

## GROUNDWATER RECHARGE INTERFACE AND NITRATE DISCHARGE: CENTRAL CANTERBURY, NEW ZEALAND

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**Abstract:** Regional surveys of groundwater quality in Central Canterbury, from 1977 to the present, show widespread occurrence of nitrate contamination primarily from agricultural land use. Two reported analyses of these surveys, in 1984 and 2002, describe a general decrease of nitrate concentration with depth from the groundwater table to uncontaminated water at about 50-80 m. This high quality water is considered to be recharge from the large rivers that originate in alpine catchments. The occurrence and location of the interface between these distinctly different groundwater bodies is important for issues of access to high quality drinking water and for the quality of groundwater-fed surface waters in the down-slope areas of the Central Canterbury Plains. This paper describes the application of a prototype regional-scale model of nitrate transport in groundwater to investigation of the likely nature of this groundwater interface and the implications for quality of surface waters in the groundwater discharge zone. Results from a 2-D horizontal groundwater flow model indicate that river recharge is the major source of groundwater, from leakage rates per kilometre of river reach that are less than 1% of mean annual river flow. The vertical distribution of groundwater contaminant transport was examined with a combination of stream-function analysis of flow, and mixing-cell model simulation of contaminant dispersion. The results of a model demonstration with a realistic land use pattern, for a typical groundwater flow path, illustrate the formation of a dispersive, concentration interface between the two groundwater bodies. The demonstration example also shows how average nitrate-N concentration of about 8 mg/L in recharge from agricultural land use contributes to mean concentrations of 2-3 mg/L in the groundwater discharge zone, due to the influence of river recharge.

**Key Words:** Contaminant transport, stream function, mixing cell

## 1 INTRODUCTION

The Central Canterbury Plains (Figure 1) is a 2020 km<sup>2</sup> portion of the larger Canterbury Plains, bounded by the Waimakariri and Rakaia Rivers that cross the plains from the Southern Alps to the east coast of the South Island of New Zealand. The aquifers underlying these plains comprise Quaternary glacial outwash and alluvial fans up to about 500 m thickness. Recharge of the aquifers is from the large rivers that are perched above the groundwater table over much of their course across the plains, and from drainage of soil-water in response to climate and land use. This drainage from the land surface (land surface recharge) is contaminated with nitrate from agricultural land use under dryland (600-800 mm/y rainfall) and irrigated farming, whereas the river water quality is very high because of its origin in the high rainfall (> 5000 mm/y) alpine catchments. The nitrate discharge to the underlying groundwater affects not only the drinking water quality in wells but also contributes to the spring-fed streams in the lower portions of the plains around the margins of Lake Ellesmere. Nitrate-nitrogen concentrations in many of these streams is typically in the range 1 – 5 mg/L (Environment Canterbury, 2005), which is in excess of the nitrogen requirements for natural stream ecosystems.

Based on analysis of samples from approximately 600 wells during 1977-1983, Burden (1984) reported on the variation of groundwater nitrate concentration with depth into the aquifers of the Central Canterbury Plains and related these results to sources of recharge. At depths of about 50 - 60 m below the water table, nitrate concentration data decrease to values of <1 mg/L from near-surface values of up to 15 mg/L. A more recent analysis of groundwater nitrate concentration (Hanson, 2002) reaches similar conclusions but includes caution about the nature of the relationship between concentration and depth, as well as noting some occurrences of nitrate concentration greater than half the maximum allowable value (MAV = 11.3 mg/L nitrate-N) from wells deeper than 100 m. Stewart (2003) supports the existence of a water quality transition with depth, at about 60-80 m below the water table, based on nitrate-N, CFC, oxygen-18, and tritium data from Stewart et al. (2002). These results suggest the presence of an interface between an upper layer of groundwater that is contaminated by soil-water drainage and a lower layer of uncontaminated groundwater that has originated from river recharge.

