Public Health Implications of Land Use Change and Agricultural Intensification with respect to the Canterbury Plains

A Literature Review

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Executive summary

Land use change has been occurring rapidly in recent years, both at a global scale and within Canterbury. While there is potential for this change to create wealth – indeed wealth creation is usually the purpose of land use change – there is also potential for unintended effects which may impair the health of communities. This review is intended to provide information to help anticipate and avoid negative consequences of land use change.

This literature review was originally completed in 2010 and updated in 2014. There is a focus on Canterbury and New Zealand literature where available, and international literature is also considered in the Canterbury context. There is also substantial reference to non-peer reviewed reports produced by New Zealand and International governmental and research organisations.

The history of land use change in Canterbury is as long as the history of human occupation, and has allowed substantial increases in health and wellbeing. Recently, the dairy boom has driven rapid expansion of irrigation and agricultural intensification, while the 'lifestyle block' trend and the 2010-2011 Canterbury earthquakes have driven urban expansion of Christchurch's rural fringe and satellite towns.

The majority of literature examining the health effects of land use change is focussed on water quality. There is clear evidence linking agricultural intensification and urban expansion in Canterbury with decreases in water quality and increases in the incidence of waterborne disease. Furthermore, these effects are often most keenly felt by those who can least afford mitigation measures – people who are often already vulnerable and can easily be pushed in to poor health. Poor health of any member of a community has a financial cost to all, both in the cost of treatment and the loss of productivity to the economy. Many mitigation technologies are available which, if implemented, could allow intensification of existing agricultural activities without the negative external effects on water quality. However, these mitigation technologies also add cost so may make more intensive agriculture uneconomic. Furthermore, the currently available mitigation technologies are not sufficiently effective to offset conversions of land from low-intensity activities such as forestry and dryland sheep-beef farming to high-intensity activities such as dairying or cropping.

Apart from water quality, agricultural intensification and urban expansion have the potential to affect health through increased greenhouse gas emissions, loss of biodiversity and ecosystem services, and weaker rural communities. Furthermore, agricultural intensification could also affect health through increased zoonotic disease risk and increased antimicrobial resistance. The evidence for the health implications of these effects is not specific to Canterbury, but national and international evidence demonstrates that the kinds of changes occurring in Canterbury are likely to have similar effects as elsewhere. Furthermore, although the health effects of these changes are less immediate than those of declining water quality, they could be much more substantial in the long term, so they cannot be ignored.

Land use in Canterbury will continue to change along with the global environment and economy. Care must be taken to ensure land use decisions support economic growth without creating adverse health outcomes.

Introduction

This paper is a 2014 update of a literature review originally performed in 2010. This update has been prepared in response to a request from the Protection Team, Community and Public Health, Canterbury District Health Board (CDHB). This document provides an overview of the literature relevant to water quality and public health in relation to land use change, particularly agricultural intensification. This information is then discussed in the context of the Canterbury region of New Zealand.

Public health and the environment

The World Health Organization defines environmental health as addressing

> *...all the physical, chemical, and biological factors external to a person, and all the related factors impacting behaviours. It encompasses the assessment and control of those environmental factors that can potentially affect health. It is targeted towards preventing disease and creating health-supportive environments (World Health Organization, 2010)*

The World Health Organisation (WHO) Director General emphasises that water quality is a key aspect of environmental health, fundamental to human health, equity, and economic development (Dr Margaret Chan, 2013).

Traditionally the risks to human health from water-related hazards have been viewed in terms of a dose-response relationship between the level of exposure to a particular environmental contaminant and the resulting health effects. Increasingly this definition cannot address the wide ranging environmental health issues that are emerging. Over recent decades ecological considerations have also been given increasing importance as it is recognised that the quality of human life declines along with any decline in the health of natural ecosystems.

> *Over the long term, ecological degradation either directly or indirectly degrades human health and the economy…human health and welfare ultimately rely upon the life support systems and natural resources provided by healthy ecosystems (United States Environmental Protection Agency, 1990)*

Consequently, public health advocates call for in-depth consideration and broader understandings of the links between water quality and human health. Protecting population health together with environmental quality will ensure that optimal health and well-being are safeguarded for current and future generations.

Under the New Zealand Public Health and Disability Act (2000), every District Health Board has the responsibility to:

- "improve, promote and protect the health of people and communities" [s229a)]
- "promote the reduction of adverse social and environmental effects on the health of people and communities" [s23 (1) (h)]

The CDHB partially fulfils this obligation through the participation of its Community and Public Health Division (Protection Team – Environments) in the Resource Management Act process.

The purpose of the Resource Management Act is described in Section 5 of the Act.

1. The purpose of this Act is to promote the sustainable management of natural and physical resources
2. *In this Act, sustainable management means managing the use, development, and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic, and cultural wellbeing and for their health and safety while—*
    a. *Sustaining the potential of natural and physical resources (excluding minerals) to meet the reasonably foreseeable needs of future generations; and*
    b. *Safeguarding the life-supporting capacity of air, water, soil, and ecosystems; and*
    c. *Avoiding, remedying, or mitigating any adverse effects of activities on the environment*

("The Resource Management Act," 1991)

Highlighting evidence, carrying out risk or health impact assessments and establishing the potential public health impacts in relation to notified resource consent applications is a key role for public health practitioners. Brokering evidence for wider audiences remains an on-going challenge.

Methods

This literature search and resultant literature review were originally completed in 2010. The review sought to identify literature that would provide guidance and support for submissions relating to the impact of land use change and agricultural intensification on the Canterbury Plains, with a particular focus on water quality. The evidence presented, both from overseas and from other regions within New Zealand, is considered in the context of the acknowledged complexity and multiple unknowns of hydrology throughout the Canterbury Plains.

The literature review was updated in 2014, although the scope of the update was limited to the effect of agricultural intensification on water quality and subsequent effects on public health.

The original literature search was conducted during 2010. Databases and search engines were searched on multiple occasions [Pubmed, Ovid (covering Medline, EMBASE & PsycINFO), Cochrane Library, Science Direct, Proquest Science databases, Wiley InterScience, EBSCO, Google Scholar and Google] using combinations of the following keywords:

rural	factory farming (cubicle)
agriculture	Canterbury
dairying (dairy)	New Zealand
farming (farm)	global
livestock	environment (al)
land-use, land use/change	impact
intensification (intensive)	ecosystem

A number of papers were identified and the reference lists associated with these studies were also searched and several additional papers were identified. Documents were sourced from the Ministry for the Environment, the Ministry of Agriculture and Forestry, Environment Canterbury, Lincoln University and the electronic resource library previously established by the Healthy Environments Team at Community and Public Health (C&PH). Publication lists from numerous international bodies concerned with agriculture and/or water and/or human health outcomes were also searched for literature relevant to this investigation including the World Health Organization and the United Nations' Food and Agriculture Organization.

The literature search was repeated in early 2014, limited to publications occurring since 2010 and related to water quality and public health. The additional information gathered was incorporated into the existing literature review. Where new topic areas were identified, the scope of the search was increased to include literature published prior to 2010.

Multiple technical reports and other documents were sourced presenting the evidence relating to the degradation of the environment as a result of agriculture and agricultural intensification; there is, however, a dearth of evidence about what this may ultimately mean for the health of humankind and the health of planet Earth itself.

Literature review

What are land cover and land use?

The physical and biological surface of the earth is known as ‘land cover’ and involves describing the Earth in terms of water, different vegetation types, bare soils and any artificial structures. For example, satellite imagery for New Zealand from 1996/1997 and 2001/2002 has resulted in New Zealand’s land surface being classified into 42 different land cover classes (Todd & Kerr, 2009).

‘Land use’ can be described as the human modification of the natural environment to create a modified environment including fields, pastures and settlements. These activities alter the land surface processes such as biogeochemistry, hydrology and biodiversity (Food and Agriculture Organization, 2014).

Alterations to land cover and changes in land use date to prehistory and reflect both the direct and indirect consequences of humans seeking to secure resources essential to life. Although human modification of the Earth’s terrestrial surface has occurred for thousands of years the current rates of change are causing unprecedented impacts on ecosystems (Houghton, 1994).

There can be multiple triggers for land use change including biophysical factors such as droughts or storm events and socio-economic factors including economic crisis or high demand for a particular product (Lambin et al., 2001). Land use change can result in increased sensitivity of the human-environment systems to climatic fluctuation which can ultimately contribute to further land degradation (Dale, 1997).

What is intensification?

Intensification of agriculture is currently characterised by high inputs of capital, labour, high yielding crop varieties, irrigation, increasing mechanisation or the extensive use of technologies such as pesticides and fertilisers relative to land area to ensure increases in food production (The World Bank Agriculture and Rural Development Department, 2009).

Global trends in land use

Over the last 10,000 years, expansion and intensification of agriculture has created huge increases in wealth and subsequently human health and the global human population, although often at the cost of degraded environments (Larsen, 2006). During the last 200 years intensification and population growth has been occurring at unprecedented rates, in large part due to irrigation of farmland (Scully, 2002). This rapid growth in food production has contributed to a dramatic increase in living standards, but has also created concern that the potential environmental impacts could begin to outweigh the benefits.

The World Health Organisation, as part of the Millennium Ecosystem Assessment, prepared a health synthesis entitled ‘Ecosystems and Human Well-being’ (2005a). The report suggests that modern societies have become less aware of their fundamental dependency on nature’s goods and services as the foundation of life and health, with their dependency “displaced in space and time, and therefore poorly recognised”. Furthermore, as global populations increase, health risks are no longer limited to localised exposure to pollutants but also arise from “broader pressures on ecosystems, from depletion and degradation of freshwater resources, to the impacts of global climate change on natural disasters and agricultural production”.

