Groundwater Flow and Nitrogen Transport Modelling of the Northern Lake Taupo Catchment

Environment
Waikato

REGIONAL COUNCIL

Acknowledgements

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Abstract

The quality of water in Lake Taupo is under threat, largely as a result of increasing nitrogen from farming in its catchment. Groundwater flow and contaminant transport modelling of the northern catchment using Modflow and MT3D indicate a substantial lag (~250 years) before the full impact of current land use would be realised. Pre-development nitrogen flux from groundwater is estimated at about 65 tonnes annually with about 3,500 tonnes of nitrogen stored in the groundwater system. After 35 years of 'current' land use, annual flux from groundwater is estimated to be nearly 175 tonnes at which stage about 10,000 tonnes of nitrogen may be stored in the system. Under simulated equilibrium with current land use, about 300 tonnes of nitrogen flux from the northern groundwater system is predicted annually. Some 18,000 tonnes of nitrogen would be stored in the aquifer. Assumptions include conservative nitrogen transport and porosity reflecting fracture flow in the deeper western model domain. Modelling results are broadly consistent with water age dating and stream water quality information. The extent of groundwater denitrification, currently thought to be minor, is an aspect of ongoing research. The proposed reduction of the manageable load to the lake was found to provide useful mitigation in the northern catchment.

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Executive summary

Although water quality in Lake Taupo is very high, it is under threat from increasing nitrogen loads, predominantly from farming within its catchment. Groundwater flow and contaminant transport modelling were undertaken to estimate the potential future nitrogen load from groundwater and the time it will take current land use impacts to reach equilibrium. The efficacy of a proposed mitigation initiative was also investigated.

A three dimensional, finite difference, numerical model was used to investigate groundwater flow and nitrogen transport in the northern Lake Taupo catchment. Flow conditions were modelled in steady state, with contaminant transport being transient.

Flow calibration indicates groundwater velocities range from about 0.02 to 1.5 m d^{-1} with highest rates in simulated fractured ignimbrite and porous media in the Kinloch area. A groundwater recharge regime is dominant throughout the model and streams predominantly gain flow from groundwater. Groundwater travel time to the lake is in excess of 100 years along some flow paths.

Modelling indicates that, before farm development, about 65 tonnes of nitrogen was discharged from the northern catchment to the lake annually, with some 3,500 tonnes of nitrogen being stored in the groundwater system.

Current land use effects were simulated by modelling farm influenced nitrogen loading for a period of 35 years. Conservative nitrogen transport was assumed, given current knowledge of denitrification in the catchment. A reasonable fit was achieved with observed groundwater concentrations and the trend in the Whangamata Stream. Nitrogen concentrations in the Mapara Stream were, however, over-estimated. The discharge of nitrogen from groundwater after 35 years is estimated at about 175 tonnes annually, at which time about 10,000 tonnes of nitrogen would be stored in the groundwater system.

Extended model simulation, assuming current land use loading, indicates a period of about 250 years is required for full nitrogen equilibrium to be achieved. The estimated mass of nitrogen stored in the northern catchment groundwater system at that stage would be about 18,000 tonnes, with an annual flux of about 300 tonnes. It is predicted that about half the total manageable nitrogen load of 235 tonnes is still to come. Given about 30 per cent of groundwater discharge to the lake from the northern catchment is via streams, the contribution still to come from direct groundwater seepage is estimated to be about 90 tonnes. This assumes no attenuation and is hence worst-case, given some denitrification can be expected.

Representation of fractured ignimbrite in the west of the model by lower porosity formation has the effect of reducing the capacity of the groundwater system to store nitrogen, hence nitrogen flux to the lake increases more quickly.

Modelling of a proposed initiative to reduce manageable nitrogen load was carried out by reducing farm loading by 20 per cent after 35 years of simulation at current rates. The calculated nitrogen concentration trends indicate that this is a useful mitigation measure. It would limit the extent of long-term loading increase otherwise expected from northern catchment groundwater.

The modelled 20 per cent reduction in nitrogen load predicted an immediate small reduction in nitrogen concentration in the Whangamata Stream, yet a subsequent slow long-term increase. The mechanism for this in a stream with a mean residence time of about 80 years was investigated. This was done by differentiating nitrogen input to the model before and after the 20 per cent reduction in manageable nitrogen load. It was found that mixing could account for the predicted trend, as ambient groundwater was slowly flushed out of the groundwater system. Reasonable agreement was found

between the model predicted fraction of younger groundwater and that calculated using tritium dating techniques.

The application of modelling supported by age dating techniques is shown to be effective in demonstrating the insidiously slow, long-term build-up of nitrogen mass in, and increased flux from, the northern catchment groundwater system.

1   Introduction

Although Lake Taupo is a very large oligotrophic lake (622 km^2), its very high water quality is under threat. For several years there has been scientific evidence indicating that, not only the clarity and purity of lake water is declining, but nutrient loads from streams are increasing (Vant and Huser, 2000). There is also evidence of increasing land use impacts on groundwater, particularly in the northern and western lake sub-catchments. Higher nitrate nitrogen concentrations are found in groundwaters dominated by recent recharge (Hadfield et al., 2001).

Historically, the lake has extremely low levels of nitrogen, which has limited the growth of nuisance plants in the waters. Much of the traditional scrub land around the lake has, however, been converted to agricultural uses over the last 50 years. This has led to increases in nitrogen being leached and transported to the lake.

Groundwater is the primary link for the transport of nutrients derived from land use activities to the lake. Land use impacts are clearly evident in groundwater. Further decline in water quality can be expected as a result of land use intensification.

In order to protect the lake, which is a national icon, impacts from land use must be restricted. An Environment Waikato regional policy variation has been proposed to reduce the manageable nitrogen load to the lake by 20 per cent with the aim of ultimately maintaining the lake's water quality at its current level. Actions proposed to achieve a 20 percent reduction in nitrogen load include:
- conversion of pastoral land to alternative low-nitrogen land use, such as forestry
- the introduction of new farming practices and crops to the catchment
- upgrades to wastewater systems around the lake.

The mass contribution of nitrogen from many streams in the Lake Taupo catchment continues to increase (Vant and Smith, 2004). These streams are typically base flow dominated. This, and the direct contribution of groundwater to the lake, are expected to cause further decline in lake water quality until it comes into equilibrium with land use changes. The extent and lag of these impacts are the subject of this investigation.

This report describes the development and findings of a three-dimensional numerical flow and nitrogen transport model of the northern Lake Taupo catchment.

2   Scope and purpose

The purpose of developing this numerical flow and contaminant transport model is to improve Environment Waikato's understanding of the groundwater systems in the northern catchment and their contribution of nitrogen to the lake. This enables more informed management of the lake catchment and provides a more solid basis for expectations in respect to mitigation of land use effects.

A separate numerical model of the western catchment was also being developed with Geological and Nuclear Sciences Limited. The northern and western catchments of Lake Taupo are modelled, as these areas have the greatest potential for agricultural intensification (Ministry of Agriculture, 1997). The development of two separate models allows comparison of approaches and results, enabling exchange of ideas between model developers and corroboration of findings.

The objectives of the modelling are to answer the following three basic management questions identified during development of the conceptual model:
- how long will it take for the effects of current land use to reach an equilibrium?
- what nitrogen load is still to reach the lake?
- what effect may a 20 per cent reduction in manageable nitrogen loading have?

The above questions require the following hydrogeologic aspects to be addressed in the flow and contaminant transport modelling:

Flow
- groundwater flow directions
- groundwater flux
    – contribution of groundwater to streams
    – contribution of groundwater directly to the lake
    – groundwater travel time.

Contaminant transport
- nitrogen concentrations and mass loadings resulting from various land use scenarios
- contaminant (nitrogen) travel times and temporal trends
- contaminant (nitrogen) transformations / losses.

Transformation via denitrification is the subject of ongoing work, although relevant preliminary results are discussed.

3 Hydrogeologic characteristics

3.1 Introduction

This section provides an overview of the hydrogeologic characteristics of the northern lake catchment, including groundwater quality and age. This information is the basis for model input discussed in sections six and seven. It is not intended as detailed hydrogeologic reporting.

3.2 Hydrogeology

The local geology, which comprises young (<0.4 Ma) locally derived rhyolitic pyroclastic formations, can be broadly divided into three main groupings. The Whakamaru Group ignimbrites toward the west are poorly to moderately welded with low fracture density, but flow is nevertheless essentially fracture controlled. To their east is a more complex grouping considered loosely as 'rhyolitic pyroclastics'. These are both overlain by unwelded Oruanui Ignimbrite, which varies in thickness generally up to some 30 metres. There are two notable occurrences of rhyolite flow materials associated with dome development on either side of the Whangamata Valley (Kinloch). A geologic map of the area is presented in Figure 1 and cross-section in Figure 2 (Hadfield et al., 2001).

Hydraulic conductivity for the simplified hydrogeologic groupings was estimated from pump and slug testing undertaken, where possible, within the northern catchment. The results of these are summarised in Table 1 below. Hydraulic conductivity was generally found to be low. Localised Whakaroa Rhyolite in the Kinloch area exhibits highly variable fracture dependent permeability ranging up to 120 m d^{-1} and has been the target of more recent water supply development. Fractured rock formations, particularly Whakamaru ignimbrite extending westward in the model domain to the Kawakawa sub-catchment, have substantially lower aquifer storage and higher flow velocities, as discussed in subsequent sections.

The map contains a key with the following entries:

Key

- post-Taupo alluvium (<1.8 ka)
- Taupo ignimbrite (1.8 ka)
- Oruanui ignimbrite (26.5 ka)
- Huka Falls Formation (>26.5 ka)
- Rhyolite lava domes (3–320 ka)
- Rhyolite dome-related pyroclastics
- Rhyolite pyroclastics (26.5 to 150 ka)
- Basalt and andesite lavas and pyroclastics (100 to 300 ka)
- Rhyolite pyroclastics (150 to 320 ka)
- Whakamaru-group ignimbrites (320–340 ka)
- Pre-Quaternary rocks (>1.6 Ma)

Lake Taupo

Taupo Catchment Geology Map

0 2 4 Kilometres

Topographic data obtained from Eagle Technology Group Ltd under licence from Land Information New Zealand (LINZ).
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Figure 1: Geological map (Hadfield et al., 2001)

Figure 2: Geological cross-section of the northern catchment (Hadfield et al., 2001)

Legend:
- Taupo ignimbrite (1.8 Ka)
- Oruanui ignimbrite (26.5 ka)
- Huka Falls Formation (>26.5 ka)
- Rhyolite lava domes (3 - 320 ka)
- Rhyolite pyroclastics (26.5 to 150 ka)
- Basalt and andesite lavas and pyroclastics (100 to 300 ka)
- Rhyolite pyroclastics (150 to 320 ka)
- Whakamaru-group ignimbrites (320 - 340 ka)
- Pre-Quaternary rocks (>1.6 ka)

Table 1: **Summary of aquifer hydraulic conductivity estimates from testing (m d^{-1})**

Formation	Median	Mean	Standard deviation	Minimum	Maximum
Whakamaru ignimbrite	0.01	0.38	0.66	0.007	2.02
Rhyolitic pyroclastics	0.93	2.16	3.89	0.008	17.3
Oruanui ignimbrite	0.31	2.91	4.65	0.03	13.9

Leakage values have been determined from aquifer flow testing at four locations as listed below (Table 2). Further indication of variable connection between hydraulic units comes from head differences between paired wells at the same location (Table 3) and groundwater age dating. Dating of groundwaters at paired monitoring wells 72.1009 and 72.383 show estimated mean residence times of 2 and 41 years respectively. Paired wells 72.331 and 72.377 were estimated to have mean residence times of 24 and 177 years respectively. These dates indicate variable vertical leakage within the domain. Aquitards, such as paleosols can be locally important.

Table 2: **Aquifer leakage estimates**

Well	Map reference	Formation	Well depth (m)	Leakage (d^{-1})
72.383	T17:546-839	Oruanui	38.4	0.02
72.463	T17:488-850	Whakamaru	130.0	0.005
68.317	T17:606-812	Pyroclastic	104.0	0.00013
68.718	T17:642-806	Oruanui	67.0	0.029

Groundwater flow in the study area is consistent with topography although more subdued (Figure 3). The catchment divide is critical for management implications. There is no discrepancy evident between surface and groundwater catchments. For example, the western end of Whangamata Road coincides with the northern Lake Taupo catchment divide. Investigation here, at the Waihora airstrip, immediately on the southern side of the road, showed the water table slopes southward toward the lake. The distribution of contaminants in groundwater at that site supports the indicated groundwater flow direction (Standish, 2004). This is consistent with the larger scale groundwater contour mapping. Surface seepages on the southern side of the Whangamata Rd at that location flow toward the lake via the Waihora Stream, whereas on the other side of the road flow is to the north via the Mangakino Stream

Table 3: **Hydraulic head differences**

Wells	Map reference	Respective well depths (m)	Head difference (m)
72.1008 and 72.514	T17:546-844	8 and 171	3.0
72.402 and 68.718	T17:642-807	51 and 67	8.0
72.354 and 72.352	T17:616-855	4.16 and 152.4	46.4
72.331 and 72.513	T17:508-847	5.8 and 160	2.4
72.1388 and 72.348	T17:570-837	7.5 and 97.5	80.3
72.1389 and 68.430	T17:613-849	26.2 and 168	58.0

A recharge regime exists generally throughout the area with a highly significant positive relationship between depth to static water level and well intake depth. The lake acts as a sink for groundwater, which is recharged from rainfall in the catchment.

Figure 3: Lake Taupo's northern catchment groundwater flow and piezometric surface.

3.3 Water chemistry

Groundwater from 44 wells sampled for analysis of nutrients and major ion chemistry showed relatively uniform sodium bicarbonate dominated groundwater, typical of rh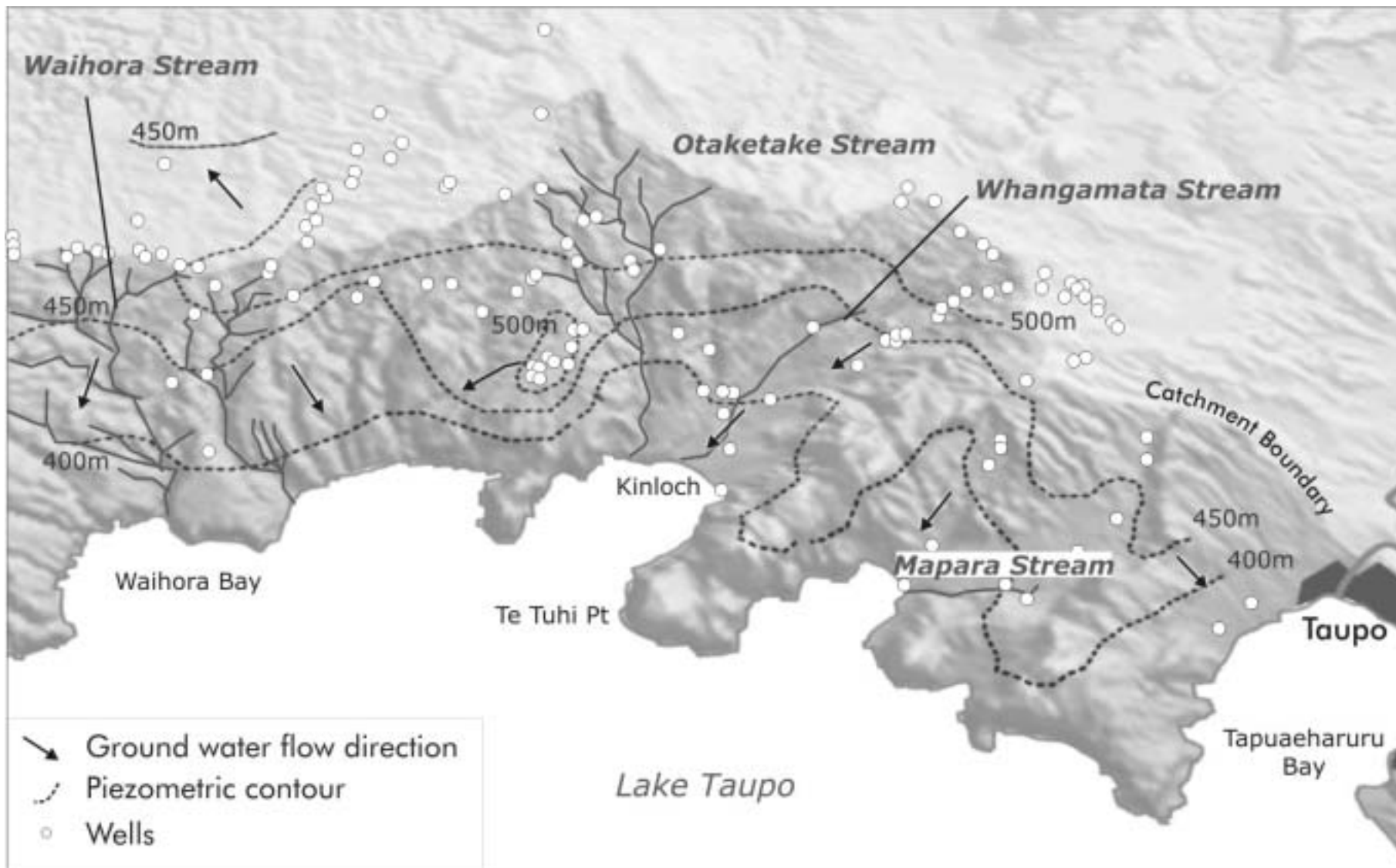yolitic formation (Hadfield et al., 2001). There was, however, clear evidence of land use impacts with nitrate-N commonly elevated above ambient (or undeveloped) conditions. Empirical data from older groundwater, samples from undeveloped areas, as well as modelling suggests ambient nitrogen concentrations of less than 1 g m^{-3}. Very high nitrate-N (~36 g m^{-3}) at one well was found to result from point source contamination from a nearby woolshed (Hadfield and Barkle, 2004).

Anaerobic or poorly aerobic conditions are indicated at about 20 per cent of wells sampled by the nitrate ammonia couple, presence of dissolved iron and manganese and Eh/pH measurement (Hadfield et al., 2001). Preliminary investigation of denitrification was undertaken by injecting nitrate and bromide into five available shallow piezometers, using the method of Trudell et al., (1986). This indicated that where redox conditions exist for denitrification, it may nevertheless be limited by a lack of available carbon (Appendix III). One site of five where tests were undertaken in shallow groundwater showed substantial denitrification. This was influenced by the uncommon presence of peat. Denitrification may be more significant in deeper groundwater isolated from the water table. There is further evidence of groundwater denitrification in the study area, as mentioned in section 8.

