

Prediction of Groundwater Nitrate Contamination after Closure of an Unlined Sheep Feedlot

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ABSTRACT

Nitrate contamination of groundwater by a sheep feedlot in Hawke's Bay, New Zealand led to closure of the feedlot in 1998. However, knowledge of the processes controlling how long the contamination will remain and an analysis of whether the current land use (vineyard) will also impact groundwater quality are required to assess long-term groundwater quality issues after feedlot operations cease. To determine the fate of NO_3^- following land use changes and the rate of reduction of contamination that may be expected under natural conditions following such changes, we compared the chemical concentrations of NO_3^- -N, Cl, and alkalinity (HCO_3^-) in the groundwater and rivers from surveys conducted in 1994 and 1995 with sampling conducted in 2001. Profile sampling of total N and C of the <2-mm size fraction in the vadose zone from two sites was used in a one-dimensional soil-plant-atmosphere system model (SPASMO) program to predict N loading in future years and predict how long it would take to improve the groundwater quality. Groundwater sampling in 1994 and 1995 determined that the highest NO_3^- -N concentrations were under the feedlot ($>140 \text{ g m}^{-3} \text{ NO}_3^-$ -N) and down gradient. In 2001, 3 yr after the feedlot closed, the Cl concentrations had increased in down-gradient wells but remained similar to the 1994 survey in other wells. There has been a decrease in NO_3^- -N concentrations in most wells, compared with the peak NO_3^- -N concentrations recorded in the 1995 survey, but an increase compared with 1994. Alkalinity concentrations in wells located within the influence of the feedlot are approximately 150 g m^{-3} lower than in surrounding wells. This indicates that nitrification reactions are affecting the HCO_3^- concentrations in the feedlot-influenced wells. However, the HCO_3^- concentrations of some of these wells are increasing, indicating that nitrification could be slowing down and the aquifer is beginning to recover. SPASMO modeling indicates that NO_3^- contamination from the site will continue for the next 3 to 5 yr. The impact of NO_3^- leaching due to current land use practices is likely to be much less than the feedlot. The model predicts there will be an improvement in groundwater quality in the next 3 to 5 yr as NO_3^- from the feedlot eventually leaves the vadose zone profile and mixes into the unconfined aquifer.

CONFINEMENT OF LARGE NUMBERS of animals in feedlots can result in large loadings of animal feed- and waste-related substances to the environment, and the consequences of waste-management practices at feedlots on ecosystem viability and human health are poorly understood (Wilde et al., 2000). Potential effects of feedlots on the quality of surface water, groundwater, and

air, as well as the implications of such effects on human health, pose issues of international concern that require science-based assessment and response. Most of the research concerning contamination from feedlot wastes concentrates on cattle (*Bos taurus*) and swine (*Sus scrofa*) feedlots (e.g., Arnold and Meister, 1999; Ham and DeSutter, 2000). Relatively little work on groundwater contamination from sheep (*Ovis aries*) feedlots has been conducted globally.

The proper management of feedlot wastes has been an issue in the USA and Australia for a number of years (Gilbertson et al., 1981; Young et al., 1982, 1994), but this problem was only recently identified in New Zealand. The wastes are of major concern because they carry significant nutrient loads of both N and P, as well as some heavy metals such as Cu, Zn, and Fe. In New Zealand, the concentration of NO_3^- -N in groundwater is one of the main concerns with feedlot waste.

For most of the history of European settlement in New Zealand, agriculture-based industries such as dairy and sheep farming have been dominant land uses. Except for small, localized areas on farms, such as sheep and cattle dips for the control of parasites (Hadfield and Smith, 1999), most of the environmental problems related to water quality have been as diffuse, nonpoint source issues related to overgrazing or fertilizing. Changes to farming practices in the 1980s brought about by international market pressures led to the introduction of cattle and sheep feedlots in New Zealand that are used to grain-feed animals before either slaughter or live shipment overseas (Young et al., 1994). Because of this trend, pollution problems related to the livestock industry now include point-source specific problems related to feedlot waste runoff and its impact on groundwater.

From July 1986 to 1998, a sheep feedlot was established for weaning the sheep off grass and onto pellets in preparation for live sheep exports. The feedlot, located approximately 2 km southwest of Maraekakaho Township, was about 1 km^2 and held up to 80 000 sheep for up to 10 d at a time. Rosen and McNeill (1996) described the feedlot as unlined, having no provision to contain run-off or run-on and had little vegetative cover. Such intense stocking of the land was shown to have an adverse impact on the shallow unconfined aquifer under the feedlot (Rosen et al., 1995; Rosen and McNeill, 1996; Rosen, 1996), primarily because of the large quantity of sheep excretions. This was demonstrated by measured NO_3^- -N concentrations of $>140 \text{ g m}^{-3}$ and Cl concentrations $>100 \text{ g m}^{-3}$ in the groundwater down gradient of the feedlot (Rosen and McNeill, 1996). The documentation of NO_3^- concentrations in the groundwater that ex-

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Abbreviations: HBRC, Hawkes Bay Regional Council; SR, stocking rate.

ceeded the New Zealand drinking water standards by more than 10 times the maximum acceptable value of $11.3 \text{ g m}^{-3} \text{ NO}_3\text{-N}$ led the Hawkes Bay Regional Council (HBRC) to impose stricter conditions for the operation of the feedlot. The owners of the feedlot decided to close the operation in 1998 rather than meet the new conditions.

The land once used as the sheep feedlot was turned into a vineyard in July 1998 and the HBRC became concerned about land use intensification in the region. Areas of particular interest were (i) to gain an understanding of the fate of NO_3 contamination following land use changes and (ii) to determine the rate of reduction of contamination that may be expected under natural conditions following such changes. The purpose of this study was to examine the current state of the groundwater under the old feedlot and to determine the potential for contaminants that are held in the soil above the groundwater table to be transported into the groundwater system. In particular, we characterized the quantity of mineralizable N in the unsaturated zone that could continue to act as a potential source of NO_3 contamination under the former feedlot. We use this information to predict the concentrations of NO_3 in the groundwater during the next 5 yr using a simple one-dimensional model. This enables us to generically evaluate the impact of feedlots on alluvial groundwater systems after closure and to determine the contribution of present land uses to the NO_3 concentrations in groundwater. This type of information is essential for evaluating the risks of confined sheep feeding operations on long-term groundwater contamination.

MATERIALS AND METHODS

Location and Setting

Maraekakaho is a small farming settlement near the confluence of the Maraekakaho and Ngaruroro Rivers, approximately 30 km southwest of Napier in the Hawkes Bay region of New Zealand's North Island (Fig. 1). Rosen et al. (1995) described the geology of the Maraekakaho area as Pliocene to Holocene-age sedimentary fluvial and marine deposits. Greywacke gravel terraces formed and reworked by the Ngaruroro River dominate the Maraekakaho area. South of the Maraekakaho area stands a Neogene-age limestone hill, which is located on the south side of Maraekakaho Stream.

Soils

Soil information was obtained from data in New Zealand Soils Database (Hewitt, 1998). The Takapau A series silt loam is the dominant soil in the area of the feedlot. This soil is derived from the parent Ngatarawa sandy loam. This is a well-drained Fluventic Udic Ustochrept soil. The parent material is a fine greywacke alluvium with minor tephra over greywacke gravels that have a stone fraction of approximately 85% below 0.5 m.

Hydrology

Groundwater is the main source of water for domestic, school, and farm use. Groundwater levels range from about 110 m above mean sea level in the west to 83 m above mean sea level to the east. The groundwater table is about 15 m below the ground surface under the former feedlot, but may

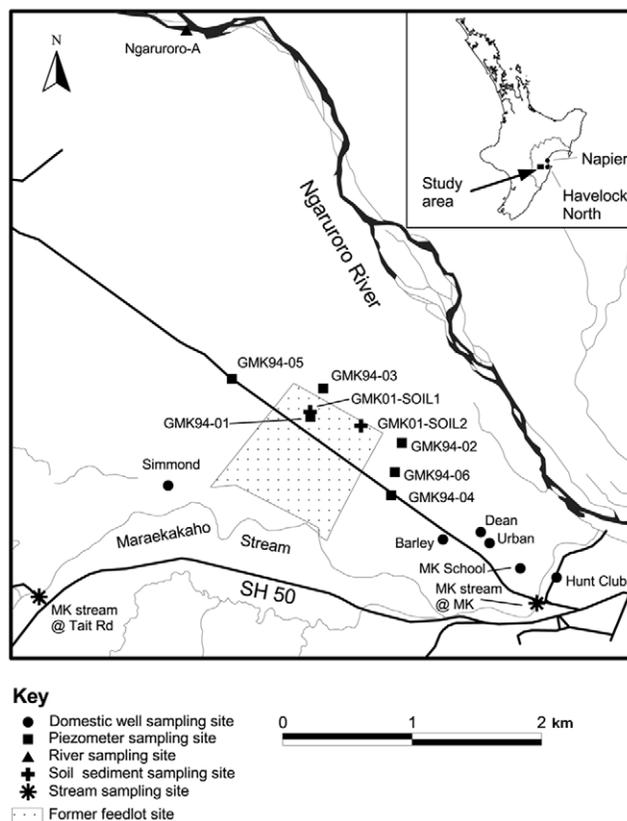


Fig. 1. Maraekakaho area showing the former feedlot site and sampling sites.

be <5 m below the land surface to the east. Rosen (1996) mapped groundwater flow as toward the east-northeast (toward the Ngaruroro River). These measurements were based on groundwater level measurements and elevations of the Maraekakaho Stream and Ngaruroro River.

Rosen and McNeill (1996) showed that the average gradient of the stream water level in Maraekakaho Stream is approximately 0.005 in the study area. This compares with the average Ngaruroro River gradient of 0.004 at the same distance above the confluence with the Maraekakaho Stream. A higher gradient in the Maraekakaho Stream means that, at similar distances upstream of the confluence, Maraekakaho Stream water levels are above those of the Ngaruroro River. The piezometric contours estimated from river and groundwater levels indicate that groundwater is traveling generally toward the Ngaruroro River and away from the Maraekakaho Stream in the study area. Any water leaking from the Maraekakaho Stream, if it leaked out of the active channel, would tend to travel underground to emerge in the Ngaruroro River channel. Stream gaugings on the Maraekakaho Stream show that it loses approximately 80 L s^{-1} of flow from Tait Road to near its confluence with the Ngaruroro River. Surface flow is completely lost between the Tait Road Bridge and a location near a gauging site. If approximately 80 L s^{-1} of flow is lost from the Maraekakaho River below Tait Road Bridge, then this water, assuming no extraction, will travel through the gravels in the direction of the Ngaruroro River.

