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Nitrate and Sediment Fluxes from a California Rangeland Watershed

David J. Lewis, Michael J. Singer,* Randy A. Dahlgren, and Kenneth W. Tate

ABSTRACT

Long-term water quality records for assessing natural variability, impact of management, and that guide regulatory processes to safeguard water resources are rare for California oak woodland rangelands. This study presents a 20-yr record (1981–2000) of nitrate-nitrogen ($\text{NO}_3\text{-N}$) and suspended sediment export from a typical, grazed oak woodland watershed (103 ha) in the northern Sierra Nevada foothills of California. Mean annual precipitation over the 20-yr period was 734 mm yr^{-1} (range 366–1205 mm yr^{-1}). Mean annual stream flow was 353 mm yr^{-1} (range 87–848 mm yr^{-1}). Average annual stream flow was $48.1 \pm 16\%$ of precipitation. Mean annual $\text{NO}_3\text{-N}$ export was $1.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (range 0.18–3.6 $\text{kg ha}^{-1} \text{ yr}^{-1}$). Annual $\text{NO}_3\text{-N}$ export significantly ($P < 0.05$) increased with increasing annual stream flow and precipitation. Mean daily $\text{NO}_3\text{-N}$ export was $0.004 \text{ kg ha}^{-1} \text{ d}^{-1}$ (range 10^{-5} to $0.55 \text{ kg ha}^{-1} \text{ d}^{-1}$). Mean annual suspended sediment export was $198 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (range 23–479 $\text{kg ha}^{-1} \text{ yr}^{-1}$). There was a positive relationship ($P < 0.05$) between annual suspended sediment export, annual stream flow and precipitation. Mean daily suspended sediment export was $0.54 \text{ kg ha}^{-1} \text{ d}^{-1}$ (range 10^{-4} to $155 \text{ kg ha}^{-1} \text{ d}^{-1}$). Virtually no sediment was exported during the dry season. The large variation in daily and annual fluxes highlights the necessity of using long-term records to establish quantitative water quality targets for rangelands and demonstrates the difficulty of designing a water quality monitoring program for these ecosystems.

APPROXIMATELY 3 million hectares of oak woodlands and savanna in the interior valleys and foothills of California are used extensively for cattle (*Bos taurus*) grazing, firewood production, wildlife habitat, and other land uses (Bolsinger, 1987; Griffin, 1988; Standiford and Howitt, 1993). Urbanization is rapidly occurring across parts of the region with potentially negative consequences for wildlife habitat, water quality, and other ecosystem services (Wacker and Kelly, 2004). These rangelands contribute significant surface drinking water supplies, with two-thirds of all drinking water reservoirs in the State located within grazed oak woodland/annual grassland ecosystems (FRRAP, 1988). Concerns about drinking water quality degradation and declines in fisheries, such as coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*), have led to the listing of associated water bodies (e.g., rivers, reservoirs) as impaired following Section 303d of the Federal Clean Water Act (Lewis et al., 2002). Impairment indicates that the

water body is not achieving its designated beneficial use(s) (e.g., municipal drinking water, cold water fisheries habitat). During the listing process, water resources agency staff must identify the cause of the impairment (e.g., excessive N and sediment) and land use(s) generating the impairment. Once a water body is listed as impaired, water resources protection agencies must develop and implement watershed–water body specific maximum allowable loading targets known as total maximum daily loads (TMDLs) for the specific nonpoint source pollutant of concern.

Development of the allowable loading limit and a watershed-scale strategy to achieve it depends on accurate estimation, among other things, of: (i) loading rates from specific land uses as well as natural background levels within the watershed; (ii) load reductions required from each land use to achieve the beneficial use(s); and (iii) watershed management scenarios that achieve the required load reductions (USEPA, 1991, 1999a, 1999b). Load estimations can and have been made from watershed specific water quality data, from watershed–water quality simulation models developed and calibrated with data representative of local watershed conditions, and from remote techniques such as aerial photographic review of historic and current erosion features (CRWQCB, 2002). By definition, a TMDL is the total maximum daily load (kg d^{-1}) of specified constituents that a water body can receive and still meet its designated beneficial use. However, in application a TMDL can take the shape of an annual load estimation for sediment (CRWQCB, 2002), or a seasonal calculation for N and P derived from a theoretical or empirical watershed-scale water quality model (ADEQ, 2000).

Long-term, watershed-scale water quality data are invaluable in the TMDL development process, and in evaluation of progress toward achieving loading reductions and beneficial uses of listed water bodies. In general, there are insufficient long-term data available to confidently estimate loading rates for specific land-use types, or to develop and calibrate watershed-scale simulation models to estimate water body response to various load reduction scenarios. This is certainly the case for California's oak woodlands and annual grasslands, where significant annual, seasonal, and storm-event variability in watershed-scale flow and water quality have been demonstrated at multiple spatial scales (Tate et al., 1999; Lewis et al., 2000, 2002; Holloway and Dahlgren, 2001; Ahearn et al., 2004; Dahlgren et al., 2005). The TMDL development guidelines acknowledge the need to account for inherent variability (e.g., annual weather fluctuation) in the establishment of maximum allowable loading targets via the addition of a margin of safety to the maximum

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Abbreviations: LOD, limit of detection; TMDL, total maximum daily load.

allowable loading estimate (USEPA, 1991). However, without long-term records of weather, streamflow, and water quality, it is extremely difficult to establish credible margins of safety that incorporate natural variability into estimates of daily, seasonal, or annual loading estimates, without creating undue economic hardship and managerial constraints for land managers.

