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TECHNICAL COMPLETION REPORT

Project Number UCAL-WRC-: W-866
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University of California Water Resources Center

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# Abstract

Previous research has shown that vegetative buffer strips are effective at protecting water quality. Most of the research has been conducted on the east coast. Our research goals are to determine whether native perennial grasses serve to restore native biodiversity while simultaneously capturing both sediment and nutrients from adjacent conventional row-cropped agriculture. In particular, we are evaluating the efficacy of buffer strips in Mediterranean climate. Buffer strips bordering Elkhorn Slough, draining into the Monterey Bay National Marine Sanctuary, have received one of three treatments: annual non-native grasses, perennial native grasses, and an unseeded treatment of weedy volunteers. We measured sediment movement as part of an erosion study and quantifying nitrogen and phosphorus pools in soil, surface water, soil water, groundwater and vegetation. In addition, we have looked at the mechanisms and processes involved in nitrogen transformation to understand the fate of the nitrogen. Preliminary results from the erosion study indicate that the annual treatment was most effective in preventing erosion in the first year, whereas the perennial grasses were more effective in the second and third years. Results from the groundwater study indicate a significant decrease in nitrogen concentration with an increase in buffer strip length. Hydrologic tests are being analyzed to understand the pathways and dilution interactions. Nitrogen dynamics suggest that gaseous loss is an important pathway for the loss of nitrogen from the field. The relative contribution of trace gasses to the atmosphere needs further investigation.

**Project Number**: W-866 Start Date: July 1, 1996 Duration: Two Years

**Title**: Effects of Perennial Grass Buffer Strips on Movement of NPS Pollutants from Cropland to Wetland

**Investigators**: Robert Curry, Research Professor and Margaret Fitzsimmons, Professor

**Key Words**: Ground water quality, water pollution, non-point source, erosion, nitrate, phosphorus

## Problem

Water quality and aquatic environments are heavily impacted by agricultural chemicals and other non-point source pollutants (NPS). NPS pollutants are diffucult to control and to quantify because they are diffuse and often ephemeral. Many common agricultural practices result in sediment loss and fertilizer, herbicide, fungicide and pesticide runoff, which ultimately degrades water quality (Lant, et al., 1995; U.S. Department of Agriculture, 1993; Karr et al., 1978). Agriculture is the largest NPS of pollution to U.S. waterways, accounting for approximately half of all water pollution (Natural Resource Council, 1991). Studies conducted in California support this conclusion (Meyers, et al., 1985).

Understanding and restoring ecosystem function, such as nutrient cycling, in degraded areas is a challenge facing scientists today (Christensen, et al., 1996; Bradshaw, 1984). Although Tansley defined ecosystem function in 1935, the mechanisms controlling many ecosystem processes are not well understood. In ecosystems and watersheds, there is a natural balance of nutrients which are cycled through inputs, weathering, soil development, and water flow (Curry, 1977). Anthropogenic activities have significantly disturbed this balance by altering vegetative cover, accelerating erosion, and introducing pollutants. Mitigating these effects is both a biological necessity and a political priority. In addition to the adverse impacts of pollution, the loss of native species is of major concern. Thus restoration efforts seek to restore both native species and ecosystem function. Finding a simultaneous solution to these problems has been the motivation for this study.

The NPS pollutants targeted in this study, primarily nitrogen and phosphorus, derive from fertilizer inputs. We have investigated the movement of these nutrients through the soil, water, and vegetation, and investigated their fate. Nitrogen (N) as ammonium and/or nitrate and phosphate are commonly applied in fertilizers. Ammonium-N and orthophosphorus are largely insoluble and relatively immobile (Vanderdeelen, 1995). However, they adhere to soil particles and are transported with sediments, becoming particulate-born pollutants. Nitrate-N is soluble and is rapidly transported with water movement (Garrels, et al., 1973, Armstrong, et al., 1975). Both plants and microbes require nitrogen and phosphorus which they assimilate from soil or water reservoirs. When mineral nutrients stored in soil or litter are no longer demanded by plants and animals, these nutrients may be lost from the system through runoff into the watercourses (Curry, 1977). For example, in the dynamics in the Hubbard Brook study, nitrate concentrations were 41-times higher in surface runoff than the undisturbed condition in the first year and 56-times higher in the second year (Likens, et al., 1970). Nitrate has been shown to be toxic in high concentrations, and is presently regulated with an action level of 10 mg L-1. Typically, nitrate-N concentrations above 1 mg L-1 are considered to be derived from anthropogenic activities (Meybeck, et al., 1990).

In addition to nutrient loss from cropland, this study focuses on erosion. Sediment loss contributes to water quality degradation and modification of channel geomorphology, potentially filling shallow wetlands. Erosion has many important economic consequences for humans, namely removal of productive topsoil, damage to roads and fields, eutrophication, and the silting of rivers and wetlands (Dunne, 1978). In addition, the insoluble chemical contaminants bound to sediments are transported into surface waters through the process of erosion. Historic compounds are also mobilized when large gullies occur, excavating large volumes of soil and transporting them down gradient. Although DDT has been banned since the early 1970s, high concentrations of DDT remain in the upland soils surrounding the estuary. As recently as 1995, floods facilitated the transport of eroded sediments, which ultimately resulted in the crash of the caspian tern population. There were 140 nesting pairs and no hatchlings survived. Results from egg and fetus analysis attributed DDT as the primary cause. Clearly, erosion continues to be a significant threat in the watershed.

Another component of this project is looking at the species composition and nitrogen dynamics in the vegetative buffer strips, particularly in a mediteranean climate such as the central California coast. Vegetative buffer strips (VBS) successfully remove excess nitrogen from agricultural runoff in many parts of the country (Karr & Schlosser, 1978; Peterjohn & Correll, 1984; Lowrance et al., 1984; Dillha, 1989). Agricultural research and recent ecological research demonstrate that species composition can control nitrogen cycling, yet species composition has not been addressed in most buffer strip research. In areas that can support a wide range of species, species composition may determine the effectiveness of nutrient removal. We tested grass buffer strips that receive runoff from row crops along the central coast of California. By comparing three treatments, our experiment was designed to characterize the buffer nitrogen dynamics in the context of a mediterranean climate.

Much of the documented success of VBS come from the Atlantic coastal plain. Reductions in groundwater nitrate provide evidence that VBS retain or transform nitrogen thereby improving water quality. Investigators cite three processes for VBS success: Plant assimilation, microbial uptake, and denitrification. The relative importance of these processes vary spatially (among sites and within sites) and temporally (inter-annually and intra-annually). Haycock and Pimay suggested that microbial decomposition of detritus was an important sink for nitrogen during the winter months when plants were dormant. Dillaha et al. (1989) found VBS could act as a sink and a source when nutrients previously trapped were later released. These studies have been done in a temperate climate, where rainfall is evenly distributed through the year. In regions with a mediterranean climate where precipitation is restricted to the winter months, such as California, VBS nitrogen dynamics are likely to be very different.

Species composition can determine nitrogen availability. Plants vary in the quantity and form of nitrogen assimilated. Nitrogen fixation rates can vary by plant or bacteria species. Finally, plants can modulate nitrogen mineralization through their litter quality. The ability of litter quality to influence mineralization processes has received recent interest from ecologists and has been well documented in cover crop research. Plant lignin and nitrogen content and C:N ratio are factors that determine mineralization rates in agroecosystems and natural systems.

Prior to European settlement and the invasion of exotic annuals, perennial grasses were important in California grasslands. Remnant populations have been found in areas protected from disturbance, e.g. railroad right-of-ways (Clements, 1934) and farm borders (Bugg and Anderson 1992, per obs.). However, the trend has been to herbicide or disk non-cropped areas for weed control. The resulting loss of vegetative cover increases the potential for erosion and the off-farm movement of non-point source pollutants. We compared three vegetative buffer treatments: Perennial grasses, common barely (often planted along farm borders by growers), and a weedy treatment similar to the vegetation adjacent to many farms. If perennial grasses were more effective in nitrogen retention, their use as VBS could provide additional incentive for restoration activities.

## Research Objectives

Our overall objectives of this study were to 1) test if VBS are effective at uptaking and immobilzing NPS pollutants and protecting the water quality of Elkhorn Slough; 2) test if VBS will reduce erosion on steep slopes, thereby protecting the long-term sustainability of the soil resource; 3) determine the minimum size requirements for effective VBS function; 4) increase our understanding of specific mechanisms controlling nutrient cycling through time, as mediated by the VBS; 5) determine if native perennial grasses can be used for both water quality protection and restoration; 6) investigate the existing groundwater hydrology and water-borne pollutant concentrations of the study site, and hydrologic changes associated with implementation of the buffer strips; and 7) provide data and recommendations to management agencies to facilitate VBS implementation .

## Regional Location

Elkhorn Slough is the major coastal wetland and National Estuarine Research Reserve adjacent to the Monterey Bay National Marine Sanctuary. Elkhorn Slough was designated as an environmentally sensitive habitat in the 1976 California Coastal Plan, and more than 567 hectares of the estuary are in the National Estuarine Research Reserve System. Soil erosion on the steep slopes surrounding the estuary and high nutrient input from the agricultural activities were identified as the primary threats affecting water quality in Elkhorn Slough (SCS, 1984). Approximately 26% of the Elkhorn Slough watershed is in agricultural production, most on highly erodible sandy soils, with strawberry cultivation accounting for the single greatest crop in production (Mountjoy, 1996). In 1984, the former Soil Conservation Service (SCS) conducted a study which examined erosion generated from strawberry fields in the Elkhorn Slough watershed. Approximately 75% (128,900 tons/year) of the anthropogenic erosion in the watershed was attributed to strawberry production (SCS, 1984). Costs of erosion were estimated at over $3 million per year, or $791 per 0.4 hectares of strawberry land (SCS, 1984). The report recommended a number of economically viable practices for erosion control which included the use of non-native VBS. This study seeks to investigate the use of native VBS as an effective mitigation strategy for both erosion control and reducing nutrient inputs.

## Site Description

Azevedo Ranch in the Elkhorn Slough watershed, is owned jointly by The Nature Conservancy and the Monterey County Agricultural and Historical Land Conservancy. These groups acquired the land as a research and demonstration site for sustainable agricultural and land management practices (Marcus, 1991). The ranch encompasses 60 hectares, approximately 36 hectares of which are currently in strawberry or flower cultivation. The slopes where the VBS are planted comprise approximately 1.2 hectares bordering a small salt marsh with restricted tidal flushing that drains into Elkhorn Slough (Figure 1). In the past, strawberries were grown on both the slope and the terrace, extending to within several meters of the wetland boundary. As of July 1995, crop production was limited to the flat terraces, enabling the slopes to be used for investigating the efficacy of VBS at mitigating surface runoff generated on the terrace.

The surface soils are sands and sandy loams, with dune sands and beach sands underlying the surface soils in discrete layers, interspersed with clay. A relatively impermeable continuous argillic horizon exists below these layers at depths ranging from 60 cm to 360 cm below ground level, depending on slope elevation. The argillic horizon underlies the site, creating an aquiclude and a seasonally perched water table. It is unknown if the clay layer results from pedogenesis or former marine sediments. We have traced the movement of subsurface water and dissolved nutrients down-gradient along a coarse particle layer, perched on the argillic horizon, until it reaches the subsurface waters associated with the marsh ecosystem. The upper water- bearing zone is a perched water table which will be referred to in this report as groundwater.