3.4 Dating

The age of groundwater in the northern and western Lake Taupo catchments was estimated by measuring the cosmogenic isotope tritium (half life 12.3 years) in samples, as well as atmospheric trace gases (CFCs and SF$_6$), which can assist to resolve age ambiguity. Age interpretation of groundwater depends on mixing processes underground. Groundwater and stream samples comprise a mixture, rather than a discrete age, reflecting variable flowpaths (Maloszewski and Zuber, 1982). It is useful to estimate not only the mean residence time (MRT) but also the fraction of water recharged since farming was established in the Taupo area in the 1960s (% young fraction (YF)). An indication of potential future nitrogen concentration may be obtained by considering an increase inversely proportional to the % YF (Hadfield et al., 2001).

A total of 25 groundwater samples from the northern and western catchments were analysed for tritium, CFCs and some for SF$_6$ tracer concentrations. MRTs ranged from about 2 to 177 years. The median MRT for northern wells sampled was about 45 years. Higher nitrate-nitrogen concentrations were shown to occur in younger groundwater (Figure 4). Not all of the young waters, however, have high nitrogen concentrations as the farming influence is not uniform across the study area and groundwater from some wells sampled may be influenced by anaerobic conditions. Also older MRT waters can have nitrate concentrations slightly above ambient condition due to some input of younger groundwater. The median %YF of groundwater samples from the northern catchment was just over 50% (Morgenstern in prep.).

Tritium samples were also collected from 11 streams' mouths in the northern and western sub-catchments in 2001/02. MRT ranged from about 2 to 85 years with the oldest (and lowest % YF) occurring in the 'rhyolitic pyroclastic', north-eastern sub-catchments. Streams in the western, Whakamaru Group, sub-catchments, which have steeper relief and higher rainfall were found to have younger MRTs. Northern catchment streams modelled range in MRT from 42 to 84 years with the exception of the Waihora Stream which has a MRT of about 10.5 years (Morgenstern, in prep.). Potential future nitrogen loads from these streams may be approximated based on the percentage of farming input to come (Vant and Smith, 2004). Tritium sampled at multiple sites along streams within nine selected sub-catchments showed little age variation within catchments, which implies there are uniform drainage patterns (Piper, 2004).
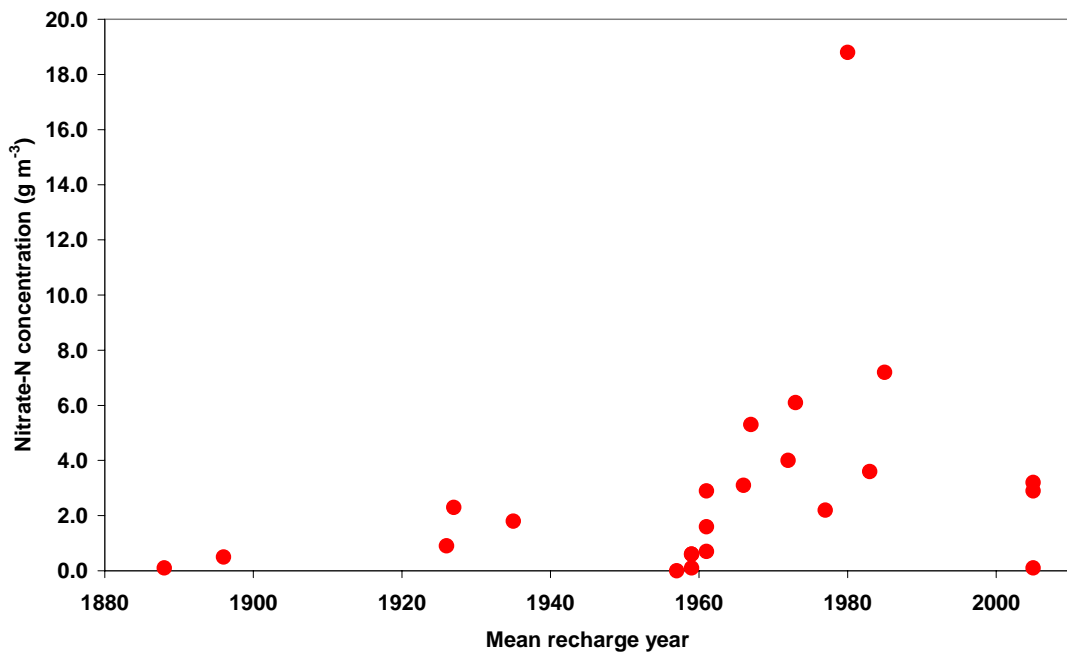
Figure 4: Nitrate-N concentration of groundwater samples versus mean recharge year

4 Conceptual model

Models are simplifications of the natural system and include numerous assumptions. The flow model is based on steady state conditions, due in part to there being limited available temporal data, but also to limit model complexity with a focus on contaminant transport. Contaminant transport is modelled in transient condition to enable lag and travel times to be investigated.

The model domain extends from the Waihora sub-catchment in the west to the outlet of Lake Taupo into the Waikato River in the east. Flow is from the topographically high catchment boundary toward the lake, which acts as a local sink (Figure 5).

Figure 5: Domain topography (ten times vertical exaggeration)

Inputs to the model are effective rainfall and solute (nitrogen) leaching from the land surface. Recharge is estimated from available rainfall data and indicates a strong orographic influence. Recharge to the model increases westward across the model domain from 450 to 900 mm annually, with evapotranspiration being about 50 to 65 per cent.

Contaminant (nitrogen) is added as spatially variable diffuse loading, without consideration of localised point sources. The concentration of contaminant addition is determined on the basis of characteristic nitrogen loadings from mapped land use and effective rainfall.

The Mapara, Whangamata, Otaketake and Waihora Streams are included in the model (Figure 3). Flow from other streams is less well known, although collectively significant. For example, flow in the Omoho, Tutaeaua and Kawakawa Streams together is about 25 per cent of the modelled stream flow. Almost all stream reaches gain water via seepage from the model domain.

Water use has not been included in the model as it was considered to be relatively insignificant in respect to flow and transport conditions.

Calibration datasets include water levels measured at water supply wells and monitoring piezometers throughout the domain. Nitrogen concentrations were also measured at numerous wells to provide transport calibration (Appendix I). Measured flow and water quality within the various reaches of the four streams listed above also constitute important calibration data.

A trade-off was recognised between developing a more complex model with potential convergence problems versus a simpler model with limitations representing aspects including stream variation and spatial variation in nitrogen transport. Complex topography in the study area provided added challenges. The use of multiple layers nevertheless provides advantages in demonstrating vertical solute (nitrogen) transport and enables calibration of vertical hydraulic gradients at paired wells.

Scenarios modelled comprise the following:
- steady state flow conditions
- ambient nitrogen transport (concentrations and loading)
- nitrogen transport after 35 years of farming development to simulate current conditions
- equilibrium conditions reached under continued current land use loading (investigation of lag)
- equilibrium conditions reached assuming an instant 20 per cent reduction in current land use loading (investigation of a proposed mitigation policy).

5 Model construction

The software used was Visual Modflow from Waterloo Hydrogeologic. This package is based on the Modflow and MT3D models, developed by the USGS, which uses a finite difference method. The technical basis of this model is described in Appendix II. This model was chosen predominantly because it has been widely used, verified and accepted as an industry standard. There are, however, disadvantages compared to a finite element model in respect to flexibility of model construction.

5.1 Model grid

Modflow requires the discretisation of the model domain into gridded cells. The domain was divided into 80 rows and 160 columns of 250 metre x 250 metre cells (Figure 6). One of the limitations of the finite difference method compared to a finite element approach is finer cells cannot be introduced in an area of interest without carrying

these through the entire row or column. A regular grid was used to avoid unnecessary complexity given the diffuse nature of the transport issue.

Figure 6:   Model grid

The model base is 300 metres above mean sea level, which is nearly 60 metres below the water level of the lake. Evidence to date indicates that most of the direct groundwater seepage to the lake occurs within a water depth of 10 metres (Hector, 2004 and Gibbs 2005). The upper surface of the model was imported from a 100 metre grid digital elevation model.

The model comprises three layers. The vertical division of the domain in this way allows vertical hydraulic gradients to be investigated. More importantly, it enables the vertical migration of nitrogen solute to be simulated more realistically than would be the case given a single layer. Single layer construction would result in an instant vertical distribution of surface derived contaminant throughout the domain.

Whilst ideally vertical layer division would follow detailed hydrogeologic patterns, this was not practical in the northern catchment. Although small scale changes, such as the occurrence of paleosols, may be locally important, they are not able to be modelled distributed as they would occur. Substantial variation in aquitard properties is indicated by aquifer hydraulic testing and water age dating (Hadfield et al., 2001). There is, however, generally insufficient log information to describe subtle changes in formation characteristics that may nevertheless be of local hydraulic importance. As a result, it was decided to introduce a 20 metre thick middle layer across the domain to represent potential aquitard occurrence. Properties of this layer are determined through model calibration and hence able to be constrained by observed vertical hydraulic gradients - nitrogen distribution and to a lesser extent water age dating. The effective result of this approach is that the same leakage characteristics of, for example, a very thin low permeability layer would be represented by slightly higher vertical permeability over a 20 metre thickness.

5.2 Boundary conditions

The perimeter of the model is defined by no flow boundaries around the edge of the northern sub-catchments representing a topographic high. Available information indicates that the groundwater and surface flow divides coincide. The lake is represented by a constant head boundary at a relative mean sea level of 357 metres, which is an approximate long-term mean.

The four streams modelled: Mapara, Whangamata, Otaketake and Waihora were represented as river cells (Figure 7). After initial calibration the Whangamata Stream was represented as a series of wells for subsequent modelling. This was because leakage to much of the stream, apart from discrete springs, is very small and leakage calculated to river boundary cells was very sensitive to changes in head.

Figure 7: **Modelled river boundaries (from left to right Waihora, Otaketake, Whangamata and Mapara).**

6 Steady state flow modelling

6.1 Inputs

An aquifer property of primary importance for model input is hydraulic conductivity. Where possible through calibration, similar patterns have been used for hydraulic conductivity, as are expressed in the geological mapping (Figures 8 to 10). Of particular note is that poorly fractured Whakamaru ignimbrite underlies most of the western side of the domain. To the east, groupings of rhyolitic pyroclastic formations dominate the deeper geology. Overlying these is the widespread unwelded Oruanui ignimbrite, with more surficial and higher permeability Taupo ignimbrite largely limited to the valley areas.

Hydraulic conductivity (horizontal) in the model ranges from 0.1 m d^{-1} to 20 m d^{-1} in layer 1; 0.01 m d^{-1} to 40 m d^{-1} in layer 2 and 0.5 m d^{-1} to 100 m d^{-1} in layer 3. Median hydraulic conductivity from flow testing for the Whakamaru ignimbrite, 'rhyolitic pyroclastic' grouping and Oruanui ignimbrite were 0.01 m d^{-1}, 0.93 m d^{-1} and 0.28 m d^{-1} respectively. The model input properties generally reflect these measured data. No horizontal anisotropy is assumed in the model and vertical permeability is generally about one order lower than the horizontal. Leakage estimates from pumping tests (varying from 0.02 to 1x10^{-4} d^{-1}) and measured vertical head differences (ranging from 3 to 80 metres) as well as groundwater dating results, indicate variable connection between hydraulic units. Although this level of detail is not readily reflected within the model construct, considerable variation is introduced though property input.

Recharge to the model was estimated from available rainfall and evaporation data. Rainfall was automatically recorded at six rain gauges in the Lake Taupo catchment (Table 4). The annual average rainfall for the six rain gauges is 1,720 millimetres. The spatial distribution of annual average rainfall is strongly correlated with topographic elevation. This relationship is shown in Figure 11, where mean annual rainfall increases significantly with altitude.

Rainfall was estimated as elevation 3.15 X + 8.23 from the regression relationship in Figure 11. It is estimated that about 35 to 50 per cent of rainfall is effective as recharge after allowing for evapotranspiration and quickflow. As a result, five recharge zones were introduced into the model increasing with domain elevation from 450 millimetres to 900 millimetres of annual recharge. It was decided not to introduce a more complex distribution as the recharge concentration for contaminant transport must be manually entered, and it could not be justified on the basis of other available information.
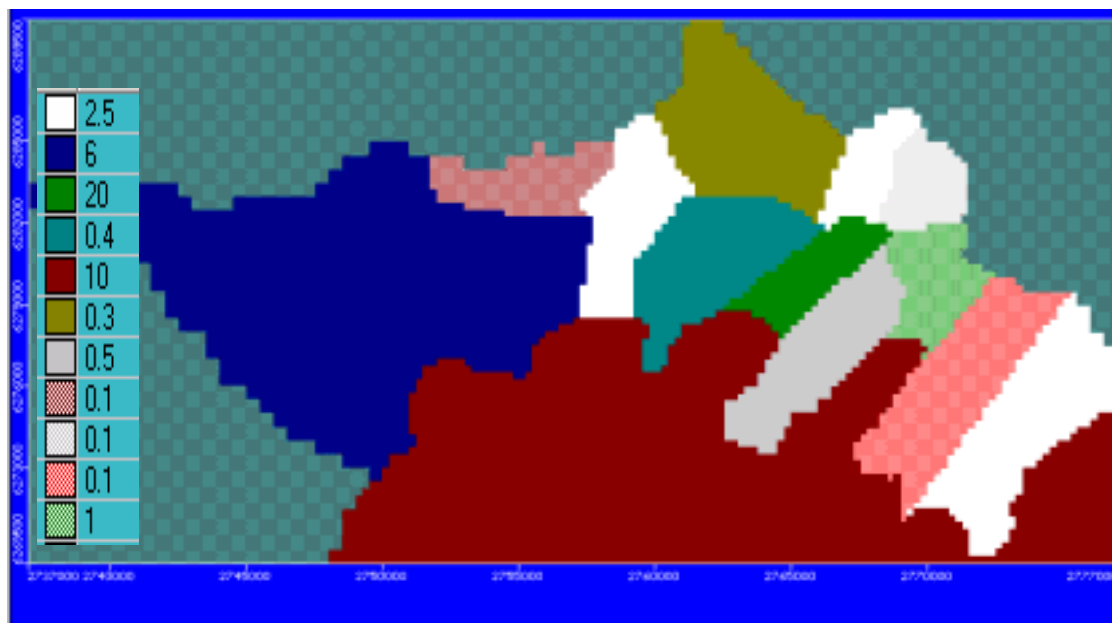
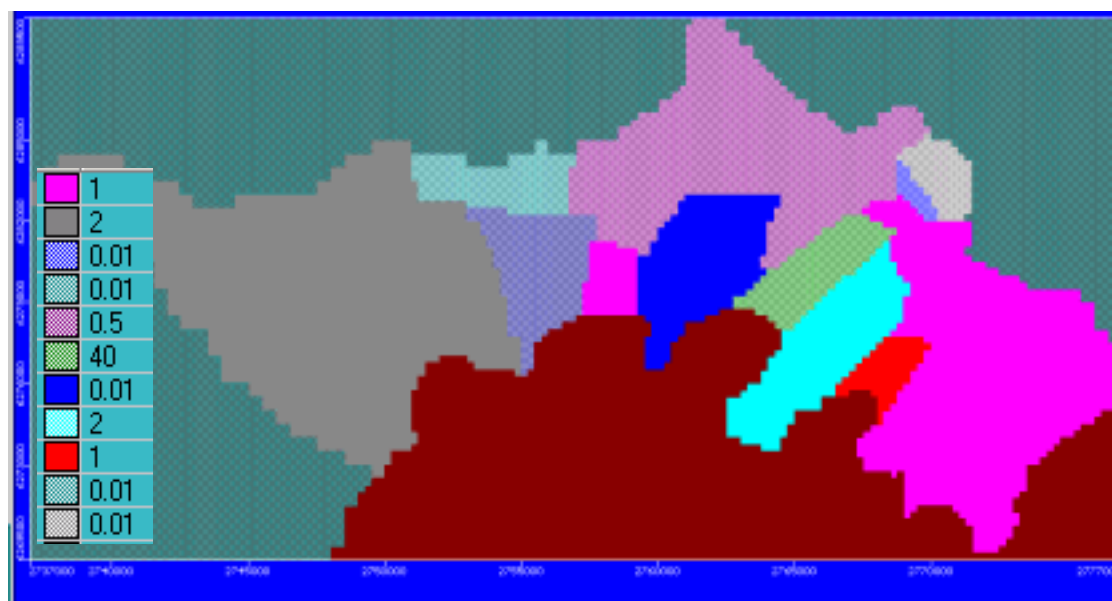**Figure 8**   **Hydraulic conductivity distribution in layer 1 (K_{xy} m d^{-1})**

Figure 9:   **Hydraulic conductivity distribution in layer 2 (Kxy m d-1)**
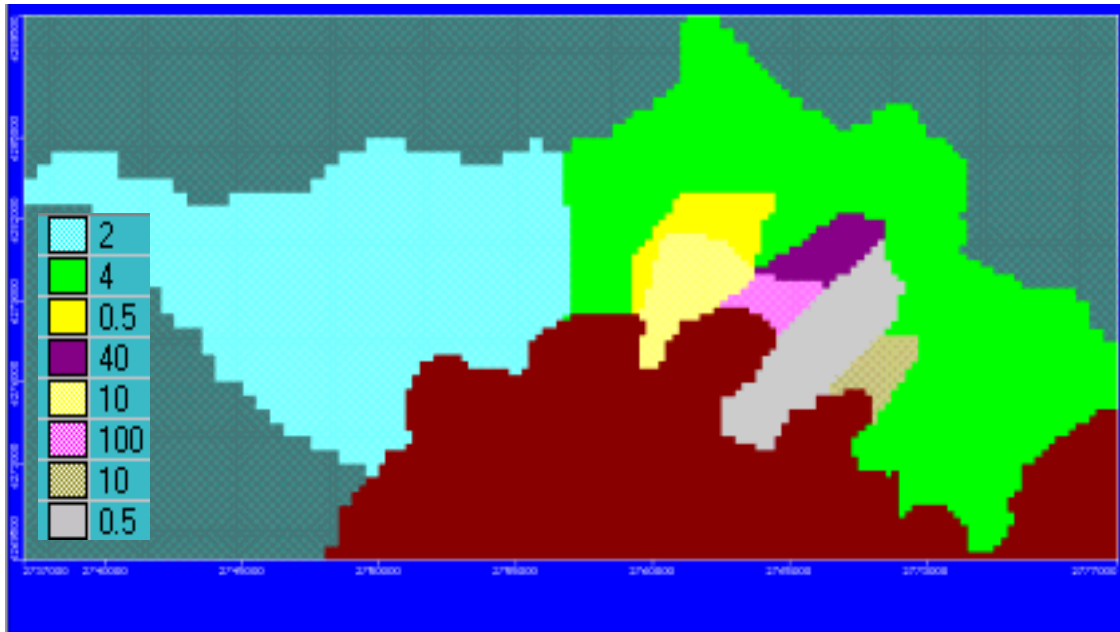
Figure 10: Hydraulic conductivity distribution in layer 3 (Kxy m d-1)

Table 4: Rainfall sites and summary of records

Site name and location	EW site number	Site altitude (m amsl)	Record duration	Record annual mean (mm)	Record annual standard deviation (mm)	Mean 1999 to 2001 (mm)
Waihaha River T18:414-739	1106.1/2	545	1977 - 1995	1564	259	1496*
Mangaokewa (Waipa) S20:248-014	414.21	488	1990 - 2001	1810	234	1604
Wairakei U17:814-825	1131.199	340	1999 - 2001	1149	5	1149
Tauranga – Taupo T19:695-392	971.2	670	1976 - 2001	1992	299	1935
Te Porere (Wanganui) T19:332-352	1849.1	700	1963 - 2001	2295	346	2353
Kuratau T18:427-546	282.3	450	1994 - 2001	1494	259	1395

∗   based on linear relationship between annual mean rainfall for the complete records and for 1999 –
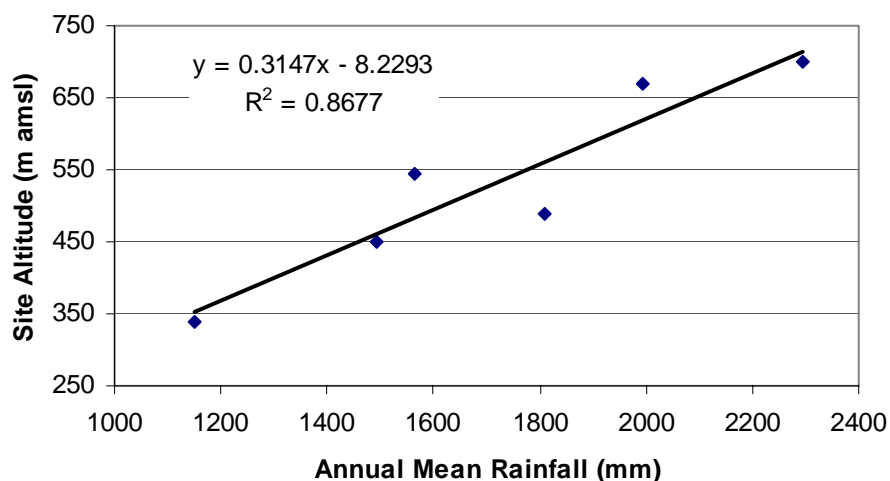   2001 for the other sites.

