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Shallow groundwater quality on dairy farms with irrigated forage crops

Thomas Harter*, Harley Davis [1], Marsha C. Mathews [2], Roland D. Meyer [3]

Department of Land, Air, and Water Resources, University of California, One Shields Avenue, Davis, CA 95616-8628, USA

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Abstract

California's dairies are the largest confined animal industry in the state. A major portion of these dairies, which have an average herd size of nearly 1000 animal units, are located in low-relief valleys and basins. Large amounts of liquid manure are generated and stored in these dairies. In the semi-arid climate, liquid manure is frequently applied via flood or furrow irrigation to forage crops that are grown almost year-round. Little is known about the impact of manure management practices on water quality of the extensive alluvial aquifers underlying these basins. The objective of this work is to assess nitrate and salt leaching to shallow groundwater in a relatively vulnerable hydrogeologic region and to quantify the impact from individual sources on dairies. The complex array of potential point and nonpoint sources was divided into three major source areas representing farm management units: (1) manure water lagoons (ponds); (2) feedlot or exercise yard, dry manure, and feed storage areas (corrals); and (3) manure irrigated forage fields (fields). An extensive shallow groundwater-monitoring network (44 wells) was installed in five representative dairy operations in the northeastern San Joaquin Valley, CA. Water quality (electrical conductivity, nitrate-nitrogen, total Kjehldahl nitrogen) was observed over a 4-year period. Nitrate-N, reduced nitrogen and electrical conductivity (EC, salinity) were subject to large spatial and temporal variability. The range of observed nitrate-N and salinity levels was similar on all five dairies. Average shallow groundwater nitrate-N concentrations within the dairies were 64 mg/l compared to 24 mg/l in shallow wells immediately upgradient of these dairies. Average EC levels were 1.9 mS/cm within the dairies and 0.8 mS/cm immediately upgradient. Within the dairies, nitrate-N levels did not significantly vary

* Corresponding author. Fax: +1-530-752-5262.
E-mail addresses: thharter@ucdavis.edu (T. Harter), mcmathews@audavis.edu (M.C. Mathews), rdmeyer@ucdavis.edu (R.D. Meyer).
[1] Regional Water Quality Control Board, Sacramento, CA 95827, USA.
[2] University of California Cooperative Extension, Stanislaus County, Modesto, CA 95358, USA.
[3] Fax: +1-559-752-1552.

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across dairy management units. However, EC levels were significantly higher in corral and pond areas (2.3 mS/cm) than in field areas (1.6 mS/cm) indicating leaching from those management units. Pond leaching was further inferred from the presence of reduced nitrogen in three of four wells located immediately downgradient of pond berms. The estimated minimum average annual groundwater nitrate-N and salt loading from manure-treated forage fields were 280 and 4300 kg/ha, respectively. Leaching rates for ponds are estimated to be on the order of 0.8 m/year, at least locally. Since manure-treated fields represent by far the largest land area of the dairy, proper nutrient management will be a key to protecting groundwater quality in dairy regions overlying alluvial aquifers.

Keywords: Groundwater; Contamination; Dairy; Nitrate; Salinity; Nitrogen; Manure

1. Introduction

Manure handling and disposal practices on animal farming operations (AFOs) are currently undergoing critical revision to reduce their impact on water quality. In the United States, protection of surface waters dominate the national discussion. But in the lower relief, semi-arid watersheds of California's dairy basins (Sacramento Valley, San Joaquin Valley, Tulare Basin, Chino Basin, see Fig. 1), groundwater quality impacts are a primary concern, particularly salt and nitrate leaching. The alluvial and fluvial basin fill aquifers of these large watersheds (10^3-10^5 km^2) are a major source of irrigation water. They are also the almost exclusive source of domestic and municipal drinking water in the area. Agricultural activities in general and dairy operations in particular, have been identified as a potentially significant source of nitrate contamination in groundwater of these basins (Lowry, 1987; Mackay and Smith, 1990; Burow et al., 1998; Boyajian and Ross, 1998). In California, dairies are by far the largest confined animal industry with a total herd size of 1.5 million dairy cows (more than 15% of the national herd size). In 1999, the average size of California's 2200 dairy farms was over 650 milk cattle (approximately 870 animal units) not including dry cows, heifers, and calves (CDFA, 2000; RWQCB, 1999). The annual production of liquid dairy manure is estimated to be approximately 120 million m^3 (Van Eenennaam, 1997; Shultz, 1997).

Little is known about direct groundwater quality impacts from the many elements of dairy manure management operations. As a result, little guidance is available on how to effectively prevent groundwater leaching and how to monitor groundwater quality within AFOs. The objective of this paper is to provide a comprehensive assessment of shallow groundwater quality (specifically, nitrate and total salts) under the immediate, unmitigated influence of dairies with manure management practices typical for semi-arid California. The organization of this paper is as follows: The next section provides a brief characterization of the complex structure of a dairy as a potential nonpoint source of groundwater nitrate and salinity. Then we describe a multi-year field monitoring study of five dairies. The study is designed to assess groundwater quality at the water table beneath dairies in one of the most vulnerable hydrogeologic dairy regions (presumably a worst case scenario). Results of the nitrate monitoring are analyzed to understand potential groundwater quality impacts from the various components of the dairy manure management

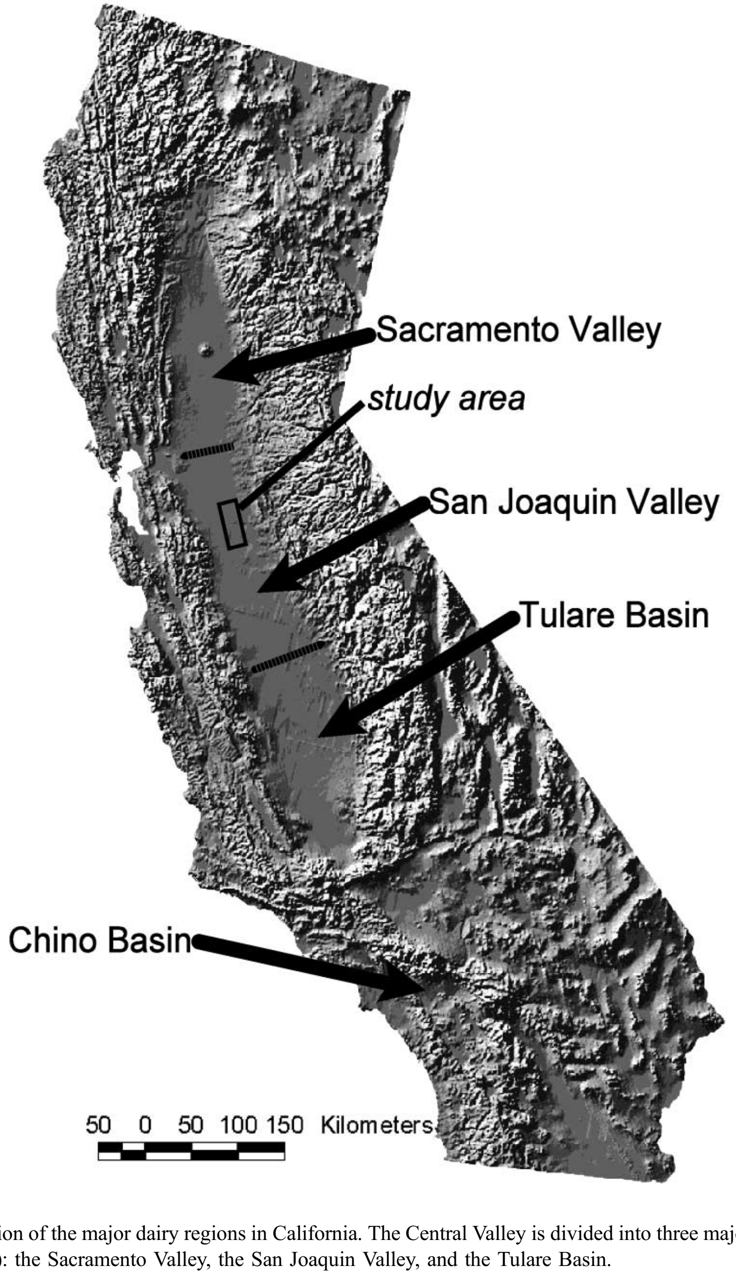

Fig. 1. Location of the major dairy regions in California. The Central Valley is divided into three major sub-basins (dotted lines): the Sacramento Valley, the San Joaquin Valley, and the Tulare Basin.

operation. The paper provides a first step towards a more comprehensive assessment of shallow groundwater quality across all potential pollutant sources within dairies (and similar AFOs).