This long-recognised occurrence of groundwater nitrate in the Central Plains has led to this region being selected as a pilot study area for the multi-agency programme, Integrated Research for Aquifer Protection (IRAP, 2004). This programme addresses research and technology for managing agricultural land use on the alluvial plains of New Zealand. The focus is on groundwater quality and the consequent effects on surface waters. The present paper describes progress to date in identifying and quantifying sources of groundwater recharge in the study area, estimation of nitrate leaching from land surface recharge, and construction of a prototype mathematical model of contaminant transport in the groundwater. This model is then used to examine the characteristics of a dispersive interface between contaminated and uncontaminated groundwater and the implications for water quality of groundwater-fed streams.

## 2 METHODS

Within the IRAP programme a decision has been made to model contaminant transport in groundwater on the basis of steady-state water flux. The reason for this decision is that the mesh size of a computational grid would be constrained by the requirement to simulate dispersive transport, and the number of cells for a three-dimensional model of a 2000 km<sup>2</sup> region would therefore be very large. For such a large number of computational cells, run times of a transient groundwater flow model are unlikely to be feasible for the purposes of the end users. The fundamental principle adopted for the modelling strategy is that a steady-state model would be able to address two questions: what is the long-term effect of a proposed change in land use; and what is the response time for this change? The latter question can be addressed by means of spatial distributions of contaminant transport times within a steady flow model. The option of using a steady water flux but transient transport model can be considered when computational times are tested at regional scale.

The following methods are a further development of the approach taken by Di et al. (2005) for their pilot scale study of land use impacts on groundwater quality, which was also for the Central Canterbury Region. In that study, the region is considered as a group of zones within each of which the interface between nitrate-contaminated groundwater from the land surface and uncontaminated groundwater from river recharge is described by the analytical solution to dispersion between upper and lower concentration boundaries. This one-dimensional approach, within each zone, does not include horizontal advective contaminant transport. In contrast, the present paper describes the use of stream-function modelling and mixing-cells to simulate advective-dispersive contaminant transport.



Figure 1. Central Canterbury, showing the locations of long-term observation wells and an example groundwater flow path (curved arrow) from the Rakaia River to Lake Ellesmere.

## 2.1 Nitrate Discharge from Agricultural Land Use

As a parallel development within IRAP, a comprehensive model of agricultural land use (FarmSim) is being constructed that predicts nitrate leaching through the vadose zone to groundwater from a range of crops, animal grazing and farm management options, on a daily computational basis, at paddock scale. The discharge of nitrate to groundwater will be accumulated as average annual values of drainage (mm/y) and associated nitrate concentration (mg/L), lumped at a horizontal scale corresponding to each farming enterprise.

For the purposes of demonstrating the current state of the groundwater transport model (AquiferSim) average annual values of drainage and nitrate concentration have been estimated from summaries of published values. Table 1 shows the selection for land uses in Central Canterbury that were applied to the model demonstration.

## 2.2 Identification of Groundwater Recharge

Land surface recharge (vertical drainage from the bottom of the soil) is estimated to an acceptable accuracy by means of daily water-balance models aggregated to annual amounts. The data required are daily values of climatic parameters, soil-water capacities, and models of evaporation by crops. In contrast, the available data about recharge to groundwater from the rivers that intersect their alluvial plains range from sparse to non-existent. Although river recharge can be the dominant source of groundwater in alluvial aquifers, the associated leakage rates from the contributing rivers may be within the gauging error of streamflow measurement even over reaches of several kilometres. An alternative approach is to use groundwater models to identify river recharges as a sub-set of the unknown parameters.

The selected method for this Central Canterbury example was developed by Bidwell (2003), and used for the pilot study of Di et al. (2005), in which the piezometric records of groundwater wells are analysed to separate out the average piezometric effects of river recharge and land surface recharge. The separation process is determined by the dynamics of aquifer response, for which most of the piezometric variation is caused by variation in land surface recharge whereas the response to variation in river recharge is heavily damped and treated as a steady value. Monthly values of piezometric data from 12 wells in Central Canterbury (locations shown in Figure 1), with records up to about 25 years, were analysed to produce 12 pairs of response data for the groundwater flow model.

