

# Nitrogen Dynamics of Spring-fed Wetland Ecosystems of the Sierra Nevada Foothills Oak Woodland<sup>1</sup>

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## Abstract

Spring-fed wetlands are small, highly productive, patchy ecosystems nested within the oak woodland/annual grassland matrix of the Sierra Nevada foothills. In an effort to place these wetlands in a landscape context, we described seasonal variation (1999-2000 growing season) in nitrogen cycling parameters at 6 spring-fed wetland sites of the Sierra Nevada foothill oak woodlands. Ammonium and nitrate pool sizes covaried inversely with nitrate dominance (about 3  $\mu\text{g N g dry soil}^{-1}$ ) in autumn/winter during plant dormancy and ammonium dominance (about 6  $\mu\text{g N g ds}^{-1}$ ) in spring/summer when wetland plants were beginning yearly production cycles. Nitrate is highly mobile in the vadose zone, but can accumulate in the absence of plant growth during dry summer months in the annual grasslands surrounding these wetland systems. With the onset of autumn rains, much of this nitrate may be flushed down slope to spring systems. Spring-fed wetlands may act as either as sources, sinks, or as transformers of this exogenously derived nitrogen.

## Introduction

Ecosystem losses of nitrate are of great ecological and economic concern. Eutrophication of aquatic ecosystems can lead to algal blooms and oxygen depletion (Schlesinger 1997). Nitrate pulses in the Mokelumne River watershed have induced downstream eutrophication leading to fish kills in lower river reaches (Holloway and others 1998). From a human health perspective, excessive nitrate concentrations in drinking water (>20 ppm nitrate-N) can cause methemoglobinemia (Bluebaby syndrome, Fan and others 1987, Fan and Steinberg 1996), an ailment which is potentially fatal to infants. The current water standard for allowable nitrate is based on protection from methemoglobinemia. In 1962, the U.S. Public Health Service recommended a maximum permissible level of 10-ppm nitrate-N in drinking water (McKee and Wolf 1963). However, no case of methemoglobinemia has been identified as related to concentrations below about 100 ppm nitrate-N (Fan and Steinberg 1996); 10 ppm nitrate-N is a conservative maximum permissible level. While conservative, this level is deemed appropriate because drinking water is only one of many potential sources of nitrate ingestion. Nitrate spikes (1 to 100 ppm nitrate-N) have been consistently detected during early season storm events in watershed-wide effluent in California hardwood rangelands (Huang 1997, Holloway

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and others 1998, Lewis 1999, Tate and others 1999), indicating the potential for these watersheds to act as a net nitrate source to downstream ecosystems.

During summer on California's annual grasslands, the absence of plant uptake, coupled with waning microbial-nitrogen demand, results in mineral nitrogen accumulation (mostly as nitrate) in surface soil horizons (Jackson and others 1988, Dahlgren and Singer 1994, Firestone 1995). With the advent of autumn rains, nitrate becomes highly mobile in the vadose zone (Schimel and others 1989); microbes and germinating annual plants may take up some of this nitrate, but much may be lost via leaching to depths below the rooting zone of annual plants (about 30 cm). Davidson and others (1990) showed that, indeed, some nitrate immobilization occurred with initial autumn rains, however, only at microsites where ammonium was in short supply, as ammonium is the preferred form of mineral nitrogen by most organisms.

Spring-fed wetlands in Californian oak woodlands are first-order riparian areas which are usually situated at slope breaks where steeper slopes give way to more gradual terrain. Nutrient cycling within spring-fed wetlands should be quite different from the surrounding annual grasslands given frequent soil saturation, high primary productivity, and species composition and phenological differences. Furthermore, the typical location of springs (at slope breaks) leaves them ideally situated to act as nutrient traps. We sought to 1) describe intra-annual variability in spring-fed wetland N dynamics, 2) explore the potential for nitrate inputs to spring-fed wetlands from the surrounding annual grassland/oak woodland matrix, and 3) measure spring-fed wetland nitrate outputs.

