

**IN THE MATTER OF**

the Resource Management Act  
1991

**AND**

**IN THE MATTER OF**

applications by Central Plains Water  
Trust to:

Canterbury Regional Council for  
resource consents to take and use  
water from the Waimakariri and  
Rakaia Rivers and for all associated  
consents required for the  
construction and operation of the  
Central Plains Water Enhancement  
Scheme

Selwyn District Council for resource  
consents to construct and operate  
the Central Plains Water  
Enhancement Scheme

**AND**

**IN THE MATTER OF**

a notice of requirement by Central  
Plains Water Limited to:

Selwyn District Council for the  
designation of land for works  
associated with the construction and  
operation of the Central Plains  
Water Enhancement Scheme

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**BRIEF OF EVIDENCE OF MURRAY EDWIN CLOSE**

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## **Qualifications and experience**

1. My full name is Murray Edwin Close and I am a research scientist specialising in groundwater quality, employed by the Institute of Environmental Science and Research Limited (“ESR”). I live in Christchurch.
2. I graduated from the University of Canterbury with a Bachelor of Science degree with Honours in 1978, and from the University of Oklahoma with a Master of Environmental Science in 1987.
3. From 1979 to 1988, I worked as a water quality research scientist at the Hydrology Centre of the Ministry of Works and Development in Christchurch. From 1988 to 1992, I was a water quality research scientist with DSIR Geology and Geophysics. Since 1993 I have worked at ESR as the Programme Leader (and Principal Scientist since 1998) in Groundwater Contamination. In this role I have researched the leaching of pesticides into groundwater, the attenuation of contaminants in groundwater, the impact of land use on groundwater quality, and undertaken both regional and national groundwater quality assessments. My knowledge and experience gained from the research have been sought by local authorities to assist them with a number of issues related to groundwater quality and contamination.
4. I have read the code of conduct for expert witnesses set out in Environment Court practice note, and confirm that I have complied with the code in the preparation of my evidence.

## **Scope of Evidence**

5. I have been asked to give evidence on the likely impacts on groundwater quality in the area of the CPWES as a consequence of the Scheme proceeding. My discussion will cover the potential for denitrification of nitrate inputs to occur and the likely impacts on pesticide levels and microbial levels in groundwater
6. The first section of my evidence will cover the potential for denitrification (transformation of nitrate into nitrous oxide and nitrogen gas) in the vadose zone (unsaturated zone between the soil and groundwater) and in the groundwater of the Central Plains Scheme area. Denitrification would result in removal of nitrate and therefore less nitrate being leached into the groundwater and reaching receiving waters
7. The second section of my evidence will look at microbial loading onto pasture from intensive dairying, the removal of microbes through the soil, vadose

zone and groundwater, the impacts of irrigation methods on groundwater contamination and likely impacts of the proposed scheme on microbial groundwater quality. Microbes from dairy shed effluent and point sources such as septic tanks will also be discussed.

8. The third section of my evidence will cover pesticide loading from anticipated land uses, pesticide removal through soil, the vadose zone and groundwater, existing levels of pesticide contamination in Canterbury groundwater, and an assessment of the impact of the proposed scheme on pesticide contamination of groundwater. The likely effects on groundwater quality in the Scheme area are then summarised and the final section of my evidence draws some conclusions.
9. I have relied on published studies in the scientific literature, published technical reports, and investigations carried out by myself and other scientists at ESR. The published reports and papers are referenced where appropriate. More detailed information is contained in Close et al. (2007b - ESR Technical Report CSC0716) and is available on request.

### **Estimation of Nitrate Reduction and Dilution in the Vadose Zone**

#### **Key Points**

- There is some evidence that denitrification can occur in the alluvial gravel vadose zones that occur in Canterbury.
- However, the amounts of denitrification can not be quantified at this stage and are thought to be minor, so a conservative assumption has been made that denitrification in the vadose zone is zero.

10. The vadose zones in the vicinity of the Central Plains Water Enhancement Scheme (CPWES), as with the majority of the Canterbury Plains, consist mainly of hardened 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 have relatively low carbon concentrations, especially in the Central Plains area away from the coast. As the rate of denitrification is mainly controlled by the abundance of denitrifying bacteria, the quantities of organic carbon and a low oxygen content in the relevant environment (Holden and Fierer, 2005), the nitrate-reducing capacity in the CPWES vadose zone type of environment is regarded as being minimal (Close et al., 2001).

