

**REPORT ON ENVIRONMENTAL EFFECTS OF CENTRAL PLAINS
WATER IRRIGATION SCHEME**

CONTAMINANT LOADING AND REDUCTION ESTIMATES

by

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for

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NITRATES

Estimation of Nitrate Reduction and Dilution in the Vadose Zone

The vadose zones in the vicinity of the Central Plain Water Irrigation Scheme (CPWIS), as with the majority of the Canterbury Plains, consist mainly of indurated sandstone/mudstone gravels with varying proportions of sand, silt, and clay (NCCB and RWB., 1983). Though there has been little research on the transport and fate of solutes in coarse gravel vadose zones, these environments are generally well oxygenated and are likely to contain few nitrogen reducing bacteria due to low carbon concentrations, especially in the Central Plains area away from the coast. As the rate of heterotrophic denitrification (main form of denitrification) is mainly controlled by the abundance of denitrifying bacteria, quantities of organic carbon and oxygen contents of environments (Holden and Fierer, 2005), the nitrate reducing capacity in the CPWIS vadose zone type of environment is regarded as being minimal (Close et al., 2001).

Denitrification in vadose zones has been reported (Holden and Fierer, 2005), though reports commonly show denitrification has tended to decrease rapidly with depth e.g., Aritola (1997) reported a decrease by a factor of 10,000 for a depth interval of 0-14m beneath fertilised soil, though this decreased to a factor of 100 below sewage sludge-amended soil. Onsoy et al. (2005) investigated nitrate transport and fate in a deep alluvial sand vadose zone in California. They indicate that denitrification may occur at local sites throughout the vadose zone but it is not likely to be a major process and could not account for low nitrate concentrations deep in the vadose zone. Low nitrate concentrations were thought to be mainly due to rapid transport and dilution of a small portion of the overall leached nitrates through preferential pathways to groundwater.

Some preliminary results (unpublished ESR) from a nitrate leaching experimental site close to the scheme area at Lincoln show concentrations of nitrous oxide gas (indicating nitrate reduction) in a coarse gravel vadose zone, similar to that within the scheme area. However, recent findings indicate that these gases were being formed in the lower part of the soil zone (with higher organic carbon concentrations and

presumably microbial activity) and then most likely transported through the vadose zone with pore water movement. It does not appear that any nitrate is actually being reduced to nitrogen gases within the sandy gravel vadose zone itself.

Observations from the Lincoln leaching experiment, which was drip-irrigated, suggest that a proportion of leachate is transported rapidly through preferential pathways to depth with little to no dilution, whilst the bulk of the leachate moves through the matrix at a slower rate, also with little dilution. This process is most likely dependant on the amount of water leaching through the profile.

At present the authors are not aware of any validated empirical evidence of denitrification occurring in gravel vadose zones such as those found in the CPWIS, therefore nitrate reduction capacity of the vadose zone is assumed to be zero. As mentioned previously, leachates with temporarily elevated nitrate concentrations can be diluted to lower concentrations as they move through the vadose zone by mixing with more dilute in-situ vadose zone pore waters (Onsoy et al., 2005).

Estimation of Nitrate Reduction and Dilution Capacity of the Aquifers and Aquitards.

Pang et al. (2006), studied the impacts of septic system discharge on nitrate and microbial quality of groundwater systems in the Canterbury Plains (similar aquifer system to CPWIS). They performed a sensitivity analysis that showed that nitrate reduction in gravel aquifers was most sensitive to hydraulic conductivity, which affects the observed concentrations by increased groundwater flux and dilution of the recharge. Six parameters were selected for sensitivity analysis — hydraulic conductivity, adsorption coefficient for NH_4^+ , rate coefficient for the transformation of NH_4^+ to NO_3^- , longitudinal dispersivity, NH_4^+ input concentration and discharge rate of the effluent. A cumulative effect of nitrate was shown for a series of septic tanks down a flowpath (Figure 1). High flux environments tended to dilute nitrate concentrations more effectively than low hydraulic conductivity environments, where nitrate levels remained higher for longer periods and were more likely to accumulate to higher levels with further downstream impacts. Accordingly the dilution capacity of aquifers (with relatively high hydraulic conductivities) will be greater than the aquitards where concentrations of nitrate are likely to remain relatively stable. It

should be noted that the capacity of groundwater to dilute nitrate is limited, and a very large distance is needed for elevated levels of nitrate to reduce to the background level. While dilution is a factor in lowering nitrate concentrations in aquifers, this is only when the diluting water has low nitrate concentrations. Where up-gradient groundwater also has high nitrate concentrations the dilution effect will be minimal. Transference of these study results is appropriate for situations of point sources with low nitrate concentrations in up-gradient water, such as caused by an increased number of septic tanks situated on a clean groundwater body.

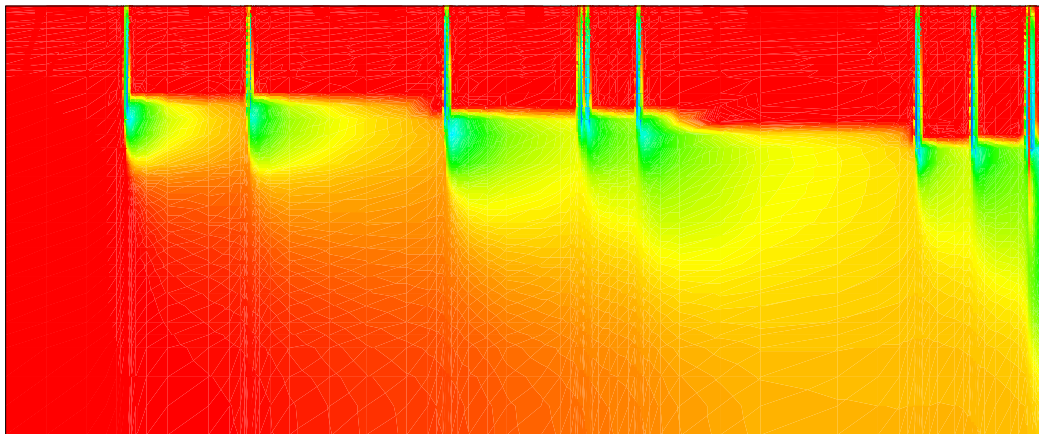


Figure 1. Predictive model showing cumulative impact of septic tanks on nitrate concentrations in groundwater (Pang *et al*, 2006).

Nitrate concentrations have been shown to decrease with depth in aquifers (Freeze and Cherry, 1979). This is often due to reducing or anaerobic conditions at depth, reducing nitrates to ammonia or nitrogen gases. Reducing conditions are generally confined to the coastal areas of Canterbury, therefore reduction of nitrates at depth due to reducing conditions is unlikely for most of the Canterbury Plains, including the CPWIS. In Canterbury, low nitrate concentrations in groundwater at depth are mainly a reflection of low nitrogen content water in recharge from the foothills and major rivers.

Scott and Thorpe (1986) in their review of groundwater resources between the Rakaia and Ashburton rivers noted that deeper water, which was older and had a larger component of river water, had lower nitrate and chloride concentrations, though higher bicarbonate concentrations than water near the water table. In Environment

Canterbury's review of nitrates in groundwater (ECAN, 2002) no correlation was found between depth and nitrate concentration, though wells deeper than 150 metres had concentrations consistently less than 5 mg/L.

The nitrate reduction capacity of aquifers and aquitards in the vicinity of the CPWIS with respect to denitrification is likely to be minimal, with the main process in lowering nitrate concentrations being mixing and dilution. Greater dilution will occur in higher conductivity aquifers where up-gradient (mixing waters have low nitrate concentrations).

MICROBES

Microbial loading for dairy land use

The loading of microbes to the soil surface from farming related practices is related to a number of factors including: animal type, stocking rates, concentrations of microbes in animal faeces, and area of land farmed. A number of studies have been carried out which include assessing loading of microbes in New Zealand environments (Tian et al., 2002; Wilcock et al., 1999). Below is an assessment of potential loading rates using some average figures, assuming that all land proposed to be in pasture will be used for dairying (worst case scenario).

Several studies have provided estimates of the number of times cattle excrete in a 24-hr period. These estimates vary from 9 ± 2.5 times to 16.1 times on average in a 24 hr period (Table 1). The average of the studies detailed in Table 1 gives a value of 12.5 defecations every 24 hrs.