### 2.3 Groundwater Flow Model - Horizontal

A steady-state, finite-difference, groundwater flow model was implemented within Microsoft Excel spreadsheet for the region (Figure 1) bounded by the foothills, Waimakariri and Rakaia Rivers, Banks Peninsula, Lake Ellesmere, and the sea coast. The computational mesh size was 2.5 km x 2.5 km, and the iterative calculation mode within Excel was used to solve the finite-difference flow equations. Boundary conditions were set as: no-flow at the foothills; specified-head around Banks Peninsula (drainage system), lake, and coast; and a simple, linear river-aquifer function applied to the river boundaries, dependent on river bed elevations estimated from 1:50,000 topographical map.

The model parameters were held to a minimal set, given that there were only 24 data values available for calibration. The selected parameters were a bulk value of aquifer transmissivity  $T$ , maximum river recharge per length of reach for the Waimakariri ( $R_W$ ) and Rakaia ( $R_R$ ), with half this value diverted to each side of the rivers, and a conductance parameter  $C$  in the river-aquifer function:

$$\text{River recharge} = \min[0.5R_{W \text{ or } R}, C \times (\text{piezometric gradient at river boundary})] \quad (1)$$

The 12 values of river recharge piezometric response were used to estimate the parameter ratios ( $R_W/T$ ), ( $R_R/T$ ), and ( $C/T$ ), because transmissivity cannot be identified with unknown recharge in a steady-flow model. Then the 12 values of total piezometric response were used to estimate transmissivity  $T$ , in conjunction with the water balance model estimate of land surface recharge for the region. A uniform value of 200 mm/y was used for the whole region. The total piezometric response was used rather than the land surface recharge response values because of the influence of river-aquifer interaction. The Excel Solver tool was used for optimisation of model parameter values, with minimising sum-of-squared-errors as the objective function. For the purposes of the initial demonstrations, the effects of pumped abstractions have not been included in the recharge estimates.

### 2.4 Groundwater Flow Model - Vertical

The proposed strategy for relating groundwater quality effects to particular locations of land use is to conduct a gradient search of the piezometric surface of the horizontal groundwater flow model in order to identify the groundwater flow path through the location of interest. The vertical distribution of groundwater flow along this flowpath is modelled by means of the stream-function method (e.g., Bear and Verruijt, 1987). The stream-function approach is being increasingly recognised (e.g., Zheng and Bennet, 1995) for its value in modelling contaminant transport because of the accuracy of the solutions (Frind and Matanga, 1985) and the ability to directly calculate contaminant transport paths and travel times. Stream-function theory applies only to steady-state flow and the computational methods are the same as for piezometric head models (Fogg and Senger, 1985), including heterogeneous aquifer properties, but with different kinds of boundary conditions.

The flux-type boundary conditions around the vertical section of the aquifer may include known recharges or be calculated from the slope of the piezometric surface of the horizontal flow model. Discharges can be set to maintain water balance and located at best estimates of positions such as spring-fed surface waters. Pumped abstractions need to be specified on a boundary, because flow cannot enter or leave a stream-function domain, and the placement of these boundaries is part of the model "art".

### 2.5 Contaminant Transport Model

Advective-dispersive contaminant transport in the two-dimensional, vertical, steady groundwater flow model is simulated by means of the mixing-cell principle (e.g., Rao and Hathaway, 1989). The theory of the mixing cell is based on conservation of water and contaminant mass, and complete mixing, within each computational cell of the model. The amount of dispersion is controlled by selecting the dimensions of the cell. In the AquiferSim prototype model, the horizontal and vertical components of water flux entering and leaving each cell are copied from the stream-function model and used with adjacent cell concentrations for iterative computation of the steady-state concentration distribution of the 2-D vertical domain. In this approach to date, transverse horizontal dispersion has been ignored but there is potential for introducing this component by mixing between the vertical planes of adjacent groundwater flow paths, but this would be relevant only where there is a concentration gradient. For the present study, the primary concentration gradient is vertical due to contrasting qualities of predominantly horizontal groundwater flow from different recharge sources.

The demonstration model of the Excel prototype is 200 cells long and 50 cells deep, and was used to simulate the vertical section through a groundwater flow path (Figure 1) that is 50 km long and 300 m deep. There are no restrictions on the shape of this vertical domain but in the absence of information about the

thickness profile of the Central Canterbury aquifer a rectangular shape has been assumed. The cell dimensions, which control dispersion, are therefore 250 m horizontal and 6 m vertical. A contrast between horizontal and vertical hydraulic conductivity was specified as  $K_H/K_V = 100$ .