The report highlights various issues including:

- The causal links between environmental change and human health are complex
- Continuing exploitation of ecosystem services together with the general decline of most ecosystems is unsustainable and likely to lead to changes that will be irreversible
- Human societies do benefit from restructuring and managing various ecosystems; although the true cost is increasing with the degradation of 60% of ecosystem services
- Many of those adversely affected by declining ecosystem services are highly vulnerable; this occurs in areas where the need for ecosystem services exceeds supply (e.g. availability of potable water), where there is declining agricultural yield, or where deforestation alters the dynamics of infectious disease transmission
- Demand for livestock products is increasingly met by intensive production systems. Although achieving higher levels of production, these intensive systems pose risks for both ecosystem and human health. The risks are associated with high levels of animal waste, increased pressure on existing systems to provide animal feed (high demand for water and nitrogen fertiliser etc.) and the risk of infectious disease outbreaks such as BSE, SARS and avian influenza
- Inorganic chemical compounds and persistent organic pollutants in food and water pose human health risks. Both natural and human actions can result in the release of toxins which can cause a multiplicity of health effects
- Pharmaceutical products or residues in the environment, which may be released through sewage and solid waste disposal, are an emerging environmental issue. Such wastes can only be partially removed using current processes, posing health risks that are yet to be quantified
- Increases in nitrogen and phosphorus within ecosystems due to land-based human activities are contributing to reduced water quality over much of the globe; fertiliser use has contributed significantly to this problem, posing a threat to human health through impacts on water supplies, toxic algal blooms and the impacts of eutrophication on aquatic creatures and human food sources

(World Health Organization, 2005a)

The livestock sector has emerged as an area of significant growth (within the wider agricultural sector) and currently provides one-third of global human protein intake. Changing patterns of consumption, together with growing populations and increasing incomes are key drivers in the increasing demand for livestock products. The FAO (2006) predicts that global meat production will more than double from the 1999/01 levels of 229 million tonnes to 465 million tonnes in 2050 with demand for milk increasing from 580 to 1043 million tonnes. The increasing demand for agricultural products is likely to continue to drive intensification in Canterbury.

Internationally (The World Bank Agriculture and Rural Development Department, 2009), and to a lesser extent in New Zealand (Macfie, 2014), agricultural intensification in developed countries is moving towards large, indoor “factory farms” where livestock are raised in close confinement at high density in order to produce products such as meat, milk and eggs.

In 2003, the total global water abstraction was 3862 km^3 (Fujiwara, 2012), or about one tenth of total annual global freshwater flow of 37300 km^3 (Dai & Trenberth, 2003). The rate of abstraction has increased rapidly in the last century and this increase is forecast to continue (Figure 1). Irrigation and intensification has caused decreases in water quality around the world, in developed and developing countries, and in arid and humid regions (Adediji, Adewumi, & Ologunorisa, 2011;

Allums, Opsahl, Golladay, Hicks, & Conner, 2012; Jingjing, Luoping, & Ricci, 2012; Struyf et al., 2012; Van Liew, Feng, & Pathak, 2012).

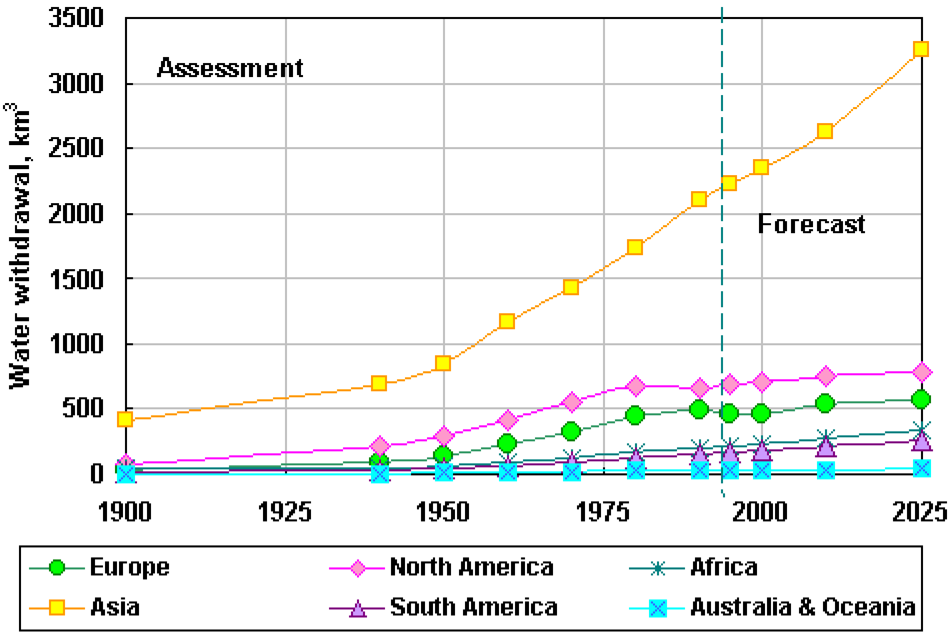

Figure 1. Total annual water abstraction by continent. From (Shiklomanov, 1999).

The intensifying livestock sector has emerged as one of the most significant contributors to our most serious environmental problems. This has become evident at every level – from local to global (Food and Agriculture Organization, 2006). The WHO estimates that 60% of ecosystem services are being used unsustainably or being degraded (World Health Organization, 2005a).

Livestock's contribution to environmental problems is on a massive scale…the impact is so significant that it needs to be addressed with urgency…The environmental impact per unit of livestock production must be cut by half, just to avoid increasing the level of damage beyond its present level (Food and Agriculture Organization, 2006).

Clear evidence exists that agriculture places a serious burden on the environment. Agriculture is the largest consumer of water (United Nations World Water Assessment Programme, 2012b), is the main source of nitrate pollution to ground and surface water and is also the principal source of ammonia pollution (United Nations World Water Assessment Programme, 2012a). The agricultural sector is also a significant contributor to phosphate pollution of waterways (United Nations World Water Assessment Programme, 2012a) and to the release of greenhouse gases (Vermeulen, Campbell, & Ingram, 2012). In Canterbury, and New Zealand as a whole, agriculture accounts for an even greater proportion (Ministry for the Environment, 2012).

Quantifying the impact of agriculture on the environment has proved challenging and is 'not an exact science' (Food and Agriculture Organization, 2003). Much debate exists about the spatial extent of agro-environmental impacts, the current and long-term biophysical effects and the economic consequences of the impact of agriculture. International literature has tended to focus on land degradation and water pollution, although there have been some attempts made to estimate the economic impacts of degradation (e.g. Economy, 2011; Requier-Desjardins, Adhikari, & Sperlich, 2011).

Globally, intensification is inevitable in the face of increasing demand for food from a growing population. By reducing the need to clear new land, intensification may also play a part in the on-going conservation of forests and wetlands. Within New Zealand, intensification could help drive economic growth, which could in turn lead to health gains. The key will be whether or not the environmental degradation which tends to follow from agricultural intensification can be mitigated.

Figure 2. Canterbury region showing towns and major roads (Environment Canterbury, 2010a).

The Canterbury plains

Canterbury is a region of New Zealand's South Island which covers 4.2 million hectares. The western half of the region is mountainous and sparsely populated, while the eastern half contains smaller hills and the 750,000 hectare Canterbury plains (Figure 2). The majority of Canterbury's 566,000 residents live on the northern plains in Christchurch city (Environment Canterbury, 2010a).

History of land use in Canterbury

Prior to the arrival of humans, the Canterbury plains were covered in forest and wetlands. However, by 1840 most forests had been cleared and the plains, primarily covered by short tussock, were able to support a substantial population (Environment Canterbury, 2010a).

Following the arrival of the Canterbury Association, European settlers drained swamps, replaced native tussock with English ryegrass and clover, and established sheep farming on the majority of the plains. Large lease-holders became very wealthy, but many small landholders were little more than subsistence farmers (Wilson, 2012c).

From the 1870s much sheep grazing land was converted to wheat farming. At the same time, races were constructed to bring stock water to dry areas, supporting an increase in stocking rates and beef farming (Wilson, 2012b). Mixed sheep, beef and cropping was the primary land use until World War II, generating considerable wealth, allowing the construction of further infrastructure, and improving the health of the growing population (Wilson, 2012a).

Large scale irrigation on the plains began with the construction of the Rangitātā diversion race in 1945, allowing further intensification of farming between the Rangitātā and Rakaia rivers. Further irrigation continued to be established, especially since the mid-1990s as high export returns for dairying have driven intensification (Wilson, 2012a). Recent growth of irrigation has been especially rapid, with the area consented for surface water irrigation rising from 268,000 hectares in 2005 to 600,000 hectares in 2009, of which approximately 500,000 hectares of consents are actually irrigated (Environment Canterbury, 2011). Excluding non-consumptive use for hydroelectricity generation, the vast majority of water abstracted in Canterbury (89% of surface and groundwater respectively) is used for irrigation (Glubb, Earl-Goulet, & Ettema, 2012). The rapid growth in irrigated area is reflected in a massive increase in dairy cattle population, primarily at the expense of sheep population (Table 1).

Table 1. Livestock populations in the Canterbury Regional Council region in 1994 and 2012. Livestock populations are counted as numbers of adult animals on 30 June. Animals less than one year old are counted separately (Statistics New Zealand).

Animal	1994 Population	2012 Population
Dairy cattle	212,492	1,200,293
Beef cattle	452,832	470,746
Sheep	9,747,488	5,348,010
Deer	258,980	291,783
Pigs	151,681	168,815

Although covering a much smaller area than agricultural land use, industrial and urban residential areas are also significant land uses in Canterbury. Christchurch city has a long history of expansion over the surrounding plains. The city was established in 1851 in the area bordered by Barbadoes, St Asaph, Antigua and Salisbury Streets and Rolleston Avenue. However, by 1855 it had already begun to spread to fill the area between the four avenues. For the next century Christchurch continued its expansion across the plains, driven by unrestrained economic and social forces. The sprawl was eventually halted in 1948 when the Christchurch Metropolitan Planning Committee established land use zones and a green belt, preventing the city from engulfing more of the surrounding rural land. However, the passing of the Resource Management Act in 1990 opened the way for expansion of Christchurch into the 'green belt' (Christchurch City Council, 2005). New low-density suburbs such as Prestons, Waitikiri, Highfield, and Wigram Skies are now appearing on the plains surrounding Christchurch. This process has been accelerated by the evacuation ("red zoning") of some eastern Christchurch suburbs following the 2010-2011 Canterbury earthquakes (Statistics New Zealand, 2014).

Parts of the plains are also being urbanised through the expansion of 'Greater Christchurch' satellite towns such as Rolleston, Lincoln, Pegasus, and Rangiora. These areas were undergoing rapid growth prior to the Canterbury earthquakes, and this growth has been further accelerated by the evacuation of Christchurch suburbs. Between the 2006 and 2013 censuses, the population of Selwyn district increased by 11,000 people (32.6%), and Waimakariri district increased by 7000 people (16.7%), primarily in the satellite towns (Statistics New Zealand, 2014).

Geology and hydrology

Human use of land is influenced by the nature of the land, but the land is also influenced by the use to which it is put – with many land uses causing changes in soil fertility, rates of erosion, or even the shape of the land itself. Less obvious, but just as important, is the way that land use interacts with water. Many human activities rely on a ready supply of high quality water, and many can also affect the quality of water or the way that water flows through the landscape. As such, to understand land use it is important to first understand the geology and hydrology of the landscape.

