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Land use and land cover influence on water quality in the last free-flowing river draining the western Sierra Nevada, California

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Abstract

Land use and land cover across 28 sub-basins within the Cosumnes Watershed, CA (1989 km^2) were correlated to nitrate-N and total suspended solids (TSS) loading between water years 1999 and 2001. The impact of human development on stream water quality was evident as both agricultural area and population density predicted TSS loading in a linear mixed effects model. In contrast to the TSS model, the nitrate-N loading model was more complex with agriculture, grassland, and the presence or absence of waste water treatment plants (WWTPs) all contributing. The lack of correlation between population density and nitrate-N loading indicates that human habitation of the landscape does not impact stream nitrate levels until a WWTP is built within the sub-basin. During dry water years the models predict a linear reduction in TSS loading but the correlations to agriculture and population density remain positive. In contrast, nitrate is positively correlated to grasslands during average water years and negatively correlated during dry water years. Analysis of constituent fluxes from the upper watershed versus the lower watershed indicates that silica is derived primarily from the uplands and that during dry water years the upper watershed is an important source of dissolved organic carbon (DOC) and nitrate. The lower watershed contributes the majority of the sediment and nutrients during both dry and average water years, the one caveat being that during dry years the lower basin becomes a nitrate sink.

Keywords: Cosumnes River; Non-point source pollution; Landscape water quality relations; Land use; Impoundment; Water quality modeling

1. Introduction

River water chemistry is controlled by numerous natural and anthropogenic factors. These controls on stream chemistry can either be spatially diffuse (e.g. interflow through organic rich soils, return flow from extensive row crop cultivation) or concentrated

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(e.g. a hydrothermal spring, a waste water treatment plant [WWTP] outfall). Calculating these latter inputs is relatively simple as direct measurements can be made at the source, but attributing stream water chemistry to non-point sources (NPS) is much more difficult (Baker, 2003), as complications of scale are inevitable. To address these problems researchers have taken a landscape approach, subdividing watersheds into various combinations of land use/land cover (LULC) and monitoring the outflowing waters. Some results have been qualitative in nature, simply stating, for example, that agriculturally dominated areas produce elevated concentrations of dissolved salts (Smart et al., 1998) and nutrients (Keeney and Deluca, 1993; Pekarova and Pekar, 1996; Turner and Rabalais, 2003), while other studies have used GIS data and regression analysis or ordination to examine relationships between LULC and suspended sediment (Allan et al., 1997; Bolstad and Swank, 1997; Hill, 1981; Johnson et al., 1997) and nutrients (Allan et al., 1997; Arheimer et al., 1996; Basnyat et al., 1999; Hill, 1981; Omernik et al., 1981; Osborne and Wiley, 1988; Sliva and Williams, 2001).

The majority of these studies conclude that agricultural land use strongly influences stream water nitrogen (Arheimer and Liden, 2000; Johnson et al., 1997; Smart et al., 1998), phosphorus (Arheimer and Liden, 2000; Hill, 1981), and sediments (Allan et al., 1997; Johnson et al., 1997). Conversely, Osbourne and Wiley (1988) found that urban land use determined soluble reactive phosphorus in an east Illinois watershed while agriculture was only a secondary predictor of both N and P. Basnyat et al. (1999) and Sliva and Williams (2001) came to similar conclusions in their studies of urban land-use impacts in Alabama and Ontario. However, using urban cover as a NPS for nutrients can give spurious results, as much of the cover in urban areas is impervious and drainage is frequently routed to WWTPs (which may or may not be in the same basin), then discharged to local rivers as point sources. To avoid this complexity Hill (1981) excluded watersheds with WWTPs from his study of 22 watersheds in Ontario and as a result found no correlation between urban coverage and stream water phosphorus. Obviously there are difficulties associated with quantifying urban impacts on water quality using GIS based techniques.

The issue of seasonality also gives rise to variation in results in LULC-water quality analyses. Many authors have argued that the analysis should be conducted during the storm season, as it is during this time that the landscape is most intimately connected with local waterways (Arheimer and Liden, 2000; Basnyat et al., 1999; Bolstad and Swank, 1997). Others have addressed this issue by analyzing stream chemistry during multiple seasons and conducting separate correlations with LULC for each season (Johnson et al., 1997; Osborne and Wiley, 1988). The results of course indicate that, dependent upon season, LULC has a varying impact on water quality. Osborne and Wiley (1988) noted that median nitrate concentrations were correlated with agricultural practices during the high-flow spring period, but then became correlated with urban land cover during the low-flow summer and autumn. Though seasonality has been addressed, few if any studies have compared dry and wet year stream chemistry with LULC gradients. The need for inter-annual analysis is especially acute in California as variability in yearly precipitation can be extreme, and the impact on water quality substantial.

Though there have been many studies linking LULC to river water quality, we know of none that have been conducted in Californian watersheds, where a distinct Mediterranean climate makes water quality dynamics unique among other locales in the United States. This study examines the impact of LULC on sediment and nitrate loading in both dry and average water years (a water year runs from October to September), taking into account the presence of the sub-watersheds that receive effluent from WWTPs, in order to quantify which land-use types drive chemical loading in the waterways of the Cosumnes River Watershed. Additionally, chemical and sediment flux analysis of the upper and lower watersheds was used to determine if and when the upper watershed is an important source of potential pollutants. In the western Sierra Nevada, a watershed study of this type and scale is only possible in the Cosumnes Watershed as every other drainage contains major impoundments which create discontinuity (Stanford and Ward, 2001; Ward and Stanford, 1983) in the longitudinal land–water connection that is essential for such analyses to be valid.

