

## 2. Baseline Groundwater Quality

The concept of 'baseline' groundwater quality is typically utilised to characterise the natural water chemistry of water contained in an aquifer system prior to influence by human (anthropogenic) impacts. Knowledge of groundwater quality baseline therefore provides a datum against which changes in the chemical composition of groundwater due to human influences can be assessed. For example, definition of 'baseline' groundwater quality is a key component of the European Water Framework Directive for groundwater quality management applied across the European Union<sup>1</sup>. Under this management approach 'baseline' groundwater quality is defined as:

*'the range in concentration (within a specified system) of an element, species or chemical substance present in solution which is derived from natural geological, biological or atmospheric sources'* (Shand *et al.* 2007)

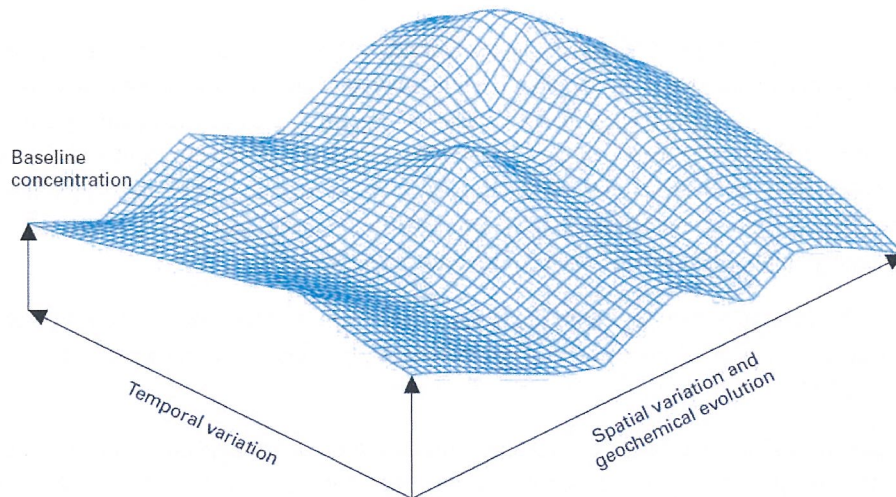
As previously noted, the concept of 'baseline' groundwater quality applied to resource consent conditions for the CPW scheme is different in that it encompasses the current and future state of groundwater quality in, and down gradient of, the CPW scheme area, in the absence of effects associated with the CPW scheme.

While baseline groundwater quality is a relatively simple concept to describe, it is much difficult to describe in quantitative terms. This difficulty arises because groundwater quality is not static but varies dynamically across an aquifer system (even in the absence of significant land use effects) with location, depth and over time. This variability is increased in situations where land use and abstraction alter the loading of chemical constituents contained in water recharging the aquifer system as well as the natural pattern of groundwater flow. The concept of temporal and spatial variability in baseline groundwater quality is illustrated in **Figure 2** below.

The following section provides a hydrogeological and geochemical overview of the aquifer systems underlying the Central Plains area and illustrates some of the processes which influence spatial and temporal variability in groundwater quality.

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<sup>1</sup> <http://ec.europa.eu/environment/water/water-framework/>



**Figure 2. Illustration of spatial and temporal variations in baseline groundwater quality resulting from variations in landscape, climate, chemical inputs and residence time (from Shand *et al.*, 2007)**

## **2.1. Hydrogeology and geochemistry of the Central Plains area**

### **2.1.1. Hydrogeology**

The Central Plains comprise a broad alluvial fan surface which extends between the Rakaia and Waimakariri rivers from the foothills in the west to the coast south of Banks Peninsula. The subsurface geology of this area comprises alluvial gravel materials deposited and reworked by the braided alpine rivers in response to climatic variation associated with glacial and interglacial cycles during the late Pleistocene (Brown and Webber, 2001). These alluvial materials host an extensive aquifer system which is extensively utilised for domestic, public and municipal water supplies and which also provides baseflow to rivers and streams in lowland areas.

Along the coastal margin, the aquifer system comprises a sequence of permeable gravel layers separated by fine-grained marine and estuarine silt deposits accumulated during periods of sea level highstand during late Pleistocene interglacial periods. These silt layers form low permeability aquitards which limit the upward flow of water resulting in flowing artesian conditions in coastal areas. Further inland, the alluvial materials comprise a thick sequence of heterogeneous alluvial materials containing layers of relatively coarse, well sorted gravels interspersed with lower permeability gravel materials containing variable amounts of silt and sand. Various investigations (e.g. Davey, 2006) have attempted to group water-bearing units encountered at similar depths in individual wells into discrete 'aquifers' based on well screen depths (as a proxy for permeability) separated by less permeable silt-bound gravel aquitards. However, evaluation of aquifer test data from the Central Plains area indicates a degree of vertical leakage between individual water-bearing layers suggesting

the entire aquifer system is hydraulically connected, at least to some degree (SKM, 2008, Williams and Lough, 2009).

Recharge to the Central Plains groundwater system is derived from two main sources; seepage from the alpine rivers (including a lesser volume from the Selwyn River and tributaries) and infiltration of rainfall and irrigation water (including stock water races) across the surface of the plains. Across the upper sections of the plains both the Rakaia and Waimakariri Rivers are incised into the surrounding alluvial fan materials by as much as 50 metres and appear to be perched above the surrounding water table. However, significant flow losses are observed from their middle reaches to the coast. These losses are estimated at up to 15 m<sup>3</sup>/sec for the Rakaia River and 10 m<sup>3</sup>/sec from the Waimakariri River (Scott and Thorley, 2009), a portion of which flows through the coastal section of the Central Plains aquifer system.

Discharge from the groundwater system primarily occurs through abstraction and discharge from springs and lowland streams.