Geology of the Canterbury plains

The Canterbury plains consist of deep sediment deposits overlaying faulted greywacke bedrock (Figure 3). Marine and estuarine silt deposits from the Cretaceous period to early Pleistocene period (approximately 145 to 5 million years ago) overlay the bedrock and create a relatively impervious and uniform surface 300 to 600 metres below present day ground level. Overlaying the silt layers is a layer of highly porous fluvial gravel deposited by glaciers and rivers during the late Pleistocene. Near the coast, the layers of porous fluvial gravel are interspersed with relatively impervious layers of marine and estuarine sediment deposited as sea levels have fluctuated over the last few million years (Figure 4 and Figure 5). To the east, Banks Peninsula is the remnants of two basaltic volcanoes last active over 7 million years ago (Taylor et al., 1989).

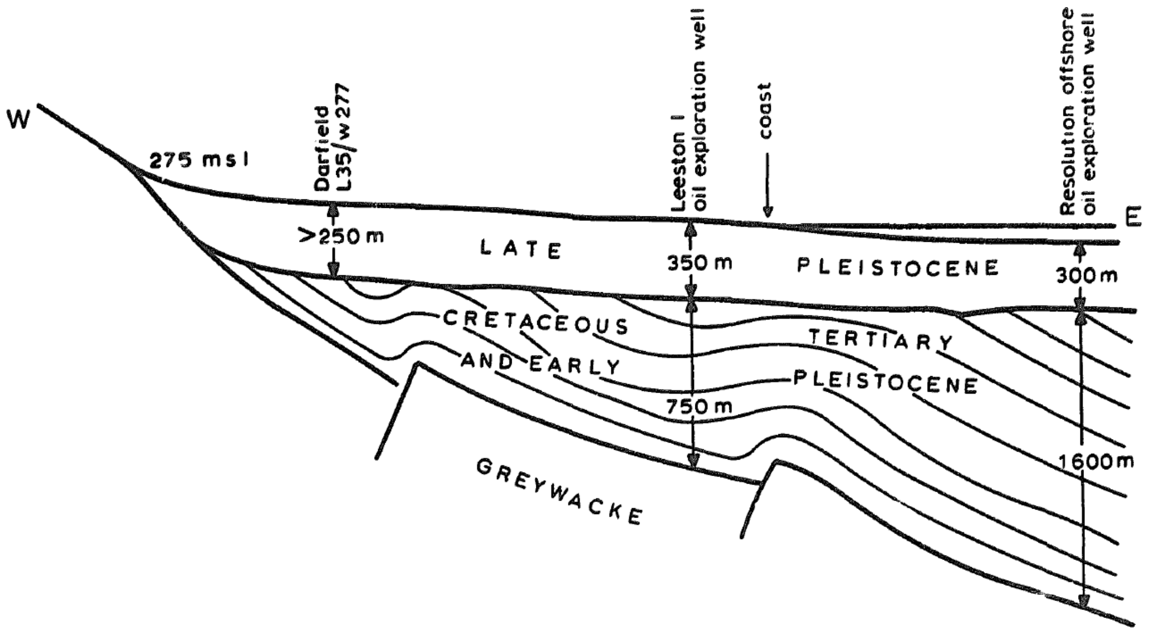

Figure 3. Longitudinal section summarising the geological strata from the Canterbury foothills to the coast. From (Taylor et al., 1989).

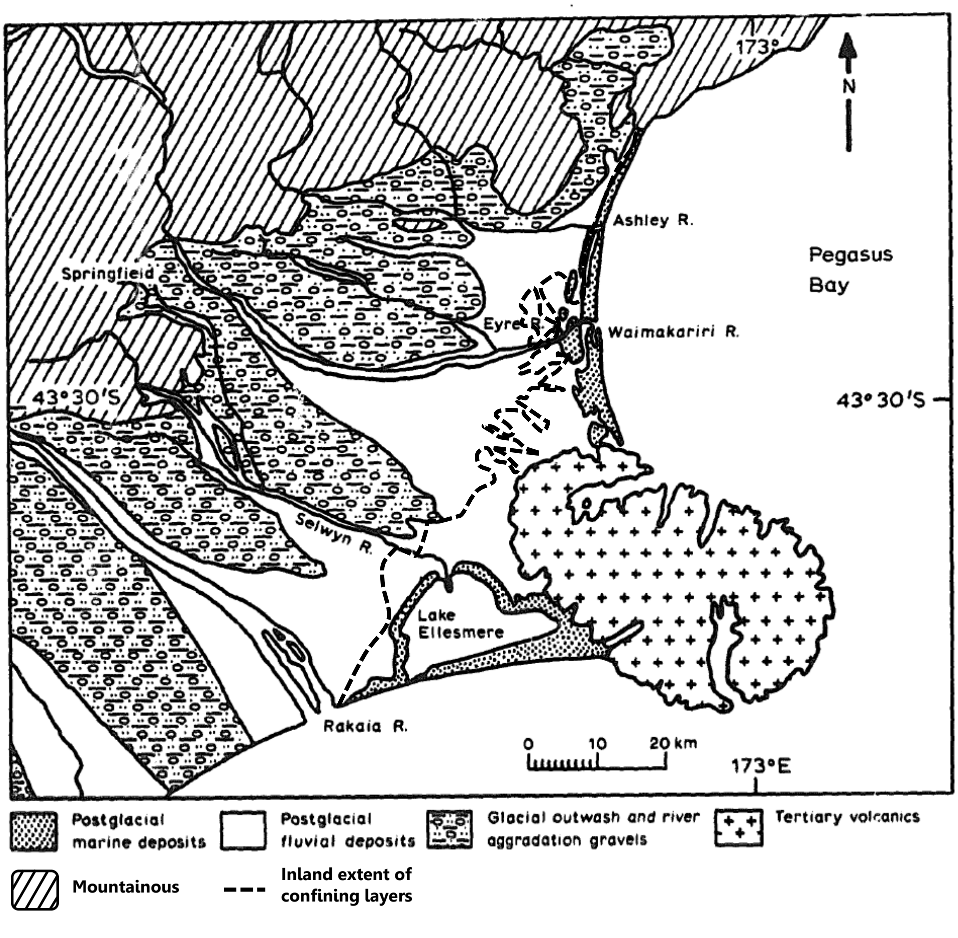

Figure 4. Present day surface deposits on the northern Canterbury plains. Adapted from (Taylor et al., 1989).

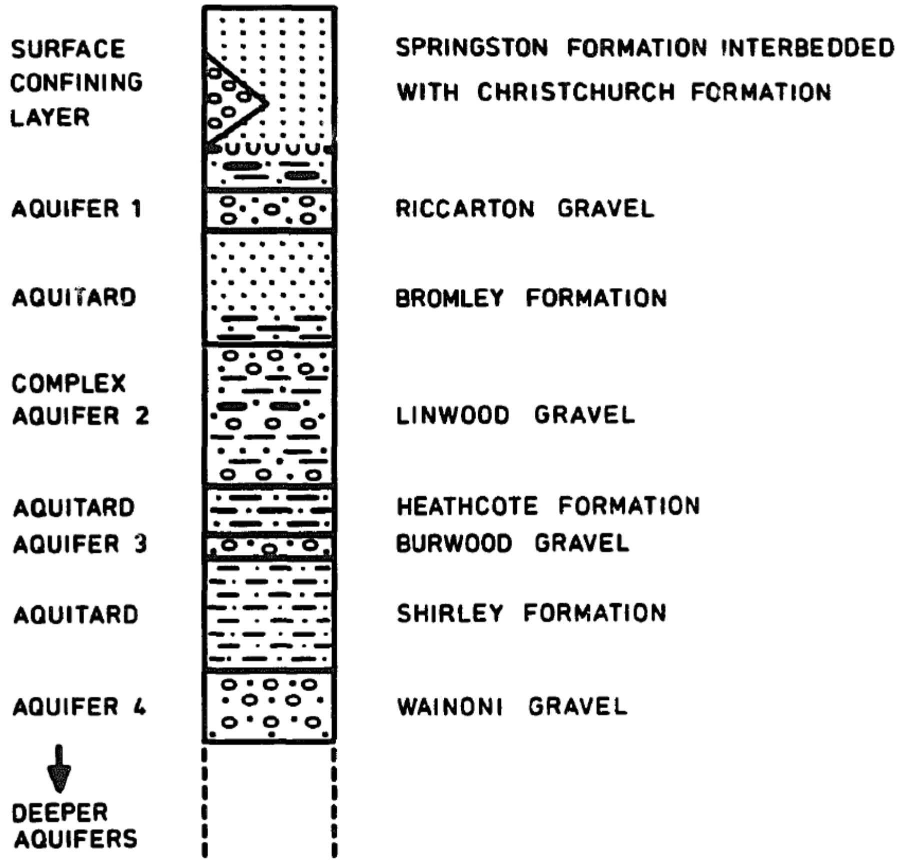

Figure 5. Sequence of aquifers and aquitards underlying Christchurch.

Hydrology of the Canterbury plains

The major hydrological features of the Canterbury plains are a large, contiguous unconfined aquifer in the porous gravel of the inland plains, and a series of confined or semi-confined artesian aquifers in the costal gravel layers interspersed with marine sediment (Stewart, Van der Raaij, Trompetter, & Environmental Monitoring, 2002; Taylor et al., 1989). The details of this system, and the areas which exhibit other characteristics, are described below.

The porous gravel covering the majority of the inland plains is one large unconfined aquifer. However, for convenience the aquifer is divided into five named aquifers based on the predominant sources of recharge and discharge in different areas (Figure 6). The gravel also exhibits substantial anisotropy (more rapid horizontal flow than vertical flow), probably due to buried water courses allowing faster flow than the surrounding sediment, and flow is generally in a south-easterly direction (Figure 6). Deeper water tends to be older than shallower water.