Figure 11: Mean annual rainfall versus rain gauge elevation for sites listed in Table 4.

Flow information for the four streams, Mapara, Whangamata, Otaketake and Waihora, was primarily derived from gaugings by Piper (2004). It includes gaugings undertaken on three separate occasions at several locations within the stream catchments during 2004. Locations are illustrated in Figure 12 and selected measured flows are listed in Table 7.

Almost all stream reaches were found to be gaining. The majority of flow in the Mapara Stream is derived from the lower part of the catchment. In contrast, flow in the Whangamata Stream is predominantly derived from discrete springs in the upper catchment. Of the four, the largest flow is from the Waihora Stream in the western catchment, which receives higher rainfall. This is reflected in greater specific discharge.

Water balance estimates indicate that, unlike streams in the western Lake Taupo catchment, there is a considerable proportion of direct seepage of groundwater to the lake in the northern catchments modelled. Direct groundwater contribution estimated from water balance residuals are listed in Table 5 for the stream sub-catchments modelled. The importance of indirect groundwater input via streams is also indicated by the very large baseflow component.

Table 5:   Estimated direct groundwater discharge to the lake and baseflow contribution to streams (Piper, 2004)

Catchment	% discharge as groundwater	% baseflow in streams
Mapara	81	90
Whangamata	89	90
Otaketake	85	88
Waihora	47	90

Direct seepage of groundwater to the lake from the Mapara and Whangamata catchments was investigated by Hector (2004) and subsequently by Gibbs (2005). These studies show that only about 4,000 m^3 d^{-1} of the 73,000 m^3 d^{-1} direct seepage estimated from the water balance is accounted for within one metre depth in the lake shore. Subsequent deeper investigation, however, indicates most of the remaining seepage occurs within 2 to 6 metres of lake depth (Gibbs, 2005).

The longest record of regular flow monitoring exists for the Whangamata Stream. It includes data from 1976 to 1979 (Schouten et al., 1981) and measurements by NIWA from 1995 onward (Howard-Williams and Pickmere, 2002). The mean flow for the longer record is 109 l s^{-1} compared to less than half this rate measured by Piper (2004) in 2002-3. The lower 2002-3 flow rates were used for model calibration. Mean residence time of baseflow in the Whangamata Stream is estimated to be about 80 years.

The river boundaries of the four modelled streams illustrated below are shown in Figure 7. The rate of leakage is determined primarily by head differential in the model, but also by an assumed regular conductance.
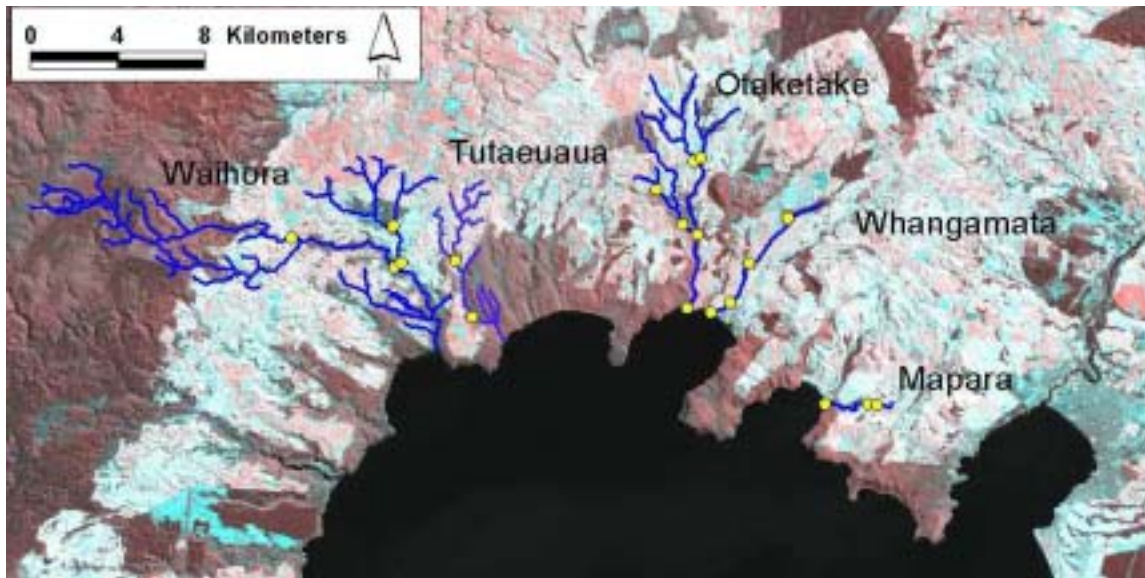
Figure 12   **Location of stream gauging and sampling sites (Piper, 2004)**

6.2 Calibration and results

The steady state flow model was calibrated against observed groundwater levels and stream flows using the inverse solution method. Parameter optimisation was initially undertaken using the non-linear parameter estimation and predictive analysis program (PEST). However, manual comparison was ultimately relied upon. A total of 54 observation wells were included in the calibration and a normalised root mean square fit of under 8 per cent was achieved (Figure 13). Calibration residuals were normally distributed. It is noted that there are no wells located within layer 2.

Figure 14 shows the predicted piezometric surface for layer 1 with velocity vectors. The similarity to the piezometric surface derived from observed data is evident by comparison with Figure 3. Horizontal groundwater velocity ranges from about 0.02 to about 1 m d^{-1} in porous media with highest velocities predicted in the Kinloch area, associated with the highest permeabilities. This is consistent with the highest observed hydraulic conductivities from field aquifer flow testing. Groundwater velocity in the simulated fractured Whakamaru ignimbrite on the western side of layer three ranges from about 0.3 m d^{-1} to 1.5 m d^{-1}. Some groundwater travel times to the lake in excess of 100 years are evident in layer 1 (Figure 15). Stream capture by gaining streams, although generally common, is minor in the Whangamata and Mapara catchments.

A more subdued piezometric head distribution is predicted in layer 3. A recharge regime occurs almost exclusively throughout the domain. This is illustrated by downward movement in vertical cross-section in Figure 16.
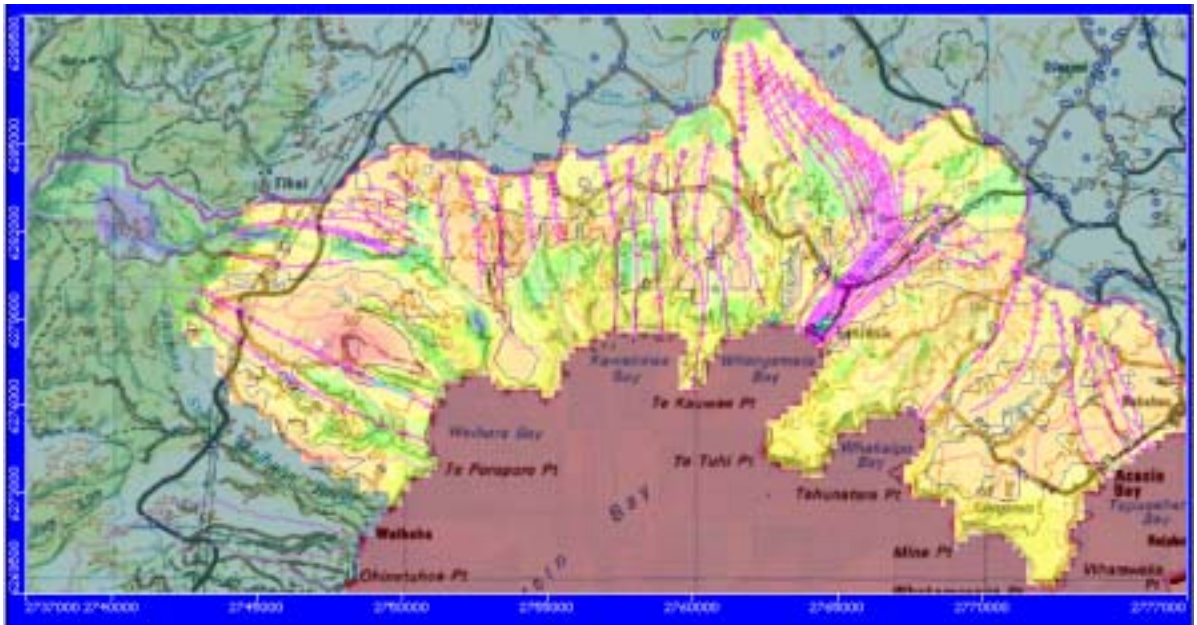
Figure 13: Predicted versus observed groundwater levels for 54 monitoring wells.

Figure 14: Predicted piezometric contours, velocity vectors and location of observation wells.

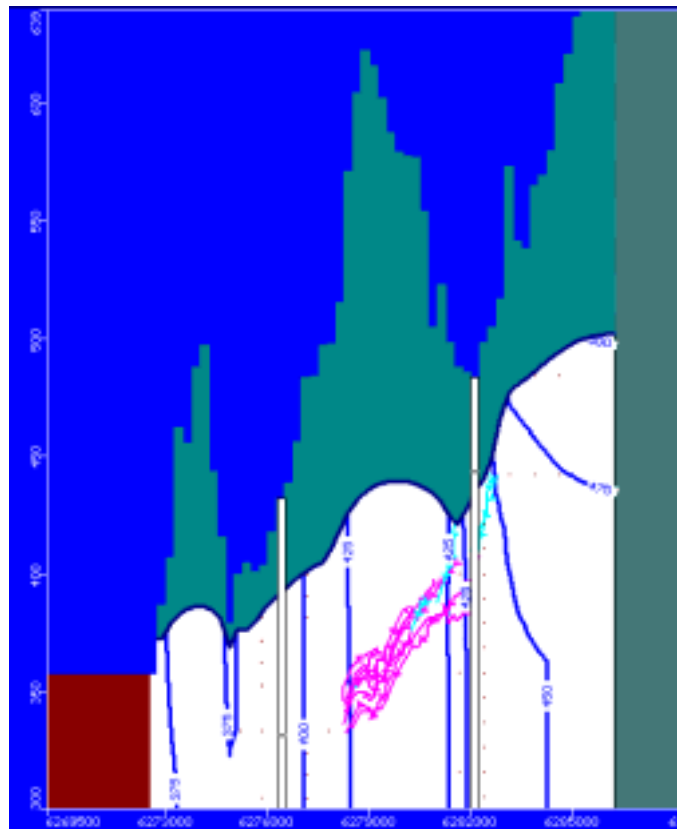**Figure 15:** **Simulated flow paths with 10 year time steps.**

Figure 16: **Vertical section illustrating typical recharge gradients** (10 times vertical exaggeration).

Another useful means of calibration is to compare vertical head differentials between paired wells for magnitude and direction. Table 6 below shows that although the listed head differentials show the correct vertical direction of flow, the magnitude matches are variable.

Table 6:   Predicted and observed hydraulic head differences at paired wells.

Wells	Observed head difference (m)	Predicted head difference (m)
72.1008 and 72.514	10.1	26.4
72.354 and 72.352	46.4	58.3
72.331 and 72.513	2.4	0.38
72.1388 and 72.348	80.3	59.5
72.1389 and 68.430	58	56.2

Stream-flow calibration was undertaken by matching flows within stream reaches observed by Piper (2004). Streams were influent throughout all reaches calibrated and leakage was found to be sensitive to stage height. A very good match was achieved, as illustrated as listed in Table 7. It was important that groundwater leakage rates to streams were well calibrated for subsequent mass transport and concentration calculations.

Table 7:   Predicted and observed stream flows (l s^{-1})

Stream flows[a]	Observed	Simulated
Mapara mouth (1)	75	**79**
Mapara (2)	25	**25.4**
Whangamata mouth (1)	47	**46.8**
Otaketake mouth (1)	80	**96**
Otaketake (2) east tributary	20	**18**
Otaketake (3) west tributary	50	**42**
Waihora confluence	**675**	**697**

[a] refer to Figure 12 for site locations numbering from the mouth

It is noted that only four selected streams were included in the model. This was due to there being limited information for some other streams. This may affect calibration of the piezometric surface very locally in those parts of the domain.

6.3   Sensitivity analysis

Sensitivity analysis of both the flow and contaminant transport models was undertaken predominantly by individually varying model parameters and noting changes in the degree of fit expressed by the objective function. For the steady state flow model, the objective function is the root mean square of the differences between groundwater level measured at 54 wells and corresponding simulated levels. The results of sensitivity analysis are summarised in Table 8.

The flow model is most sensitive to changes in recharge, horizontal conductivity of layer 3, vertical hydraulic in layer 2 and horizontal conductivity in layer 1, respectively. It is noted that the model is more sensitive to a change in all conductivities than recharge. The horizontal conductivity is more sensitive in the y direction or general direction of flow toward the lake.

Table 8: Flow model sensitivity analysis

Model parameter varied by 30 %[1]	% change in RMS fit
Recharge	35
Horizontal hydraulic conductivity layer 1	13
Horizontal hydraulic conductivity layer 2	1.2
Horizontal hydraulic conductivity layer 3	23
Horizontal hydraulic conductivity all layers	40
Vertical hydraulic conductivity layer 1	1
Vertical hydraulic conductivity layer 2	19
Vertical hydraulic conductivity layer 3	0.06
Anisotropy k_y/k_x layer 1	6.5
Anisotropy k_x/k_y layer 1	4
Anisotropy k_y/k_y layer 2	0.8
Anisotropy k_x/k_y layer 2	0.2
Anisotropy k_y/k_x layer 3	15
Anisotropy k_x/k_y layer 3	0.8

[1] all changes in parameter values were reduction with the exception of recharge.

Due to the constraint of thin saturated upper layers in a complex topography, significant increases in conductivity or decreases in recharge for sensitivity analysis required these to be undertaken as a coupled process. Although this resulted in little change in flow calibration there were implications for contaminant concentration.

7 Transient contaminant modelling

7.1 Introduction

The modular three-dimensional multi-species transport model (MT3D), originally developed by Zheng (1990), was used to simulate nitrogen transport within the northern catchment. The technical basis of the model is described in Appendix II.

Ambient conditions were initially simulated as a starting point for subsequent modelling of land use effects. Estimated 'current' (2002) nitrogen loading rates were then modelled for a period of 35 years to approximate present conditions. The effects of long-term nitrogen loading at the current rate were also investigated by continuing simulation using current loading until an equilibrium was reached.

The effectiveness of a 20 per cent reduction in manageable loading rate in mitigating water quality degradation was examined by modelling a 20 per cent reduction in nitrogen loading from land uses across the domain. This reduction commenced after 35 years of 'current' loading. The period of 35 years was used to simulate the current land use loading situation. Farming was becoming established in the area a little before this time. However, the model assumes instant establishment of the current farming intensity and hence a shorter period is considered appropriate.

The model was also used to differentiate between 'older' and 'younger' nitrogen transport to provide further calibration with, and understanding of, water age dating. Nitrogen transport was assumed to be conservative as discussed in the following section.

7.2   Transport properties

Nitrogen contaminant was modelled as a conservative solute and hence no sorption or kinetic reactions were specified. In this respect the model output may be considered worst case. Although, in reality, conservative transport may not always be assumed, very detailed information would be required to confidently introduce attenuation processes to the model. Also, information to date, including preliminary field work reported in Appendix III, indicates that active denitrification, although significant, is likely to be minor in much of the catchment. More research is, however, needed to substantiate this. It appears that even where anaerobic aquifer conditions are inferred active denitrification may be limited due to lack of dissolved organic carbon. Some localised denitrification is evident associated with wetlands in the lower reaches of some streams, such as the Tutaeaua (Sukias et al., 2004). Model calibration could be used to infer rates of denitrification. Attenuation is, however, considered separately as a potential adjustment to modelled load in Section 8.

Ambient or predevelopment nitrate-nitrogen concentrations were established for subsequent investigation of land use impacts by first achieving an equilibrium ambient condition throughout the model. To do this, a loading rate of 2 kg N ha^{-1} y^{-1} was applied to the whole domain as a solute source, by applying a spatially variable rainfall concentration.

The rainfall concentration was estimated by dividing estimates of typical nitrogen loading for land use by the rate of effective recharge. This implies an inverse linear relationship between recharge and concentration for a given land use. Although this is not necessarily the case, work by Green and Clothier (2002), indicates that it is a useful approximation. Ambient recharge concentrations ranged from 0.25 mg l^{-1} to 0.44 mg l^{-1}.