### 2.1.2. Geochemistry

The earliest groundwater quality investigations in the Central Plains area were carried out by the North Canterbury Catchment Board (NCCB, 1983). Using these data Burden (1984) divided the Central Plains into three chemical zones:

- Zone 1 - areas adjacent to the Rakaia, Selwyn and Waimakariri rivers exhibiting low Nitrate-N concentrations associated with river recharge flux;
- Zone 2 - unconfined groundwater to a depth of approximately 60 metres below the water table across a majority of the inland plains away from the riparian margins of the major rivers which exhibited elevated, and seasonally variable Nitrate-N and Chloride concentrations associated with land surface recharge
- Zone 3 - all groundwater across the Central Plains area located more than 60 metres below the water table containing low Nitrate-N and Chloride concentrations.

Taylor *et al.*, (1989) utilised groundwater chemistry data (including oxygen and hydrogen isotope ratios) to distinguish between rainfall and river recharge. While analysis of sample results generally supported the water quality zonation proposed by Burden (1984), the detection of <sup>18</sup>O depleted water suggested a contribution of alpine river seepage to recharge of deep groundwater, particularly near the coastal confined aquifer system. Stewart *et al.*, 2002 also identified a recharge contribution from alpine rivers to deeper groundwater in coastal areas of the Central Plains. Groundwater in the Christchurch artesian system was identified as being primarily derived from Waimakariri River recharge, while deep groundwater near Lake Ellesmere was interpreted as being sourced from a combination of Selwyn River losses and rainfall recharge. Further south, deeper groundwater in coastal areas was interpreted to be primarily sourced from the Rakaia River. In contrast, the <sup>18</sup>O enriched signature of shallow groundwater in inland areas removed from the major river systems was interpreted to reflect land surface recharge by low altitude rainfall.

Regional-scale nitrate transport modelling undertaken by Bidwell (2005) in the Central Plains area suggested a dispersive interface separating groundwater from two separate recharge sources; deeper

groundwater sources from river recharge containing low nitrate concentrations and shallower groundwater dominated by land surface recharge containing elevated nitrate concentrations. The relatively low nitrate concentrations observed in the lowland groundwater discharge zone were attributed to the dilution of shallower, higher nitrate groundwater with upwelling of deeper, low nitrate river recharged groundwater toward the coastal margin.

Hanson and Abraham (2009) analysed chemical and isotope data from 78 wells of varying depths along two transects across the Central Plains. The main conclusions of this investigation include:

- Groundwater from two different recharge sources; alpine river seepage and land surface recharge, can be clearly identified based on differences in water chemistry;
- Groundwater seepage from the alpine rivers flows into deeper parts of the aquifer system and re-emerges near the coast. Groundwater recharged by land surface infiltration remains in shallower parts of the aquifer system (<100 m below the water table) and discharges in the upper reaches of lowland streams. The depth of the interface between groundwater derived from land surface and river recharge varies spatially across the Central Plains area;
- Groundwater chemistry does not appear to be significantly influenced by stratigraphy of the inland plains suggesting hydraulic connection and circulation of groundwater between different water-bearing layers.

## **2.2. Factors influencing spatial and temporal variation in groundwater quality**

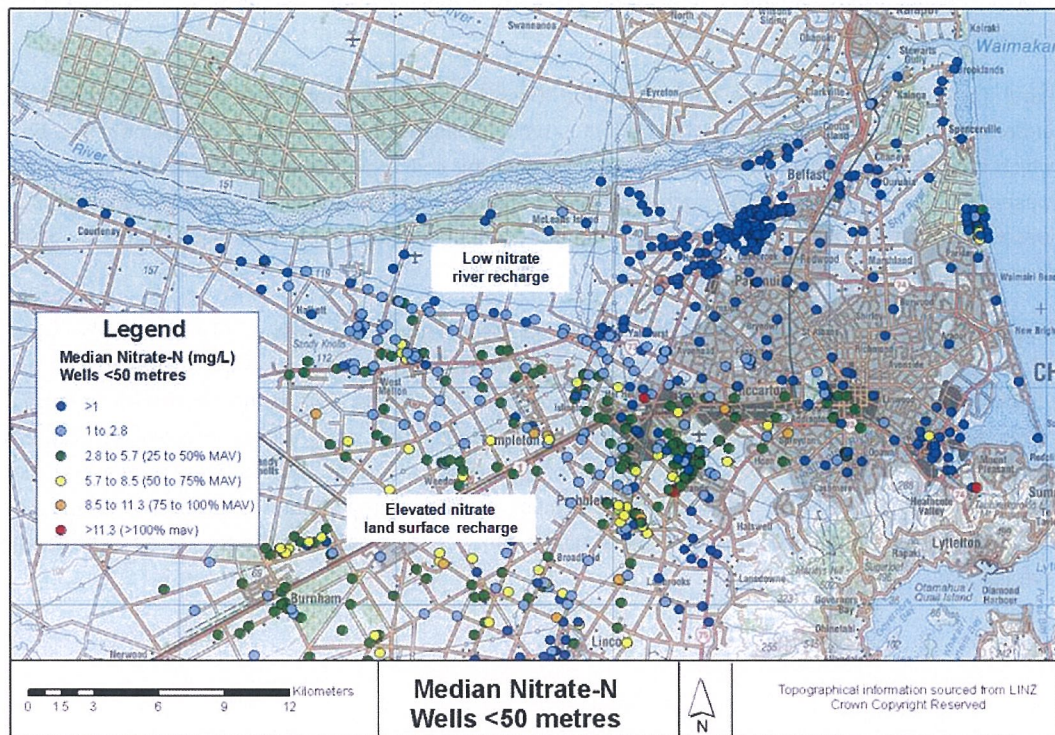
The chemical (and microbial) composition of groundwater at any given point within an aquifer system reflects the cumulative effects of a number of a large number of factors ranging from the source and composition of recharge water, the timing and magnitude of recharge flux, chemical and biological processes in the unsaturated zone as well as dilution, dispersion, redox processes, water-rock interaction and rate of groundwater flow along any given flow path within the saturated zone.