The purpose of this paper is to report a unique, continuous 20-yr record (1981–2000) of $\text{NO}_3\text{-N}$ and suspended sediment export from a typical, grazed annual rangeland watershed (103 ha) in the northern Sierra Nevada foothills of California. Impairment by nutrients and sediments, along with microbial pathogens, are the primary water quality concerns on California rangelands. We report water quality data as daily and annual loading rates with the goal that they have utility for: (i) development of maximum allowable loading rates from rangeland watersheds; (ii) incorporation of inherent temporal variability in development of margins of safety for maximum allowable loading rate estimations; and (iii) development of loading rate reduction targets for annual rangeland watersheds. In addition, we report statistical relationships identified between precipitation and stream flow, and annual suspended sediment and $\text{NO}_3\text{-N}$ fluxes.

METHODS

Study Area

The study site is located on the University of California's Sierra Foothill Research and Extension Center, approximately 96 km northeast of Sacramento (Fig. 1) ($39^\circ 15' \text{ N}$, $121^\circ 17' \text{ W}$). Elevations in the 103 ha Schubert watershed range from 152 to

427 m. The average slope is 18% with a range from 2 to 50%. Climate is Mediterranean with more than 90% of annual precipitation falling from November to March as rainfall. Over the 20-yr monitoring period, mean annual rainfall ($\pm 1 \text{ SD}$) was $734 \pm 253 \text{ mm yr}^{-1}$ (range 366–1205 mm yr^{-1}) and median annual rainfall was 684 mm yr^{-1} . Mean annual stream flow ($\pm 1 \text{ SD}$) was $353 \pm 225 \text{ mm yr}^{-1}$ (range 87–848 mm yr^{-1}) and median annual stream flow was 330 mm yr^{-1} . The mean annual air temperature has been 15°C .

Vegetation at the site is typical of Sierran foothill oak woodlands that are dominated by blue oak (*Q. douglasii* Hook and Arn.) with minor interior live oak (*Q. wislizenii* A.D.C.) and foothill pine (*Pinus sabiniana* Douglas) (Griffin, 1988). Vegetation patterns across the watershed are a mosaic of open grassland and woodland (Epifanio, 1989; Jansen, 1987). Blue oak density ranges from 90 to 200 trees per hectare with approximately 70% average canopy cover for the watershed. Annual grasses and legumes dominate the ground cover, with differing species diversity under the oak canopy compared to open grasslands (Jackson et al., 1990; Jansen, 1987).

Soils in the watershed are Ruptic-Lithic Xerochrepts on the steep side slopes and Mollic Haploxeralfs on the more level areas (Lytle, 1998). Soils formed in residuum from basic meta-volcanic (greenstone) bedrock (Beiersdorfer, 1979) are shallow to moderately deep, medium textured, gravelly, and rocky. Surface horizons have a high organic C content (63 g kg^{-1}) and a relatively high infiltration rate (1.8 cm h^{-1}) (Huang, 1997). Soils are rich in Fe oxides, with a mean dithionite-extractable Fe content of 31 and 35 g kg^{-1} in the A/AB and Bt horizons, respectively. They have very stony clay subsoils with an average clay content of 400 g kg^{-1} and a saturated hydraulic conductivity of about 0.68 cm h^{-1} (Huang, 1997). The horizon in contact with the weathered bedrock had the lowest saturated hydraulic conductivity (0.012 cm h^{-1}), which in conjunction with the shallow bedrock (1–1.5 m depth) limits deep percolation of water (Dahlgren and Singer, 1994).

The watershed is used for light to moderate seasonal beef cattle grazing primarily from January to March and August to October. Average stocking rate over the 20-yr period was $0.22 \text{ animal unit months ha}^{-1}$ (Lewis, 1998). Grazing levels on the watershed for the 20-yr record fell within recommended levels for moderate grazing, as defined by October residual dry matter levels ($600\text{--}1000 \text{ kg ha}^{-1}$), on annual rangelands given site specific land slope and annual rainfall levels (Clawson et al., 1982; Bartolome et al., 2002). In addition to normal grazing management, 1352 oaks were selectively harvested from 15 ha in the northern upland portion of the watershed between July 1984 and May 1986 (Epifanio, 1989). Trees were removed during the spring and summer months when soils were dry. Trees were not removed from drainage-ways or from slopes steeper than 30%.

Streamflow Measurement and Water Quality Sampling

Hydrology and water quality have been measured in the watershed since 1978 with dependable records starting in 1981. The watershed was instrumented with a 0.912-m (3-foot) Parshall flume for storm flow measurements and a tandem 0.304-m (1-foot) 90° V-notch weir for base flow measurements (Lewis et al., 2000). Stage height was continuously recorded with stage height recorders located within stilling wells connected to the flume and weir. Stage height was converted to flow using standard methods (Linsley et al., 1982). Rainfall amount and intensity were recorded with tipping bucket rain gauges and recorders at 198, 290, and 328 m elevations within the watershed. Using this continuous hydrologic record, annual rainfall

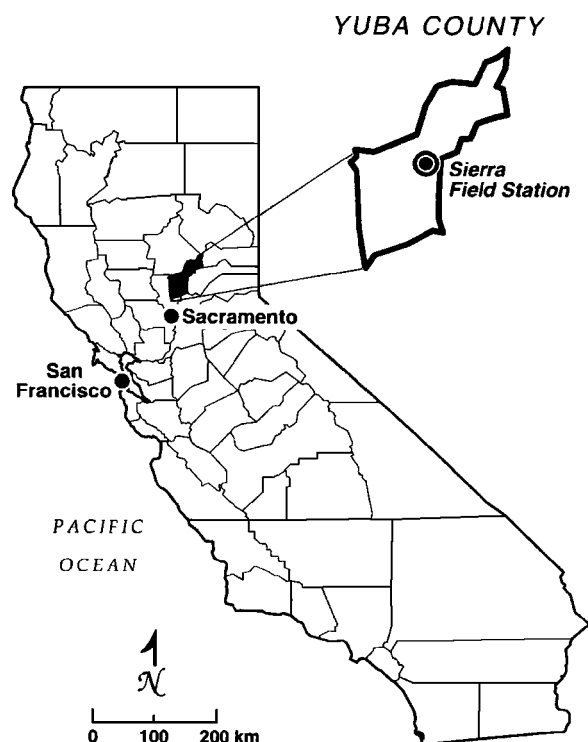


Fig. 1. Location of the Sierra Foothills Research and Extension Center in northern California.

and runoff were calculated as the sum of all rainfall events or discharge measured during each water year (1 October–30 September). No storm hydrographs were missed during the 20-yr period, but water flow during base-flow periods was not measured at times due to equipment failure. Flows for these times were estimated as linear changes between measured flows.