Row crops that include strawberries and cut flowers are fertilized using a combination of methods and materials. Following plant-out or seeding, slow release NPK (nitrogen, phosphorus, and potassium) fertilizers containing ammonium and nitrate are applied in a narrow band at a shallow soil depth. Through the spring and summer production seasons, calcium nitrate and urea are routinely injected into drip irrigation systems. Preliminary studies found that ammonium and phosphate are primarily transported to the marshes adsorbed to soil particles. Nitrate is dissolved in surface runoff and subsurface water that discharges into salt marshes. Algae blooms in the salt marsh suggest a eutrophic conditions.

## Experimental Design

The plots were located on a 1.4 Ha area, adjacent and down gradient from row crops on a slope catena, at an upland-wetland zone transition. The treatment area was subdivided into nine adjacent plots. The area was topographically surveyed to create equivalent plots, with plot boundaries following the topographic fall line to minimize contamination across treatments. For the experimental plots, we used a randomized complete block design with three blocks, each consisting of three treatments. Each plot was approximately 0.12 Ha in size, 25-30 meters cross slope and 40 m down slope, although the actual slope lengths beyond the plot boundaries range from 50-80m. The plots are located down slope from strawberry (or flower) beds allowing surface and subsurface influent flow from the row-cropped beds. Each plot was randomly assigned to one of three treatments: 1) native perennial grasses (a mix of Nassella pulchra, Bromus carinatus and Deschampsia caespitosa), 2) non-native annual barley grass (Hordeum vulgare), and 3) an unseeded treatment consisting of volunteer weedy vegetation, recruited from seed rain and the remaining seeds in the soil seed bank. The perennial seeds were collected from within the region to maintain the genetic integrity of the native grasslands. The barley was selected because it is commonly used by farmers in the watershed. The unseeded plot provides an analog for common management practices if a farmer did not plant a VBS.

The installation of the VBS occurred on October 22, 1995. The plots had previously been planted in strawberry beds and was not suitable for seedling establishment. Farm roads with compacted soils and gullies occurred in several places in the study area. To increase success, we filled gullies and had the roads ripped with farm equipment. The buffer area was chisel plowed, disked, and land planed. We irrigated the plots in September to deplete the weed seed bank. After two weeks, the field was then disked to dislocate the germinating seeds. The grass seeds were hand broadcast and buried by light disking before seasonal precipitation began. All plots were irrigated in November to improve seed germination before heavy rains began. The grasses established successfully during the winter and spring of 1996. The barley was reseeded in November 1996 and 1997 as the original crop was mowed in May 1996 for pest control and June 1997 for weed control. Mowing occurred after Hordeum had set seed. The unseeded plot was herbicided in January 1997 to control for vegetative treatment differences.

## Methodology:

### *An Edaphic Approach to evaluating the efficiency of Vegetative Buffer Strips in preventing erosion*

Soil erosion was quantified using three complimentary methods: discrete point analysis with erosion pins, continuous points with an erosion meter bar, and gully analyses. Erosion pins were used to measure changes in surface micro-topography (Dunne and Leopold, 1978; Lehre, 1982; Goudie, 1981; Griggs, 1988). 1.2 cm diameter steel reinforcement bars were installed in transects along the slope (Figure 2a). Six to nine pins, spaced 3.5 m apart, compose one transect. There are five transects per plot at distances of 5 m, 10m, 20m, and 40m from the top of the slope, and at the base of the slope.

The pins were surveyed at the time of installation and measured for a baseline height above the ground (from the top of the pin to the surface of the soil). A steel washer was placed on each pin to distinguish between the processes of erosion and deposition (Dunne and Leopold, 1978). When erosion occurs, the washer falls to become flush with the new surface elevation. Net deposition is determined by measuring the height of soil above the washer. When washers are buried, the washer can no longer move and only net change can be determined. It was determined after the first year of data collection that the presence of the rebar was affecting the hydrologic pathways and causing preferential erosion or deposition. Therefore, a new method was developed.

To supplement the discrete point data, a surface topographic analyzing rod (STAR) was designed to provide a more continuous data set and capture changes in topography resulting from rills and small gullies. This instrument design is a modified combination of a point contact bar and erosion frame or bedsteads (Goudie, 1987, Foster et al., 1991, Leonard and Clark, 1993). The instrument was designed to fit over the erosion pin on one end with an adjustable leg on the other side. A compass and level are attached to provide consistent and repeatable locations for precise determination of surface topographic changes over time (Figure 2b). A nail was installed in the soil where the base of the adjustable leg rests to ensure precise replication in the future. Holes are spaced every ten cm on the meter bar and a wooden dowel with metric gradations is used to measure surface height over a one meter distance to the south/southwest of each erosion pin. All 315 pins are surveyed annually.

In addition to sheet and rill erosion, gullies can remove large volumes of soil. While the STAR provides data on the surface topographic changes, the amount of soil lost through gullies is not captured with this method. All gullies are measured each year to determine maximum soil volume displaced, using a cross section analysis (SCS, 1984; Swanson, 1989; Heede, 1976). Width and depth of each gully is measured every meter beginning at the upslope end of the gully down to its base. Soil volume is calculated and correlated to vegetative treatments and specific soil parameters. Furthermore, each gully is reanalyzed each year to evaluate trapping effciency of the VBS.

## *A Hydrological Approach to evaluating the efficiency of Vegetative Buffer Strips*

### *Method for Ground Water collection and Measurements*

Piezometers are vertical standpipes used to measure total head at a point in the aquifer or perched water table (Dunne and Leopold, 1978, Topalidis, 1983). I have installed 27 five-cm diameter piezometers to monitor the groundwater throughout the study area. Three wells were placed in each of the treatment plots, at the top, middle and bottom of the slope, corresponding to the 5 m, 20 m and 40 m buffer strip lengths. In addition, nine wells were installed in the row-crops above each plot to serve as controls. These nine wells were only available intermittently, depending on the agricultural activities. The monitoring well placement will allow comparison of groundwater qualityto be detected both from slope top to bottom and between treatments. Monitoring wells were hand augered using a 7.6-cm diameter bucket auger. Boring holes were advanced until the argillic horizon was encountered, and completed at a depth of 15 cm into that clay layer. The piezometers were constructed using five-cm diameter 0.01- inch machine slotted schedule 40 PVC piping (Harlan, et al., 1989). The well casing was surrounded with a 2/16 sand pack and the top 15 cm were sealed with a water-tight bentonite slurry, in accordance with EPA standards (Barcelona, et al., 1983). The piezometers were screened at depths sufficient to accommodate groundwater fluctuations in both drought conditions and high precipitation years.

Four data loggers with pressure transducers will be used to monitor groundwater levels continuously in a diamond shape configuration placed in one treatment block (see

Figure 3). These data will enable determination of downslope (and any cross slope) movement of water. Depth to water in all wells were also monitored manually with an electronic water level sounder to determine the elevation of the water table relative to mean sea level. All wells were surveyed and converted to NGVD, relative to mean sea level. The water level surveys were conducted on a weekly sampling regime through the duration of the rainy season, and continued until the wells were dry. All water level surveys were conducted prior to disturbing the water table. A series of slug tests were conducted in the spring of 1998 to estimate hydraulic conductivity (Freeze and Cherry, 1979; Oberdorfer, et al., 1990) and to provide insight into the rate of nutrient movement.

Groundwater samples were collected on a weekly basis, to minimize the potential for missing high nutrient leaching events. Surficial aquifers are the most susceptible to rapid and sometimes dramatic changes in quality, often related to human-induced pollution (Pettyjohn, 1982). Prior to collecting groundwater samples, standing water in the well was purged by hand, using a PVC or teflon bailer. Groundwater samples were collected from the monitoring wells after four casing volumes had been evacuated. If the well was purged until dry, the well was sampled after 80% recovery of its static water level. Water samples were collected in plastic bottles, stored on ice, and transported back to the laboratory for analysis. Groundwater samples were analyzed for nitrate, ammonium and phosphate concentrations using ion chromatography.

### *Method for Surface Water collection and Measurements*

I have monitored the water quality of the surface runoff during storm events by collecting grab samples as possible from overland flow. I measured the nutrient concentrations into, in the middle, and exiting the plots. Grab samples of surface runoff were collected during storm events when the soil conditions were saturated and antecedent rainfall created conditions which facilitated overland flow.

Due to soil and hydrologic flow path heterogeneity, it was difficult to collect influent and effluent samples from each of the nine plots using this method. Therefore, a second method was designed to collect composite samples. A surface collector was installed at the top and bottom of each plot to intercept overland flow and capture runoff. The surface collectors consist of two plastic 480 ml Odwalla juice containers connected together at the lip of the bottles. The lower bottle was cut around the rim to enable a ping pong ball to be installed inside. It was then sealed with duct tape. A 6 cm X 9 cm slit was cut into one of the four sides of the upper bottle. The collector was then buried so that the open slit faced in the upgradient direction and was flush with the soil profile. Therefore the bottom half with the ping pong ball was buried beneath the surface. This design allowed overland flow to enter the unit over time, and when the bottle was full, the ping pong ball would float until it obstructed the entrance and prevented loss of water or continued collection. In addition, the unit was contained except for the slit, preventing rain from diluting the sample, or soil water to overflow the sides and dilute the sample. A vacuum pump was used to empty the containers after storm events. Water samples and sediment load were analyzed as described earlier.

Changes in water quality both vertically along the slope and in each treatment across the slope as a result of the VBS treatments will be determined by chemical analysis. Surface flow will be quantified (see 2C) to correct for dilution effects. A Dionex Ion Chromatographer will be used to determine the concentrations of the following anions: nitrate, ammonium and phosphate.

As water movement governs the transport and concentrations of the NPS pollutants, the overland flow entering each plot will be quantified. These data will be used in conjuction with the nutrient and sediment capture data. As the topography and land use above plots varies from plastic covered strawberry beds to row crops to heavily compacted farm roads, the volume of runoff varies significantly. Overland flow was determined volumetrically (Heede, 1987) during a synoptic sampling event (Coyne, *et al.*, 1995; Curry, pers comm). A total of 40-60 volunteers were spaced first at the top and then bottom of each plot. Five to seven volunteers per plot intercepted overland flow entering and exiting plots for two minute intervals during a rain storm. This exercise was repeated three times. The water was composited in a five gallon bucket. Total volume was determined in the field with graduated cylinders. The water was agitated and one liter was subsampled randomly. The subsample was brought back to the laboratory for chemical analyses of surface water quality and quantification of the sediment load suspended in the water (Coyne, *et al.*, 1995; Heede, 1987, Stevens, 1992). Sediment load was quantified with vacuum filtration. The overland flow was quantified annually to measure changes in runoff with land use activities, as the grower rotated crops and grew both strawberries and flowers during this three year study.

### *A Biological Approach to evaluating the efficiency of VBS*

#### *Method for vegetation sampling*

Within each of the 5 m, 10 m, 20 m, and 40 m buffer lengths a 30 $cm^2$ quadrat will be tossed randomly three times. Percent cover of both the target species and other species will be estimated visually twice per year on a ranking scale ranging from <1 to 100% (Dethier, et al. 1993). The above-ground biomass within the quadrat will be destructively sampled, dried for 24 hours at 100 C, and weighed to 0.01 grams (Tillman, et. al., 1996). The dried plant tissue was ground and analyzed to determine total nitrogen. This allowed comparison of the grasses ability to capture excess nitrogen. Quantifying the nitrogen flow will confirm whether the VBS function as net sinks or sources of nutrients. This sampling occurred annually, at the end of the growing season. Results from the first sampling event indicated that *Hordeum* established a dense cover quickly (80-90%) while the perennials had percent covers ranging from 10-90%.