Land Use

The feedlot was in operation from 1986 to 1998. Seven residents and landowners were interviewed about past and current land use practices around the former feedlot to deter-

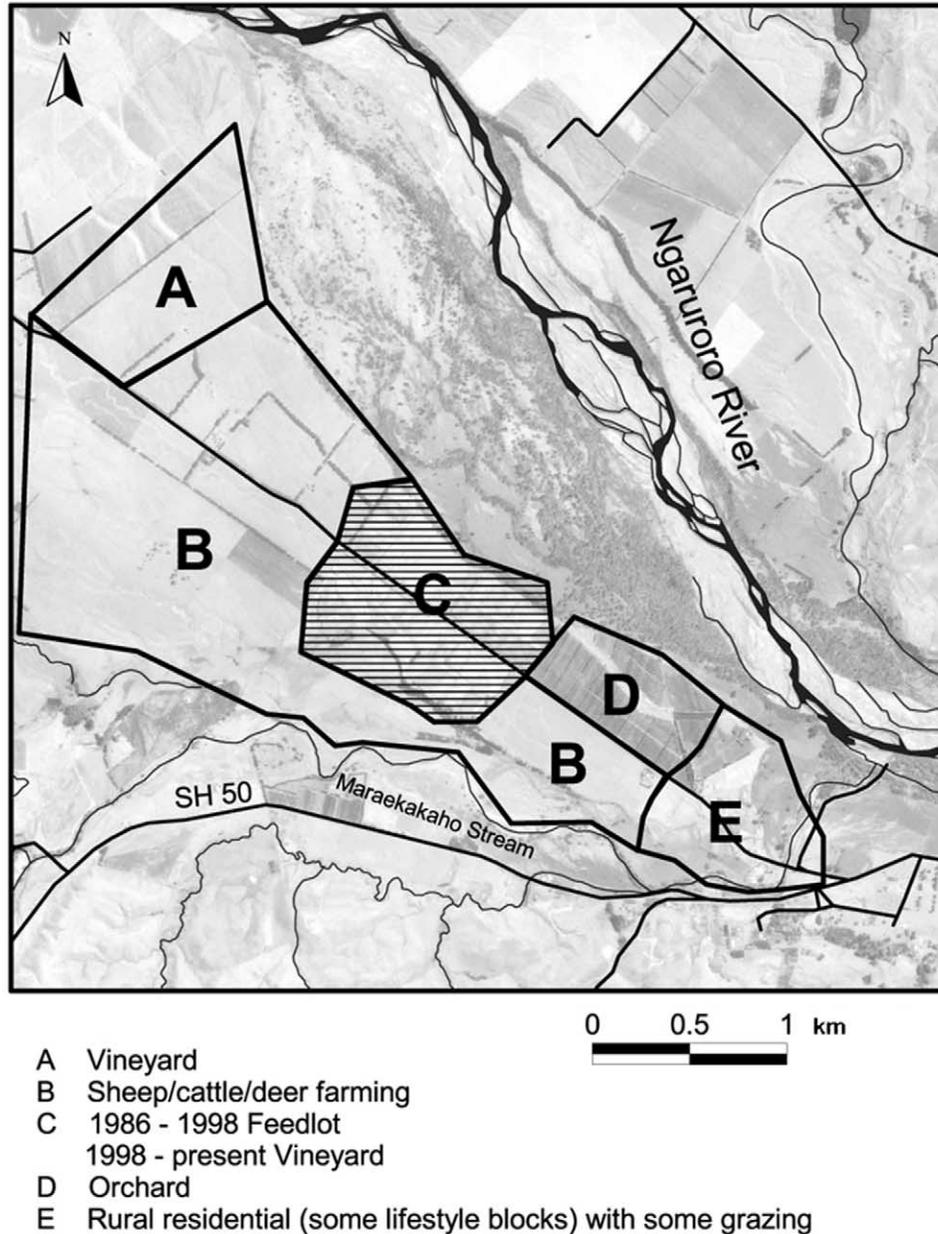


Fig. 2. General land use types in the area. Area C includes both the feedlot land use and the current vineyard land use and so appears differently than in Fig. 1.

mine if any observed groundwater chemistry changes were caused by differences in land use that occurred between the 1995 groundwater sampling (Rosen and McNeill, 1996) and the most recent (2001) sampling.

Figure 2 shows general land uses in the Maraekakaho area at the time the feedlot was in operation. Cattle, deer (*Cervus elaphus*), and sheep farming dominated the area, with a small residential area near the Maraekakaho Stream Bridge. When the feedlot was operating it was located approximately 2 km up Keru Road from the Maraekakaho Bridge. An orchard occupies a large area southeast of the former feedlot and borders the Ngaruroro River. A vineyard was established around 1995 to 1996; it is approximately 1.5 km northwest of the former feedlot. Before this, the land use was sheep and cattle grazing.

Not many land use practices have changed since 1995 in the Maraekakaho area. The most notable change was conver-

sion of the feedlot to a vineyard (Zone C, Fig. 2) in July 1998. Also, there has been some additional residential development near the Maraekakaho Bridge (Zone E, Fig. 2). These changes have several implications for groundwater quality. Although sheep are not contributing new nutrient inputs to the feedlot since the conversion of the feedlot to a vineyard, the vines do require fertilizer and minerals. However, this nutrient input is considered small relative to the feedlot input. Rainfall is probably the main process driving the movement of nutrients through the soil profile. The long-term average rainfall in the Maraekakaho area is approximately 800 mm yr⁻¹ (Rosen and McNeill, 1996), with less in summer months. Irrigation to a vineyard probably will not increase drainage and movement of nutrients through the soil because it is used only to replace water uptake by the vines. The vines are irrigated at a rate of 2.7 mm plant⁻¹ every second day during the summer (December–March); however, water is applied only when the

Table 1. Summary of well depths for the wells sampled in this study.

Site	Depth	Screen length
	(from ground surface)	(above 1 m sump)
m		
GMK94-01		
Shallow	16.4	5
Intermediate	19.7	1
Deep	27	1
GMK94-02		
Shallow	8.5	5
Intermediate	12.5	1
Deep	17.5	1
GMK94-04		
Shallow	11.4	5
Intermediate	17.6	1
Deep	22.5	1
GMK94-05	21	5
GMK94-06	13.8	5
Hunt Club	unknown	unknown
Urban	unknown	unknown
Dean	unknown	unknown
Barley's	unknown	unknown
Simmond	unknown	unknown
School	unknown	unknown

plants need it. In dry years the vineyards may be irrigated almost every day, in wet years there is very little irrigation. A rate of 2.7 mm d⁻¹ is the HBRC-set limit for irrigation to grapes (*Vitis* spp.). The long-term average irrigation rate for the vineyard is approximately 69 mm yr⁻¹.

All residential houses are on septic tanks. An increase in the number of houses near the Maraekakaho Bridge also means an increase in the number of septic tanks. This may cause an increase in nutrients leaching into the shallow groundwater <5 m below land surface to the east of the former feedlot. New wells around the feedlot area have been drilled for commercial and domestic water supplies. One of these new wells has been drilled approximately 200 m southeast of site GMK94-02 and is used to irrigate the orchard. If improperly constructed, these new wells may provide preferential pathways for surface water to drain to the groundwater.

Groundwater Sampling

Characterization of groundwater quality beneath the feedlot before its opening is difficult due to lack of data (Rosen et al., 1995). The GMK94 series of piezometers in and around the feedlot were installed in May through June 1994 to obtain data directly below the feedlot and immediately up and down gradient (Rosen, 1996). Four of the GMK94 sites were drilled with three nested piezometers at different depths (Table 1) in the aquifer (Rosen and McNeill, 1996). Groundwater samples were collected from these piezometers and domestic wells in June, August, and October 1994, and in October 1995. These same wells were resampled in May 2001 to determine if groundwater quality has improved since the feedlot was closed in 1998.

Rosen et al. (1995) and Rosen (1996) reported data on a total of 12 groundwater sites, two stream sites on the Maraekakaho Stream, and historic data for the Ngaruroro River (Fig. 1). Six of the sites (GMK94 01-06, which included the nested piezometers) were drilled as monitoring wells for the project; the other groundwater-monitoring sites are domestic water-supply wells.

In this study, all 12 groundwater-sampling sites were located and, together with an upstream (of the former feedlot) Ngaruroro River site and upstream and downstream sites on the Maraekakaho Stream (Fig. 1), sampled in accordance with procedures outlined in Rosen et al. (1999). Groundwater sampling included purging piezometers and wells of at least three

Table 2. Chemical analyses of the May 2001 sampling. All analyses except pH in.

Location	pH	HCO ₃	Cl	NO ₃ -N	NH ₄ -N
GMK94-01					
Shallow	6.2	50	28	32	0.2
Intermediate	6.2	40	26	31	<0.01
Deep	6.2	35	26	26	<0.01
GMK94-02					
Shallow	6.2	44	79	49	<0.01
Intermediate	6.2	45	76	48	<0.01
Deep	6.3	45	68	45	<0.01
GMK94-03					
Shallow	6.5	79	21	6.8	<0.01
Intermediate	6.8	83	14	2.7	0.02
Deep	6.7	77	14	5.3	0.09
GMK94-04					
Shallow	6.2	43	39	26	<0.01
Intermediate	6.5	66	34	24	<0.01
Deep	6.6	73	23	15	<0.01
GMK94-05	6.3	46	14	10	<0.01
GMK94-06	6.3	70	81	48	0.02
Hunt Club	7.5	280	37	0.23	<0.01
Dean	7.2	195	27	4.4	<0.01
Simmond	7.6	188	19	0.67	<0.01
Barley's	7.4	172	28	8.6	<0.01
Urban	7.7	235	32	1.9	0.02
School	7.4	249	37	0.94	<0.01
Maraekakaho Stream, Maraekakaho	7.8	275	37	0.18	<0.01
Maraekakaho Stream, Tait Road	8.1	305	35	0.03	<0.01
Ngaruroro River	6.8	54	5.9	0.11	<0.01

times the casing volume. Samples were collected as close to the wellhead as possible in wells with existing pumps. A 250-mL, untreated, polyethylene bottle and a 100-mL field-filtered (0.45 µm) polyethylene bottle were filled at each site. All samples were chilled below 4°C and analyzed for Cl, NO₃-N, and NH₄-N using an ion chromatograph, and for alkalinity (reported as HCO₃) using an autotitrator (Table 2). Consistent results for repeat analyses and repeat sampling at daily intervals of NO₃-N and Cl for many of the samples indicates the reproducibility of our results. Tables of the results from all previous sampling can be found in Rosen et al. (1995), Rosen and McNeill (1996), and Reeves et al. (2001).

Soil and Sediment Sampling

Soil samples at two sites were collected by augering to 0.5 m below ground surface. A coarse gravel layer, which could not be augered, occurred at 0.5 m below ground surface. An air-rotary drill rig was used to drill from 0.5 to 12 m. The drilling technique forced the outer casing approximately 200 mm ahead of the drill bit to minimize sediment loss and to ensure samples were representative of the sediments at a specified depth. Sediments were blown up the inside of the casing, with compressed air, and into a plastic bucket. Drilling was done in 0.5-m intervals to a depth of 12 m at each site. Thus, sediment collected in the plastic bucket is considered to be a composite sample for the 0.5-m interval. At the end of each interval, a small random sample of sediment from the bucket was placed and stored in a sealed plastic bag. The bucket and the casing were cleaned with air before starting the next 0.5-m interval.

Samples from the <2-mm size fraction were analyzed for total N and total C with a Leco FP-2000 analyzer (Leco Corp., St. Joseph, MI) according to procedures described by Blake-more et al. (1987).¹ The sample was combusted at 1050°C in a

¹ The use of product names in this paper are for identification purposes only and do not imply endorsement by the USGS or any of the authors' institutions.

Table 3. Soil analyses of the GMK01-SOIL1 composite sample from the former feedlot site.