2. Characterization of potential salt and nitrogen sources within dairies

The primary potential source of groundwater nitrate and salt within dairy systems is manure. In the semi-arid to arid climate of the Western United States, manure management practices differ in many ways from those in the colder climates of traditional dairy regions in the north-central and north-eastern US or in Central Europe. In California's basins, precipitation of 200–500 mm annually occurs only during the cool, but mostly frost-free winter months. Dairies commonly use flushed freestalls in open barns, surrounded by uncovered corrals (exercise yards, animal holding areas) (Meyer et al., 1997). Manure in the freestalls is flushed into a liquid manure storage pond (also called a "lagoon", henceforth referred to as "pond"). Pond manure water is recycled for flushing. Manure water in the anaerobic storage pond contains from 2000 to 4500 mg/l total salts, and 200–1000 mg/l total nitrogen with one third to two thirds of the N as ammonia and the remainder in the organic form (Mathews et al., 2001). Manure solids from the flush are separated from the liquid portion in settling basins or by mechanical devices. Manure solids, including those scraped off corral areas are stored on-site for composting, land application, and for use as bedding material. Often, manure solids are hauled off-site by truck. Cow wash and milk barn operations continuously add fresh water to the liquid manure recycling system, thereby gradually filling the storage pond. Intermittent runoff from the corrals is also captured by the recycling system and stored in the pond.

The diluted liquid manure is eventually applied to adjacent forage crop land via flood or furrow irrigation system (Schwankl et al., 1996; Meyer et al., 1997). Irrigations with liquid manure typically occur during the late fall to create pond storage capacity for the winter, during the rainy winter months if runoff collection exceeds pond storage capacity, in the spring during pre-irrigation, and intermittently on summer crops. Irrigated fields comprise the majority of the land area within a typical dairy (several tens to a few hundreds of hectares, Fig. 2). Crops are grown almost year-round. Most dairies grow corn (maize) silage during the summer followed by fall planting of cereal grains (oats, *Avena sativa*, wheat, *Triticum* sp., or barley, *Hordeum* sp.), which is harvested as forage in early spring. In some regions this double cropping system is rotated with alfalfa (lucerne, *Medicago sativa*) or other crops that occasionally receive applications of diluted liquid manure.

Dairy operators have commonly managed the land application of manure as a waste disposal system, not as a nutrient management system. Application to fields has therefore been dictated not by seasonal crop nutrient demands but primarily by the capacity and layout of the irrigation system, by pond storage capacity, and by the type of crop (some crops are perceived to be too sensitive for manure application). Often, commercial fertilizer is applied in addition to manure to meet the perceived nutrient requirements of the crop (Schwankl et al., 1996; Meyer et al., 1997; Mathews et al., 1999; Meyer and Schwankl, 2000).

Hydrologically, these dairy systems represent a complex conglomeration of multiple potential point and nonpoint sources for nutrient and salt leaching to groundwater. Potential sources include: freestalls, corrals, underground pipelines and storage facilities of the waste recycling systems, the manure solids storage area, the feed storage area,

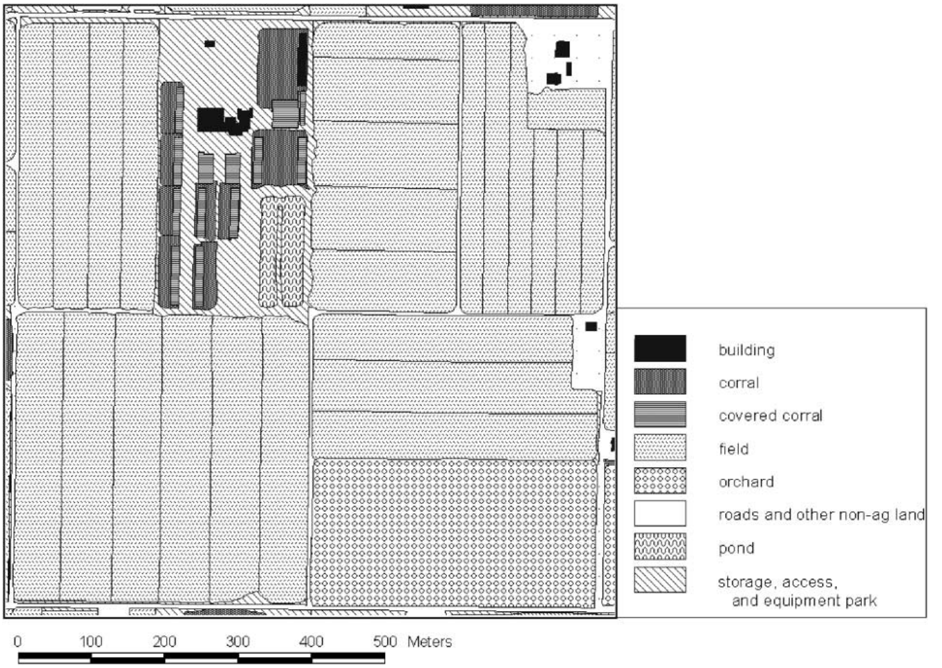

Fig. 2. Map of a typical freestall dairy with corrals, feed storage areas, solids storage area, liquid manure storage pond (lagoon), residences, and forage crop land. Total dairy area is approximately 64 ha.

settling ponds and liquid manure storage ponds, land application of manure, and commercial fertilizer applications on crop land (Fig. 2). Septic systems for residences within the dairy may also leach nitrate to groundwater. Dairies are not the only sources of groundwater nitrate. Residential septic system, commercial fertilizer applications outside AFOs, and surrounding urban activities (golf courses, septic systems, municipal waste application) are potential sources of groundwater nitrate in these dairy regions as well.

A simplified structure of the complex assembly of potential sources is introduced here. Potential and known source areas are grouped by the three major operational management units within the dairy: "corrals", which include all animal holding areas (feedlots, freestalls, exercise yard, milkbarn), "ponds" including settling ponds and liquid manure storage ponds (also known as storage lagoons), and "fields", which are irrigated cropping areas receiving liquid manure water either regularly or intermittently. These dairy management units are distinguished from the "upgradient" areas outside and upgradient of the dairy property.

Most research work to date has focused on evaluating nitrate leaching from specific AFO manure management components or from specific manure management practices, although some studies address regional impacts (a summary is contained in Table 1). We are not aware of any comprehensive groundwater quality assessment across all management units of an AFO. Little is known about the degree of spatial and temporal

Table 1
Examples of AFO-related groundwater quality studies including pond and field leaching studies and regional surveys in AFO regions

Citation	Nitrogen source	Location	Study design	Observed range of N	Comments
Culley and Phillips, 1989	small-scale unlined earthen manure pits with liquid dairy manure	Ottawa, Canada	one pair of pits at each of three locations w/ different soils (clay, clay loam, sand); nested soil water suction lysimeters at 1–3.5 m below surface	NO_3–N from less than 2 to 60 mg/l	no effective self-sealing; water quality strongly dependent on soil type
Davis et al., 1973	dairy lagoons	Merced County			lagoon "self-sealed"
McCurdy and McSweeney, 1993	dairy lagoons	Sauk County, WI	morphologic and pedogenic analysis of existing dairy liner		
Ham and DeSutter, 1999	swine-waste lagoons	near Ulysses, KS	three lagoons, seepage rate computed by water balance over a 5- to 7-day period	seepage rate 0.8–1.1 mm/day; lagoon total N concentration: 700–900 mg/l	
Korom and Jeppson, 1994	dairy lagoons	Heber Valley, UT	two farm lagoons, located in coarse alluvial deposits; soil water suction lysimeter samples over a 2-year period	seepage rate 13–91 mm/day; NO_3–N from less than 5 mg/l to over 150 mg/l; mean concentrations of 37–128 mg/l	some sealing after initial construction
Meyer et al., 1972	dairy lagoons	Merced County			lagoon "self-sealed"
Ritter and Chirnside, 1990	animal waste lagoons (hog, beef)	Delmarva Peninsula, Delaware	monitoring well network around each of two farm lagoons; depth to groundwater 0.6–3 m; 3-year observation period	3-year mean of NH_4–N from 8 to 970 mg/l; of NO_3–N from less than 1 to 40 mg/l	

Chang and Entz, 1996	solid cattle manure application to crop	Lethbridge, Alberta	20-year split-plot experiment; one irrigated site, one non-irrigated site; three tillage treatments; four manure application rate treatments; clay loam soil. Soil core samples once per year to depth of 150 cm	no leaching loss of NO_3-N in non-irrigated soil except in high precipitation years; leaching losses of 100–300 kg/ha per year under irrigated conditions	long-term N mineralization: 56% of applied N
Joshi et al., 1994	liquid manure application to crop	Goodhue County, MN	3-year field study, split plot, randomized block design, two tillage treatments, three fertilization treatments, silt loam soil. Manure application by slurry injection at 284 kg N/ha per year. Equivalent mineral fertilizer at 200–240 kg/ha per year. Weekly soil water quality monitoring at 1.5 m depth with suction cup samples (April–November)	NO_3-N concentration at 1.5 m from less than 20 to 60 mg/l	manure treatments yielded lower NO_3-N than mineral fertilizer treatments
Spalding et al., 1993	human sewage sludge amendments to corn field	Platte River Valley, NE	water quality survey in groundwater monitoring well network to monitor nitrate plume emanating from field after 8 years of sludge application	NO_3-N within plume: up to 40 mg/l	nitrogen isotope ratios indicate that some denitrification occurs in a clayey silt layer at 10 m depth