Stream-water samples were collected during rain-free periods, as well as on a storm-event basis, from the center of the stream 10 m upstream from the flume and weir using an ISCO (ISCO, Lincoln, NE) automatic pump sampler. Sampling frequency over the 20-yr record was more intense during storms (about one sample every 1–4 h) than during rain-free periods (about one–two samples per 7 d). This sampling protocol is consistent with the concept of flow proportional sampling. Samples were refrigerated at 3°C through completion of all analyses. Samples were filtered through a 0.45- μm membrane filter and analyzed for pH, electrical conductivity, total alkalinity, suspended solids, and Ca^{2+} , K^+ , Na^+ , Mg^{2+} , Cl^- , NH_4^+ , NO_3^- , ortho- PO_4^{3-} , and SO_4^{2-} concentrations. Only suspended sediment and $\text{NO}_3\text{-N}$ data are presented here. The weight of suspended sediments was determined by the difference in weight of the 0.45- μm filter before and after filtering a known volume of water (LOD = 0.1 mg L^{-1}). Nitrate concentrations were determined using a Dionex ion chromatograph (LOD = 0.01–0.04 mg N L^{-1}).

Constituent fluxes or loads were calculated using a period-weighting method (Moldan and Cerny, 1994). This method multiplied the average of the concentration at the beginning and end of a time interval by the discharge volume during that time interval. Any missing baseflow sample concentrations were estimated from the 20-yr average base flow concentration for each constituent. Daily loads were then summed to generate annual loads. This method was chosen based on recommendations from studies comparing various flux calculation methods (Semkin et al., 1994; Dann et al., 1986). The period-weighting method is commonly employed in watershed studies (e.g., Likens et al., 1995) and provides daily and annual load estimations.

Statistical Analysis

Standard probability plots were created using log transformed daily $\text{NO}_3\text{-N}$ and sediment masses. Daily values were estimated using the period-weighting method described previously. Simple linear regression analysis was used to determine relationships among annual precipitation, stream flow, and suspended sediment and $\text{NO}_3\text{-N}$ fluxes from the watershed. Annual suspended sediment and $\text{NO}_3\text{-N}$ loads were \log_{10} transformed to provide a normal distribution of the data.

RESULTS AND DISCUSSION

Precipitation and Stream Flow

Precipitation and stream flow varied significantly over the 20-yr record (Fig. 2). Mean annual rainfall (± 1 SD) was $734 \pm 253 \text{ mm yr}^{-1}$ (range 366–1205 mm yr^{-1}) and mean annual stream flow (± 1 SD) was $353 \pm 225 \text{ mm yr}^{-1}$ (range 87–848 mm yr^{-1}). Average annual stream flow, as a percentage of precipitation was $48.1 \pm 16\%$ and ranged from 19 to 76%. Annual stream flow increased significantly with increased annual precipitation (Table 1). We found no measurable change in stream flow that could be attributed to selective removal of oak trees early in the 20-yr study period (data not shown).

Precipitation amount and timing during the rainy season were important factors regulating the stream flow/rainfall ratio. Over the course of the rainy season, the first storms produced no appreciable change in stream flow (Lewis et al., 2000). A mean precipitation depth of 47 mm (range 26–69 mm) was required to “prime” the watershed and generate a measurable stream response to precipitation after the dry summer. Once the watershed soils reached field capacity, each storm event produced a rapid response in the stream hydrograph. This is illustrated by the 1984 water year (October 1983–

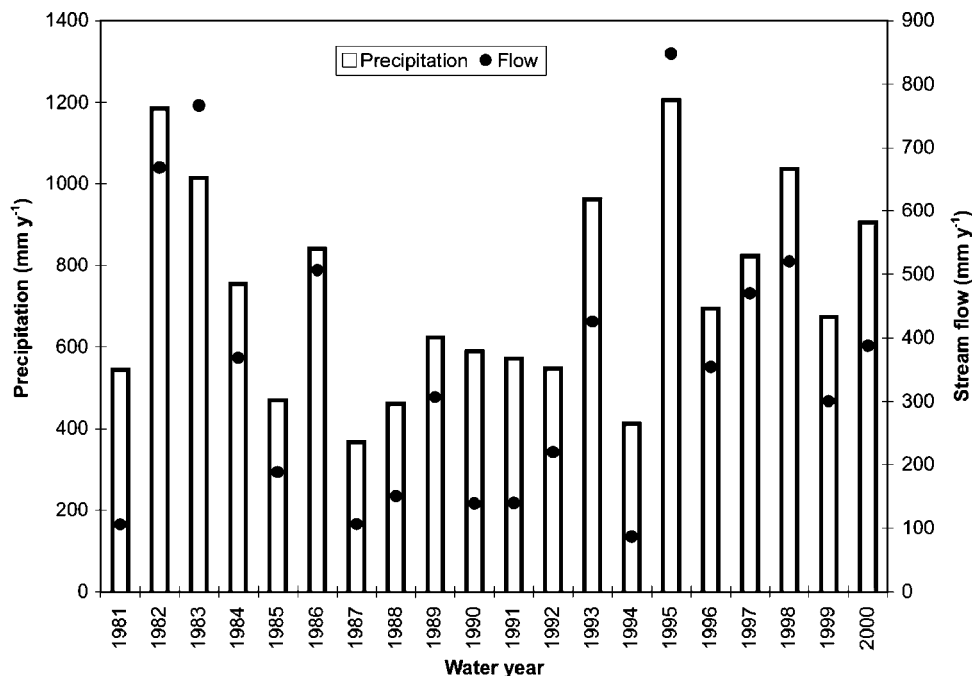


Fig. 2. Precipitation and stream flow in the Schubert watershed for water years 1981 to 2000.