11. Denitrification in vadose zones has been reported (Holden and Fierer, 2005). Reports commonly show denitrification tends to decrease rapidly with depth. 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.
12. 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. The results so far do not indicate whether the nitrate was being reduced to nitrogen gases in the subsoil (and then being transported through the vadose zone with pore water movement) or within the sandy gravel vadose zone itself. The amounts of denitrification could not be quantified in the field, and laboratory experiments are continuing to quantify the likely amounts of denitrification in this environment.
13. 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 dependent on the amount of water leaching through the profile.
14. At present I am not aware of any quantification of denitrification occurring in gravel vadose zones such as those found in the CPWES, therefore nitrate reduction capacity of the vadose zone is conservatively 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.

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

#### **Key Points**

- There will be little reduction in nitrate concentrations in the CPWES groundwater resulting from denitrification because the groundwater is generally well oxygenated.

- The main reduction in nitrate concentrations in the aquifers will be from mixing and dilution

15. Pang et al. (2006) studied the impacts of septic tank system discharge on nitrate and microbial quality of groundwater systems in the Canterbury Plains. They performed a sensitivity analysis that showed that reduction in nitrate concentrations in gravel aquifers was most sensitive to hydraulic conductivity, which affects the observed concentrations by increased groundwater flow and dilution of the recharge. A cumulative effect of nitrate was shown for a series of septic tanks and wells down a flowpath. High flow 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. While dilution is a factor in lowering nitrate concentrations in aquifers, this is only relevant when the diluting water has low nitrate concentrations. Where up-gradient groundwater also has high nitrate concentrations the dilution effect will be minimal. These study results can be appropriately applied to 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.
16. 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, which reduces 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 CPWES. 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.
17. 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 than water near the water table. In Environment Canterbury's review of nitrates in groundwater (Hanson, 2002) wells deeper than 50 m tended to have lower nitrate concentrations compared to shallower wells and wells below 150 metres had concentrations consistently less than 5 mg/L, although no statistically significant correlation was found between depth and nitrate concentration,.
18. The nitrate reduction capacity of aquifers, which are groundwater systems that produce lots of water, and aquitards, which are low-permeability units that can

transmit water slowly from one aquifer to another, in the vicinity of the CPWES 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

### Key Points

- There are a lot of bacteria that are deposited onto pasture by dairying;
- There is significant die-off of microbes in the soil and under spray irrigation there is minimal leaching of bacteria;
- There needs to be care exercised with application of dairy shed effluent. Estimates indicates that a 4-log reduction of bacteria from spray irrigation of effluent is likely for a soil depth of 0.7 m;
- While border strip (or border dyke) irrigation can result in extensive leaching of microbes, spray irrigation, such as centre pivot or Briggs Rotarainer, will result in minimal leaching of bacteria;
- There is significant removal of microbes in both the vadose zone and groundwater systems (more uncertainty exists for the vadose zone) under dryland or spray-irrigated situations;
- While septic tanks and other point sources provide a significant risk to microbial quality of shallow groundwater in the local area, attenuation processes in the groundwater mean that these effects are not cumulative.
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### Microbial loading for dairy land use

19. The loading of microbes to the soil surface from farming-related practices relates 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).

20. The average daily faecal output of a dairy cow can be estimated at 25 kg. The average number of cows per hectare on irrigated pastures is approximately 2.9 cows/ha (MAF, 2000; Wilcock et al., 1999) and this gives an average *Escherichia coli* loading of  $4.7 \times 10^6 / \text{m}^2 / \text{day}$  (4.7 million bacteria per square meter per day). Variation in stocking rate from 2.5 to 4.0 cows per ha would

change the loading from 4.6 to  $6.5 \times 10^6$  per  $m^2$ . This is very conservative as it assumes no surface die-off, and that all bugs deposited infiltrate with leached water. Die-off in cowpats and on the land surface will be significant and will depend on a number of environmental conditions including temperature, water content, predators and ultraviolet 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). Using a conservative mean die-off value (0.062) from Howell et al. (1996) with the daily inputs of  $4.6 \times 10^6$ , the *E. coli* concentrations level out at approximately  $7.4 \times 10^7/m^2$ . Assuming a very large drainage event (90 mm) and all surviving faecal coliforms being leached, this converts to a leachate concentration of approximately  $8.2 \times 10^4$  cfu/100ml. Averaged over a year of drainage (assuming 600 mm) the derived faecal coliform concentration in drainage water is  $1.2 \times 10^4$  cfu/100ml (cfu = colony forming units). 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 depends 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.