### **2.2.1. Recharge source**

As previously noted, the Central Plains aquifer system is recharged by a combination of infiltration of local rainfall and flow loss from the Rakaia, Waimakariri and Selwyn rivers. While recharge from surface waters typically contains relatively low concentrations of dissolved ions, recharge via soil infiltration (i.e. land surface recharge) is often enriched with respect to parameters such as sodium, potassium, chloride and sulphate, and other chemical species which accumulate during passage of water through the soil zone. Depending on land use, recharge infiltrating through the soil zone may also contain elevated concentrations of Nitrate-N, in contrast with the relatively low concentrations typically present in recharge from rivers and streams. As a consequence, the relative proportions of aquifer recharge derived from different recharge sources can directly influence spatial variations in groundwater quality across an aquifer system.



The influence of recharge source on groundwater quality in the Central Plains is illustrated on **Figure 3** below which shows the a clear transition between low Nitrate-N groundwaters recharged by seepage losses from the Waimakariri River to the west of Christchurch and higher Nitrate-N groundwaters dominated by land surface recharge to the south. A similar surface influence on groundwater quality due to the dominance of river recharge is observed along the margins of the both the Rakaia River and the lower Selwyn River, as well as in deeper groundwaters (typically >50 metres below ground) toward the coastal margin.



**Figure 3. Median Nitrate-N concentrations in wells <50 metres deep to the west of Christchurch city illustrating the influence of the relative contribution of river recharge and land surface recharge on median groundwater Nitrate-N concentrations**

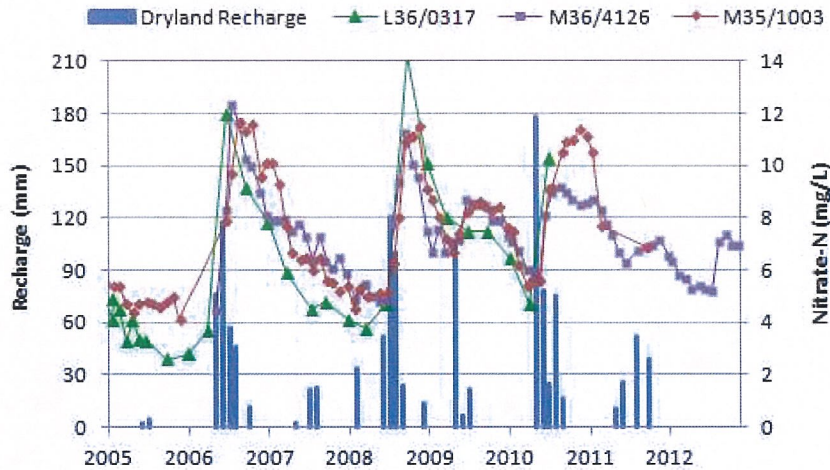
As noted in **Section 2.2.1**, this differentiation between river recharge dominated groundwaters containing low concentrations of dissolved ions (including Nitrate-N) and those derived from land surface recharge containing elevated concentrations of dissolved ions is a major control on the spatial and depth distribution of groundwater quality across the Central Plains.

### 2.2.2. Recharge Flux

Land surface recharge under dryland conditions typically occurs between late autumn and early spring when soils are at or near field capacity for an extended period. During these conditions, rainfall in excess of soil water holding capacity can initiate drainage through the underlying vadose zone. Under irrigated land use, maintenance of higher soil moisture levels throughout the year acts to initiate additional soil moisture drainage. For example, results of soil moisture modelling undertaken by Environment Canterbury (David Poulsen, *pers comm.*) indicate average annual recharge approximately doubles in mid-plains area (e.g. West Melton/Dunsandel) from around 220 mm under dryland conditions to 430 mm under irrigation. Significant recharge events may also be initiated following heavy snowfalls (Close, 2009).

Land surface recharge acts to flush dissolved ions (including nutrients) through the soil zone into underlying groundwater. This leaching effect is typically associated with land management practices which enable accumulation of mobile contaminants in the soil zone and may also occur due to direct losses associated with activities such as winter grazing or soil cultivation. Depending on land use, losses from the soil zone can be exacerbated due to relatively low rates of nutrient uptake by plants over the winter period,

Many, although by no means all, shallow wells in the Central Plain area exhibit seasonality which appears associated with episodic land surface recharge inputs. For example, **Figure 4** shows temporal variations in groundwater Nitrate-N concentrations in three wells in the Central Plains area in response to calculated land surface recharge. The data show a significant increase in Nitrate-N concentrations immediately following periods of significant recharge, with concentrations declining during intervening periods. This suggests significant contaminant inputs to groundwater may be episodic and with a significant proportion of nitrate flux occurring in response to significant recharge events.