The wet weight of a single defecation has also been estimated to weight in the range from 1.48 kg to 2.7 kg (Haynes & Williams 1993), resulting in an average of 2.02 kg. Combining the average number of defecations in a 24 hr period (12.5) and the average weight of each defecation (2.02 kg), the daily faecal output by cattle can be estimated at 25 kg.

Table 1 The mean number of defecations by cattle in a 24 hr period

| Stock Type | Mean number of defecations per day | Reference |
|----------------|------------------------------------|--|
| Dairy Cow | 11.6 | Castle <i>et al.</i> , (1950) ¹ |
| Dairy Cattle | 12.2 | Hancock (1950) ¹ |
| Dairy Cattle | 12 | Goodall (1951) ¹ |
| Dairy Cattle | 15.4 | Hardison <i>et al.</i> , (1954) ¹ |
| Dairy Cattle | 12 | Petersen <i>et al.</i> , (1956) ¹ |
| Dairy Cattle | 11.6 | McLusky (1960) ¹ |
| Dairy Cattle | 12 | Davies <i>et al.</i> , (1962) ¹ |
| Dairy Cattle | 16.1 | Wardrop (1963) ¹ |
| Beef Steer | 10.5 | Weeda (1967) ¹ |
| Dairy Cattle | 11 | Frame (1971) ¹ |
| Dairy Cattle | 13.9 | MacDiarmid & Watkins (1972) ¹ |
| Dairy Cattle | 15 | Hafez and Bouisson, (1975) |
| Dairy Cattle | 9 | Sahara <i>et al.</i> (1990) |
| Average | 12.5 | |

Note:

1. References from Castle *et al.* (1950) to MacDiarmid & Watkins (1972) are listed in Haynes and Williams (1993).

As *E. coli* is the faecal indicator measured for Drinking Water Standards we make an assessment of its loading to the area. The average number of cows per hectare on irrigated pastures is approximately 2.9 cows/ha (MAF, 2000; Wilcock *et al.*, 1999), average cow produces 12.5 pats per day (Table 1) with each containing 1.3E09 *E. coli* (Wilcock *et al.*, 1999).

Table 2. Estimation of microbial loading depending on stocking rates.

| Stocking Rate (Cows/ha) | Pats/day | <i>E. coli</i> /pat | Load <i>E. coli</i> /m ² | Load <i>E. coli</i> /ha |
|-------------------------|----------|---------------------|-------------------------------------|-------------------------|
| 2.5 | 12.5 | 1.3E09 | 4.60E06 | 4.60E10 |
| 2.9 | 12.5 | 1.3E09 | 4.71E06 | 4.71E10 |
| 3.5 | 12.5 | 1.3E09 | 5.69E06 | 5.69E10 |
| 4.0 | 12.5 | 1.3E09 | 6.50E06 | 6.50E10 |

The *E. coli* loading /ha/day $\sim 12.5 \times 2.9 \times 1.3E09 = 4.71E10/\text{ha} = 4.71E06/\text{m}^2/\text{day}$ and $1.68E09/\text{m}^2/\text{year}$ (see Table 2 for variations due to different stocking rates). This is extremely conservative as it assumes no surface die-off, and all bugs deposited infiltrate with leached water. Die-off in cowpats and on the land surface will be variable depending on a number of environmental conditions including temperature,

water content, predators, and UV light exposure. Tian et al. (2002) derived a die-off rate for their Hamilton study of 0.025-0.098, using data from Howell et al. (1996) who demonstrated faecal coliform die-off at between 0.04-0.084. Bolton et al. (1999) reported a 5-log reduction of *E. coli* O157:H7 in 43-50 days in samples decanted onto grassland (approximately 0.1 reduction/day). Using a conservative mean value (0.062) from Howell et al. (1996), we derive a curve for concentrations of *E. coli* over time as shown in Figure 2.

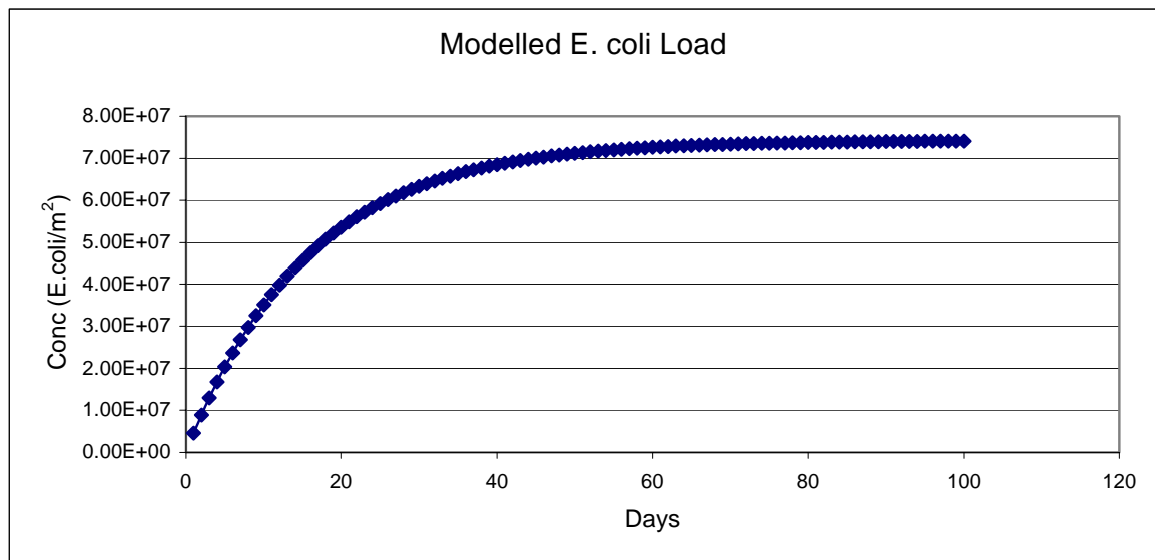


Figure 2. Modelled *E. coli* concentrations over time taking into account daily additions (10^6) and die-off (0.062/day).

The curve in Figure 2 indicated concentrations level out at approximately $7.4E07/m^2$. Assuming a very large drainage event (90 mm) and all surviving faecal coliforms being leached this converts to a leachate concentration of approximately $8.2E04$ cfu/100ml. Averaged over a year of drainage (assuming 600 mm) the derived faecal coliform concentration in drainage water is $1.2E04$ cfu/100ml. However these concentrations are unlikely to occur as a high percentage of microbes are likely to remain within the cowpats at the surface during each leaching event. The amount released into infiltrating water is dependant on the energy and volume of the rainfall/irrigation, the age of the cowpat and the soil type. It must be noted that these numbers are estimates as relatively little is known about the survival and release to

infiltrating water of microbes in New Zealand environments. ESR is currently undertaking research on these aspects of microbial retention, release and die-off.

The New Zealand drinking water guideline is <1 cfu/100ml. As such in the worst case-scenario where all viable bacteria are leached from the surface a 5-log reduction is required from the cowpat, to the infiltrating water and through the soil and vadose zone, to reduce the concentrations to below drinking water guidelines.

Microbial reduction capacity of soils

The microbial reduction capability of soils in the CPWIS is mainly dependant on the flow regime applied to them. The longer the bacteria remain within the soil profile the more likelihood of die-off occurring. Therefore, irrigation regimes with higher application rates result in shorter time of transfer and subsequent increased likelihood of contamination of groundwater. In general bacteria usually decline to negligible numbers in soils after 2-3 months with indicator bacteria such as *E. coli* persisting for longer (Gerba and Bitton, 1984). Factors affecting the survival time of bacteria in soils are outlined in Table 3.

Table 3. Factors Affecting Survival of Enteric Bacteria in Soil (modified from Gerba and Bitton (1984)).

| Factor | Comments |
|---------------------------|---|
| Moisture Content | Greater survival time in moist soils and during times of higher rainfall/irrigation |
| Moisture Holding Capacity | Survival time is less in lower water holding capacity soils |
| Temperature | Longer survival at lower temperatures i.e. longer survival in winter as compared to summer |
| pH | Shorter survival in acid (pH 3-5) than in alkaline soils |
| Sunlight | Shorter survival time in sunlight, i.e. at soil surface |
| Organic Matter | Increased survival and possible regrowth when sufficient amounts of organic matter are present. |

Under spray irrigation leaching of microbes through soils is generally limited, due mainly to filtration of bacteria as water percolates through finer soil matrix material.

