in the Tahoe Basin with values predicted by Lewis's regression equations. The latter predicted average NO_3-N and TN values that were, respectively, 4.3 and 2.4 times our average measured values. The predicted values also overstate the relative importance of NO_3-N: about 19% of TN by the Lewis (2002) equations, and 12% from our data. The HBN stations are sampled monthly, whereas the Tahoe stations are generally sampled at least weekly during snowmelt runoff, so sampling error in the USGS HBN dataset might explain some of the differences. More likely explanations are that Lewis's threshold for atmospheric deposition was set high enough that it includes areas with significant atmospheric DIN deposition (Coats and Goldman, 2001), or that some of the HBN watersheds are dominated by hardwoods rather than conifers (Northrup *et al.*, 1995). As mentioned above, at our precipitation gage near the west shore of Lake Tahoe, wet DIN deposition (1981-2013) averaged only 1.16 kg/ha/yr and wet + dry deposition averaged 1.93 kg/ha/yr (see Figure 4). Other anthropogenic N sources in the "minimally disturbed" HBN watersheds are also possible.

Phosphorus Loads

Figure 3b shows the average annual yields for TP and SRP, 1989-2011, for the primary stations. Note the importance of Blackwood and Ward Creeks, and the dominance of TP compared to SRP. Most of the P in transport is bound with sediment, and the variance of TP loads is much greater than the variance of

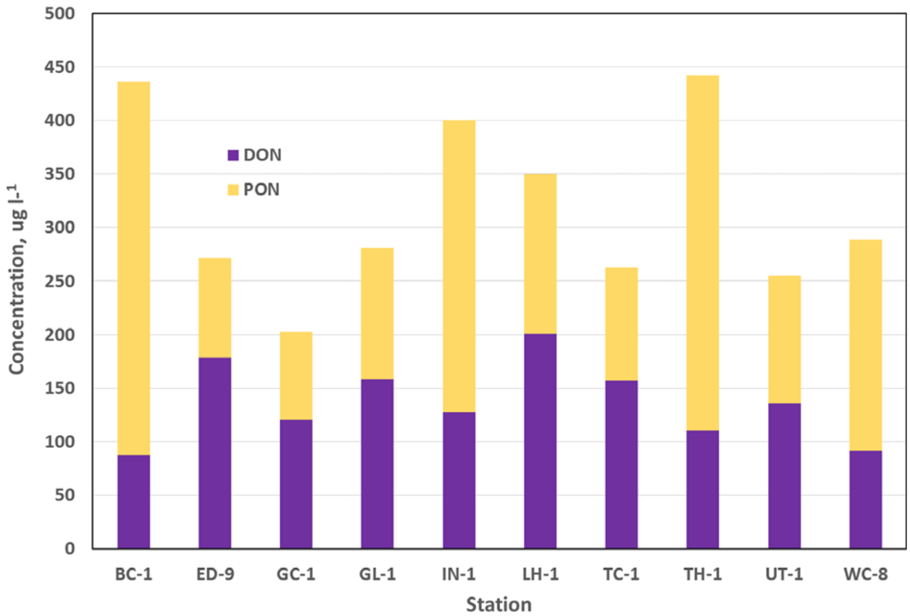

FIGURE 5. Discharge-Weighted Mean Dissolved Organic Nitrogen (DON) and Particulate Organic Nitrogen (PON) Concentration for the 10 Primary Stations, Averaged over 14 Water Years. See Table 1 for station names.

SRP loads. This is because the concentrations of SRP are strongly controlled by biological demand and by equilibrium reactions (adsorption and precipitation) with iron and aluminum sesquioxides in the soil.

The controls on the release of SRP from TP vary by watershed. Figure 6 shows plots for six primary stations of average annual PP concentration in suspended sediment *vs.* average annual discharge-weighted SRP concentration (calculated as annual load divided by annual discharge). Note that the watersheds are somewhat segregated. The axes in this plot are analogous to those in a plot of a Freundlich isotherm, with concentration in adsorbant plotted against concentration of adsorbate in solution (Daniels and Alberty, 1966). The concentration of TP in suspended sediment, however, is typically one to two orders of magnitude higher than that adsorbed by basin soils in laboratory experiments (Susfalk, 2000), as suspended sediment contains P that is liberated from unweathered minerals and organic matter in the acid digestion as well as P that is adsorbed.