Land use loading rates for dairy, drystock, forested and native flora (ambient) conditions were by necessity manually loaded into the model. Recharge concentrations largely reflect the land use pattern illustrated in Figure 17. Typical land use loading rates of 14 kg N ha^{-1} y^{-1} for drystock and 40 kg N ha^{-1} y^{-1} for dairy were used, based on work by Ledgard (2000a,b). Although land use loading was simplified in this way for manual model input, more detailed analysis by Brown et al., (2002) produced a very similar total loading (less than 10 per cent higher).

The recharge concentrations ranged from 0.25 mg l^{-1} to 7.3 mg l^{-1}. The scenario of a 20 per cent reduction in manageable nitrogen load was introduced through a comparable decrease in recharge concentration with a similar distribution.

Nitrogen removal via in-stream uptake or algal mats associated with zones of deeper direct groundwater seepage to the lake (Gibbs et al., 2005) is beyond the scope of the groundwater model, but influences nitrogen load to the lake.

Total and effective porosity inputs to the model were varied to determine the effect of simulated fracture transport and sensitivity more generally. Values of 0.25 and 0.3 were used in the model for porous sediments. Fractured ignimbrite was simulated by introducing total and effective porosities of both 0.01 and 0.05. The latter preferred estimate allowing some unwelded component, is based on Davis (1969). Effective porosity is directly proportional to travel times predicted by Modpath and total porosity influences MT3D concentration predictions, as described in Section 7.4.

A range of longitudinal dispersivities was introduced into the model. Transverse and vertical dispersivity were assigned ratios of 0.1 and 0.01 of the longitudinal magnitude. The magnitude of dispersivity is considerably less important in diffuse contaminant modelling than point source. It does, however, influence vertical migration.

7.3   Calibration and results

Ambient conditions were achieved by running the model initially for 800 years to ensure equilibrium nitrogen concentrations were achieved throughout the entire domain using a loading rate of 2 kg N ha^{-1} y^{-1}. Equilibrium was reached within 300 years with 66 tonnes of nitrogen estimated to be discharging from the groundwater system annually.

Just under 5 kilotonnes of ambient nitrogen is estimated to be stored in the modelled groundwater system if it is comprised completely of porous media. The introduction of simulated fractured Whakamaru ignimbrite west of Omoho sub-catchment with a porosity of 0.05, reduces the mass of nitrogen stored in the aquifer under ambient conditions to about 3.5 kilotonnes. The nitrate-N distribution in layer 1 is illustrated in Figure 18.

Modelled concentrations after 35 years of simulated current land use were compared with observations from 31 wells and historic water quality trends in the Mapara and Whangamata Streams. Highest nitrate concentrations of about 7 g m^{-3} were predicted in an area of dairy land use just west of Kinloch. Calibration using simplified land use data produced a normalized root mean square of <16%. Increasing dairy farm loading in the Kinloch area to 60 kg N ha^{-1} y^{-1} improved the fit to <14% and removing four wells with groundwater chemistry indicating likely anaerobic condition improved the fit further to <12%. Residuals are approximately normally distributed. One well (68.320) was excluded from the calibration as the nitrate concentration of > 30 mg l^{-1} at this site is known to result from point source contamination from a nearby woolshed.

Despite further improvement in calibration being readily achieved by expedient manipulation of recharge concentration spatially, emphasis was instead given to scenario and sensitivity testing using more simplified land use loading. Also although a slight improvement in flow calibration results from concomitant increase in both recharge rate and hydraulic conductivity, related decreases in contaminant concentration produce a poorer contaminant transport fit.

The modelled concentration distribution after 35 years at the current land use regime is shown in plan view for layer 1 in Figure 19. Increased nitrogen concentrations are apparent as a result of farming with a notably higher area of nitrogen concentration in the middle of the domain (Kawakawa Rd area) associated with dairying activities. Figure 20, indicates that after 35 years the deeper groundwater in layer 3 has essentially been little affected.
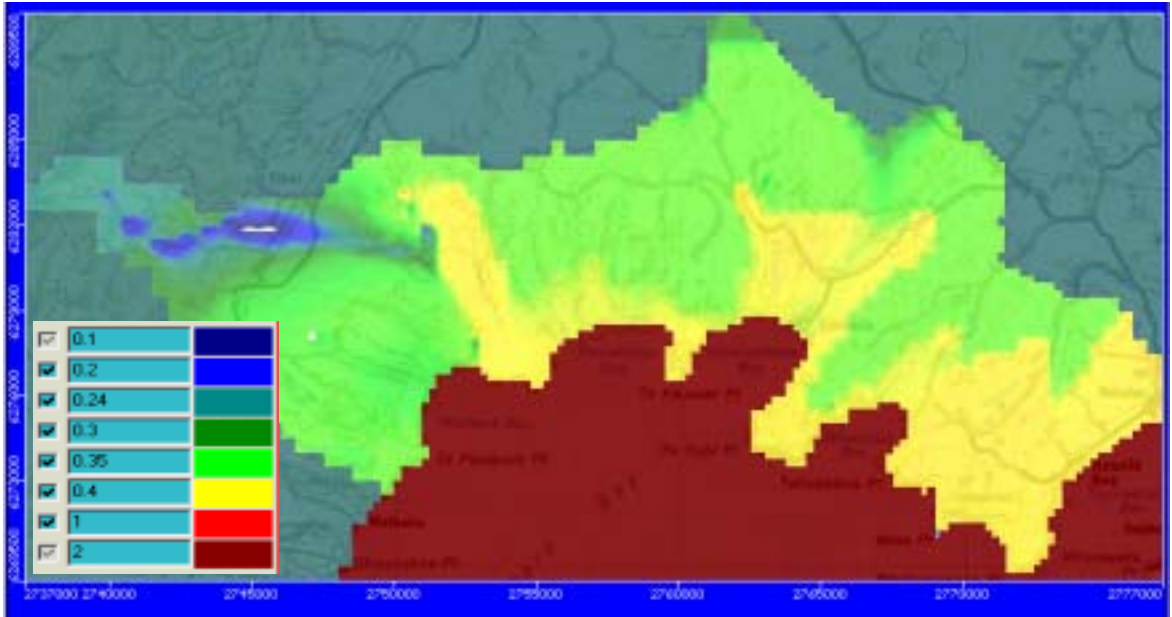
Figure 17: Northern catchment land use

Figure 18: Nitrate-N concentration in layer 1 ambient condition (mg l^{-1})
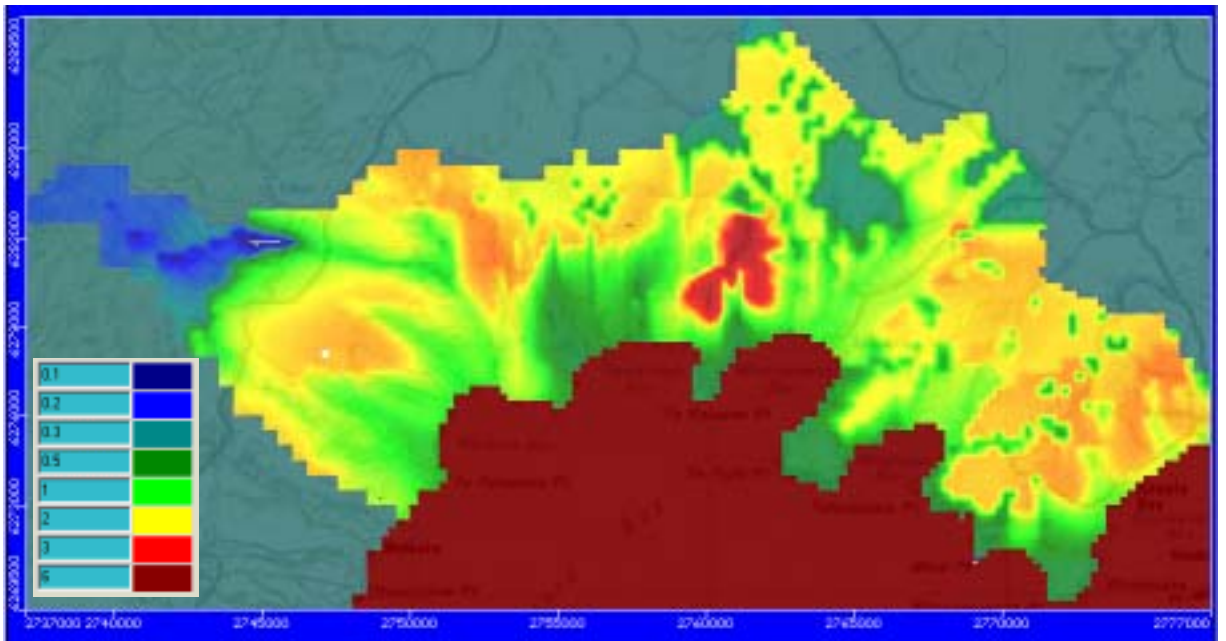

Figure 19: Nitrate-N concentration (mg l^{-1}) in layer 1 after 35 years of current land use.
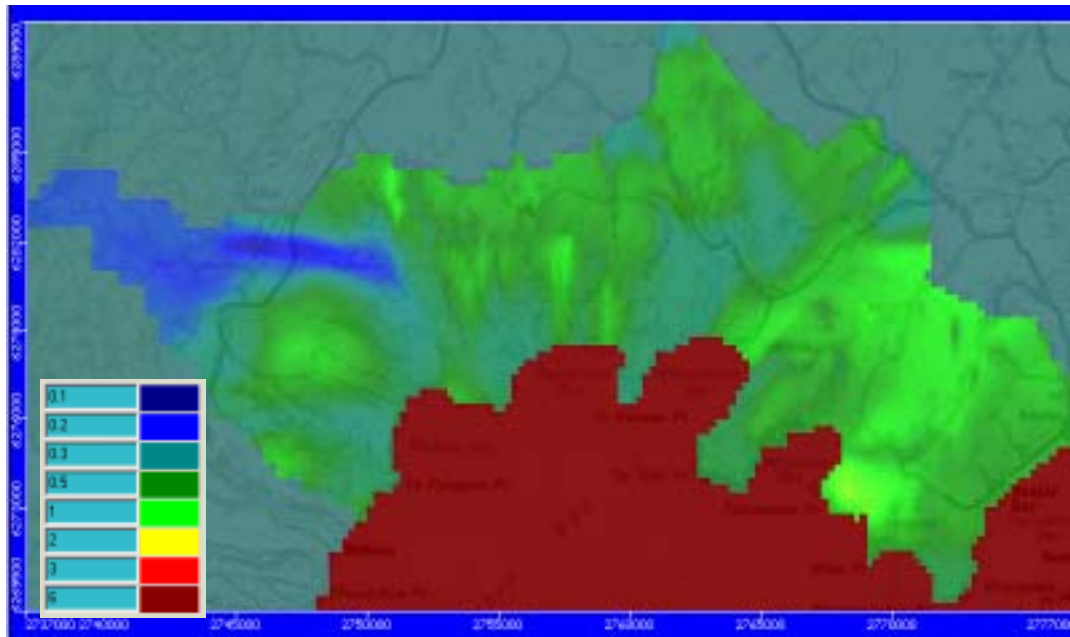
Figure 20 **Nitrate-N concentration (mg l^{-1}) in layer 3 after 35 years of current land use.**

Figures 21 and 22, show nitrate-N concentrations after 250 years in plan view for layers 1 and 3 respectively. Concentrations are generally higher in lower rainfall areas in the east of the domain and in dairying areas along Kawakawa Rd.

The temporal trend of calculated nitrate-N flux from the groundwater system is illustrated in Figure 23. Figure 24, shows a similar trend in nitrogen mass stored within the groundwater system. These trends predict very slow equilibration (~250 years) of nitrogen contaminant within the domain. An important part of the substantial lag in the system is the time it takes land use activities to build the very large nitrogen mass in the groundwater store to achieve equilibrium.

On the basis of assumptions described in the modelling, nitrate-N leaching to the lake would equilibrate at about 300 tonnes annually. At that stage some 18 kilotonnes of nitrogen would be present in groundwater within the domain. 'Current' annual nitrogen flux of about 175 tonnes is indicated after 35 years of simulated farming, with about 10 kilotonnes being stored in the groundwater system. If fractured ignimbrite porosity of 0.01 were used, these numbers would vary slightly, such that the 'current' scenario would see just over 180 tonnes of nitrogen flux annually from the aquifer with about 9.5 kilotonnes in storage. Sensitivity is discussed further in Section 7.4.
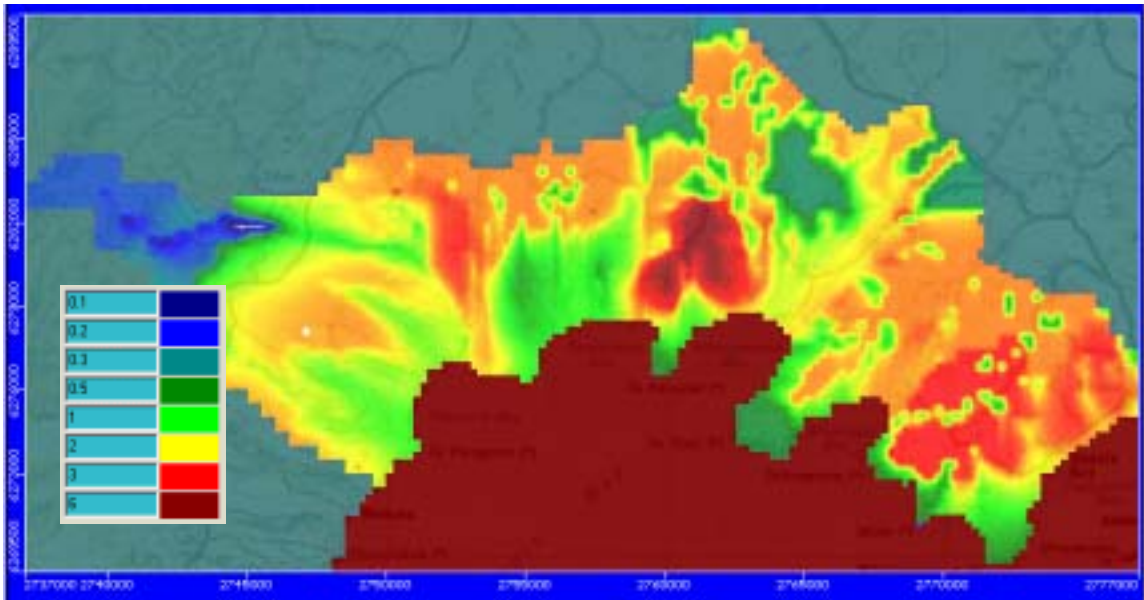
Figure 21: Nitrate-N concentration (mg l^{-1}) in layer 1 after 250 years of current land use.
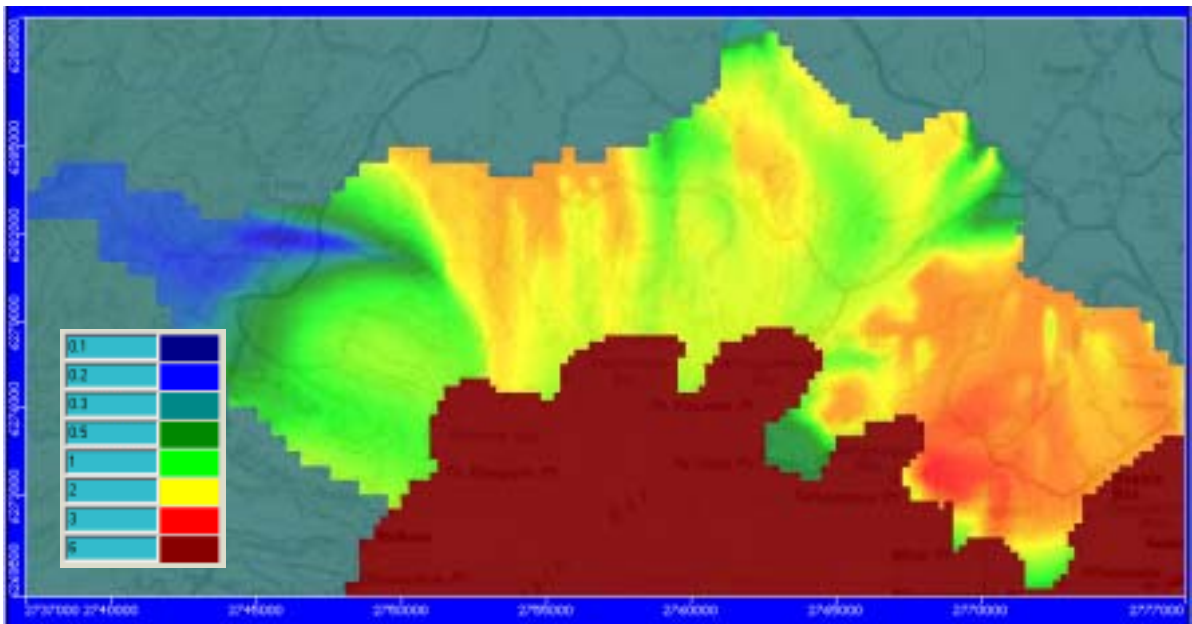

Figure 22 Nitrate-N concentration (mg l^{-1}) in layer 3 after 250 years of current land use.