Table 1. Linear regression equations depicting relationships between annual rainfall (P), stream flow (Q), total suspended solid load (TSS), and nitrate-nitrogen load ($\text{NO}_3\text{-N}$) discharged from the Schubert Watershed over a 20-yr period (1981–2000).

Dependent variable	Linear regression equation	p value	r^2
Stream flow (Q) [†]	$Q = -258.14 + 0.83*(P)$ [‡]	<0.001	0.88
Total suspended sediment (\log_{10} TSS) [§]	$\log_{10} \text{TSS} = 1.00 + 0.0015*(P)$ $\log_{10} \text{TSS} = 1.49 + 0.0017*(Q)$	<0.001	0.68
Nitrate nitrogen (\log_{10} $\text{NO}_3\text{-N}$)	$\log_{10} \text{NO}_3\text{-N} = -0.39 + 0.0007*(P)$ $\log_{10} \text{NO}_3\text{-N} = -0.23 + 0.0009*(Q)$	0.036 0.006	0.22 0.34

[†] Annual stream flow discharged from the Schubert Watershed (mm).

[‡] Annual rainfall (P) for Schubert Watershed (mm).

[§] \log_{10} transformed annual total suspended sediment load discharged from Schubert Watershed ($\text{kg ha}^{-1} \text{yr}^{-1}$).

^{||} \log_{10} transformed annual nitrate-nitrogen load discharged from Schubert Watershed ($\text{kg ha}^{-1} \text{yr}^{-1}$).

September 1984), which had near average precipitation for the 20-yr record. Four early season precipitation events totaling 37.8 mm produced only a small increase in the stream flow. The next 8 mm produced a notable rise in the stream hydrograph and later precipitation events produced large and distinct increases in the hydrograph (Fig. 3). In the spring, the hydrologic response of the stream declined as storm size and intensity declined along with an increase in evapotranspiration due to increasing temperature and plant growth (data not shown). Thus, water yields are greatest for winter season precipitation events following soil water replenishment in the fall and before elevated evapotranspiration rates in the spring.

Nitrate-Nitrogen

We found considerable variability in the year-to-year $\text{NO}_3\text{-N}$ export from the watershed (Fig. 4). Mean

annual $\text{NO}_3\text{-N}$ export from the watershed was $1.6 \text{ kg ha}^{-1} \text{yr}^{-1}$ (range $0.18\text{--}3.6 \text{ kg ha}^{-1} \text{yr}^{-1}$) (Fig. 4). Median $\text{NO}_3\text{-N}$ export was $1.2 \text{ kg ha}^{-1} \text{yr}^{-1}$. Daily $\text{NO}_3\text{-N}$ export was also highly variable, with the greatest loads occurring between November and March (Fig. 5b). Mean daily $\text{NO}_3\text{-N}$ export was $0.004 \text{ kg ha}^{-1} \text{d}^{-1}$ (range 10^{-5} to $0.55 \text{ kg ha}^{-1} \text{d}^{-1}$) (Fig. 5b). Nitrate was the only significant form of mineral N exported by the stream from the watershed; average $\text{NH}_4\text{-N}$ export was $<0.1 \text{ kg ha}^{-1} \text{yr}^{-1}$. While annual $\text{NO}_3\text{-N}$ fluxes reached values as high as $3.6 \text{ kg N ha}^{-1} \text{yr}^{-1}$, loss of NO_3 from the watershed was substantially less than the combined inputs of mineral N ($\text{NH}_4^+ + \text{NO}_3^-$) in the bulk precipitation (mean 9.4 and maximum $17.8 \text{ kg N ha}^{-1} \text{yr}^{-1}$). As a result, there is an apparent net retention of mineral N within the watershed. We did not measure organic forms of N or gaseous losses, and have not accounted for N exported by animal consumption of the grass, so we were not able to calculate a complete mass balance for N in the watershed. We expect volatilization losses to be small. Herman et al. (2003) measured gas emissions of $<1 \text{ ng N cm}^{-2} \text{h}^{-1}$ and concluded that N volatilization losses from this watershed were not an important pathway for N loss.

Annual $\text{NO}_3\text{-N}$ load was positively and significantly ($P < 0.05$) related to both annual precipitation and stream flow (Table 1). The years with the highest export were a function of a combined very high flow volume and high $\text{NO}_3\text{-N}$ concentrations during some storm events. Daily export was often high during the early portion of the rainy season when $\text{NO}_3\text{-N}$ concentrations are generally highest in stream water (Fig. 5a, 5b, and Fig. 6). In California oak woodlands, stream water NO_3 concentrations are highest during the early rainfall season due to a temporal asynchrony in the N cycle (Tate et al., 1999; Holloway and Dahlgren, 2001; Jackson et al., 2006). Once early season rainfall begins to produce a significant change in the stream hydrograph, it also flushes N from the system