### *Evaluating the role of root systems at accessing dissolved nutrients and uptaking nutrients beneath the VBS*

Roots were sampled in July, 1997 quantitatively for root biomass. Soil samples were collected from at the top, middle and bottom of each plot, at the end of each growing season. The sampling was stratified into three target depths. A bucket auger was used to collect samples from the top depth of 1-15 cm bgl. A slide hammer was used to collect samples from the depths of 60-75 cm bgl and 120-135 cm bgl. This stratification was designed to distinguish the root depths from annual plants compared to the perennial treatments. The soil samples were presoaked in a dispersing solution (sodium metaphosphate) to allow separation from the roots. A screen and tweezers were then used to capture all the roots within the samples. (Bohm, 1979; Boyer, unpublished data 1996). Roots will be dried for 24 hours at 60 C and weighed for total biomass on a scale to 0.0001 grams.

In addition, roots were quantified according to presence or absence in 1996 and 1998. This method determined the rooting depth to allow comparison between the treatments and facilitate in understanding the groundwater chemistry. Using a hand held auger, holes were advanced and each sample was examined for root presence. The holes were contined until no roots were found in the sample. Three holes per plot were advanced, at the top, middle and bottom of each plot. This work will be completed during August, 1998.

### *Evaluating the nitrogen dynamics in vegetative buffer strips*

Soil cores were collected to 15 cm in depth. Five soils samples (pooled from approximately 10 sub-samples and mixed) were collected from each plot. The samples were collected along the topographic contours, 0 meters, 5 meters, 10 meters, 20 meters, and 40 meters, down slope from the farm road that separates the row crops from the VBS. Soils were also sampled in the row crops, approximately 10M up slope from the road. We increased the sampling frequency during the fall and winter months to improve resolution of the seasonal changes on soil nitrogen. The soils were extracted with 2 M $K_2SO_4$ in a 2.5:1 extracting soil ratio. Soil moisture was determined gravimetrically, drying the soil for 24 hours at 100°C. Bulk density was measured with intact cores from 27 locations, four times over the study period, and were averaged for each plot and slope location. Nitrate and Ammonium were determined colorimetrically using a Lachat flow through injection analyzer and are reported in areal units to a depth of 15 cm.

36 porous cup lysimeters were installed 60 cm below the soil surface. Four were installed at the bottom of each buffer plot. Several months after installation and a pretreatment, 40 kPa vacuum was applied to the lysimeters every week and water samples were removed the following week. Water samples were put on ice and analyzed

immediately or were stored frozen until they could be analyzed. A Dionex ion chromatograph was used to quantify the concentrations for the following anions: chloride, nitrite, bromide, nitrate, and sulfate. Phosphate was rarely above the detection limit set at a sensitivity of 100 microsiemens and can not be reported.

Species composition in terms of percent cover was measured using 0.25 meter quadrats in a stratified random sample design. The plots were stratified along the slope into 30 $m^2$ sub-units and 15 random locations for the quadrat was located plot subunit. Percent cover was measured in February 1996, February 1997, May 1997, July 1997, March 1998, and May 1998. Percent cover is reported as an average within each year. Plant biomass was collected by destructive harvesting within 0.25 $m^2$ quadrats. The quadrats were randomly located in each plot along three slope locations, 5 meters, 20 meters, 40 meters from the farm road. At each slope locations, three samples were taken. Harvested plants were separated into two groups, planted species (Hordeum sp. or perennial species) and weedy (unplanted, volunteer species). The plants were dried at $60^{\circ}$ c for 24 hours and weighed. Then plants were ground to 0.5 mm using a Wiley mill. TKN digestion was used to determine the nitrogen content.

Concurrent research on sediment capture in the VBS prevented whole plant harvesting and direct estimates of nitrogen translocation between above and below ground biomass in perennial vegetation. Therefore we assumed that the nitrogen left in the above ground biomass during the summer dormancy approximated the net annual gain of nitrogen in the perennial grasses.

### *Statistical Analysis*

Soil and soil water inorganic ions distributions had long tails and were log transformed. All data were analyzed using a general linear model with SAS software (proc GLM). Using a complete random block design, treatment was the only factor that could be considered as an independent and randomized variable. To analyze the overall treatment affect by date, spatial data were considered sub-samples and averaged to avoid pseudo-replication. Thus, treatments were evaluated on each date independently. Soil water anions were also analyzed using a general linear model using a repeated measured analysis. In all cases, probabilities less than 0.1 are reported but are not considered statistically significant unless probabilities were less than 0.05.

## Principal Findings and Significance:

### *Results of species composition and nitrogen dynamics*

Seed germination rates were high immediately after seed burial. The annual treatment grew fastest through the late fall. By the time the first rain storms arrived, the annual treatment had little exposed ground cover and the culms were 30-40 cm in height. The weedy treatment and perennial treatment had much higher amounts of exposed ground and the height of the plants were in most cases less than 10 cm. However, by February 1996, there were not treatment differences in the percent of exposed ground (Table 1).

The treatments were significantly different in percent cover of annual and perennial vegetation. In year one and two the annual treatments consisted of greater than 65% annual grass species, significantly more than the other two treatments. In the third year, annual forbs become more important with 47% cover while annual grasses were only 30%. These changes reflect the relative importance of Hordeum vulgare which had a cover of 64%, 55%, and 2.2% in the first, second, and third year respectively. Lolium latifolia with (20% cover) increased its importance each year as Hordeum declined. The perennial treatment had only 37% cover of perennials in the first year but then jumped to 77% in the second year and dropped to 40% in the third year. Each year the perennial treatment had significant higher cover of perennial grasses, dominated by Bromus carinatus. In the weedy treatment, annual forbs were able to become established early. The forbs Erodium botrys (17%), Trifolium hirtum (10%), and Sonchus oleraceious (7%) recruited from the seed bank. Also, Poa annua and Convolvus arvensis (10%) were important plants in the first spring but in the second year were rarely detected. Medicago had become important in the second and third year of the study, with 17% and 32% respectively. From the start of the experiment, nitrogen fixers especially Medicago polymorpha and Trifolium hirtum, become increasingly important. They were present in all treatments and by the third year, both the annual and perennial treatment had 20% N-fixer cover, while the weedy had 34% N-fixers, although there was no statistically significant difference detected.

The nitrogen content of above ground Hordeum vulgare was significantly lower than the perennial or other species. The perennial grasses contained 2.4 % nitrogen while the annual contained 1.4% (Table 2).

In soils sampled immediately after sowing seeds, soil nitrate concentrations were high, approximately 15 ppm across the entire site. After the rainy season began soil nitrate concentration declined until the rains ceased (Figure 2). This general pattern repeated each year of the study in all the treatments. In the second year of the study, the soil nitrate concentrations diverged due to a treatment affect. During the fall, the weedy and annual treatment had the highest soil nitrate concentrations. After winter rains began, the annual treatment had lower soil nitrate. Table 5 shows the average soil nitrate concentrations during late fall and early winter from the second and third year of the study.

In general, soil ammonium concentrations followed the same seasonal pattern as soil nitrate concentrations, with a higher degree of variability and no significant treatment differences (Figure 3). The noise associated with soil ammonium does not allow an interpretation that suggests any treatment difference.

There was a weak relationship between soil nitrate and soil water nitrate concentrations,

$$[\text{Soil water NO3-N}] = 4.3 + 3.7 * [\text{Soil NO3-N}]$$

where 27% ($p<0.0001$) of the variation in soil water nitrate concentration was explained by soil nitrate concentrations (Figure 4). A five day lag between the soil sample and lysimeter samples maximized the regression coefficient.

There were no significant treatment effects for chloride, bromide, sulphate using a repeated measures analysis. However, there were significant differences in nitrate-N concentrations through the second and third year (year 2: $p=0.0033$, year 3: $p=0.00174$). The soil water nitrate concentrations were highest each fall (Figure 5). In the fall of 1996, the weedy treatment had nitrate levels in excess of 50 ppm-N, while the annual and perennial treatments had concentrations approximately 10 ppm-N. As winter progressed nitrate concentrations declined. The annual treatment was significantly lower ($p<0.05$).

The nitrate concentration for the weedy treatment steadily declined during the spring until most of the lysimeters became dry. The perennial treatment had a noticeable increase in soil nitrate concentrations through the end of the spring. In the fall of 1997 all the treatments had initially high nitrate concentrations that declined quickly as the winter progressed. As in the previous year, the weedy treatment had the highest nitrate levels through the early spring, while the annual treatment had the lowest. The differences were not statistically significant in the third year.

The nitrate:chloride ratio in the lysimeter water has the same pattern as the soil water nitrate data (Figure 3). In the fall of 1996, the weedy treatment had the highest nitrate-N:chloride but steadily declined through the spring until it approached the same concentration as annual treatment in late spring. The annual treatment had the lowest nitrate-N:chloride concentrations in both 1996-7 and 1997-8. The perennial treatment had an intermediate nitrate-N:chloride ratio until the spring when its nitrate-N:chloride ratio increased to the highest among the treatments.

There were lower mineralization rates in the annual treatment (p=.067) in 1997. Since the results were not strongly significant, we measured nitrification potentials the following year. However, these measures did not find any treatment differences (Table 3). Figure 6 shows the sample locations for all soil and water collection sites.

Microbial biomass-N tended to be higher in the annual treatment in the winter of the second year, but the difference were not significant when comparing the three treatments (Table 4).

### *Discussion of species composition and nitrogen dynamics*

There were strong seasonal trends in soil and soil water nitrate. The oscillation between a source and sink of nitrogen follows patterns of nitrogen availability found in other grassland studies in California (Jackson et al 1984). Furthermore, as suggested in the other research, nitrogen availability can vary within very short time periods. For example, in the third year; nitrate and ammonium concentrations spiked within a two-week period.

Species composition in vegetative buffers played an important role in the nitrogen dynamics in VBS. After the initial spike in soils inorganic nitrogen pools and the soil water, there is evidence of a high capacity in the annual treatment to assimilate nitrogen, which is more pronounced in the nitrate-N:chloride ratio in the soil water. The nitrogen sink in the annual treatments followed the pattern in soil microbial biomass-N in both the second and third years. This suggests that the treatment effect by the annual vegetation was not based on the nitrogen uptake of the current years growth but by the above and below ground litter, as it provided substrate for microbial activity. Another piece of evidence to support this conclusion is based on the different plant nitrogen content, which showed much lower nitrogen for the barley than for all the other plants sampled. As seen in many studies, the nitrogen content of plants can play an important role in the decomposition process and nitrogen dynamics.

Another interesting pattern was found in the perennial treatment each spring. Although nitrogen content did not change much in the perennial treatments in the soil, the soil water seemed to increase in leached nitrate. This may be due to a change in mineralization, immobilization, or assimilation rates. If mineralization or immobilization rates had changed, the leaky pattern should be seen in the soil as well as the soil water. However, the nitrogen increases were limited to the soil water suggesting that there is a change in the lower part of the soil profile. We believe there was an allocation shift in the perennial vegetation preparing for summer dormancy. It appeared that the perennial plants stopped assimilating soil nitrogen and began translocating above ground nitrogen to roots earlier than we predicted. These allocation patterns need to be better defined, because it may be unique to Bromus carinatus, which dominated the perennial grasses in our site.

### ***Results of the Erosion study***

Results of the erosion pin method did not differ in either year at any slope position ( See Figure 7: $p>0.05$ in all cases using a repeated measures one-way ANOVA). It was determined that the presence of the rebar affected the erosional and hydrologic pathways. Therefore, a new instrument called the STAR was developed to increase the accuracy of the measurement, as discussed in the methods section. The STAR data is presently being analyzed.

Gully development within the annual treatment was significantly lower than the both the perennial and unseeded treatments in the first year ( See Figure 8: $p<0.05$ using a one-way ANOVA and a Bonferoni multiple comparison). However by year two, no new gullies developed in any treatment.