Two factors might explain the differences between watersheds in SRP yield (load per unit area) in relation to TP. First, Al-sesquioxides such as allophane and Fe-oxides weathered from andesitic volcanic rock have a strong capacity to fix and retain P. Poorly developed granitic soils have a much lower capacity to fix P, although their fixation capacity increases over time as iron and aluminum-rich minerals weather to oxides and sesquioxides. Susfalk (2000) and Johnson *et al.* (1997) compared the ability of volcanic soils and granitic soils in the Eastern Sierra Nevada to fix and retain P. They found that the extractable-P concentration in granite-derived soils was up to three orders of magnitude greater than that in andesite-derived soils, even though andesitic rocks contain a slightly higher average P concentration than granodiorite—1,150 *vs.* 810 ppm (Porder and Ramachandran, 2012). The soils of Ward Creek (WC-8) and Blackwood Creek (BC-1) are virtually all andesitic, formed either from in-place bedrock or andesitic alluvium/colluvium, whereas the soils of Edgewood Creek (ED-9), Trout Creek, and the UTR are dominated by granitic parent material. Many of the other catchments present a mix of parent materials, making it difficult to find a significant regression relationship between SRP yield and percent volcanic or granitic rock. A further complicating variable is the influence of hydrologic flow paths and weathering rate. Soils along preferential flow paths will weather faster and develop P-fixation capacity faster than better-drained soils (Susfalk, 2000). Interflow through an organic-rich O horizon may mobilize high concentration of both N and P (Miller *et al.*, 2005).

The second factor that may influence the SRP and TP yields of LTIMP watersheds is urban development. In a study of stormwater runoff pollutant concentrations in small developed catchments, Coats

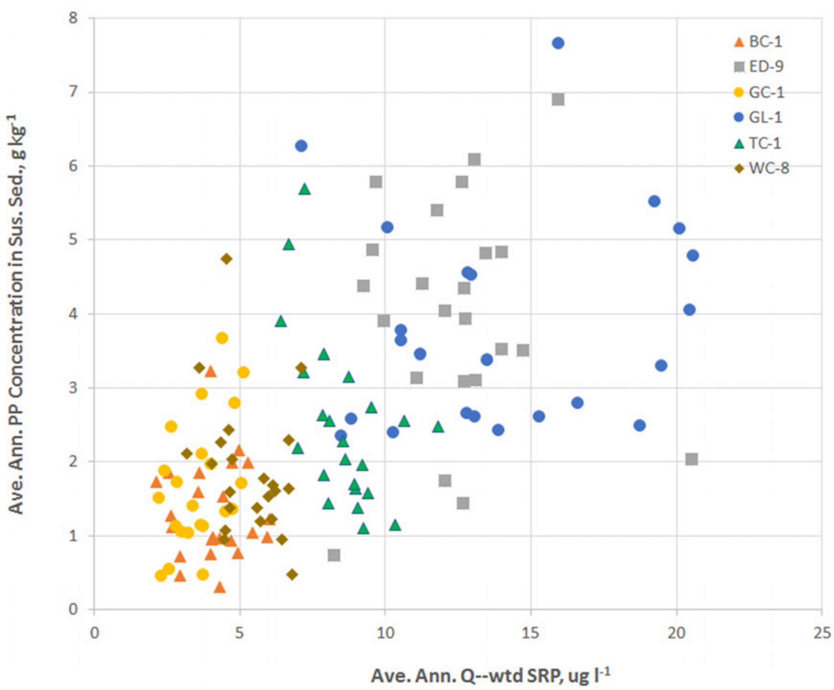

FIGURE 6. Average Annual Total Phosphorus (TP) Concentration in Suspended Sediment *vs.* Average Annual Soluble Reactive Phosphorus (SRP) Concentration in Stream Water, for Six Primary LTIMP Stations. See Table 1 for station names.

et al. (2008) found that the percent of area classed as impervious residential explained 49% of the ln(SRP) discharge-weighted mean concentration. In the LTIMP dataset, three watersheds have percent developed areas greater than 10%. These are Edgewood Creek (ED-9), and the lower subwatersheds of Incline Creek and the UTR.