The proposed policy initiative to achieve a 20 per cent reduction in manageable nitrogen load to the lake was also modelled. This was done by running the current land use loading for 35 years and then reducing the manageable load instantly by 20 per cent. The resultant curves presented (Figures 23 and 24) show that this initiative is a very useful mitigation measure. Modelled output from the total groundwater system continues to increase rather than decline, but reaches a lower equilibrium mass loading.
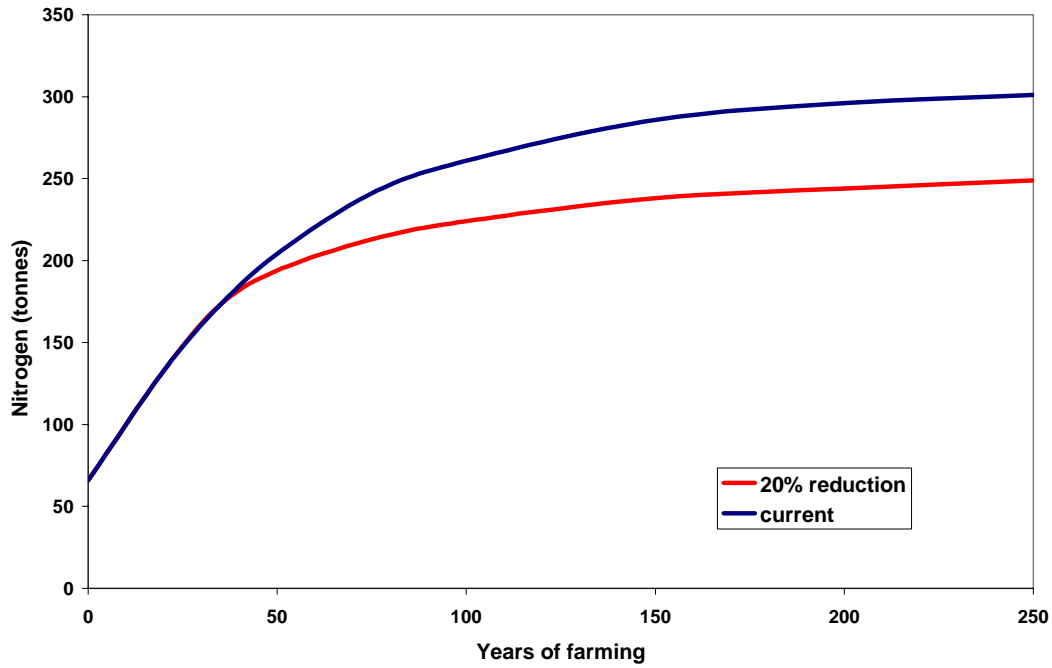
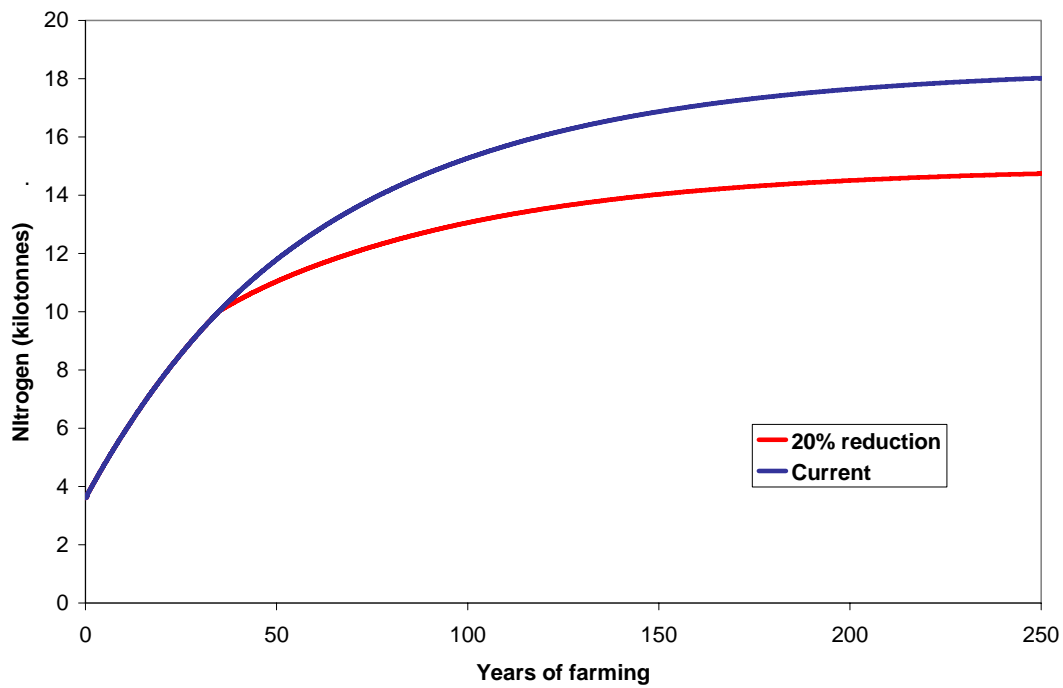**Figure 23:**   **Calculated annual nitrate-N flux to the lake**

Figure 24:   **Calculated nitrate-N mass in the groundwater system**

Temporal nitrogen concentration trends within the groundwater system vary depending on location. Examples of such variations are illustrated in Figures 25 and 26.
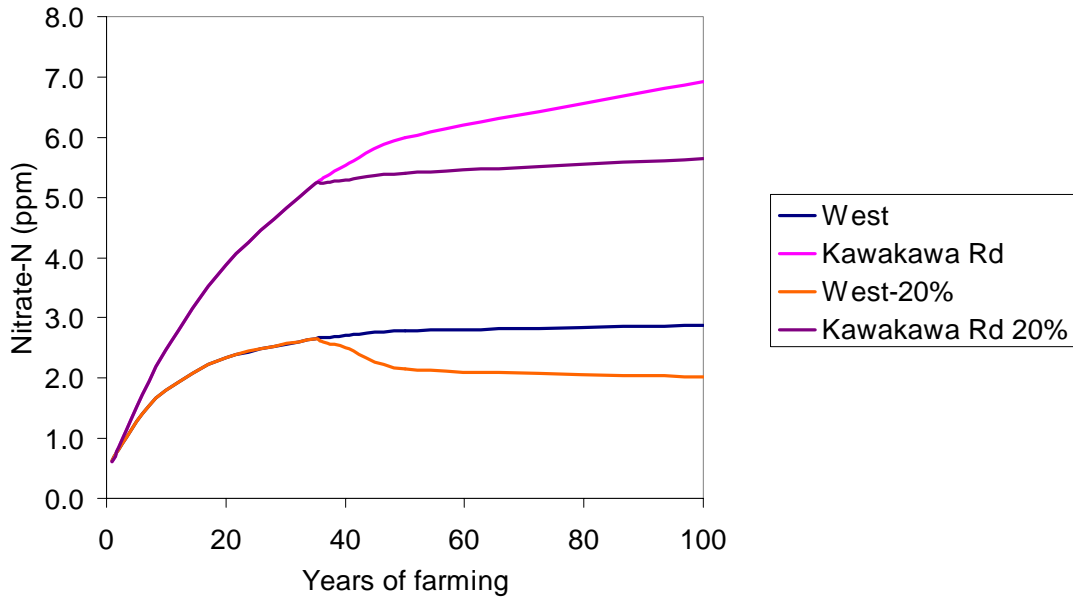
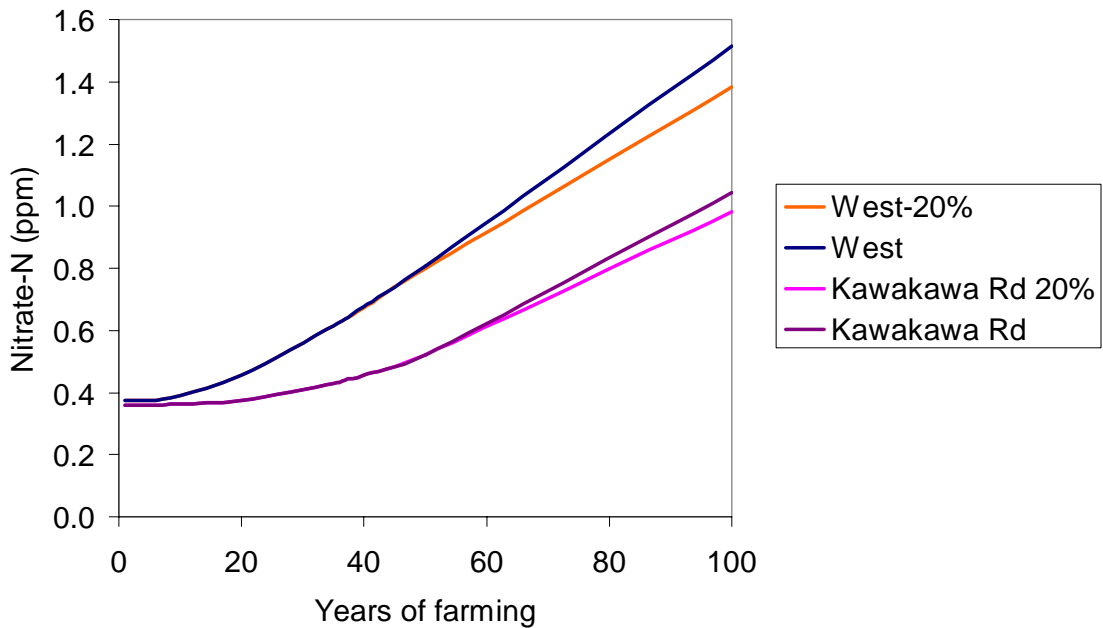**Figure 25:** Example nitrate-N concentration trends in Layer 1

Figure 26: Example nitrate-N concentration trends in Layer 3

Modelled surface water concentrations were calculated from flow budgets and mass flux using an external beta version transport budgeting mass flux tool from Waterloo Hydrogeologic. Simulated and observed temporal trends for the Whangamata and Mapara Streams are illustrated in Figures 27 and 28 respectively.
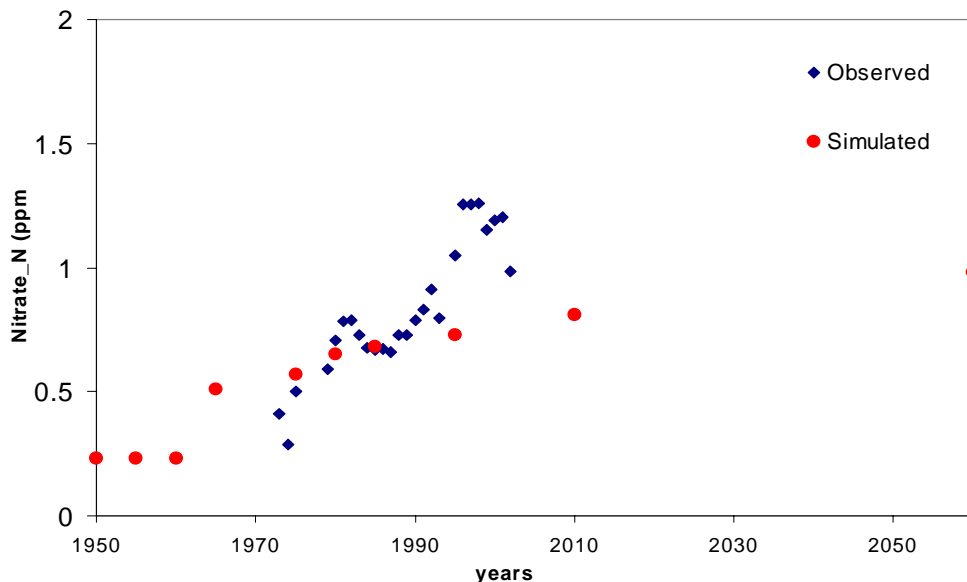
Figure 27: **Predicted and measured nitrate-N concentrations at the Whangamata Stream mouth.**

The predicted Whangamata Stream concentrations show a reasonable fit with observed quality, given assumptions including a single step increase to farm loading. The ambient nitrate-N concentration calculated at 0.23 ppm is close to the earliest stream measurements. Equilibrium reached at 1.05 ppm after about 150 years, however, is less than highest stream measurements and implies a larger nitrogen source within the catchment than is represented in the model. Attenuation within the stream or riparian zones is not accounted for by the model and would accentuate discrepancies.

In contrast, simulation of nitrate concentrations in the Mapara Stream significant over-estimated nitrate-N concentrations (Figure 28). The simulated ambient concentration of 0.43 ppm from a land use loading of 2 kg N ha^{-1} y^{-1} is higher than the earliest observed concentration of 0.18 ppm. This implies that significant denitrification may be occurring in this catchment. Nearly two thirds of the stream flow is generated in the bottom one kilometre of the stream catchment below a wetland. It is possible that there is substantial nitrogen loss within the wetland. The time series curve indicates that equilibrium is reached after about 100 years.

The effect of a 20 per cent reduction in manageable nitrogen load on the Whangamata Stream was investigated by simulating an instant reduction after 35 years of current land use. The resulting curve is indicated in Figure 29. This shows an immediate response, but a continued longer term slow increase. This initiative would, however, result in useful mitigation of the degradation otherwise predicted.

Modelling was used to illustrate how an immediate response in nitrate-N concentration can result from a 20 per cent reduction when such long overall lags are predicted. This was achieved by differentiating 'new' nitrogen leaching into the system from 'older' nitrogen already in storage at that time. The resulting curve (Figure 30) shows that although there is essentially an instant response, full flushing of older nitrogen from the higher loading rate would take a considerable time.
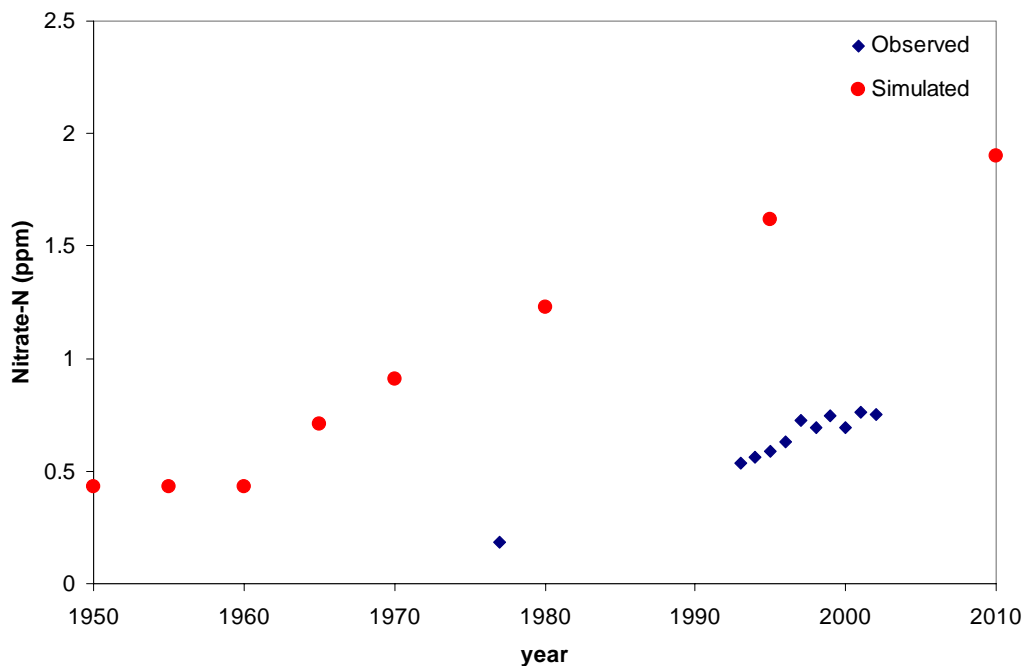
Figure 28: Predicted and measured nitrate-N concentrations at the Mapara Stream mouth.
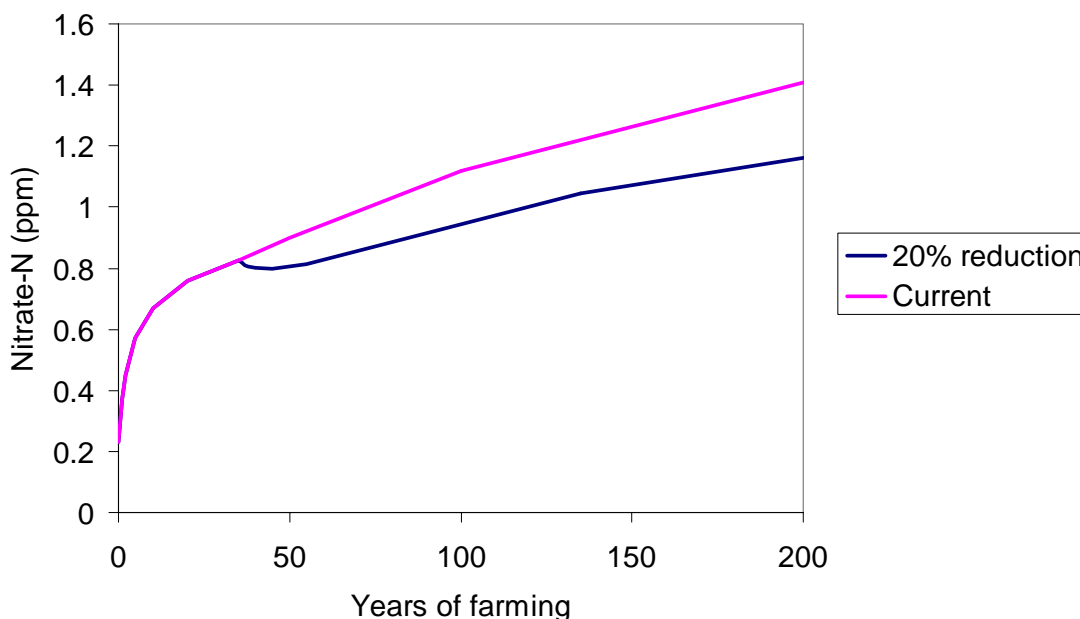
Figure 29: Predicted nitrate-N concentrations at the Whangamata Stream mouth comparing continuation of current land use with a 20% reduction.

The mixing of 'new' nitrogen with 'older' nitrogen in the model is in good general agreement with tritium dating of the Whangamata Stream by Geological and Nuclear Sciences for Environment Waikato (Morgenstern, 2007). The tritium interpretation suggests the Whangamata Stream has a mean residence time of about 80 years with about 24 per cent of water being less than 45 years old.

Figure 30: Predicted percentage of new nitrate-N input to the Whangamata Stream after an instantaneous 20% reduction in loading.

7.4 Sensitivity analysis

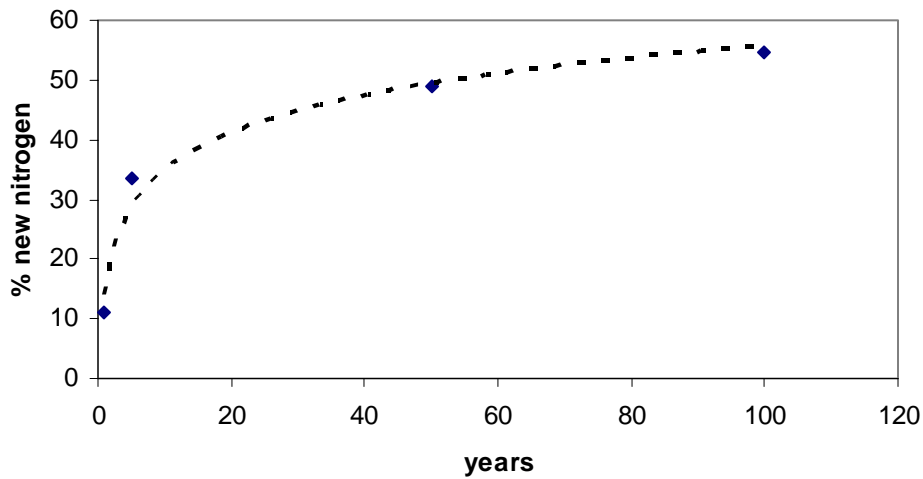Sensitivity analysis for the contaminant transport model was performed by individually varying model parameters and noting changes in the degree of fit based on the objective function. For the contaminant transport model, the objective function is the root mean square of discrepancies between predicted and observed nitrogen concentrations at the 31 wells used for calibration. The results of sensitivity analysis are summarised in Table 9.

Comparison using the objective function was made after 35 years of simulated 'current' land use nitrogen input with initial equilibrium ambient conditions. The parameters varied were: recharge concentration and rate, dispersivity, porosity and conductivity. Transformation (denitrification) was not considered.

The transport model was most sensitive in order of importance to changes in recharge concentration, conductivity, total porosity, recharge rate and dispersivity. Very high dispersivities (longitudinal of up to 150m) gave a slightly better fit.