(continued on next page)

Table 1 (*continued*)

Citation	Nitrogen source	Location	Study design	Observed range of N	Comments
Stout et al., 1997	urine and feces application in pasture	central Pennsylvania	3-year study on field installed drainage lysimeters beneath N-fertilized orchardgrass, silt loam soil; urine and feces treatments	NO_3–N leaching losses: 100–300 kg/ha under urine treatment (18–31% of urine-N applied); only small losses under feces treatment (2% of N applied)	
Boyajian and Ross, 1998	sub-regional survey	Tulare County, CA	single water quality survey of 60 domestic and irrigation wells within 25 km^2 in dairy farming community	NO_3–N: less than 1 mg/l to 50 mg/l; average for AFO wells: 17 mg/l; non-AFO wells: 8 mg/l	depth to water table approximately 30 m; highly heterogeneous alluvial deposits in the unsaturated zone
Lowry, 1987	sub-regional survey	Hilmar, Merced County, CA	single water quality survey of 69 domestic wells within 90 km^2 in dairy farming community	NO_3–N: less than 1 mg/l to 38 mg/l; average 17 mg/l to 11 mg/l, decreasing with well completion depth (<15, 15–45, 45–60 m); 16 of 26 dairy wells exceed 10 mg/l	
Wildermuth Environmental, 1999	sub-regional survey	Chino Basin, Riverside County, CA	comprehensive groundwater study for management purposes and water rights administration; from 1960, a significant part of the basin has almost exclusively been occupied by dairies	NO_3–N in production wells throughout dairy area: from less than 1 to 66 mg/l (1991–1995)	
Fryar et al., 2000	regional survey	Southern High Plains, TX	soil core, soil gas, and groundwater sampling in and around playas receiving animal waste runon to characterize denitrification	NO_3–N in soil corings: less than 1 to 19 mg/kg, on core with up to 160 mg/kg	denitrification limits N loading to groundwater

Goss et al., 1998	regional survey	Ontario, Canada	stratified random water quality survey of 1292 (out of 500,000) farm water wells, four per township	14% of wells have NO_3-N > 10 mg/l; highest frequency of "contamination" in shallowest wells; no association of NO_3-N with feedlot occurrence
Hudak, 2000	regional survey	Texas	county-by-county survey of well water nitrate; association with well depth, aquifer type, county-wide land use	feedlot density (high in Texas panhandle, SW Texas) not related to nitrate levels
Oenema et al., 1998	regional survey	Netherlands	national groundwater monitoring program in sandy soil regions, annual samples taken at lowest groundwater level during 1992, 1993, and 1995; analysis by overlying farming system	dairy farming (>2.8 animal units/ha): 55, 51, and 25 mg NO_3-N/l in 1992, 1993, and 1995; mean N surplus: 410 kg/ha/year
Rudolph et al., 1998	regional survey	Ontario, Canada	2-year water quality survey of multilevel monitoring wells at 144 farms selected from Goss et al. (1998); monitoring wells installed in cultivated fields near farm wells; variable soil types (clayey soils, highly heterogeneous soils, sandy soils)	45% of wells exceeded 10 mg NO_3-N/l; max. concentration 87 mg/l; avg. concentration decreases from 10 mg/l near water table to 3 mg/l at 6.5 m; approximately 1/3 of wells located in fields that receive manure; there, significantly higher NO_3-N levels

variability of shallow groundwater nitrate (or salt) concentration within an AFO or the integrated dynamics and mixing of nitrate from multiple sources in groundwater underneath an AFO. As a result, groundwater monitoring guidelines for AFOs lack a critical scientific basis and monitoring results are subject to usually speculative interpretation.

3. Methods

The groundwater quality investigation focuses on the occurrence and distribution of nitrate, reduced nitrogen (ammonia and organic nitrogen, measured as total Kjehldahl nitrogen, TKN), and salts (measured as electrical conductivity, EC). Our approach is threefold:

1. A 4-year groundwater monitoring program was implemented on dairy facilities with an average of approximately 1000 animals units and 60 ha of crop fields (longitudinal study).
2. Data are analyzed through a combination of statistical analysis and simplified hydraulic model analysis (as opposed to, e.g., site specific groundwater modeling).
3. Comparison to and analysis of data reported in the literature are used in lieu of cross-sectional study (i.e., dairy/AFO industry survey) to determine the representative value of the site study with respect to similar operations in the region and elsewhere.

3.1. Study area, climate, soils and hydrogeology

Our study area is the central-eastern portion of the northern San Joaquin Valley. The San Joaquin Valley is one of the three large sub-basins in California's Central Valley, a low relief structural basin from 60 to 100 km wide and 700 km long (Fig. 1). The study area comprises the low alluvial plains and fans bordered by the San Joaquin River to the west, tertiary upland terraces to the east, the Stanislaus River to the north, and the Merced River to the south. The region has a long history of nitrate and salt problems in groundwater (Page and Balding, 1973; Lowry, 1987). The climate is mediterranean with annual precipitation of 290 mm, practically all of which occurs between late October and early April. The area is characterized by featureless topography with slopes of less than 0.2%.

The upper basin fill contains the main aquifer and consists of primarily quaternary older alluvial and fluvial deposits with some interbedded hardpan and lacustrine deposits. Alluvial deposits are a few hundred meters thick near the valley trough and pinch out towards the eastern edge of the valley floor. East of the valley-trough, the unconfined to semi-confined aquifer in the upper 100–200 m of the basin fill serves as the major regional groundwater production zone. There, groundwater generally flows from the east–northeast to the west–southwest following the slope of the landscape. The average regional hydraulic gradient ranges from approximately 0.05% to 0.15%. The water table at the selected facilities is between 2 and 5 m below ground surface. Hydraulic conductivity (K) of the shallowest aquifer material has been estimated from slug tests. Measured K values range from 1×10^{-4} to 2×10^{-3} m/s (Davis, 1995), which is consistent with the predominant texture of the shallow sediments. The geometric mean K value at the monitoring sites is 5×10^{-4} m/s.

Soils formed on flood plains and wind modified alluvial fans. The dominant surface texture is sandy loam to sand underlain by silty lenses, some of which are cemented with lime. Some soils may have a slight accumulation of clay in their subsoil. Water holding capacity is low. Where the water table is high, large community drainage systems with shallow groundwater pumps are used to good effect. Because of the high infiltration capacity of the soils, border flood irrigation of forage crops has historically been the dominant cropping system among dairies in the study area. Low salinity surface water from the Sierra Nevada (EC: 0.1–0.2 mS/cm) is the main source of irrigation water.

Three hydrogeologic criteria made the area particularly suitable as a field laboratory for investigating recharge water quality from AFOs: The high groundwater vulnerability reflects a worst case scenario. The shallow groundwater table allowed for relatively low cost access. Travel times in the unsaturated zone are short. Most importantly, the relatively small long-term fluctuations in water level (1–2 m) allowed us to sample a well-defined, vertically very narrow zone immediately below the water table using a fixed depth monitoring well network.

3.2. Dairy sites and monitoring well network

The five selected facilities are among leading facilities in the region with respect to herd health, product quality, and overall operations. Improvements in manure and pond management have continually occurred since the inception of the project. The dairies are located in a geographic and hydrogeologic environment that is representative of many other dairies on the lowlands of the northern San Joaquin Valley. The manure management practices employed at these dairies over the past 35 years, particularly with respect to corral design, runoff capture, and lagoon management, have been recognized by industry, regulators, and university extension personnel as typical or even progressive relative to other California dairies (Schwankl et al., 1996; Meyer et al., 1997; Mathews et al., 1999). Over the past 30–40 years, the herd size on these dairies has continually grown from less than 100 at their inception to over 1000 animal units in the 1990s. In 1993, between 6 and 12 shallow groundwater monitoring wells were installed on each dairy (designated as dairies *V*, *W*, *X*, *Y*, *Z*; 44 well total) Monitoring wells are strategically placed (a) upgradient and downgradient from fields receiving manure water, (b) near wastewater lagoons (ponds), and (c) in corrals, feedlots, and storage areas (henceforth referred to as "corrals"). Wells are constructed with PVC pipe (5 cm diameter) and installed to depths of 7–10 m. The wells are screened from a depth of 2–3 m below ground surface to a depth of 10 m. Water samples collected from monitoring wells are representative of only the shallowest groundwater.

3.3. Water quality sampling protocol and analysis

From June 1993 through August 1994, preliminary well samples were taken on four sampling dates. Between November 1995 and November 1999, 35 well sampling campaigns took place on an approximately 5- to 6-week basis. At each sampling campaign, groundwater levels are determined using a calibrated groundwater level meter with an accuracy exceeding 0.005 m. Well water is then purged with a submersible pump and continuously monitored for field water quality parameters (including temperature, electric conductivity, and dissolved oxygen). The field water quality probes are calibrated before each sampling trip and checked against calibration standards at the completion of the sampling trip. Water samples are collected after field water quality stabilizes or after a minimum of 5 well volumes of water has been removed. The depth at which the submersible pump is located during sampling is identical from sampling date to sampling date. It varies from well to well depending on the average depth to water. The pump intake is located at least 1.5 m above the bottom of the well and at least 1.5 m below the water table.