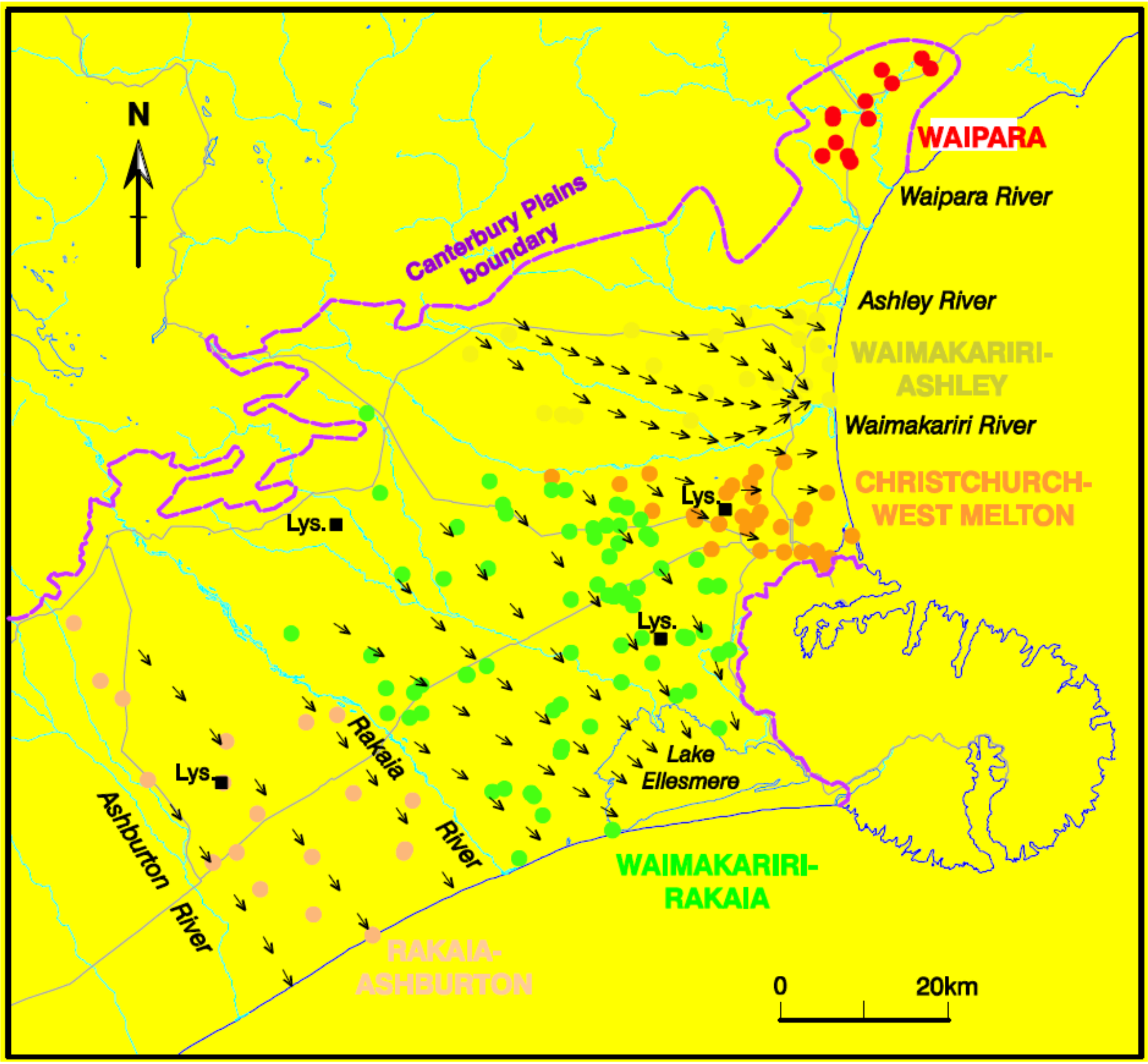

Figure 6. Well sampling sites and inferred shallow groundwater flow directions for each of the five Canterbury Plains aquifers. Inferred deep groundwater flow is generally similar to shallow flow, with the notable exception of some deep flow from the Waimakariri-Ashley aquifer under the Waimakariri River to the Christchurch-West Melton aquifer. From (Stewart et al., 2002)

As the unconfined aquifer nears the coast it encounters the relatively impervious confining layers of marine and estuarine sediment. These aquitard layers inhibit vertical flow between gravel aquifers. Thicker marine sediments off the coast prevent undersea discharge in all but the shallowest aquifer. As such, gravel aquifer water flowing south-east cannot discharge off the coast, and instead creates a pressure gradient with greater artesian head at deeper depths. Water then escapes upwards either via abstraction through wells or via leaky confining layers to the shallowest aquifer to be subsequently discharged through springs (Figure 5).

Recharge for shallow groundwater on the majority of the plains is from low-land rainfall infiltration either directly or via foot-hills rivers. Recharge from rainfall and foot-hills river water carries with it contaminants leached from the surface. The rate of leaching of these contaminants in to groundwater depends on the way the land is used and the soil type. Near the coast, heavy soils reduce the rate of rainwater infiltration and increase the transit time of rainwater, which increases the potential for nutrient uptake by plants and denitrification by soil bacteria, and in turn reduces groundwater contamination. However, reduced infiltration also promotes runoff of contaminated

Box 1. The Christchurch city water supply.

The unique hydrology of the Christchurch aquifers mean that the Christchurch city water supply is high quality and is likely to remain so provided that contaminated infiltration does not increase in the small area of plains known as the "Christchurch-West Melton Groundwater Zone".

The Christchurch city water supply is drawn from artesian wells drilled to at least the first confined aquifer (Riccarton Gravel) or deeper. Recharge for these aquifers is primarily from the Waimakariri river east of Halkett. The Waimakariri is a large river with the majority of its flow emanating from its alpine catchment. As such, it has sufficient volume of clean water to dilute the relatively small quantities of contaminants which enter the river on the plains. The Christchurch aquifer also receives a small amount of rainfall infiltration recharge from the plains between Halkett and Christchurch city. Land use in this "Christchurch-West Melton Groundwater Zone" is subject to additional controls intended to protect the quality and quantity of water entering the Christchurch aquifer (Environment Canterbury, 2009), although some shallow wells in this zone do show evidence of minor contamination from rainfall infiltration (Environmental and Scientific Research, 2014). Once the water reaches the confining layers under Christchurch itself, the pressure of up-gradient groundwater creates an artesian head so the water flows upwards to the surface. This artesian zone is resistant to contamination from the surface because any contaminants which do infiltrate to the first aquifer quickly re-surface in springs rather than continuing to infiltrate to lower levels (Scott & Hanson, 2013).

water to surface waterways during rainfall events. Further inland, light soils allow rapid infiltration which tends to increase groundwater contamination, decrease nutrient uptake by plants and decrease denitrification, but also inhibits runoff of contaminants into surface waterways.

Deep groundwater and shallow groundwater near alpine rivers receive a large recharge contribution from alpine river water infiltration. This water is of very high quality and is likely to remain so provided the quality and flow of alpine rivers is maintained.

In contrast to most of the plains, shallow groundwater in the Christchurch – West Melton aquifer receives the majority of its recharge from Waimakariri river infiltration. Whereas groundwater and most surface water on the plains flows in a South-Easterly direction, the Waimakariri veers eastwards in the vicinity of Halkett. River infiltration downstream of Halkett, instead of flowing along the course of the river, flows South-East towards Christchurch (Figure 7), providing high quality groundwater in shallow aquifers in this area (Box 1). However, there is some mixing with rainfall-infiltration derived water. In South Christchurch, rainfall-infiltration water from the central plains rises upon reaching the Port Hills bedrock, and near Belfast deep groundwater from the Northern plains rises upon reaching the confining layers.

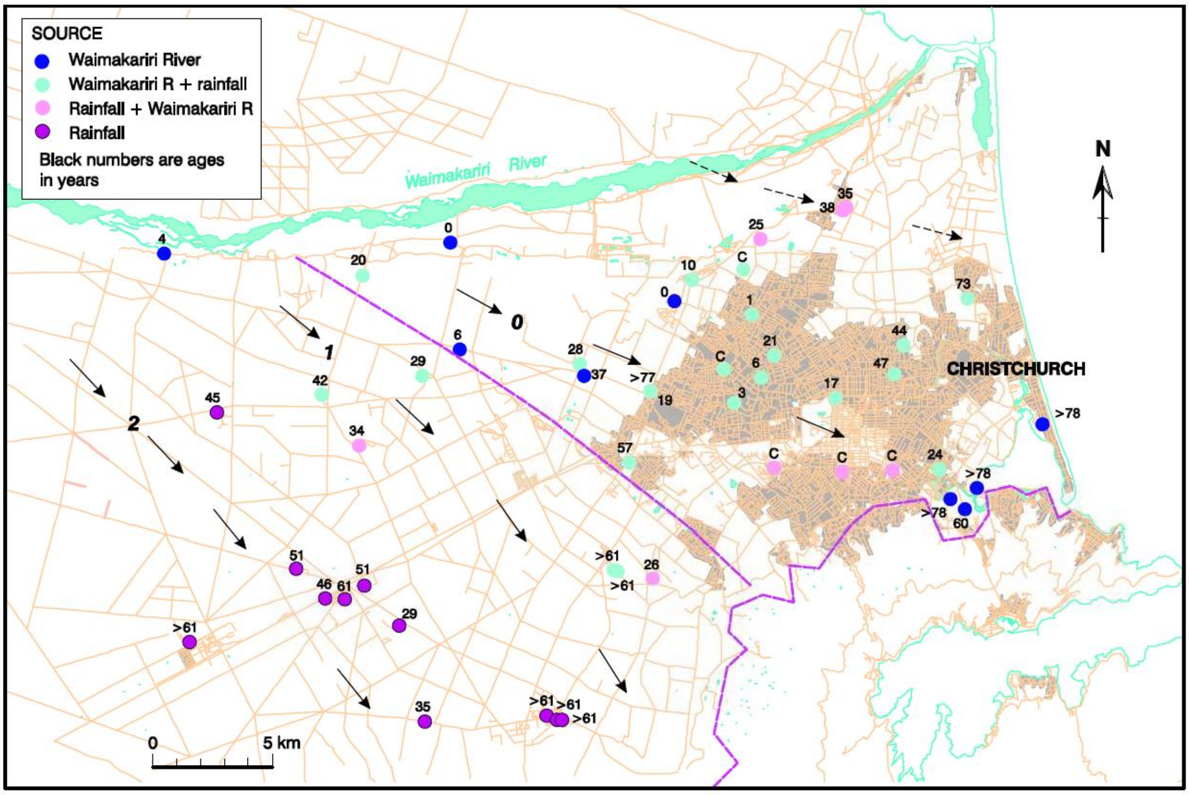

Figure 7. Map of Christchurch West Melton System showing ages in years and sources of groundwaters, based on hydrochemical measurements. The symbol C means the sample contained excess CRCs and no age could be determined. Arrows show flow directions in upper aquifers, dashed arrows flow at deeper levels. (Stewart et al., 2002).

On the plains between the Rakaia and Ashburton rivers, many wells receive a majority of their recharge from infiltration of irrigation water from the Ashburton – Lyndhurst Irrigation Scheme (Stewart et al., 2002). The hydrology of this area is a clear demonstration of the potential for irrigation to increase infiltration through the soil, thereby increasing nutrient leaching. The groundwater in the area down-gradient from the Ashburton – Lyndhurst Irrigation Scheme has some of the highest nitrate concentrations of any groundwater in Canterbury (see Box 4 on page 33).