Site	Interval from	Interval to	Stones >2 mm	Total C	Total N	Sampling method
	m					
GMK01-SOIL1	0.08	0.1	18	9.31	0.91	augered
	0.2	0.3	1	2.99	0.29	augered
	0.5	1	36	0.79	0.07	drilled
	1	1.5	71	0.55	0.05	drilled
	1.5	2	71	0.14	0.01	drilled
	2	2.5	84	0.15	0.01	drilled
	2.5	3	74	0.11	0.01	drilled
	3	3.5	86	0.12	0.01	drilled
	3.5	4	75	0.11	0.01	drilled
	4	4.5	51	0.12	0.01	drilled
	4.5	5	73	0.08	0.01	drilled
	5	5.5	83	0.08	0.01	drilled
	5.5	6	79	0.08	0.01	drilled
	6	6.5	74	0.09	0.01	drilled
	6.5	7	68	0.07	0.01	drilled
	7	7.5	74	0.07	0.01	drilled
	7.5	8	80	0.08	0.02	drilled
	8	8.5	86	0.08	0.02	drilled
	8.5	9	82	0.10	0.02	drilled
	9	9.5	52	0.08	0.01	drilled
9.5	10	56	0.08	0.01	drilled	
10	10.5	57	0.08	0.01	drilled	
10.5	11	70	0.07	0.01	drilled	
11	11.5	82	0.09	0.02	drilled	
11.5	12	80	0.10	0.01	drilled	

stream of O₂, and the CO₂ produced was measured by infrared detection while the N gas was measured by thermal conductivity. The instrument was calibrated using standard EDTA and reference soil samples. Data from the soil sampling sites are presented in Tables 3 and 4.

Leachate Modeling (SPASMO Model)

A numerical modeling study was used to run the risk assessment model SPASMO described earlier (Green et al., 2001) to predict how much NO₃ leaching would occur under a sheep feedlot and a vineyard at Maraekakaho. The quantitative risk assessment model used 30 yr of daily weather data obtained

Table 4. Soil analyses of the GMK01-SOIL2 composite sample from the former feedlot site.

Site	Interval from	Interval to	Stones >2 mm	Total C	Total N	Sampling method
	m					
GMK01-SOIL2	0.08	0.1	1	7.74	0.77	augered
	0.4	0.5	0	2.78	0.23	augered
	0	1	62	0.91	0.08	drilled
	1	1.5	73	0.22	0.02	drilled
	1.5	2	73	0.12	0.01	drilled
	2	2.5	59	0.10	0.01	drilled
	2.5	3	72	0.09	0.01	drilled
	3	3.5	70	0.09	0.01	drilled
	3.5	4	75	0.09	0.01	drilled
	4	4.5	75	0.08	0.01	drilled
	4.5	5	82	0.09	0.01	drilled
	5	5.5	84	0.09	0.01	drilled
	5.5	6	81	0.08	0.01	drilled
	6	6.5	87	0.10	0.02	drilled
	6.5	7	81	0.08	0.01	drilled
	7	7.5	69	0.07	0.01	drilled
	7.5	8	85	0.08	0.02	drilled
	8	8.5	74	0.07	0.01	drilled
	8.5	9	79	0.08	0.01	drilled
	9	9.5	84	0.07	0.01	drilled
9.5	10	70	0.07	0.01	drilled	
10	10.5	80	0.07	0.01	drilled	
10.5	11	71	0.07	0.01	drilled	
11	11.5	75	0.08	0.01	drilled	
11.5	12	82	0.08	0.01	drilled	

from the HortResearch Centre at Havelock North (Fig. 1). The model used measured hydraulic and physical property data for the Takapau A series silt loam soil (see Soils above) that was obtained from data in the New Zealand Soils Database (Hewitt, 1998). Model output is presented to illustrate the amount and concentration of NO₃ that is likely to leach through the soil profile both before and after a change in land use from a sheep feedlot to a vineyard.

The computer model links the mechanisms of soil water flow through the root zone with the complex N transformations that result from natural processes, and those resulting from the application of N fertilizer, N uptake and recycling by the vegetation, and the returns of dung and urine from the sheep. The model calculates the balance and movement of water and N through a one-dimensional vertical slab of soil that extends from the surface to a depth of 7 m. The 7-m depth was chosen because it is likely to be above the water table depth, and SPASMO cannot handle dilution with groundwater. Although groundwater levels may be considerably deeper than 7 m, there is likely to be very little attenuation of NO₃ below 7 m. A description of the N mass balance portion of the model is given below and a full mathematical description of the SPASMO model is given in Green et al. (2001).

The flow of water through the soil profile is simulated using a capacity model (Hutson and Wagenet, 1993) in which soil water is divided into mobile and immobile phases. The mobile domain is used to represent the soil's macropores (e.g., old root channels, worm holes, and cracks), and the immobile domain represents the soil matrix. After any rainfall or irrigation events, water is allowed to percolate through the soil profile. However, this percolation only occurs when the soil slab is above field capacity. The infiltrating water first fills up the immobile domain and, once this domain is full, it then refills the mobile domain as the water travels progressively downward through the soil profile. Subsequently, on days when there is no significant rainfall, there is a slow approach to equilibrium between the mobile and immobile phases, driven by a difference in water content between the two domains.

The model is run on a daily time step, and the calculations are made in the following sequence:

1. Subtract evaporation, transpiration and N uptake from each soil segment.
2. Add and subtract the N involved in the various transformation processes.
3. Partition N between solution and sorbed fractions, assuming complete equilibrium between the mobile and immobile phases.
4. If there is rain or irrigation, then perform the leaching process.
5. Redistribute water and solute vertically, according to water potential and solute concentration.
6. Repeat the solute partitioning.

SPASMO solves the combined transport equations for water flow and the fate of mineral N that is assumed to comprise urea, NH₄-N, and NO₃-N both in solution and adsorbed onto the soils' clay and mineral particles.

The N transport component of SPASMO is based on a N balance equation that accounts for plant uptake, the application of mineral fertilizer, exchange and transformation processes in the soil, losses of gaseous N to the atmosphere, and the leaching of N below the root zone. The SPASMO model considers both the organic N (i.e., in soil biomass) and the mineral N (i.e., NH₄ and NO₃ in solution) contained in the soil and the plant biomass. Dissolved NO₃ is considered to be mobile and to percolate freely through the profile, being car-

ried along with the invading water. The movement of dissolved NH_4 is retarded as it binds to the mineral clay particles of the soil. The soil can receive inputs of organic C and N from decaying plant residues, which is added to the litter layer of the topsoil, and inputs of mineral fertilizer which are applied to the soil surface during the spring and summer periods. Animal urine is assumed to be in the form of urea that is applied uniformly to the pasture surface. Animal dung is recycled back to the soil's organic matter, where it slowly decomposes and releases mineral N in the form of NH_4 .

Pasture roots play a key role in the water and N dynamics of the root zone. The amount of soil N removed by the pasture is determined by the growth of the above- and below-ground dry-matter, multiplied by their respective N concentrations. Daily biomass production is modeled using a potential production rate per unit ground area, G ($\text{kg m}^{-2} \text{d}^{-1}$) that is related, via a conversion efficiency, ϵ (kg MJ^{-1}), to the amount of solar radiant energy, Φ ($\text{MJ m}^{-2} \text{d}^{-1}$), intercepted by the grass shoots using the equation:

$$G = \epsilon \Phi f_T f_N f_W \quad [1]$$

Here f_T , f_N , and f_W are response functions that range between zero and unity depending on temperature, plant N, and soil water status, respectively (Eckersten and Jansson, 1991). The value of G depends on the daily sunshine and temperature, plus the leaf-area index of the pasture, and is moderated by the soil's water and N status (King, 1993; Thornley et al., 1995). Pasture growth can achieve a maximum only if soil water and soil N are nonlimiting.

A simple allometric relationship is used to partition the daily biomass production into the growth of the foliage and roots. Plant biomass is expressed in terms of the balance between growth and senescence of the plant organs.

Allocation to the roots, α_R , depends on the leaf N content $[N]_F$, having a minimum value $[\alpha_{R0}]$ at a maximum leaf concentration $[N]_{F\text{max}}$, and increasing as N_L decreases (Eckersten and Jansson, 1991), and is written as

$$\alpha_R = \alpha_{R0} + 1 - \{1 - [(N]_{F\text{max}} - [N]_F)/[N]_{F\text{max}}\}^{0.5} \quad [2]$$

SPASMO accommodates seasonal changes in dry-matter allocation that are associated with changes in the nutrient status of the pasture. For simplicity, the corresponding seasonal changes in senescence rates, γ , have not been included in the model because we are concerned with the long-term consequences of these allocation patterns.

The model assumes that plant growth will achieve the maximum potential only if soil water and soil N (NO_3^- and NH_4^+) are nonlimiting. The net uptake of N from the soil is set equal to the amount of N incorporated into the new biomass minus the fraction of N that has been retranslocated, λ , from the old or senescing tissues.

The pasture's demand for N is set by the maximum N content of the root $[N]_{R\text{max}}$ and leaf $[N]_{F\text{max}}$ material. During active growth the pasture tries to supply the new leaf and root material with N corresponding to these maximum concentrations. The potential uptake of N from the soil, P_D ($\text{kg ha}^{-1} \text{d}^{-1}$) is defined as

$$P_D = (n_F G - \lambda_F \gamma_F F) [N]_{F\text{max}} + (n_R G - \lambda_R \gamma_R F) [N]_{R\text{max}} \quad [3]$$

This uptake can only be met if sufficient N exists in the soil. Otherwise both $[N]_F$ and $[N]_R$ will be reduced in low-N soils, and pasture growth will be curtailed.

The daily uptake of N from the soil is assumed to be proportional to the depth-wise distribution of the fine roots, and the total amount of NO_3 (N) and NH_4 (A) in each soil layer

(Johnsson et al., 1987). The potential uptake of NO_3 is calculated as

$$P_N = \min \left[\rho_R(z) \frac{N}{N + A} P_D, f_M N \right] \quad [4]$$

based on the relative root fraction in the layer, $\rho_R(z)$; the proportion of total mineral N as NO_3 , N (mg L^{-1}); and the total growth requirement for N, P_D . However, the actual uptake of NO_3 is limited to a fraction, $f_M = 0.10$, of the total NO_3 available in each layer. Ammonium uptake, P_A , is calculated in a similar way, by assuming P_A to be proportional to the relative amount of NH_4 in solution, A (mg L^{-1}).

If the total N uptake from a given soil layer is below the potential rate, P_D , then a compensatory increase in uptake is allowed from other layers deeper in the root zone (Johnsson et al., 1987). This is achieved by adding a fraction (c_{um}) of the deficit to the potential uptake from the next soil layers where more mineral N may be available. Surface roots are the most active (Clothier and Green, 1994) and they preferentially extract soil water and nutrients from the upper soil layers. As water and N stresses develop, the uptake action of roots typically switches to the deeper roots, because more water and nutrients are available there.

Allocation of the daily total N uptake to the new roots and leaves is based on the idea that roots receive N first, until they reach their maximum concentrations, and then that N is allocated to the leaves. Whenever soil N becomes limiting, a reduction factor f_N is used to reduce the total N uptake. This reduction function also effectively reduces the leaf N contents and alters the dry-matter allocation pattern (Eckersten and Jansson, 1991). Pasture growth parameters in this study have been chosen to generate appropriate levels of dry-matter production; that is, the model simulates yields between 10 and 15 Mg dry matter (DM) ha^{-1} from an irrigated pasture and adds about 1000 kg DM for every 100 kg N ha^{-1} of N fertilizer.