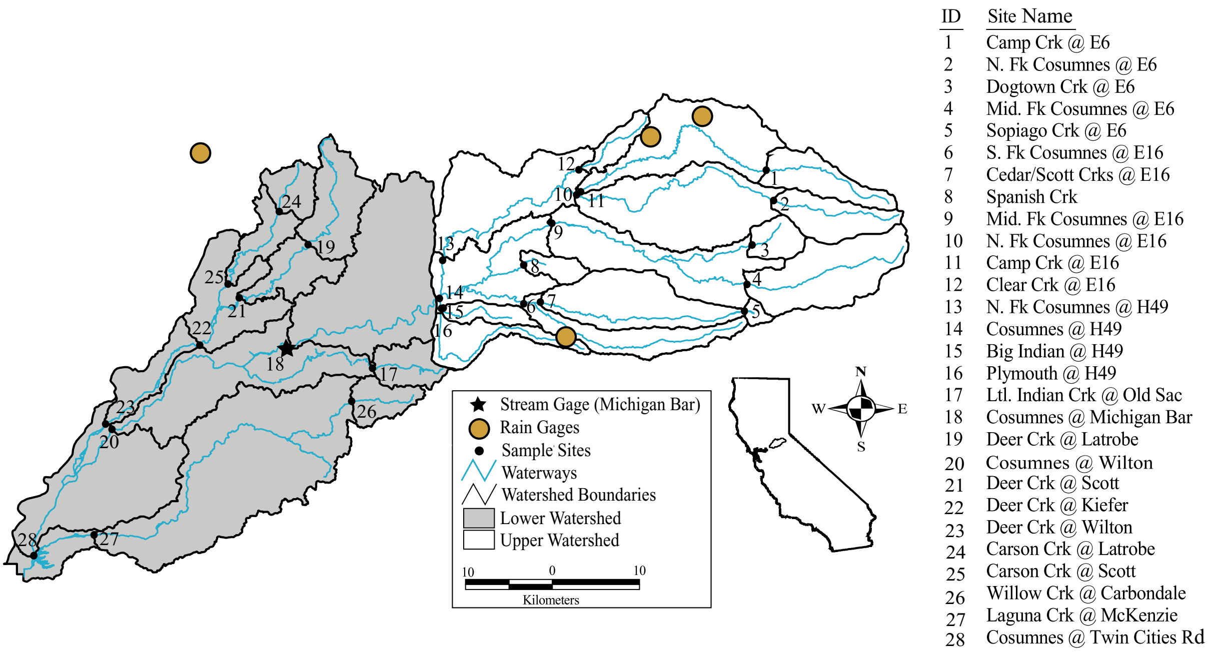

Fig. 1. Cosumnes River Watershed with names and locations of the 28 sampling locations, four rain gages, and the one river gage used in this study.

2. The study area

The Cosumnes River Watershed is located southeast of Sacramento, CA and has a drainage area of 1989 km^2 (Fig. 1). The watershed begins south of Lake Tahoe at 2400 m and descends westward into the Central Valley, joining with the Mokelumne River at 2 m above sea level. Because the Cosumnes Watershed drains the western flank of the Sierra Nevada the waterways pass through three distinct geologic zones. The upper watershed is underlain by granodiorites of the Sierra Nevada batholith, the middle reaches pass through a metamorphic belt, and the lower reaches meander through valley fill sedimentary units. The human population is sparse in the uplands and some logging of the coniferous forest is the only significant land use. The middle reaches of the Cosumnes River wind their way through oak woodland habitat, where grazing and viticulture are the dominant land uses. Annual grasslands dominate the lower Cosumnes Watershed as the river descends to its confluence with the Mokelumne River and the important aquatic habitat of the Bay-Delta ecosystem. Land use in the lower reaches is dominated by production agriculture (e.g. row crops and viticulture) (Fig. 2). Population distribution is weighted toward the lower watershed

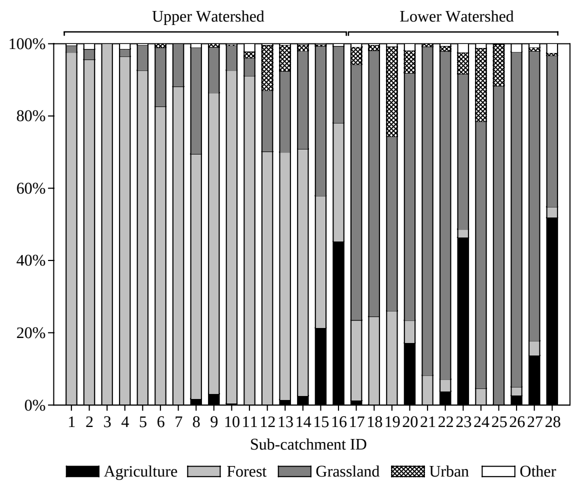

Fig. 2. Landscape characteristic gradients for each of the 28 sub-basins used in the study. Note the predominance of grasslands in the lower basin and coniferous forest in the upper basin.

where suburban encroachment from the city of Sacramento is spurring rapid growth, while gold rush mountain towns account for some elevated population numbers in the middle reaches.

In the Mediterranean climate of central California there is a strong seasonal cycle with virtually all of the annual precipitation occurring between December and March. Average precipitation in the upper watershed is 804 mm yr^{-1} while approximately 445 mm yr^{-1} falls in the lowlands. The Cosumnes River, as gaged at Michigan Bar (Fig. 1), has a long-term (1907–2002) mean daily discharge of 14.4 $m^3 s^{-1}$ (USGS gage #11335000).

Water sampling stations were located at 28 sites throughout the Cosumnes River Watershed (Fig. 1). The sites were selected across a range of land uses, geology types, and stream orders. The two sites with the most extensive urban development (sites 19 and 24) have 25 and 20% urban coverage respectively; each of the two sub-watershed contains a WWTP which services areas that extend beyond the confines of the drainages.

3. Methods

3.1. Water collection and analysis

Grab samples were collected from 28 sites every 2 weeks from October 1998 to September 2001. Total suspended solids (TSS) were measured from a 500 ml sample collected from the thalweg of the river and at approximately the mid-depth of the water column. The 500 ml subsample was filtered through a pre-weighed glass fiber filter; the filter was dried at 60 °C for 24 h and weighed again, the difference being the mass of sediment in the water sample. A separate 125 ml sample was filtered through a 0.2 μm polycarbonate membrane (Nuclepore) and stored at 4 °C through completion of analyses. Ion chromatography (Dionex 500x; CS12 cations; AS4A anions) was used to measure nitrate-N and phosphate-P with a detection limit of 6.0 ppb. A Dohrmann UV-enhanced persulfate TOC analyzer (Phoenix 8000) was used in the analysis of dissolved organic carbon (DOC) with a detection limit of 50 ppb. Total phosphorous (TP) was analyzed from a persulfate-digested split of unfiltered sample (Yu et al., 1994), the digested sample was measured with the ammonium molybdate method (Clesceri et al., 1998) using a Hitachi U-2000 spectrophotometer with a detection limit of 5.0 ppb. Total nitrogen (TN) was measured on a persulfate-digested split of unfiltered sample on a Carlson autoanalyzer (Carlson, 1978, 1986) with a detection limit of 50 ppb.