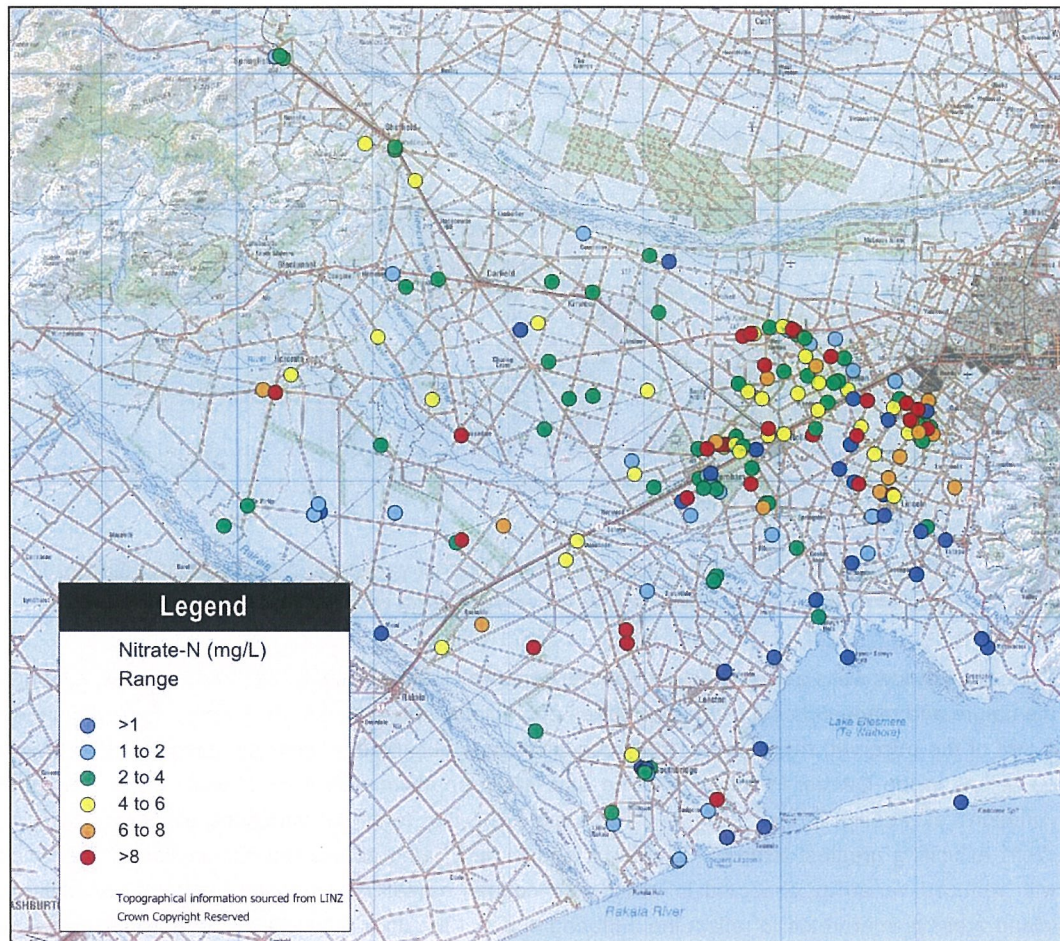


**Figure 4. Calculated land surface recharge and temporal variation in groundwater Nitrate-N concentration in three wells in the Central Plains area**

Typically, shallow wells exhibit greater temporal variability in water quality than deeper wells, although the nature and magnitude such changes varies appreciably between individual wells. **Figure 5** shows a plot of the observed range of Nitrate-N concentrations observed in wells sampled on more than 6 occasions in the Central Plains area. The data show approximately 70% of wells exhibit temporal variations exceeding 2 mg/L, with 40 % of wells exhibiting temporal variations exceeding 4 mg/L. Wells exhibiting greatest temporal variability tend to be located across the mid-section of the plains with reduced variability observed in inland and coastal areas. The reduced temporal variability in inland areas is interpreted to reflect the buffering effect of the thick unsaturated zone, while in coastal areas the influence of low nitrate river recharge is likely to moderate seasonal variations in Nitrate-N concentrations

The magnitude of temporal variations in groundwater quality means it can be difficult to determine a 'representative' measure of water quality at any given point in time as it may be significantly influenced by the variable recharge input occurring from land surface recharge over preceding months and/or years.





**Figure 5. Range of Nitrate-N concentrations observed in wells sampled more than 6 times in the Central Plains area.**

### 2.2.3. Unsaturated zone thickness

Contaminants are transported from the land surface to underlying aquifers through the unsaturated zone. The rate at which mobile contaminants (such as nitrate) infiltrate to groundwater is dependent on the thickness and hydraulic properties of the unsaturated zone. In areas with a thick unsaturated zone or where the unsaturated zone materials have a high effective porosity and/or low permeability it may take considerable time for recharge from the land surface to reach groundwater. Conversely, in areas with a shallow water table or where the unsaturated zone materials have a low effective porosity and/or are highly permeable, recharge from the land surface may reach underlying groundwater rapidly.

Recent research in the UK (e.g. Wang, *et al.*, 2012) suggests that in many aquifer systems there is an appreciable lag on the scale of years to decades between nutrient loss from the root zone and the arrival of peak nitrate concentrations in underlying groundwater. Current work in the Southland



Region (Scott Wilson, Lincoln Agritech, *pers comm.*) has suggested unsaturated zone lag times of the order of years to decades under New Zealand conditions.

This issue was addressed by Close (2010) who reviewed available information from the Canterbury Region. Conclusions of this study include:

- Rapid transmission of recharge may occur through the unsaturated zone under saturated conditions (e.g. following significant recharge events or associated with flood irrigation);
- A number of wells in locations with varying unsaturated zone thickness showed a relatively rapid response in groundwater nitrate concentrations following recharge events. However, it was not clear if the observed response reflected rapid transmission of land surface recharge to the water table (i.e. saturated or bypass flow) or reflected displacement of water from the base of the vadose zone (i.e. piston flow);
- Calculations based on mean annual recharge and profile available water for selected locations in Canterbury indicated a wide range of vadose zone travel times. For example, at West Melton calculated travel time was 6.6 years with a range between 0.3 to 27.8 years;

**Figure 6** shows the spatial distribution of mean depth to groundwater based on water level monitoring information recorded by Environment Canterbury. The figure shows depth to groundwater is generally less than 20 metres from the mid-plains to the coast (where groundwater levels become artesian). However, in inland areas, particularly in the vicinity of Darfield and Te Parita, depths to groundwater may reach 100 metres. Given a median (dryland) land surface recharge of between 200 and 300 millimetres over much of the Central Plains area, the potential exists for significant lag occur between nutrient loss from the soil zone and arrival at the water table, particularly in inland areas where the water table is in excess of 50 metres below ground.