The decomposition of soil biomass and animal manure adds to the amount of mineral N in the soil. This process is known as mineralization, and it is modeled by dividing the soil organic matter into three pools—a fast cycling litter pool, an almost stable humus pool, and a manure pool (Johnsson et al., 1987). The relative amount of organic-N in these three pools changes daily to reflect inputs of fresh biomass and manure, and the losses of older biomass and manure as it decomposes. The N demand for the internal cycling of soil C and soil N is regulated by the C/N ratio of the soil biomass, r_o , which is one of the model inputs.

Decomposition of soil litter carbon (C_L) is assumed to be a first-order process and is specified by a rate constant (K_L) that is influenced by temperature and soil moisture. The products of decomposition are CO_2 , stabilized organic material (humus), and, conceptually, microbial biomass and metabolites. The relative amount of these products is determined by a synthesis efficiency constant (f_E) and a humification fraction (f_H). The following mass balance equations, which represent the inputs minus the outputs of soil C and soil N, are used to model the turnover of C and N in the litter pool:

$$\frac{\partial C_L}{\partial t} = [(1 - f_H) f_E - 1] K_L C_L + F_{C,L} \quad [5]$$

$$\frac{\partial N_L}{\partial t} = (1 - f_H) f_E \frac{1}{r_o} - \frac{N_L}{C_L} K_L C_L + F_{N,L} \quad [6]$$

where F represents the amount of fresh organic matter added to the soil biomass. A similar set of equations describes the turnover of C and N in the manure pool:

$$\frac{\partial C_M}{\partial t} = [(1 - f_H)f_E - 1]K_M C_M + F_{C,M} \quad [7]$$

$$\frac{\partial N_M}{\partial t} = \left[(1 - f_H)f_E \frac{1}{r_O} - \frac{N_M}{C_M} \right] K_M C_M + F_{N,M} \quad [8]$$

Lastly, a set of mass balance equations is used to describe the turnover of C and N in the humus pool:

$$\frac{\partial C_H}{\partial t} = f_E f_H K_L C_L - K_H C_H + F_{C,H} \quad [9]$$

$$\frac{\partial N_H}{\partial t} = \frac{f_E f_H}{r_O} K_L C_L - K_H N_H + F_{N,H} \quad [10]$$

Decomposition of soil humus (C_H) is assumed to follow first-order kinetics with a specific rate constant (K_H) that depends on temperature and soil moisture. The other terms in these mass balance equations have already been described above.

All C and N turnover reactions can result in a net production (mineralization) or a net consumption (immobilization) of NH_4 , depending on the C/N ratio of the biomass, r_O , in the three pools. From a consideration of mass balances, any increase in $NH_4^+ - N$, due to mineralization, must equal the decrease in organic N from the three organic matter pools. Thus, the following mass balance equation is solved for mineralization:

$$\frac{\partial NH_4^+}{\partial t} = \left[\frac{N_L}{C_L} - \frac{f_E}{r_O} \right] K_L C_L + \left[\frac{N_M}{C_M} - \frac{f_E}{r_O} \right] K_M C_M + K_H N_H \quad [11]$$

Mineralization occurs whenever $\partial NH_4^+ / \partial t > 0$, otherwise immobilization will occur. The model also recognizes that if no NH_4 is available for immobilization, then NO_3 can be used according to the following equation:

$$\frac{\partial NO_3^-}{\partial t} = -\frac{f_E}{r_O} (K_L C_L + K_M C_M) \quad [12]$$

During all simulations reported here we chose typical values for most of the parameters: the rate constants were set equal to $K_L = 0.015 \text{ d}^{-1}$, $K_M = 0.015 \text{ d}^{-1}$, and $K_H = 0.00005 \text{ d}^{-1}$; constant values were used for the efficiency of C turnover, $f_E = 0.5$; the humification fraction, $f_H = 0.2$; and the C/N ratio of the soil biomass, $r_O = 10.0$, as suggested by Johnsson et al. (1987).

Transformation processes in the soil are regulated by abiotic response functions involving soil temperature and soil moisture. A Q_{10} relationship (Bunnell et al., 1977) is used to express the effect of temperature:

$$f_T(z) = Q_{10}^{\frac{T(z) - T_B}{10}} \quad [13]$$

where $T(z)$ is the soil temperature for the layer, T_B is the base temperature at which f_T equals 1, and Q_{10} is the factor change in rate due to a 10° change in temperature.

The soil moisture factor decreases, on either side of an optimum level, in drier soil or in excessively wet soil (Johnsson et al., 1987); that is,

$$f_w(z) = f_s + (1 - f_s) \left[\frac{\theta_s(z) - \theta(z)}{\theta_s(z) - \theta_H(z)} \right]^M \quad \theta_H(z) < \theta(z) < \theta_s(z),$$

$$f_w(z) = 1 \quad \theta_L(z) < \theta(z) < \theta_H(z),$$

$$f_w(z) = \left[\frac{\theta(z) - \theta_w(z)}{\theta_L(z) - \theta_w(z)} \right]^M \quad \theta_w(z) < \theta(z) < \theta_L(z), \quad [14]$$

where θ_s is the saturated water content; θ_H and θ_L are the high and low water contents, respectively, for which the soil moisture factor is optimal; and θ_w is the minimum water content for process activity. The factor f_s defines the relative effect of moisture when the soil is completely saturated, and M is an empirical constant.

All N-transformation processes in the soil are assumed to be first-order with rate constants that are regulated by both temperature and moisture status of the soil. The hydrolysis of urea, U (mg L^{-1}), to NH_4 , A (mg L^{-1}), is modeled as

$$\frac{dU}{dt} \Big|_{U \rightarrow A} = -k_1 f_T(z) f_M(z) U \quad [15]$$

and depends on a potential rate (k_1). The transfer of NH_4 to NO_3 , N (mg L^{-1}), is modeled as

$$\frac{dA}{dt} \Big|_{A \rightarrow N} = -k_2 f_T(z) f_M(z) \left[A - \frac{N}{n_q} \right] \quad [16]$$

and depends on the potential rate (k_2), which is reduced as the NO_3/NH_4 ratio (n_q) of the soil is approached. If $A < N/n_q$, then no transfer of NH_4 to NO_3 takes place.

Denitrification is the transfer of NO_3 to gaseous N (N_2 and N_2O) products. This is an anerobic process, and consequently, it is highly dependent on soil aeration. Soil moisture is used as an indirect expression of the O_2 status of the soil. The influence on the denitrification rate

$$f_D(z) = \left[\frac{\theta(z) - \theta_D(z)}{\theta_s(z) - \theta_D(z)} \right]^d \quad [17]$$

is expressed as a power function that increases from a threshold point (θ_D) to a maximum at saturation (θ_s), where d is an empirical constant. No denitrification occurs below the threshold point. The denitrification rate for each layer is modeled as

$$\frac{dN}{dt} \Big|_{N \rightarrow \text{gas}} = -k_3(z) f_T(z) f_D(z) \left[\frac{N}{N + c_s} \right] \quad [18]$$

and depends on a potential denitrification rate (k_3), the soil aeration status (f_D), and the same temperature factor (f_T) used for the other biologically controlled processes. The rate constant k_3 is assumed to be a linear function of soil organic C (Smith and Arah, 1990). The factor c_s is the NO_3 concentration, where the denitrification rate is 50% of the maximum and all other factors are optimum.

The N transport model allows for an input of mineral N in the form of urea, NH_4 , or NO_3 . This option allows us to simulate different forms of mineral fertilizer that are broadcast onto the soil surface. We can also simulate the return of animal urine by assuming that it is all in the form of urea.

Once the urea or urine is applied to the soil surface, its fate is determined by two competing processes:

- losses due to hydrolysis of urea to NH_3
- losses due to the drainage of urea through the soil profile

It is assumed that all of the urea enters the soil, and that any surface runoff of urea is negligible. The total mass of urea, M_U (mg m^{-2}), in each soil slab of thickness z_R (mm) is found by solving the following mass balance equation:

$$\frac{dm_U}{dt} = z_R \frac{d\theta R_U U}{dt} = X_{U,i} - (k_1 z_R \theta U + W_p U) \quad [19]$$

where U (mg L^{-1}) is the concentration of urea in soil solution, θ ($\text{m}^3 \text{ m}^{-3}$) is the soil's volumetric water content, $X_{U,i}$ (mg m^{-2})

is the mass of urea added to the i th segment ($X_{U,i} = 0$ if $i > 1$), k_1 (1/d) is the rate-constant describing the hydrolysis of urea to NH_4 , $W_p U$ (mm d^{-1}) represents the percolation of dissolved urea through the soil. Different concentrations are a reflection of soil water content and the mass of urea that is added during each urine event. Urea is rapidly hydrolyzed to NH_4 , in a matter of a few days.

The fate of dissolved NH_4 is determined by six competing processes:

- inputs from the mineralization of the soil biomass
- retardation due to the adsorption of NH_4 to the soil particles
- losses due to the nitrification of NH_4 into NO_3
- losses due to the volatilization of NH_3 gas
- losses due to the drainage of NH_4 through the soil slab
- losses due to plant uptake

The total mass of NH_4 , M_A (mg m^{-2}), in each soil slab of thickness z_R (mm) is found by solving the following mass balance equation:

$$\frac{dM_A}{dt} = z_R \frac{d\theta R_{A,A}}{dt} = (X_{A,i} + S_M + k_1 z_R \theta U) - [k_2 z_R \theta (A - N/n_q) + f_v z_R \theta A + P_A + W_p A] \quad [20]$$

where A (mg L^{-1}) is the concentration of NH_4 in soil solution, $X_{A,i}$ (mg m^{-2}) is the total mass of NH_4 added to the i th layer ($X_{A,i} = 0$ if $i > 1$), S_M (mg m^{-2}) is rate of mineralization, P_A ($\text{mg m}^{-2} \text{d}^{-1}$) is the rate of plant uptake, k_2 (1/d) is a rate constant to describe the nitrification of NH_4 to NO_3 , and $W_p A$ ($\text{mg m}^{-2} \text{d}^{-1}$) represents the percolation of dissolved NH_4 through the soil slab. Here f_v represents the fraction of NH_4 that is lost by volatilization of NH_3 to the atmosphere. For simplicity, it is assumed that $f_v = 0.05$ in the top 10 cm of soil and equals zero elsewhere. $R_A = (1 + \rho K_D/\theta)$ is the retardation factor for NH_4 , ρ (kg L^{-1} soil) is the soil's dry bulk density, and K_D (L kg^{-1}) is the distribution coefficient that determines how much NH_4 gets adsorbed to the cation-exchange sites of the soil. A standard batch-isotherm technique was used to determine the adsorption of NH_4 (Vogeler et al., 1997). A linear isotherm fitted to the data over the concentration range 0 to 40 mg N L^{-1} , yielded a distribution coefficient, K_D , of 3.3 L kg^{-1} . This value indicates that NH_4 will be strongly adsorbed by the Takapau sandy loam soil.