3.2. GIS analysis

ArcGIS 8.3 Desktop GIS software was used to determine the relative composition of land use and population characteristics within the Cosumnes River sub-watersheds. All datasets were converted to a common digital format, using a common coordinate system (Albers Equal Area). Multi-source Land Cover Data (California Department of Forestry and Fire Protection, 2002b), also referred to as FVEG (Forestry-Vegetation management concentration), were reclassified into Agriculture, Forest, Grassland (Oak woodland habitat was classified with grassland), Urban, and Other. The FVEG was then converted from raster to a vector dataset and intersected with previously-derived watershed boundaries. GIS tools were used to calculate the area of each land cover type within each sub-watershed, which was subsequently divided by the watershed area to derive the percentage of the watershed covered by each type (Fig. 2). Year 2000 Census Blocks (California Department of Forestry and Fire Protection, 2002a) were intersected with the same watershed boundaries. GIS tools were used to calculate the total population for each watershed, which was subsequently divided by the watershed area to get the population density (people/km^2). A geologic map (Wagner et al., 1981) was digitized for The Nature Conservancy and provided to the Cosumnes Research Group as a GIS dataset. The map was simplified into three categories—igneous, metamorphic, and sedimentary—and percent coverage of each was calculated for the sub-basins.

3.3. Hydrologic modeling

Two models were used in the estimation of flow from each of the 28 sub-watersheds. A Precipitation Runoff Modeling System (PRMS) with input from three rain gages in and around the upper basin (Fig. 1) was used to estimate flows in the upper watershed, while HEC-HMS, an updated version of the United States Army Corps of Engineers (USACE) rainfall–runoff model, was used in the lower basin. HEC-HMS used input from the one rain gage adjacent to the lower basin. Two separate models were used because the PRMS model for the upper basin existed prior to the study, applying the PRMS model to the lower basin would have been inefficient due to the paucity of gages in the lower basin. HEC-HMS was more suited for the conditions in the lower basin. Both models were calibrated to the Michigan Bar gaging station located in the middle of the watershed (Fig. 1). After calibration the PRMS model had a relative standard error of 11.2% while the HEC-HMS model had an error of 17%.

The PRMS model, was developed by the United States Geological Survey (Leavesley et al., 1983). It is a conceptual, distributed parameter model capable of continuous simulations. There are over 50 modules used to describe hydrologic function which can be used in any combination in the construction of a model for a particular region.

HEC-HMS is a conceptual, lumped parameter model that utilizes a graphical user interface to build a watershed model and to set up the precipitation and control variables for simulation (USACE, 2000). The watershed model contains seven different elements describing sub-watersheds, channel segments, diversions, reservoirs, sources, and sinks. The sub-watershed element has routines to compute losses due to infiltration, transforming the rainfall to runoff, and baseflow.

Estimated daily discharges for each sub-basin were multiplied by linearly interpolated constituent concentrations of TSS and nitrate (see Fig. 3 for the data range and median). The resultant flux numbers were summed by year, in-flowing fluxes were subtracted, and finally divided by sub-basin area (Fig. 4).

3.4. Statistical analysis

The data structure is a longitudinal, cross-sectional survey of TSS and nitrate-N annual fluxes repeated yearly for 28 sub-basins over 3 water years. In order to account for co-dependence introduced by repeated measurements on the 28 experimental units (sub-basins) we used linear mixed effects (LME) analysis to examine the relationship between LULC factors and TSS and nitrate-N flux in dry and average water years (Pinheiro and Bates, 2000). This approach allows for robust, simultaneous evaluation of associations between response variables and environmental gradients while accounting for the repeated measures embedded in the data structure (Atwill et al., 2002; Bedard-Haughn et al., 2004; Tate et al., 2003). Separate statistical models were developed for TSS and nitrate. A backwards stepping approach was employed to isolate a final model with only significant ($p<0.05$ for TSS, $p<0.10$ for nitrate) independent variables included. For each model, the initial fixed independent variables were LULC variables (%forest, %agriculture, %grassland, %urban, population density, %sedimentary, %igneous, %metamorphic, elevation, and presence or absence of WWTP), annual rainfall (average or dry), and interaction terms for LULC variables and rainfall. Sub-basin identity was treated as a grouping variable to account for repeated measures. Insignificant main effects remained in the model if interaction terms containing the main effect were significant. Both dependent and independent data were natural log transformed to meet the assumptions of normality, as determined via graphical evaluation of standard diagnostic graphs. Goodness-of-fit of final significant statistical models was evaluated by scatter plot and simple linear regression of observed data against equivalent model prediction.

In addition to a LME model, simple linear regression was also conducted. The water quality and geographical data were not normally distributed across the sub-watersheds so a non-parametric test (Kendall's Tau) was used to determine significance of simple linear regressions between environmental variables and response variables.

4. Results

4.1. Linking land use and river water quality

Multiple linear regression was used to quantify the magnitude, direction, and significance of relations between individual environmental variables and loading of nitrate-N and TSS. During the average precipitation years of 1999 and 2000 the environmental variables most strongly related to nitrate-N