Development of a groundwater interface within this contaminant transport model was quantitatively tested for a simple case. In this test, river recharge into the top upstream cell was set equal to the total uniform land surface recharge into the top row of cells, and all groundwater discharge exited uniformly through the downstream vertical column of cells. The resulting vertical concentration distribution  $C(z)$  was examined by hypothesising that the location and dispersion of the interface is determined as the cumulative effect of surface recharge transporting a step change in concentration  $C_0$  as a one-dimensional vertical process. The vertical concentration profiles at horizontal locations of 50, 100 and 200 cells were fitted to the analytical solution (Banks, 1974):

$$\frac{C(z)}{C_0} = \frac{1}{2} \left\{ \operatorname{erfc} \left[ \frac{z-F}{(2z_0F)^{0.5}} \right] + \exp \left[ \frac{2z}{z_0} \right] \operatorname{erfc} \left[ \frac{z+F}{(2z_0F)^{0.5}} \right] \right\} \quad (2)$$

in which  $F$  is the depth of the interface and  $z_0$  is the vertical cell dimension. The expected value of vertical dispersivity for this 1-D mixing-cell model is  $0.5 z_0$ .

As a demonstration of the effect of a mixture of land uses on the quality of groundwater and groundwater-fed surface waters, an arbitrary but realistic pattern of agricultural land uses and river recharge was applied to the model (Table 1). The value of Rakaia recharge in Table 1 (30.68 m/y) was calculated as being half (Section 2.3) of the model parameter value  $R_R$  (Table 2) allocated to a 500 m<sup>2</sup>/m recharge strip (two horizontal cells). Groundwater discharge was allocated to a two kilometre margin of land surface (8 horizontal cells) and a depth of 24 m (four vertical cells) at the downstream end of the model, to represent groundwater-fed surface waters.

Table 1. Recharge and water quality parameters used for the model demonstration.

Land use	Recharge (m/y)	Nitrate-N (mg/L)	Model cells.	Flowpath distance (km)
Rakaia recharge	30.68	0	2	0.5
Sheep	0.15	3	38	10.0
Dairy	0.25	12	38	19.5
Forest	0.10	1	38	29.0
Sheep	0.15	3	38	38.5
Crops	0.20	15	38	48.0
Discharge zone			8 horiz., 4 vert.	50.0