The fate of any NO_3 in the soil water is determined by the following six processes:

- inputs of NO_3 from fertilizer application
- inputs from the nitrification of NH_4
- retardation due to the adsorption of NO_3 ($= 0$ in most mineral soils)
- losses from denitrification
- losses due to plant uptake
- losses due to the drainage of N beyond the root zone

The total mass of N, M_N (mg m^{-2}), in each soil slab of thickness z_R (mm) is found by solving the following mass balance equation:

$$\frac{dM_N}{dt} = z_R \frac{d\theta R_{N,N}}{dt} = (X_{N,i} + k_2 z_R \theta A) - \left(k_3 z_R \theta \left[\frac{N}{N + c_s} \right] + P_N + W_p N \right) \quad [21]$$

where N (mg L^{-1}) is the concentration of NO_3 in soil solution, $X_{N,i}$ (mg m^{-2}) is the total mass of NO_3 added to the i th layer ($X_{N,i} = 0$ if $i > 1$), k_3 (1/d) is a rate constant to describe denitrification losses, P_N ($\text{mg m}^{-2} \text{d}^{-1}$) is the rate of plant

uptake, and $W_p N$ ($\text{mg m}^{-2} \text{d}^{-1}$) represents the drainage of NO_3 through the soil slab. Denitrification is considered to be a microbial process that is rate-limited by the amount of soil organic C (the energy source) and mineral N (the nutrient source) available to the microbes.

RESULTS

Groundwater Quality

Rosen and McNeill (1996) used HCO_3 , Cl, and NO_3 -N as indicators of groundwater contamination from the feedlot and used HCO_3 as an indicator of different geological sources of the groundwater. In this study NH_4 -N also was measured as an indicator of reducing conditions in the aquifer. Most wells were below the detection limit for NH_4 -N, indicating that the aquifer is oxidized. Rosen (1996) demonstrated that dissolution of limestone bedrock in the hills around Maraekakaho Stream is a major source of HCO_3 to the stream and the groundwater near the stream. However, the nitrification reaction:



yields acidity (2H^+), which will act to reduce the HCO_3 and pH of the groundwater (Rosen and McNeill, 1996). Therefore, it is important to assess the concentrations of HCO_3 and pH in the groundwater as an indicator of nitrification and as a source of limestone-derived water.

The indicators chosen do not specifically identify that the feedlot is the sole source of groundwater contamination. Nitrate-N and Cl could be contributed by other farming practices in the surrounding area. However, the pattern and distribution of the various chemical indicators suggest the feedlot is the likely source of the contamination (Rosen and McNeill, 1996; Rosen, 1996).

Chloride concentration maps are not presented here because they show similar patterns to the NO_3 concentrations. Rosen (1996) showed Cl concentration maps of the data collected before 1995.

Comparison of Previous Groundwater Results with May 2001 Sampling

Rosen et al., (1995), Rosen (1996), and Rosen and McNeill (1996) showed that the groundwater under the Maraekakaho area is a mixture of Maraekakaho Stream water, Ngaruroro River water, and rainwater. The groundwater flow direction was mapped flowing to the east-northeast toward the Ngaruroro River under the feedlot, with an approximate 10-m head difference between the southwest and the northeast boundaries of the feedlot. Elevated NO_3 -N concentrations were highest under the feedlot in 1994 and 1995, and the distribution of elevated concentrations was consistent with calculated groundwater flow directions down gradient of the feedlot.

Rosen and McNeill (1996) found the highest NO_3 -N and Cl concentrations in the GMK94 piezometers located in the feedlot and down gradient of the feedlot in August 1994, only 3 mo after the June 1994 samples were taken. In particular, Piezometers GMK94-01 and GMK94-06 had NO_3 -N concentrations $>100 \text{ g m}^{-3}$ at this time. Elevated NO_3 -N and lower HCO_3 levels in wells and streams coincided with rainfall events with

very little lag time (Rosen, 1996). This indicates that rainwater percolation is the main processes for washing accumulated soil NO_3 into the unconfined groundwater aquifer. However, because percolation is rapid, direct rainwater recharge alone cannot account for the long-term spatial variations in groundwater chemistry found by Rosen and McNeill (1996). Groundwater $\text{NO}_3\text{-N}$ concentrations were generally low when sampled in June 1994 ($0.3\text{--}2\text{ g m}^{-3}$) at the up-gradient sites and sites around the Maraekakaho bridge area, and generally higher ($21\text{--}40\text{ g m}^{-3}$) under and immediately down gradient of the feedlot.

Chloride concentrations were also elevated above background values down gradient of the feedlot. However, there may have been some contribution from the down-gradient orchard. Alkalinity contours over the feedlot area show that there is little evidence of soil ammonification reactions causing increased HCO_3 concentrations in the groundwater.

Figures 3 and 4 show contoured diagrams of HCO_3 and $\text{NO}_3\text{-N}$ for both the May 2001 and June 1994 sampling runs. Where multiple piezometer nests occur at a site, the average concentration was used for the $\text{NO}_3\text{-N}$ maps, and the deep piezometer value was taken for contouring HCO_3 . This is because there is little variation between the concentrations detected in the deep versus shallow piezometers, except for HCO_3 (see the following section). This is consistent with the method used in Rosen et al. (1995). Figures 5A, 5B and 6A, 6B show time trends of HCO_3 and $\text{NO}_3\text{-N}$, respectively, since the June 1994 sampling.

Alkalinity

The May 2001 HCO_3 concentrations in groundwater range from 35 g m^{-3} (GMK94-01 Deep) to 280 g m^{-3} (Hunt Club) (Table 2). The distribution of the HCO_3 values and resultant contours (Fig. 3) are very similar to the June 1994 sampling, where groundwater wells located close to the Maraekakaho Stream have higher alkalinities than wells further away. In general, HCO_3 concentrations have remained about the same, except at Sites GMK94-01, GMK94-02, and GMK94-05, where concentrations are approximately one-half of the June 1994 measurements.

Some differences in HCO_3 with aquifer depth can be seen at the multilevel nested piezometer sites. Sites GMK94-01 and GMK94-04 have the greatest variations of 15 g m^{-3} (decreasing with depth) and 30 g m^{-3} (increasing with depth), respectively. The other two sites (GMK94-02 and GMK94-03) show little variation with depth. The observations above are generally consistent with those observed in the June 1994 sampling round, except for Site GMK94-02. The June 1994 sampling round showed a large difference in alkalinities between the shallow (86 g m^{-3}), intermediate (340 g m^{-3}), and deep (244 g m^{-3}) piezometers at Site GMK94-02. However, these higher concentrations may have been caused by insufficient purging of the deeper wells (Rosen and McNeill, 1996). By the August 1995 sampling, HCO_3 concentrations in this well were 40 to 47 g m^{-3} . The May

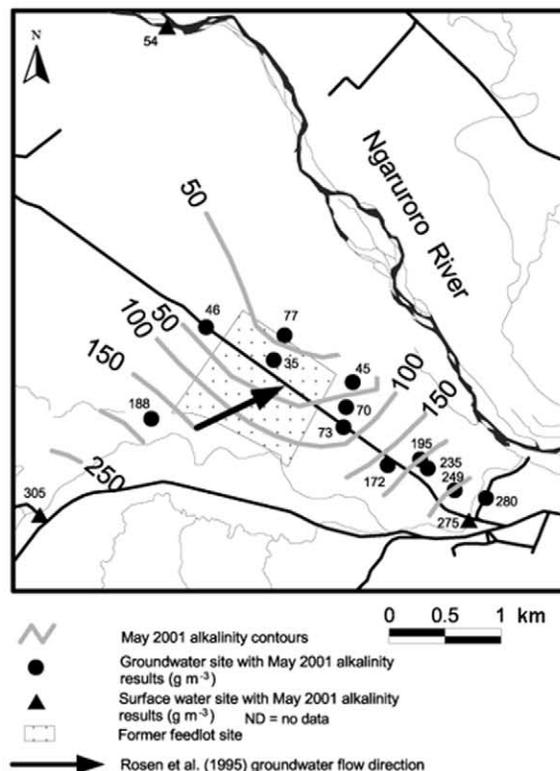
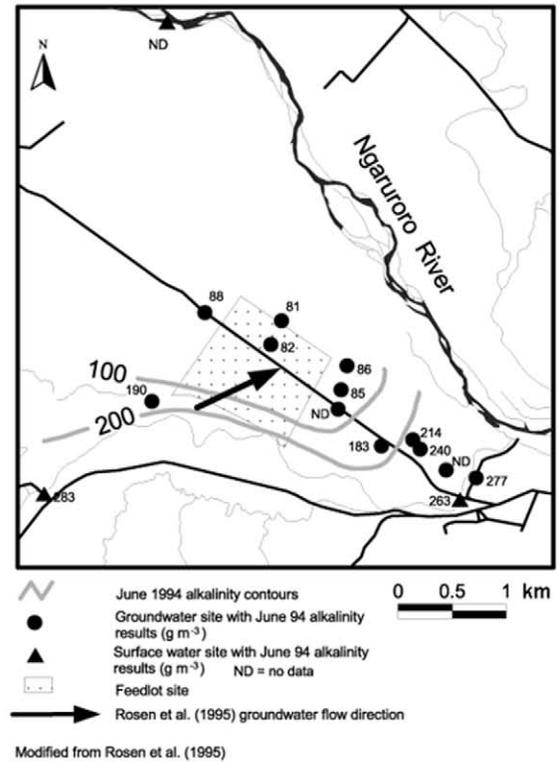


Fig. 3. Alkalinity contour maps for June 1994 and May 2001 sampling rounds. The concentration of the deep piezometer was used for the nested GMK94 piezometers in these maps.

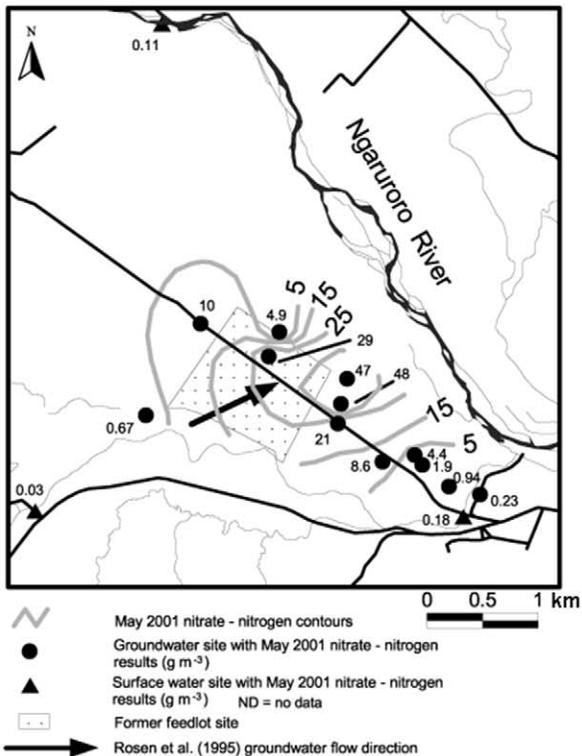
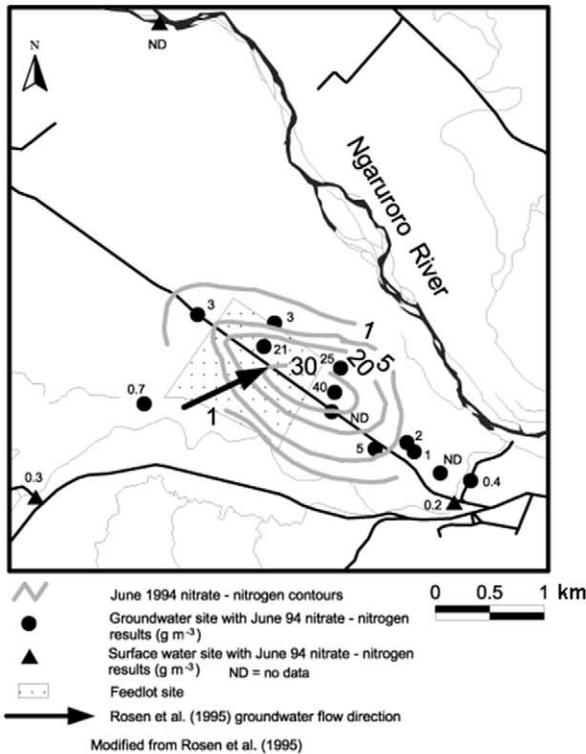


Fig. 4. Nitrate-N contour maps for June 1994 and May 2001 sampling rounds. The average concentration of all three piezometers was used for the nested GMK94 piezometers in these maps.