## Methods

### *Study Area*

Owned and managed by the University of California for more than 30 years, the Sierra Foothill Research and Extension Center (SFREC) is located on the western slope of the Sierra Nevada foothills in Yuba County, CA (*fig. 1*). SFREC covers 2300 ha of steep to rolling landscape, 90-600 m above sea level. Annual precipitation at SFREC averages 72 cm·yr<sup>-1</sup> with maximum and minimum air temperatures in the region ranging from 32.0°C in July to 2.2°C in January. Soils in this area are generally shallow and classified as Auburn (loamy, Oxidic, Thermic, Ruptic-Lithic Xerochrepts) and Argonaut (fine, mixed, Thermic Mollic Haploxeralfs) series (Herbert and Begg 1969).

*Figure 2* depicts four arbitrarily chosen spring sites from four distinct watersheds at SFREC—Poleline Ridge (PR), Fireline Ridge (FR), Forbes Valley (FV), and Campbell Roadside (CR). Two of these sites (FV & CR) supported two distinctive plant communities—cattail (FVT & CRT) and graminoid (FVP & CRP) dominant. Separate plots were established for each community type X site combination, resulting in six sampling units or plots.

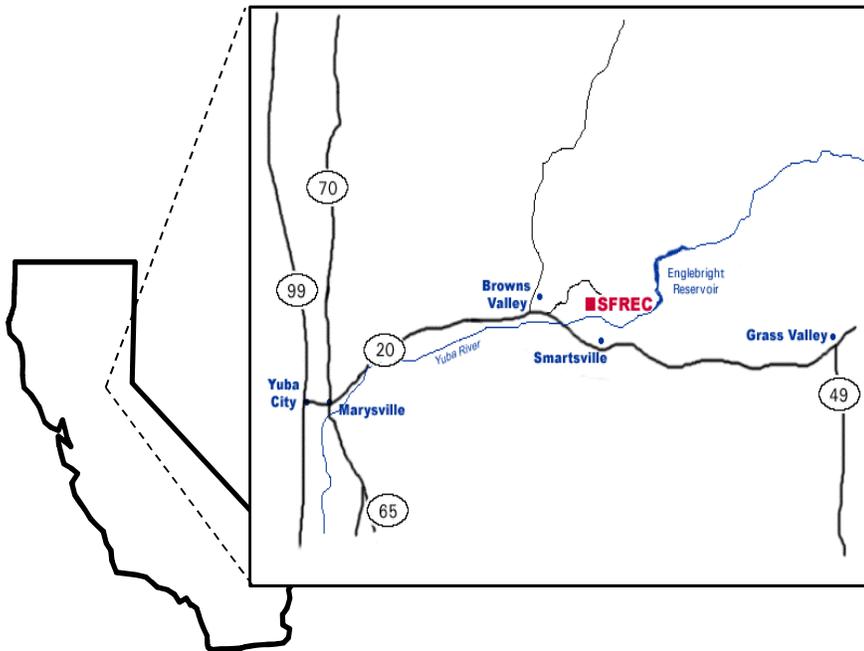


Figure 1—Sierra Foothill Research and Extension Center (SFREC).