21. The New Zealand drinking water guideline is  $<1$  cfu *E. coli* /100ml. 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. So there are a lot of bacteria being deposited on the pasture that have the potential to leach. In the next section I consider the factors that reduce these bacterial numbers.

### **Microbial reduction capacity of soils**

22. The microbial reduction capability of soils in the CPWES mainly depends 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 residence time within the soil and a subsequently 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 include moisture content, soil temperature and pH, organic matter and sunlight (if on the soil surface).

23. 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 CPWES (Close et al., 2005) indicate that counts of *E. coli* are reduced to a mean of <0.5 – 0.7 cfu/100ml at 1.2 m depth under a regular regime of 50 mm of spray irrigation using a Briggs Rotarainer system (Table 1). 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. This means that under normal conditions with Briggs Rotarainer spray irrigation there is little leaching of bacteria. The second line in Table 1 considers the situation of irrigation followed immediately by a heavy rainfall of 30 mm. In this situation significant bacteria leached to a depth of 1.5 m. The numbers were about 10 to 1000 fold less than the potential numbers estimated in section 20, indicating that, even with heavy rainfall, substantial reductions in bacterial numbers had taken place.

Table 1. 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> (cfu/100 ml)	Max <i>E. coli</i> (cfu/100 ml)
~ 50	16	< 0.5 – 0.7	< 0.5 – 7
~ 80	4	36 – 275	200 – > 2000

### Dairy Shed Effluent

24. A study on leaching of dairy shed effluent is currently being carried out by ESR/Lincoln University using intact soil cores in lysimeters. This study 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 (Jiang et al., 2007).

25. Scientists from Landcare Research have published a number of papers on movement of bacteria and viruses from dairy shed effluent through a variety of soils (Aislabie et al., 2001; McLeod et al., 2003; McLeod et al., 2001). These experiments have been run with an effluent application of 25 mm at 50 mm/hr, followed by continuous leaching with spray irrigation rates of 5-10 mm/hr. In these experiments differential microbial movement was attributed to differences in soil structure, ie. where bypass flow (where the water flows in cracks and worm holes and by-passes most of the soil) 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, which increased filtration and decreased leaching of bacteria (Aislabie et al., 2001). Simulation of these Landcare Research lysimeter leaching data (Pang et al., 2007) indicates that log reduction per m varies in the soils tested, with mean values of 2–3 log reduction/m. The silt loam soils had a 2 log reduction/m value and the sandy loam soils reduction value ranged from 2.4–3 log reduction/m.

26. 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 encourage bypass flow. Soils of the CPWES are generally well drained and, under efficient spray irrigation, there is likely to be less bypass flow occurring as found in a study by Aislabie et al., (2001) on microbial leaching for well-drained soils. The 4-log reduction of bacteria from dairy shed effluent through a depth of 0.7 m is likely to be typical for the CPWES area.

### **Impact of Irrigation Method**

27. The type of irrigation method is an important parameter for the leaching of microbes through stony silt loam soils in Canterbury. Sinton et al. (1997) indicate that significant leaching of microbes through soils and vadose zones occurs from the application of sewage effluent under border strip (also known as border dyke) irrigation conditions, with large numbers of bacteria (up to >2400 cpu/100ml) recorded in down-gradient wells. In a study of dairying under border strip irrigation, *E. coli* indicator bacteria were found in 75% of samples and the pathogenic bacteria *Campylobacter* was found in 12% of samples from shallow groundwater (Close et al., 2007a).
28. In the CPWES the irrigation method will be spray and therefore there will be much greater attenuation in the soil zone, more like the Templeton silt loam column study (reduction of approximately 4-log/ metre). There is uncertainty in the loading rate around the percentage of microbes leached from the cowpats into the soil during leaching events. This percentage leached into the soil depends 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, with higher energy events being more likely to increase the number of microbes in infiltrating water.
29. I have measured microbial contamination in shallow groundwater over a 6-year period at the Lincoln University Dairy Farm using Centre Pivot spray irrigation

(Close et al, 2007c). This is described more fully in paragraph 33 and indicates that there is little groundwater microbial contamination from dairying with a centre pivot irrigation system.