### 3 RESULTS

#### 3.1 Identification of groundwater recharge

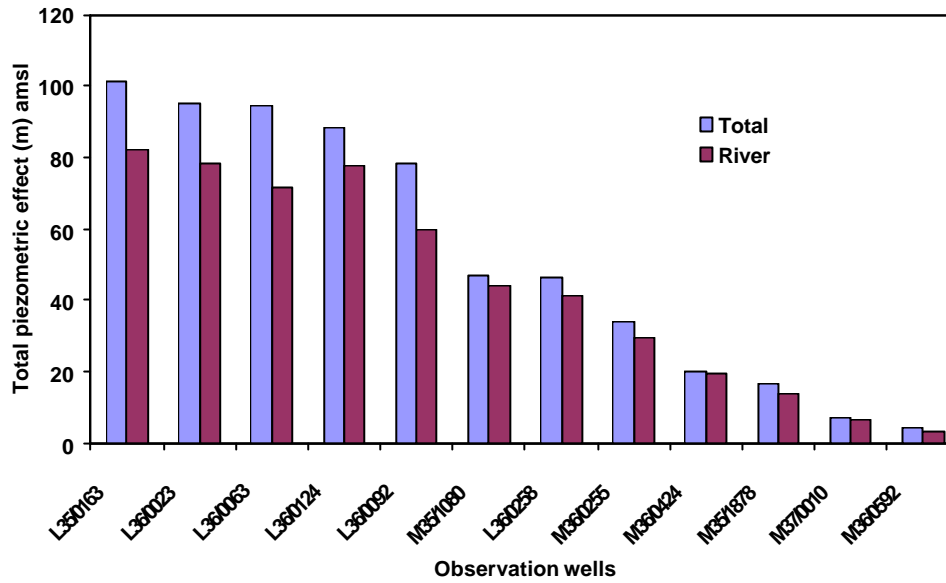
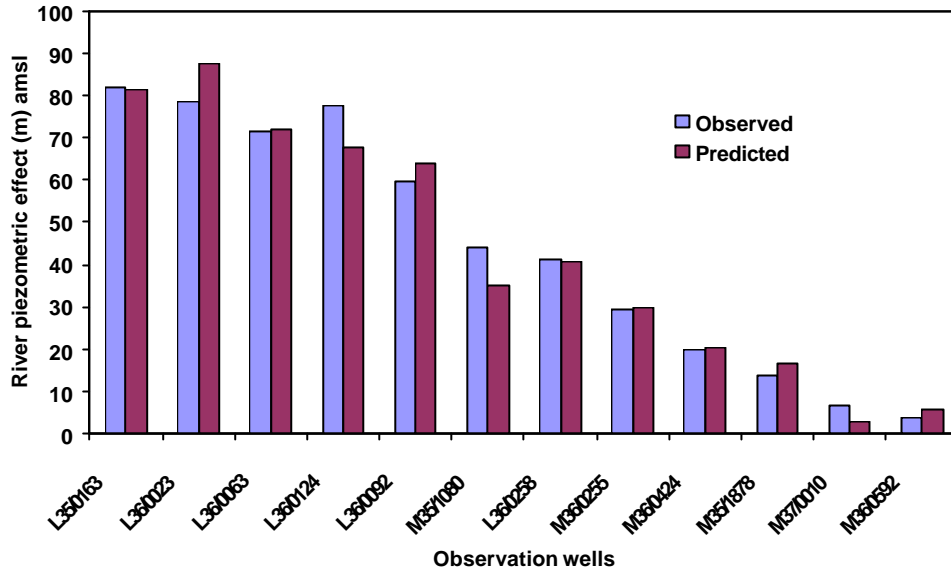


Figure 2. Steady-state piezometric effects, relative to mean sea level, of river recharge and total recharge for observation wells in Central Canterbury at locations shown in Figure 1.

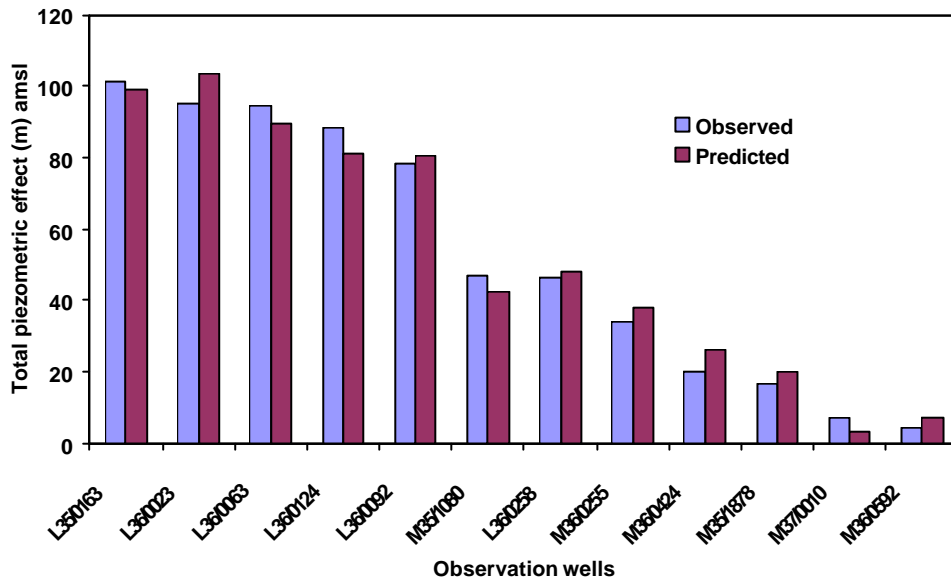
#### 3.2 Groundwater flow model – horizontal

Table 2. Parameter values of the steady-state, 2-D horizontal, groundwater flow model.