Results from experiments on a typical dairy farm at Te Pirita within the CPWIS (Close et al., 2005a) indicate that counts of *E. coli* are reduced to a mean of <0.5 – 0.7 MPN/100ml (MPN = most probable number) at 1.2 m depth under a regular regime of 50 mm of spray irrigation using a Briggs Rotarainer system (Table 4). The soil type at Te Pirita classified as a stony Lismore, which is silt loam with a depth of approximately 20-30cm underlain by coarse sandy gravel.

Table 4. Summary of preliminary results from spray irrigation near Te Pirita. Note that 200 ml samples were analysed for *E. coli*

| Irrigation & Rainfall (mm) | Number of events | Mean <i>E. coli</i> (MPN/100 ml) | Max <i>E. coli</i> (MPN/100 ml) |
|----------------------------|------------------|----------------------------------|---------------------------------|
| ~ 50 | 16 | < 0.5 – 0.7 | < 0.5 – 7 |
| ~ 80 | 4 | 36 – 275 | 200 – > 2000 |

Dairy Shed Effluent

Another study currently being carried out by ESR/Lincoln University shows an approximate 4-log reduction in faecal coliform concentrations through a Templeton Silt Loam (depth of 700 mm), under spray irrigation (50 mm at 25 mm/hr) after application of animal effluent (pers comm. ESR).

In New Zealand, scientists from Landcare Research have published a number of papers on movement of bacteria and viruses through a variety of soils (Aislabie et al., 2001; McLeod et al., 2003; McLeod et al., 2001). These experiments have been run under dairy shed effluent application conditions (25 mm at 50 mm/hr), with continuous spray irrigation rates of 5-10 mm/hr until at least 1 pore volume of leachate has been collected (Aislabie et al., 2001). In these experiments differential microbial movement was attributed to differences in soil structure, ie. where bypass flow occurred bacteria were leached. Poorly drained soils tended to have coarse subsoil structures with macropores which enabled bypass flow, whilst well drained soils had finer more uniformly porous structure that minimised bypass flow and allowed matrix flow, which increased filtration and decreased leaching of bacteria (Aislabie et al., 2001). Simulation of these Landcare Research lysimeter leaching data

(ESR unpublished data) indicates that log reduction per m varies in the soils tested, from mean values of 1.98 – 2.98 log reduction/m (Table 5).

Table 5. Modelled microbial log-reduction results for a variety of soils with applied dairy shed effluent followed by irrigation water. Experimental data from Landcare Research.

| Soil name | Texture | Column length (m) | log reduction/m | Mean |
|------------------|-----------------|--------------------------|------------------------|-------------|
| Waikiwi | Silt loam | 0.5 | 2.24 | |
| Waikiwi | Silt loam | 0.5 | 2.22 | |
| Waikiwi | Silt loam | 0.5 | 1.94 | 2.13 |
| Waikiwi | Silt loam | 0.5 | 2.21 | |
| Waikiwi | Silt loam | 0.5 | 2.14 | |
| Waikiwi | Silt loam | 0.5 | 2.62 | 2.32 |
| Waikoikoi | Silt loam | 0.5 | 2.23 | |
| Waikoikoi | Silt loam | 0.5 | 2.69 | |
| Waikoikoi | Silt loam | 0.5 | 2.09 | 2.34 |
| Waitarere | Sandy | 0.7 | 2.41 | |
| Waitarere | Sandy | 0.7 | 2.08 | |
| Waitarere | Sandy | 0.7 | 2.89 | 2.46 |
| Waitarere | Sandy | 0.7 | 2.29 | |
| Waitarere | Sandy | 0.7 | 1.96 | |
| Waitarere | Sandy | 0.7 | 2.77 | 2.34 |
| Lismore | Silt loam | 0.7 | 0.99 | |
| Lismore | Silt loam | 0.7 | 2.43 | |
| Lismore | Silt loam | 0.7 | 2.53 | 1.98 |
| Manawatu | Fine sandy loam | 0.7 | 3.26 | |
| Manawatu | Fine sandy loam | 0.7 | 2.40 | |
| Manawatu | Fine sandy loam | 0.7 | 3.28 | 2.98 |
| Templeton | Silt loam | 0.4 | 2.56 | |
| Templeton | Silt loam | 0.4 | 1.85 | |
| Templeton | Silt loam | 0.4 | 1.56 | 1.99 |

These reduction values are lower than those derived from the ESR/Lincoln University column leaching studies of the Templeton silt loam. This is probably because of the increased and continuous irrigation applied in the Landcare Research experiments, which would enable bypass flow. In the Templeton study, bypass flow is likely to be minimised under bypass flow. Soils of the CPWIS are generally well drained and therefore under efficient spray irrigation there is likely to be minimal bypass flow occurring as found in a study by Aislabie et al., (2001) on microbial leaching for well-drained soils.

Impact of Irrigation Method

The type of irrigation method is an important parameter as other studies done in stony silt loam soils in southern Canterbury. Pang et al. (2005) indicate that leaching of microbes through soils and vadose zones occurs under border strip irrigation conditions, with large numbers of bacteria (up to >2400 cpu/100ml) recorded in down-gradient wells. The pathogenic bacteria *Campylobacter* was also often found in shallow groundwater under these flood irrigation conditions.

In the CPWIS the irrigation method will be spray and therefore there will be greater attenuation in the soil zone, more like the Templeton silt loam column study (reduction of approximately 4-log/ metre). In addition there is uncertainty mentioned in loading capacity around the percentage of microbes leached from the cowpats into the soil during leaching events. This percentage leached into the soil is dependant on two main factors; 1) age of cowpat, older cowpats are less likely to release microbes than fresh cowpats; and 2) the intensity of the irrigation and rainfall, higher energy events are likely to increase the number of microbes in infiltrating water.

Microbial reduction capacity of vadose zone

Little work to date has been carried out assessing the reduction capacity of coarse sandy gravel vadose zones. However, due to the coarse nature of the vadose zone materials (mainly gravel and sand) the main attenuation for microbes through the vadose zone is likely to result from a combination of both the time of transport in combination with filtration. Pang et al. (2006) noted that microbes were not readily retarded in gravel aquifers and used a sensitivity approach to determine an overall removal rate coefficient (combined die-off and filtration, though mostly filtration over this short time period) of $1.3 \log \text{ day}^{-1}$ for unsaturated gravels, though conditions for this modelling involved a constant supply of effluent solution from a septic tank system, not dairy irrigated pastures as is being considered here. The flow regime would have been semi-saturated from the septic tank effluent pulses. Microbial attenuation under general dairy farming systems are likely to be greater than in the septic tank environment as the flow is unsaturated and the vadose zone sediments will not be severely contaminated. Contaminated sediments tend to reduce attenuation and

filtration as noted by greater transporting distances of microbes in contaminated aquifers (Pang et al. 2005).

In a septic tank study by Sinton (1986), carried out at Burnham, faecal coliforms reduced 2.51-log in 9.5 m thickness of unsaturated zone. This gives a removal rate of 0.265 log/m (equivalent of 3.77 m for a 1-log reduction). This study was undertaken with intermittent near saturated flow through the vadose zone (worst case scenario). Bacteria leaching from the soil zone under dairy land use would require less distance per log-reduction of microbes due to the lower moisture content and associated conductivity.

Laboratory studies recently undertaken by ESR on the survival of different microorganisms in saturated gravel material indicate the following die-off rates:

- Faecal coliforms (including *E. coli*) - factor of ~3 in 3 months
- *Campylobacter* ; - $\sim 10^4$ reduction in 2 weeks

These studies show that die-off of faecal coliforms in vadose zone environments is a relatively slow process (in comparison to some other bacteria such as *Campylobacter*), and filtration through the soil zone generally provides the greatest reduction of bacteria through the profile, depending on flow velocities.

As there is only limited data of this type there is significant uncertainty about the transference of these calculations to a wider area of land under dairy management practices. However, it is likely that removal of bacteria in the CPWIS will be higher in soils and the vadose zone due to the intermittent microbial loading and lower rainfall/irrigation than has been applied in the studies above.