### **Microbial reduction capacity of vadose zone**

30. Little work has been carried out to date 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 die-off due to the time of transport and filtration. Pang et al. (2006) noted that microbes were not readily retarded in gravel aquifers and estimated an overall removal rate coefficient (combined die-off and filtration) of  $1.3 \text{ log day}^{-1}$  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 is 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.
31. In a study of septic tanks by Sinton (1986), carried out at Burnham, faecal coliforms reduced 2.5-log in 9.5 m thickness of unsaturated zone. This gives a removal rate of 0.27 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.
32. Laboratory studies recently undertaken by ESR on the survival of different micro-organisms in unsaturated gravel material indicate the die-off rate for faecal coliforms (including *E. coli*) is approximately a factor of 3 in 3 months, and for *Campylobacter* there is approximately a  $10^4$  reduction in concentrations in 2 weeks
33. 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.

34. As there are 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 CPWES 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.
35. Data from some current dairy farms suggest higher removal rates from cow pat to leachate concentrations in the vadose zone. As highlighted earlier, results from experiments at Te Pirita under typical dairy farming conditions (Close et al. 2005) indicate mean *E. coli* counts are reduced below drinking water guidelines in the vadose zone at 1.2 m (< 0.5 – 0.7 cfu/100ml) under a regular application of 50 mm with a Briggs Rotarainer irrigation system (Table 1). As the Te Pirita dairy farm is a typical dairy farm it can be assumed that the microbial loading is in the region of  $7.4 \times 10^7/\text{m}^2$ . When heavy rainfall occurs in conjunction with irrigation (giving a total input of around 80 mm) some leaching of faecal coliforms was observed.
36. The groundwater has been monitored over the past 6 years at the Lincoln Dairy Farm, which has a centre pivot irrigation system (Close et al. 2007c). A total of 651 and 693 groundwater samples have been analysed for faecal coliforms and *Campylobacter*, respectively, over this period. There have been 14 groundwater samples out of a total of 257 (5.4%) from up-gradient wells with detections of faecal coliforms at concentrations of 1 cfu/100 ml or greater, and 11 groundwater samples out of a total of 394 (2.8%) from down-gradient wells with detections of faecal coliforms at concentrations of 1 cfu/100 ml or greater. Most faecal coliforms concentrations have been 1–3 cfu/100 ml. There have been five detections of *Campylobacter* over the 6-year period. Three were from down-gradient wells (0.7%), and two were from up-gradient wells (0.7%). 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 6 years of operation.
37. 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**

38. 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 of  $< 1$ , indicating that bacteria are travelling at a slightly faster rate than the average groundwater velocity through preferential pathways. A review of the literature indicates that microbes have been recorded up to 900 m down-gradient of the release site in aquifers at Burnham, similar to those within the CPWES.
39. Pang et al. (2005) reported a gradation of microbial reduction capacity in aquifers depending on the aquifer physical properties and contamination status. The results of the study indicated that a 1-log reduction would require 17–40 m travel in clean coarse gravel aquifers, 240–530 m travel in contaminated coarse gravel aquifers, 5–9 m travel in clean sandy fine gravel aquifers, 5–18 m travel in contaminated sandy fine gravel aquifers, and 5–7 m travel in contaminated river and coastal sand aquifers.
40. This suggests that in generally clean coarse gravel aquifer environments such as those generally in the CPWES a 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 from a point source.
41. Under the new land use scenarios there will be a slight increase in the area where the groundwater table will be within 5 m of the land surface (for further details see evidence presented by AquaLinc Research Ltd). 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. In evidence presented by Julian Weir, AquaLinc Research Ltd, he states that there will be a small increase (3.6

– 6.5%) of the total land area that has groundwater levels within 5 m of the ground surface. An additional 1.2 – 2.4% of the total land area will have groundwater levels within 1 m of the ground surface.

42. It is expected that for the aquifer environment in the CPWES, microbial contamination is unlikely to have a cumulative impact and will be localised to contaminant sources, as is currently the case. Only low counts of microbes 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 to below drinking water guideline levels in a relatively short distance (10s to 100s of m).