Suspended Sediment Loads

Figure 3c shows the average annual yields for SS, 1989-2011, for the primary stations. As with N and P, Blackwood Creek is the largest contributor of suspended sediment per unit area. Three factors may account for this. First, the headwall of Blackwood Canyon includes an area of rapidly eroding and poorly vegetated volcanic tuff and breccia commonly called “the badlands” (Stubblefield, 2002; see Figure SI-2). Second, it has the highest per unit area discharge of the LTIMP watersheds; and third, it has a history of overgrazing, heavy logging, gravel extraction, and channel destabilization during the 19th and 20th Centuries (Murphy and Knopp, 2000; Swanson, 2002).

Attribution of Nutrient and Sediment Loads to LULC Variables

Table 7 shows the results of the efforts to attribute nutrient and sediment load residuals to land use and land cover variables. The *p*-values from the stepwise regression are reported, but we consider the values from the permutation test to be more realistic. Definitions of the variables included in the final models are shown in Table 8. TKN loads are positively related to local roads and negatively related to flow path-weighted slope. TP loads are directly related to percent anthropogenic impervious surface area (AIS), and SS loads are directly related to percent AIS but negatively related to “riparian rivers.” The SRP:PP ratio is inversely related to total annual discharge and AIS. Both SRP and TP loads are associated with land development, but PP increases more rapidly than SRP with developed area and total discharge. This is consistent with biogeochemical buffering of SRP concentration (Coats *et al.*, 2008).

The importance of roads and impervious surface area for loads of TKN, TP, and SS is not surprising and corroborates findings of the Tahoe TMDL (Lahontan and NDEP, 2010) and others (Schueler, 1994; Arnold and Gibbons, 1996; Jones *et al.*, 2001; Groffman *et al.*, 2004). Streams and rivers that intersect with mapped riparian vegetation might be expected to produce less suspended sediment than streams lacking in riparian vegetation, as the vegetation may trap suspended sediment and reduce streambank erosion (Stubblefield *et al.*, 2006; 2NDNATURE, 2011). The negative influence of flow-path-weighted slope (see Table 8) on TKN, however, begs for an explanation. This measure of slope weights areas near the lake more heavily. It may be confounded with development, hence misleading because low slopes near the lake tend to be heavily developed.

A larger sample size, clearer delineation of soil parent material type, or higher percent area developed in some watersheds might permit detection of more significant relationships between constituent loads and LULC variables. The use of time as a covariate would have been interesting, but we do not have repeated measurements of LULC variables. Nevertheless, the study of factors controlling N, P, and sediment yields at the small watershed scale appears to be a fruitful line of inquiry.

Time Trends

Because of the possibility that a change in time of sampling could create apparent time trends in total load, we consider here load residuals based only on

TABLE 7. Results from Stepwise Multiple Regression and Permutation Significance Tests of Annual Constituent Load Residuals (after regression with mean annual discharge) *vs*. Land Cover and Geomorphic Variables. The SRP:PP ratio is not represented as a regression residual, but is transformed using the logit function.

Response Variable	Explanatory Variables	R^2	Stepwise Reg., *p*	Permutation Test, *p*
NO_3-N	+Anthropogenic Impervious Surface area, pct (AIS)	0.168	0.081	0.678
TKN	+local roads; −flow-path-wtd. slope	0.61	0.0005	0.041
SRP	+alluvial rivers; +disturbed high-hazard lands	0.45	0.013	0.378
TP	+AIS	0.46	0.0014	0.016
SSC	+AIS; −riparian rivers	0.57	0.0011	0.060
SRP:PP ratio	−AIS; −Log(Total Ann. Q)	0.66	0.0003	0.022
	Disturbed High-Hazard Lands, Log(Total Ann. Q)	0.47	0.009	0.383

Note: SSC, suspended sediment concentration; PP, particulate phosphorus.

TABLE 8. Definitions of Explanatory Variables Included in the Land Use/Land Cover Regression Models.