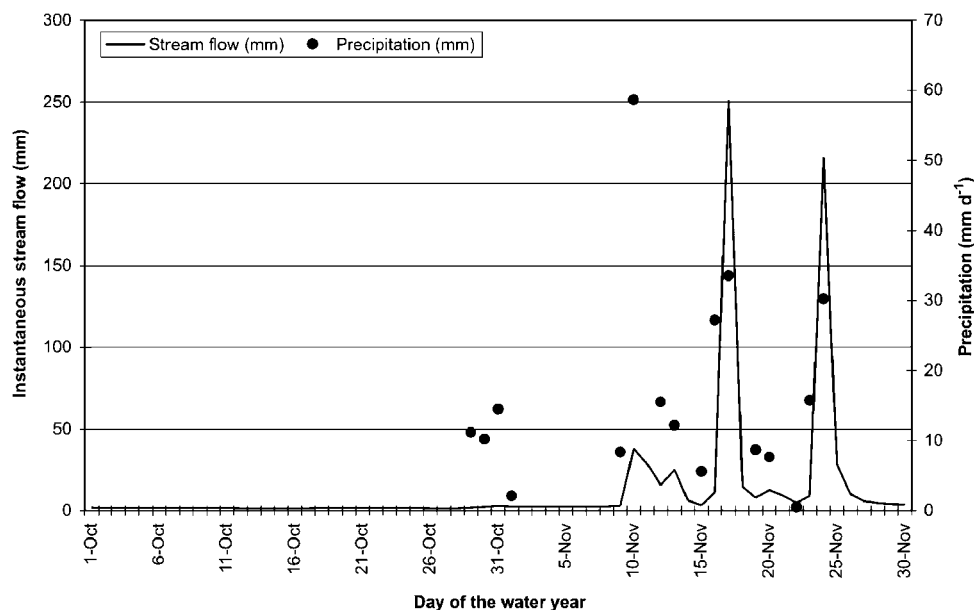


Fig. 3. Early season stream flow and precipitation for 1984 illustrating early season “priming” of the watershed soils.

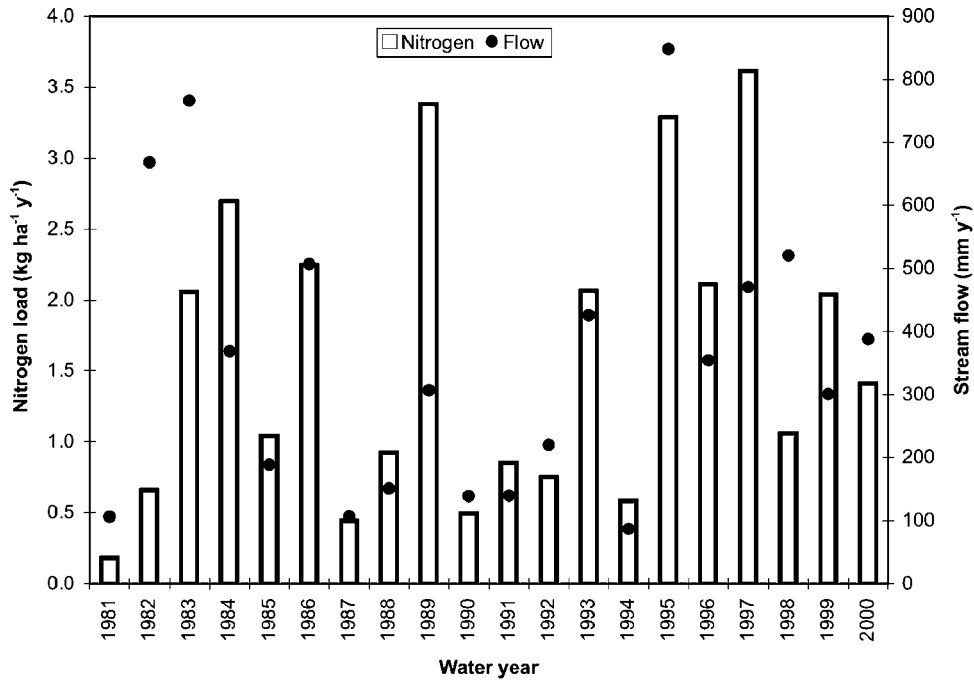


Fig. 4. Annual N export and stream flow from the Schubert watershed 1981 to 2000.

(Fig. 6). Subsequent storms have lower N concentrations as the season progresses (Fig. 6). The reason for this behavior is that NO_3^- accumulates in the soil during the summer and fall periods, when plant uptake and leaching processes are very low. Nutrient uptake by oak trees is

limited during the summer due to the soil water deficit and they become dormant in the fall and remain inactive until late March. Similarly, annual grasses die in June, and seeds germinate with the first fall precipitation. However, their growth is limited by cold temperatures until about the

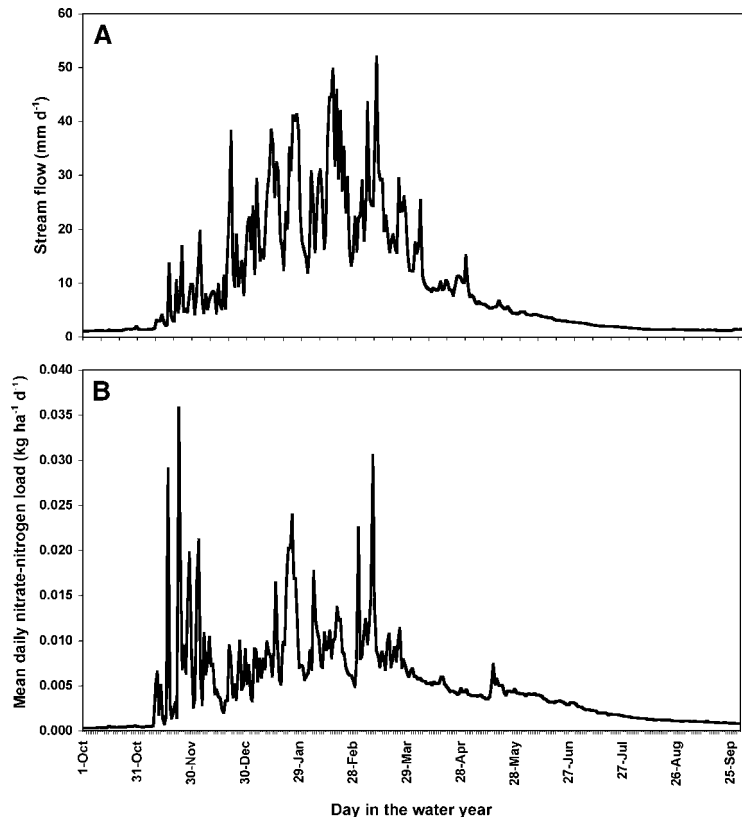


Fig. 5. (A) Mean daily stream flow for the Schubert watershed 1981 to 2000. (B) Mean daily nitrate-nitrogen ($\text{NO}_3\text{-N}$) load for the Schubert watershed 1981 to 2000.