### **Other Sources of Microbes in Groundwaters**

43. Sinton (2001) in his assessment of microbial contamination of New Zealand aquifers states that microbial contamination of groundwater in the Canterbury region has been attributed to poor well-head protection, septic tank systems and land application of animal effluents. It is likely that these three issues provide the greatest risk to the microbial quality of groundwater in the CPWES. The general diffuse contamination of groundwaters originating from general grazing (cowpats) has been discussed earlier in my evidence and this is less likely to be a concern, where spray irrigation is used, than point source pollution aspects of this land use. Grazing animals give rise to three forms of point sources including offal pits, dairy-shed effluent, and animal containment areas (Sinton 2001).
44. Where land-use intensifies and the population subsequently increases, septic tanks can represent a significant source of human effluent to groundwater. In semi-rural and rural areas, a high proportion of households have septic tank systems. Most of the properties in the Selwyn District are on septic tank systems (around 6000 in 1992) (O'Boyle, 1992). A typical septic system discharges into a subsurface disposal structure, such as a soakage trench or deep pit. Soakage pits generally connect to the porous gravel media in the Canterbury area, through which effluent can leach to groundwater. A number of studies have been carried out on the effects of effluent disposal to soakage pits (Mulcock and Noonan, 1973; Sinton, 1982; Sinton, 1986). In the study by Sinton (1986) there was evidence of groundwater mounding underneath a soakage pit, and samples collected from a well located down gradient in the unconfined aquifer exhibited diurnal fluctuations of faecal coliform concentrations in response to periods of effluent discharge.

45. The current General Authorisation for Sewage Tank Effluent Disposal (Canterbury Regional Council 1991) indicates that only 600–900 mm of sand with a mean grain size of approximately 0.5 mm is required in the base of seepage trenches to remove pathogenic bacteria. Due to a lack of appropriate sand, soakage trenches in Canterbury are often filled with coarser sand, not compliant with the particle size requirements of the guideline. There is little information on the effectiveness of this method, and any interactions with shallow groundwater tables (pers comm. M. Leonard, ESR, 2006). The required separation distances between bores and discharge systems are stipulated in the General Authorisation document (and the Transitional Regional Plan). As explained below I consider this is not an issue that needs to be addressed for CPWES, as the increased number of septic tanks is unlikely to need mitigation.
46. While septic tanks provide a significant risk to the microbial quality of shallow groundwater, Pang et al. (2006) has modelled the effects of a series of septic tanks in a down-gradient direction to assess the cumulative effects of effluent disposal (Figure 1). Their findings indicate that, while faecal coliforms are likely to contaminate shallow aquifers in a localised effect, attenuation processes such as die-off and filtration ensure that there is not a cumulative effect for bacteria.

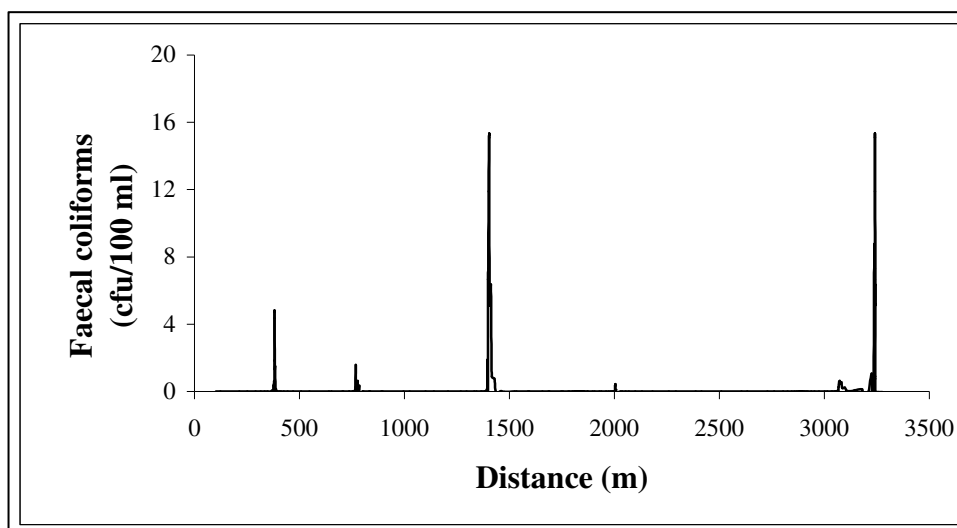


Figure 1. Simulated effect of septic tank effluent on down-gradient groundwater quality showing that the effect is only localised (10's to 100's m). Each increase corresponds to inputs from a separate septic tank system.