Explanatory Variable	Definition and Source
AIS area, percent	Anthropogenic Impervious Surface Area from IKONS coverage; Minor and Cablk, 2004
Local roads, km/km	U.S. Forest Service local roads; Luck *et al.*, 2002
Alluvial rivers, km	Intersection of rivers and streams with alluvium in soils map; Rogers, 1974; Luck *et al.*, 2002
Riparian rivers, km	Intersection of rivers and streams with riparian vegetation; Luck *et al.*, 2002
Disturbed high-hazard lands, percent	Percent of watershed in Tahoe Regional Planning Agency Land Capability Classes 1-3 (high hazard) that is developed; Bailey, 1974; Cartier *et al.*, 1994
Flow-path-wtd slope	Average of pixel slopes from 10-m DTM discounted for distance from watershed outlet, by the formula Slope*e^{-kd}, where k = 0.1, and d = flow-path distance (km) to outlet; Luck *et al.*, 2002

daytime samples (see Coats and Lewis, 2014). The significant trends for load residuals based on daytime samples were as follows (see Table SI-1 in Supporting Information for full station names):

1. **SS**: ED-3, IN-1, TH-1, UT-1
2. **TP**: GC-1, IN-2, TH-1, UT-1, WC-8
3. **TKN**: none
4. **NO_3-N**: BC-1, GC-1, IN-1, IN-3, TH-1, UT-1, WC-8
5. **SRP**: GC-1, TC-1, TH-1

All significant trends are downward, with the exception of SRP at TC-1. The significant trend at IN-2 is for a short 14-yr period of record ending in 2001. Most stations have a decreasing pattern of NO_3-N loads, but only the seven listed above are statistically significant at $p \leq 0.0025$. Figures 7 and 8 show the trends for NO_3-N and SS load residuals, respectively. The curves through the data are from a LOESS smoothing (Cleveland and Devlin, 1988).