Water samples are cooled and stored at 1 °C for analysis of NO_3–N and total Kjehldahl nitrogen (TKN, measuring the sum of ammonia-N and organic N). For quality control, blank, duplicate, and diluted duplicate samples are prepared in the field from approximately every 10th well water sample. NO_3–N determination is by diffusion–conductivity analyzer (Carlson, 1978). TKN is determined by the wet oxidation of the water samples using standard Kjeldahl procedure with sulfuric acid and digestion catalyst (Keeney and Nelson, 1982).

Four wells were selected to test the sensitivity of measured water quality to the vertical location of the pump within the well and to duration of pumping. The NO_3–N concentration in the selected wells ranged from 31 to 70 mg/l. The wells were purged (60 l, equal to 5 well volumes) and sampled within 0.6 m of the well bottom, at the middle of the screen length, and near the top of the well screen at three separate sampling dates that were 2 days apart. No significant differences were observed between the three depths (less than 5%). At a fourth sampling date, 4000 l (approximately 350 well volumes) were purged from each of these wells and samples taken after removing 80, 200, 400, 800, 2000, and 4000 l. Groundwater quality (NO_3–N, EC, pH, temperature) changed insignificantly over the extended purging period (less than 5%). The routine sampling protocol is therefore considered to provide a representative depth integrated sample of shallow groundwater within 1–2 m around the monitoring well.

3.4. Hydraulic analysis of the monitoring well source area

The source area of a monitoring well is defined as the area from where well water originates as recharge (Fig. 3). A geometric approximation provides a simplistic but useful estimate of the upgradient linear extent of the monitoring well source area, L [m] (Fig. 3):

$$L = d_{\mathrm{MW}}\, q/R \tag{1}$$

where R [m/s] is the net recharge rate in the source area, q [m/s] is the average groundwater discharge rate, and d_{MW} [m] is the length of the screened interval of the monitoring well below the water table. The major source of recharge on and around these dairies is percolation from irrigation water on irrigated crop land during the summer months and rainfall during the winter months. Based on hydraulic gradient data

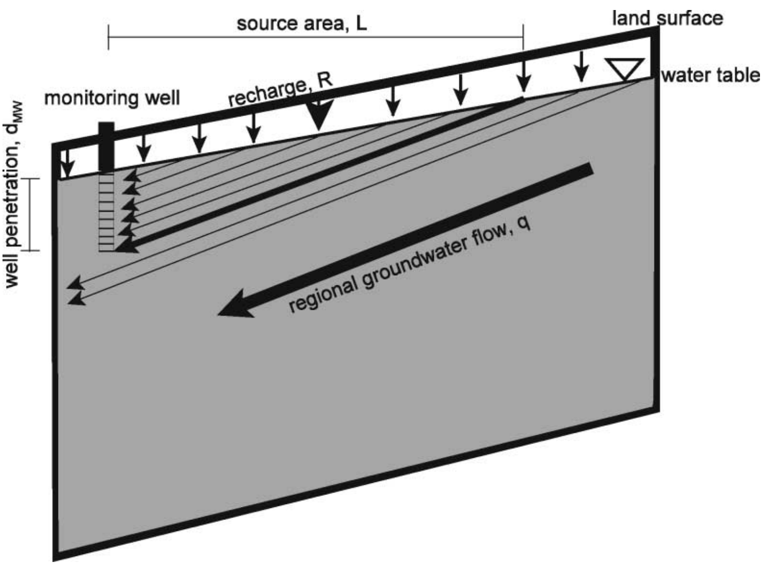

Fig. 3. Simplified shallow aquifer cross-section along the regional groundwater gradient. If uniform recharge rate, R, groundwater discharge rate, q, and monitoring well screen depth, d_{MW}, are known, the size of the source area can be estimated from Eq. (1).

and hydraulic conductivity estimates, long-term average q is on the order of 5×10^{-7} m/s. Recharge, R, to groundwater from irrigated fields ranges from 0.45 to 0.6 m/year (approximately 1.5×10^{-8} m/s). For d_{MW} ranging from 5 to 8 m, the upgradient extent of the source area within irrigated field is therefore estimated to range from approx. 150 m to a few hundred meters, well within the size of most fields.

Leakage rates from corrals and ponds are not known. Therefore, Eq. (1) cannot be applied to estimate the origin of water in corral and pond monitoring wells. However, if long-term water quality analysis of two wells that are a distance, y [m], apart along the groundwater flow path leads to the conclusion that the downgradient well receives completely different source water ($y>L$), then the minimum infiltration rate in the area between the two wells can be computed by re-arranging Eq. (1):

$$R > qd_{MW}/y \tag{2}$$

Eqs. (1) and (2) assume that recharge in the source area is uniform, that groundwater flow is unidirectional, and that recharge and flow rates are constant over time. It neglects the influence of shallow drainage wells or tile drainage. Drainage wells are located downgradient of two dairies (V and W), which were converted to tile drainage in spring 1999. Nearby agricultural and other deep production wells may also contribute to locally non-uniform groundwater gradients. Downgradient influence is considered negligible except near irrigated fields: The vertical downward movement of water from the irrigation application through the unsaturated zone to the water table temporarily raises the water table under the field, thus locally reversing the groundwater flow and impacting monitoring wells that are within several meters upgradient of the field. The lateral (transverse) extent of the upgradient source area is considered to be on the order of a few meters to a few tens of meters and depends on subsurface heterogeneity and temporal changes in groundwater flow direction.

3.5. Management unit classification of wells

The average areas occupied by the three management units "corral", "pond", and "field" (see above) on each dairy are 2.5, 0.9, and 59 ha, respectively. Based on water table maps and the simplified source area model (1), all wells were classified by the management unit located immediately upgradient from the well. Because of their location along access roads, several wells are located at the boundary of two management units and were given a separate "multiple" management units classification (Table 2). "Field" wells were subdivided into those upgradient of the corral and pond areas ("upper field") and those lateral or downgradient from the corral and pond areas ("lower field"). Wells immediately upgradient of the dairy property are considered to belong to a separate "upgradient" management unit. Actual land use within 1000 m "upgradient" from the dairy properties include neighboring dairies with and without manure-treated fields, almond orchards, vineyards, and forage crops with no manure treatment.

3.6. Statistical analysis

Descriptive statistical analyses are carried out for EC and for the sum of measured NO_3–N plus measured TKN concentration, denoted hereafter as total N:

$$\text{total N} = c_{NO_3-N} + c_{TKN}$$

Unless otherwise mentioned, TKN concentrations are negligibly small for purposes of this study (less than 5 mg/l, see below), and N concentrations are equal to NO_3–N concentrations.

Differences between groups of monitoring wells are tested for statistical significance using two steps: First, the 4-year average, m_i, is computed separately for each well, i. Second, a one-way analysis of variance (ANOVA, Davis, 1986) is performed on the set of m_i for specific effects (dairy, management unit). Hence, each m_i is considered an individual

Table 2
Number of monitoring wells within each management unit designation (in parenthesis: number of monitoring wells within each management unit designation when ambiguous management unit location ("multiple") is dropped) and mean, standard deviation, and coefficient of variation of total nitrogen concentration for all individual samples collected, grouped by dairy management unit (including "multiple")

Management unit	Number of wells	Mean N [mg/l]	No. of obs.	Std. dev. N [mg/l]	Coefficient of variation
Upgradient	5	23.5	168	16.3	0.69
Field	18	60.7	589	39.9	0.65
Corral	10 (15)	64.2	322	34.7	0.54
Pond	2 (6)	48.7	49	22.7	0.47
Multiple	9	75.4	274	40.7	0.54
All	44	59.4	1402	39.0	0.66

sample and group means are actually means of the 4-year average of all wells within a group. Individual sampling data are not considered for the ANOVA because well sampling data collected over time are not statistically independent of each other.