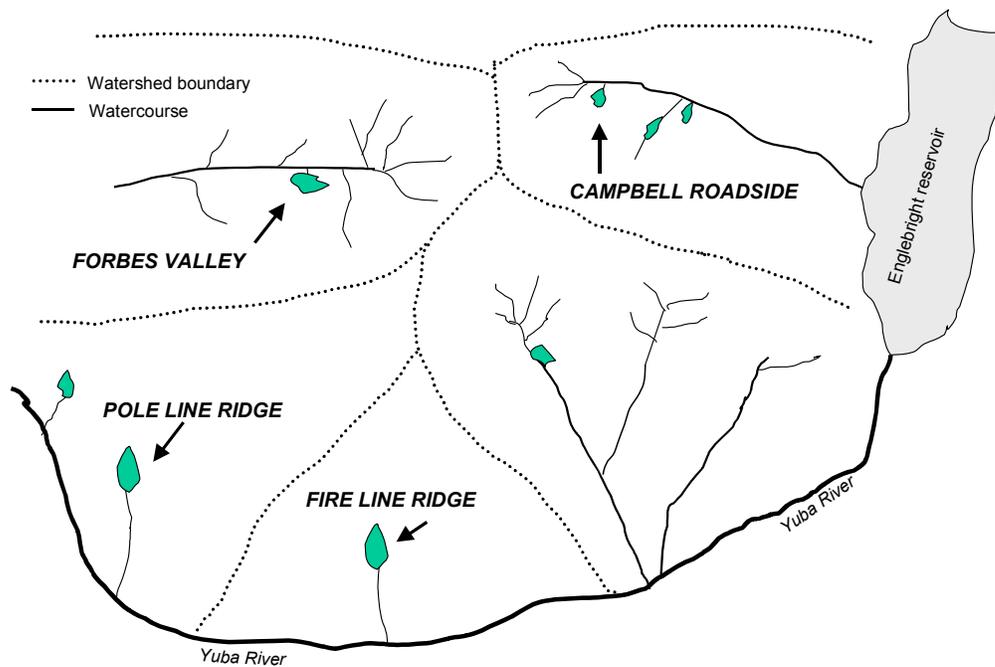


Figure 2—Location of significant spring-fed wetland sites at SFREC.

### **Wetland Soil N Pools**

Wetland soils were collected in September and December 1999 and March and June 2000 from each plot for determination of standing soil N pools. Six soil cores (5-cm diameter X 20-cm deep) were randomly removed from each plot and immediately placed in coolers for transport to the University of California, Berkeley. Cores were stored for a maximum of 48 hours at 4°C prior to hand homogenization to remove roots and rocks. Ten grams of soil were weighed out in duplicate: one subsample for gravimetric water content determination (24 h @ 105°C), and one subsample for immediate extraction in 2M KCl. Extracts were frozen prior to ammonium (method #12-107-06-2-A) and nitrate (method #12-107-04-1-B) determination on the Lachat autosampler (Zellweger Analytics).<sup>3</sup>