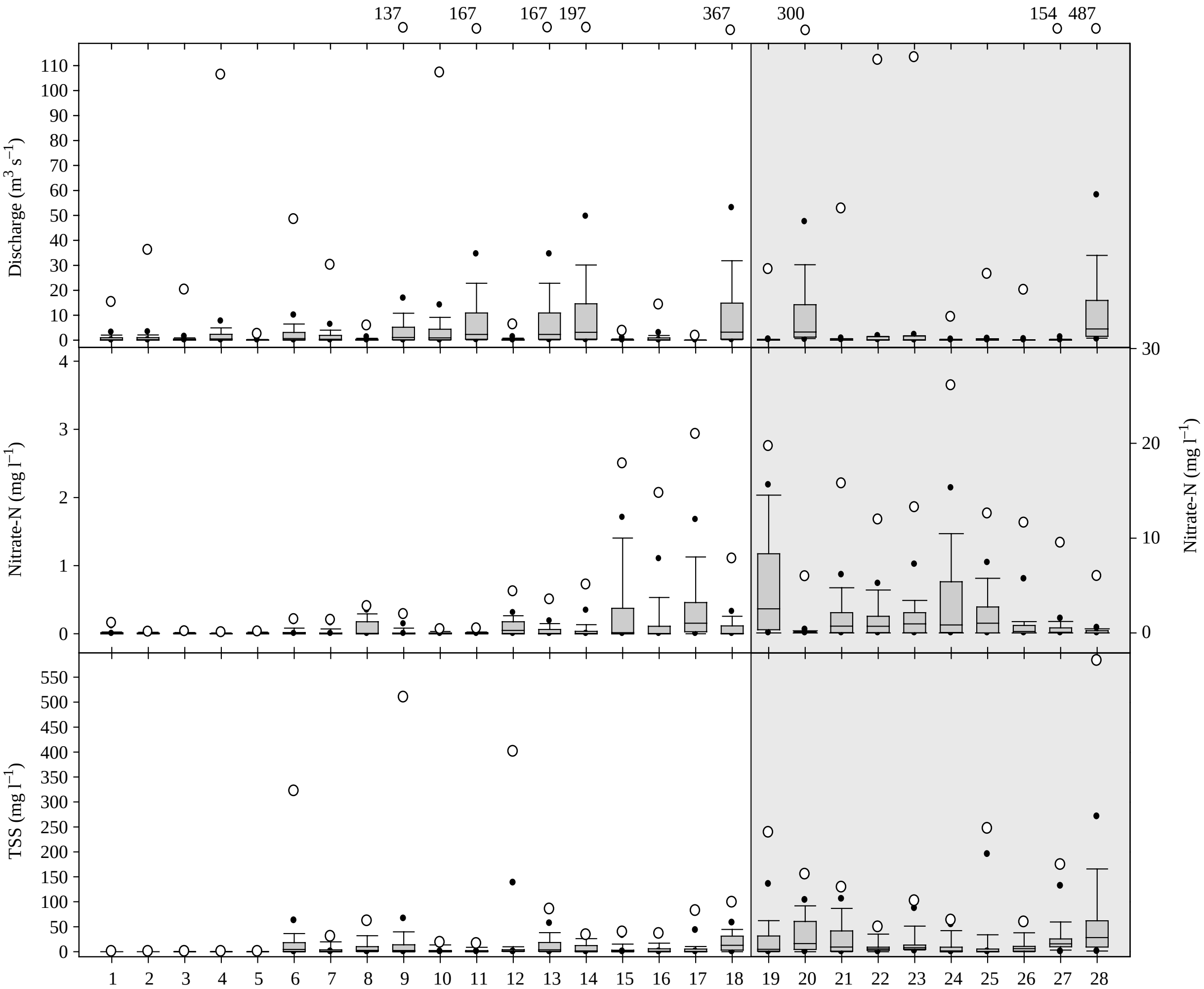

Fig. 3. Boxplots representing nitrate-N, TSS, and discharge data distributions (1999–2001) from the 28 sub-basins. The open circle represents the maximum value obtained, the closed circle the upper 95% and lower 5%, whiskers are the upper 90% and lower 10%, boxes are the upper 75% and lower 25%, median line is displayed in each box. Lower watershed sites are in the shaded region. Note the change of scale for nitrate between the upper and lower watershed.

loading were population density, %grassland, elevation, and %forest (Table 1). For TSS the most important environmental drivers included %urban, %agriculture, and elevation. During 2001, nitrate was only correlated to population density while the relationship between TSS and %agriculture weakened.

In order to formulate a more rigorous predictive model for chemical loading from our environmental data, a linear mixed effects model was constructed for water years 1999–2001 (the data from water year 1999 is not displayed because the results were very similar to those in 2000) (Fig. 5). Geologic variables were not significant in any of the model runs so we excluded these from our analyses and focused on LULC. TSS loading was predicted by a linear combination of %agriculture, %agriculture × rainfall, population density, and rainfall (Fig. 5a). Population density was the driving factor with a coefficient of 0.588 while %agriculture played a slightly less important role (coefficient = 0.516). The rainfall term indicates that TSS loading is greatly reduced