Table 9: Transport model sensitivity analysis

Model parameter varied	Percent change	% change in RMS fit
Recharge concentration	-30	13.4%
Recharge concentration	+30	4.7%
Dispersivity	-30	0.7%
Dispersivity	+30	0.3%
Total porosity	-30	4.2%
Total porosity	+30	6.7%
Recharge rate	+30	1.7%
Conductivity k_{xy} layers 1 and 3	-30	6.3%
Conductivity a/a and L2 k_z also	-30	7.4%

Travel times calculated by the Modpath module are directly proportional to effective porosity whereas nitrate-N concentrations calculated by the MT3D transport module vary dependent on total porosity. The effect on simulated transport of varying total porosity was further tested by introducing a range of values to the model.

Figure 31, shows the effect of representing fractured ignimbrite in the western deeper aquifers with porosities of 0.01 and 0.05. Higher annual nitrogen flux to the lake occurs more quickly with fractured rock formations included, given the reduced capacity of the groundwater system to store nitrogen. It is noted that there is a less marked difference between the 'fractured rock porosity' trends.
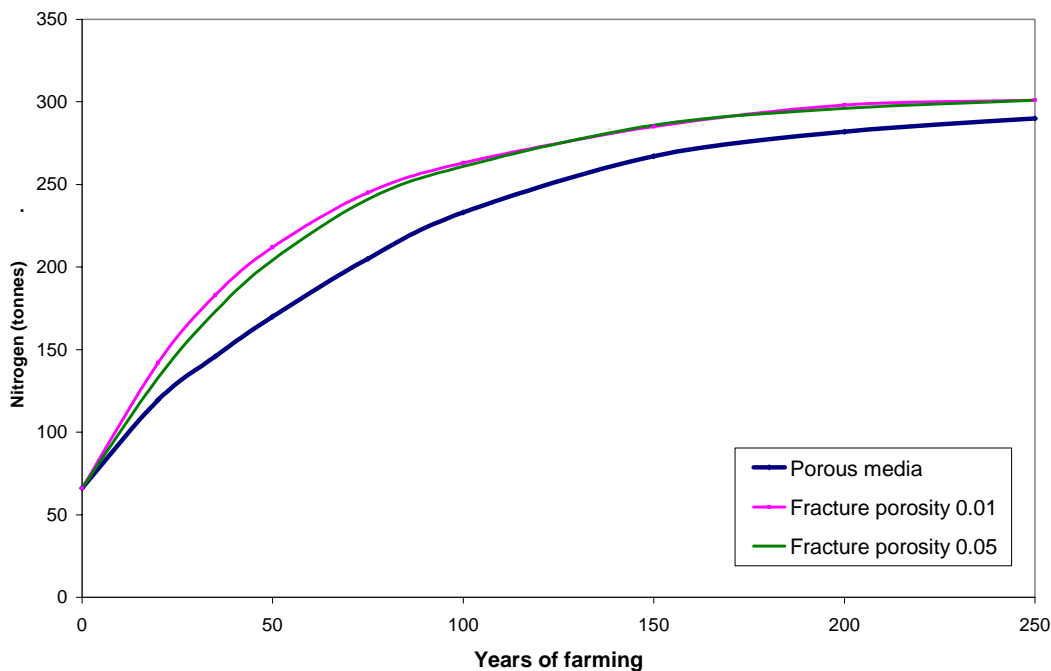
Figure 31: **Predicted annual nitrate-N flux to the lake from groundwater assuming uniform porous media porosity of 0.25; fracture porosity of 0.01 and 0.05 in the deeper western domain.**

Figure 32, illustrates the effect of porosity variation on calculated nitrate-N concentrations in the Whangamata Stream. It is apparent that variation of this parameter influences the slope of breakthrough, but the ultimate equilibrium concentration is unaffected. A contaminant transport study on the eastern side of Lake Taupo by Hadfield (1995) indicated an effective porosity value of 0.38 to 0.43 in pumice sediments. Total porosity for porous media in the model study area is likely to be close to 0.3. A more pronounced effect on breakthrough to the Whangamata Stream would be obtained by using lower porosity reflecting fractured rock.
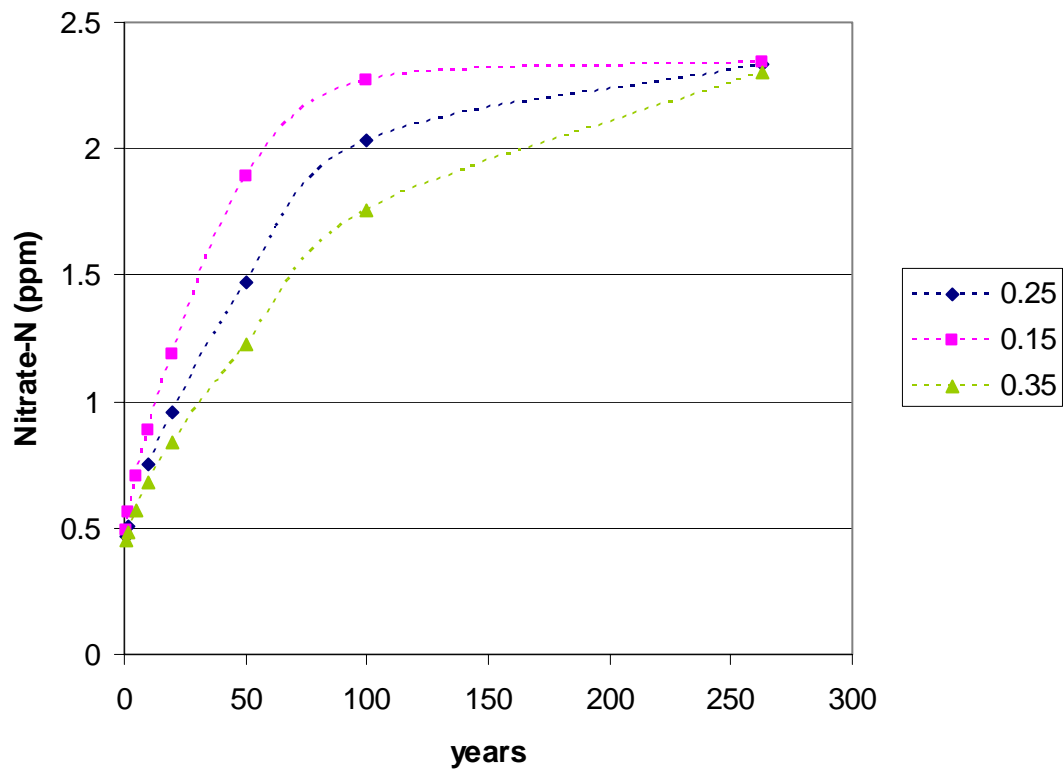
Figure 32: **Predicted nitrate-N concentration in the Whangamata Stream at selected total porosity values.**

Figure 33, shows the effect of representing fractured ignimbrite in the western deeper aquifers with porosities of 0.01 and 0.05 on the total mass of nitrogen stored in the groundwater system modelled. Several kilotonnes less nitrogen are stored in a partially fractured system.
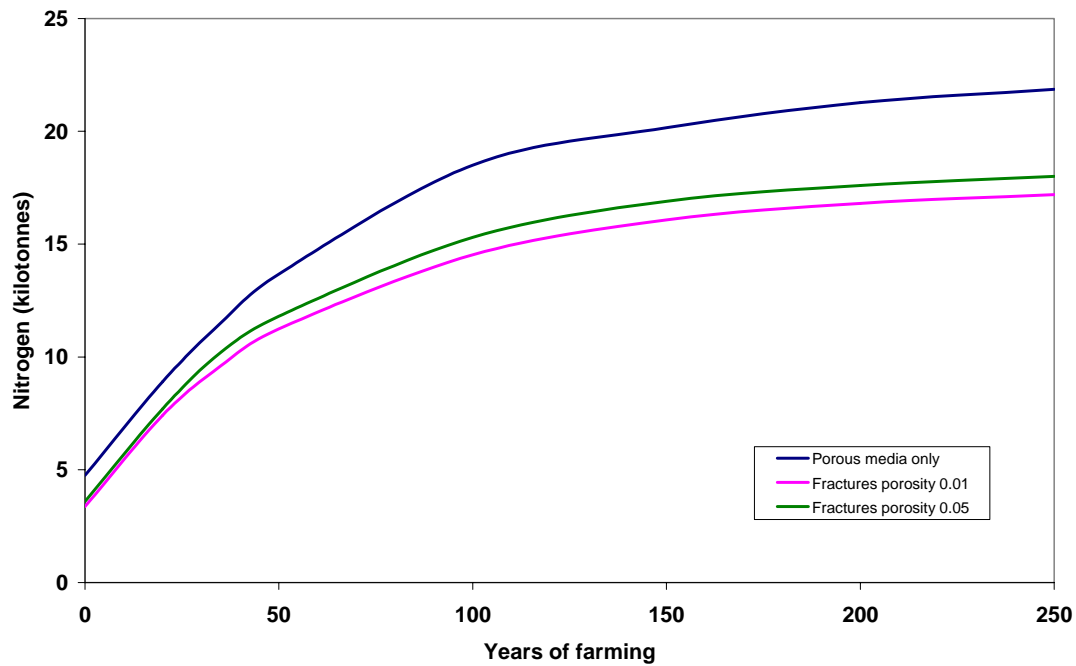
Figure 33: **Predicted nitrogen mass in the modelled groundwater system assuming uniform porous media porosity of 0.25; fracture porosity of 0.01 and 0.05 in the deeper western domain.**

8 Discussion – implications and predictive uncertainties

Models are inevitable simplifications of reality, based on numerous conceptual assumptions. The degree of appropriate complexity is dependent upon the objectives, the amount of information required to achieve a reasonable simulation and the availability of data. Whilst more detail could potentially be included, the reported modelling enables the fate and transport of nitrate (at this stage conservative) to be investigated, including spatial variation and temporal trends.

One of the inevitable challenges in the modelling was how to effectively differentiate geological units and practically represent these in the model, where relevant, for flow definition. The highly variable topography and constraints of model structure required to achieve convergence in Modflow limit the complexity possible at this scale. Complex topography is more readily represented by the finite element method and convergence is less problematic with a single layer model. The use of a multi-layer model, however, enables much more realistic simulation of transport, particularly vertically through the domain.

Although more detailed spatial variation and transient simulation could usefully be undertaken, it is important to initially test broader concepts and provide simplified predictions. The modelling reported here was not intended for farm scale investigation.

Apparent nitrogen loss of some 50 per cent is observed between estimated leaching rates from Overseer (Ledgard, 2000a,b) and observed surface water loads (Elliot and Stroud, 2001). This may be the result of a number of processes including attenuation in the vadose zone, denitrification in groundwater, assimilation in the riparian zone and in-stream uptake. This discrepancy may also be explained in part by lags. This would not, however, account for losses in streams with young waters. Such attenuation processes are not accounted for in the modelling reported here.

There are indications of denitrification occuring in the study area. Nearly 20 per cent of wells sampled show likely anaerobic conditions, which could lead to denitrification. The persistence of nitrogen in the oxidised form at some wells, where Eh/pH ratios suggest there is potential for denitrification, however, may be due to low organic carbon availability.

Other indications of reducing conditions include the degradation of CFCs and traces of methane found at some wells during groundwater age investigation (Morgenstern in prep.). Nitrate reduction in groundwater is also evident at the vadose research site in the Lake Taupo catchment (Stenger et al., 2007). Denitrification is reported in wetlands in the same Tutaeaua catchment (Sukias, 2004) and other wetlands at Turangi and Waimarino in the Lake Taupo catchment (Rosen et al., 1998). There is also some indication that even where oxic conditions predominate, there may be potential for some micro-environment nitrogen attenuation (Wang et al., 2003).

Denitrification would mitigate the magnitude of land use effects, but is unlikely to influence temporal trends. The introduction of such transformation within the model was not considered justified at this stage. This is given preliminary field investigation results reported in Appendix III, and the inferred minor denitrification from model calibration. More information is required to introduce denitrification rates into model transport.

Unsaturated transport through the vadose zone is also not included in the model and is yet another lag in the natural system. Similarly preferential flow paths such as rock fractures or natural piping are not addressed specifically. They are, however, generally accounted for in larger scale calibration through averaged hydraulic conductivity and

addressed in part by the introduction of some porosity variation. Preferential flow paths may be expected to locally reduce travel times.

Assumptions such as instantaneous farming development would accelerate effects. Exclusion of some streams from the model would conversely locally increase average flow paths and travel times. Although land use loading was simplified for model input, similar total loading (less than 10 per cent higher) was estimated by Brown et al., (2002) for the domain area.

The above assumptions may all be varied to investigate conceptual uncertainty. Inevitably there are, however, practical constraints which limit the use of multiple configurations to address such uncertainty. The basis for conceptual assumptions, such as the use of three continuous layers, is reported to enable this approach to be considered.

There are uncertainties in any modelling associated not only with the simplified conceptual basis but also with field measurement and parameter input. Uncertainty in field measurement is relatively small. Well heights have been measured to 0.02 m accuracy, water levels to a similar accuracy and water quality to standards imposed by Telarc registration. Whilst other errors may relate to temporal variation, overall greater uncertainties are introduced through conceptual modelling assumptions. Estimation and spatial extrapolation of parameters such as evapotranspiration and hydraulic conductivity also introduce significant uncertainty addressed in part by the sensitivity analysis.

The principal benefit of this modelling work has been to demonstrate the large time scale and insidiously gradual effects resulting from changes in diffuse land use loading. As stated, there is a substantial lag between land use change and the establishment of new equilibrium effects on groundwater and subsequently effects on the lake. The lag relates in large part to the relatively small addition of nutrient from leaching compared to the large store of groundwater in the aquifer system. Modelling also enables the mixing of 'new' and 'older' nitrogen in the system to be investigated and compared with age dating predictions. The combined application of modelling and dating techniques provides greater confidence in investigating and simulating the temporal and spatial effects of land use change.

Apart from investigating the efficacy of proposed mitigation measures, modelling is also useful for interpreting monitoring information by providing predicted response for comparison. For example, on the basis of the modelling assumptions described, a uniform 20 per cent reduction in manageable nitrogen load from land use would provide useful mitigation. Overall, nitrogen flux from northern catchment groundwater would nevertheless continue to increase, but much less significantly.

The modelling reported relates only to part of the Lake Taupo catchment and, therefore, predictions do not relate to the total lake catchment. The northern catchment is, however, considered to be the area providing the greatest future increase in nitrogen load due to the large component (~70%) of direct groundwater flow to the lake and old age of the streams (Morgenstern, 2007).

It is predicted that about half the total manageable nitrogen load of 235 tonnes from the northern catchment is still to come. The majority of this is expected via direct groundwater seepage. Only about 30 per cent of groundwater contribution from the northern catchment occurs indirectly via streams. The nitrogen load still to come from direct groundwater seepage is therefore estimated to be about 90 tonnes annually. This assumes no attenuation and is hence worst case, given that at least some minor denitrification can be expected.

There is a much smaller component of direct groundwater seepage from the western lake catchment and pastoral land use in the remaining (eastern) catchment is relatively

minor. The nitrogen load to come from direct groundwater seepage from the total lake catchment is estimated to be between 110 and 160 tonnes annually. This allows for about 15 per cent nitrogen attenuation. Current total nitrogen load to the lake from all sources is estimated to be a little over 1,300 tonnes annually.

A 20 per cent decrease in manageable nitrogen load to Lake Taupo is proposed by regional government (Environment Waikato, 2005). A fund has been established to buy farmland to convert from pastoral to forestry or other low nitrogen leaching use. Landowners in the area would have their current nitrogen output capped and would be able to increase this only by offsetting elsewhere through a nitrogen trading system. Even before this initiative was promulgated, active conversion of farms to dairying was effectively stopped by market concerns about future land use constraints.

9   Further research

Transient flow modelling could usefully be undertaken to provide greater insight into, and understanding of, recharge and transport processes. This would, however, require additional information and involve greater computational detail and hence time. Given that greater flexibility would be needed in model construction, it may be better to undertake this using the finite element method and more limited sub-catchments. Temporal groundwater level and chemistry data records are becoming sufficiently long to make this practical. Also, it would be helpful if additional local climate data became available. Relatively long stream records and other information for the Whangamata catchment would make this an obvious candidate for sub-catchment analysis.

The extent and distribution of dentrification within the Lake Taupo catchment groundwater is an important aspect warranting further research. Although there are many indications of denitrification, direct measurements of attenuation rates required for modelling are limited. Further single and dual well tracer testing would be useful.

Vadose zone processes are not represented in the current model and would be a further lag in the natural system. This is an aspect of considerable ongoing research by Lincoln Environmental (Barkle et al., 2005).

10   Summary

Three dimensional, numerical modelling undertaken using Visual Modflow and MT3D shows a considerable time lag between land use change and maximum nitrogen flux from groundwater. Flow calibration indicated groundwater velocities ranging from about 0.02 to 1.5 m d^{-1}. The highest flow rates occur in the Kinloch area, consistent with field estimates.

A groundwater recharge regime exists throughout the model. The Mapara, Whangamata, Otaketake and Waihora Streams modelled are influent and base flow dominated. With the exception of the Waihora catchment, direct seepage of groundwater to the lake from these catchments predominates. Groundwater travel time to the lake along some flow paths is in excess of 100 years.

Modelling indicates that nitrate-N concentrations under ambient conditions are about 0.25 to 0.5 g m^{-3}. Annual nitrogen discharge from groundwater in the northern catchment is estimated to be some 65 tonnes under ambient conditions with just under 3,500 tonnes estimated to be stored in the groundwater system.

Annual nitrogen discharge from current land use, modelled by increasing nitrogen loading as an instantaneous step and running this for 35 years, is estimated to be about 175 tonnes. It is predicted that some 10,000 tonnes of nitrogen would be stored in the groundwater system after this period of time.

Calibration of the transport model was improved by increasing the land use loading of dairy farming in the Kinloch area to 60 kg N ha ha^{-1} y^{-1} and removing four wells with groundwater chemistry indicating likely anaerobic conditions. A reasonable match was observed with temporal nitrogen data from the Whangamata Stream. This was slightly underestimated, whereas the Mapara Stream concentration was overestimated. The over-estimation may be the result of wetland denitrification and in-stream processes.

The full effect of current farming, was also modelled by continuing the current land use loading rate until equilibrium was observed. This took about 250 years, at which stage annual nitrogen flux from groundwater in the northern catchment was estimated to be about 300 tonnes y^{-1}, with about 18,000 tonnes of nitrogen being stored in the groundwater system. It is predicted that about half the total manageable nitrogen load of 235 tonnes is still to come. About 30 per cent of groundwater discharge to the lake from the northern catchment is via streams. The contribution still to come from direct groundwater seepage is therefore estimated to be about 90 tonnes, assuming no attenuation.