4. Results and discussion

4.1. Sample distribution

Fig. 4 summarizes the major statistical sample parameters for total N from 39 wells observed between November 1995 and November 1999 (not including measurements at five "upgradient" wells). The broad distribution of the 4-year arithmetic mean nitrogen

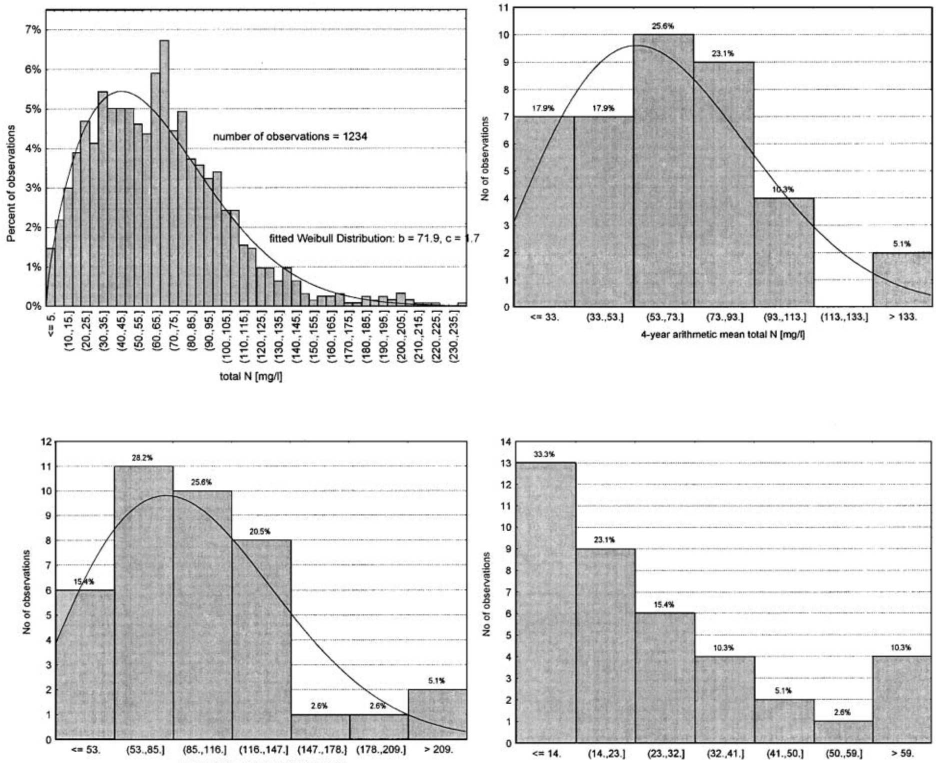

Fig. 4. Top left: Histogram of N concentration distribution in 1234 samples from 39 wells (not including "upgradient" wells). Mean: 64.3 mg/l, median: 61.5 mg/l, lower quartile: 35.2 mg/l, upper quartile: 85.9 mg/l, standard deviation: 38.6 mg/l skewness coefficient: 1.06. A Weibull distribution is fitted to the histogram values using maximum likelihood estimation. The Weibull distribution function is defined as: $f(x)=c/b[x/b]^{c-1}e^{-[x/b]^c}$. The scale and shape parameters, b and c, are estimated to be 71.9 and 1.71. Histogram of the 4-year arithmetic means (top right) and 95th percentile (bottom left) of individuals wells (excluding upgradient wells). The histograms have been fitted with a Weibull distribution using maximum likelihood estimation (means: $b=72.7$, $c=2.1$; 95th percentiles: $b=109.6$, $c=2.1$). Histogram of the spread between 4-year upper and lower quartile (bottom right) of individual wells (excluding upgradient wells).

concentrations at individual wells indicates that a large degree of spatial variability exists within the network. The same observation is made for the distribution of the 95th percentile (second highest) concentration at individual wells. Temporal variability of N at individual wells is almost as large as spatial variability across the network. Differences between minimum and maximum concentration observed at individual wells range from 15 to over 200 mg/l. The differences between the upper and lower quartile N at individual wells are more resistant to outliers, yet still exhibit a broad distribution (Fig. 4).

4.2. Seasonality and long-term variations

Monthly mean N concentrations vary significantly over time although the 4-year observation period (1995–1999) is too short to detect significant long-term trends (Fig. 5). Mean N concentrations during the most recent summer were significantly lower (less than 45 mg/l during three consecutive sampling dates) than during any of the previous years, when mean N ranged from approximately 55 to approximately 70 mg/l. It remains to be seen, whether this is a significant trend that can be related to recent improvements in field nutrient and corral management. Monthly mean EC values vary from 1.7 to 2.3 mS/cm, also with no significant temporal trend.

At the onset of the study, we anticipated that seasonal influences would be significant due to the pronounced seasonal contrasts in the hydrologic regime and manure applica-

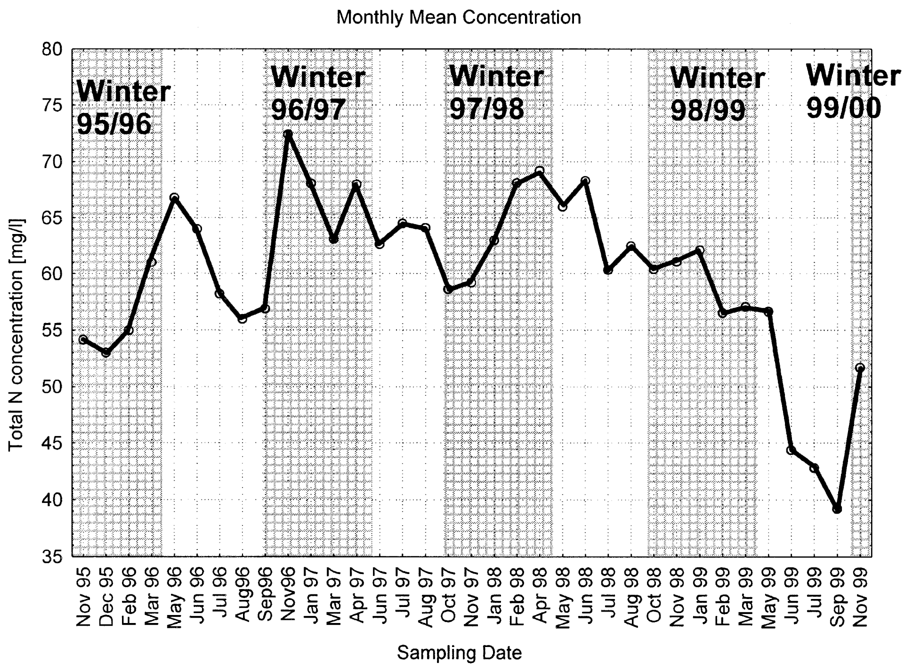

Fig. 5. Long-term behavior of N concentrations averaged for each sampling date.

tions (irrigation during the summer with significant crop uptake of nutrients, fall and winter land application of manure, winter rainfall). But seasonal averages of total N and EC in each management unit for September–November (fall), December–February (winter), March–May (spring), June–August (summer, main irrigation season) show only minimal and statistically not significant differences (Fig. 6).

4.3. Upgradient groundwater

The five wells classified as "upgradient" (i.e., upgradient of the dairy fields and corrals) average 23.5 mg/l N (all as NO_3–N; 168 observations), significantly lower than the on-site wells (64.3 mg/l), but over twice the drinking water standard of 10 mg/l. Individual observations at these five wells range from 1 to 77 mg/l. Similarly, the average electrical conductivity in upgradient wells is much lower (0.81 mS/cm) than in non-upgradient wells (1.89 mS/cm). The upgradient groundwater quality is primarily determined by neighboring activities. Two dairies are immediately downgradient from other dairy facilities. The remaining three dairies are surrounded by commercially fertilized forage fields and orchards.

4.4. Differences between dairies

Dairies *V* and *W* have a slightly shallower depth to water (4-year average: 2.4 m) than dairies *X*, *Y*, and *Z* (4-year average: 3.3 m). More importantly, the soils on V and

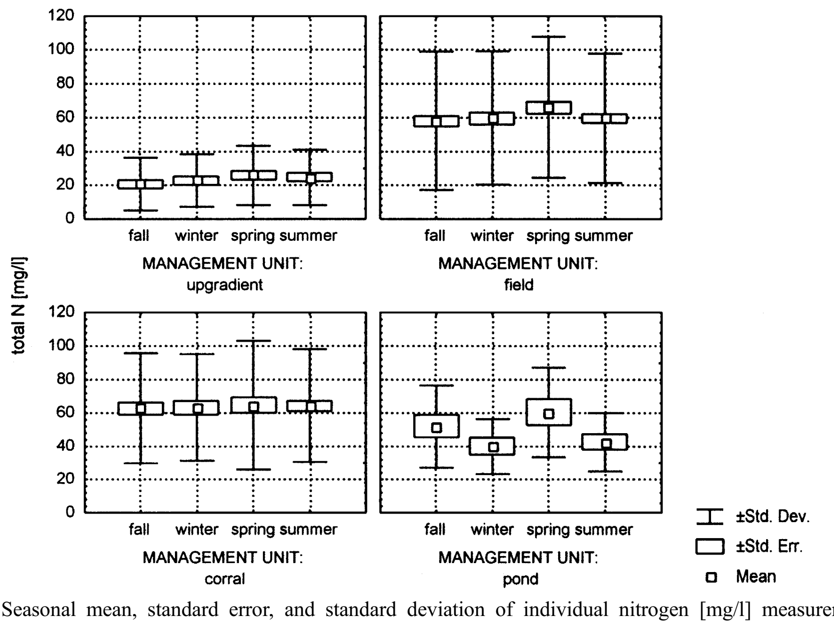

Fig. 6. Seasonal mean, standard error, and standard deviation of individual nitrogen [mg/l] measurements, grouped by time of sampling (fall, winter, spring, summer) and by dairy management unit.