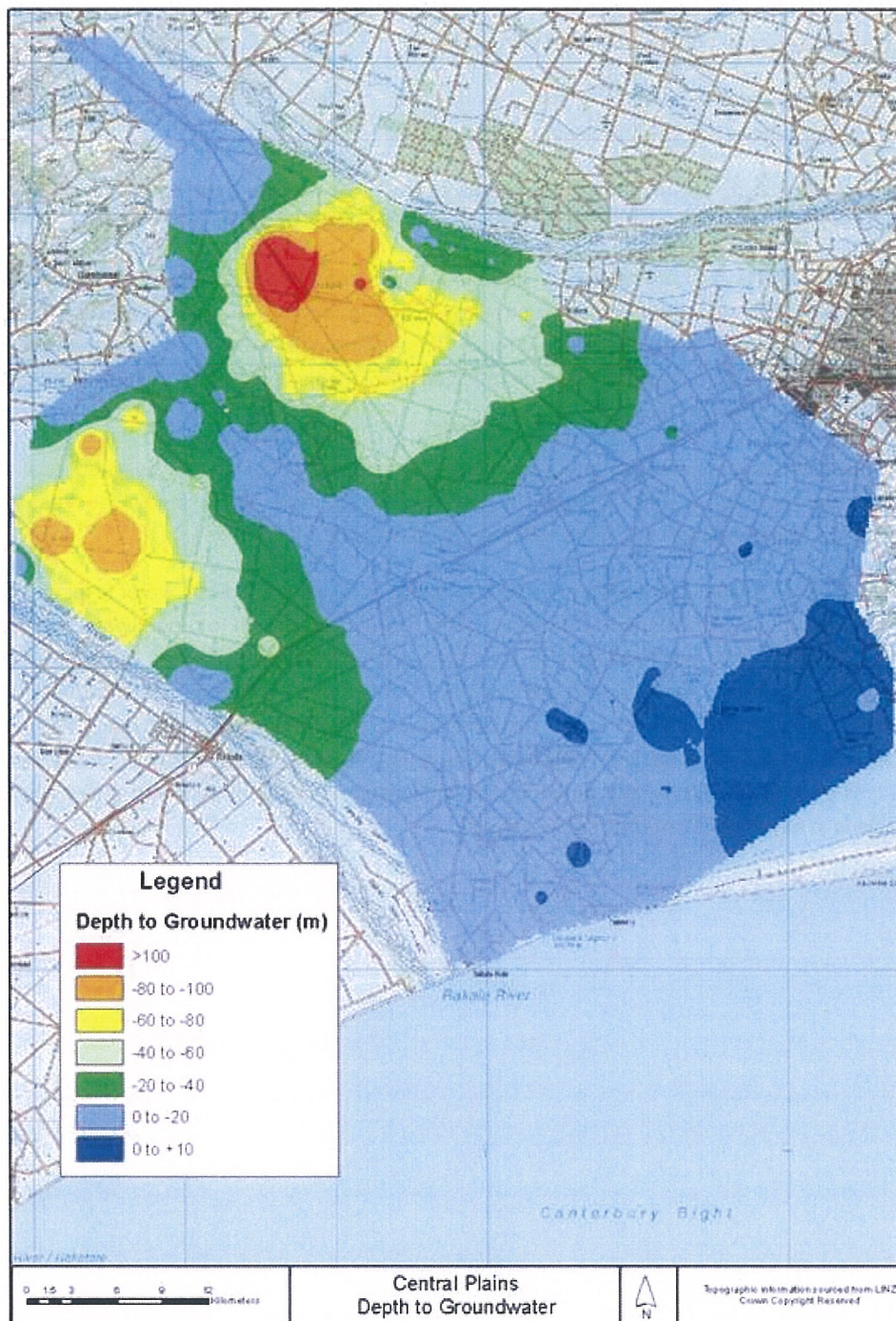


Figure 6. Mean depth to groundwater in the Central Plain area

#### 2.2.4. Water-rock interaction

As groundwater flows through an aquifer system a variety of hydrogeochemical processes may alter its composition. Interaction with geological materials forming the host aquifer (termed *water-rock interaction*) commonly influences natural groundwater chemistry. In general, surface water and recently recharged groundwater tend to be dominated by calcium (Ca) and bicarbonate ( $\text{HCO}_3^-$ ) ions due to the dissolution of carbonate minerals and the production of carbon dioxide by soil microbes which lowers the pH of water infiltrating through the soil zone. However, with increasing age (referred to in terms of *residence time*), groundwater commonly becomes more enriched with parameters such as sodium (Na) and chloride (Cl) as a result of the progressive dissolution of minerals from the aquifer media.

As groundwater moves through an aquifer system it may undergo many other chemical reactions which involve changes in chemical state and facilitate the gradual evolution of water quality along a particular groundwater flow path. These reactions include:

- Acid-base reactions;
- Precipitation and dissolution of minerals;
- Sorption and ion exchange;
- Biodegradation

In the Central Plains area, aquifer materials are predominately derived from silicic greywacke parent rock. These materials are relatively resistant to weathering and exert a relatively minor influence on overall chemical composition of groundwater. For example, **Figure 7** shows a piper plot of major ion chemistry from approximately 500 wells in the Central Plains area. The data show a relatively tight clustering of water chemistry indicating limited geochemical evolution of groundwater within the aquifer system. A majority of wells exhibit bicarbonate-type groundwaters with the Sodium and Potassium the major cations, trending toward Calcium and Magnesium in deeper wells.

The observation of limited geochemical evolution of groundwater in the Central Plains area is consistent with investigations reported by Hanson and Abraham (2009). This study identified relatively minor increases in the relative concentration of major ion species along a flow path from the Waimakariri River near Courtenay to the coast near Lake Ellesmere.