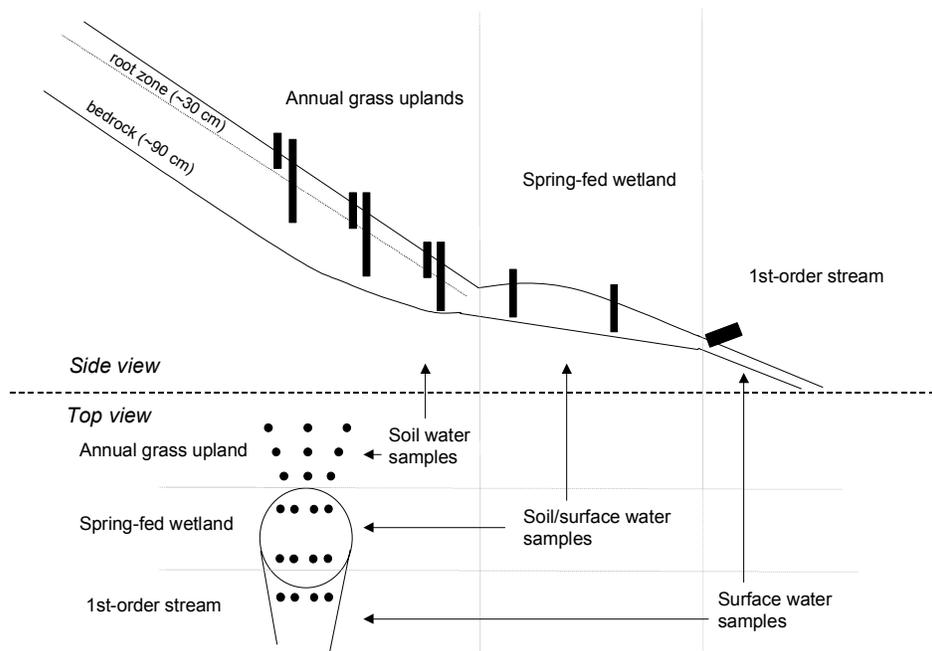
### **Soil and Surface Water Sampling**

To assess the potential for nitrate losses from annual grassland uplands, we collected leachate from three transects (one directly upslope, two at 45° angles; *fig. 3*) of preinstalled, porous cup leachate samplers (Model 1900, Soilmoisture Equipment Corp., Santa Barbara, CA) radiating up slope from each of two spring sites (PR & FV). Leachate samplers were installed in March 2000 at 5-m intervals along each transect at depths of about 30 and about 90 cm to sample from below the annual grassland rooting zone and just above bedrock (*fig. 3*). Lateral flow of water and nutrients is mainly confined to these two soil horizons at SFREC, but is differential due to soil textural class separation (Dahlgren and Singer 1994). Hand vacuum pumps were used to apply negative pressure to soil water samplers (about 70 cbars). Soil water samplers were left under pressure until the following sample period (about 4 weeks) so that concentrations from a given date reflected conditions from the previous sampling date. In several trials, the volume of water extracted reached an asymptote after 2 to 6 days, i.e., little more could be withdrawn from the soil matrix. We were not concerned about denitrification of standing samples because the porous cups of these lysimeters are impermeable to organic matter which is requisite as a labile carbon source for denitrifying bacteria (Firestone 1980, Firestone and Davidson 1989).

When available, upland leachate and wetland surface water were collected on a monthly basis throughout the 2000/01 rain season. Within wetlands, about 10 surface- water samples were collected to assess nitrate output from these systems and to compare output with upland soil nitrate dynamics. All water samples were immediately placed in coolers for transport to UC Berkeley whereupon they were frozen until analysis for nitrate at UC Berkeley's Department of Agriculture and Natural Resources laboratory using the method treated by Carlson (1978).

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<sup>3</sup> Mention of trade names or products is for information only and does not imply endorsement by the U.S. Department of Agriculture.



**Figure 3**—Soil and surface water sampling layout from side and top views.

## Results

### *Wetland Dynamics*

Gravimetric water content (GWC) patterns for wetland soils were site specific (*fig. 4*). Interestingly, the CRT and CRP plots were more similar to the wetter plots (PR and FVT) in autumn and winter, but drier and more similar to the driest plots (FR and FVP) in spring and summer. All but FVT showed an apparent autumn rain response, i.e.,  $GWC_{\text{December}} > GWC_{\text{September}}$  for gravimetric water content. Both CRT and CRP displayed steadily declining GWC after brief autumn response, while PR and FVT steadily increase GWC through winter and spring months. Finally, FR and FVP initially responded to autumn rains, but underwent very gradual decline throughout the growing season; overall, these two plots were significantly drier.

During plant dormancy in autumn and winter, nitrate was the dominant form of mineral nitrogen ( $3.27 \mu\text{g NO}_3^- \text{-N g dry soil}^{-1}$  and  $2.63 \mu\text{g NH}_4^+ \text{-N g dry soil}^{-1}$ , *fig. 5*), whereas ammonium was roughly double nitrate in spring and summer ( $5.08 \mu\text{g NO}_3^- \text{-N g dry soil}^{-1}$  and  $6.30 \mu\text{g NH}_4^+ \text{-N g dry soil}^{-1}$ ), when wetland plants were beginning yearly production cycles.

Spring-fed Wetland N Dynamics—Jackson and Allen-Diaz

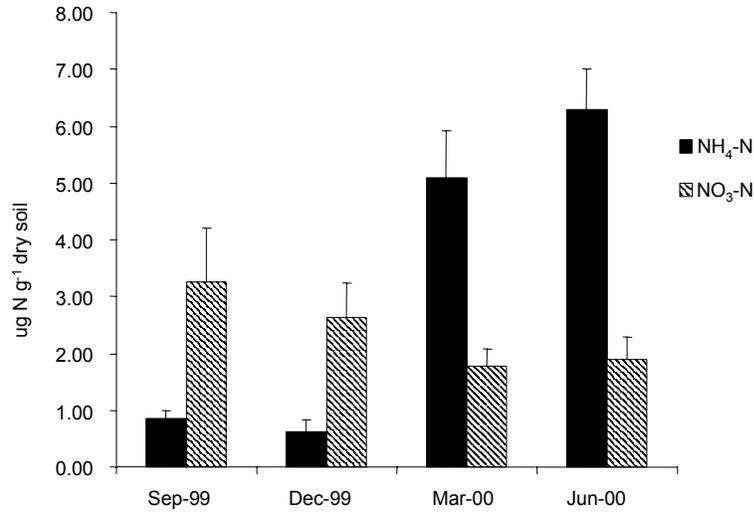


Figure 4—Mineral nitrogen pool sizes averaged over six spring-fed wetland plots.