Parameter	Value
Waimakariri River recharge – $R_W$	0.89 m <sup>3</sup> /s per km of river
Rakaia River recharge – $R_R$	0.97 m <sup>3</sup> /s per km of river
Conductance – $C$	8400 m <sup>2</sup> /d
Transmissivity – $T$	18000 m <sup>2</sup> /d



(a)



(b)

Figure 3. Model predictions of steady-state, piezometric response to, (a) river recharge and (b) total recharge, for the 12 observation wells shown in Figure 1

### 3.3 Groundwater flow model – vertical

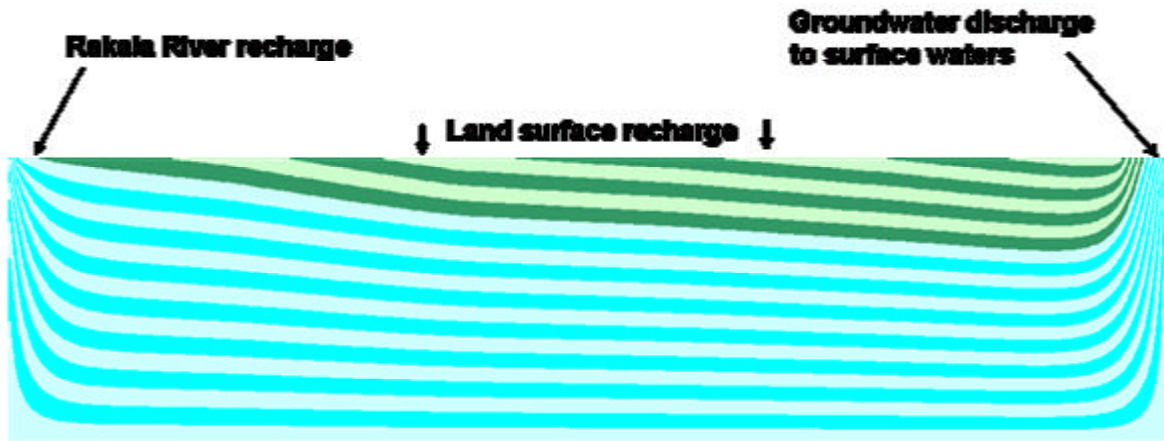


Figure 4. Stream-function model of the vertical distribution of groundwater flow along the flow path in Figure 1. The upper (darker) zone is groundwater flow from land surface recharge, and the lower zone is from river recharge.

### 3.4 Contaminant transport model

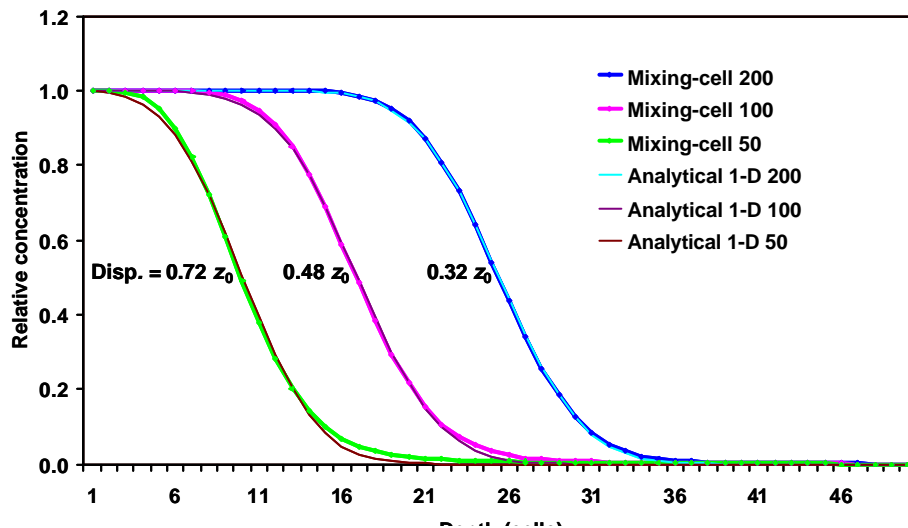


Figure 5. Comparison between mixing-cell model and 1-D vertical analytical model predictions of vertical concentration profiles at 50, 100, and 200 cells horizontal distance along the interface between groundwater from land surface and river recharge