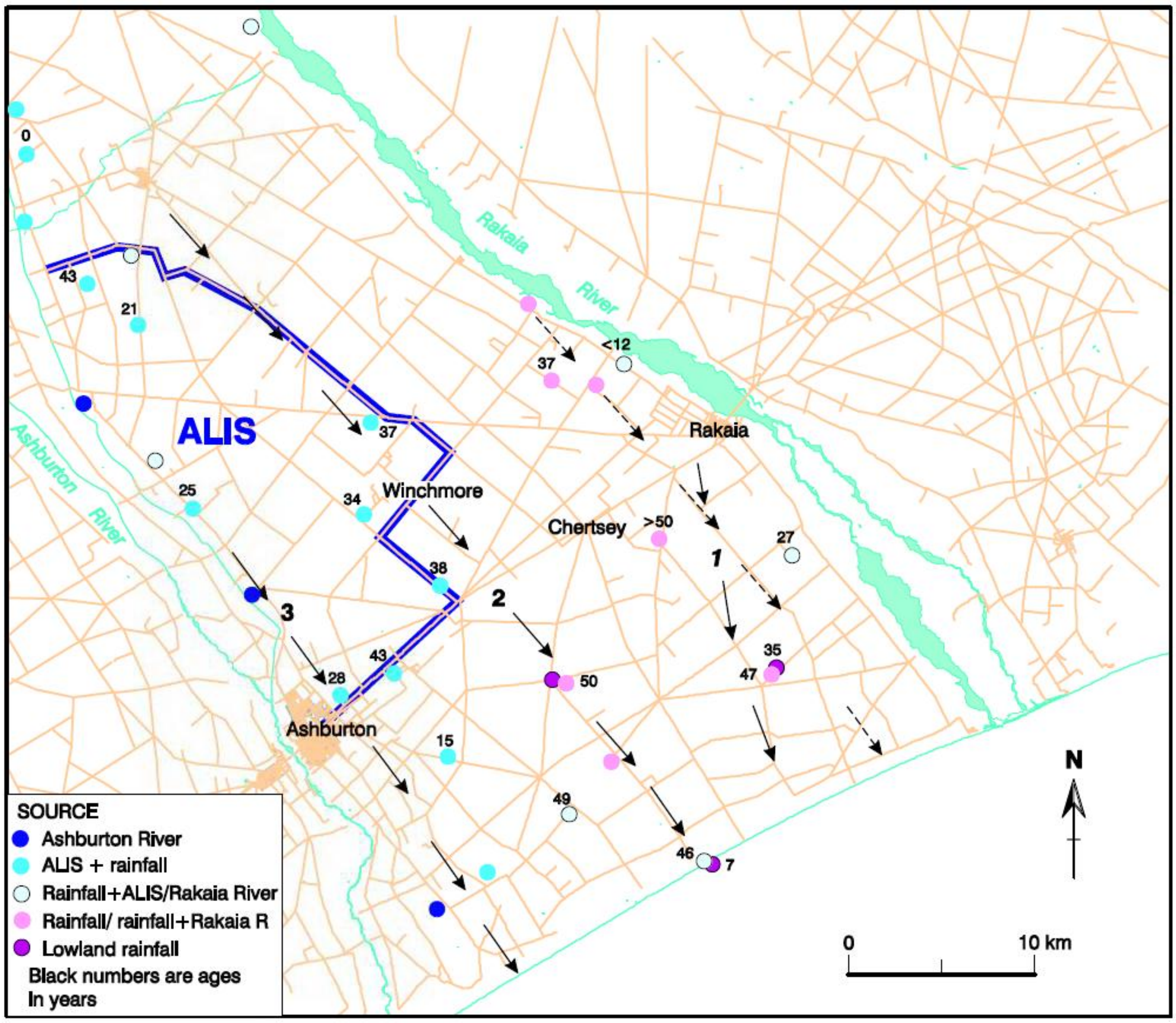

Figure 8. Map of the Rakaia-Ashburton Plains showing ages in years and sources of groundwaters, based on hydrochemical measurements. Arrows show flow directions in upper aquifers, dashed arrows flow at deeper levels. From (Stewart et al., 2002).

The Waipara basin exhibits its own hydrologic characteristics. The majority of the basin consists of small confined or semi-confined shallow aquifers, with larger confined aquifers at deeper levels. The only unconfined areas are found beneath stream channels (Stewart et al., 2002).

Health effects of land use change and agricultural intensification

Zoonotic disease emergence and transmission

Importance for health

Emerging zoonoses have the potential for catastrophic human health impacts due to a lack of immunity to novel infections. Recent examples of emerging zoonotic diseases are avian influenza, including the 1918 pandemic (Taubenberger & Morens, 2006), bovine spongiform encephalitis (BSE), and Nipah virus. Lingering zoonoses, while unlikely to be catastrophic, also have a substantial health burden. Lingering zoonoses having a substantial impact in some parts of the world today include brucellosis, dog rabies and various parasitic diseases (World Health Organization, 2014b). The United States Centers for Disease Control estimate that approximately 75% of emerging human pathogens, and 60% of all human pathogens, are zoonotic (Centers for Disease Control, 2014).

Effect of land use change

Land use change and intensification can increase the risk of emergence and transmission of zoonoses by bringing people into closer contact with livestock, bringing livestock into contact with new environmental conditions, and increasing the potential for disease transmission amongst herds (World Health Organization, 2005b). The effects on zoonotic disease emergence of any one form of land use change in any one area are unpredictable.

Within Canterbury, the most common zoonotic diseases are enteric infections such as campylobacteriosis, cryptosporidiosis, and *E. coli* 0157. The burden of disease from these illnesses is high; Campylobacteriosis is the most frequently notified disease in New Zealand and is responsible for between 18,000 and 36,000 illnesses nationwide annually (Ball, 2006). Although these diseases are frequently of animal origin, they are usually transmitted through drinking water so they are discussed more fully in the "Reduced water quality – Pathogens" section of this document. Nevertheless, it is important to acknowledge that even within Canterbury, rates of zoonotic enteric disease such as campylobacter are higher in areas with more intensive animal farming (Close, Dann, & Ball, 2008; Kaboré et al., 2010).

In addition to persistent pathogens, novel zoonotic disease can also emerge in New Zealand. The Whataroa virus is endemic in native bird populations near the township of Whataroa on the West Coast of the South Island. Whataroa virus is suspected of causing several unseasonal rural influenza-like-illness outbreaks (Tompkins, Paterson, Massey, & Gleeson, 2010).

Zoonotic disease outbreaks are often attributed to increased contact between humans and animals following changes in agricultural practice. A World Health Organisation report (World Health Organization, 2005b) describes an outbreak of Japanese encephalitis in Malaysia in the 1990s that was the result of clearance of forest for pig farming, bringing pigs in to contact with infected flocks of bats. The virus spread from the bats to domestic pigs and subsequently to humans. Similarly, another outbreak of Japanese encephalitis in Sri Lanka was attributed to the promotion of pig grazing for supplementary income amongst small holder rice farmers, thereby exposing the pigs to an irrigated ecosystem which was an ideal breeding ground for disease vectors.

Ilea (Ilea, 2009) notes that small-scale livestock agriculture can cause damage to the environment but does not usually have an impact on the health of those raising the animals, whereas working on an intensive livestock farm is associated with significantly greater health risks. With a focus on ethical concerns this paper presents evidence from recently published studies, demonstrating the effect of intensive livestock production on both global climate and human health.

Highlighted examples include serious respiratory problems among industrial farm animal production (IFAP) workers, including chronic bronchitis and non-allergic asthma (Donham et al., 2007), eye and respiratory health risks due to air emissions from intensive livestock farms for both workers and those living near IFAP (Thu et al., 1997; Wing et al., 2008; Wing & Wolf, 2000).

Donham et al. (2007) sums up the findings of multiple research papers describing the physical health impacts of the confinement environment (particularly in relation to swine operations) on workers. It is evident that at least 25% of these workers suffer from respiratory diseases (including bronchitis, mucus membrane irritation, asthma-like syndrome and, in extreme cases, acute respiratory distress syndrome). Anecdotal reports that a small proportion of workers develop acute respiratory symptoms early in their employment and have to leave the workplace immediately have been substantiated in the work of Dosman et al. (2004). Organic dust toxic syndrome also occurs episodically in over 30% of swine workers and is related to high concentrations of bioaerosols in livestock buildings. Assessments of indoor air quality in these buildings reveal high concentrations of hydrogen sulphide, ammonia, inhalable particulate matter and endotoxin (Doekes et al., 1998).

Increased rates of respiratory symptoms among those living in the vicinity of swine operations have been documented by Thu et al. (1997) when compared with residents in areas of low-density livestock operations. Lower concentration and secretions of immunoglobulin A were found among swine IFAP neighbours (Avery et al., 2004 in Donham et al., 2007). This occurred during periods of moderate to high odour (compared with times of low or no odour) and is suggestive of a stress-mediated response to malodour (Shusterman, 1992).

Impaired mental health has been linked to living in proximity to large-scale IFAP operations. North Carolinians living near IFAP operations had higher self-reported levels of depression and anxiety during odour episodes (Bullers, 2005, Schiffman et al., 1995 in Donham et al., 2007). A study by Thu et al. (1997) did not corroborate these findings although in that study residents were not asked to report their mental state during an actual odour episode.

The Pew Commission Report entitled 'Putting meat on the table: Industrial Farm Animal Production in America' (2008) determined that industrial farming posed 'unacceptable' risks for public health and the environment. The Pew Commission brought together 15 Commissioners and was a project of the Pew Charitable Trusts and the Johns Hopkins Bloomberg School of Public Health. The potential public health effects of IFAP were identified as disease and the transmission of disease, the potential spread of pathogens (animal to human) and mental and social impacts. The report highlights an example of the more serious harm that can occur - an IFAP facility has been implicated in a 2006 *Escherichia coli* multi-state outbreak in which three people died and 200 became ill (Pew Commission on Industrial Farm Animal Production, 2008).

Patz et al. (Patz et al., 2004) presenting the findings of The Working Group on Land Use Change and Disease Emergence cite human-induced land use change as the primary driver for a range of infectious disease outbreaks and emergence events. Land use change is also identified as a modifier of the transmission of endemic infections. Recent research has shown that forest fragmentation, urban sprawl and loss of biodiversity are all linked to an increased risk of Lyme disease in north-eastern regions of the United States (Schmidt & Ostfeld, 2001). Changes in agricultural practices and expansion have been associated with the emergence of Nipah virus in Malaysia and cryptosporidiosis in North America and Europe (Patz et al., 2004)

Antimicrobial resistance

Importance for health

The World Health Organisation (World Health Organization, 2014a) describes antimicrobial resistance as a "global heath security emergency that is arising due to emergence of microorganisms that are no longer treatable because of their resistance to virtually all available antimicrobial treatment options". People hospitalised with antimicrobial infections have approximately twice the death rate of people hospitalised for similar non-resistant infections, take longer to recover, have higher treatment costs, and experience more ongoing effects. Untreatable infections also threaten the effectiveness of other aspects of modern healthcare such as chemotherapy and organ transplants (World Health Organization, 2014a).