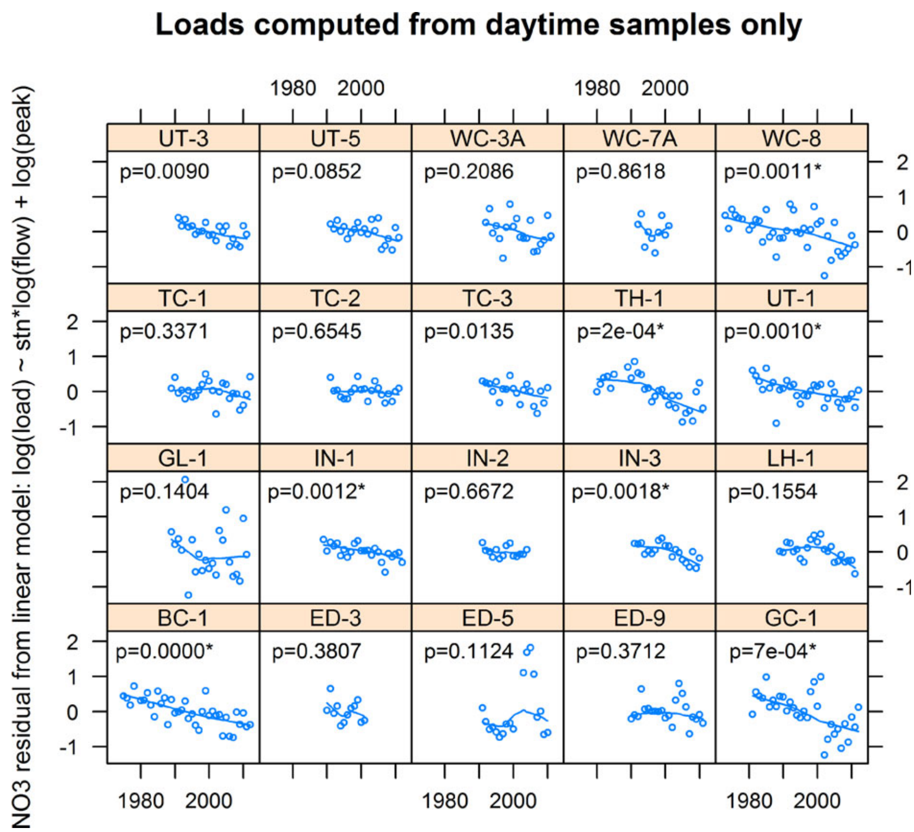

FIGURE 7. Trends in Annual Loads of Nitrate-N after Accounting for Interannual Variation in Total and Maximum Daily Runoff. See Table 1 for station names. Curves are fitted by LOESS (Cleveland and Devlin, 1988). An asterisk by the p-value indicates that the adjusted Mann-Kendall test for linear trend was significant at the 0.0025 level.

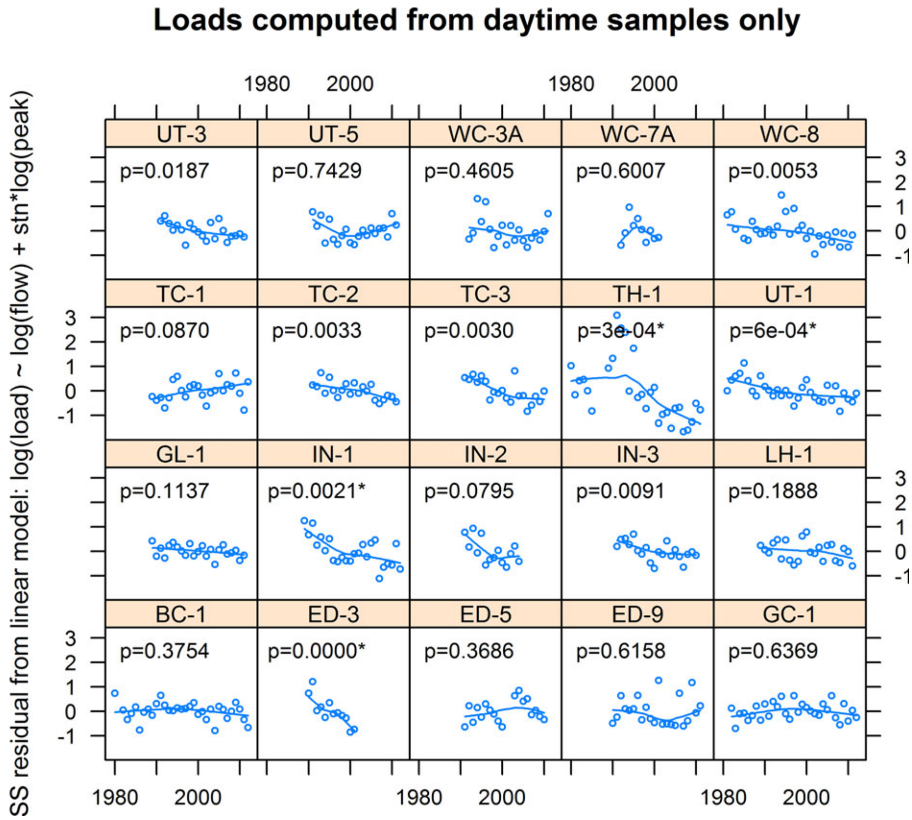

FIGURE 8. Trends in Annual Loads of Suspended Sediment after Accounting for Interannual Variation in Total and Maximum Daily Runoff. See Table 1 for station names. Curves are fitted by LOESS (Cleveland and Devlin, 1988). An asterisk by the p-value indicates that the adjusted Mann-Kendall test for linear trend was significant at the 0.0025 level.

While SRP had significant trends at three locations, one (TC-1) was upward and two (GC-1 and TH-1) were downward, but none were very steep. The pattern for many stations starting in 1985 is similar to that of TP, with declining load residuals reversing around 2003. Plots of load residuals for SRP, TP, and TKN are shown in the Supporting Information. Figure 9 shows the trends for residuals pooled across stations, for each constituent. For NO_3-N, the trend for the pooled residuals is linear, highly significant ($p < 10^{-5}$), and represents an average decrease of 2% per year since the 1970s or a total reduction of about 51%. For SS, the trend in pooled residuals is also highly significant ($p < 10^{-5}$). Because small and large watersheds are equally weighted in these calculations, the true changes in total NO_3-N and sediment discharge to the lake may differ from these values. The very low p-values are a result of the very large sample sizes, which, together with the LOESS curve fits, suggest strongly that downtrends in TP and SRP have reversed. The p-values were only computed for the two constituents (SS and NO_3-N) that exhibited monotonic trends and may be somewhat understated because there is a potential for within-station dependency among the residuals. We do not have an explanation for the apparent oscillations in TKN; a truly periodic pattern would be unexpected.