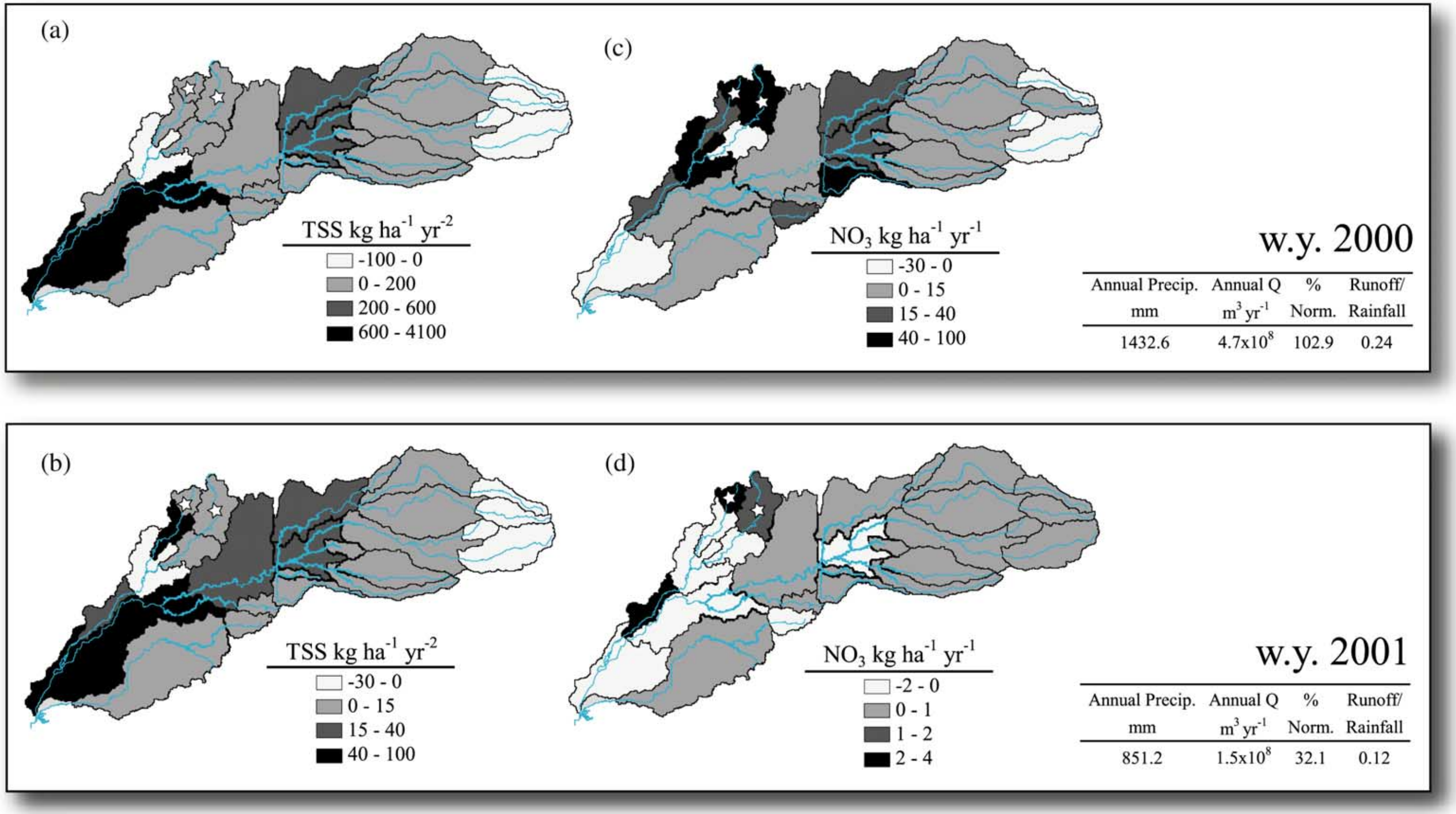

Fig. 4. TSS and nitrate-N loading distribution across the watershed in an average (2000) and dry (2001) water year. Stars indicate the presence of WWTPs. Note differences in legend scales between years. The loading distributions in water year 1999 were very similar to those found in 2000, for simplicity only the results from 2000 to 2001 are reported.

Table 1
Linear regression results using Kendall's Tau *b* computed on log transformed data

	1999 and 2000		2001	
	NO_3^-	TSS	NO_3^-	TSS
Agriculture	0.34*	0.32*	–	0.31
Forest	−0.49*	−0.24	–	−0.50*
Grassland	0.45*	0.24	–	0.45*
Urban	0.35*	0.30*	–	0.50*
Pop. density	0.43*	0.24	0.31	0.37

The * indicates Prob > Tau b < 0.01, otherwise < 0.05, – indicates an insignificant relation. A Kendall's Tau b of 1.0 is a perfect fit.

during dry years, while the %agriculture × rainfall interaction indicates that this is especially true in agricultural lands. In order to understand the results of a LME model it is frequently helpful to graph the results with hypothetical dependant variables. When hypothetical population densities and %agricultural coverage are input into this model the results give a clear picture of how TSS varies with land use (Fig. 6). As population increases TSS loading is predicted to increase, especially in a wet year, while an increase in %agriculture is predicted to have a lesser effect on the resultant TSS loading.

The LULC-nitrate model is more complex and the fit not as strong as that obtained for the TSS model (Fig. 5). Nitrate was explained through a combination of %agriculture, %grassland, %grassland × rainfall, and the presence or absence of WWTPs (Fig. 5b). Because the two sub-basins that included WWTPs (basins 19 and 24) exhibited such high nitrate-N loading (Fig. 4) the WWTP term was a strong predictor of nitrate-N loading, but outside of these two highly urbanized basins, %urban or population density did not predict nitrate-N loading. Instead, a combination of grassland and agricultural land drove nitrate-N loading in both average and dry years.

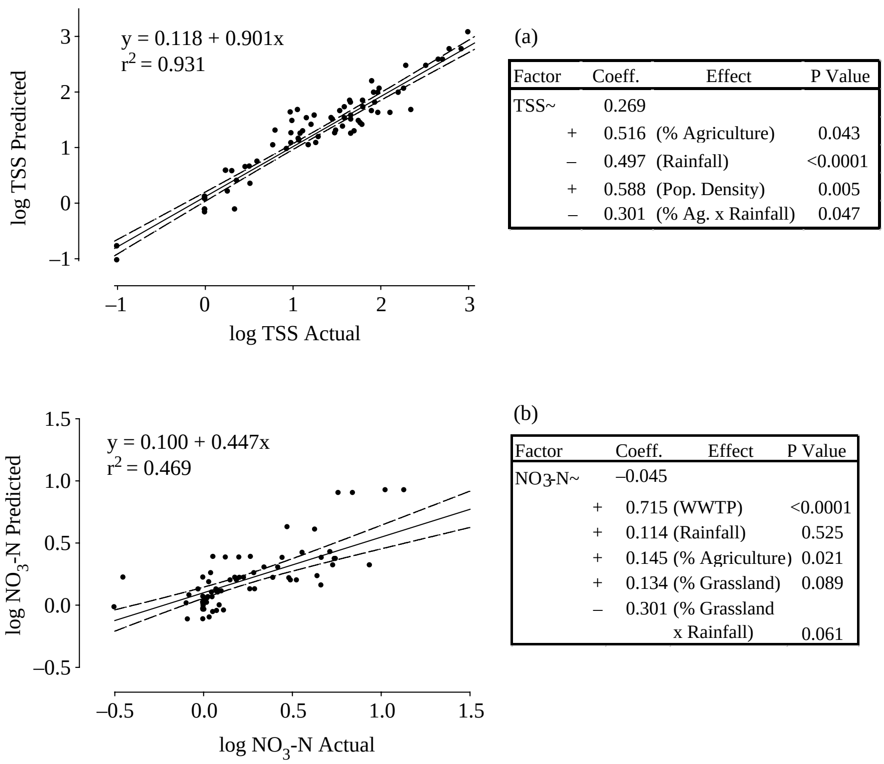

Fig. 5. Results of LME model analysis for (a) TSS and (b) nitrate-N. Model fit is graphically depicted with 95% confidence limits, while the linear equation is presented in the accompanying tables. All units are kg ha^{-1} yr^{-1}.