Effect of land use change

The high-density confinement of animals in unhygienic conditions may impair growth rates due to the infectious load they may be exposed to. In many parts of the world, antimicrobials are used to control infection in densely housed animals, thereby removing this limitation to growth. Even in developed countries, growth promoting antibiotics are used in very large quantities - over half of antimicrobials used in the United States are fed to farm animals. The utilisation of clinically significant antibiotics in animal agriculture may accelerate the rate of production of antibiotic resistant human isolates. The WHO have called for a reduction in the use of antibiotics in animal agriculture due to the transmission from animals to humans of resistant strains of Salmonella, Campylobacter, Enterococci, and *E. coli* (Horrigan, Lawrence, & Walker, 2002).

There are abundant examples of antibiotic resistant bacteria associated with industrial farm animal production (IFAP) in the literature. The Santa Ana watershed in California contains dense human and cattle populations, including high numbers of IFAP operations, and surface water quality in the watershed routinely exceeds maximum acceptable value (MAV) for nutrients and pathogens by an order of magnitude. Ibekwe et al. (2011) found different patterns of antibiotic resistance occur in E. Coli in reaches dominated by IFAP operations and by urban areas, reflecting different patterns of antibiotic use in these populations. These results demonstrate that any use of antibiotic leads to resistance.

Zhang et al. (2012) found detectable levels of macrolide and sulphonamide antibiotics in water samples from the middle and lower reaches of the East River in Dongjiang, China. The spatial distribution of the various human and animal antibiotics, combined with a principal component analysis, demonstrate that the primary source of sulphonamides was from rural areas with intensive agriculture. As the concentration of sulphonamides (predominantly administered to animals) was much greater than that of macrolides (only administered to humans), this study demonstrated that agriculture was the main source of antibiotics in the East River (Zhang et al., 2012).

Hayes, English, Carr, Wagner, and Joseph (2004) found that multidrug resistance was evident in 53% of *enterococcus faecium* and *enterococcus faecalis* collected from poultry litter. Resistant pathogens were found to persist for at least four months without digestion or formal composting. Tetracycline resistant genes were found to be highly persistent in hog waste lagoons, soils and groundwater where the hog waste had been spread. The detection of tetracycline resistance genes was much higher in wells down-gradient of and closer to the lagoons than more distant wells (Mackie et al., 2006).

Resistant *E. coli* together with resistance genes have been identified in groundwater sources (drinking water sources) near hog farms in North Carolina, Maryland and Iowa. In addition, antibiotics have been found regularly in surface waters at low levels (micrograms per litre range, Sarmah, Meyer, & Boxall, 2006). Sarmah et al. (2006) reported that in some cases as much as 80% of orally administered antibiotics pass through the animals unaltered. There is concern that residues of both antibiotics and antibiotic-resistant bacteria found in bacteria-rich lagoons and in waste spread on fields as a source of fertilizer is easily transportable in both surface and groundwater through both leaching and runoff.

There is some risk that resistant bacteria produced in IFAP could be aerosolised and spread to surrounding areas. The high-volume fans utilised in food animal production facilities result in considerable quantities of particulate matter (<10 µm in size) being moved into the external environment. In the United States, resistant bacteria and antimicrobial drugs have both been detected up to 150m downwind from swine facilities (Gibbs et al., 2006). Campylobacter strains with the same DNA fingerprint as those colonising broilers have also been identified up to 30m downwind of broiler facilities in the United Kingdom (Bull et al., 2006).

Reduced water quality

Intensifying agriculture and expanding urban areas are two land use trends which pose special problems for water quality because they tend to promote non-point source discharge of pollutants. Non-point sources are difficult to manage because there is usually no feasible way to collect and remove contaminants before they enter waterways or groundwater, yet they often represent substantial agricultural impacts on water quality (Journeaux, 2003).

In OECD countries (Journeaux, 2003), including New Zealand (NIWA, 2003), agriculture is the principal land use affecting water quality, despite most countries having tight controls on point source agricultural discharges (e.g. piggeries, cattle feed lots, and chicken batteries). Throughout the OECD, waterways flowing through agricultural regions have greater faecal coliforms, E. coli, pesticide, suspended sediment, turbidity, nitrogen, and phosphorus concentrations than forest streams in the same country (Journeaux, 2003). Furthermore, the risk having of poor water quality is increased by urban and agricultural land use, and within agricultural areas by the intensity of agriculture (Brown & Froemke, 2012; Preston, Alexander, Schwarz, & Crawford, 2011).

Irrigation is a direct pre-cursor to more intensive agricultural systems and there is a direct link between irrigation and increased adverse effects on water bodies. The impact of irrigation on natural ecosystems is multi-dimensional:

- Enables increased land area to be used for agriculture, especially intensive agriculture
- Increased application of contaminants – more fertilisers and pesticides are required to make use of the increased water supply and more effluent is generated with increased stocking rates
- Greater runoff and leaching of contaminants – saturated soil promotes runoff of contaminants to surface waterways and leaching of contaminants to groundwater
- Reduced river flows and groundwater levels – abstraction reduces the remaining environmental water. Reduced river flows can change freshwater ecosystems by reducing the frequency of flushing flows or increasing the length and frequency of very low flow periods. Reduced river flows and groundwater levels can result in less dilution of contaminants that enter these waterways, thereby increasing contaminant concentrations
- Replacement of natural vegetation by pasture or crops can increase surface runoff and decrease replenishment of groundwater aquifers

(Journeaux, 2003).

Within Canterbury, a health impact assessment of the Central Plains Water Scheme was undertaken by the Community and Public Health (CPH) Division of the CDHB in 2008 (Humphrey, Walker, & Porteus, 2008). The Central Plains Water Scheme was a project designed to provide irrigation for 60,000 hectares of Canterbury land, to encourage intensification of farming. The Central Plains Water Trust applied for 64 (originally 55) resource consents to take, use and discharge water, and relating to land use changes, and was granted 31 consents (see http://ecan.govt.nz/get-involved/consent-projects/central-plains-water/Pages/Default.aspx for more information). The CPH health impact assessment examined the impact of the Central Plains Water Scheme on three determinants of health; water quality, regional and local economy and employment (including distribution of wealth), and social connectedness of communities. Overall, the health impact assessment of the Central Plains Water Scheme found that the potential risks to the health of Cantabrians outweighed the probable financial benefits to a few people (Humphrey et al., 2008).

Pathogens

An analysis of the state and trends in water quality of Canterbury's rivers and streams carried out by ECAN identified the increasing contamination of lowland waterways with faecal indicator bacteria as a key water quality theme (Stevenson, Wilks, & Hayward, 2010).

Importance for health

Pathogen contaminated water can result in human infection through drinking water, contact recreation, and consumption of food from contaminated water.

The most important route of infection is through drinking water, especially as contaminated municipal water supplies have the potential to infect many people (Box 2). In New Zealand, a 2006 report used two different methods to estimate that there were between 18,000 and 34,000 cases of campylobacter in New Zealand attributable to drinking contaminated water each year, or an annual incidence of 473 to 894/100,000 population per year (Ball, 2006). Furthermore, the authors noted that these figures were likely to be under-estimates due to low notification rates and considering only the most common pathogen (campylobacter).

The importance of disease caused by drinking contaminated water is further increased by the way this disease burden is distributed. In New Zealand in 1996, water supplies for the most deprived 10% of urban area census meshblocks were 3.76 times more likely to be high risk than were water supplies for the least deprived 10% of urban area meshblocks, primarily due to poor water quality in small rural water supplies and private supplies sourced from shallow groundwater (Hales, Black, Skelly, Salmond, & Weinstein, 2003). Despite the introduction of new drinking water standards in 2005[1], the 2012-13 annual water quality report (Environmental and Scientific Research, 2014) indicates that smaller water supplies (which tend to serve more deprived areas, Hales et al., 2003) still have the highest rates of pathogen contamination, suggesting that more deprived communities still experience greater risk of infection from contaminated drinking water. That is, the poorest communities have the least ability to pay for water treatment, so must instead suffer poorer health. In contrast, the financial benefit of polluting activities primarily accrues to polluters, but the cost of treating water is shared across entire communities. The 2008 Springston gastroenteritis outbreak is a good example of poor water quality imposing a financial cost on a community (Box 3).

Contact recreation also contributes a substantial burden of disease. A study which obtained 750 water samples over 15 months from each of 25 contact recreation sites around New Zealand

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[1] Pre-existing water supplies servicing less than 10,000 people were not required to comply with the updated standard during the 2012-2013 reporting period.

estimated an annual incidence of campylobacter infection attributable to contact recreation at 325/100,000 population (Till, McBride, Ball, Taylor, & Pyle, 2008). Again, because this study considered only campylobacter, this figure is likely to be an under-estimate of the overall burden of disease from contact recreation in contaminated water.

The literature search conducted for this review found no assessment of the New Zealand burden of disease from consuming food gathered from pathogen contaminated water. However, Shuval (2003) estimated that globally the consumption of shellfish resulted in 40 million cases per year of hepatitis A and E, with 40,000 deaths and 40,000 cases of long term disability. In Australia, contaminated shellfish were responsible for over 2000 notified viral and bacterial infections between 1990 and 2000, and likely a great deal more non-notified or undiagnosed infections (Sumner & Ross, 2002). In New Zealand, a joint agency research project found frequent, repeated norovirus contamination of shellfish in Tauranga harbour during a one year monitoring period (Schole et al., 2009). Given New Zealand's strong seafood consuming traditions, it seems likely that there is a substantial burden of disease from consuming pathogen contaminated shellfish in New Zealand.

Box 2. The Darfield and Walkerton outbreaks

A heavy rainfall event in Walkerton, Ontario, in May 2000 caused increased contamination of the shallow groundwater source for the town water supply, overwhelming water treatment (chlorination) and contaminating the water supply with *E. Coli.* (0157:H7) and C*ampylobacter Jejuni*. Insufficient water quality monitoring by the water supply operator meant that the contamination and lack of treatment were not detected until spate of gastrointestinal illnesses alerted the Medical Officer of Health. A boil water notice was issued, but not before the town had been exposed to contaminated water for days. As a result, an estimated 2300 people became ill (almost half the town's population), while 65 people were hospitalised, 27 developed haemolytic uraemic syndrome[1] and seven died. The subsequent inquiry attributed the tragedy to negligence by the water supply operator and a laissez faire regulatory framework for environmental and water quality issues (O'Connor, 2002).