Data from some current dairy farms suggest higher removal rates from cow pat to leached concentrations in the vadose zone. As highlighted earlier, results from experiments at Te Piritā under typical dairy farming conditions (Close et al. 2005a) indicate mean *E. coli* counts are reduced below drinking water guidelines in the vadose zone at 1.2m (<0.5 – 0.7 MPN/100ml) under a regular application of 50 mm with a Briggs Rotarainer irrigation system (Table 4). As the Te Piritā dairy farm is a

typical dairy farm it can be assumed that the microbial loading is in the region of $7.4E07/m^2$. When heavy rainfall occurs in conjunction with irrigation (giving a total input of around 80 mm) some leaching of faecal coliforms was observed.

The groundwater has been monitored at the Lincoln Dairy Farm, which has a centre pivot irrigation system, over the past 4 years (Close et al. 2005b). A total of 457 and 499 groundwater samples have been analysed for faecal coliforms and *Campylobacter*, respectively, over this period. There have been 10 groundwater samples out of a total of 179 (5.6%) from up-gradient wells with detections of faecal coliforms at concentrations of 1 cfu/100 ml or greater, and 6 groundwater samples out of a total of 278 (2.2%) from down-gradient wells with detections of faecal coliforms at concentrations of 1 cfu/100 ml or greater. Most faecal coliforms concentrations have been 1 or 2 cfu/100 ml. There have been three detections of *Campylobacter* over the four-year period. Two were from down-gradient wells (0.7%), and the other sample was from an up-gradient well (0.5%). This indicates that the centre pivot spray irrigation system at the Lincoln Dairy Farm has had little or no impact on the microbial quality of groundwater in the first four years of operation.

In summary there should be little impact from dairying irrigated with spray irrigation on the microbial quality of groundwater. The studies referred to in this section indicate that there is a greater risk of microbial leaching from application of dairy shed effluent and care is required to ensure that application amounts are not excessive and the dairy shed application is well managed. The reductions observed in the soil and vadose zone should ensure that microbial levels are low provided that there is a depth to groundwater > 5 m. For shallower groundwater tables there would probably be an increased risk of contamination for large rainfall events, particularly if combined with irrigation events. There are very few studies on the removal of microbes in the vadose zone and it is difficult to further refine the critical depth for removal of microbes. As stated in the next section, there is significant removal of microbes in the groundwater, which would ensure that any microbial contamination would not be cumulative but would tend to be localised.

Microbial Reduction capacity for aquifers/aquitards

Column and field experiments carried out in alluvial gravel aquifer environments by Pang et al. (2003) and Sinton et al. (2000), indicate retardation factors of *E. coli* and faecal coliforms, less than 1 indicating that bacteria are travelling at a faster rate than the average groundwater velocity, through preferential pathways. A list of microbial travel distances observed in gravel aquifer studies is shown in Table 6.

Table 6 indicates that microbes have been recorded up to 900m down-gradient of the release site in aquifers at Burnham, similar to those within the CPWIS. Pang et al. (2005) reported a gradation of microbial reduction capacity in aquifers depending on the aquifer physical properties and contamination status. The study considers the distance required to achieve a 7-log reduction in microbial concentration (to comply with guideline standards assuming worst case scenario of initial concentration of 10^6 cfu/100 mL). The results of the study indicated that a 7-log reduction would require 125-280 m travel in clean coarse gravel aquifers, 1.7-3.9 km travel in contaminated coarse gravel aquifers, 33-61 m travel in clean sandy fine gravel aquifers, 33-129 m travel in contaminated sandy fine gravel aquifers, and 33-47 m travel in contaminated river and coastal sand aquifers.

This suggests that in generally clean coarse gravel aquifer environments such as those generally in the CPWIS a 7-log reduction will occur in approximately 125-280 m, therefore 1-log reduction will occur in approximately 17-40 m. Should aquifers become contaminated by effluent material over a long period (for example, in the vicinity of a septic tank), this aquifer travel distance may increase for 1-log reduction of bacteria to approximately 240-530 m. This would only be a localised effect.

It is expected that for the aquifer environment in the CPWIS microbial contamination is unlikely to have a cumulative impact and will be localised to contaminant sources. Only low counts are likely to leach through the soil and vadose zone under spray irrigation methods, and microbial contamination of groundwater is likely to be minimal where sufficient depth (>5 m) to groundwater occurs. Where microbial contamination of groundwater does occur it is likely to be reduced below drinking water guideline levels in a relatively short distance.

Areal distribution of microbial leaching under future landuse scenario.

Under the new land use scenarios the greatest concern would be the depth to groundwater, as in some areas the groundwater table is predicted to be at or above the land surface. Where water tables are shallow (<5 m), leaching of viable microbes is likely in large leaching events, but where groundwater tables are deeper (approximately > 5 to 10 m) and spray irrigation occurs with good efficiencies, leaching of microbes to groundwater will be minimal to non-existent.

Figure 3 highlights the minimum depth to water for 80% of the time across the CPWIS (i.e. 20% of the time the water levels are expected to be at these depths or shallower). The areas highlighted in red and yellow are at higher risk from microbial leaching. In general, leaching of microbes to groundwater is most likely to occur in areas of dairying with poor irrigation efficiencies and shallow groundwater tables.

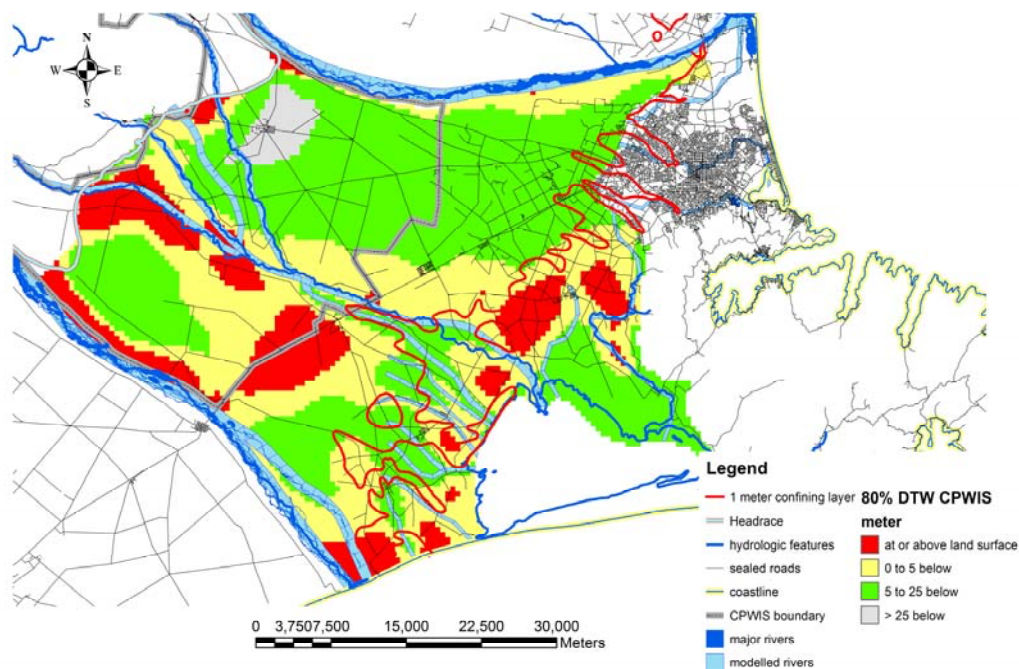


Figure 3. Minimum depth to water for 80% of the time under proposed CPWIS.