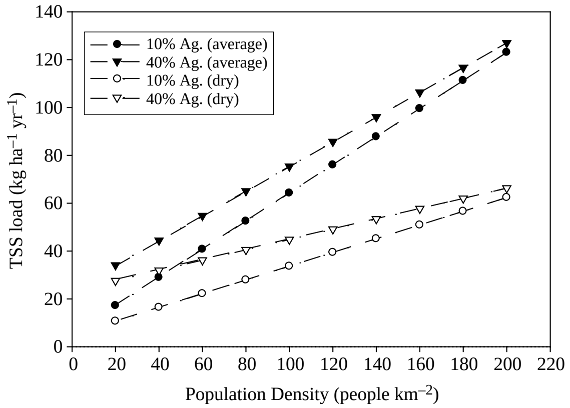

Fig. 6. TSS linear model with hypothetical %agriculture and population density numbers input to depict output results for dry and average water years.

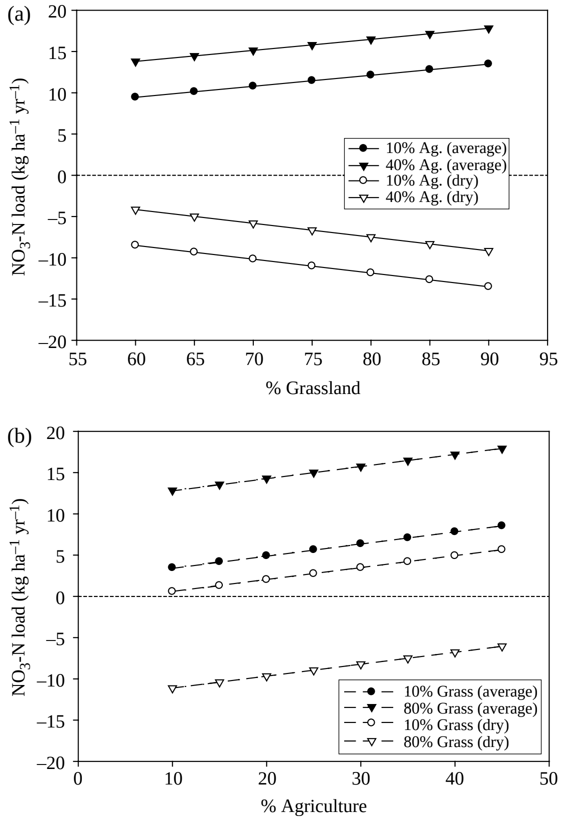

Fig. 7. Nitrate-N model for dry and average water years with hypothetical %grassland and %agriculture depicted (a) without WWTPs and a %grassland *x*-axis and (b) with WWTPs and a %agriculture *x*-axis.

The correlation between %agriculture and nitrate-N loading was clear as loading increased with increasing agricultural coverage in both average and wet years (Fig. 7). Grasslands however had a more complex relationship with nitrate-N loading. During average years grassland was a positive predictor of stream nitrate-N loading, but during dry years grassland was negatively correlated to nitrate-N loading (Fig. 7). While nitrate-N loading changed markedly between wet and dry years, the relative contribution of each sub-watershed to TSS loading remained relatively constant inter-annually (Fig. 4).

4.2. Uplands versus lowlands

Though chemistry varied across each of the 28 sub-watersheds, upland drainages tended to deliver dilute, clear waters to the lowlands, while lower elevation sub-watersheds produced more turbid waters with elevated levels of constituents (Fig. 3). In the upper watershed median TN for the 16 sub-watersheds was 0.15 mg l^{-1} while median EC was 62.9 μS cm^{-1} (data not shown). This contrasted sharply with the lower watershed where median concentrations of TN and EC over the three years of this study were 0.83 mg l^{-1} and 192.5 μS cm^{-1}, respectively. Median TSS concentrations of 1.6 mg l^{-1} in the upper watershed and 8.7 mg l^{-1} in the lower watershed indicated that the lower watershed is the primary source of sediment in the Cosumnes. TP and nitrate concentrations were elevated in the lowlands especially in the Deer Creek sub-watershed (containing WWTPs), while Si and DOC concentrations were more evenly distributed throughout the watershed (data not shown).

During water year 2000 (an average precipitation year) the upper watershed produced a water flux of 4.45×10^{11} L yr^{-1} (Fig. 8). Though the lower watershed delivered half as much water (2.14×10^{11} L yr^{-1}) in this year, chemical concentrations in lowland areas (Fig. 3) were high enough to cause the lower watershed to be the primary source of solutes and sediment in the basin. While the upper basin proved to be an important source of Si, far more TSS was delivered by the lower basin; in water year 2000 (an average water year) the lower basin delivered

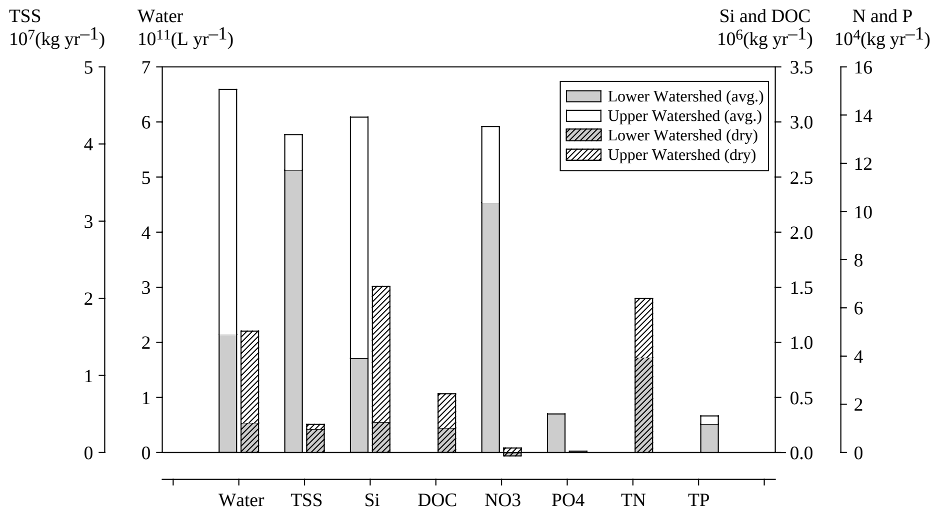

Fig. 8. Fluxes of nine constituents and water from the upper and lower Cosumnes Watershed in water year 2000 (average) and water year 2001 (dry). Note the variable scales on the four *y*-axes.