Sediment deposition at the upper boundary of the plot (0 meter buffer length) in the perennial treatment was significantly higher than in both the annual and unseeded treatments in the second year ( See Figure 9: $p<0.05$ using a one-way ANOVA and a Bonferoni multiple comparison).

Vegetative cover in the first year ranged from 80-97% in the annual treatment, 10-80% in the perennials and 10-80% in the unseeded plots. In the second year, vegetative cover ranged from 60-90% in the annuals treatment, 90-100% in the perennials and 60-95% in the unseeded treatments.

### *Discussion of Erosion Results*

The annual treatment established a dense cover in the first year early in the rainy season, while the perennials were slower to establish. The lack of plant cover increased the susceptibility of the perennial and unseeded treatments to erosion, as shown by the significant differences in gully development. The annual treatment had sediment deposition at buffer lengths of 5 meter, 10 meter, and 20 meter, while both the perennial and unseeded treatments had erosion at those buffer lengths; however these differences were not statistically significant due to high variance.

By year two, the perennials provided a dense cover, while the annuals did not agressively reseed. Even with reseeding, the annuals did not establish as well. The unseeded treatments had a cover of weedy species at a similar density to the annual treatment. Therefore, the perennial treatment was most effective at immobilizing sediments within the first several meters of the buffer. However, these differences were only statistically significant at the 0 meter buffer length ( the upper boundary of the plot), due to high variance.

The high variance was determined to result from the discrete point method of data collection. The STAR was designed to provide a continuous data set over an area of one meter, enabling a more representative assessment of the changes in surface topography, including both rills and gullies. Data collection for these data sets will be completed in August of 1998. The STAR data has not been analyzed, and is therefore not included in this report.

Based on the preliminary results of this study, additional research was necessary to make management decisions regarding species composition and erosion control. Therefore, a collaboration was initiated between the Natural Resources Conservation Service, the Elkhorn Slough National Estuarine Research Reserve, and myself to implement a new experiment testing a mixture of annuals and perennials. This new experiment was planted in September of 1997 and consists of seven treatments, with four replicates of each treatment. The treatments include seeding densities of 50% annual and 50% perennial plots, 75% annual and 25% perennial plots and 100% each plots. Vegetative cover will be monitored over two years to determine if the annuals will be effective at holding the soil in the first year and allowing the perennials to succeed in the following years, to maximize both short and long term erosion control. Data will be collected at this new site through June of 1999.

*Results and Discussion from the Groundwater study*

The groundwater table ranges from 18 feet below ground level (bgl) on the agricultural terrace, to 12 feet bgl at the top of the slope, 8 feet bgl mid-slope and 3 feet bgl at the bottom of the slope. Due to the continuous argillic horizon, the perched water table creates an upper water-bearing zone. In 1996-1997, the rainy season was short but intense. The majority of the rainfall occurred over a period of six weeks. The elevation of the upper water-bearing zone is directly correlated to soil texture and precipitation. As a result of the short rainy season, the water table elevation was insufficient to collect water samples from all wells in the study. However, there was sufficient data to determine several trends. In 1996-1997, there was no significant treatment effect in groundwater quality. However, there was a significant effect of slope position in groundwater quality. The concentration of nitrate-nitrogen in groundwater samples decreased from the top(a) of the slope to the bottom (c)of the slope . Figures 10,11, and 12 show the groundwater nitrate concentration in the perennial, annual and unseeded treatments, by slope position, respectively, in the upper water-bearing zone beneath the vegetative buffer strips.

In 1997-1998, there was an extended rainy season due to the effects of El Nino, enabling a more continuous and representative data set to be collected. These results have not been statistically analyzed yet (see Figures 13, 14 and 15 for groundwater nitrate concentration in the perennial, annual and unseeded treatments, respectively). In addition, Figures 16 and 17 show the groundwater phosphorus concentration in the perennial and annual treatments, respectively, at each slope position.

There was a trend showing a decrease in nutrient concentration from the top of the slope (a) to the bottom of the slope (c). While the chemical analysis is completed and presented here, this project will continue through June of 1999. Statistical analysis is pending. In addition, several hydrologic tests were conducted during the 1997-1998 rainy season to understand the hydrologic pathways and groundwater quality data. Tests were conducted on infiltration, hydraulic conductivity, overland flowpaths and flowrates, and groundwater elevation monitoring. These data will be used in conjunction with the groundwater nutrient analyses to characterize the rate of migration of the target compounds. Dilution and soil residence needs to be evaluated before it can be determined conclusively if the buffer strips are effective at removing nutrients and whether the species composition of the buffer strips affects the efficacy of the buffer strips. Funding for this work comes from a different source.

## Overall Significance

Agroecosystems contribute approximately half of the nitrogen that leads to elevated nitrogen concentrations in surface water in the U.S. Nitrogen movement from diffuse sources has created public health hazards a nd adverse ecological impacts. The Federal Water Pollution Control Act (1972) was enacted to improve water quality and was successful in reducing point sources of pollution. However, diffuse sources could not be regulated by the command and control approach used in point source reductions, and States had little incentive to develop alternative regulations. The Coastal Zone Management Act Reauthorization Amendment (1989) outlined the problem of diffuse sources, also known as nonpoint source pollution (NPS), and mandated states to development management strategies to reduce nonpoint source impacts on coastal waters. As a result, the States have explored two strategies: Source reduction and transport reduction.

Source reduction is the most robust and effective approach to reduce nonpoint source pollution. Furthermore, it can be implemented easily when economically viable alternatives exist. For example, the banning of phosphate detergents in the Chesapeake Bay watershed has successfully improved water quality. The reduction of fertilizer use would limit nutrient runoff from cropped fields, however, this approach has a severe political obstacle: perceived loss to farm incomes. As an alternative, transport reduction techniques have become popular, since they do not regulate farm management *per se* but the processes that allow pollution to leave the farm field. Vegetative buffer strips have demonstrated the ability to capture pollutants as they leave the farm field and before surface waters are contaminated.

Based on this research two issues arise in applying VBS for nitrogen retention in California: First, species composition does influence nitrogen dynamics. Therefore, farm border must be managed properly to limit the invasion of weedy species, i.e. legumes, from invading. This management requirement causes additional burden to farmers and may severely limit adoption rates. Second, it appears that vegetative buffers do not protect soil water from elevated nitrogen levels. This conclusion suggests that VBS can not provide the protection that they do along the east coast context. Therefore, we recommend that future research projects focus on farm nitrogen management to protect water quality from nitrogen contamination in California.

The buffer strips are effective at reducing soil erosion on steep slopes, in addition to trapping sediments eroding off flat terraces or steep slopes up gradient from the buffer strips. Thus, as an erosion control strategy, the buffer strips are a good management practice. In terms of the ability of the buffer strips to remove excess nutrients from the surface or groundwater, additional information is necessary. As stated above, hydrologic tests were conducted and these data are presently under analysis. Dilution, hydrologic flowpaths, and soil residence needs to be evaluated before it can be determined conclusively if the buffer strips are effective at removing nutrients and whether the species composition of the buffer strips affects the efficacy of the buffer strips. Funding for this work comes from a different source.

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Table 1. Percent cover of plant functional groups and selected species. Differing letters signify significant differences between treatments (p<0.05).

| | 1996 | | | 1997 | | | 1998 | | |
|---|---|---|---|---|---|---|---|---|---|
| | Treatment | | | Treatment | | | Treatment | | |
| | Annual | Perennial | Weedy | Annual | Perennial | Weedy | Annual | Perennial | Weedy |
| **Annual Grasses** | 65.8a | 4.4b | 12.5b | 63.2a | 10.1b | 29.5b | 30.2a | 19.2b | [illegible] |
| Avena barbata | 0 | 0 | 0 | 3.7 | 0 | 0.6 | 6.5 | 0.08 | 0 |
| Hordeum vulgare | 64.4 | 0 | 0 | 54.6 | 0 | 6.2 | 2.2 | 0 | 0.6 |
| Lolium latifolia | 0 | 0 | 0 | 2.9 | 8.1 | 11.6 | 20 | 18.7 | 13.3 |
| Poa annua | 1.1 | 4.4 | 12.3 | 0.4 | 0.2 | 1.4 | 0 | 0 | 0.1 |
| Polypogon monspeliensis | 0 | 0 | 0 | 3.5 | 1.6 | 7.8 | 1.5 | 0.4 | 4.2 |
| **Perennial Grasses** | [illegible] | 37.7a | 0b | 0.8b | 76.8a | 3.2b | 0.6b | 40.8a | 4.6b |
| Bromus carinatus | 0 | 37.7 | 0 | 0.7 | 74.8 | 3.1 | 0.7 | 37 | 4.6 |
| Deschampsia ceaspitosa | 0 | 0 | 0 | 0.1 | 1.8 | 0.02 | 0 | 1.5 | 0 |
| Nassella pulchra | 0 | 0 | 0 | 0.02 | 0.09 | 0.04 | 0 | 2.2 | 0.01 |
| **Annual Forbs** | [illegible] | 20.4ab | 37.4a | 20.9b | [illegible] | 57.4a | 47.5ab | 30.8b | 66.6a |
| Erodium botyrs | 1.6 | 11 | 17.3 | 1.4 | 0.6 | 18.7 | 4.8 | 1.4 | 16.4 |
| Geranium dissectum | 0 | 0 | 0 | 1.1 | 0.2 | 2.2 | 7.8 | 1.1 | 3.4 |
| Medicgo polymorpha | 0 | 0 | 0 | 9.5 | 0.01 | 27.4 | 13.8 | 16.7 | 31.5 |
| Picris echiodes | 0 | 0 | 0 | 3.5 | 0.1 | 1.5 | 12.9 | 2.1 | 4.2 |
| Sonchus oleraceous | 0.9 | 0.8 | 7.4 | 0.9 | 0.02 | 0.6 | 4.4 | 5.5 | 7.4 |
| Trifolium hirtum | 3.4 | 4.8 | 10.4 | 1.3 | 0.004 | 2.5 | 1.1 | 2.2 | 0.6 |
| Vicia sativa | 0.5 | 0 | 0.1 | 1.5 | 0.02 | 0.2 | 3.7 | 0.3 | 2.1 |
| **Perennial Forbs** | [illegible] | 6.0a | 10.0b | [illegible] | [illegible] | 0.6b | 8.5a | [illegible] | 2.0a |
| Convovuls arvensis | 0.1 | 6 | 9.9 | 0 | 0 | 0 | 1.5 | 0 | 0 |
| Epilobium sp. | 0 | 0 | 0 | 2.4 | 0.03 | 0.5 | 7 | 0.4 | 0.8 |
| **N-fixers** | [illegible] | 5.3a | 11.0a | [illegible] | [illegible] | [illegible] | 19.0a | 19.2a | 34.2a |
| **Shrubs/trees** | [illegible] | 0a | 0a | 0.1a | 0a | 0.2a | 0a | 0a | 0a |
| **Bare ground** | [illegible] | [illegible] | 41.2a | [illegible] | [illegible] | 2.0a | 3.0a | [illegible] | 0.8a |
| **Detritus** | [illegible] | 0a | 0a | [illegible] | [illegible] | 1.9a | [illegible] | [illegible] | [illegible] |

Table 2. Above ground nitrogen content. Differing letters signify significant differences between treatments (p<0.05).

| Vegetation Analyzed | Nitrogen Content (w/w) |
|---|---|
| Native Perennial (mixed spp.) | 0.0279 (0.217) |
| Hordeum vulgare | 0.0143 (0.429) |
| Other species | 0.0261 (0.167) |