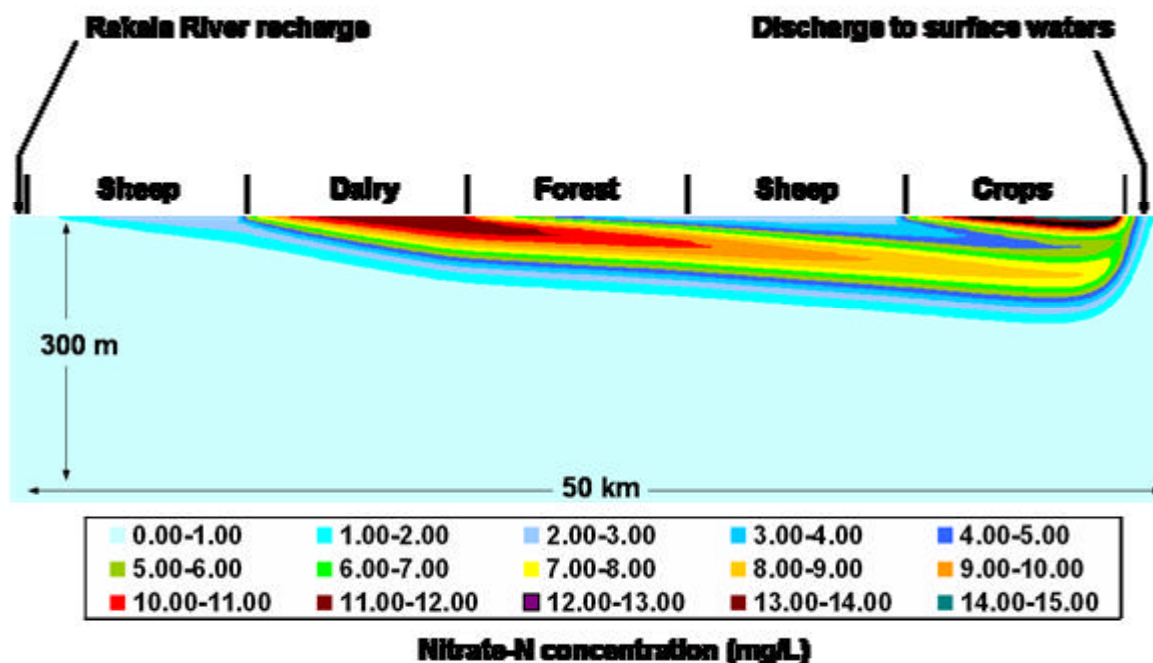


Figure 6. Vertical distribution of nitrate-N concentration along the groundwater flow path in Figure 1.

#### 4 DISCUSSION AND CONCLUSIONS

The significance of river recharge to the Central Canterbury aquifers, relative to total recharge, is indicated by the magnitudes of the respective steady-state piezometric effects, relative to mean sea level, in Figure 2. For the five wells in the upper part of the Central Plains (higher piezometric values) the piezometric effect of river recharge, relative to sea level, is at least 75 % of the total effect. The proportionate effect is even more in the wells of the lower plains, but this is mainly due to the damping effect of river-aquifer interaction on response to land surface recharge in this part of the region.

The four-parameter, steady-state, horizontal, groundwater flow model predicts these piezometric effects (Figure 3) with errors of up to 12 % for river and 8 % for total effect, for the five upper plains wells. Prediction is generally poorer for the lower plains wells because of interaction effects and the expected smaller amplitudes of the piezometric record closer to the coastal boundaries. The quality of these model predictions gives some confidence in the estimates of the maximum river recharge parameters. In the upper plains, where the rivers are perched above the groundwater table, the (assumed uniform) recharge rates are 0.89 m<sup>3</sup>/s/km (Waimakariri) and 0.97 m<sup>3</sup>/s/km (Rakaia). These values of recharge per kilometre of river reach are, respectively, 0.74 % and 0.47 % of mean annual river flow. Therefore, even over a reach of 10 km the flow loss from the rivers is comparable to the expected errors of flow gauging. This means that accurate measurement of river recharge is very difficult, although it is the more significant groundwater source. The infiltration rate of this recharge through the river bed, for the assumed model structure (Table 1), is 84 mm/d.