2001 results show HCO_3^- concentrations ($44\text{--}45 \text{ g m}^{-3}$) are about the same as the August 1995 sampling and are the same in the three piezometers at GMK94-02 (Table 2).

One explanation of the relatively low HCO_3^- concentrations measured in the GMK94 piezometers is that the nitrification reaction produces H^+ ions (see Eq. [22]) that could react with the HCO_3^- in solution to form CO_2 gas. This would explain why the piezometers within and directly down the groundwater gradient from the feedlot have the lowest HCO_3^- concentrations (which are lower than HCO_3^- concentrations of the Ngaruroro River) and the lowest pH (between 6.0 and 6.3 for GMK94-01, GMK94-04, and GMK94-06). GMK94-05 may have low HCO_3^- concentrations because of diffusion from the feedlot source, or because of other up-gradient sources of NO_3^- contamination. Higher HCO_3^- concentrations (by about 40 g m^{-3}) and slightly higher pH values (by up to 0.2 pH units) measured in May 2001 in the deeper piezometers at GMK94-04 (Fig. 7) and at GMK94-06 indicate that nitrification reactions could be slowing down and the deeper part of the aquifer is beginning to recover. With time, it is expected that HCO_3^- concentrations in all the GMK94 piezometers should increase.

Nitrate-Nitrogen

The May 2001 $\text{NO}_3\text{--N}$ contour pattern is consistent with the previously documented contour patterns (Rosen et al., 1995; Rosen, 1996; Rosen and McNeill, 1996), with a plume moving to the east-northeast (Fig. 4).

Nitrate-N concentrations are higher than the June 1994 sampling in all groundwater wells (where measured) except at the Maraekakaho Stream and Tait Road sites, which remained the same. However, most wells have lower concentrations than the maximum $\text{NO}_3\text{--N}$ concentrations measured by Rosen and McNeill (1996) in August 1995.

Rosen and McNeill (1996) reported higher $\text{NO}_3\text{--N}$ concentrations in the GMK94 sites during the August 1994 to August 1995 sampling (except at GMK94-02, which had the highest concentrations measured in May 2001), including $\text{NO}_3\text{--N}$ concentrations of 7 to 9.3 g m^{-3} at Site GMK94-05. The largest $\text{NO}_3\text{--N}$ increases during this time period were at GMK94-01 and GMK94-06 (Fig. 6). Nitrate-N concentrations measured in the May 2001 sampling ranged from 0.03 g m^{-3} (Maraekakaho Stream and Tait Road) to 48 g m^{-3} (GMK94-02 Intermediate and GMK94-06) (Table 2). Piezometers in and directly down the groundwater gradient of the former feedlot site had elevated $\text{NO}_3\text{--N}$ concentrations compared with up-gradient piezometers (Fig. 4). The two up-gradient piezometers, Simmond well and GMK94-05, had a large difference in concentrations (0.67 g m^{-3} and 10 g m^{-3} , respectively). The N concentration in GMK94-05 also increased by about 3 g m^{-3} $\text{NO}_3\text{--N}$ from 1994 to 2001, indicating some N input up gradient of this piezometer. The highest concentrations of $\text{NO}_3\text{--N}$ in May 2001 (Fig. 6, Table 2) are found at Sites GMK94-06 and GMK94-02, both in the orchard and immediately down the groundwater gradient of the former feedlot.

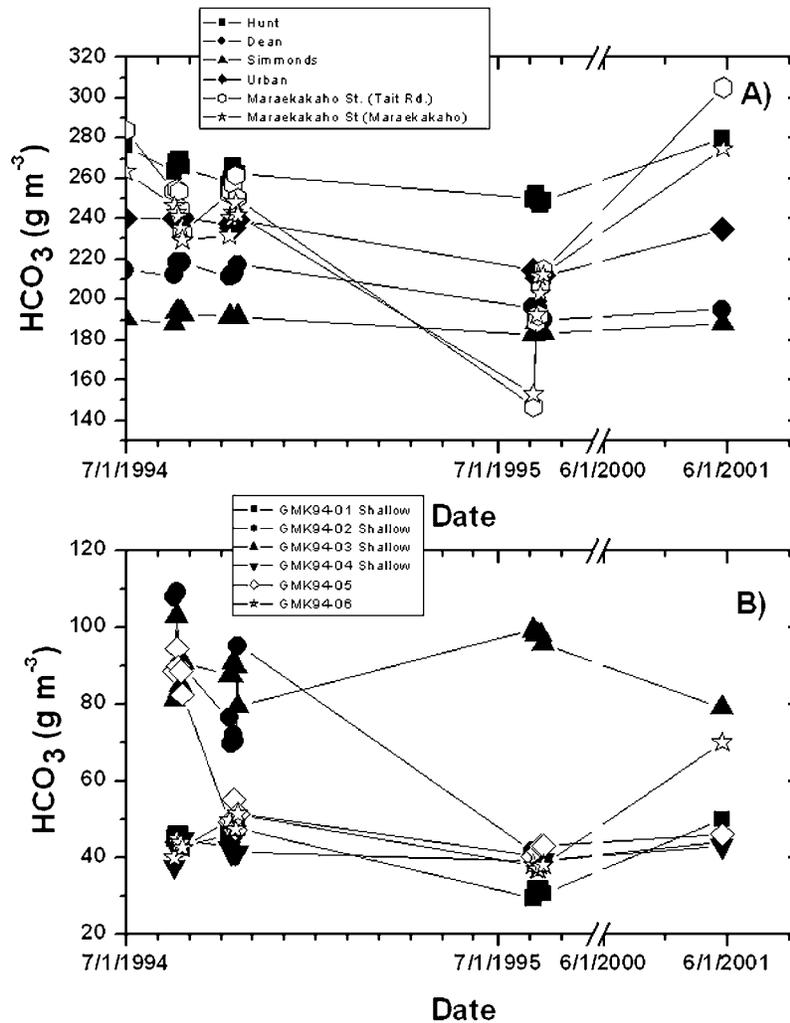


Fig. 5. Groundwater HCO₃ trends through time from June 1994 and May 2001 for selected wells and the Maraeakaho Stream: (A) domestic wells and Maraeakaho Stream and (B) GMK94 series shallow wells.

Nitrate-N concentrations at the sites near the Maraeakaho Township are $<5 \text{ g m}^{-3}$ except for Barley's well, which was measured at $8.6 \text{ g m}^{-3} \text{ NO}_3\text{-N}$ in 2001. Nitrate-N concentrations in the multilevel piezometers GMK94-01, GMK94-02, GMK94-03, and GMK94-04 generally decrease with aquifer depth (Table 2, Fig. 7). Typically, variations in $\text{NO}_3\text{-N}$ concentration with depth are $<6 \text{ g m}^{-3}$, except for GMK94-04, where there is an 11 g m^{-3} decrease from the shallow to the deep piezometer in the May 2001 sampling. All multilevel piezometers showed a decrease in $\text{NO}_3\text{-N}$ concentration from the shallow to the deep piezometers. This decrease in $\text{NO}_3\text{-N}$ concentration was not observed in the 1994 and 1995 sampling because N inputs from the active feedlot were large and penetrated deeply into the aquifer. Because the constant source of NO_3 input has decreased (to just soil input) the deeper part of the aquifer is becoming less contaminated more quickly than the upper part of the aquifer, and the NO_3 entering the groundwater table now is not penetrating as deeply as when the feedlot was in operation.

Although the deeper parts of the aquifer have lower concentrations of $\text{NO}_3\text{-N}$ than the shallow parts of the

aquifer, all three piezometers at GMK94-02 and the deep piezometer GMK94-03 show increasing trends in $\text{NO}_3\text{-N}$ from 1994 to 2001. These two piezometers are directly down gradient from feedlot inputs, indicating that $\text{NO}_3\text{-N}$ is still traveling down gradient to these sites.

Comparison of the groundwater $\text{NO}_3\text{-N}$ results from the 1994 and August 1995 sampling with the May 2001 sampling indicate $\text{NO}_3\text{-N}$ is stored in the unsaturated zone and percolates to the groundwater after large recharge events. This is because high concentrations of these anions persist in the groundwater 3 yr after the main source of these anions (the feedlot) was eliminated.

Chloride

Chloride patterns and values are similar in comparisons of the June 1994 and May 2001 data. The main feature in each contour map is the high Cl concentrations observed at Sites GMK94-02 and GMK94-06, which are both immediately down the groundwater gradient, but outside of the feedlot.

A new groundwater well, used to irrigate the orchard

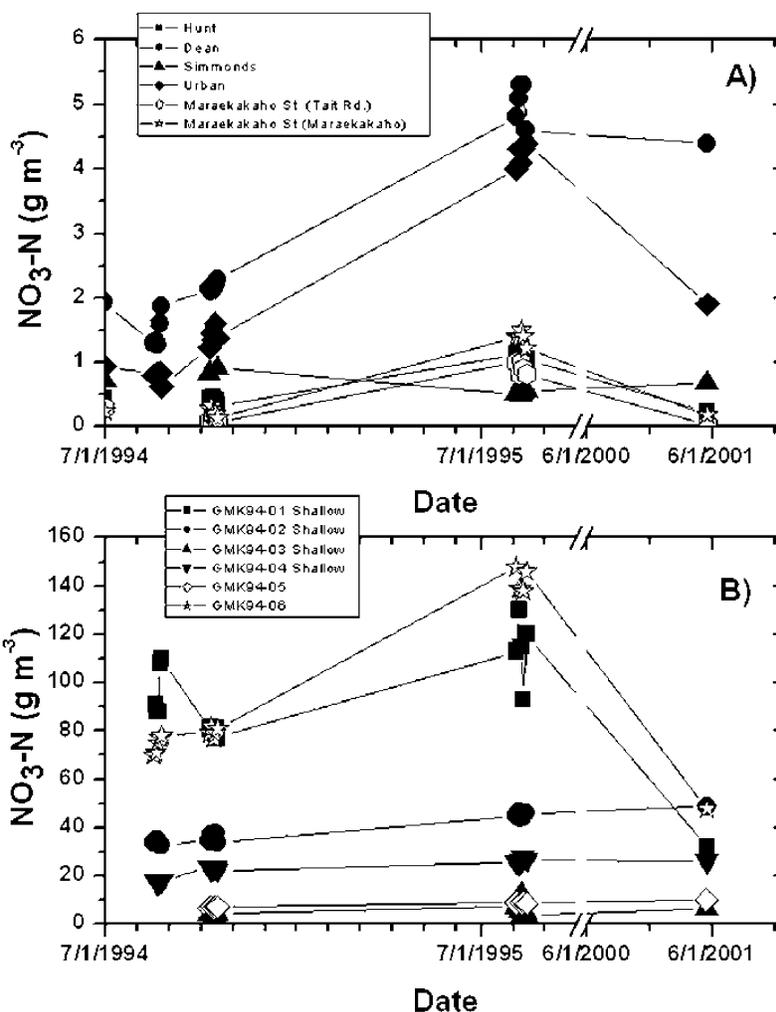


Fig. 6. Groundwater NO₃-N trends through time from June 1994 and May 2001 for selected wells and the Maraekakaho Stream: (A) domestic wells and Maraekakaho Stream and (B) GMK94 series shallow wells.