The modelling indicates that a proposed initiative to reduce the manageable nitrogen load from the lake catchment by 20 per cent would effectively mitigate the extent of increase otherwise expected in the northern catchment. It would not, however, given the assumptions, result in long-term remediation to current loading rates.

A predicted small instantaneous reduction in nitrogen concentration to streams was observed after reducing loading rate by 20 per cent before long-term concentrations continued to increase. This phenomenon was investigated by differentiating between nitrogen introduced to the model before this initiative from subsequent nitrogen input. Although a small amount of 'new' nitrogen appeared in streams immediately, full flushing of the older groundwater from the system took a long time (>100 years). This is in general agreement with water dating results in respect to mean residence times and mixing.

Land use effects in the Lake Taupo catchment are insidious, given substantial lags and subtle change. Groundwater modelling supported by water dating is useful in revealing long term trends and impacts. The substantial lag reflects the time required to replace old pristine groundwater with nitrogen enriched water from farming.

Further transient modelling is suggested to provide greater understanding of recharge and transport processes, particularly given useful temporal groundwater level and quality records are now becoming available.

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Appendix I: Well Input Details

Hydrol Number	Map Reference	Well depth (m)	Casing depth (m)	WL* (m)	RWL (m)	Geology	Nitrate-N* (ppm)
68.82	T18:699-752	38.0	32.5	18.68	398.32	Rhyolite pyroclastic	
68.244	U17:724-833	90.0	58.0	43.72	513.58	Rhyolite pyroclastic	
68.264	T17:647-806	45.1	39.3	12.40	400.56	Rhyolite pyroclastic	0.41
68.301	T17:656-803	53.6	49.0	38.40	365.03	Rhyolite pyroclastic	2.48
68.317	T17:605-811	104.0	73.0	54.00	506.37	Rhyolite pyroclastic	5.69
68.320	U18:749-747	62.0	43.0	15.34	369.18	Oruanui Ignimbrite	16.5
68.364	U17:706-843	161.5	97.5	70.47	521.17	Rhyolite pyroclastic	
68.377	U17:717-838	90.0	84.0	68.75	519.58	Oruanui Ignimbrite	
68.387	T19:482-497	128.0	96.0	79.41	418.75	Huka Formation	<0.02
68.423	T17:586-866	120.7	76.2	92.34	509.05	Whakamaru Ignimbrite	0.15
68.430	T17:613-848	134.1	73.4	69.80	479.71	Rhyolite pyroclastic	
68.446	T17:601-834	135.4	80.0	84.00	488.30	Rhyolite pyroclastic	
68.656	T17:675-812	40.0	32.5	29.84	420.98	Rhyolite pyroclastic	1.92
68.660	T17:643-817	37.7	19.5	20.35	420.90	Rhyolite	0.85
68.718	T17:642-806	67.1	58.5	26.32	389.17	Rhyolite pyroclastic	
68.801	T17:488-850	130.0	65.1	23.50	489.56	Whakamaru Ignimbrite	0.04
68.872	T17:645-800	72.0	39.0	30.55	362.61	Rhyolite pyroclastic	
68.915	T17:697-829	43.0	33.5	9.00	500.40	Oruanui Ignimbrite	1.59
68.964	T18:506-533	5.0	2.0	2.76	356.50	Post Taupo lacustrine	0.09
68.981	T17:479-849	109.0	65.2	47.00	452.13	Whakamaru Ignimbrite	
68.1000	T18:420-547	spring	spring	spring	spring	Whakamaru Ignimbrite	1.04
72.158	T17:613-812	277.6	169.8	167.50	396.52	Rhyolite pyroclastics	0.48
72.172	T17:681-819	148.0	98.5	40.10	437.66	Rhyolite pyroclastic	
72.173	T17:682-820	148.0	97.5	44.55	437.97	Rhyolite pyroclastic	
72.174	T17:683-821	148.0	97.5	55.67	426.58	Rhyolite pyroclastic	
72.331	T17:508-847	5.8	2.8	3.71	490.66	Oruanui Ignimbrite	3.64
72.332	T18:679-752	5.7	2.7	5.12	356.39	Taupo Ignimbrite	2.91
72.334	U18:741-741	18.0	NA	9.30	367.20	Taupo Ignimbrite	<0.02
72.335	T18:494-615	NA	NA	61.45	478.59	Whakamaru Ignimbrite	
72.338	U17:723-835	72.0	64.0	49.13	513.62	Rhyolite pyroclastic	
72.341	T17:566-833	116.7	36.3			Whakamaru Ignimbrite	1.15
72.343	T17:606-808	177.3	95.7	45.32	507.54	Whakamaru Ignimbrite	
72.348	T17:570-836	97.5	NA	83.24	446.99	Rhyolite pyroclastic	
72.349	T17:700-831	65.0	59.0	17.25	500.98	Rhyolite pyroclastic	
72.351	T17:633-845	NA	NA	*25.20*	504.71	Rhyolite pyroclastic?	

Hydrol Number	Map Reference	Well depth (m)	Casing depth (m)	WL[*] (m)	RWL (m)	Geology	Nitrate-N[*] (ppm)
72.352	T17:616-855	152.4	121.9	48.80	504.30	Rhyolite pyroclastic	0.27
72.354	T17:616-855	4.2	1.2	2.52	550.68	Oruanui Ignimbrite	1.45
72.356	T17:685-821	42.0	37.0	17.60	465.61	Rhyolite pyroclastic	1.94
72.358	U18:702-786	101.2	94.8	95.24	428.64	Rhyolite pyroclastic	
72.359	T18:686-763	157.9	43.6	69.27	362.69	Rhyolite pyroclastic	
72.361	T17:693-826	NA	NA	8.12	488.65	Rhyolite pyroclastic	
72.362	T17:645-807	NA	NA	6.00	412.20	Rhyolite pyroclastic	
72.363	T18:644-778	3.0	1.0	1.32		Post Taupo lacustrine	0.42
72.364	U17:665-824	spring	spring	spring	spring	Rhyolite lava	1.58
72.369	T17:520-876	46.5	1.2	2.67	469.02	Whakamaru Ignimbrite	5.56
72.370	U17:716-833	NA	NA	41.59	518.44	Rhyolite pyroclastic?	
72.371	T17:615-843	128.0	25.0	91.30	447.74	Rhyolite pyroclastic	
72.374	U17:721-831	51.8	39.0	35.47	516.10	Oruanui Ignimbrite	7.18
72.377	T17:508-847	50.3	45.4	4.51	489.92	Whakamaru Ignimbrite	<0.02
72.379	T17:527-843	38.0	32.6	2.61	507.40	Whakamaru Ignimbrite	2.39
72.381	T17:535-836	49.0	32.6	13.29	519.29	Whakamaru Ignimbrite	6.2
72.383	T17:546-839	38.4	32.6	8.80	539.23	Whakamaru Ignimbrite	3.55
72.384	T17:555-847	110.0	35.0	25.91	531.02	Whakamaru Ignimbrite	1.71
72.389	T17:564-868	57.5	12.8	6.23	528.78	Whakamaru Ignimbrite	1.6
72.390	T17:552-833	120.0	38.4	56.22	479.63	Whakamaru Ignimbrite	<0.02
72.391	T17:608-814	93.0	47.2	51.80	512.60	Rhyolite pyroclastic	
72.392	T17:609-813	93.0	65.2	51.40	513.22	Rhyolite pyroclastic	9
72.394	T18:527-612	175.1	NA	160.65	365.07	Whakamaru Ignimbrite	0.08
72.395	T18:524-616	156.7	NA	152.69	365.05	Whakamaru Ignimbrite	
72.397	T18:501-616	NA	NA	59.05	488.70	Whakamaru Ignimbrite	6.85
72.398	U17:720-812	NA	NA	66.29	477.90	Whakamaru Ignimbrite	
72.399	T18:511-562	6.0	NA	0.39	358.06	Post Taupo lacustrine seds	2.39
72.400	U17:728-827	NA	NA	40.82	513.70	Rhyolite pyroclastic?	
72.401	U18:702-788	NA	NA	106.30	431.42	Rhyolite pyroclastic	0.18
72.402	T17:642-806	NA	NA	18.65	397.51	Rhyolite pyroclastic	
72.403	T17:572-876	NA	NA	40.00	522.59	Whakamaru Ignimbrite?	
72.407	U18:702-747	NA	NA	3.58	408.69	Rhyolite pyroclastic?	0.12
72.408	T17:694-828	NA	NA	10.67	489.12	Rhyolite pyroclastic	
72.412	T17:522-846	45.1	NA	12.55	494.56	Whakamaru Ignimbrite	
72.413	T17:516-857	NA	NA	23.31	467.98	Whakamaru Ignimbrite	
72.414	U17:725-831	NA	NA	41.04	514.06	Rhyolite pyroclastic?	1.87

Hydrol Number	Map Reference	Well depth (m)	Casing depth (m)	WL[*] (m)	RWL (m)	Geology	Nitrate-N[*] (ppm)
72.415	T17:614-823	NA	NA	72.27	511.80	Rhyolite pyroclastic?	
72.424	T18:482-522	11.5	4.8	3.28	429.74	Post Oruanui lacustrine seds	0.63
72.426	T17:607-888	134.0	19.5	108.02	512.23	Whakamaru Ignimbrite	
72.431	T17:528-806	48.0	45.7	38.19	428.37	Whakamaru Ignimbrite	1.09
72.434	T18:646-790	17.0	11.3	11.29	359.38	Post-Oruanui pyroclastics	2.41
72.435	U18:732-784	NA	NA	39.48	481.27	Rhyolite pyroclastic?	<0.02
72.466	T17:605-811	100.0	58.6	48.80	511.19	Rhyolite pyroclastics	

[*] Information collated summer 2000

Appendix II   Model Technical Basis

Modflow is a computer program that simulates three-dimensional groundwater flow through porous media using a finite difference method (McDonald and Harbaugh, 1988). The partial-differential equation of ground-water flow used in MODFLOW is:

$$\frac{\partial}{\partial x}\left(K_{xx}\frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_{yy}\frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_{zz}\frac{\partial h}{\partial z}\right) + W = S_s\frac{\partial h}{\partial t} \tag{1}$$

where:

K_{xx}, K_{yy} and K_{zz} are values of hydraulic conductivity along the x, y, and z coordinate axes, and assumed to be parallel to the major axes of hydraulic conductivity (L/T);

h is the potentiometric head (L);

W is a volumetric flux per unit volume representing sources and/or sinks of water, with W<0.0 (negative) for flow out of the groundwater system, and W>0.0 (positive) for flow in (T^1);

S_s is the specific storage of the porous material (L^{-1}); and

t is time (T).

This equation (1) when combined with boundary and initial conditions, describes transient three-dimensional groundwater flow in a heterogeneous and anisotropic medium, provided that the principal axes of hydraulic conductivity are aligned with the coordinate directions.

The groundwater flow process solves equation 1 using the finite-difference method in which the groundwater flow system is divided into a grid of cells (Figure II.1). For each cell, there is a single point, called a node, at which head is calculated. The finite-difference equation for a cell (McDonald and Harbaugh, 1988) is:

$$
\begin{aligned}
&CR_{i,j-\frac{1}{2},k}\left(h^m_{i,j-1,k}-h^m_{i,j,k}\right) \;+\; CR_{i,j+\frac{1}{2},k}\left(h^m_{i,j+1,k}-h^m_{i,j,k}\right) \\
&+\; CC_{i-\frac{1}{2},j,k}\left(h^m_{i-1,j,k}-h^m_{i,j,k}\right) \;+\; CC_{i+\frac{1}{2},j,k}\left(h^m_{i+1,j,k}-h^m_{i,j,k}\right) \\
&+\; CV_{i,j,k-\frac{1}{2}}\left(h^m_{i,j,k-1}-h^m_{i,j,k}\right) \;+\; CV_{i,j,k+\frac{1}{2}}\left(h^m_{i,j,k+1}-h^m_{i,j,k}\right) \\
&+\; P_{i,j,k}h^m_{i,j,k} \;+\; Q_{i,j,k} \;=\; SS_{i,j,k}\left(DELR_j \times DELC_i \times THICK_{i,j,k}\right)\frac{h^m_{i,j,k}-h^{m-1}_{i,j,k}}{t^m-t^{m-1}}
\end{aligned}
\tag{2}
$$

where:

$h^m_{i,j,k}$, is head at cell I,j,k at time step m (L);

CV, CR, and CC are hydraulic conductances, or branch conductances, between node I,j,k and a neighbouring node (L^2/T);

$P_{i,j,k}$ is the sum of coefficients of head from source and sink terms (L/T); -

$Q_{i,j,k}$ is the sum of constants from source and sink terms, with $Q_{i,j,k}$ <0-0 for flow out of the groundwater system, and $Q_{i,j,k}$ >0.0 for flow in (L^3/T)

$SS_{i,j}$k is the specific storage (L^{-1});

DELR, is the cell width of column j in all rows (L);

DELC, is the cell width of row i in all columns (L);

$THICK_{i,j,k}$ is the vertical thickness of cell I,j,k (L); and
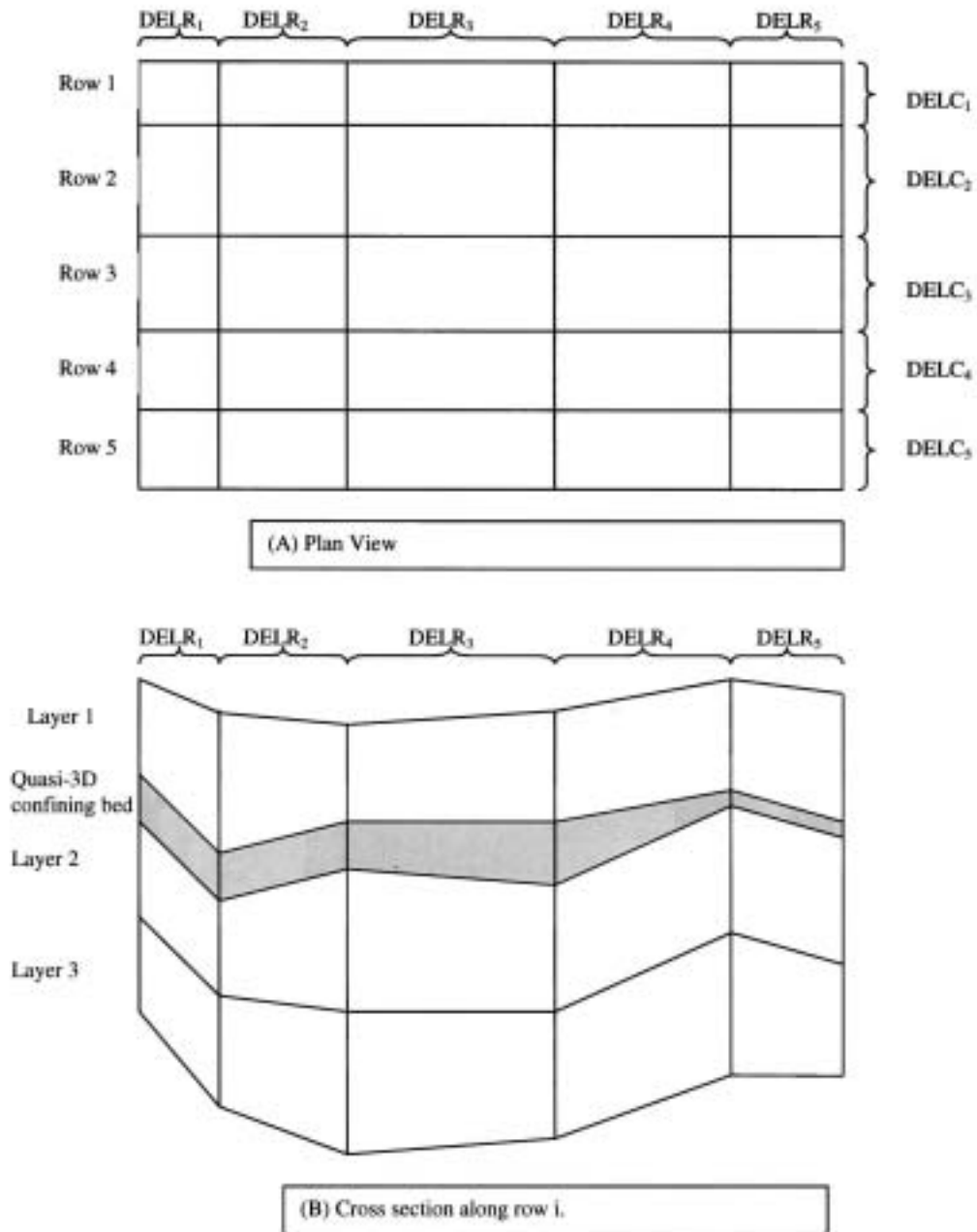
t^m is the time at time step in (T).

Figure II.1: Finite difference grid with (A) plan view and (B) cross-section view.

The technical basis of the MT3D modelling software and underlying fundamentals of contaminant transport modelling are described by Zheng (1990). The partial differential equation describing three-dimensional transport of contaminants in groundwater can be written as:

$$\frac{\partial(\theta C^k)}{\partial t} = \frac{\partial}{\partial x_i}\left(\theta D_{ij}\frac{\partial C^k}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(\theta v_i C^k) + q_s C_s^k + \sum R_n \qquad \text{Eqn (3)}$$

where

C^k	is the dissolved concentration of species k, ML^{-3};
θ	is the porosity of the subsurface medium, dimensionless;
t	is time, T;
x_i	is the distance along the respective Cartesian coordinate axis, L;
D_{ij}	is the hydrodynamic dispersion coefficient tensor, L^2T^{-1};
v_i	is the seepage or linear pore water velocity; LT^{-1}; it is related to the specific discharge or Darcy flux through the relationship, $v_i = q_i/\theta$;
q_s	is the volumetric flow rate per unit volume of aquifer representing fluid sources (positive) and sinks (negative), T^{-1};
C_s^k	is the concentration of the source or sink flux for species k, ML^{-3};
$\sum R_n$	is the chemical reaction term, $ML^{-3}T^{-1}$.

The effect of biochemical and geochemical reactions on contaminant transport can be introduced for the chemical reaction term. Chemical reaction can be expressed in terms of aqueous-solid surface reaction (sorption) and first order rate reaction by:

$$\sum R_n = -\rho_b \frac{\partial \overline{C}^k}{\partial t} - \lambda_1 \theta C^k - \lambda_2 \rho_b \overline{C}^k \qquad \text{Eqn (4)}$$

where

ρ_b	is the bulk density of the subsurface medium, ML^{-1};
$\overline{C}^k$	is the concentration of species k sorbed on the subsurface solids, MM^{-1};
λ_1	is the first-order reaction rate for the dissolved phase, T^{-1};
λ_2	is the first-order reaction rate for the sorbed (solid) phase, T^{-1}.