Table 6. Transport of micro-organisms in coarse-grained gravel aquifers in the literature.

| Reference | Field site | Microbes | Furthest distance detected (m) | Velocity (m/day) interpreted from c_{\max} |
|------------------------|--|---|--------------------------------|--|
| Noonan & McNabb (1979) | Sewage effluent irrigation site at Burnham, Canterbury, NZ | Coliphage T4, $\Phi X174$ | 920 | 230 |
| Sinton (1980a) | Sewage effluent irrigation site at Burnham, Canterbury, NZ | <i>E. coli</i> (H ₂ S ⁺), <i>B. stearothermophilus</i> | 920 | 56-153 |
| Sinton (1980b) | Sewage effluent irrigation site at Burnham, Canterbury, NZ | <i>E. coli</i> J6-2, PB922 | 570 | 207-240 |
| Pang et al. (1998) | Uncontaminated Burnham, Canterbury, NZ | <i>B. subtilis</i> spores | 86 | 50-85 |
| Sinton & Close (1983) | Sewage effluent irrigation site, Templeton, Canterbury, NZ | <i>E. coli</i> J6-2 | 240 | 156-196 |
| Sinton et al. (1997) | Sewage effluent irrigation site, Templeton, Canterbury, NZ | <i>E. coli</i> , MS2 | 401 | 144-182 |
| Thorpe et al. (1982) | Roys Hill, Heretaunga Plains, NZ | <i>E. coli</i> (H ₂ S ⁺) | 125 | 116-240 |
| Thorpe et al. (1982) | Ngatarawa, Heretaunga Plains, NZ | <i>E. coli</i> (H ₂ S ⁺) | 50 | 12-42 |
| Rossi et al. (1994) | A postglacial sand and gravel aquifer in Switzerland | T-4 coliphages | 163 | 11-205 |
| Rossi et al. (1994) | A postglacial sand and gravel aquifer in Switzerland | Phage f1 | 113 | 11-132 |
| DeBorde et al. (1999) | Sandy gravel aquifer, Erskine, near Missoula, Montana, USA | MS2, PRD1, $\Phi X174$ | 30 | 18-36 |
| DeBorde et al. (1999) | Sandy gravel aquifer, Erskine, near Missoula, Montana, USA | Attenuated polio | 19 | 33-59 |
| Woessner et al. (2001) | Sandy gravel aquifer, Erskine, near Missoula, Montana, USA | MS2, PRD1, $\Phi X174$, attenuated polio | 21.5 | 115-172 |

Pesticide loading for land use scenarios

The fate and transport of a variety of pesticides are modelled for differing land uses. Prior to the modelling an assessment of the pesticides most commonly used for a particular land use were assessed and are presented in Table 7. A number of different land uses were assumed including:

- dairying (irrigated);
- irrigated arable (wheat, peas and potatoes); and

Sources of information included 'Trends in Pesticide Use in New Zealand: 2004'. (Manktelow et al., 2005); NovaChem Manual 2004/05; a review of nitrates and pesticides in NZ groundwater systems (Close et al., 2001); and personal communications with various experts in the field. These sources indicated a variety of herbicides, fungicides and insecticide use for most land uses.

Pesticide reduction capacity of the soil layer

The main factors influencing the leaching of a chemical is its mobility, as indicated by the organic carbon partition coefficient (K_{oc}) and its persistence, as shown by its half-life ($T_{1/2}$). The agro-chemicals applied vary widely in these properties and hence in their leaching potential. Agro-chemicals with high K_{oc} values (>100) are generally sorbed within the soil layer, whilst agro-chemicals with low K_{oc} values may be more easily leached with drainage water. The organic carbon content of the soil plays a considerable role in the attenuation of agro-chemicals through the soil zone, with soils and layers with higher organic carbon contents being more capable of attenuating leached chemicals. Barbash and Resek (1996) indicate that adsorption dominates attenuation processes when organic fraction of the soil exceeds 0.1%.

The half-life of an agro-chemical affects its ability to persist in the soil and environment. Chemicals with long half-lives are able to remain within the soil and environment, in their original structure, for considerable periods of time. Chemicals with short half-lives are more prone to breaking down, and do not persist in the environment. Table 7 indicates the K_{oc} and $T_{1/2}$ of agro-chemical applied in the management of the land uses modelled. Those chemicals with very high K_{oc} values or

Table 7. Agro-chemicals used in a variety of different land uses. Note: chemicals in rows highlighted have been modelled.

| Land use = dairy | | | | | | | | |
|----------------------------|------------------------|-------------|-------------|------|------------|-------|-------------------|------------------|
| Pesticide | active ingredient (ai) | Type | formulation | unit | ai applied | units | K _{oc} | T _{1/2} |
| 2,4D ester | 2,4D ester | herbicide | 520 | g/L | 468 | g/ha | 48 | 32 |
| Basagran | bentazone | herbicide | 480 | g/L | 162 | g/ha | 35 | 27 |
| Harmony | thifensulfuron-methyl | herbicide | 750 | g/kg | 6.7 | g/ha | 28 | 11 |
| clopyralid | clopyralid | herbicide | 300 | g/L | 13.5 | g/ha | 36 | 13 |
| dimilin 25W | diflubenzuron | insecticide | 250 | g/kg | 5.6 | g/ha | 8700 | 8 |
| Land use = wheat | | | | | | | | |
| Pesticide | ai | Type | formulation | unit | ai applied | units | K _{oc} | T _{1/2} |
| Cougar | diflufenican | herbicide | 100 | g/L | 75 | g/ha | <i>Not listed</i> | |
| - | isoproturon | herbicide | 500 | g/L | 375 | g/ha | <i>Not listed</i> | |
| glean | chlorsulfuron | herbicide | 750 | g/kg | 11.3 | g/ha | 36 | 35 |
| karate zeon | lambda-cyhalothrin | apicide | 250 | g/L | 15 | g/ha | 180000 | 43 |
| opus | epoxiconazole | fungicide | 125 | g/L | 187.5 | g/ha | <i>Not listed</i> | |
| amistar | azoxystrobin | fungicide | 250 | g/L | 210 | g/ha | | |
| Land use = peas | | | | | | | | |
| Pesticide | ai | Type | formulation | unit | ai applied | units | K _{oc} | T _{1/2} |
| Sencor | metribuzin | herbicide | 750 | g/kg | 281 | g/ha | 52 | 47 |
| MCPB | MCPB | herbicide | 385 | g/L | 1010 | g/ha | 540 | 10 |
| Bravo | chlorothalonil | fungicide | 500 | g/L | 1125 | g/ha | 5000 | 48 |
| Protek | carbendazim | fungicide | 500 | g/L | 750 | g/ha | 350 | 60 |
| Reglone | diquat | herbicide | 200 | g/L | 525 | g/ha | 500 | 1000 |
| Land use = potatoes | | | | | | | | |
| Pesticide | ai | Type | formulation | unit | ai applied | units | K _{oc} | T _{1/2} |
| Linuron | Linuron | herbicide | 450 | g/L | 1012 | g/ha | 496 | 82 |
| Karate Zeon | lambda-cyhalothrin | insecticide | 250 | g/L | 15 | g/ha | 180000 | 43 |
| Amistar WG | azoxystrobin | fungicide | 500 | g/kg | 187 | g/ha | <i>Not listed</i> | |
| Mancozeb | Mancozeb | fungicide | 800 | g/kg | 2160 | g/ha | 6000 | 43 |

Notes: ai = active ingredient.

very short half-lives were not modelled as it was assessed that their potential to leach was minimal.

Pesticide Leaching Model Simulations

Simulations were carried out for the chemicals highlighted in Table 7 using the LEACHM solute transport model to assess the likely leaching concentrations and masses from the applied chemicals. LEACHM (Version 4.1) is the general acronym (Leaching Estimation and Chemistry Model) that refers to a suite of models describing the 1-dimensional (vertical) storage, transmission, and dissipation of water and solutes in the soil profile (Hutson and Wagenet 1995). LEACHP, one of the modules within LEACHM, describes sorption, movement, and degradation of pesticides. This model uses finite difference approximation to solve the Richard's equation for water flow and the convection-dispersion equation for solute transport. Water retention, matric potential and hydraulic conductivity values must be specified in the model: this is accomplished by modification of Campbell's equation to allow for the exponential function at the wet end to be replaced by a parabolic function at high potential for better representation of water retention characteristics (Hutson and Cass 1987). Degradation and transformation of pesticides is accounted for using first-order kinetics, while sorption isotherms are assumed to be in equilibrium and linear. A more detailed description of LEACHM can be found in Hutson and Wagenet (1995). Daughter (or transformation) products were not considered in these simulations. They can be included in future simulations if required although information on their mobility and degradation characteristics is much more limited compared to the parent compounds. Consideration of any daughter products is unlikely to change the general conclusions regarding possible pesticide contamination in the area.