Table 3. Mean Gross mineralization rates with lower 95% confidence limits (LCLM) and upper 95% confidence limits (UCLM). Means with the same letters are not significantly different ($p<0.05$).

| | Treatment | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Annual | | | Perennial | | | Weedy | | |
| Date | mean | LCLM | UCLM | mean | LCLM | UCLM | mean | LCLM | UCLM |
| 10/6/97 | 692a | -1017 | 2401 | 1136a | 64.9 | 2207 | 1570a | 253 | 2888 |
| 11/12/97 | 1162a | -532 | 2857 | 885a | -433 | 2204 | 1245a | 256 | 2234 |
| 12/15/97 | 332a | -52.1 | 716 | 1394a | -499 | 3289 | 1033a | -931 | 2998 |

Table 4. Microbial Biomass Nitrogen with lower 95% confidence limits (LCLM) and upper 95% confidence limits (UCLM). Means with the same letters are not significantly different ($p<0.05$).

| | | Annual Treatment | | | Perennial Treatment | | | Weedy Treatment | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Method | Date | mean | LCLM | UCLM | mean | LCLM | UCLM | mean | LCLM | UCLM |
| F.I | 10/6/97 | 2.74a | 1.22 | 4.26 | 2.84a | 0.88 | 4.8 | 0.62b | 0.41 | 1.19 |
| F.I | 11/12/97 | 2.86a | -0.73 | 6.45 | 2.93a | 0.64 | 5.21 | 1.34a | -2.82 | 5.5 |
| F.I | 12/15/97 | 7.72a | 4.79 | 10.64 | 3.93a | -1.82 | 9.68 | 5.90a | 0.65 | 11.16 |
| F.E. | 2/21/97 | 3.62a | 2.98 | 4.27 | 2.26a | 0.15 | 4.38 | 2.41a | -0.08 | 4.89 |
| F.E. | 10/24/97 | 2.41a | -0.43 | 5.26 | 2.75a | 1.57 | 3.93 | 2.37a | 0.23 | 4.51 |
| F.E. | 11/10/97 | 2.08a | -1.53 | 5.69 | 1.94a | -3.05 | 6.93 | 3.16a | -4.32 | 10.63 |
| F.E. | 12/19/97 | 6.52a | 3.06 | 9.98 | 7.36a | 0.77 | 13.95 | 6.36a | -2.48 | 15.2 |
| F.E. | 2/10/98 | 8.48a | -0.14 | 17.09 | 8.23a | -0.41 | 16.88 | 5.81a | 1.35 | 10.26 |
| F.E. | 4/23/98 | 2.36a | -9.68 | 14.4 | 3.43a | 0.92 | 5.93 | 2.98a | -16.74 | 22.71 |
| F.E. | 6/11/98 | 3.36a | -11.51 | 18.23 | 2.64a | -8.71 | 14 | 3.14a | -2.36 | 8.65 |

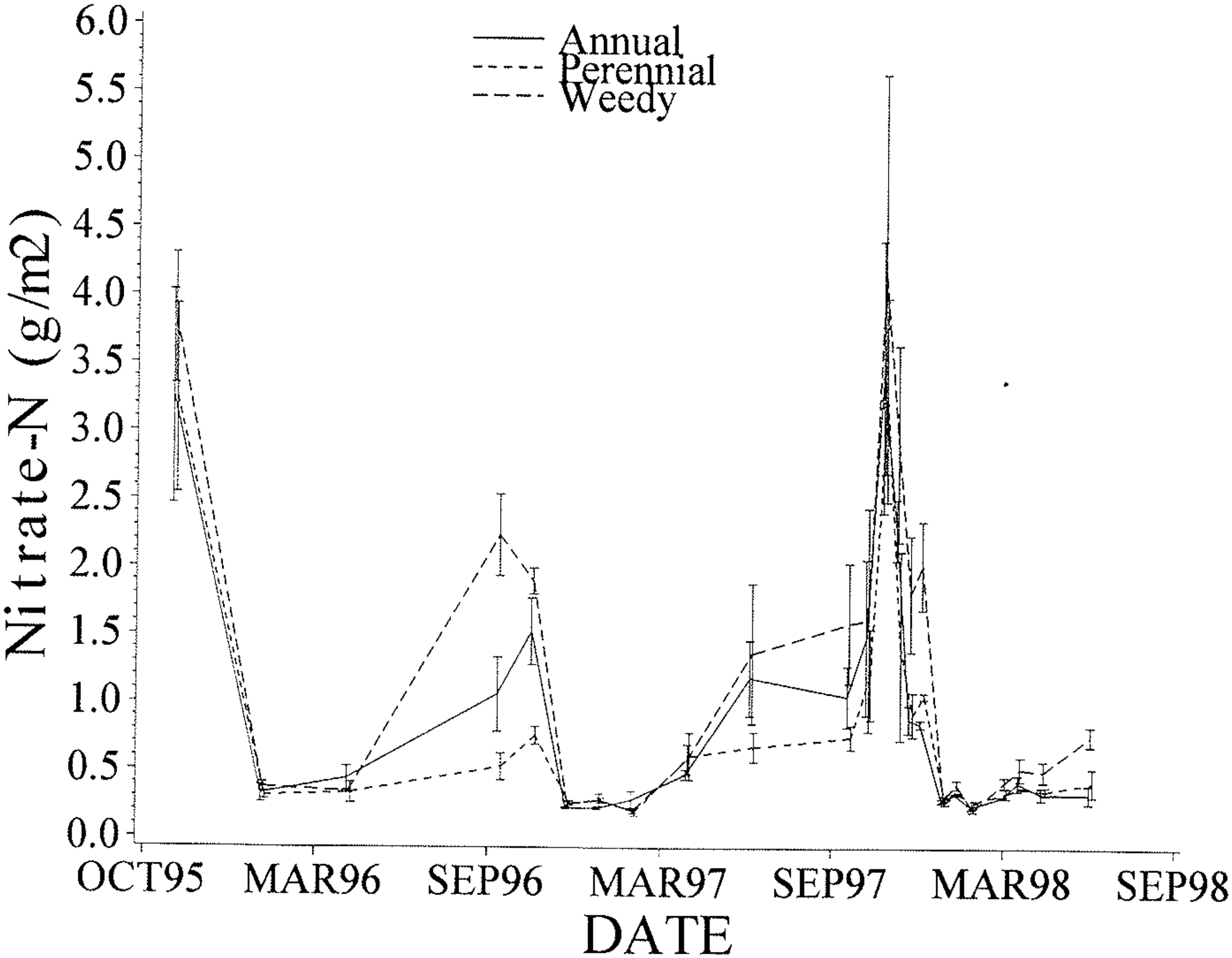


Figure 1. Average Treatment Soil Nitrate Concentrations.

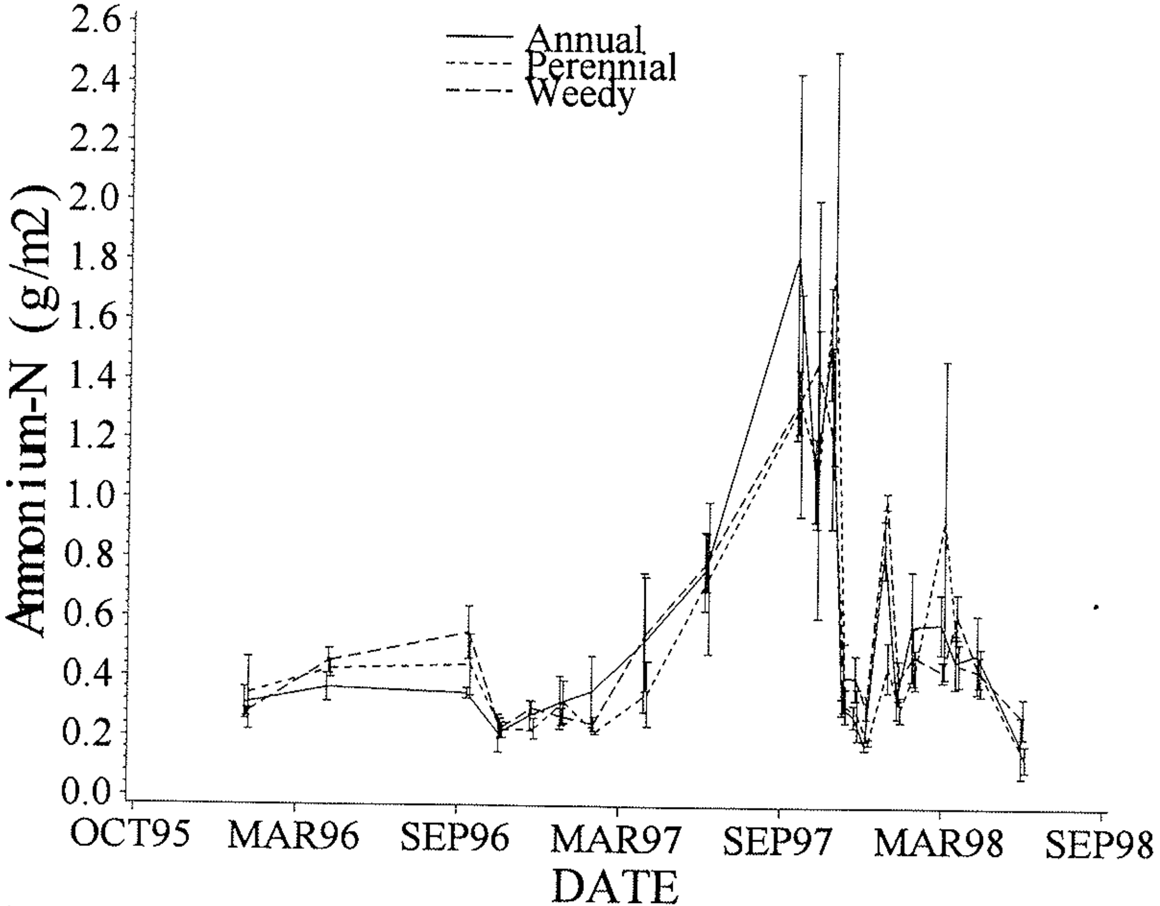


Figure 2. Average Treatment Soil Ammonium Concentrations.

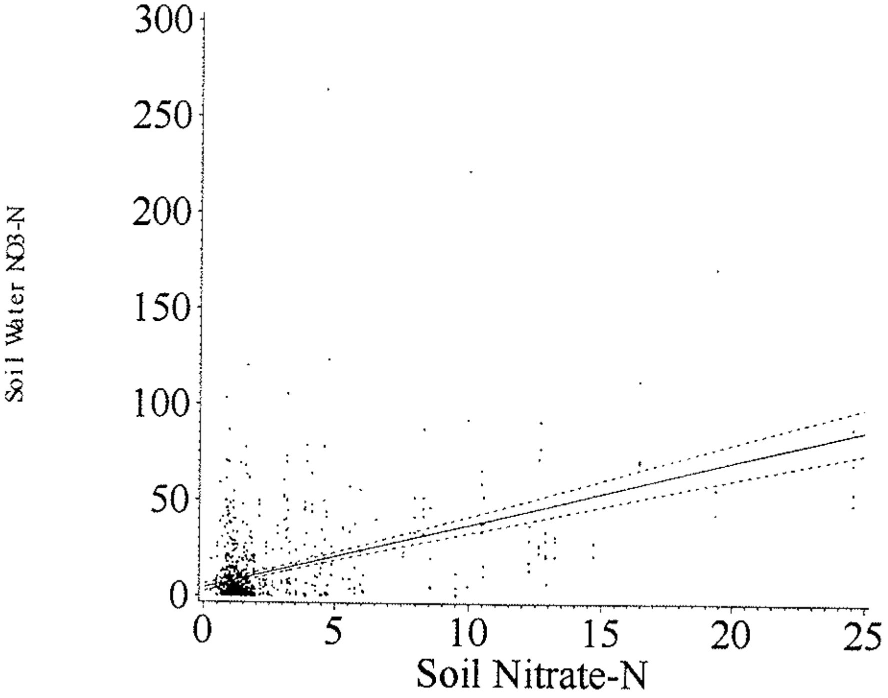


Figure 3. Soil Nitrate and Soil Water Nitrate Concentrations.