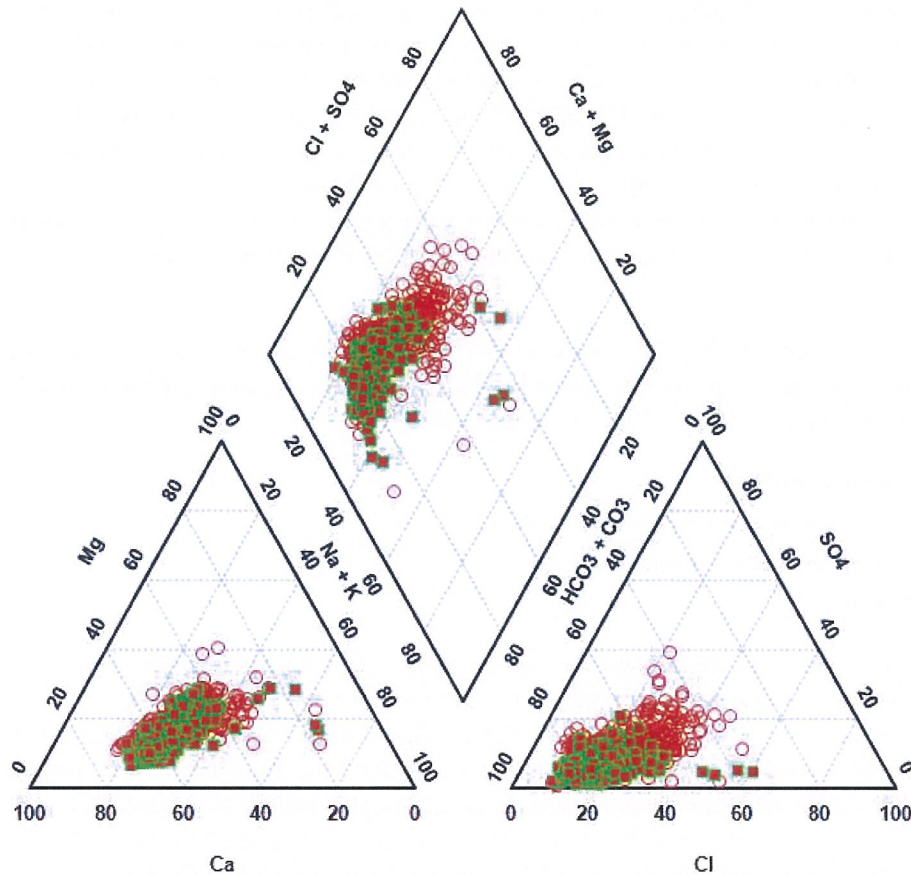
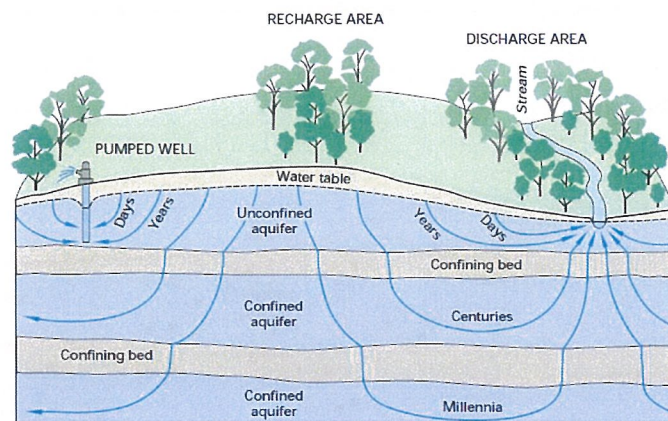


Figure 7. Piper diagram of groundwater quality in the Central Plains area. (Note: open circles include samples from wells <50 metres and filled squares wells >50 metres deep)

### 2.2.5. Groundwater residence time

Under natural conditions groundwater moves in three dimensions through an aquifer system from areas of recharge to points of discharge. The route taken by groundwater flowing through any particular part of an aquifer system is termed the groundwater flow path and may vary in extent from tens of metres to many kilometres in the horizontal direction (following the natural piezometric gradient) and up to the total aquifer thickness in the vertical dimension. **Figure 8** shows a schematic representation of the length and travel time of various flow paths through an aquifer system.



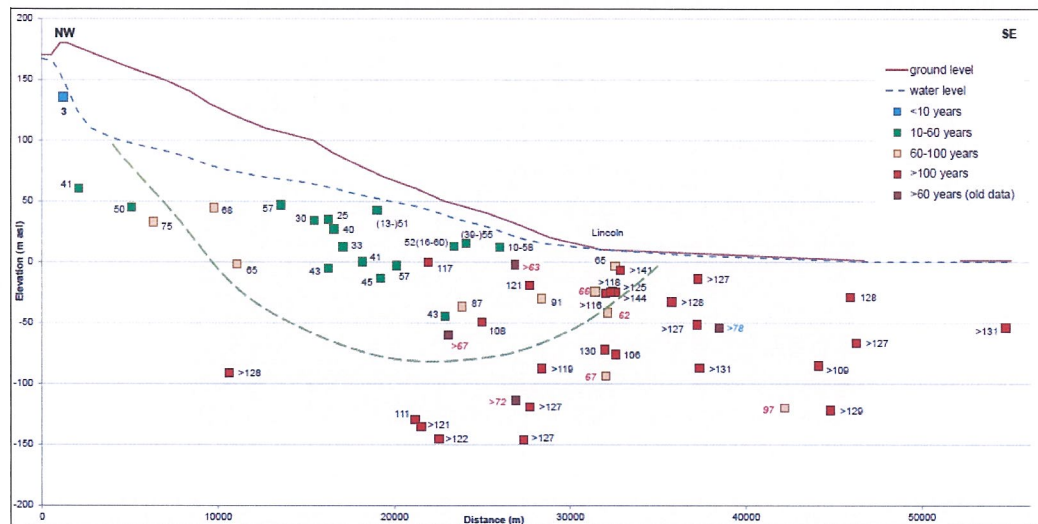


**Figure 8. Schematic representation of the length of groundwater flow paths and groundwater residence time in an aquifer system (USGS, 1998)**

The age (or residence time) of groundwater in an aquifer system progressively increases along a groundwater flow path from the point of recharge to the point of discharge. The increase in residence time provides greater opportunity for the water-rock and geochemical processes described in preceding sections to occur.

In many aquifer systems groundwater flow in the horizontal direction predominates over flow in the vertical direction. This means groundwater commonly flows more rapidly through the upper parts of an aquifer system with comparatively slow rates of groundwater circulation through deeper levels of the aquifer. The resulting gradation in groundwater age along deeper flow paths is often reflected in progressive changes in groundwater quality with depth. As a result, development of a conceptual level understanding of the three dimensional nature and rate of groundwater flow through an aquifer system is particularly important to developing linkages between observed changes in groundwater quality and potential causal mechanisms (such as land use activities).