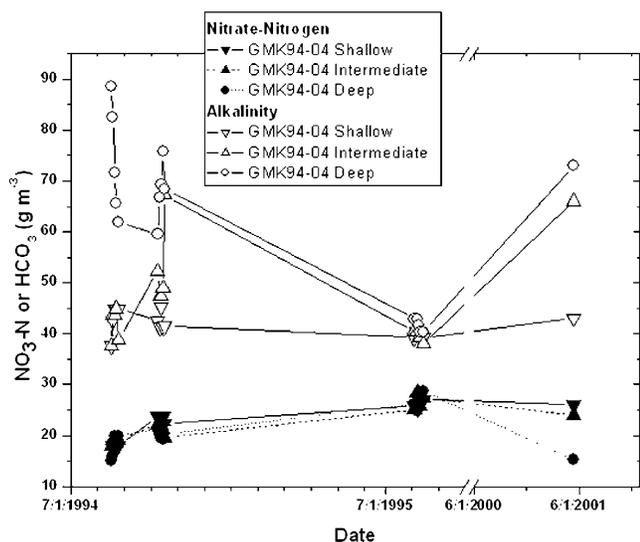


Fig. 7. Plot of NO₃-N and alkalinity variations with depth in the aquifer for piezometer GMK94-04. Nitrate-N concentrations decrease with depth, and HCO₃ increase with depth.

east of the vineyard, is likely to have elevated NO₃-N and Cl concentrations given that it is located in the NO₃-N and Cl plumes. When the pump is switched on, draw down from this well may alter the groundwater flow direction toward this well, and may provide a source of NO₃-N and Cl to the soils in the orchard. This then has the possibility of leaching to groundwater beneath the orchard and recycling the high NO₃-N and Cl concentrations in the area for a longer period of time than if the well had not been drilled.

Chloride concentrations in the May 2001 sampling ranged from 5.9 g m⁻³ in the Ngaruroro River to 81 g m⁻³ at GMK94-06 (Table 2). Elevated concentrations occurred under and immediately down gradient from the former feedlot site where a concentration of 81 g m⁻³ was observed at Site GMK94-06. An average concentration of 74 g m⁻³ was observed at GMK94-02. Typical background concentrations are 19 and 14 g m⁻³ at the up-gradient sites, Simmond's well and GMK94-05, respectively. Average Cl concentrations in wells sampled in Maraekakaho Township west of the mouth of Maraekakaho Stream (30 g m⁻³) and water collected from Maraekakaho Stream (36 g m⁻³) are similar.

No large difference (<10%) in Cl concentrations with depth was observed at Site GMK94-01 (Table 2). The other multilevel piezometers, GMK94-02, -03, and -04 all showed decreasing Cl concentrations with depth with concentrations that varied from 11, 7, and 16 g m⁻³ over vertical distances of 9, 1.6, and 11.1 m, respectively.

Soil Quality

The data collected during and after feedlot operations on the groundwater concentrations of HCO₃, NO₃-N, and Cl indicate that the groundwater down gradient of the feedlot is becoming less contaminated. To determine how quickly the groundwater quality would recover, we collected soil N data below the former feedlot to quantify how much N could continue to percolate into the groundwater with time. The soil data presented below provides data on how N moves through the unsaturated zone above the groundwater table. These results are used in our models, as described below, to provide predictions of groundwater quality with time.

Soil Sampling under the Former Feedlot

How much N is likely to leach under the former feedlot in the future is a question that can be answered by sampling the soil profile. In April 2001 we collected samples from two sites (Fig. 1) to a depth of 12 m to establish the depth-wise profile of total N (Tables 3 and 4). These soil samples revealed a total N content of between 0.01 and 0.02% (g/g), a total C content of between 0.07 and 0.12%, and a C/N ratio of between 4 and 7 at depths below about 3 m. Total N includes both organic N, which is immobile, and mineral N, likely to be in the form of NO₃. The C/N ratio is calculated on the fine earth fraction (<2-mm size fraction) only. The problem here is to estimate the amount of NO₃ that remains in the soil profile. This estimate can be made by comparing the measured C/N ratio (4–7) with the C/N ratio typical of a sandy soil (C/N = 10–12; Smith and Mullins, 1991). A very low C/N ratio, as measured under the feedlot site, indicates an excess of N relative to C in the soil profile. Any excess N is likely to be NO₃-N.

The C/N ratio measured under the former feedlot at both soil sampling sites was quite low at depths below about 7 m in the soil profile. Nevertheless, given the measured total C content of the soil, and assuming a typical C/N ratio of 12, we calculated that more than one-half of the total N below 7 m is likely to be NO₃ that originated from the former feedlot. The vertical profile of the C/N ratios for GMK01-SOIL1 and GMK01-SOIL2 are shown in Fig. 8. The bulge in low C/N ratios approximates the depth to which the NO₃ has probably leached during the previous 3 to 4 yr. This profile depth is consistent with output from the SPASMO model. Site GMK01-SOIL2 has a similar profile in total N, but our calculations of NO₃ results in a smaller bulge at the 7- to 10-m depth, possibly due to errors in our simple procedure to estimate the C/N ratio. It could also be true that the GMK01-SOIL2 site may have received fewer nutrients from the feedlot operations because the site was located on the edge of the former feedlot.

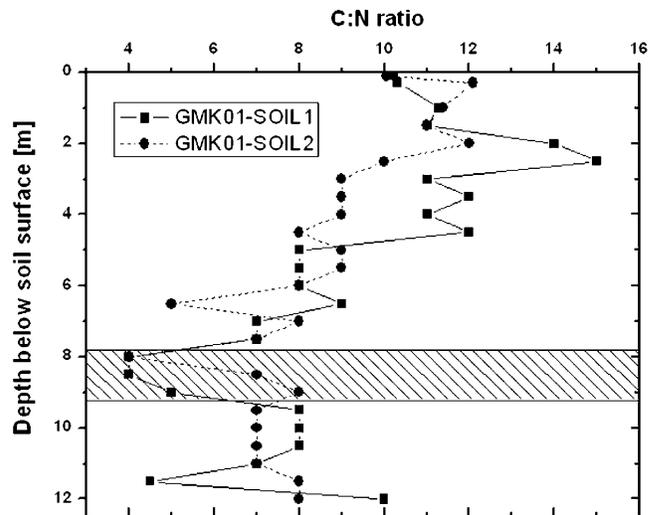


Fig. 8. Carbon/nitrogen ratios with depth in the GMK01-SOIL1 and -SOIL2 profiles. The boxed area near the 8-m depth corresponds to low C/N ratios that represents the depth to which we estimate that excess feedlot N has penetrated.

Modeled Effects of Land-Use Change on Nitrogen Balance

To predict the effects of the feedlot after closure and to predict what effect the conversion of the feedlot site to a vineyard may have on groundwater quality, we used the SPASMO model to simulate N inputs to the groundwater assuming (i) the feedlot still existed, (ii) conversion of the feedlot to a vineyard, and (iii) N inputs from a vineyard. These different scenarios provide predictions of the effects that different land uses would have on N concentrations in the groundwater in the Maraekakaho area.

Feedlot Nitrogen Balance

Historically, the feedlot at Maraekakaho was operated for about 13 yr, between 1986 and 1998, with the animals being grazed just once or twice each year. We do not have records of the actual grazing dates, so for the purpose of demonstration we have assumed here that a single 10-d grazing event occurred each October for 13 consecutive years. Thereafter, in the model we altered the land use from a feedlot to a vineyard, and reran the calculations of NO₃ leaching under the vineyard using the next 14 yr of weather data. We used this second simulation to assess the potential for continued NO₃ leaching under the former feedlot.

The calculations assume a sheep stocking density of 800 head ha⁻¹ and a standing time of 10 d. The feedlot is a dryland pasture that is grazed just once per year, in the month of October. Pasture growth and N cycling within the root zone are predicted using daily sunshine, temperature, and rainfall at the site, with each process being moderated by the moisture and nutrient status of the soil (King, 1993). Sheep feed requirements are met first by the available pasture and then by pellet supplements, which accounts for most of the feed given to the sheep. We have no information on the quality and quantity of the pellet feed, so we have made some ap-

proximations to estimate the dung and urine returns to the soil. These approximations are based on normal levels of feed intake and assume simple ratios of N excretion by the grazing sheep (Whitehead, 1995). Each sheep urinates 3 L d^{-1} , in the form of urea at a concentration of 10 g L^{-1} . The daily return of fecal N from each sheep is taken to be 10 g N d^{-1} (Parsons et al., 1991). At a stocking rate (SR) of $800 \text{ sheep ha}^{-1}$, the total urine N loading is $24 \text{ kg N ha}^{-1} \text{ d}^{-1}$ and a total fecal N loading of $8 \text{ kg N ha}^{-1} \text{ d}^{-1}$. Over the 10 d that the sheep are on the feedlot, the total N loading is assumed to be 240 kg N ha^{-1} in the form of urine and 80 kg N ha^{-1} in the form of dung. Thus, each 10-d grazing event adds about 320 kg N to the feedlot pasture. These rates indicate that there is the potential for excessive NO_3 leaching to occur under the sheep feedlot.

The feedlot is assumed to be a dryland pasture that is grazed just once per year, in October. However, based on our observations of the feedlot when it was in operation, very little pasture grass was growing in the enclosures. This indicates that our model will be a conservative estimate of N leaching to the groundwater.

Pasture growth and N cycling within the root zone are predicted using daily sunshine, temperature, and rainfall at the site, with each process being moderated by the moisture and nutrient status of the soil (Green et al., 2001). The feedlot is assumed to be 1 km^2 in size and to have 80 000 sheep on the pasture site for a period of 10 d. Thus, the SR is set at $800 \text{ sheep ha}^{-1}$, which is very high compared with a SR of 10 to 20, which is more typical of a grazed sheep pasture. The task here is to estimate the feed intake and the corresponding amount of dung and urine that is returned to the pasture by the grazing sheep.

The feedlot at Maraekakaho was operated between 1986 and 1998, and animals were grazed once or twice each year. Because we do not have records of the actual grazing dates we have assumed here, for the purpose of demonstration that a single 10-d grazing event occurred each October for 13 consecutive years. Thereafter, in the model we altered the land use from a feedlot to a vineyard, and ran the calculations of NO_3 leaching under the vineyard using the next 14 yr of weather data. We used this 27-yr simulation to assess the potential for continued NO_3 leaching under the closed feedlot after changing to vineyard land use.

The temporal pattern of NO_3 leaching under the sheep feedlot is shown in Fig. 9. For these calculations we used a low initial concentration of 5 g m^{-3} in the soil profile. For this simulation, the site was assumed to be operating as a feedlot for approximately the first 13.5 yr (Days 1 to 5000) and then was converted to a vineyard for the next 13 or so years (Days 5000 to 10 000).