47. Therefore, while shallow groundwater immediately down-gradient of a septic tank should be treated with caution, a large persistent plume of microbial contamination is unlikely to eventuate from a series of septic tanks along a flowpath. These localised effects have been contemplated by both the Transitional Regional Plan through the General Authorisation and the PNRRP through the permitted activity rule WQL9. New septic tank treatment systems will be required to comply with a set of conditions within the Plans to be permitted, and if they do so, the effects on the environment will be minor, as contemplated by the permitted activity status.
48. The disposal of offal and carcasses by burial in pits is common farming practice in New Zealand. These pits carry the potential for disease transmission via groundwater to both domestic and stock bores (Sinton, 2001). A Ministry of Agriculture and Fisheries report (MAF, 1991) provides guidelines, which only require the bottom of the pit to be 1 m above the groundwater table. This is unlikely to be sufficient distance to attenuate all potentially leaching microbes, and there is also the potential for water tables to rise and fall significantly in the Canterbury area. A more appropriate document is an Environment Canterbury report (Canterbury Regional Council 1999), which reviewed the available literature and recommended a distance of > 3 m between the base of the pit and the groundwater table, and that the pit be located a minimum of 250 m up-gradient of any water supply well.

In summary, microbes associated with general grazing are likely to attenuate and die-off, either at the surface or as they infiltrate through the soil and vadose zones. This assumes that irrigation efficiencies will be high, as the lower the irrigation efficiency the higher likelihood of microbial leaching to groundwater. The management of potential point sources appears to be a greater issue regarding the potential contamination of shallow aquifers. The effects of point source pollution are likely to be confined to the local area and accumulation from a number of point sources such as septic tanks should not lead to cumulative effects.

## Impacts of CPWES on Pesticides in Groundwater

### Key Points

- Pesticide leaching simulations carried out for bentazone, chlorsulfuron, carbendazim, and linuron using the LEACHM solute transport model indicated that concentrations likely to leach below 55 cm were low..

- There was likely to be some further reduction in the vadose zone.
- There would significant reduction in the groundwater system mainly from dilution and mixing.
- The results from the national surveys of pesticides in groundwater show that 10-15% of shallow wells have low levels of pesticides detectable, with most levels well below the drinking water MAV.
- Overall there is likely to be little or no problem from pesticides resulting from the CPWES.

### **Pesticide loading for land use scenarios**

49. The fate and transport of a variety of pesticides has been modelled for two proposed land uses. Prior to the modelling an assessment of the pesticides most commonly used for a particular land use was made (Close et al., 2007b).

Two different land use scenarios were assessed:

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

50. 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 insecticides are used for these land uses.

### **Pesticide reduction capacity of the soil layer**

51. The main factors influencing the leaching of a chemical are 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 (attached to the soil particles) 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.

52. 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. The pesticides with very high  $K_{oc}$  values or very

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

### **Pesticide Leaching Model Simulations**

53. Simulations were carried out for bentazone (herbicide used in dairying), chlorsulfuron (herbicide used in wheat), carbendazim (fungicide used in peas), and linuron (herbicide used in potatoes) 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 one-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. A more detailed description of LEACHM can be found in Hutson and Wagenet (1995). Daughter (or transformation) products were not considered in these simulations. 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.
54. Simulations were carried out over the period from January 1 2001 to 25 May 2005. The soil simulated was an Eyre shallow silt loam, which is characteristic of the area. Soil physical and chemical input parameters were sourced from Landcare Research. Climate and irrigation data were collected from the following sources. All temperature and evapotranspiration data used is taken from the NIWA Darfield weather station. Rainfall data were sourced from both Darfield and Hororata. The irrigation data was derived from modelling of an irrigated Te Pirita soil (sourced from Aqualinc Ltd, 2006). A summary of the rainfall and irrigation data used is presented in Table 2, indicating that Hororata gets significantly more rainfall than Darfield. The mean annual rainfall for the years modelled 2001–2005 is close to the long term mean for 1960–2005.

Table 2. Comparison of rainfall and irrigation totals for the two rainfall datasets used (Hororata mean annual rainfall from 1960-2005 is 855 mm).