If runoff is changing as a result of climate change, there are probably corresponding changes in loading that these analyses would not detect. However, using the same methods as for loads, we found no systematic trends in annual maximum daily flow or total annual discharge.

The new load calculation methods were able to identify more trends in total loads than was the simple rating curve method. For NO_3-N, only one significant trend was identified using rating curve estimates, compared to seven significant trends using the PWS method (Table 9). For all constituents but NO_3-N, regression models from which residuals were derived for trend analysis have smaller variance when computed from loads estimated by our new methods rather than from simple rating curves. The smaller variance did not result in identification of more trends, just different ones that should be more reliable.

The occurrence of so many downward trends in loads, especially for NO_3-N, is striking. We hypothesize that the trends in NO_3-N load residuals are caused by long-term recovery from logging and

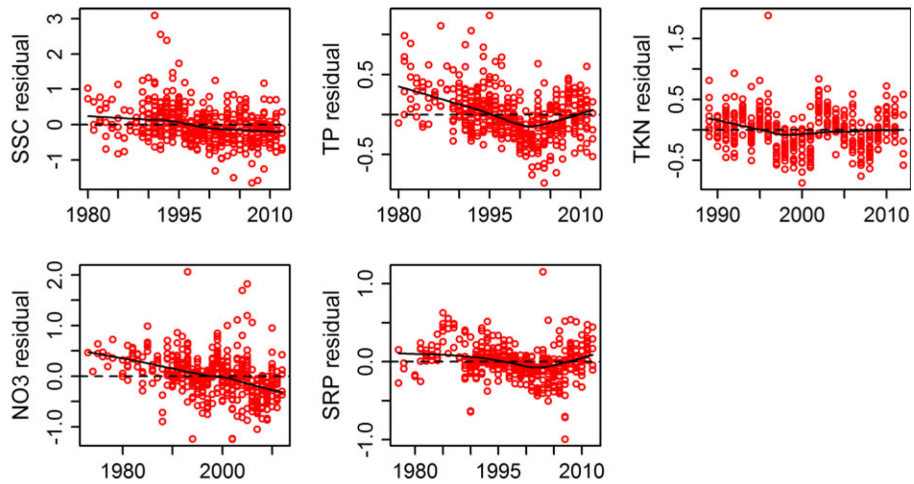

FIGURE 9. Trends in Annual Loads of Suspended Sediment (SS), TP, Total Kjeldahl Nitrogen (TKN), Nitrate-N (NO_3-N), and SRP after Accounting for Interannual Variation in Total and Maximum Daily Runoff. Residuals are pooled from all 20 Lake Tahoe Interagency Monitoring Program stations.

TABLE 9. Comparison of Trend Tests for Annual Loads Estimated by Simple Regression and New Methods.

	Constituent	$\log(c) \sim \log(q)$	New Methods
Standard error of residuals tested for trend	NO_3-N	0.4201	0.4209
	TKN	0.3580	0.3233
	SRP	0.2459	0.2218
	TP	0.3606	0.3089
	SS	0.6451	0.5470
Stations with significant trends ($p < 0.0025$)	NO_3-N	UT-1	UT-1, BC-1, GC-1, IN-1, IN-3, TH-1, WC-8
	TKN	TH-1	None
	SRP	GC-1, TC-1[1], TH-1, IN-3	GC-1, TC-1[1], TH-1
	TP	GC-1, IN-2, TH-1, IN-1	GC-1, IN-2, TH-1, UT-1, WC-8
	SS	IN-1, TH-1	IN-1, TH-1, UT-1, ED-3

[1]Uptrend. All other significant trends are downtrends.

overgrazing in the 19th Century and first half of the 20th Century (Murphy and Knopp, 2000). Essentially, the forests are aggrading (accumulating biomass and nutrients), and becoming more effective in retaining N. In the case of Blackwood Canyon, recovery may involve a shift from N-fixing alders toward conifers, which produce a litter and humus layer with high carbon-nitrogen ratio and low rates of nitrification (Coats *et al.*, 1976). The long-term trend toward warmer temperatures could accelerate plant growth and contribute to closing of nutrient cycles.