Another heavy rainfall event in Darfield, Canterbury, in August 2012 caused increased contamination of the shallow groundwater source water for the town water supply, overwhelming water treatment (chlorination) and contaminating the water supply with *Campylobacter*. Insufficient water quality monitoring by the District Council meant that the contamination and lack of treatment were not detected until spate of gastrointestinal illnesses alerted the Medical Officer of Health. A boil water notice was issued by the District Council, but not before the town had been exposed to contaminated water for days (Community and Public Health, 2012). As a result, an estimated 1283 people (two thirds of the town's population) became ill (Sheerin, Bartholomew, & Brunton, 2013). The striking similarities between the Walkerton and Darfield outbreaks illustrate the potential for a massive outbreak with high mortality to occur in Canterbury. It is fortunate that the campylobacter strain responsible the Darfield outbreak did not have the same mortality rate as the *E. Coli.* (0157:H7) responsible for the Walkerton outbreak.

[2] Haemolytic uraemic syndrome – a serious and potentially life-threatening complication characterised by renal failure (due to toxins released by the bacteria attacking the kidneys) it is believed that 5 – 10% of those suffering *E. coli* O157:H7 will develop haemolytic uraemic syndrome

Box 3. Springston campylobacter outbreak

A boil water notice for the Springston water supply was issued on 7 March 2008 in response to a series of transgressions of the microbial drinking water standards. Numerous notified cases of gastroenteritis in the town led to an outbreak investigation which revealed 42 cases and a relative risk of gastroenteritis of 16 in Springston residents compared to controls of Selwyn District Council staff. In response to the outbreak, the Selwyn District Council commissioned a deeper well. There were no further transgressions of the drinking water standards or notified cases of gastroenteritis in the four weeks between the new well coming online and the publication of the report (Community and Public Health, 2008). The need to commission a new well is a clear example of a financial cost from lower quality source water.

The use of contaminated water for irrigation of food crops also presents a health risk. The presence in irrigation water of heterotrophic bacteria and faecal coliforms at concentrations higher than the EU recreational water standard results in high levels of pathogens on vegetables irrigated with the polluted water, necessitating washing with clean water prior to sale to avoid adverse public health effects (Puto, 2012). Similar problems have occurred with the use of reclaimed waste water to irrigate crops near Melbourne (Barker-Reid, Harper, & Hamilton, 2010).

Effect of land use change

There is ample evidence that agricultural intensification and growth of urban areas each contribute to increasing pathogen loads in ground and surface water. These problems are compounded by the poor pathogen attenuation of the gravelly soil and gravel aquifers underlying most of Canterbury.

The distances that dangerous concentrations of pathogens can travel in groundwater depends on the pathogen type, the rate of entry into the groundwater, and the soil type. Pang et al. (Pang, 2009) studied pathogen removal rates in Canterbury and found that viral pathogens can travel much further than bacteria or other cellular pathogens. They comment on soil types in relation to pathogen removal:

> *The subsurface media that are most effective at microbial removal, thus suitable for effluent land disposal and would require smaller setback distances, are allophanic soils, pumice sand, fine sand, and highly weathered aquifer rocks, while the least effective subsurface media are structured clayey soils, stony soils, coarse gravel aquifers, fractured rocks, and karst limestones (Pang, 2009)*

In Canterbury, almost all groundwater travels in coarse gravel aquifers with low pathogen removal, so bacteria and viruses which reach the aquifer can travel a long way (Pang, 2009). The surface soil layer is often of silty or sandy loam so some attenuation of pathogens can occur while surface water is leaching through the surface soil layer (Close, Noonan, Hector, & Bright, 2010). However, sub-surface pathogen sources, such as leaky septic tanks or infiltration around bore heads, are not subject to surface-layer attenuation so pose an especially high risk for ground water quality.

On Canterbury dairy farms, surface layer attenuation may well be enough to remove bacterial risk under well managed irrigation. Close et al. (2008) showed that the use of border-strip irrigation resulted in increased concentrations of both *E. coli* and *Campylobacter* in nearby groundwater. However, in a four-year follow up study, carefully controlled travelling spray irrigation (55mm application) did not produce any detectable decrease in the bacterial quality of downgradient groundwater at 1.5-10 metres depth (Close et al., 2010), although there was substantial contamination with the application of 80mm of irrigation, either as irrigation or due to irrigation plus an unexpected rain event. Both studies were carried out on silty loam over sandy-gravelly loam soil (with moderate bacterial removal rates).

The results of the Close et al. (2010) study are encouraging, but require careful interpretation due to the limited range of pathogens measured, the possibility of different surface layer transmissibility in different areas, and the potential for weather events to apply greater surface water loads than irrigation alone. The transmissibility of enteroviruses and bacteriophages in soil is much greater than that of bacteria (Pang, 2009), raising the possibility that these pathogens could still leach into groundwater under spray irrigation. The authors (Close et al., 2010) acknowledge this limitation but argue that there is no evidence that animal faeces contain human pathogenic viruses, so it is appropriate to consider only bacterial transmission under irrigation. Furthermore, although travelling spray irrigators allow for careful application of irrigation water, Close et al. (2010) found substantial contamination when unexpected rain fell the day after irrigation application. A rain event close to 80mm total will also likely cause substantial pathogen infiltration, although no such rainfall events occurred during the four years of the study (Close et al., 2010). Perhaps most importantly, the spray irrigation study (Close et al., 2010) was carried out at one location (the Lincoln University Dairy Farm), so it is likely that some areas on the plains, especially old river channels, will experience greater pathogen infiltration due to lesser surface soil depth or greater surface soil pathogen transmissibility than the study site.

The pathogen removal properties of Canterbury soils could be challenged if industrial farming practices become more common. Some aspects of industrial dairy farming, such as herd homes, are already in use in Waikato (Macfie, 2014). The Pew Foundation in the United States commissioned a report on the health effects of industrial farm animal production (IFAP) in that country (Pew Commission on Industrial Farm Animal Production, 2008). IFAP systems raise large numbers of animals in confinement buildings (similar to herd homes), increasing production but also increasing rates of within-herd disease transmission and potentially increasing the impact on humans either by animals directly or their waste. Effluent, harbouring pathogens and chemical contaminants, is generally untreated (or minimally so) before being sprayed on fields as a fertilizer where it may contaminate air, water and soils. Furthermore, to minimise transport to market costs, IFAP systems have tended to be concentrated in areas where they have the potential to impact on human population centres. The report notes that IFAP systems do not aspire to balancing the natural productivity of the land in order to provide feed crops but instead import feed and medicines to ensure that animals are market-ready in the shortest possible time. Animals, as well as their waste, are concentrated and consequently exceed the capacity of the land to either provide feed or absorb waste (Pew Commission on Industrial Farm Animal Production, 2008).

Epidemiological studies which demonstrate an association between agricultural intensity, especially cattle density, and enteric disease notifications highlight the importance of the intensity of farming in determining population rates of enteric disease. Within Canterbury, a study based on EpiSurv notification data found that the risk of *Campylobacter* notifications (adjusted for age and socioeconomic status) was higher in regions with dairying and major irrigation schemes compared with two control group regions; control group 1 (CG1) = areas with dairying but without major irrigation, and control group 2 (CG2) = the rest of Canterbury. The relative risks for campylobacteriosis were 1.51 (95% confidence interval 1.31-1.75) and 1.51 (95% confidence interval 1.33-1.72) for CG1 and CG2 respectively (Close et al., 2008). The fact that risk is elevated by irrigated dairy farming but not dryland dairy farming suggests that irrigation contributes to the disease burden and implicates contaminated groundwater as the disease vector. However, it is possible that this elevated risk may decrease as border-strip irrigation systems are replaced with more efficient spray irrigation systems which cause less bacterial leaching (Close et al., 2010).

A similar epidemiological study from Quebec (Kaboré et al., 2010) demonstrated that, in children less than 4 years old, the incidence of potentially zoonotic enteric diseases overall and campylobacteriosis, salmonellosis, and giardiasis individually were positively correlated with local cattle density, but not with local poultry, swine, or small ruminant density (RR Q4/Q1 = 1.79, 95% CI = 1.11–2.87), except in the case of giardiasis which was also correlated with swine density. These results suggest that, in Quebec, the presence of high intensity cattle operations cause an additional 97 cases of enteric disease per 100,000 children per year.

There is some evidence to suggest that loading of soil and benthic sediments with faecal bacteria can create a reservoir of these bacteria which can, in turn, be released back into the waterway. Chase et al. (2012) found that sediment and nearby soil in a seasonal river in Florida contained very high levels of faecal indicator bacterial (FIB, ~100 times the recreational water standard). These FIB acted as a reservoir which replenished water FIB during low flows resulting in water FIB levels increasing during low flow, despite typical concentrations during high flow (Chase et al., 2012). This observation is likely to be relevant for Canterbury's lowland streams and lakes, which often experience periods of low- or no-flow during summer.

Although agriculture is the primary land use on the Canterbury plains, urban areas are expanding and are also having increasing influence on Canterbury's water quality. Compared to agricultural activities, the literature contains much less evidence about the effect of urban areas on water quality. However, the literature search for this review identified a few studies.

Within Canterbury, the most urban spring fed streams (Avon/Ōtākaro, Heathcote/Opawaho and Styx/Purakaunui) are amongst the most highly contaminated on the plains. Sources of faecal contamination in these streams include storm water runoff, sewage overflows, and water birds (Stevenson et al., 2010).

Epidemiological evidence also demonstrates that urban impervious land cover is associated with contamination of waterways with pathogens. A study of lakes in Quebec showed that the proportion of land cover made up of agricultural (odds ratio of 11.0) or impervious surfaces (odds ratio of 5.2) were positively associated with the risk of lakes having faecal coliform concentrations above the recreational water quality limit (Turgeon et al., 2013). In light of the high burden of disease of contact recreation due to campylobacter (Till et al., 2008), and the probability that there are more cases of contact with water near urban areas with greater population, it seems likely that runoff from impervious surfaces in urban areas contributes substantially to the overall burden of waterborne disease.