A single effective porosity is assumed in the transport equations. Due to field complexity of such pore structure, it is a lumped parameter derived in model calibration with plume movement or observed solute accumulation effects. In certain situations, such as fractured aquifers or extremely heterogeneous porous media, a dual-porosity approach may be more appropriate. This can be done by defining a primary porosity for pores with mobile water where advection predominates and a secondary porosity for pores with immobile water where transport is primarily by molecular diffusion. Exchange between the mobile and immobile domains can be defined by a kinetic mass transfer equation similar to that used to describe non-equilibrium sorption.

The transport equation is related to the flow equation through Darcy's Law:

$$v_i = \frac{q_i}{\theta} = -\frac{K_i}{\theta}\frac{\partial h}{\partial x_i}$$ Eqn (5)

where

K_i is a principal component of the hydraulic conductivity tensor, LT^{-1};

h is hydraulic head, L.

The hydraulic head is obtained from the solution of the three-dimensional groundwater flow equation:

$$\frac{\partial}{\partial x_i}\left(K_i \frac{\partial h}{\partial x_i}\right) + q_s = S_s \frac{\partial h}{\partial t}$$ Eqn (6)

where S_s is the specific storage of the aquifer, L^{-1}, and q_s is the fluid sink/source term as

defined in equation 3

Implied in equations (5) and (6) is the assumption that the principal components of the hydraulic conductivity tensor i.e. K_x, K_y and K_z are aligned with x, y and z coordinate axes so that all non-principal components (cross terms) become zero. This assumption is incorporated in most commonly used finite difference groundwater flow models including Modflow (McDonald and Harbaugh, 1988).

Advective transport is dominant in field-scale contaminant transport modelling such as this. Dispersion is the spreading of contaminants by both mechanical dispersion and molecular diffusion. The latter is typically negligible in other than very fine grained media.

Boundary and initial conditions are required to solve the governing equation. Three boundary types are accommodated in MT3D. These are where:
- concentration is known along a boundary (Dirichlet condition)
- concentration gradient is known across a boundary (Neuman condition), and
- a combination of the above (Cauchy condition).

Appendix III   Field Investigation of Denitrification in Taupo Catchment Groundwater

Background

The potential for removal of nitrogen nutrient from groundwater through the process of denitrification is of fundamental importance to the consideration of land use effects and management. The aim of the following preliminary work was to trial an investigative approach used by Trudell et al., (1986) and to infer the incidence and rate of in-situ denitrification within the catchment to improve information for nitrogen transport modelling.

This approach does not account for denitrification that may be occurring in the vadose zone. This aspect is being addressed through intensively instrumented research by Lincoln Ventures at a site in the northern Lake Taupo catchment (Barkle et al., 2005).

Denitrification is evident associated with wetlands in the lower reaches of some streams e.g. the Tutaeaua (Sukias, 2004), Otaketake and Whangamata Streams in the northern catchment. Also about 20 per cent of samples from the phase 1 round of groundwater quality investigation showed the likely occurrence of anaerobic conditions which could lead to denitrification (Hadfield et al., 2001). Shallow piezometers subsequently drilled in the western catchment also showed some likely anaerobic conditions. These were sufficient to influence CFC dating analysis at some sites.

Eh/pH analyses from several sites (Figure III.1), showed that there is potential for denitrification. Nitrogen at these sites, however, occurs predominantly in the nitrate form. This persistence of the oxidised form is possibly due to relatively low dissolved organic carbon (DOC). This may effectively slow the microbially mediated denitrification process or prevent it overcoming a conversion threshold.

Method

An indication of denitrification rate in groundwater can be inferred by observing the reduction in concentration of nitrate injected into groundwater over time. An in-situ injection experiment was conducted whereby nitrate and bromide (a conservative tracer for comparison) were added to natural groundwater and re-injected into shallow piezometers.
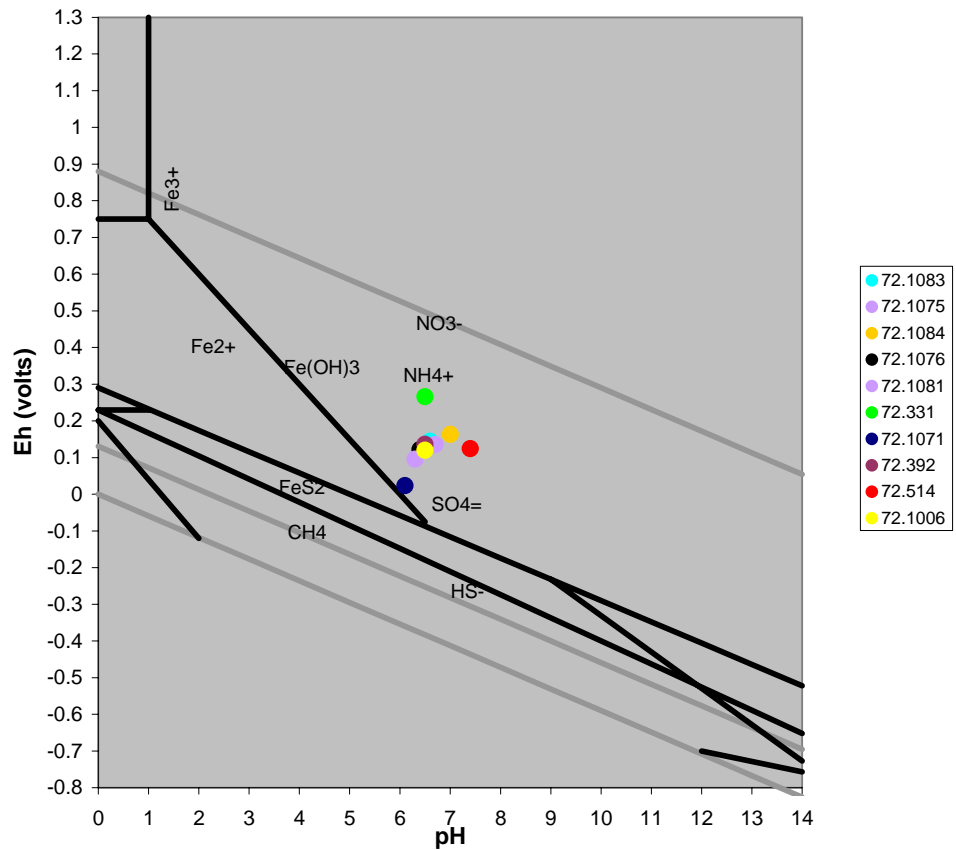
Figure III.1   Eh/pH diagram for Fe/S/N/C system at sampled wells

Initially two existing shallow piezometers in the Taupo catchment were selected (72.1007 on Tukairangi Rd and 72.1011 at Whakaipo Bay), to represent likely denitrifying (72.1007) and non-denitrifying conditions (72.1011). Critical aspects of site selection are that there is a small enough annular volume to enable practical purging of the well and that the permeability is low enough to allow denitrification to occur before the tracers migrate beyond the reach of sampling. Four further sites were subsequently tested. They comprise one piezometer specifically constructed into swampy sediments at a property on Collins Rd near Hamilton (well 72.2859) to test the method and three further existing piezometers in the Lake Taupo catchment.

Typically 190 L of groundwater was gently pumped from the shallow piezometers using submersible pumps to minimize aeration. Water level and D.O. were measured before pumping and a sample of the water pumped taken for pH, Cl, HCO_3, NO_3, NH_4, Br and dissolved organic carbon (DOC) analyses. Generally 25 g of potassium nitrate and 15 g of potassium bromide were added. A small amount of Rhodamine WT dye was also introduced for visual confirmation of injection water presence.

A sample of the injection water was taken for analysis to confirm chemical composition. The spiked water was pumped back down the piezometers through plastic tubing below the water table. Groundwater from the piezometers was subsequently sampled on an approximately daily basis for pH, Cl, HCO_3, NO_3, NH_4 and Br analyses. Dissolved oxygen and water level were also noted. Cooled but unfiltered, samples were sent to the laboratory for analysis within 24 hours.

Results

A total of five sites were tested in the Lake Taupo catchment. The first two (72.1007 and 72.1011) showed no indication of denitrification - that is, there was no significant decline in nitrate-N:Br ratio at 72.1007. The N:Br ratio at 72.1011 actually increased

markedly as additional nitrate migrating to the piezometer was sampled. Site 72.1071 (Waihaha) showed a steady increase in N:Br ratio during testing, indicating no dentrification.

A pair of piezometers was tested at Rangiatea in the western Lake Taupo catchment. The 'shallow' piezometer (72.1087) is 6.6 m deep and the deeper piezometer is (72.1072) is 21 m deep. Geology is essentially unwelded pumiceous ignimbrite but with peaty sediments logged between about 10 and 11 metres below ground. The shallow piezometer showed an increase in N:Br ratio and hence no dentrification. The deeper piezometer by contrast, showed a decline in N:Br ratio with one possible outlier (Figure III.2). The nitrate concentration, corrected by a dilution factor based on the Br concentration, declines steadily at a rate of about 0.1 g m^{-1} N hr^{-1}. Dissolved oxygen did not decrease significantly during the monitoring period of 140 hours. There was also only a slight increase in alkalinity from 37 to 44 g m^{-3}.
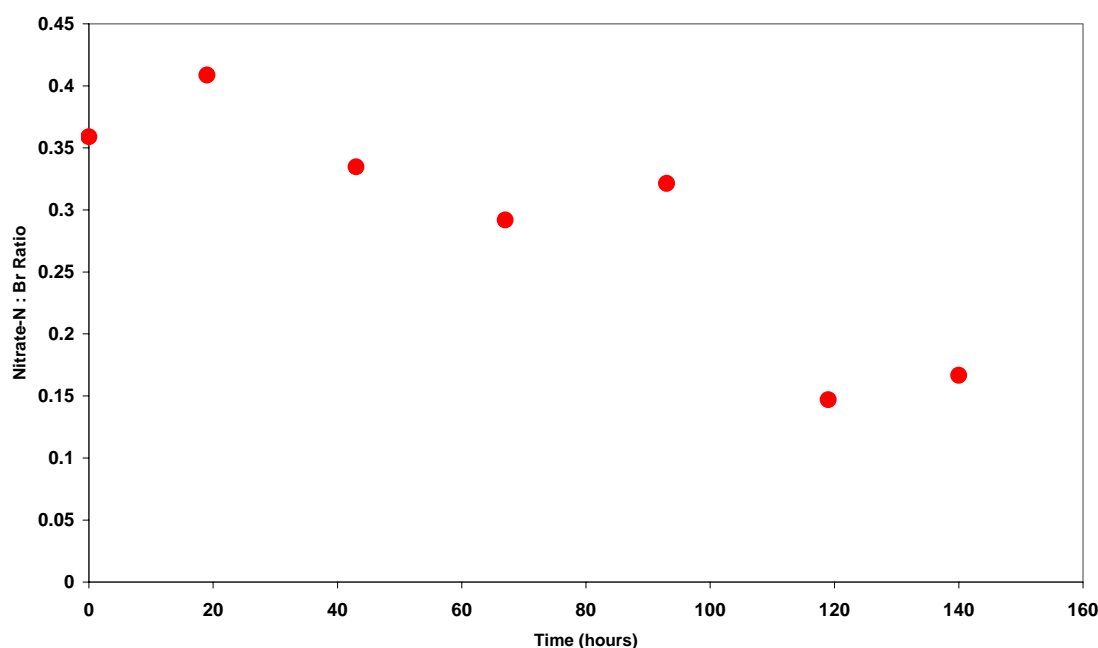

Figure III.2   Nitrate-N : Bromide ratio at site 72.1072

DOC in the 'deep' monitoring well was the highest of the five Taupo sites tested at about 4mg l^{-1}. The other sites 72.1011, 72.1007, 72.1071 and 72.1087 had DOC concentrations of 0.5, 1.0, 1.5 and 2.9 g m^{-3} respectively.

It was decided that the method should be checked by carrying out an experiment in what was considered likely to be a strongly denitrifying situation close to Environment Waikato's Hamilton office. A shallow (6 m) monitoring well (72.2859) was specifically constructed in peaty sediments at Rukuhia. A strong reduction in the ratio of nitrate-N to bromide after injection at this site indicates that substantial denitrification is occurring.

The 169 L of groundwater and tracer were injected with a bromide concentration of 61 ppm and nitrate-N concentration of 17.9 ppm. A ten-fold decrease in the N:Br ratio was apparent after about 70 hours. The rate of nitrate reduction corrected for dilution was initially about 0.5 g m^{-3} N hr^{-1} and reduced to about 0.03 g m^{-3} N hr^{-1} below a concentration of 2 g m^{-3}.

It was noted that there was about a ten-fold increase in ammonium concentration between the ambient (0.28 ppm) and injection (3 ppm) concentrations. This was similar to the discrepancy between the design mixing concentration of the nitrate concentration of 20.5 ppm and the 17.9 ppm measured in the injected water in the lab. This implies that probably very rapid denitrification of the more labile nitrogen was occurring between injector tracer mixing and laboratory measurement.

One of the likely reasons such strong denitrification was observed was the high DOC concentration of 43 ppm. There was no consistent or significant change in dissolved oxygen noted during the test. (Analytical results of the testing are documented in Environment Waikato's document 1064322.)
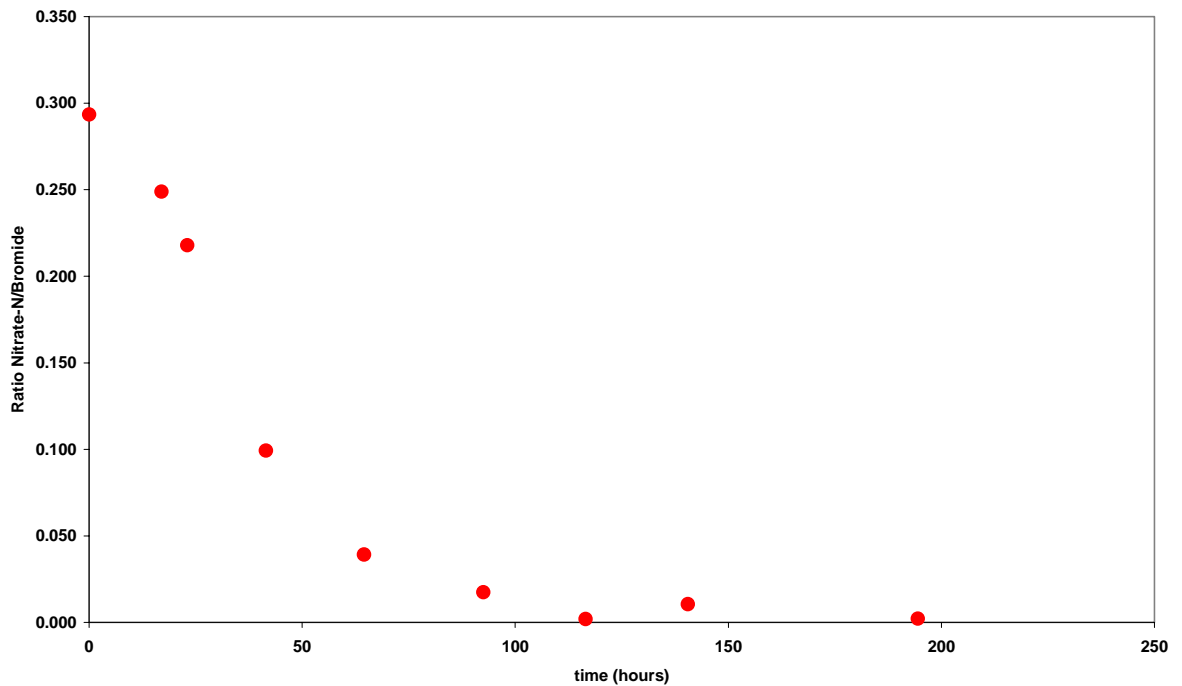

Figure III.3   Nitrate-N : Bromide ratio at site 72.2859

Discussion

The denitrification rates above are similar to literature rates such as 7.8×10^{-3} to 1.3×10^{-1} g m^{-3} N hr^{-1} of Trudell et al., (1986); 0.026 to 0.119 g m^{-3} N hr^{-1} (Focht and Joseph, 1974 in Trudell et al., 1986); 0.27 g m^{-3} N hr^{-1} (Doner et al., 1974 in Trudell et al., 1986) and 0.83 g m^{-3} N hr^{-1} (Focht et al., 1973).

Apart from anaerobic conditions, an organic carbon source is required for bacteria to metabolise nitrate and mediate the denitrification process (Starr and Gillham, 1993). The higher rates of denitrification being found where DOC is higher, is consistent with work reported by Trudell et al., (1986). Although decreasing dissolved oxygen and increasing alkalinity are generally expected during denitrification, no such trend was evident in this investigation.

Analysis of DOC from 37 sites in the Lake Taupo catchment in June 2002 showed a range in concentration from 0.5 to 13.7 g m^{-3} but is typically low, with a median concentration of 1.3 g m^{-3} and standard deviation of 2.96 g m^{-3}.

Site 72.1007, which was originally thought highly likely to be anaerobic, has subsequently been found to have slight (up to 0.44 ppm), but slowly increasing, nitrate concentrations. This is consistent with test results indicating no active denitrification at this site.

Unfiltered samples could potentially allow further microbially mediated denitrification to occur after sampling. In this respect, that lack of apparent reduction in the N:Br ratio at many of the sites is a more conservative indication that denitrification is not occurring.

One of the difficulties using this method is to find suitable experimental sites. Clearly the investigation method works only in anaerobic environments where no new nitrate-N is arriving through migration to the site. Also, the monitoring wells must be sufficiently small to enable feasibly small volumes to be pumped for tracer injection and sampling. Groundwater velocities in the 'aquifer' must also be sufficiently slow to enable denitrification to occur before migrating away from the sampling influence but fast enough to enable a reasonable volume of water to be abstracted for re-injection.

Summary

In-situ denitrification tests were carried out using five piezometers in the Lake Taupo catchment. A method similar to that of Trudell et al., (1986) was used, which involves injecting nitrate and tracers into shallow groundwater and observing the relative rate of nitrate concentration reduction. The method was checked at a peaty Hamilton site where strong denitrifying conditions were expected. A high rate (about 0.5 g m^{-3} N hr^{-1}) of nitrate attenuation was observed and the approach found to be useful in such anaerobic conditions.

Only one of the five Taupo sites (72.1072) investigated showed a decline in nitrate concentration relative to the conservative bromide tracer, indicating denitrification is occurring. Peaty sediments are reported at this site and the DOC was highest of the five Taupo sites. The observed denitrification rate of 0.1 g m^{-3} N hr^{-1} is similar to literature reported values. This preliminary investigation shows that denitrification is occurring in shallow groundwater in the Lake Taupo catchment. However, even where anaerobic conditions exist, there may not be active denitrification, probably due to a lack of available organic carbon.

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