In an effort to explain the downward trends in NO_3-N, we tested multiple regression models with the previous year's basin-wide runoff, average summer temperature, Palmer Drought Severity Index (MJJA, at Tahoe City), and current year's DIN deposition. None of the models (including interaction terms) was useful in explaining the trends.

The forest aggradation hypothesis, however, is supported by data from the United States Forest Service's Forest Inventory and Analysis program which has measured and remeasured live biomass and live basal area on forested plots in the Tahoe Basin between 2001 and 2013 (USDA Forest Service, 2015). The rate of change in live biomass on 14 of these plots averaged +1.1 tons/ac/yr. The average slope (biomass rate of change) is positive (by a one-tailed t-test) at $p < 0.02$. The average slope of basal area (on 22 plots) was 2.35 ft^2/ac/yr, positive at $p < 0.006$. It seems reasonable to assume that the rates of biomass and basal area increase from the mid-1970s (when sampling began in Ward and Blackwood Creeks) to the year 2000 are at least this great.

The downward trends in SS and TP have a different possible explanation. Simon *et al.* (2003) showed that the flood of January 1997 scoured fine sediment from streams in the Tahoe Basin, and shifted sediment rating curves downward for some streams, with a lower SS concentration for a given discharge. Except for GC-1, all of the stations for which we found downward trends in SS and/or TP were identified by Simon *et al.* as showing highly significant ($p < 0.0001$) downward shifts in their sediment rating curves. Simon found no such downward shift in the rating curve for Blackwood Creek, and for that station we found no downward trend in SS or TP load residuals. We do not see a downward step-shift in the SS and TP residuals, but it is possible that the impacts of the 1997 flood continued to increase for some years following the flood.

SUMMARY AND CONCLUSIONS

In this study, we have developed and compared different methods of calculating total constituent loads. Using the method that maximizes precision and minimizes bias, we recalculated the total annual loads of NO_3-N, NH_4-N, TKN, SRP, TP, and SS for all of the LTIMP stations over the periods of record. Significant differences in yield were found among watersheds for different forms of N and P. Differences were also found for the ratios of DIN:TN, and DON:PON and SRP:TP. Efforts to attribute differences in constituent loads to Land Use-Land Cover variables were moderately successful, and emphasized the importance of AIS as an explanatory variable.

To examine long-term trends in constituent loads, we related the annual loads to total annual discharge and annual maximum daily discharge, and analyzed time trends in the residuals. The significant downward trends in NO_3-N residuals indicate a long-term improvement (since the mid-1970s) in water quality, which we suggest may be due to long-term recovery of terrestrial ecosystems from 19th and 20th Century disturbance.

Although the work reported here represents a step forward in calculating and attributing total constituent loads in Tahoe Basin streams, considerable room for improvement remains. The composite approach of Aulenbach (2013) may offer a useful tool for further modifying and testing load calculation methods for the Tahoe Basin. Continuous turbidity monitoring combined with additional sampling of fine sediment will permit improved estimates of fine sediment particle numbers, which are a key variable in the clarity of Lake Tahoe. The use of automated sampling equipment, which has proven essential in studies of urban runoff in the basin, could provide a solution to the vexing problem of time-of-sampling bias. The ultimate value of water quality monitoring in supporting the protection of Lake Tahoe, however, will depend not just on technology and statistics, but also on the continued dedication and involvement of the resource agency staffs and political support from the public.

SUPPORTING INFORMATION

Additional supporting information may be found online under the Supporting Information tab for this article: File containing: 1. Detailed background on the Lake Tahoe Basin; 2. Methods in detail; 3. Results in detail.

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