beginning of March. Therefore, NO_3 that accumulates in the soil over the dry period is highly susceptible to leaching. The accumulated NO_3 is progressively flushed out of the soil, resulting in lower $\text{NO}_3\text{-N}$ concentrations in stream water with each subsequent storm. The export of $\text{NO}_3\text{-N}$ decreases throughout the spring as plant uptake removes NO_3 from the soil zone. The decreased $\text{NO}_3\text{-N}$ fluxes also result from a progressive decrease in baseflow from spring to summer (Fig. 5a). Thus, early to midwinter precipitation events result in the highest yields of $\text{NO}_3\text{-N}$ from the watershed.

Suspended Sediment

Mean annual suspended sediment export from the watershed was $198 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (range $23\text{--}479 \text{ kg ha}^{-1} \text{ yr}^{-1}$) (Fig. 7). Annual suspended sediment export was significantly ($P < 0.05$) related to annual stream flow and precipitation, with increasing export as annual stream flow and precipitation increased (Table 1). Mean daily suspended sediment export was $0.54 \text{ kg ha}^{-1} \text{ d}^{-1}$ (range 10^{-4} to $155 \text{ kg ha}^{-1} \text{ d}^{-1}$) (Fig. 8b). Virtually no sediment was exported from the watershed during the dry

season (Fig. 8b). Maximum daily suspended sediment export was linked to high sediment concentrations and high stream flow occurring during large storm events (Fig. 8a).

Suspended sediment exports are small relative to several other annual rangeland watersheds with similar hydrologic characteristics, but differing soil types (Lewis et al., 2002). The high concentrations of secondary Fe oxides and organic matter promote high infiltration rates and strong aggregate stability, leading to low erosivity. The high canopy coverage (70%) and adequate herbaceous cover ($600\text{--}1000 \text{ kg ha}^{-1}$ October residual dry matter) further attenuate surface runoff and erosion by limiting raindrop impact on the soil surface. Ground cover by herbaceous plants exceeds 90% in the watershed during the rainy season, helping to ensure a low erosion rate.

Implications for the Total Maximum Daily Load Development Process

These data have several important implications for the TMDL process when applied to annual rangeland water-

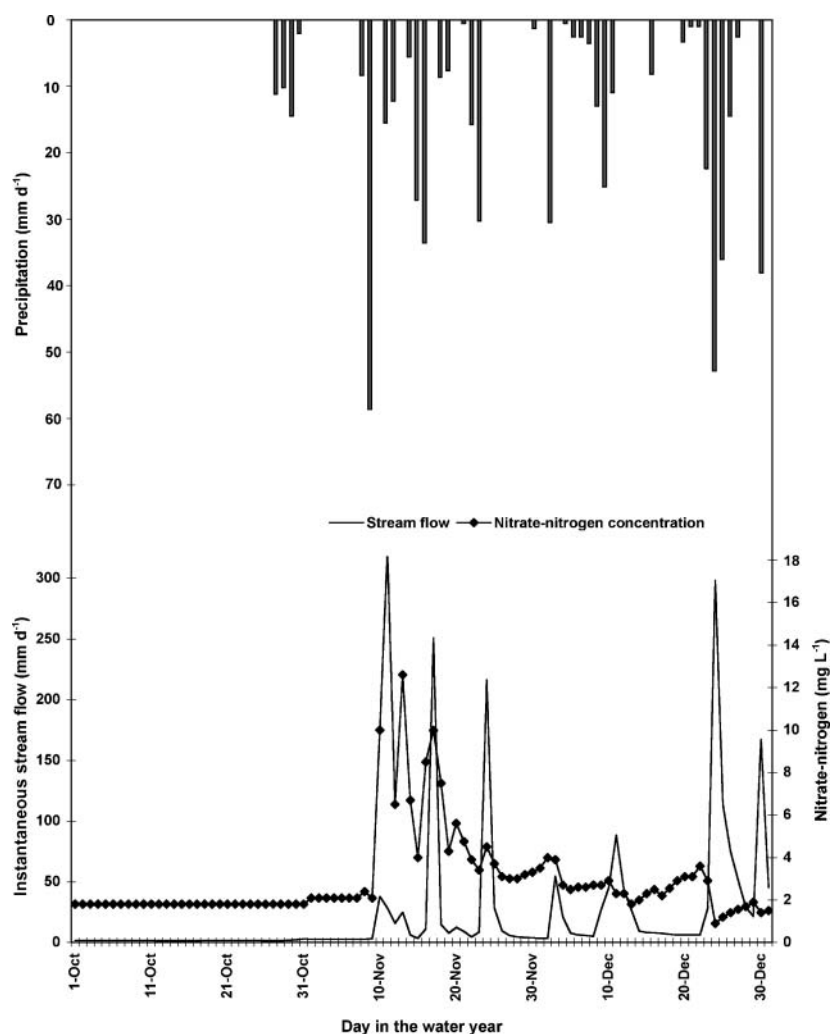


Fig. 6. Early season stream flow and nitrate-nitrogen concentration for the 1984 water year, and early season precipitation events for the 1984 water year.