800% more TSS than the upper basin. Nitrate-N and phosphate-P fluxes were also disparate with approximately 330% more nitrate-N being generated in the lower watershed and no measurable phosphate being generated by the upper watershed.

In water year 2001 a similar flux analysis was conducted, except in this dry year we included DOC, TP, and TN data (Fig. 8). Because water year 2001 was a dry year there was not much precipitation in the lowlands (410 mm); the lack of regional storms meant that orographic precipitation in the uplands became the primary source for water in the basin. This is reflected in the water budget for 2001; while the upper watershed produced $1.66\times10^{11}\,\mathrm{L\,yr^{-1}}$, the lower watershed produced only $5.39\times10^{10}\,\mathrm{L\,yr^{-1}}$. The relative contributions from the two sub-basins changed in 2001, as the upper watershed weighed more heavily in the total flux budget. The nitrate response changed considerably with the lower watershed acting as a nitrate sink while the upper watershed continued to export nitrate (though in small quantities, 1873 kg in 2001). While the lower watershed was a nitrate sink, it was acting as a source of other nutrients; the lower watershed produced 528% more TP and 299% more TN than the upper watershed during the dry year of 2001. In 2001, the upper watershed produced slightly more DOC than the lower watershed. Because there were no DOC data collected during an average water year it is difficult to say whether this is a dry year phenomenon or not. Lastly, the lower watershed—though exporting an order of magnitude less sediment than in the previous water year—remained the primary source for TSS in 2001.

5. Discussion

Analysis of water, chemical, and sediment fluxes between the upper and lower watersheds revealed the relative importance of the upper watershed in the delivery of select solutes during dry years. Silica, DOC, and nitrate (Fig. 8), all potentially important solutes in the receiving waters of the downstream Bay-Delta ecosystem—Si and nitrate as nutrients, DOC as a potential aquatic food source and contributor to trihalomethanes generated from drinking water chlorination—were sourced primarily in the uplands during water year 2001 (a dry year). During average water years the upper watershed delivers relatively insignificant amounts of nutrients and sediment when compared to the lowlands (Si being the only exception), and instead becomes an important source of dilute water (Fig. 8).

During water year 2001 the lower watershed acted as a nitrate sink. This is most likely due to two factors: (1) there was not enough precipitation in the lower watershed to effectively flush nitrate from grasslands and agricultural lands (Ahearn et al., 2004), and (2) high hydraulic residence times promoted nutrient uptake and/or denitrification in the stream channel (Triska et al., 1993a, b) and riparian zone (Groffman et al., 1996; Schade et al., 2001). The LME model revealed that grasslands are an important nitrate source during average water years and that the channels flowing through the grasslands are an important nitrate sink during dry years. Since the lower watershed is predominantly grasslands (Fig. 2), the model results are consistent with the flux comparison between the upper and lower watersheds during average and dry years.

Grasslands as an important nitrate source has been reported previously (Hart et al., 1993; Holloway and Dahlgren, 2001) and attributed to asynchrony within nutrient cycling. Instead of a continuous nitrogen feedback among senescing plants, their soils, and new growth (biotic uptake), nitrogen in annual grasslands is mineralized and accumulates in soils during the dry summer and fall months (Hart et al., 1993). With the onset of winter rains, water begins to flow through the upper soil horizons, mobilizing the accumulated nitrate (Holloway and Dahlgren, 2001) before new growth can uptake nutrients. As such annual grasslands are inherently leaky systems with respect to nitrate. The novelty of this analysis lies in the fact that if there are no storms large enough to flush the soil nitrogen pool, as in 2001, then these same grassland dominated sub-basins can become nitrate sinks. The streams flowing through the grassland ecosystems of the Cosumnes Watershed tend to have relatively wide and shade-free channels with gravel substrates and intermittent disturbance regimes, all conditions favorable for nutrient removal from the water column (Allan, 1995). It would seem that within-channel nutrient reduction outweighs nitrate input from grassland contributing areas during dry years. Fig. 9 illustrates nitrate-N inflow and outflow concentration in sub-basin 24 (a grassland dominated sub-basin). It can be seen how nitrate-N outflow concentrations were much lower than inflow concentrations, especially during the baseflow season. This vivid example of stream-channel assimilative capacity is exaggerated by high nitrate input from an upstream WWTP (7 km above the inflow station). The high N loading, depicted in Fig. 9, is only seen in two of

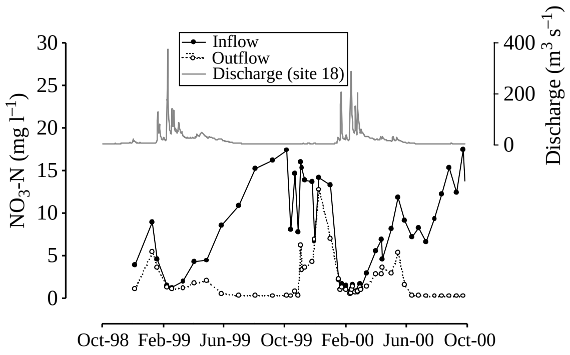

Fig. 9. Temporal nitrate-N fluctuation between the inflow and outflow of sub-basin 24. Discharge from the only gage in the basin (site 18) is graphed in the inset.

the nine grassland dominated sub-watersheds, yet still a number of these grassland watersheds act as annual nitrate sinks—the result most strikingly illustrated as fluxes when comparing Fig. 4c and d.