The stream-function plot (Figure 4) illustrates the general vertical distribution of groundwater flow patterns for the selected example flow path from the Rakaia River in the upper plains to a discharge zone near Lake Ellesmere. The flow pattern clearly shows the major contribution (66 %) of river recharge to total groundwater flow. This example flow pattern would be distorted to some extent by variations in aquifer thickness and heterogeneities in porosity and hydraulic conductivities, but the overall distribution can be expected to be a robust illustration (Frind and Matanga, 1985). The stream-function model, together with data on aquifer thickness and porosity, is the key to estimating contaminant transport times and the dynamics of water quality response to changes in land use.

The average nitrate-N concentration in groundwater discharge to surface waters is not dependent on aquifer properties but can be calculated on a mass flow basis from the information given in Table 1. The flow-weighted average nitrate-N concentration in the land surface recharge for the demonstration example is 8.24 mg/L, and with the contribution of uncontaminated Rakaia recharge water the resulting average concentration of the discharge to surface waters is 2.84 mg/L. This value is well within the range of monitored data for the spring-fed streams near Lake Ellesmere (Environment Canterbury, 2005).

The vertical distribution of nitrate contamination within the aquifer is important with regard to access to suitable drinking water. This distribution depends on aquifer properties such as thickness, distribution of porosity and hydraulic conductivity, and the dispersion process. Detailed knowledge of these properties remains poor even for regions such as Central Canterbury which have relatively good databases. Current process research within the IRAP programme is directed at improving knowledge of these aquifer properties.

The dispersive interaction between the contaminated recharge from land use and the uncontaminated river recharge has been quantified in terms of a scale length called dispersivity. This dimension cannot be measured directly in the field but is obtained as a calibrated model parameter, which is itself considered (e.g., Gelhar et al., 1992) to vary with the scale of the groundwater transport distance. Figure 5 shows a comparison between interface dispersion within the two-dimensional, vertical, contaminant transport model, caused by the selected computational grid dimensions, and an equivalent one-dimensional mixing-cell model of vertical dispersion. The results show that the vertical component of dispersivity, which is usually considered to be a transverse component of horizontal dispersion along the flow direction, can be accounted for as simple dispersion in the direction of vertical recharge from the land surface. In this case, dispersivity of the interface varies with the horizontal transport scale and appears (Figure 5) to be approaching a value that is less than the expected value of half the vertical grid dimension. These model results suggest that even after a transport distance of 50 km the nitrate concentration is decreasing in a vertical direction within the interface at maximum rates of about 10 % of the original concentration per 6 m of depth into the aquifer.

The effect of this relatively steep interface on access to clean drinking water is illustrated by the model results for the demonstration case (Figure 6). Most of the example aquifer contains uncontaminated water but deep wells are necessary for low risk access. Pumped abstraction of water from this uncontaminated region would be expected to reduce its thickness and hence increase the cost of access. The effects of various land uses can be distinguished for quite long transport distances downstream in the aquifer. In the groundwater discharge zone the model predicts nitrate-N concentrations in the range of 0.08 – 10.32 mg/L, around the flow-averaged value of 2.84 mg/L.

The results of the study to date suggest that the selected modelling approach can simulate the current knowledge of nitrate transport processes within the aquifers of the Central Plains and help explain various observations from the monitoring networks. The mixing-cell approach to contaminant transport can also be easily augmented to include first-order reaction models of nitrogen transformations, when more data about the occurrence of these processes become available. An additional feature of the model is that by representing groundwater age as a solute that undergoes first-order growth (Goode, 1996), the mixing-cell approach can be used to address questions about contaminant transport times and likely response times of changes in groundwater quality to changes in land use. However, it has become clear that this capability for modelling the nitrate transport and transformation processes is limited by knowledge about the magnitudes and distribution of basic aquifer properties and some aspects of recharge hydrology.

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