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

55. The simulated concentrations of leached pesticides were low (Table 3). Bentazone was approximately two orders of magnitude below the New Zealand drinking water Maximum Acceptable Value (MAV) and chlorsulfuron, carbendazim and linuron had very low values, though there is no MAV to compare to (see Table 3). The pesticide leaching simulations indicated that there was 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 (due to decreased residence time in the soil layer where most degradation takes place)

Table 3. 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	0.0069	*
Linuron	0.0063	*

### **Pesticide reduction capacity of the vadose zone**

56. Some reduction of pesticide mass is likely to occur in the vadose zone 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, microbial biomass decreases with depth, contributing to less decay of pesticides and longer half-lives.

57. 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**

58. 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. The organic matter content within aquifers is generally very small, with levels of 0.04% measured in Canterbury alluvial gravels (Close et al., 2001). As such pesticide sorption in aquifers is likely to be equally or more dependent 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. 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 for atrazine and picloram in an alluvial gravel aquifer similar to CPWES (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. In a longer batch study (194 days) atrazine concentrations showed a significant decrease ( $T_{1/2} = 204$  days) and picloram concentrations showed a little if any decrease.

59. Thus the degradation of pesticides in aquifers will be considerably slower than in soils, and 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.

60. As such, the main process for the reduction of pesticide concentrations is mixing by flowing groundwater 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 10–110 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 and I consider it is not needed in this case because of the low concentrations of pesticides simulated at the bottom of the root zone (section 55; Table 3).

61. 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.

62. 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 available).

### **Point Sources of Pesticides**

63. Close et al. (2001) noted that pesticides can enter groundwater systems in a number of ways including:

- By leaching of pesticides applied during normal agricultural and horticultural operations through the soil and vadose zones;
- By leaching of pesticides used in industry, including spraying along roads; railways and buildings. In some of these operations, soak holes are used to dispose of excess runoff, allowing pesticides easier access to groundwater;
- By recharge of groundwater by contaminated surface water; and
- Point source entry of pesticides into groundwater due to inappropriate storage near wells and inadequate well-head protection.

64. The first bullet point has been covered by the simulations of pesticide movement and persistence described earlier in my evidence. These simulations indicate that while pesticides applied during normal agricultural practices may leach through shallow soils in the CPWES, their concentrations at the base of the soil profile are very low. The simulation results indicated that all the pesticides modelled had concentrations at the base of the soil profile well below the MAV for the New Zealand Drinking water guidelines. Any leached concentrations of pesticides will be further diluted when they mix with underlying flowing groundwater and no further mitigations are needed.

65. The other three bullet points will not be affected by the CPWES and thus there will no additional impact on groundwater quality due to the scheme.

### **Risks for Shallow Aquifers**

66. Shallow aquifers are most at risk due to their proximity to the surface and any potentially contaminating land use practice, and also due to the unconfined nature of the shallow aquifers in this area. The soil and vadose zone provide a degree of protection from surface applied contaminants, but their effectiveness

is dependent on their physical, chemical and biological properties, as well as the conditions they are subjected to. The longer the contaminants take to transit through the soil and the vadose zone, the less the risk of contamination of the shallow unconfined aquifer providing there is a degradation mechanism, as is the case for microbes and pesticides.

67. The greatest risk of contamination of the shallow aquifers comes from the application of nitrogen fertilizers on shallow rooted crops, leaching from urine patches and effluent irrigation. There is likely to be very little if any reduction of nitrate after it exits the soil root zone. Once in the aquifer the reduction capacity is again small with dilution the only reduction process occurring in these oxidized aquifers. This dilution process can take a long time and nitrate plumes can travel long distances contaminating significant quantities of water. Effective management of the nitrogen application and irrigation is important to reduce the volumes and concentrations of nitrate in leached water and I understand this is being addressed by other witnesses.
68. The risks to shallow aquifers associated with microbial contamination are assessed to be minimal under dry-land conditions and low under dairy spray irrigation regimes, with the exception of point source pollution and where water tables are within 5 m of the surface. The main factor controlling leaching of microbes deposited through grazing (i.e. cowpats) is over-irrigation. As the CPWES will be spray irrigated, leaching of microbes under general grazing land will be minimal. However, there will be an increased risk of contamination after large rainfall events, especially where water tables are close to the surface as a result of additional irrigation.
69. Potential avenues of point source pollution such as offal pits, effluent irrigation and poorly constructed well-heads may lead to localised high concentrations of microbes in aquifers. Poorly constructed well-heads provide opportunity for contamination of wells in the moderate and deeper aquifer, while offal pits and effluent irrigation are most likely to affect shallow aquifers.
70. Where viable microbes do leach into the groundwater system it is unlikely that microbial contamination would have a cumulative effect, as highlighted by the modelling of Pang et al. (2006) (see Figure 1). Contamination events that do occur as a result of grazing are likely to occur mainly after large rainfall events; be restricted to shallow aquifers; have a localised impact; and be relatively short lived (as a result of microbial die-off). They should be adequately mitigated with the proposed Sustainability Agreement.