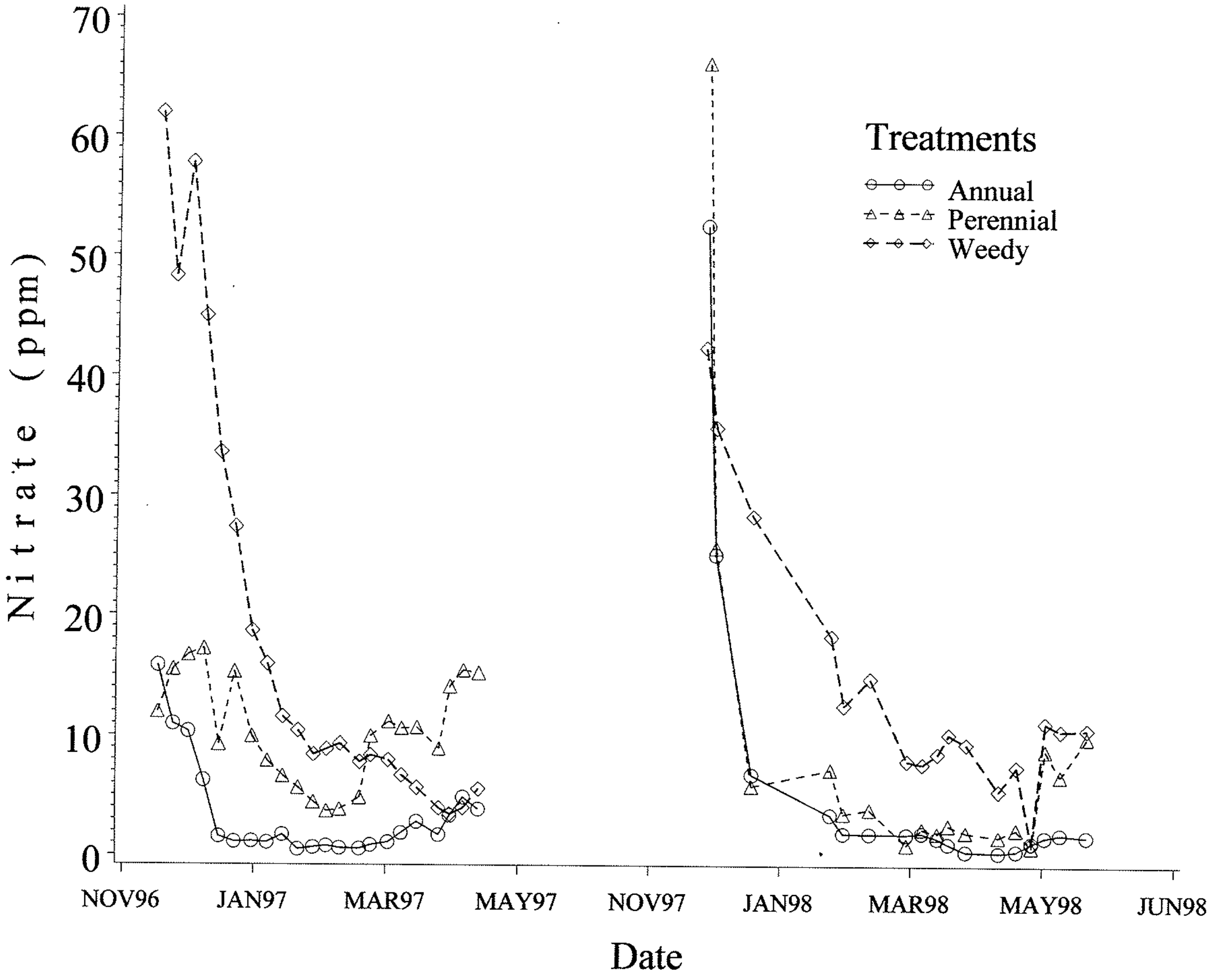


Figure 4. Soil water concentrations at the bottom of the slope.

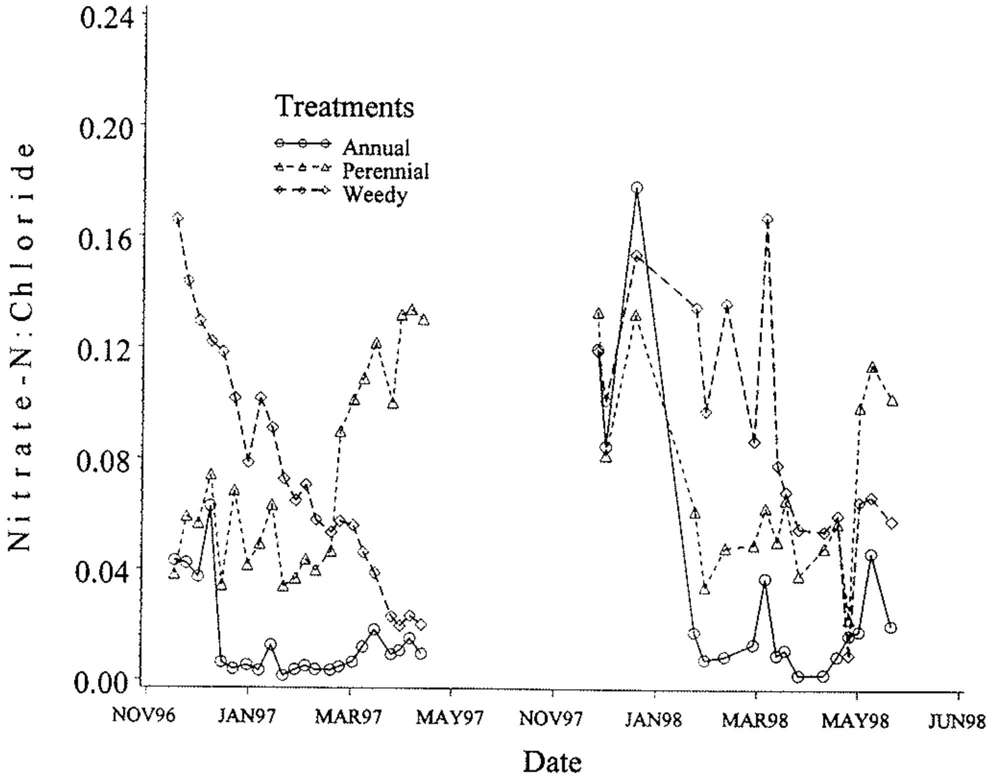


Figure 5. Nitrate-N:chloride ratio in soil water at the bottom of the slope.