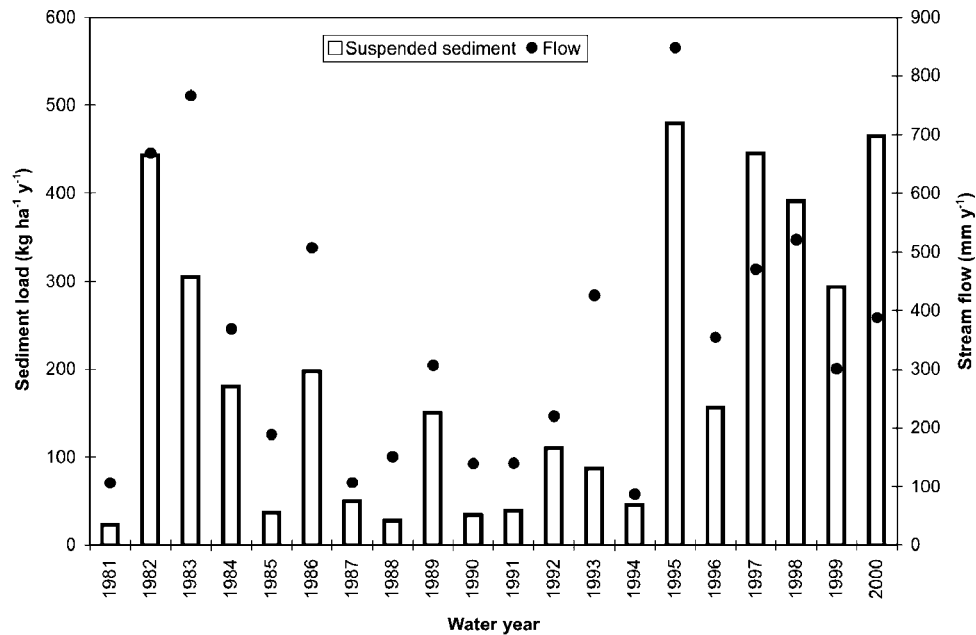


Fig. 7. Annual suspended sediment export and stream flow from the Schubert watershed 1981 to 2000.

sheds, and illustrate the overall value of investment in long-term, strategic monitoring by agencies involved in water quality management and protection. An important use of long-term hydrology and water quality data is to calibrate and validate watershed-scale water quality models to conditions typical of annual rangeland watersheds. The use of such models will be required to address issues of spatial scale and variability in source assessments, to bridge data gaps, and to evaluate the potential effectiveness of alternative watershed management scenarios to reduce pollutant loadings. To have confidence in loading estimates and watershed management plans developed during the TMDL process, the models employed to develop these estimates and plans must be evaluated against real watershed-scale data. For example, Lewis et al. (2000) were able to use a portion of this dataset to evaluate methods of applying the Curve Number rainfall-runoff model for annual rangeland watersheds providing credibility to an important hydrologic modeling component common among watershed-scale water quality models.

To the best of our knowledge, these data represent the most comprehensive, watershed scale assessment of long-term $\text{NO}_3\text{-N}$ and suspended sediment loading rates for moderately grazed annual rangeland watersheds. Under the grazing levels and climatic conditions realized during the 20 yr of data collection the watershed actually served as a sink for mineral N deposited as dry and wet atmospheric deposition. While there was certainly export of $\text{NO}_3\text{-N}$ from the watershed on a daily and annual basis, these data raise the possibility that annual rangeland watersheds sequester more N than they generate.

Approximately 90% of annual rangelands in the region are privately owned and 80% are grazed by domestic livestock (Wacker and Kelly, 2004), so an important question to address is how grazing manage-

ment influences the capacity of these watersheds to retain N and suspended sediments. It is likely that many watersheds throughout the region may be grazed more heavily than the studied watershed, and certainly some watershed management proposals call for elimination of grazing by domestic livestock. These data demonstrate that moderate intensity grazing, such as that practiced within the experimental watershed, does not contribute to water quality degradation by N or sediment compared to many agricultural and range lands.

These data also demonstrate the temporal variability in loading rate at both the daily and annual scale. Fundamentally, these data illustrate that the establishment of a set daily or annual loading rate (either as a regulatory target, or as a management expectation) ignores basic hydrologic and biogeochemical processes that dictate background N and sediment flux from these watersheds. For instance, it is clear from these data that a maximum allowable loading rate and expected loading reductions should be scaled according to season. At a minimum, one could justify different loading rates for dry versus wet season (Fig. 5 and 8). These data and the existing literature (Tate et al., 1999; Holloway and Dahlgren, 2001; Jackson et al., 2006) justify application of a different loading rate expectation for early vs. late wet season $\text{NO}_3\text{-N}$. There is an inherent flushing of $\text{NO}_3\text{-N}$ from annual rangelands due to asynchrony between N availability and plant demand, and hydrologic transport events early in the wet season. Even in the absence of grazing, California oak woodland/annual grasslands are susceptible to $\text{NO}_3\text{-N}$ leaching due to this asynchrony in the N cycle. This highlights the importance of considering the natural background contributions of pollutants, such as nutrients and suspended sediments, before setting regulatory criteria.