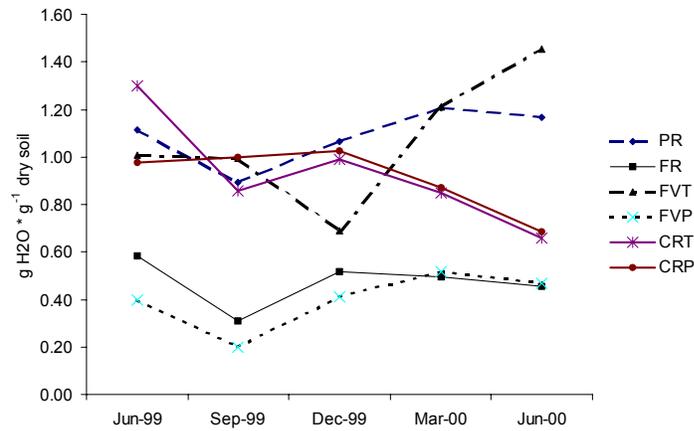
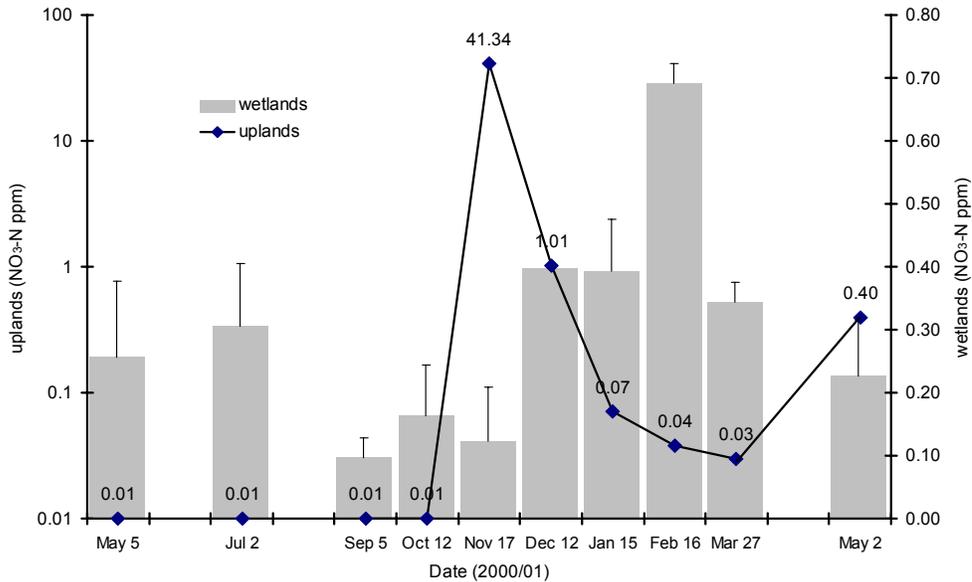


Figure 5—Gravimetric water content of six spring-fed wetland plots.

**Upland-wetland Throughput**

Surface waters of wetlands showed a general pattern of relatively constant nitrate concentrations through the summer and early autumn (about 0.20 ppm nitrate-N), with marked increase during winter months and a peak for the period representing late January through early February (0.69 ppm nitrate-N, *fig. 6*). However, nitrate patterns in upland leachate were temporally variable. We were unable to recover any upland leachate during summer and early autumn. Leachate was finally realized in mid-November, a sample period reflecting late October conditions, which coincided with a cumulative total of about 30 mm rainfall for that season (*fig. 7*). During this period, mean nitrate concentrations were exceptionally