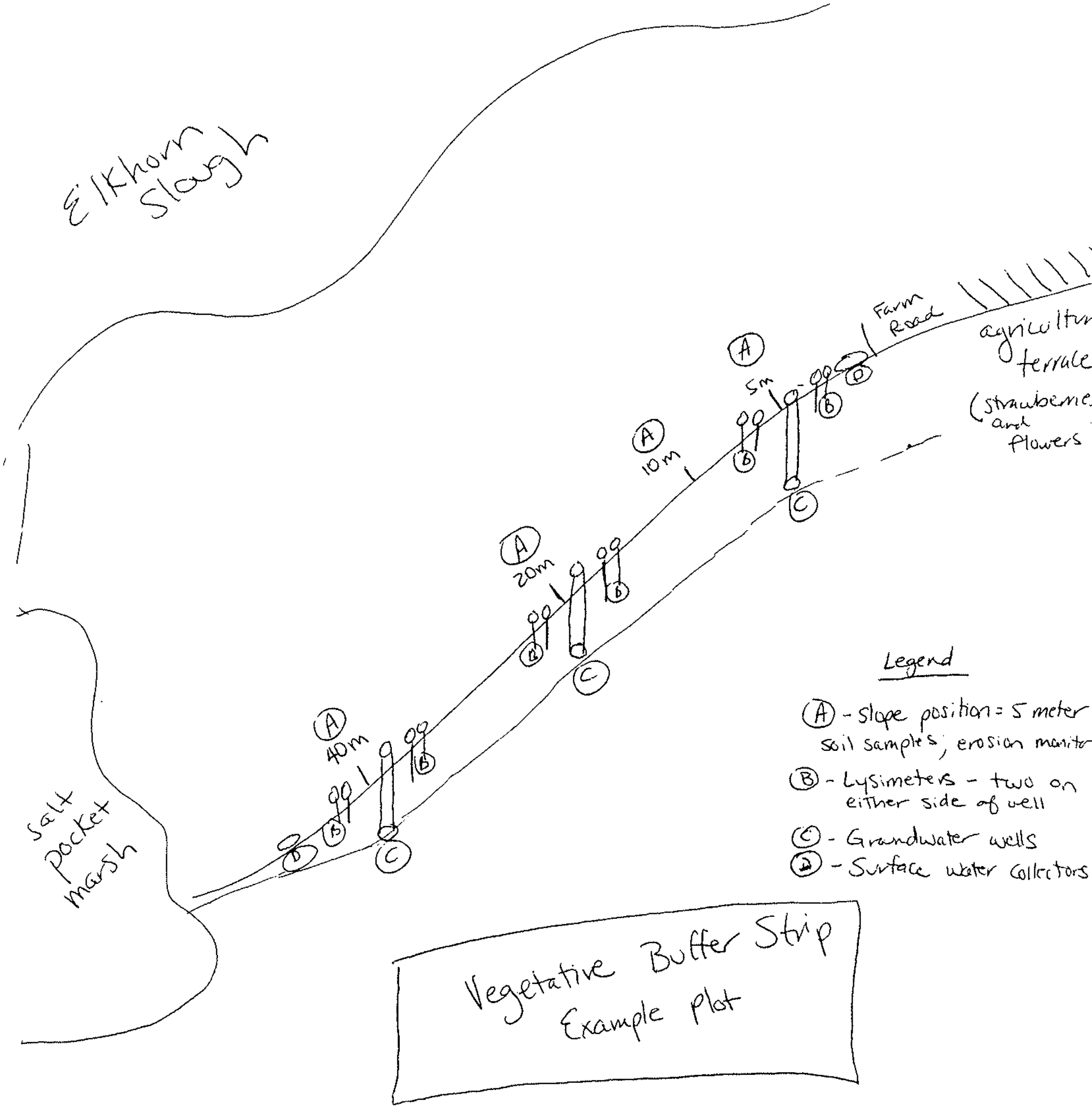


Figure 6: Sampling location map

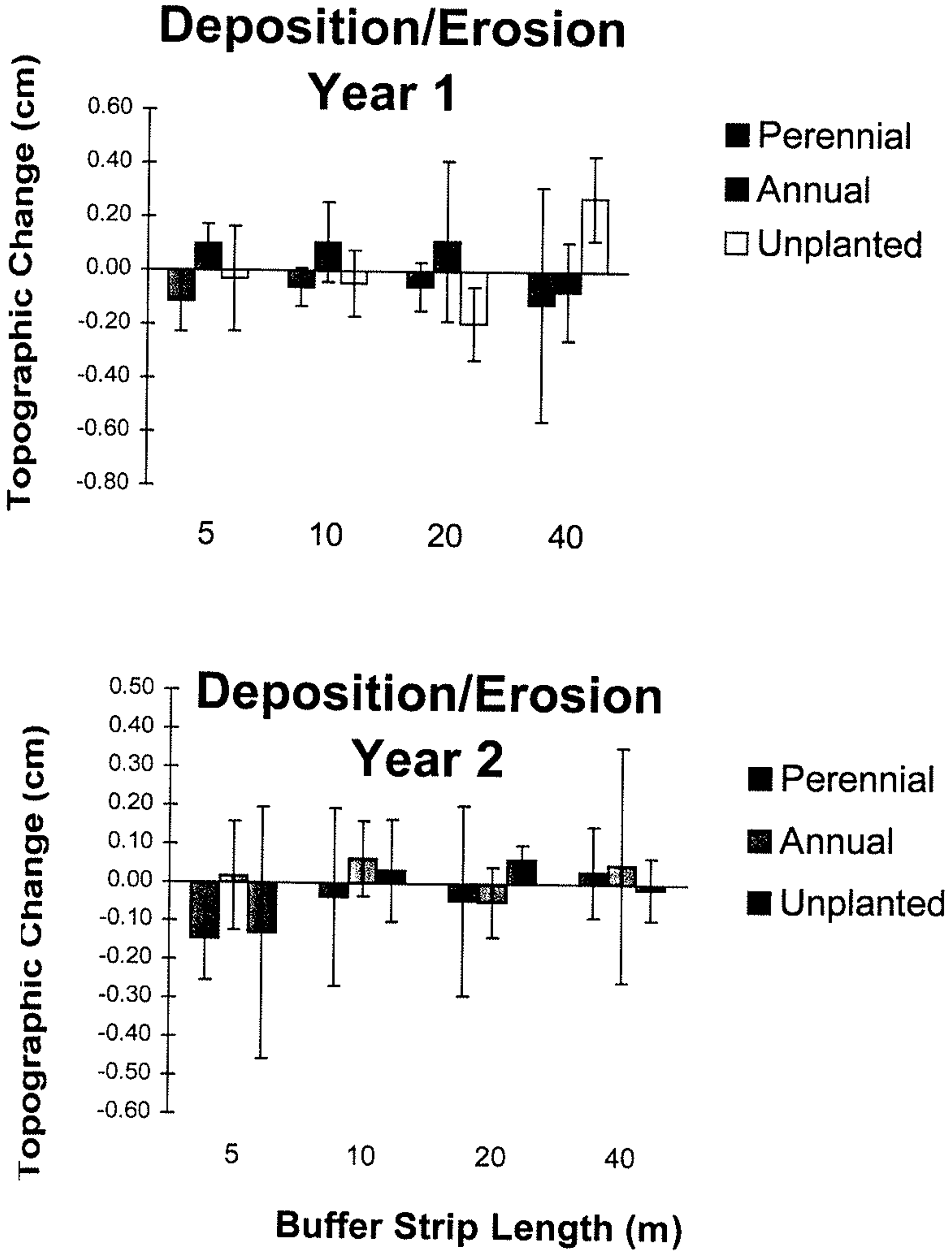


Figure 7- Deposition and Erosion at four slope positions in the first year of the study

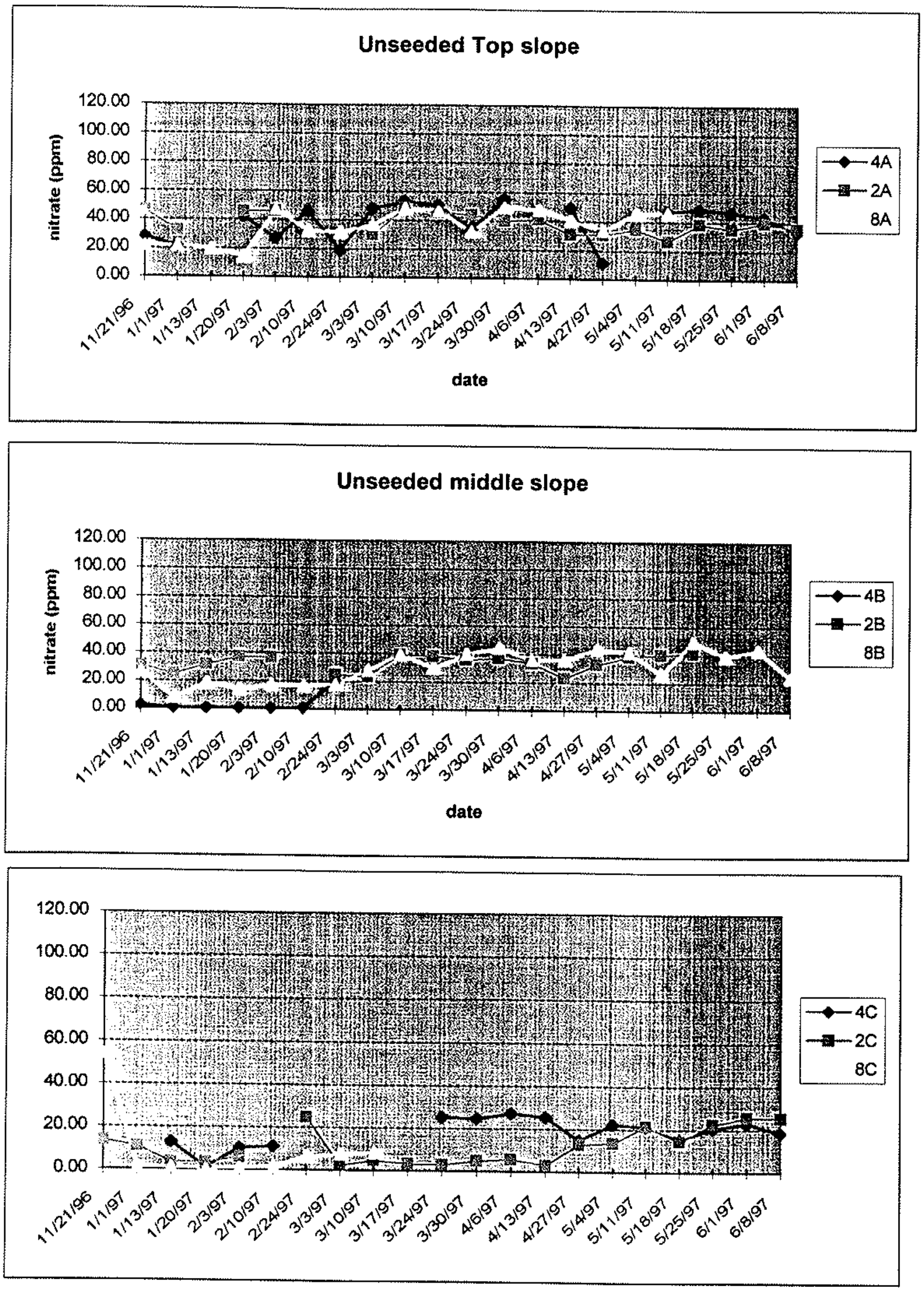


Figure 12: Unseeded groundwater nitrate 1996-1997 at top, middle and bottom slope

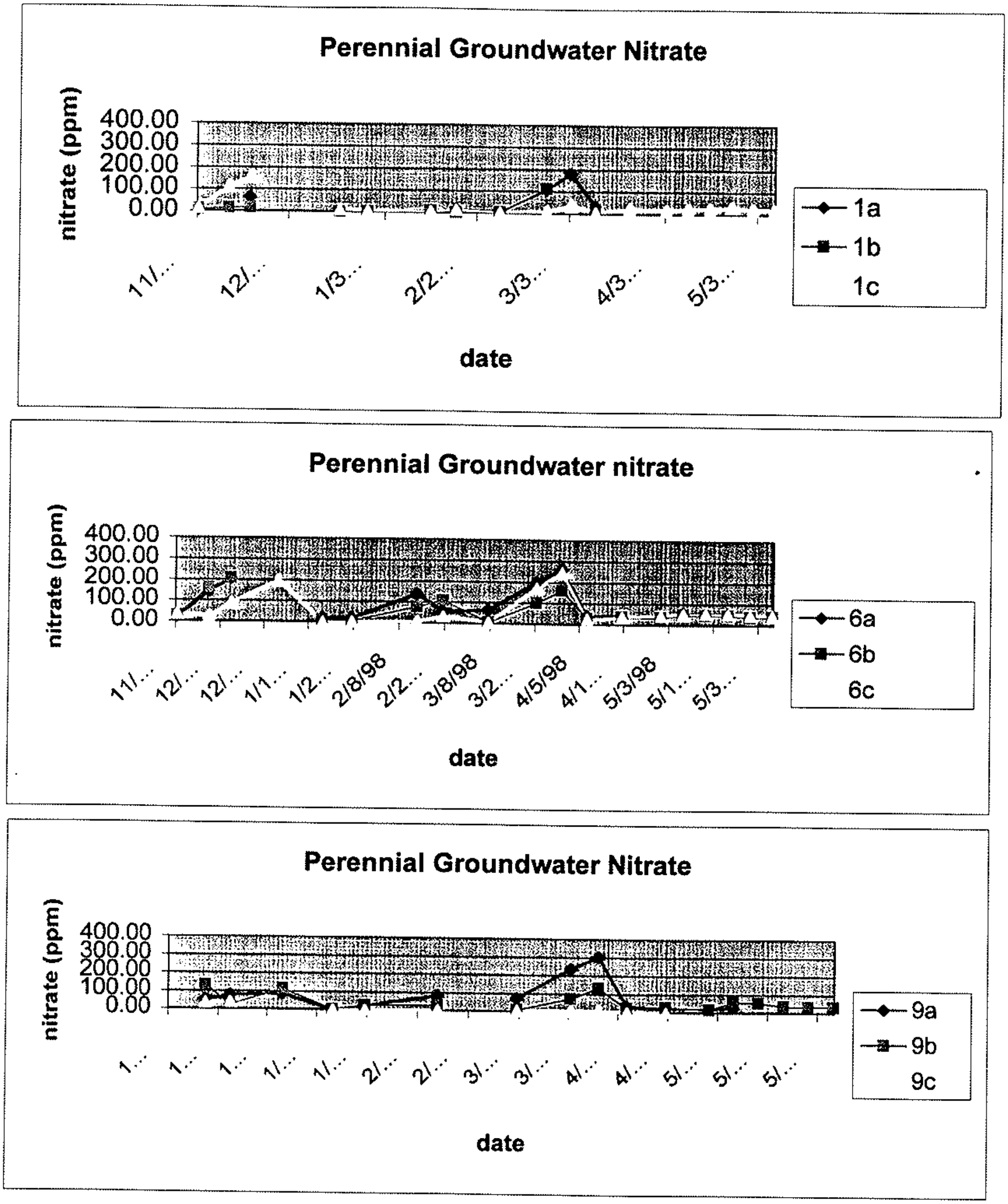
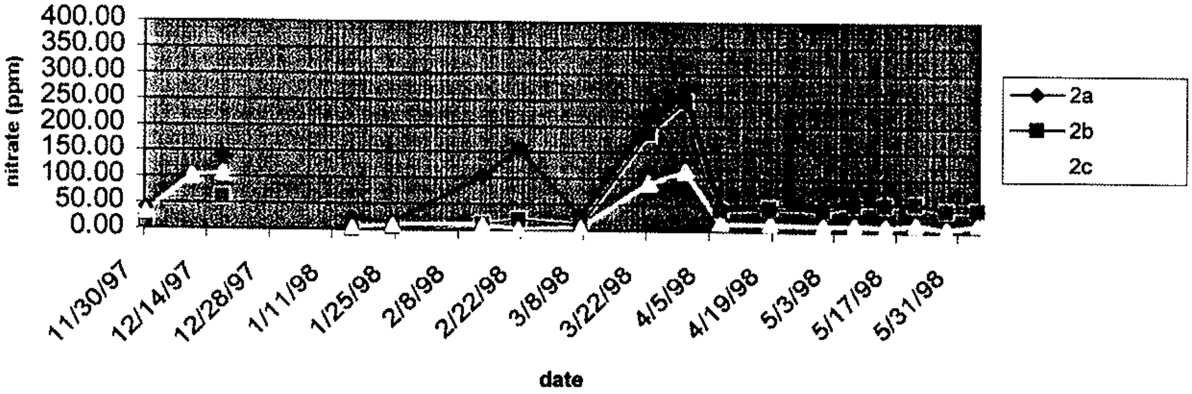
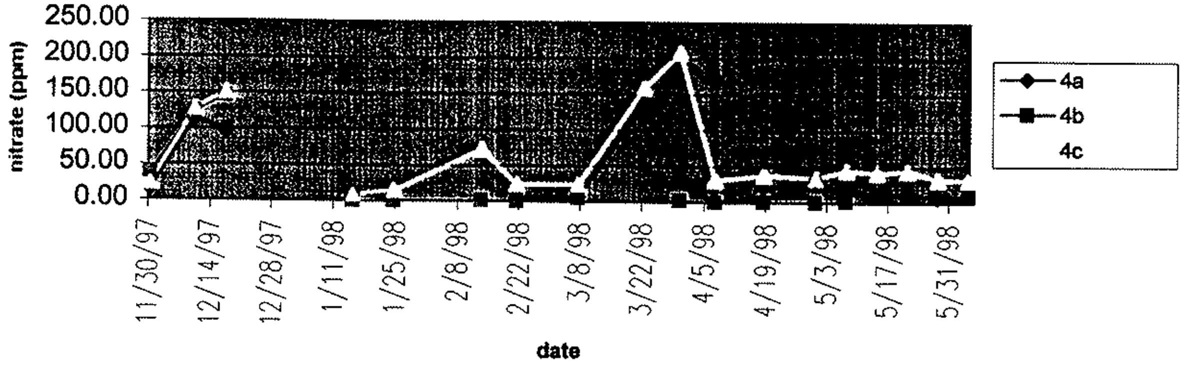