Simulations were carried out over the period from January 1 2001 to 25 May 2005, as this was the period for which constant rainfall, temperature and evapotranspiration data was available from the nearest NIWA weather station (at Darfield). A summary of the climate data used is presented in Table 8 and a comparison of rainfall data from the two sites is shown in Figure 4. This indicates that Hororata gets significantly more

rainfall than Darfield. Also noted is the mean annual rainfall for the years modelled 2001-2005 is close to the long term mean 1960-2005 indicating that rainfall data used is indicative of general conditions in the area. A summary of the scenarios simulated in LEACHM is shown in Table 9, and the breakthrough curves (concentrations leached from the base of the soil profile at 55 cm) from the simulations are shown in Figures 5-8.

Table 8. Comparison of rainfall, irrigation and leached volumes for the two rainfall datasets used.

| Location | Mean annual rain (mm) 2001-2005 | Mean annual rainfall plus irrigation (mm) |
|----------|------------------------------------|--|
| Darfield | 682 | 1340 |
| Hororata | 828 | 1487 |

Note: Hororata mean annual rainfall from 1960-2005 is 855 mm.

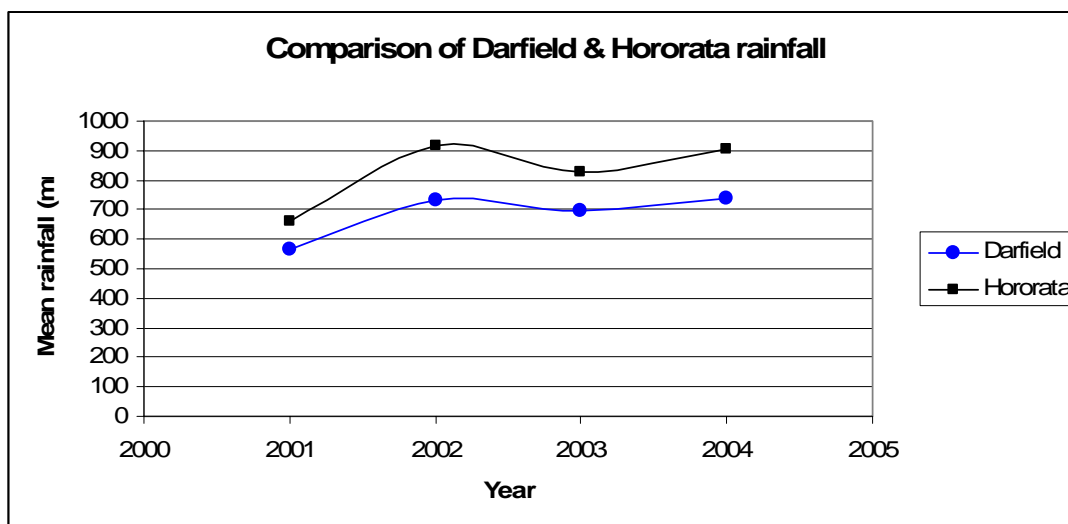


Figure 4. Comparison of mean annual rainfalls for Darfield and Hororata.

It was necessary to make a number of assumptions in setting up the models. Firstly the soil simulated was a typical shallow silt loam (Eyre soil), which is characteristic of the area. Soil physical and chemical input parameters were sourced from Landcare Research. Climate and irrigation data has been collected from a number of sources. All temperature and evapotranspiration data used is taken from the NIWA Darfield weather station. Simulations include rainfall sourced from both Darfield and Hororata. The irrigation data was derived from modelling of an irrigated Te Piritā soil (sourced from Aqualinc Ltd, 2006).

Table 9. Summary of LEACHM pesticide leaching scenarios.

| Simulation Name | Landuse Description | Simulation Dates | Rainfall source | LEACHM Pesticide simulations | | | Koc | 1/2 life | ai g/ha | Application dates |
|-----------------|---------------------|--------------------|-----------------|------------------------------|------------|-----------------------|-----|----------|---------|----------------------------|
| | | | | Irrigation Source | PET source | Pesticides | | | | |
| CPIAPEST | Irrigated Dairy | 1/01/2001-31/05/05 | Darfield | Te Pirita | Darfield | Bentazone | 35 | 27 | 162 | 1 Oct. each year 2001-2004 |
| CPIAAEST | Irrigated Dairy | 1/01/2001-31/05/05 | Hororata | Te Pirita | Darfield | Bentazone | 35 | 27 | 162 | 1 Oct. each year 2001-2004 |
| CPIBPEST | Irrigated Arable | 1/01/2001-31/05/05 | Darfield | Te Pirita | Darfield | Chlorsulfuron (wheat) | 36 | 35 | 11 | 1 Oct. each year 2001-2004 |
| | | | | | | Carbendazim (Peas) | 350 | 60 | 750 | 15 Jan each year 2002-2005 |
| | | | | | | Linuron (Potatoes) | 496 | 82 | 1013 | 1 Oct. each year 2001-2004 |
| CPIBBEST | Irrigated Arable | 1/01/2001-31/05/05 | Hororata | Te Pirita | Darfield | Chlorsulfuron (wheat) | 36 | 35 | 11 | 1 Oct. each year 2001-2004 |
| | | | | | | Carbendazim (Peas) | 350 | 60 | 750 | 15 Jan each year 2002-2005 |
| | | | | | | Linuron (Potatoes) | 496 | 82 | 1013 | 1 Oct. each year 2001-2004 |

Figures 5-8 show that there is a small difference in the effects of greater rainfall with pesticides leached using the Hororata rainfall data having slightly higher concentrations than those leached with Darfield rainfall data.

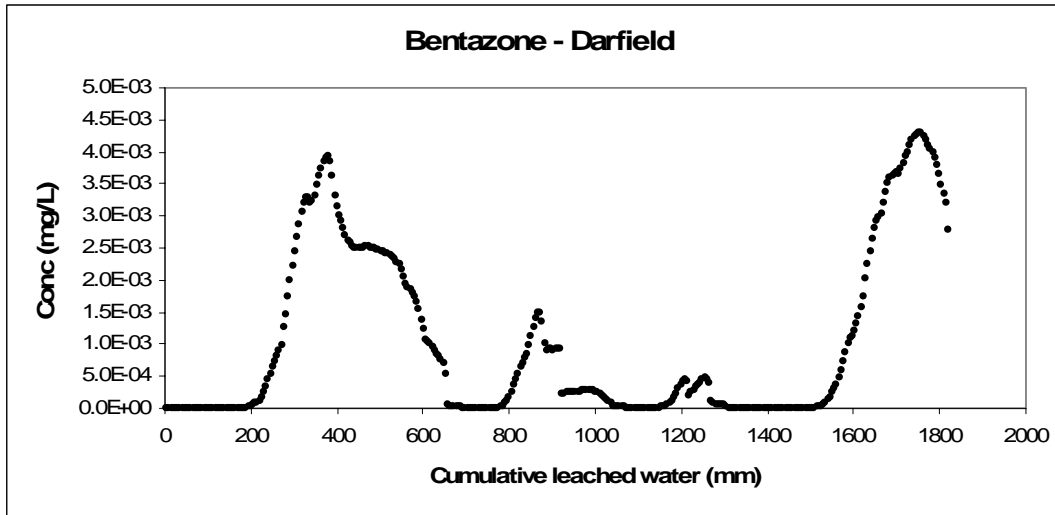


Figure 5. Simulated leaching of Bentazone (irrigated dairy with Darfield rain) showing concentration vs cumulative leached water from 01/01/2001-25/05/2005.