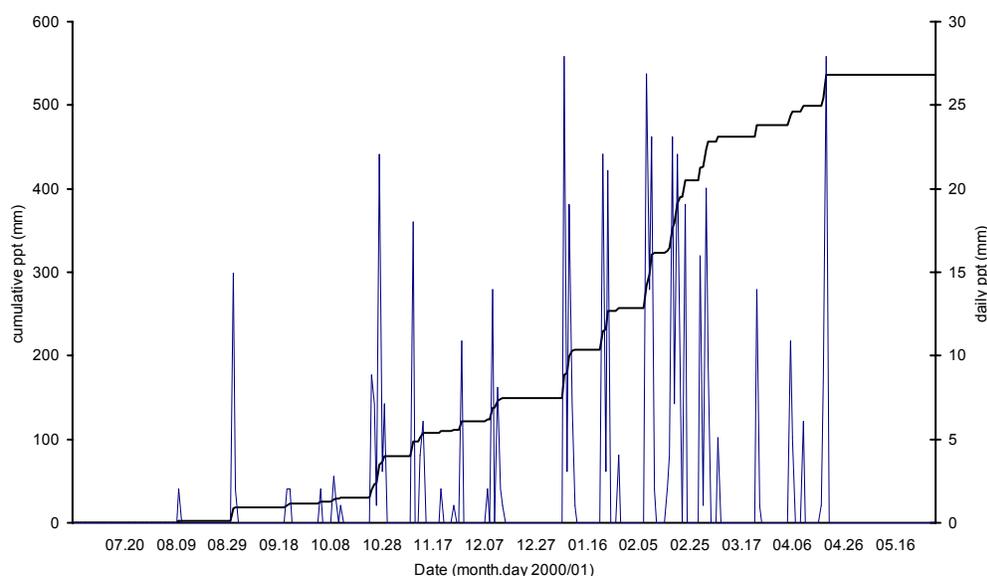
high (41 ppm nitrate-N) but decreased over the remaining winter months to less than 1 ppm nitrate-N. Nitrate concentrations from 30- and 90-cm were averaged as no significant differences were determined between these depths.



**Figure 6**—Soil and surface water nitrate concentrations in spring-fed wetlands and surrounding annual grassland upland matrix. Note log scale for primary y-axis.

## Discussion

What was the fate of nitrate observed in the annual grasslands above the spring-fed wetlands in the late October/early November period? As rainfall accumulated, effectively priming the dry uplands, a wetting front slowly moved down through the soil profile (Huang 1997). In our study, this front appeared to have transported nitrate that had accumulated during summer in the uplands below the annual grassland rooting zone. Most of this nitrate is generated in the surface soil (0-10 cm) where biological activity is concentrated (Hart and others 1993, Schimel and others 1989). At both 30- and 90-cm soil depths, we observed high nitrate concentrations at the November 17 sampling date. This sample represented nitrate-accumulating conditions that occurred since the previous sampling date (October 12). Prior to October 12, little rain had fallen (about 25 mm), but subsequently, an additional 60 mm of rain fell (October 12-November 17). Apparently this was sufficient to carry nitrate to both sampling depths. The wetting front probably was not linear and macropore flow a likely major pathway for downward transport, resulting in no differences between sampling depths.



**Figure 7**—Daily and cumulative rainfall for the 2000/01-water year at SFREC.

By December, it appeared that the front had almost completely flushed the vadose zone of this nitrate pool. A concomitant increase in January and February nitrate concentrations within the surface water of the wetlands indicated that at least some of the upland-derived nitrate probably entered the wetland area. However, we never observed nitrate concentrations  $>1$  ppm, as others have found in higher order streams (Holloway and others 1998, Tate and others 1999). We suggest four possible explanations for this apparent lack of response. First, the pulse of nitrate may have passed through the system undetected. Second, a significant dilution from non-vadose zone ground water may have occurred. Third, the nitrate may have been lost to deep percolation. Lastly, nitrate may have been sequestered by plants, microbes, and/or soil organic matter. These alternatives are discussed in the following sections.

### ***Pulse Not Sampled***

Our monthly sampling may have missed spikes of nitrate moving through the system. However, hourly sampling of Tate and others (1999) from SFREC's Lewis watersheds illustrated that major spikes (3-4 ppm) were apparent for weeks, depending on the intensity and duration of storm events. In another SFREC watershed (Schubert), which is known to have at least four spring-fed riparian zones, nitrate concentrations in stream flow were low (0.5-2 ppm nitrate-N), with spikes as high as 60 ppm nitrate-N (Huang 1997, Lewis 1999). These comparisons must be made with caution; variable source areas may contribute to these disparities. As Tate

and others (1999) discussed, total nitrogen load (concentration X flow rate per unit time) is the best metric for between-watershed comparisons.