71. The risks of contamination of shallow unconfined aquifers from pesticides is assessed to be low. Very low pesticide concentrations in recharge waters (as indicated from LEACHM modelling presented in the previous section), combined with dilution when leached water intersects the saturated zone reduce concentrations of pesticides in aquifers to between low and non-detectable values. This observation is consistent with the findings of the New Zealand National Pesticide Surveys (Close, 1993; Close, 1996; Close and Flintoft, 2004; Close and Rosen, 2001), which show that, while 10–15% of wells sampled from shallow unconfined groundwater systems in New Zealand had detectable concentrations of pesticides, concentrations were generally well below the MAV.

### **Risks to Moderate and Deep aquifers**

72. Risks to moderate and deep aquifers relate mainly to well-head protection and poor well construction issues, though nitrate contamination of deeper aquifers can occur as long-lived plumes gradually move to deeper aquifers while moving down gradient. The most direct avenue for contaminants from the land surface to the moderate and deep aquifers is down the well itself. There is also potential for contamination within wells that intersect shallow and moderate or deep aquifers. Where there is a negative pressure head difference between the deeper aquifer and the shallow aquifer water can transfer from the shallow aquifer to the deeper aquifer. Where the shallow aquifer is contaminated this can result in subsequent contamination of the deeper aquifers.

73. In general, contaminants are restricted to moving through the shallow unconfined aquifer system with the exception of long lived contaminants such as nitrates which may move down through the profile (where no confining layers exist) as they migrate down the plains. Microbial and attenuation and die-off processes confine the extent of their contamination effects to a localised area. Dilution of low leached concentrations of pesticides is likely to result in concentrations of pesticides in deeper aquifers that are non-detectable.

74. Accordingly in the upper plains only shallow groundwater is at risk from nitrate (mainly), microbes (to a lesser extent) and pesticides (to a much lesser extent). The exceptions to this are when wellhead and well construction issues arise, leading to potential direct contamination of deeper wells. In the mid-plains the risk increases to moderately deep aquifers where confining layers are not continuous (for nitrate pollution) and shallow aquifers, while in the lower plains the risks that contamination may extend to deeper aquifers are less likely due to confining layers and contributions from river recharge at depth.

## Summary and Conclusions

75. The increased water flux and proposed changes to land use in the CPWES may result in some localised effects on microbial quality due to new septic tanks or additional effluent disposal. Pesticides are less likely to be a concern for groundwater quality. The shallow unconfined aquifers are the most vulnerable to leaching of nitrates and microbes, especially where the groundwater table is less than 5m from the surface. Deeper aquifers are less at risk of contamination.
76. At present, although there is some evidence of denitrification occurring in gravel vadose zones, I am not aware of any quantification of denitrification occurring in these zones, therefore nitrate reduction capacity of the vadose zone is conservatively assumed to be zero. The nitrate reduction capacity of aquifers and aquitards in the vicinity of the CPWES with respect to denitrification is likely to be minimal, with the main process in lowering nitrate concentrations being mixing and dilution.
77. There should be little impact from dairying irrigated with spray irrigation on the microbial quality of groundwater. 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 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.
78. The effects of point source microbial pollution, such as septic tanks and offal pits, is likely to be confined to the local area and accumulation from a number of point sources such as septic tanks should not lead to cumulative effects. While septic tanks provide a significant localised risk to the microbial quality of shallow groundwater, attenuation processes such as die-off and filtration ensure that there is not a cumulative effect for bacteria.
79. Simulations of pesticide movement and persistence indicate that while pesticides applied during normal agricultural practices may leach through shallow soils in the CPWES, their concentrations at the base of the soil profile are generally very low. The simulation results indicated that all the pesticides modelled had concentrations at the base of the soil profile well below the MAV

for the New Zealand Drinking water guidelines, with further dilution occurring when the leached water reaches the groundwater table. The risks of contamination of shallow unconfined aquifers from pesticides is assessed to be very low.

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