Figure 13: Perennial groundwater nitrate 1997-1998 at all slope positions

Unseeded Groundwater Nitrate



Unseeded Groundwater nitrate



Unseeded Groundwater Nitrate

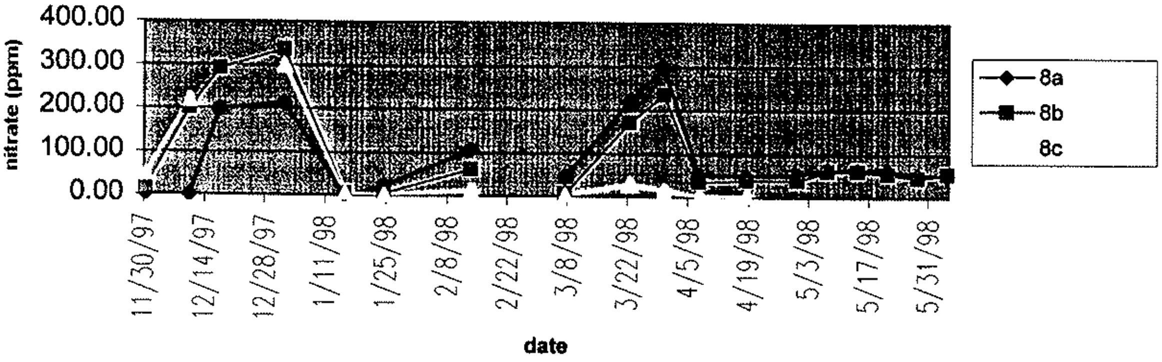


Figure15: Unseeded groundwater nitrate 1997-1998 at all slope positions

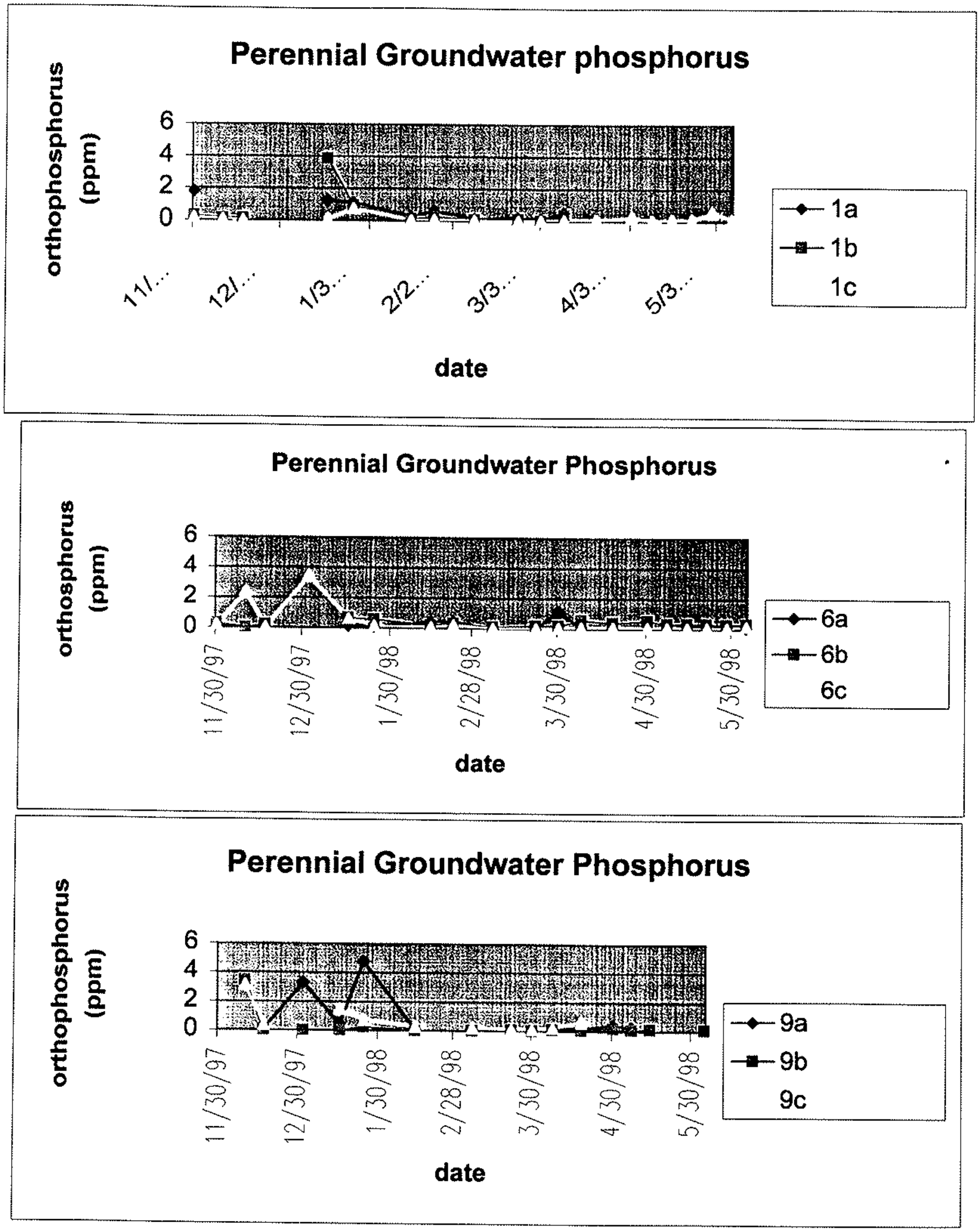


Figure 16: Perennial 1997-1998 Groundwater Phosphorus at all slope positions

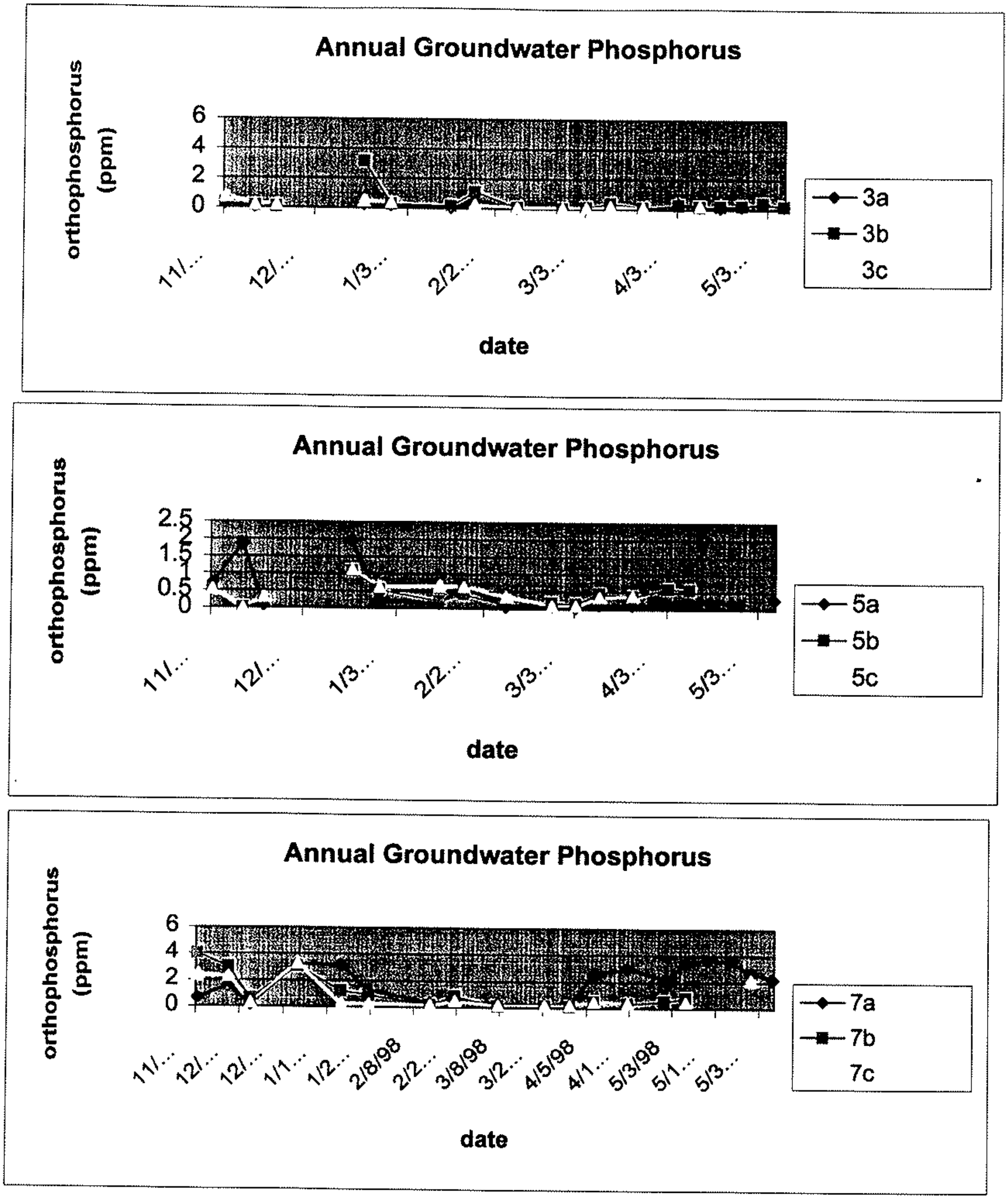


Figure 17: Annual 1997-1998 Groundwater Phosphorus at all slope positions