The temporal variability in NO_3 and suspended sediment concentrations documented by this study pro-

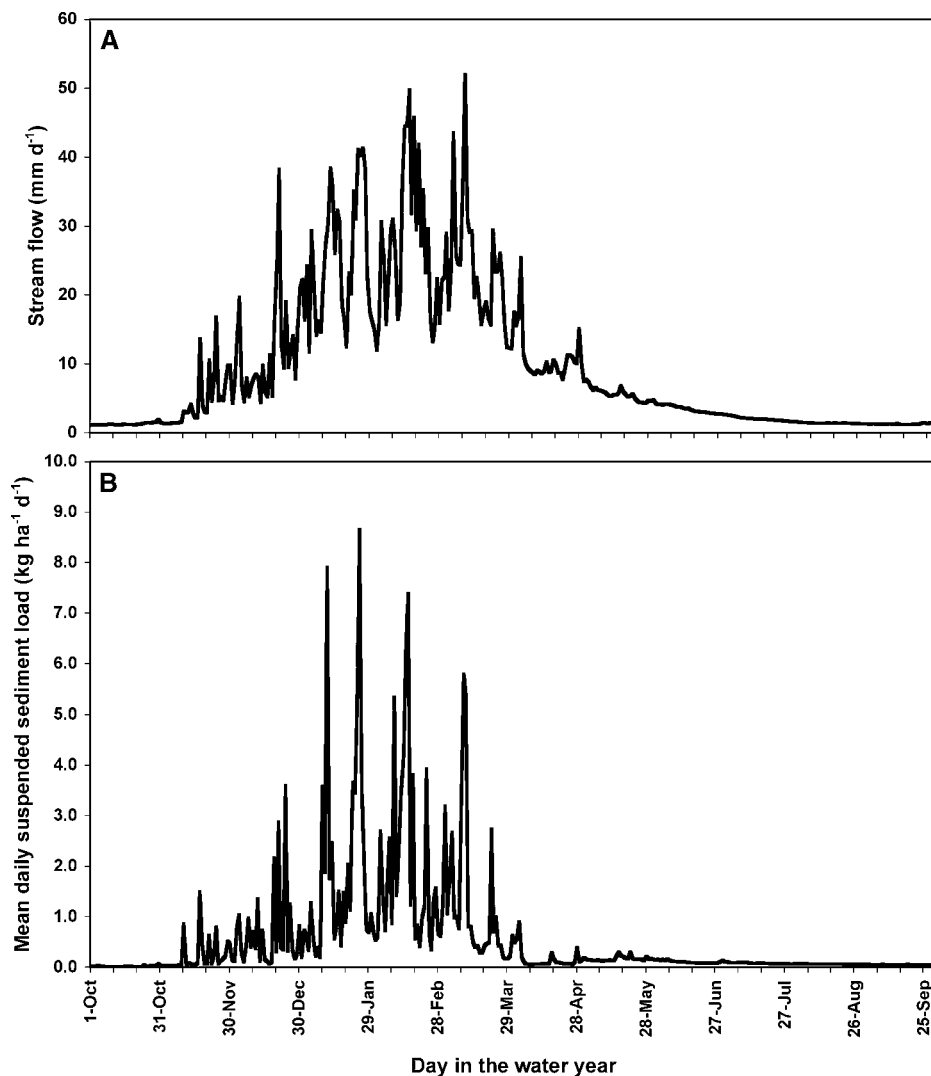


Fig. 8. (A) Mean daily stream flow for the Schubert watershed 1981 to 2000. (B) Mean daily suspended sediment load for the Schubert watershed 1981 to 2000.

vides a valuable record to assist in designing a cost-effective and accurate monitoring program. For several constituents, the majority of the annual constituent flux

is transported during a few large storm events each year. Because constituent concentrations may change appreciably during storm events, intensive sampling of

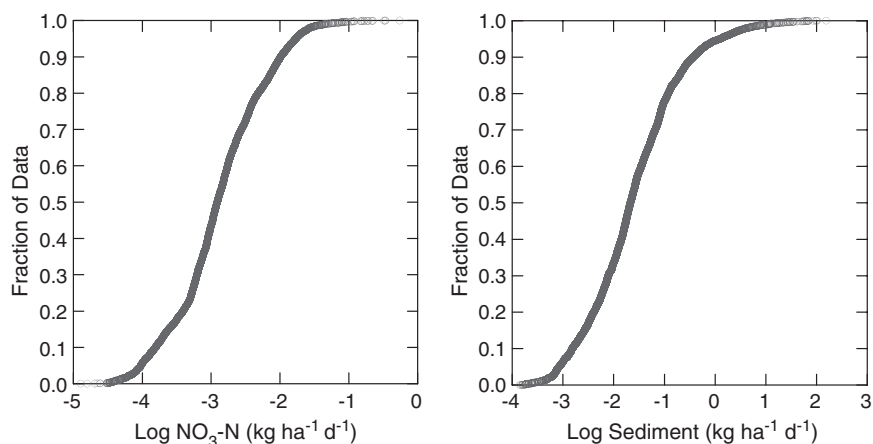


Fig. 9. Probability plots of daily nitrate-nitrogen (NO₃-N) and suspended sediment loads in stream water for the Schubert watershed.

the storm hydrograph is required to adequately characterize fluxes. In contrast, constituent concentrations during baseflow conditions typically display small changes and therefore do not need to be sampled at a high frequency.

The probability plots reported in Fig. 9 ($n = 7300$) represent a useful, risk-based approach to assigning a margin of safety to maximum allowable loadings, as well as selecting reasonable allowable loading estimates. For example, given the probability information, a watershed manager or regulator could identify a maximum allowable loading for $\text{NO}_3\text{-N}$ based on probability of occurrence rather than based on a mean or median value. Thus, a defensible risk of exceedance-based margin of safety could be selected to accommodate watershed specific risk tolerance both from a water quality and stakeholder's risk tolerance perspective.

The 20-yr record of NO_3 and sediment flux reported here illustrates the importance of long-term monitoring and data collection to facilitate management and regulatory decision-making. These data indicate that NO_3 and suspended sediment fluxes in stream water from annual rangelands are highly variable at the daily and annual time scales. Year-to-year $\text{NO}_3\text{-N}$ and suspended sediment fluxes may vary by over an order of magnitude, making long-term data records essential for evaluating watershed input-output budgets and establishment of allowable loading rates for nonpoint source constituents from rangelands.

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