We estimated September flow rates at our first-order spring-fed wetlands (area-velocity method) to range from 0.001-0.02 L·s<sup>-1</sup>, compared to the higher order stream output at SFREC's Schubert and Lewis watersheds of about 0.1-400 L·s<sup>-1</sup>. Given the seasonal variation of nitrate-N concentrations we observed in spring-fed output (0.01-1.5 ppm), and assuming a mean flow rate of 0.007 L·s<sup>-1</sup>, the calculated total N load from these systems ranges from a very conservative estimate (i.e., 0.01 ppm nitrate-N for 12 months) of 0.11 kg N·y<sup>-1</sup> to an overestimate (i.e., 1.5 ppm nitrate-N for 12 months) of 15.89 kg N·y<sup>-1</sup>. Upland areas above springs in these watersheds average about 2 ha, therefore, we estimate an average watershed load of 0.55-7.95 kg N·ha<sup>-1</sup>·y<sup>-1</sup>. Total N inputs and outputs to these oak woodland/annual grassland systems has been estimated at about 3 kg N·ha<sup>-1</sup>·y<sup>-1</sup> (Dahlgren and Singer 1994). Given the seasonal variability in nitrate concentrations, e.g., about 1 ppm observed in spring output for only about 3 months, this estimate seems quite plausible. Therefore, if pulses of nitrate >1 ppm passed through these spring-fed wetlands with any regularity, nitrogen inputs to the upland watersheds would need to be much higher to maintain overall system nitrogen balance, hence, productivity. This argues for much of the leachate nitrate observed in the annual grassland uplands in autumn being conserved within the watershed in some fashion.

### ***Dilution***

Our second alternative explanation for the apparent lack nitrate transfer from uplands to wetlands was that nitrate was diluted by non-vadose zone water, i.e., ground water. This seems unlikely given our observation of very low baseline flow (about 0.001 L·s<sup>-1</sup>) in September prior to initiation of autumn rains. This low volume was likely not sufficient to dilute the nitrate carried in about 50 cm worth of rainfall-derived, vadose flow. In other words, most of the water emanating from springs is vadose-zone derived, so likely not significantly diluted by groundwater.

### ***Deep Percolation***

Huang (1997) showed that about 7 percent of the precipitation that fell and was not evapotranspired in autumn went to deep percolation, while the remaining 93 percent moved laterally through the vadose zone until reaching surface water outlets. This 7 percent equated to about 5 mm of the total 107 mm of rainfall observed during the soil-wetting phase of early autumn. We estimated that the autumn nitrate pulse we observed was carried down through the soil profile in about 50 mm worth of rainfall, since a total of 80 mm was observed by November 12 sampling date, and about 30 percent of this was likely lost to evapotranspiration. With an additional 60 mm of rain between November 12 and December 12, we no longer observed extremely high nitrate concentrations. According to Lewis (1999), about 150 mm rainfall are needed before downward flow gives way to significant lateral flow. Nitrate may have been lost to depth during this period, or it may have remained at the wetting front before being "pushed" through to surface waters when about 150 mm rain had accumulated, i.e., late December.

## **Wetland Sequestration**

Our relatively high nitrate concentrations across all months in wetland soil extracts indicate potential for wetlands to trap inputs. The semi-saturated to saturated soil conditions should preclude nitrification of ammonium because it is an obligately aerobic process (Paul and Clark 1996). Once exogenous nitrate has found the wetland, it should be subject to any combination of plant and microbial uptake, soil organic matter adsorption, dissimilatory nitrate reduction to ammonium (Silver and others 2001), or denitrification [ $\text{NO}_3^- \rightarrow \text{N}_2\text{O}_{(g)} + \text{N}_{2(g)}$ ]. However, apparently very little of this nitrate is throughput to stream waters, or it is effectively baffled such that stream water concentrations remain low, but consistent (*sensu* Campbell and Allen-Diaz 1997). Regardless of the sequestration mechanism, exogenous nitrate likely plays an important role in maintaining the productivity of these wetland systems. Likewise, the productivity of these systems affords the important ecosystem service of nutrient conservation at the small catchment scale.