Van der Raaij (2012) reported results of an investigation to determine groundwater age along a transect from Courtenay to Lake Ellesmere (similar to that utilised by Hansen and Abraham, 2009) using a range of age-tracers (Tritium, CFC-11, CFC-12, SF<sub>6</sub> and <sup>14</sup>C). Results indicated significant variation in groundwater ages from a few years to in excess of 3,000 years. Groundwater age was observed to increase both with depth and distance along the transect. Deeper groundwater (i.e. >150 metres) was observed to be unequivocally old with ages of at least 100 years based on Tritium sampling. Shallow wells, predominantly recharged from the land surface, exhibited younger groundwater ages but showed considerable variation in the age to depth relationship. Several mechanisms were inferred to account for the variability in estimated age (i.e. residence time) in shallow groundwater including the heterogeneity of the aquifer materials, variable depth of the unsaturated zone and variable contribution of rainfall and irrigation to land surface recharge.



**Figure 9. Calculated groundwater tritium ages along a transect between Courtenay and Lake Ellesmere. The green line encloses wells with groundwater predominantly from land surface recharge (after Hanson and Abraham, 2009). Vertical scale exaggerated. (Reproduced from van der Raaij, 2012).**

Estimated groundwater age (or residence time) represents a complex mix of water of varying ages within the capture zone of an individual well. For example, younger water flowing through higher permeability zones may mix with older water contained in lower permeability sediments to yield a composite age. However, based on the available data a majority of wells in the Central Plains area appear to contain water which has been resident in the aquifer for years or decades.

The age dating data suggest that, particularly in deeper groundwater, there is likely to be an appreciable lag between activities on the land surface which contribute dissolved ions to recharge and changes in groundwater quality observed in down gradient wells. However, although typically following a general trend (i.e. older water in deeper groundwater or with increasing distance from the recharge source), the data also indicate considerable spatial and depth variation in residence time between individual wells. As a consequence, it is often difficult to reconcile observed variations in groundwater quality with specific changes in land use, particularly when the composite residence time of the water is taken into account<sup>2</sup>.

## 2.2.6. Redox processes

Oxidation-reduction reactions (referred to as *redox* reactions) are an important process influencing groundwater quality in most aquifer systems. These reactions essentially involve the transfer of electrons between ions in solution and are facilitated by bacteria in the aquifer system which utilise

<sup>2</sup> For example, samples collected from wells around 240 metres deep to the west of Darfield exhibit elevated nitrate concentrations (6 to 9 mg/L) but have calculated groundwater ages in excess of 100 years (Hanson, 2012)

this process as an energy source (McMahon and Chapelle, 2008). Bacterially mediated redox reactions involve sequential reactions with oxygen, nitrate, manganese, iron, sulphate and finally carbon dioxide, and result in the conversion of these ions to a reduced state. The resulting reduced forms of these ions include ammonia ( $\text{NH}_4^+$ ), dissolved manganese ( $\text{Mn}^{2+}$ ), ferrous iron ( $\text{Fe}^{2+}$ ), hydrogen sulphide ( $\text{H}_2\text{S}$ ) and methane ( $\text{CH}_4$ ). Due to the sequential nature of redox processes it is not uncommon to observe progressive declines in dissolved oxygen and Nitrate-N concentrations followed by the appearance of dissolved manganese ( $\text{Mn}^{2+}$ ) and ferrous iron ( $\text{Fe}^{2+}$ ) and finally methane ( $\text{CH}_4$ ) along a groundwater flow path.

Redox status of groundwater in the Central Plains was assessed using the methodology developed by Jurgens *et al.*, (2009) for wells having suitable water quality parameters (including dissolved oxygen) available. As illustrated in **Figure 10** below, this assessment showed that 195 of the 280 wells analysed (70%) were classified as oxic (i.e. having  $\text{O}_2$  reduction as the dominant redox process) while just 4% were classified as anoxic (where  $\text{NO}_3^-$ ,  $\text{Mn(IV)}$  or  $\text{Fe(III)}$  reduction is the dominant redox processes). The limited number of wells exhibiting anoxic groundwaters means that denitrification is only likely to exert a significant influence of groundwater Nitrate-N concentrations in a relatively restricted area along the northern margin of Lake Ellesmere in the vicinity of Motukarara.

However, a significant number of wells (~25%) exhibit groundwaters classified as having a mixed redox state. In a majority of these wells this condition is characterised by dissolved iron concentrations in excess of 0.1 mg/L and Nitrate-N concentrations greater than 0.5 mg/L. Given the sequential nature of redox processes, the significant number of wells exhibiting mixed redox state groundwaters suggests a contribution of groundwater to these wells from two different sources. These sources are interpreted to be nitrate-rich oxic groundwaters derived from land surface recharge moving through more permeable zones in the alluvial aquifer materials and older, more reduced groundwaters contained in lower permeability sediments containing organic carbon. This mixing of groundwaters from different sources may be increased due to vertical gradients induced by groundwater abstraction (including in the vicinity of the actual wells sampled).

As shown on **Figure 10**, no clear relationship is observed between redox state and well depth. Again this is interpreted to reflect a combination of the complex nature of groundwater flow and variable occurrence of organic carbon within the heterogeneous alluvial sediments. Comparison of the statistical distribution of Nitrate-N concentrations in oxic and mixed state groundwaters in **Figure 11** shows that while median concentrations are relatively similar in both sample populations (3.8 mg/L in oxic groundwater, 3.5 mg/L in mixed state groundwater), a significantly greater number of wells classified as oxic show elevated Nitrate-N concentrations. For example, while only 3 percent of wells classified as mixed-state exhibit Nitrate-N concentrations greater than 8.6 mg/L (i.e. 75 % of MAV), 13 percent of oxic wells exhibit concentrations in excess of this figure. This observation suggests that, even in the absence of a significant change in concentrations in shallow groundwaters, Nitrate-N concentrations in many wells exhibiting a mixed redox state may increase over time as older, reduced groundwaters are progressively replenished by younger oxic groundwaters.