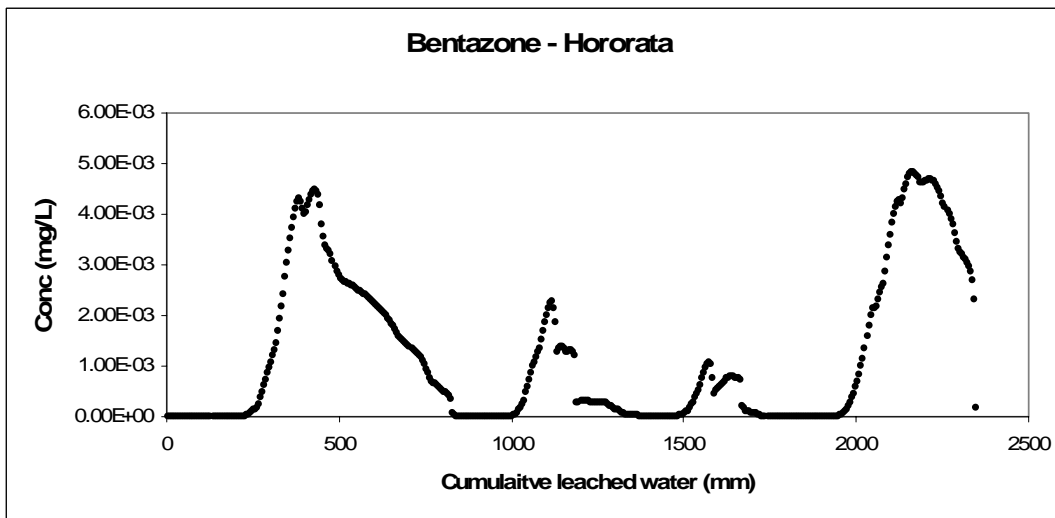


Figure 6. Simulated leaching of Bentazone (irrigated dairy with Hororata rain) showing concentration vs cumulative leached water from 01/01/2001-25/05/2005.

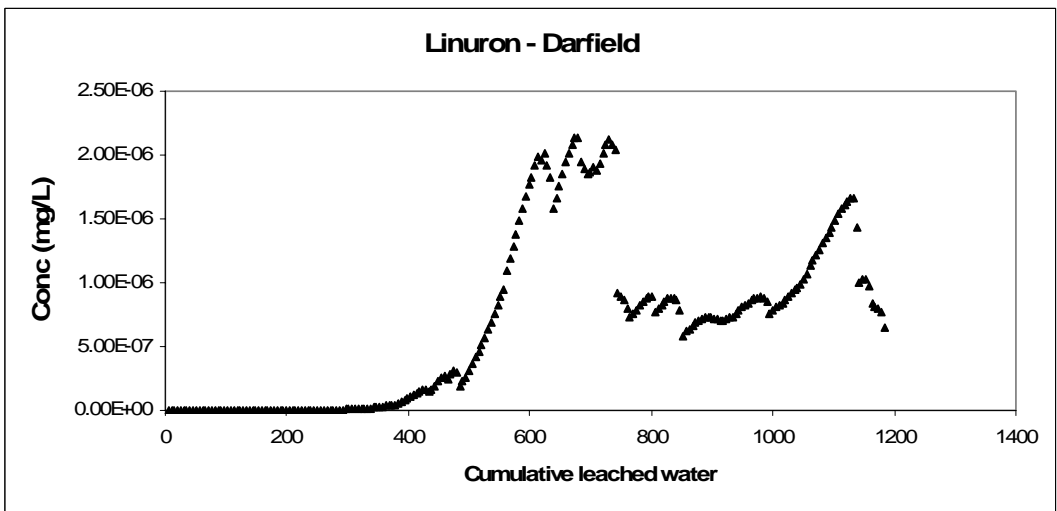
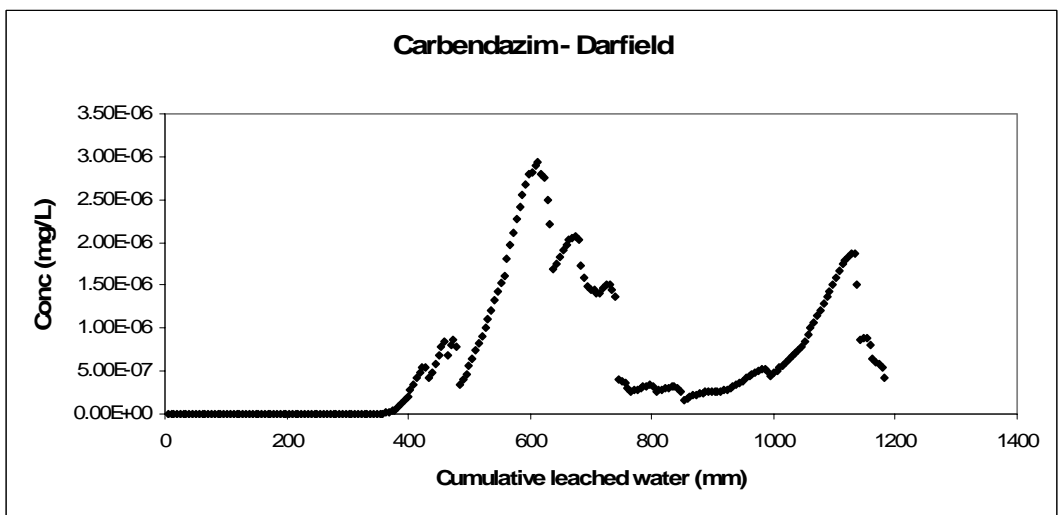
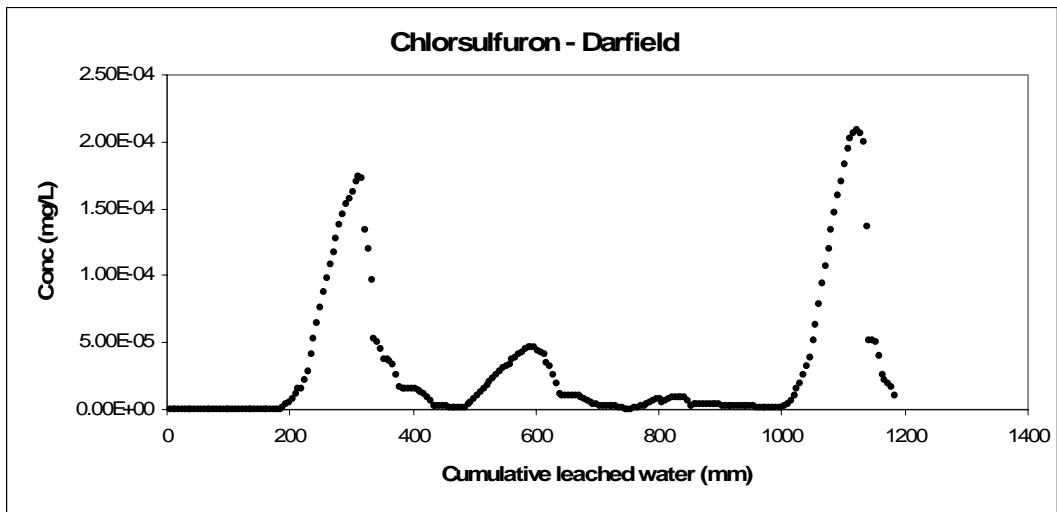


Figure 7. Simulated leaching of chemicals used for arable landuses (wheat, peas, potatoes) under irrigated conditions with Darfield rainfall, showing concentration vs cumulative leached water from 01/01/2001-25/05/2005.