W are predominantly fine sand, whereas soils on *X*, *Y*, and *Z* are predominantly sandy loams. However, the differences in groundwater nitrogen concentrations and EC between the five dairies (not including upgradient wells) are small compared to the spatial and temporal variability of concentrations within each dairy (Fig. 7a,b). The mean N concentrations averaged on individual dairies range from 45 to 81 mg/l, the mean EC from 1.6 to 2.1 mS/cm, but the ranges of concentrations found within each dairy overlap considerably. Based on the ANOVA of the $39 m_i$ not including those from the five upgradient wells, neither total N nor EC are statistically different between dairies (Table 4).

On two dairies, tile drain networks were recently installed at a depth of 2.7 m. Each spans approximately 130 ha underneath and immediately adjacent to the overlying dairy. The average NO_3–N in the monitoring wells are in good agreement with those measured in the tile drain outlets, despite the relatively small number of observation wells per dairy (6–12): between April and November 1999, observed NO_3–N concentrations in the outlets of the two tile drain networks varied between 44 and 56 mg/l, while the monthly average NO_3–N concentration in the monitoring well networks varied from 45 to 64 mg/l. The agreement shows that the average nitrate measured with these shallow monitoring well networks of 6–12 wells is a meaningful representation of the average nitrate at the water table across the entire dairy.

4.5. Regional significance and comparison to other regions

The five dairies are only a small sample of dairies in the region and even between these five dairies individual differences exist in operations design and day-to-day manure management. Yet, no statistically significant differences are observed in the overall nitrate and salt load near the water table. Since their manure management practices are considered representative or even progressive for the region, and because hydrogeologic and pedologic conditions are similar throughout the study area, it is reasonable to assume that salt and nitrogen loading to the water table occurs at similar levels on the other dairies. This is confirmed by observations in tile drain networks and shallow drainage wells, where NO_3–N levels have been found to be similar magnitude if the dominant land use in the drainage area is dairying.

Only few other surveys of similarly shallow groundwater underneath dairies or comparable AFO exist for comparison. In a national survey of water table water quality in the Netherlands, the national average for nitrate-N measured at the water table underneath AFOs sites located on well drained, predominantly sandy soils is comparable to that found in our study (1992 average: 55 mg/l; Oenema et al., 1998). A shallow monitoring well survey of farmsteads in Ontario yielded lower nitrate levels than observed at our sites, however the survey included predominantly sites with finer textured soils (Rudolph et al., 1998). More importantly, dairies in both regions (Netherlands and Ontario) employ significantly different water and manure management methods reflecting temperate, moist climate conditions: crops are not irrigated, manure is generally applied with mechanical spreaders, and cattle are often raised on pasture at relatively low animal densities. In those regions, annual variations in N concentrations at the water table are highly dependent on precipitation amounts and the timing of manure

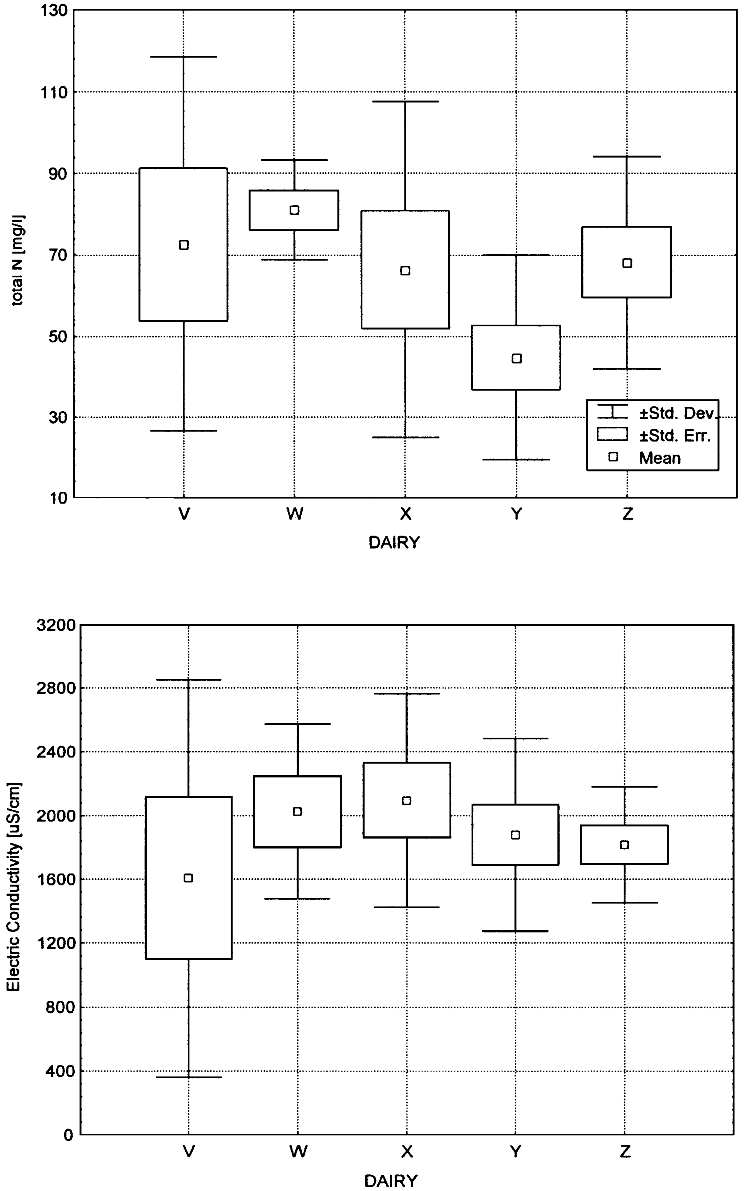

Fig. 7. (Top) Mean, standard error of the mean, and standard deviation of the 4-year average well N concentrations, grouped by dairy (not including upgradient wells). The number of non-upgradient wells in dairies *V–Z* 6, 6, 8, 10, and 9, respectively. Each well's average N is assumed to be one independent datum for computing standard errors and standard deviations. For ANOVA results, see Table 4. (Bottom) Mean, standard error of the mean, and standard deviation of the 4-year average well electrical conductivity. Same water sample population as above.

application relative to rainfall events. In contrast, the irrigated dairies of our study are subject to highly controlled water management, where irrigation and nutrient management are considered the primary control on N leaching (Schwankl et al., 1996; Mathews et al., 1999).

4.6. Nitrogen and salt leaching from dairy management units

Fig. 8 shows a statistical profile of total N and EC across management units. Management units are ordered along the main groundwater flow path through the dairies. Hydrologically, the profile is only an approximation, because the monitoring wells on individual dairies are not located along a single groundwater flow path. Also, ponds—while always adjacent to corrals—may be located either downgradient of upgradient of the corral area.

4.6.1. Fields

Because of the size of the fields (>100 m) and the shallow depth of the well (<10 m), the resulting source area (1) is rarely larger than the field. Monitoring well concentrations in "field" wells therefore represent undiluted water quality of field recharge water. The average NO_3–N concentration (62 mg/l) and EC level (1.6 mS/cm) of all (upper and lower) "field" wells are used to compute an estimate of the nitrogen and salt loading per hectare. At standard conversion rates (Hem, 1985), 1.6 mS/cm represent a total dissolved solids (TDS) concentration of approximately 960 mg/l. From the low efficiency of border flood irrigation (50–70%), from local climate data, and from irrigation application data recharge is estimated to be at least 0.45 m/year. From these data it follows that on average at least 280 kg NO_3–N/ha and 4300 kg salt/ha has been recharged to groundwater from these dairy fields in each of the last 4 years.

The field surface application itself is higher, because the NO_3–N and TDS in groundwater recharge does not include N and salt uptake in the crop and N losses due to volatilization at the soil surface or due to potential denitrification in the vadose zone. Conservatively assuming the following N and salt sinks prior to groundwater recharge: N volatilization losses of 20% of the amount applied (Van Horn et al., 1994), denitrification losses of 10% for well-drained soils, annual plant N uptake for the double cropped corn-grained system of approximately 500–600 kg/ha (Mathews et al., 1999); and annual plant salt uptake of 1200–1400 kg/ha (Karlen et al., 1988; CFA, 1995). Adding these sinks to the groundwater loading, the gross annual N application at the land surface is estimated to be on the order of 1000 kg N/ha, and the gross annual salt application is estimated to be on the order of 5500 kg/ha. These application rate estimates are consistent with recent manure nutrient and salt measurements in irrigation water considering the amount of liquid manure applied (Mathews et al., 2001).