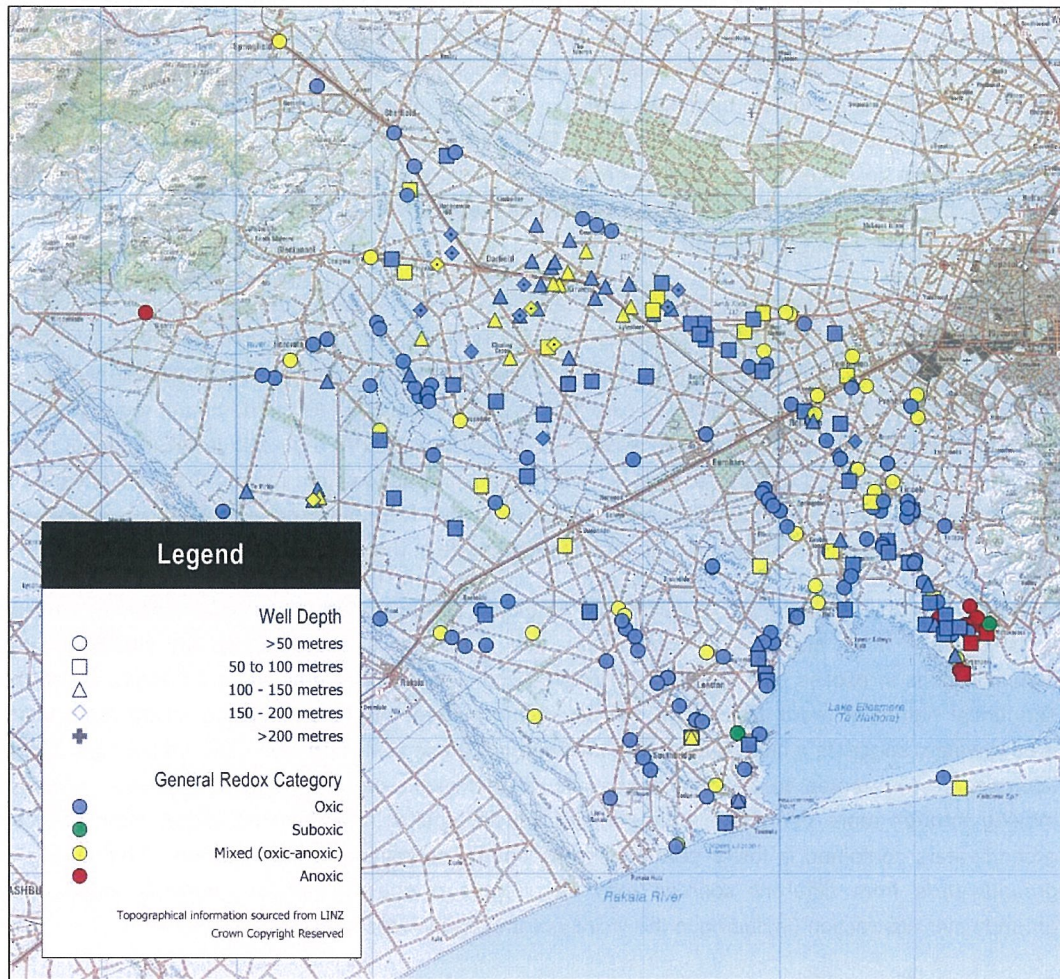
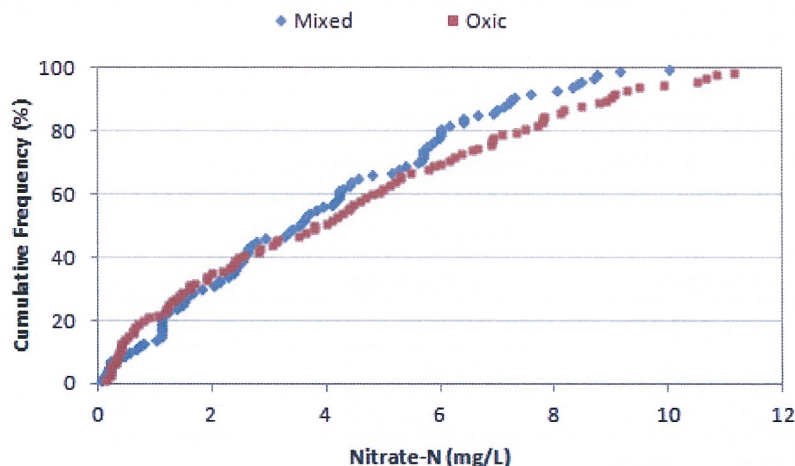


Figure 10. Redox status of groundwater in the Central Plains area assessed using the methodology developed by Jurgens et al. 2009.





**Figure 11.** Cumulative frequency of nitrate concentrations in wells exhibiting oxic and mixed (oxic-anoxic) redox states

#### 2.2.7. Land Use

As previously noted, land use and associated land management practices are the major factor influencing groundwater quality in most aquifer systems, particularly where recharge flux is dominated by land surface recharge (as opposed river seepage losses). Soluble ions and particulates (such as microbial contaminants) can be mobilised by infiltrating soil moisture and transported to the underlying water table. Contaminants may also be introduced into the unsaturated zone via specific discharges below the root zone (such as septic tanks).

The rate of Nitrate-N leaching loss varies significantly between land use type. For example, **Table 1** provides a summary of estimated Nitrate-N losses under different land uses in the Canterbury region. These figures show significant variation in the rate of loss between various land uses which can be further enhanced by factors such as land management practice and climate.

**Table 2** provides a summary of current (2011) land use across the Central Plains area. This report does not attempt to quantify historical or potential future land use over this area other than to observe that land use is dynamic and has changed significantly over time and is likely to continue to do so into the future. Such changes occur both at a paddock (e.g. crop to pasture rotation) and farm (e.g. transition from dryland to irrigated agriculture) scale. As a consequence, nutrient losses may vary significantly both spatially and over time.