Outside of grasslands, agriculture and the presence or absence of WWTPs were the dominant controls on nitrate-N loading. Basnyat et al. (1999) and Sliva and Williams (2001) both found urban coverage to be a primary nitrate predictor and indeed our analysis would have come to the same conclusion if we did not factor out those basins with WWTPs (as evidenced in Table 1 where, during average water years, population density has a stronger relation to nitrate than %agriculture). The fact that the basins with substantial urban area but no WWTPs (sub-basins 12, 13, 14, 17, 20, 23, 25) did not contribute to the nitrate-N loading LME model calls into question the implication that urban coverage can be considered an extraordinary source of nitrate NPS pollution; in our analysis it would seem that urban areas do not have a significant influence on stream nitrate-N loading until a major WWTP (a point source) is constructed within the basin. Furthermore, once such a treatment plant is constructed one has to consider where the water that reaches the plant is coming from, as inter-basin transfers are commonplace.

Our analysis indicates that agricultural coverage and population density have nearly equal effects on TSS loading. These findings are in contrast to Hill (1981) who found no correlation between LULC and TSS in a Southern Ontario watershed, but Hill excluded all sub-basins with WWTPs in his analysis, thus reducing the possibility of a correlation between population density and TSS. Others have found %agriculture to have a strong relation with TSS (Allan et al., 1997; Johnson et al., 1997), but we are not aware of any studies that have correlated population density to TSS. This may be evidence that complex studies such as these may only hold valid within spatially limited areas (Baker, 2003), or it may be a reflection of the fact that other researchers have used %urban coverage and not census derived population density as a measure of human habitation impact on water quality. Percent urban coverage fell out of our LME model while population density remained significant, so apparently they are disparate metrics of human habitation.

During 2001, a dry year, TSS loading was greatly reduced, especially from agricultural lands. This reduction is expected as large storms are needed to mobilize sediment and erode stream banks. In agricultural fields a relatively high infiltration capacity needs to be exceeded before surface runoff is initiated and sediment is efficiently exported to the channel. Infiltration is greatly reduced in impervious urban areas, so this may explain why dry year reduction in TSS loading was greater in agricultural lands than in populated areas.

Both Johnson et al. (1997) and Osborne and Wiley (1988) affirm that concentration data should be used when examining landscape influence on water quality. The assertion is that concentration is more directly linked to the health or 'integrity' of the system than loading. Yet in such analyses where hydrologically linked sub-basins are being used concentration data can only be regressed against the total upstream land use (see Hill, 1981; Johnson et al., 1997; Osborne and Wiley, 1988; Smart et al., 1998). Loading, because it is on a mass basis, can be successively subtracted from basin to basin and local environmental variables (rather than total upstream) can be used.

The most salient limitation of using loading rather than concentration is that more error is introduced when loads are calculated. Sources of error include stream gage error, flow modeling error, and error introduced by interpolating concentration data between sampling dates. A simultaneous study analyzing chemical fluxes from the Cosumnes and adjacent Mokelumne watershed found that in the Cosumnes error in flux calculations could be as high as 21.4% (Ahearn et al., in press). Error can be reduced by increasing sampling frequency during the winter, but because our study area was so large storm sampling proved difficult. The primary reason storm sampling was not conducted was that it takes a storm wave approximately 36 h to move through the Cosumnes Basin. In order to accurately compare each sub-basin sampling needs to be conducted on a comparable portion of the hydrograph. In practice this would mean following the storm wave through the basin and sampling over a 36 h period. This, of course, was not feasible during a 3 year study. Although our sampling design introduced some error we feel that disproportionate storm sampling between the sites would have further obfuscated the results. By keeping the sampling at a 2 week interval the flow at the time of sampling at a given site was randomized, the result being a cross section of all flow regimes sampled at each site. We believe that, since the crux of the experiment was to compare sub-basins, that this was the most appropriate sampling design.

5.1. Impact of impoundment on spatial watershed chemistry

Research in the adjacent Mokelumne Watershed has indicated that large impoundments can act to sever the chemical continuity between the uplands and the lowlands (Ahearn, 2004). Like many reservoirs (Hannan, 1979), the Pardee–Camanche reservoir system on the Mokelumne acts as a substantial chemical sink for most constituents (the exceptions, nitrate and phosphate, can be either retained or released). This phenomenon has been witnessed in reservoirs of varying size and function from Tennessee (Higgins, 1978) to Montana (Soltero et al., 1973) to the arid west (Kelly, 2001), and the same phenomenon is expected in other reservoirs impounding major tributaries draining the western Sierra Nevada. The reservoirs reduce the flux of most constituents coming from the upper watershed, and by doing so decrease the importance of the upper watershed to downstream water quality. In the Cosumnes Watershed we have seen how important the upper watershed is in determining downstream loading of nitrate-N and DOC during dry years, and Si during both dry and average water years. These data lead us to believe that the emplacement of dams on 19 of the 20 tributaries draining into the Bay-Delta has shifted the primary source of these dissolved constituents from the upland to the lowlands. Additionally, most urban and agricultural development has occurred downstream of the major impoundments along the Sierran front. With the reservoirs acting as sinks for many constituents and lowland LULC contributing to increased sediment and nutrient loading, the lowlands have become the primary source for NPS pollution.

6. Conclusions

This study has shown which geographic variables have the greatest control on water quality in the Cosumnes Watershed. Percent agricultural coverage had a significant influence on both TSS and nitrate-N loading. Population density also contributed to TSS loading, but did not have an impact on nitrate-N loading when sub-watersheds with WWTPs were not included in the analysis. It is concluded that in the Cosumnes Basin population density does not have a strong influence on stream nitrate-N loading until a WWTP is built within the basin. The final factor driving nitrate-N loading was %grassland. These grasslands acted as a nitrate source during average water years and the channels running through them acted as a nitrate sink during dry years. During dry years, the upper basin is an important source of nitrate and DOC. During both dry and average water years the upper basin is the primary source for Si. Assuming this holds true for other rivers draining the western Sierra Nevada, and seeing as impoundments act as sinks for dissolved species, it is concluded that the emplacement of dams on 19 of the 20 tributaries draining into the Bay-Delta has contributed in shifting the primary source of these dissolved constituents from the upland to the lowlands. Finally, these results indicate that the lowlands are the primary source of TSS, TN, TP, nitrate, and phosphate. Because these constituents are major water quality concerns, restoration/remediation efforts should be focused in lowland areas.

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