4.6.2. Corrals

Mean total N does not significantly vary between the dairy management units (Fig. 8a, Tables 2–4). The spatio-temporal variability of total N (coefficient of variation of all observations) is also similar for all management units except corrals, which have slightly smaller variability. Based on N measurements alone, it is therefore not possible to

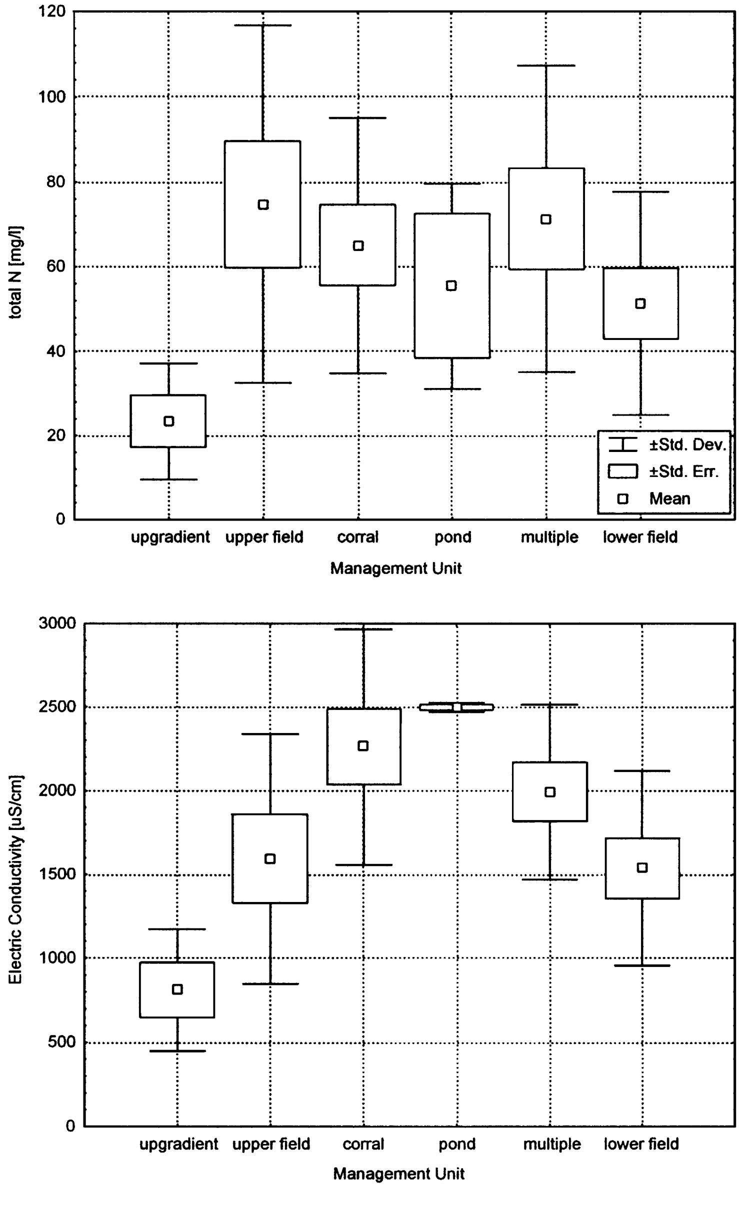

Fig. 8. Mean, standard error, and standard deviation of the 4-year average well nitrogen (top) and average well electrical conductivity (bottom) grouped by the dairy management units. From left to right, this yields a statistical profile along the general groundwater flow path. For ANOVA results, see Table 4.

determine whether groundwater underneath corrals and ponds is recharged from those management units or whether it is laterally transported to those areas from the upper field recharge areas.

In contrast, the mean EC in corrals and ponds is significantly higher than under the fields (Fig. 8b, Tables 3 and 4). The difference in EC indicates that the shallow groundwater underneath corrals and ponds is recharged, at least in some locations, directly from the corral and pond area and not in upgradient fields. When grouped not only by management unit, but also by dairy, corral EC is significantly higher than field EC on three of five dairies. Corral leaching is further confirmed by the presence of elevated N concentrations in corral wells of dairies where nitrate and salinity in groundwater immediately upgradient of the corral area is low, e.g., on dairy *Y* (Fig. 9). It is not clear, to which degree high EC underneath corrals may be the result of past vs. current management practices.

Our observations of EC and N levels underneath the corral areas are consistent with results in the few other existing studies of corral leaching. For example, higher EC underneath corrals, when compared to fields, is consistently measured in electromagnetic subsurface imaging surveys on dairies in Georgia, California, and elsewhere (Drommerhausen et al., 1995; Brune et al., 1999). The NO_3–N concentration in a small monitoring well network on three Georgia dairies ranged from 47 to 135 mg/l in the shallow-most groundwater (well completion to 1.5 m below water table, which ranged from 5.6 to 8.7 m). This range is surprisingly similar to that found in our study, even though the local shallow aquifer in that study consists of a highly weathered, clayey to loamy saprolite above fractured gneiss and granite bedrock (Drommerhausen et al., 1995). However, as in our study, corral N concentrations there are similar to those at adjacent fields making it difficult to assess the source of the corral groundwater based on nitrate measurements alone.

4.6.3. Ponds

TKN provided an additional water quality variable to discriminate between recharge water from different management units, specifically in the pond area. TKN was determined for all samples from 1995 through August 1997 (16 sampling dates). Beginning in October 1997, TKN was measured on only 16 of 44 wells, because of non-detects elsewhere. Results from the quality assurance program led us to conclude

Table 3
Mean and coefficient of variation of EC, total N, and TKN for the various management units using the 4-year mean at individual wells as a sample datum

Management unit	No. of wells	EC [μS/cm]		Total N [mg/l]		TKN [mg/l]	
		Mean	C.V.	Mean	C.V.	Mean	C.V.
Upgradient	5	810.0	0.45	23.3	0.59	2.7	0.17
Upper field	8	1593.0	0.47	74.7	0.56	3.5	0.24
Corral	10	2262.2	0.31	65.0	0.46	3.3	0.28
Pond	2	2497.3	0.01	55.4	0.44	36.8	0.15
Multiple	9	1991.9	0.26	71.3	0.51	4.5	0.55
Lower field	10	1537.2	0.38	51.3	0.51	2.5	0.25
Corral and pond	12	2301.4	0.28	63.4	0.45	8.9	1.47
Upper and lower fields	18	1562.0	0.41	61.7	0.57	3.0	0.30
All	44	1766.2	0.42	59.8	0.57	4.9	1.48

Table 4
Results of the one-way analysis of variance for various effects

Grouping variable (effect)	Effect (grouping variable)	Dependent variable	*df* effect	MS effect	*df* error	MS error	*F*	*p*	Effect statistically significant ($p<0.05$)
Dairy	dairy: *V*, *W*, *W*, *Y*, *Z*	EC [μS/cm]	4	$2.43E\times10^{5}$	34	$4.93E\times10^{5}$	0.49	0.741	
	(excluding data from	total N [mg/l]	4	$1.57E\times10^{3}$	34	$1.02E\times10^{3}$	1.54	0.214	
	"upgradient" wells)	TKN [mg/l]	4	$5.58E\times10^{1}$	34	$5.87E\times10^{1}$	0.95	0.447	
Management units	mgmt unit: upper field,	EC [μS/cm]	3	$1.35E\times10^{6}$	26	$4.38E\times10^{5}$	3.08	0.045	✓
	corral, pond, lower field	total N [mg/l]	3	$8.65E\times10^{2}$	26	$1.06E\times10^{3}$	0.82	0.495	
	(samples from all "RWQCB" wells)	TKN [mg/l]	3	$7.06E\times10^{2}$	26	$1.77E\times10^{0}$	398.77	0.000	✓
Management unit	mgmt unit: upper field,	EC [μS/cm]	3	$2.68E\times10^{6}$	48	$3.25E\times10^{5}$	8.25	0.000	✓
	corral, pond, lower field	total N [mg/l]	3	$6.53E\times10^{2}$	48	$7.91E\times10^{2}$	0.83	0.486	
	(samples from all wells)	TKN [mg/l]	3	$7.20E\times10^{2}$	48	$1.14E\times10^{0}$	632.68	0.000	✓

df: Degrees of freedom; MS: mean square; *F*: F-statistic; *p*: significance level.