Our model results show a large increase in NO_3 concentration under the feedlot between 1986 and 1998. Values in excess of 250 g m^{-3} are predicted to occur at a depth of 3 m, and values as high as 150 g m^{-3} are predicted to occur at depths of 7 m (Fig. 9). The reason for this increase is quite simple. When used as a feedlot, there was a large imbalance between NO_3 uptake by the pasture and N return in the dung and urine of the

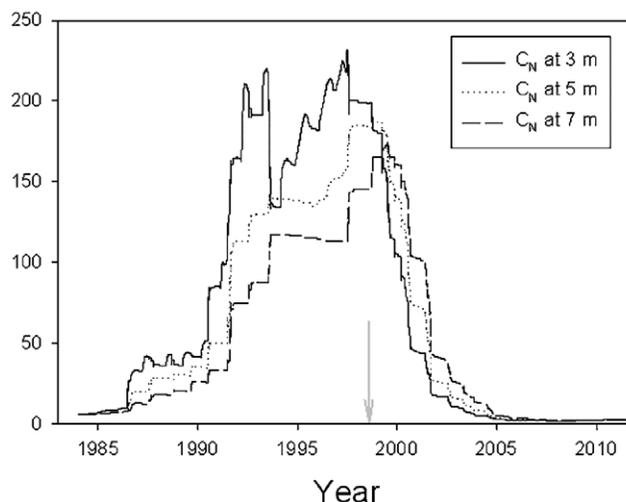


Fig. 9. Modeled NO_3 concentration for selected soil depths under an active feedlot (1986–1998) that was changed to vineyard land use for the next 14 yr. The arrow indicates when the land use was changed to a vineyard. C_N is concentration of N.

sheep. A dryland pasture will remove, on average, about 386 kg N yr^{-1} from the root zone soil. However, a large fraction of this N ($\approx 85\%$) is likely to be returned back to the soil as organic N in the form of decaying plant material as the ungrazed pasture sward senesces. The organic N that is returned to the soil biomass will slowly decompose and release mineral N in the form of NH_4 in solution. This NH_4 is strongly bound to the soil particles, yet it is quite quickly transformed into NO_3 that is mobile and will move through the soil profile along with the percolating drainage water. In this model scenario, the animals are returning some 320 kg N back to the pasture each time the feedlot is in operation. So N input is well in excess of the nutrient requirements of the pasture.

After 13 yr of a feedlot operation, the NO_3 concentrations are predicted to be in excess of 100 g m^{-3} down to a depth of at least 7 m (Fig. 9). These predictions are the result of considering just a single grazing each year and assuming that some N is taken up by pasture growth. The actual NO_3 concentrations under the feedlot may well have been much higher because there was probably more than one grazing event in some of the years and there is very little pasture available for the sheep. Toward the end of the feedlot operation, $\text{NO}_3\text{-N}$ concentrations of more than 140 g m^{-3} were recorded in the shallow groundwater at a depth of 12 m under the feedlot (Rosen and McNeill, 1996). Our model results support the measurements that show that NO_3 concentrations in excess of 10 times the New Zealand drinking water standard are likely in the drainage water under the feedlot.

Once the land use was converted from a feedlot to a vineyard we see a very rapid decrease in soil NO_3 concentrations (Fig. 9). This is because of a combination of two factors: (i) more water is draining through the soil profile under a vineyard (223 mm yr^{-1} compared with 74 mm yr^{-1} for the dryland pasture) and (ii) the total NO_3 loading from fertilizer and animals is much

reduced (30 kg N yr^{-1} compared with $>300 \text{ kg N yr}^{-1}$ under the feedlot operation). The NO_3 concentration at a depth of 3 m has reduced from >200 to $<50 \text{ g m}^{-3}$ during the first 3 to 4 yr after conversion to a vineyard. This represents a reduction of about 80% in soil NO_3 at 3 m.

Because of the stony nature of the underlying soil, and assuming piston flow, any drainage water that travels through the soil profile will advance, on average, at a rate of about 2.5 to 3 m yr^{-1} (223 mm yr^{-1} through an 8–10% transport volume). An 80% reduction in NO_3 concentration at the 7-m depth would appear likely 4 to 5 yr after the land use conversion. An impact of the NO_3 leaching from the former feedlot on NO_3 concentrations in the ground water, at depths up to 14 m, is expected to be observed for a period of up to 10 yr after closure of the feedlot. Dilution of the drainage by lateral transport of low- NO_3 groundwater will also occur. This dilution has not been estimated, but this additional low NO_3 groundwater may speed up the renovation of the groundwater quality under the feedlot site.

Vineyard Nitrogen Balance

The conceptual model used to describe the vineyard land use included grape vines alternating with grass between rows, assuming irrigation and fertilizer management practices that are typical of a commercial vineyard (Green et al., 2000). For modeling purposes, the grape vines are irrigated automatically with an aliquot of 2.6 mm of water whenever the water deficit within the root zone exceeded one-half of the total available soil water. A single dressing of $\text{CaNH}_4[\text{NO}_3]_2$ is applied once per season, in October, at a prescribed rate of 30 kg N ha^{-1} . However, no fertilizer is applied to the vineyard on the former feedlot, because of the high total N status of the soil. Pasture growth and N cycling within the root zone are predicted using daily sunshine hours, temperature, and rainfall at the site, with each process moderated by the moisture and nutrient status of the soil (Green et al., 2001).

Vegetative and fruit growth of the grapevines is simulated to estimate the total N uptake by the plants. Vine growth is modeled using daily weather data and is moderated by the moisture and nutrient status of the soil. The vines were assumed to have been thinned during late summer to reduce their vigor, and pruned in mid winter to remove the old shoots. All leaf and shoot material was recycled back into a pool of soil organic C and N that slowly decomposes, releasing inorganic N to the soil in the form of NH_4 . Nitrogen harvested in the grapes was removed from the site and represents a loss of N from the system. The water and N balance as well as the concentration and loading of NO_3 that leaches below the vineyard were calculated on an annual basis for the purpose of comparison with the sheep feedlot.

The temporal pattern of soil NO_3 under a vineyard at Maraekakaho is shown in Fig. 10. For these calculations we used an initial concentration of 5 g m^{-3} in the soil profile. A value of 1 to 3 g m^{-3} is considered a typical background solution concentration of native soils. The

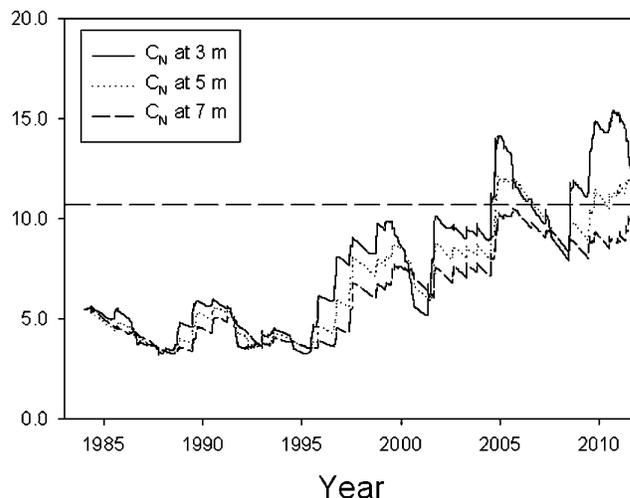


Fig. 10. Predicted NO_3 concentration for selected soil depths under a vineyard at Maraekakaho that receives an annual dressing of N fertilizer at a rate of 30 kg N ha^{-1} . The dashed line is the New Zealand drinking water standard of 11.3 g m^{-3} for NO_3 . C_N is concentration of N.

soil at Maraekakaho is of a high natural fertility, and we anticipated much higher concentrations of NO_3 would soon occur under the feedlot. On an annual basis, the grape vines are expected to remove about 61 kg N ha^{-1} from the root zone soil to support plant growth and fruit production. Two-thirds of the N uptake (i.e., 45 kg N ha^{-1}) will be returned to the soil biomass as leaf fall and stem prunings. Another 18 kg N ha^{-1} is taken off with the fruit at harvest time. Any NO_3 that does leach will be due to natural mineralization of soil organic matter or because of inappropriate application of a N fertilizer.

A good vineyard manager would probably apply just a small amount of N fertilizer whenever soil or plant analysis indicated a nutrient deficiency. In our calculations here, we have applied a regular annual dressing of 30 kg N ha^{-1} . This amount is likely to be somewhat greater than what would be applied under normal vineyard management practice because vineyard managers are not likely to fertilize every block of the vineyard, every year.

On balance, the average quantity of NO_3 that leaches through the soil profile beneath the vineyard, beyond a depth of 7 m, is modeled to be only about 13 kg N ha^{-1} . The average drainage flux of water is expected to be about 225 mm yr^{-1} . After 30 yr, the modeled NO_3 concentration in the drainage water, at a depth of 7 m, is expected to remain below New Zealand's drinking water standard, which has been set at 11.3 g m^{-3} as N for NO_3 (Fig. 10). The model results indicate that NO_3 leaching is unlikely to be excessive under a well-managed vineyard at Maraekakaho. The gradual rise in NO_3 concentration that does occur with time may indicate that the modeled annual N fertilizer application of 30 kg N ha^{-1} is too high for grapes.

CONCLUSIONS

A combination of vadose zone soil N and C measurements coupled with vadose zone modeling and observa-

tions of the hydrochemical characteristics of the aquifer below have shown that $\text{NO}_3\text{-N}$ contamination from an unlined sheep feedlot that overlies a porous alluvial aquifer will continue for several years after the closure of the feedlot. After conversion of the feedlot to a vineyard in July 1998, the vineyard will have a minimal impact on $\text{NO}_3\text{-N}$ contamination in the future. Rainfall percolation is expected to have the greatest influence on NO_3 leaching to groundwater because the vineyard uses only small amounts of irrigation water.

Our study, which compares groundwater samples collected in 1994 and 1995 with samples collected in May 2001, has shown that Cl and $\text{NO}_3\text{-N}$ concentrations peaked in the mid to late 1990s and then decreased in down gradient wells in 2001, but have remained similar overall compared with the June 1994 survey. The HCO_3 concentrations have remained similar compared with the June 1994 survey, but low HCO_3 concentrations and low pH found in wells down gradient of the feedlot site indicate that nitrification reactions are still important in the groundwater at these sites. Nitrate-N concentrations decreased between the shallow and deep piezometers at sites that have nested piezometers in the May 2001 sampling round. This is because the constant source of feedlot NO_3 inputs has decreased, so the loading to the deeper part of the aquifer has decreased and, consequently, it is becoming less contaminated more quickly than the upper part of the aquifer.

It appears that from both a modeling and a measurement perspective that more than one-half of the NO_3 leaching that is occurring below the former feedlot may already have passed into the groundwater system. Nitrate contamination from the site will continue for the next 3 to 5 yr or more, depending on the weather. In this time, there will be a continuing improvement in groundwater quality as the “ NO_3 bulge” from the feedlot eventually percolates through the soil profile and mixes with the groundwater. The modeling shows that NO_3 contamination from unlined feedlots will be excessive and that it will take the aquifer below the feedlot a minimum of 5 to 10 yr to return to near-normal conditions if no remediation is done.

Future monitoring of the site would be useful to validate our predictions and further coupled vadose and groundwater models are needed to fully predict how $\text{NO}_3\text{-N}$ is transported in the subsurface as well as to determine the role that natural attenuation may play in $\text{NO}_3\text{-N}$ transport.

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