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## **References**

- Campbell, C. G.; Allen-Diaz, B. 1997. **Livestock grazing and riparian habitat water quality: an examination of oak woodland springs in the Sierra Foothills of California**. Gen. Tech. Rep. PSW-GTR-160. Berkeley, CA: Pacific Southwest Forest and Range Research Station, USDA Forest Service
- Carlson, R. M. 1978. **Automated separation and conductimetric determination of ammonia and dissolved carbon dioxide**. Analytical Chemistry 50:1528-1531.
- Dahlgren, R. A.; Singer, M. J. 1994. **Nutrient cycling in managed and non-managed oak woodland-grass ecosystems**. Davis, CA: Final Report, Integrated Hardwood Range Management Program.
- Davidson, E. A.; Stark, J. M.; Firestone, M. K. 1990. **Microbial production and consumption of nitrate in an annual grassland**. Ecology 71:1968-1975.
- Fan, A. M.; Steinberg, V. E. 1996. **Health implications of nitrate and nitrite in drinking water: An update on methemoglobinemia occurrence and reproductive and developmental toxicity**. Regulatory Toxicology and Pharmacology 23:35-43.
- Fan, A. M.; Willhite, C. C.; Book, S. A. 1987. **Evaluation of the nitrate drinking water standard with reference to infant methemoglobinemia and potential reproductive toxicity**. Regulatory Toxicology and Pharmacology 7:135-148.
- Firestone, M. K. 1980. **Nitrous oxide from soil denitrification: factors controlling its biological production**. Science 208:749-751.
- Firestone, M. K. 1995. **Nutrient cycling in managed and non-managed oak woodland-grass ecosystems**. Davis, CA: Final Report, Integrated Hardwood Range Management Program.

- Firestone, M. K.; Davidson, E. A. 1989. **Microbial basis of NO and N<sub>2</sub>O production and consumption in soil.** In: Andreae, M. O.; Schimel, D. S., editors. Exchange of trace gases between terrestrial ecosystems and the atmosphere. John Wiley & Sons; 7-21.
- Hart, S. C.; Firestone, M. K.; Paul, E. A.; Smith, J. L. 1993. **Flow and fate of soil nitrogen in an annual grassland and a young mixed-conifer forest.** Soil Biology and Biochemistry 25:431-442.
- Herbert, F. W.; Begg, E. L. 1969. **Soils of the Yuba Area, California.** University of California, Davis: County of Yuba, CA.
- Holloway, J. M.; Dahlgren, R. A.; Hansen, B.; Casey, W. H. 1998. **Contribution of bedrock nitrogen to high nitrate concentrations in stream water.** Nature 395:785-788.
- Huang, X. 1997. **Watershed hydrology, soil, and biogeochemistry in an oak woodland annual grassland ecosystem in the Sierra Foothills, California.** University of California, Davis. Ph.D. dissertation.
- Jackson, L. E.; Strauss, R. B.; Firestone, M. K.; Bartolome, J. W. 1988. **Plant and soil nitrogen dynamics in California annual grassland.** Plant and Soil 110:9-17.
- Lewis, D. J. 1999. **Water quality and yield in a California oak woodland watershed: a 17-year study.** MS thesis. University of California, Davis.
- McKee, J. E.; Wolf, H. W. 1963. **Water quality criteria.** Publication #3-A, State Water Quality Control Board, Sacramento.
- Paul, E. A.; Clark, F. E. 1996. **Soil microbiology and biochemistry.** San Diego, CA: Academic Press.
- Schimel, J. P.; Jackson, L. E.; Firestone, M. K. 1989. **Spatial and temporal effects of plant-microbial competition for inorganic nitrogen in a California annual grassland.** Soil Biology and Biochemistry 21:1059-1066.
- Schlesinger, W. H. 1997. **Biogeochemistry: an analysis of global change.** Academic Press, San Diego.
- Silver, W. L.; Herman, D. J.; Firestone, M. K. 2001. **Dissimilatory nitrate reduction to ammonium in upland tropical forest soils.** Ecology 82:2410-2416.
- Tate, K. W.; Dahlgren, R. A.; Singer, M. J.; Allen-Diaz, B.; Atwill, E. R. 1999. **On California rangeland watersheds: timing, frequency of sampling affect accuracy of water quality monitoring.** California Agriculture 53:44-48.