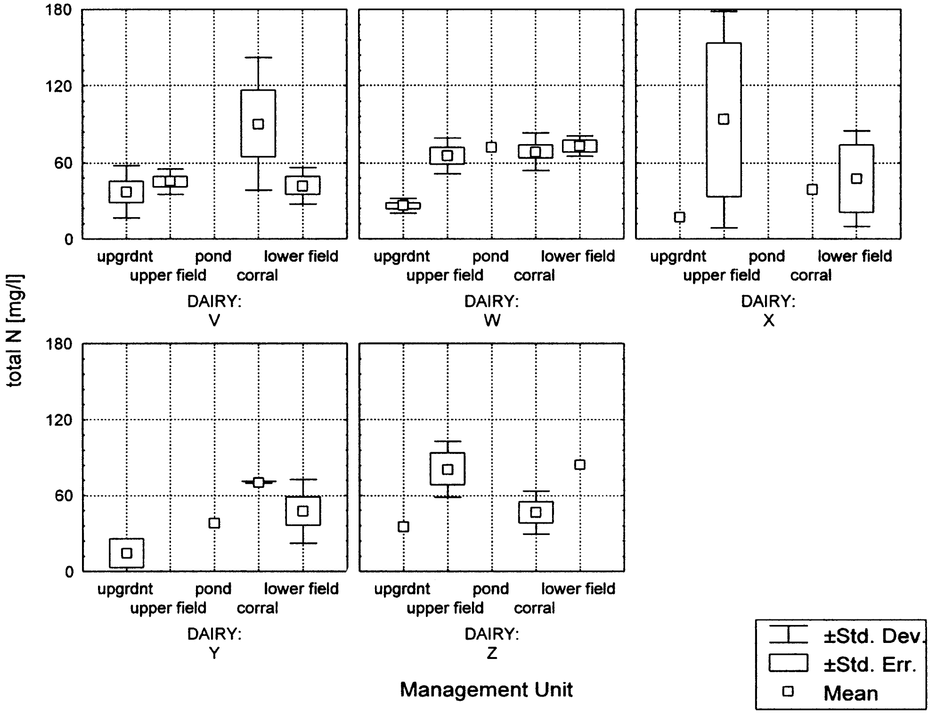

Fig. 9. Statistical nitrogen profile along groundwater path through individual dairies. Except on *V* and *W*, most groups include only one to three wells (here, dairies *V* and *W* include 35 additional wells drilled in 1999, yielding smaller standard errors).

that the analytical method used did not allow for proper TKN measurement at levels less than 5 mg/l, Also, on four sampling dates (5th, 6th, 11th, and 35th sampling dates), TKN data were considered unreliable. Of 708 valid TKN analyses on 31 sampling dates, 70 samples showed measurable levels (>5 mg/l) of TKN averaging 26.1 mg/l (std. dev.: 17.7 mg/l). The remaining 638 samples averaged 1.9 mg/l TKN (std. dev.: 1.1 mg/l), which is considered to be statistical noise due to measurement and analytical error.

Four wells had measurable TKN more than once or twice. The four wells account for 90% of all measurable TKN data. At one of these four wells (management unit: field), five of six measurable TKN concentrations were less than 10 mg/l. The remaining three wells with measurable TKN are all located within the downgradient outside slope of the berms of three separate ponds. Two of these wells are classified as pond wells, one as a well with multiple source areas (at the edge of a field, but immediately downgradient from a pond). The long-term averages of measurable TKN concentrations in the three wells are 44.6, 35.5, and 14.2 mg/l respectively. Less than measurable TKN was found on only 1 of 11, 2 of 30, and 12 of 28 sampling dates, respectively. For comparison, nitrate-N at the three wells average 34.7, 5.3, and 88.0 mg/l, respectively (all sampling dates). Beside these three wells, only one other well exists in the downgradient berm of a (fourth) pond. No measurable levels of TKN were found at that well (grouped as a well with ambiguous source area). Consideration of the actual source areas based on the hydraulic analysis (1) proved important in classifying the wells by management unit: four wells located adjacent to but upgradient of ponds did not show any TKN detections.

While the number of wells with measurable TKN data is small, the consistent long-term occurrence of TKN at pond locations is significant (Tables 3 and 4). Physico-chemical considerations also identify the pond as the likely TKN source: measurement of TKN is always accompanied by the absence of dissolved oxygen in the sample water. TKN (predominantly as NH_4, according to lab results), while stable under such anoxic conditions, is strongly sorbed. This precludes any long-distance transport (more than several tens of meters) of TKN. Furthermore, the entire N pathway in the subsurface must be anaerobic to inhibit nitrification. TKN therefore cannot originate from irrigations in upgradient fields. The distance to these fields is on the order of 100 m and more, the well-drained irrigated field soils are frequently aeraeted, and no TKN is found in "upper field" wells.

The ponds, 10–30 years old, have generally been considered self-sealed based on early experiments (Meyer et al., 1972). From the TKN levels we conclude that some leakage occurs, possibly because of macropores and fractures in the clay liner after pond water levels have been low or, at very low rates, through the liner itself as shown by McCurdy and McSweeney (1993) and by Ham and DeSutter (1999). A lower bound for the net leakage rate can be estimated using Eq. (2). We assume that d_{TKN} is at least 5 m, corresponding to the depth of the well. Pond recharge must be on the order of 0.8 m/year or more, at least locally. This value is on the same order as the generally recommended allowable seepage rate of 0.3 m/year (USDA, 1997) and at the low end of the range of leakage rates reported elsewhere (Korom and Jeppson, 1994; Ham and DeSutter, 1999; see Table 1). Resulting groundwater N concentrations are not nearly as high as observed in some studies (Ritter and Chirnside, 1990; Korom and Jeppson, 1994; see Table 1) and are significantly lower than the TKN concentrations observed in the stored liquid manure (from 200 mg/l to over 1000 mg/l, Mathews et al., 2001). EC levels in the pond wells (mean: 2.5 ms/cm) are only slightly higher than in the neighboring corral wells (Table 3, Fig. 8b).

4.7. Comparison to production well water quality

In contrast to shallow groundwater quality, NO_3–N in production wells operated on the five project dairies ranges from less than 1 to only 31 mg/l with an average of 12 mg/l. The wells are drilled to depths of approximately 40–60 m. Well screens are typically installed from 15 m on downward. These concentrations are similar to those found in a sub-regional groundwater survey of the 90 km^2 Hilmar dairy area (Lowry, 1987; see Table 1). Approximately 60 dairies and 3 poultry farms are located in that area of which almost 85% is cropland (primarily corn, winter grain, alfalfa, and almonds). In the 1980s, at least half of the crop acreage received animal waste. Two of the five project dairies are located within that study area. The difference between NO_3–N levels observed in the shallow groundwater monitoring network and those observed in the deeper production well is attributed to the following four potential factors.

- Export of a significant amount of shallow high nitrate water from tile drainage and shallow drainage wells to lined irrigation canals.
- Denitrification below the water table due to interbedded fine-grained sediments in the alluvial aquifer, as documented in other regions, for example, by Spalding et al. (1993).
- Dilution with ambient groundwater of better quality recharged on the dairy in fields without manure applications, or with off-site groundwater. The dilution partially occurs in the aquifer, but also occurs in the production wells, which typically pump water across an aquifer thickness of several tens of meters.
- Significantly longer travel time to the deeper production wells when compared to the shallow monitoring wells. Deeper production wells reflect recharge water quality several years to several decades ago, at a time when animal densities in this region were significantly smaller.

Nitrate levels in the Hilmar survey are similar to those observed in other California dairy regions with deeper water table and less permeable soils (Boyajian and Ross, 1998; Wildermuth Environmental, 1999; see Table 1).

5. Conclusion

Data collected during the past 7 years confirm that shallow groundwater quality below dairies with irrigated forage crops is degraded by high levels of nitrate and salts. The spatial and temporal variability of nitrate and TKN, but also of EC is found to be large. The exact location and extent of the source area of individual monitoring wells is difficult to determine in practice, although some useful estimates can be made. As a consequence, we find that—short of highly controlled field experiments—explanations for individual events in the water quality history of individual monitoring wells are highly speculative, particularly in the corral and pond area. Under those conditions, a multi-year monitoring effort across several dairies, and the classification of the wells by source areas (management units) via a simplified hydraulic analysis provided a useful larger scale assessment method and the data for a statistically meaningful analysis.

The assessment demonstrates that past manure applications to forage fields have the most impact on shallow groundwater quality due to the area of the manure-treated land when compared to the size of corrals and ponds (approximately 90% of each dairy property is crop land). The impact of current corral and pond designs on shallow groundwater quality remains less certain and is difficult to distinguish against the background N levels created by manure applications in surrounding fields and against historic loading from prior corral and pond management practices. Leaching from corrals can be inferred from elevated EC levels associated with corral areas and from the nitrogen profile across a subset of the five investigated diaries. Elevated levels of TKN found at the outside edge of three of the five ponds indicate that ponds are not impermeable and are subject to leaching at a rate on the order of nearly 1 m/year, even after many years of operation (no complete self-sealing). This leaching rate is consistent with federal and state guidelines for the design of pond liners. Other geochemical analysis (ion ratios, N isotope ratios) were not considered in this study, but may be a helpful tool in further evaluating leaching rates from ponds and corrals.

Across all dairy management units, overall nitrate levels observed near the water table (at depths to less than 10 m) are within the range of observations reported in similar AFO manure studies of deep soil N leaching and recharge water quality. More importantly, production well water quality at depths of 15–60 m below the land surface is significantly better than that observed in our shallow monitoring well network and comparable to levels in production wells of other California dairy regions, despite the high hydrogeologic vulnerability of our study area. But long-term impacts of past management practices and of recent changes in nutrient loading brought about by improved management practices remain to be assessed. Clearly, proper nutrient management with land applied liquid manure is the most critical factor in protecting long-term groundwater quality.

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