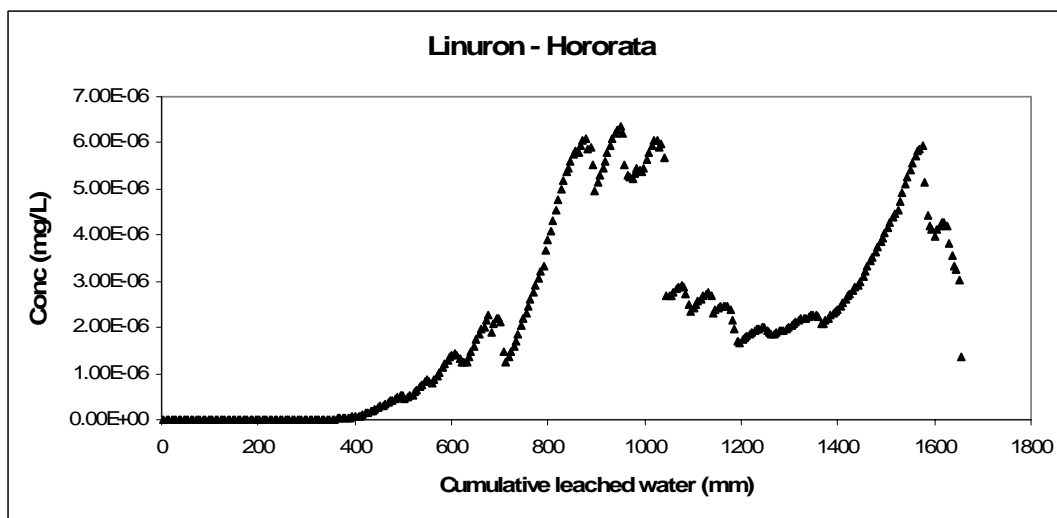
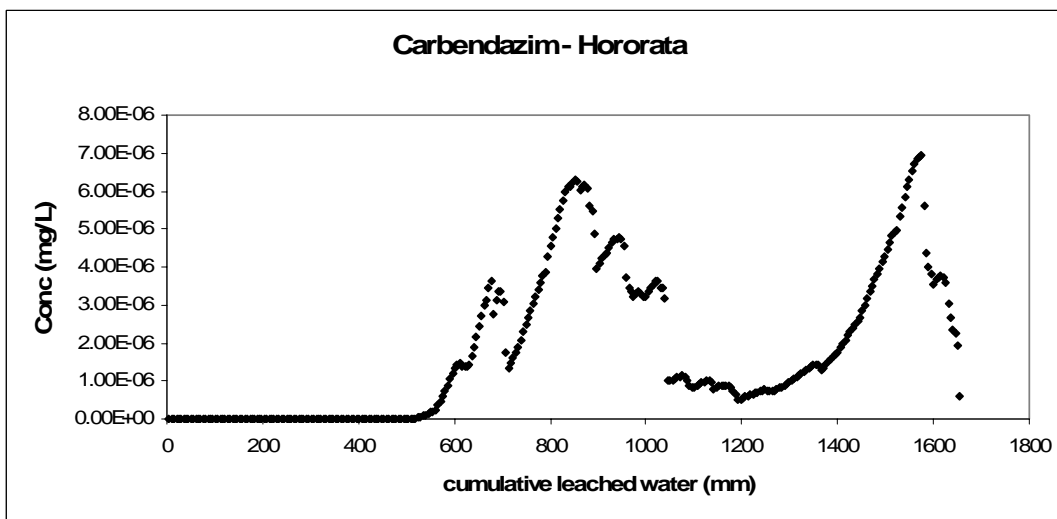
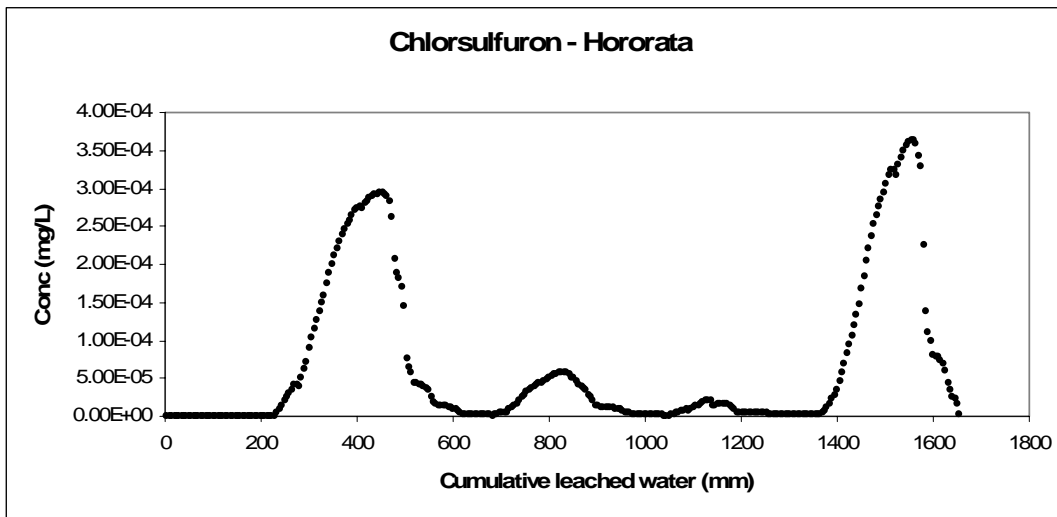


Figure 8. Simulated leaching of chemicals used for arable land uses (wheat, peas, potatoes) under irrigated conditions with Hororata rainfall, showing concentration vs cumulative leached water from 01/01/2001-25/05/2005.

The concentrations of leached pesticides in the simulations were low. Bentazone was approximately 2 orders of magnitude below the MAV and Chlorsulfuron, Carbendazim and Linuron had very low values, though there is no MAV to compare to (see Table 10).

Table 10. Comparison of maximum leached concentration with guideline MAV. Note * denotes no MAV.

| Chemical | Max. concentration at 55 cm (ug/L) | MAV (ug/l) |
|---------------|---------------------------------------|---------------|
| Bentazone | 4.8 | 400 |
| Chlorsulfuron | 0.36 | * |
| Carbendazim | 6.9E-03 | * |
| Linuron | 6.3E-03 | * |

Pesticide reduction capacity of the vadose zone

Some reduction of pesticide mass in the vadose zone is likely to occur but will be much less than in the soil zone. As the organic fraction of coarse sandy gravel vadose zones is generally expected to be quite low, little adsorption of pesticides will occur in comparison to soils. In addition, a low organic fraction leads to a low bacterial mass, contributing to less decay of pesticides and subsequently longer half-lives than within the soil profile. There will be some reduction due to dispersion and mixing in the unsaturated zone, but this is likely to be minimal, especially where groundwater tables are shallow. In cases where there are zones of higher clay or organic matter content within the vadose zone, attenuation and reduction will increase.

It is estimated that in general, the concentration and mass of pesticide leaving the soil profile will, in the most part, continue through the vadose zone to the groundwater table with minimal reduction.

Pesticide reduction capacity of the aquifers/aquitards

The factors affecting the fate and transport of pesticides in aquifer systems are generally the same as those affecting pesticides in soil. However, due to differing

conditions in aquifers the rates are often quite different. Most importantly the organic matter content within aquifers is generally very small, levels of 0.04% have been measured in Canterbury alluvial gravels (Close et al., 2001). As such pesticide sorption in aquifers is likely to be equally or more dependant on clays and other minerals. In addition the microbial population of aquifers is generally much lower than in soil, reducing the capacity to break down pesticides microbially, a major mechanism. A study of pesticide fate in a shallow sandy till aquifer in the United States, (Levy and Chesters, 1995) reported very long degradation half-lives for atrazine and desethylatrazine (3470 and 2770 days, respectively) in groundwater, which were much greater than those derived for soils. There have been similar findings in New Zealand studies for atrazine and picloram in an alluvial gravel aquifer similar to CPWIS (Pang and Close, 1999). They found no observable retardation or degradation for either pesticide over a distance of 90 m and a period of 49 hours in a field experiment. A much longer batch study (194 days) atrazine concentrations showed a significant decrease and picloram concentrations showed a slight decrease. Thus the degradation of pesticides in aquifers will be considerably slower than in soils, and in addition pesticides are likely to be only slightly retarded by the aquifer material. For the purposes of this assessment both degradation and retardation in alluvial gravel aquifers has been assumed to be zero.

As such the main process for the reduction of pesticide concentrations is mixing by advection and dispersion. This effect can be quite large in coarse alluvial gravels that often have recharge from nearby rivers (Close et al., 2001). Groundwater velocities have been measured in the Canterbury Plains ranging from 10 to 110 m/day (Pang et al., 1998). If mixing is assumed to occur in the top 1 m and assuming a porosity of 0.15, a recharge of 2 mm/day, and a groundwater velocity of 5 m/day (all conservative assumptions), then this provides a immediate dilution of around 375 times. While this does not take into account cumulative inputs over a large area, it indicates that there is significant dilution in the Canterbury Plains groundwater system. A more accurate estimate of dilution could be carried out using analytical or numerical modelling but this has not been done at this stage.

National surveys of pesticides in groundwaters carried out every 4 years from 1990 (Close, 1993; Close, 1996; Close and Flintoft, 2004; Close and Rosen, 2001) indicate

that low levels of pesticides have been detected in 10-15% of shallow unconfined wells surveyed, with higher detection rates in targeted surveys. The concentrations of most detections are very low though there were a few detections from point sources that exceed the MAV for drinking water.

Our leaching simulations are in agreement with the national surveys, indicating that, though some pesticides may be leached through the soil layer, they are likely to be at very low concentrations, well below the MAV (where applicable).

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