Urban expansion may also result in direct contamination of water with human waste. A study in Belgrade provides a good example of a situation similar to Canterbury. Growth of suburban Belgrade beyond the reaches of its municipal sewerage and water supply systems has resulted in a proliferation of private septic tanks and small scale groundwater supply schemes in the outer suburbs (as has occurred in Darfield). Sewage infiltration from septic tanks has caused over half of the drinking water supplies to fail to meet drinking water chemical or microbial standards, so citizens must implement their own in-home treatment systems or suffer high rates of waterborne disease (Petkovic et al., 2011). As discussed earlier in this section, the high pathogen transmissibility of the sub-surface gravels of the Canterbury plains (Pang, 2009) make underground discharges from leaky septic tanks an especially important issue.

Chemical contamination

Importance for health

The Ministry for the Environment guide for drinking water supply operators lists many potential sources of dangerous chemical contaminants (Nokes, 2008). However, there are no estimates of the burden of disease of chemical contamination of water in New Zealand, and very high uncertainty about this issue world-wide (Prüss-Ustün, Vickers, Haefliger, & Bertollini, 2011). The National Survey of Pesticides in Groundwater 2010 (Close et al., 2010) did not detect any pesticide contamination in five shallow wells in the Canterbury region, but heavy metals and hydrocarbons are increasingly being detected in surface waterways in urban catchments in Canterbury (Stevenson et al., 2010).

Chemical contaminants could also increase the risk of pathogen or nutrient contamination of water by altering the ecology of waterways. The presence of inorganic fertiliser and atrazine broadleaf herbicide in freshwater environments can lead to increased survival of faecal indicator organisms (Staley, Rohr, & Harwood, 2010), suggesting that these agrichemicals could increase the health risk from a given pathogen input to a waterway.

Chemical contaminants, like pathogens, may also be concentrated by filter feeding shellfish. Again, there are no reliable estimates of the burden of disease from the consumption of chemically contaminated shellfish, but shellfish in New Zealand have relatively low levels of contamination from agricultural chemicals (Scobie, Buckland, Ellis, & Salter, 1999). Similarly, shellfish in Canterbury have low levels of heavy metal contamination, generally less than 10% of the MAV (McMurtrie, 2012).

Effect of land use change

Chemical contaminants in water have received much less research attention than have nutrients or pathogens, so the overall effects of land use change are less certain.

Globally, an increase in production of 1% is, on average, accompanied by a 1.8% increase in pesticide use per hectare and a 0.8% increase in pesticide use per unit output (Schreinemachers & Tipraqsa, 2012). In developing countries, increasing pesticide use with intensification has led to drinking water pesticide concentrations above the MAV set by the United States Environmental Protection Agency (e.g. Motekar, 2011). The increase in pesticide use with intensification does tend to be smaller in higher income countries such as New Zealand, but intensification in Canterbury could still be expected to result in increased pesticide application (Schreinemachers & Tipraqsa, 2012). Nevertheless, there were no detectable pesticide residues in the six wells tested in Canterbury as part of the most recent national survey of pesticides in groundwater (Close & Skinner, 2011).

Heavy metals and hydrocarbons are increasingly being detected in surface waterways in urban catchments in Canterbury, leading to their inclusion in routine monitoring for these waterways (Stevenson et al., 2010). An increase in impervious surface area within a catchment is associated with an increased risk of contamination of waterways with heavy metals and hydrocarbons (Arnold Jr & Gibbons, 1996). As seafood gathering occurs in urban streams and their estuaries, it is also possible that increasing contamination of these waterways could lead to increased incidence of shellfish poisoning.

Nutrients

Human activity has had, and is having, a significant impact on the global nitrogen cycle. Before the industrial revolution and the green revolution, the rate of nitrogen supply to land was balanced by the rate of bacterial nitrogen fixation. Today, more atmospheric nitrogen is fixed into reactive forms by human activity than by all other terrestrial processes combined (Howarth, 2008).

Townsend et al. (2003) propose that there are largely positive consequences for net public health of the changing N cycle at lower levels. The consequences become increasingly negative as the creation and use of fixed N climbs (Figure 9).

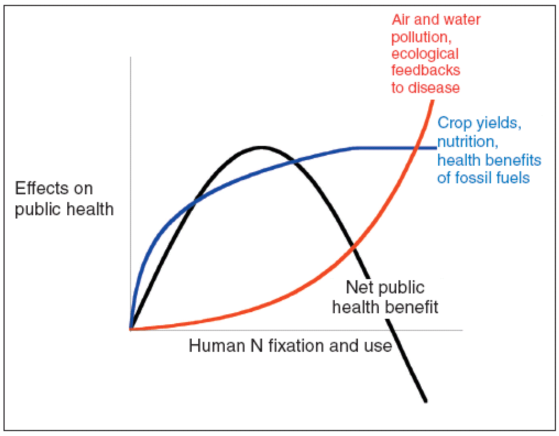

Figure 9. Conceptual model of the overall net public health effects of increasing human fixation and use of atmospheric N_2. As human N_2 fixation and use rises, the positive effects of N on crop growth, food production, and overall nutrition peak (and potentially begin to decrease). In contrast, rising N inputs create exponential increases in losses of N to air and water, deposition to natural ecosystems, and potential feedbacks between rising N availability and the ecological dynamics of environmentally harboured diseases. Source: Townsend et al. (2003)

Importance for health

The literature identifies three main issues related to the health effects of nutrient enrichment of water: algal blooms, methaemoglobinaemia, and cancer. Perhaps more importantly, nitrate is also a good indicator of the likelihood of water also being contaminated with other more toxic material, such as pesticides, pharmaceuticals, and effluent (van Grinsven, Rabl, & de Kok, 2010).

Algal blooms

High concentrations of nutrients in a river, lake, estuary or bay can increase the frequency of cyanotoxic algal blooms (Chorus & Bartram, 1999), such as have occurred recently in rivers and lakes on the Canterbury plains (Environment Canterbury, 2012). These blooms are excessive growths of algae such as cyanobacteria which may release cyanotoxins poisonous to humans and other mammals. The frequency of toxic cyanobacteria blooms has been increasing worldwide since at least the 1950s, leading the WHO to acknowledge cyanobacteria as an emerging health issue (Chorus & Bartram, 1999).

Humans may be exposed to cyanotoxins in drinking water, through the consumption of contaminated shellfish, or during contact with water such as when swimming. The specific effects of cyanotoxins from a particular bloom are hard to predict because different blooms release different toxins or may release no toxins at all, and different people may have different responses (Paerl & Paul, 2012). However, common reactions to cyanotoxin exposure include asthma, eye irritations, rashes, blistering around the mouth and nose and gastrointestinal disorders including abdominal pain, cramps and diarrhoea (Environment Canterbury, 2012).

Methaemoglobinaemia

Methaemoglobinaemia, or blue baby syndrome, is caused by the oxidation of haemoglobin to methaemoglobinaemia by nitrite ions. Under normal circumstances, less than 2% of human haemoglobin exists as methaemoglobin. However, in infants, especially those experiencing enteric infection, ingestion of nitrates leads to bacterial conversion of nitrate to nitrite in the gut, which can increase blood nitrite concentrations and increase the proportion of methaemoglobin. At 10% of haemoglobin as methaemoglobin, symptoms appear such as blue-grey skin, chocolate coloured blood, and excessive crying in infants. 60% methaemoglobin can result in coma or death (Fewtrell, 2004).

Concerns about high concentrations of nitrates in drinking water increasing the risk of methaemoglobinaemia in infants were first raised in 1945 after a series of 114 cases including 14 deaths in Minnesota. Since that time, there were no documented cases of methaemoglobinaemia in the United States in infants fed with breast milk or with formula made up using water with nitrate concentrations below the United States MAV of 45 mg/l, but at least 87 cases who had consumed formula made up using water with nitrate concentrations greater than the MAV (Fan & Steinberg, 1996). These data appear consistent with recent research in Romania which found no cases of methaemoglobinaemia in breast fed infants, and found that the incidence was greatest amongst infants < 40 months old who had been fed infant formula or vegetables prepared using water high in nitrates (Curşeu, Sîrbu, Popa, & Ionutas, 2011). However, cases of methaemoglobinaemia in infants not exposed to water above the MAV for nitrates have been observed in countries other than the United States, and Fan et al. (1996) suggest that these cases may be explained by exposure to water with poor bacterial quality. Moreover, a review of literature conducted on behalf of the WHO found no exposure-response relationship between drinking water nitrates and methaemoglobinaemia and an apparent decline in the incidence of methaemoglobinaemia since 1990 despite an increasing trend for drinking water nitrate. They concluded that nitrate was only one of a number of nutritional, environmental, and behavioural co-factors which may be associated with methaemoglobinaemia (Fewtrell, 2004). Despite the uncertainty about the mechanism, it seems appropriate to limit the consumption of water with nitrate concentrations above the MAV by infants less than 4 months old.

Cancer

The World Health Organisation's International Agency for Research on Cancer classifies nitrates as "probably carcinogenic" when ingested "under conditions that result in endogenous nitrosation" (International Agency for Research on Cancer, 2014). This classification is based on numerous studies which find an association between nitrates and cancer, including some that show a dose response effect, a plausible biological mechanism, and in the case of thyroid cancer a time trend consistent with rising concentrations of nitrates preceding development of thyroid cancer (see subsequent paragraphs for details). When viewed together, these observations suggest that drinking water nitrates may cause an increase in the risk of some cancers, particularly thyroid cancer. Nevertheless, absolute mortality from thyroid cancer has remained constant at around 0.4 deaths per 100,000 population per year, and thyroid cancer remains relatively rare with 5.1 new cases per 100,000 population in 2010 (Ministry of Health, 2013).

There is good evidence for an association between water supply nitrate concentration and the incidence of some cancers. In particular, three large studies from the United States provide high-quality evidence. The Iowa Women's Health Study assessed the association between water supply nitrate concentration and risk of various cancers in 21,977 Iowa women. They found positive associations between nitrate concentration and the incidence of bladder and ovarian cancers, but negative associations with uterine and rectal cancers (Weyer et al., 2001). Another study following the same cohort for a longer time found a positive dose-response association between water supply and dietary nitrate concentrations and incidence of thyroid cancer ($p = 0.02$ for trend) (Ward et al., 2010). Compared with women who had never used water with nitrate-N concentration >5mg/L (22 mg/L total nitrate) as their primary drinking water supply, women who had used water with >5mg/L nitrate-N for the primary drinking water supply for at least 5 years were 2.6 times more likely to develop thyroid cancer (95% CI 1.09 to 6.19) (Ward et al., 2010). A similar positive dose-response association between dietary nitrate intake and thyroid cancer was then found in males in a much larger cohort, the 490,194 person National Institutes of Health-American Association of Retired Persons Diet and Health Study (Kilfoy et al., 2011).

It should be noted that the ecological design of the studies cited above can demonstrate an association, but is not well suited to demonstrating causation. In particular, these designs are prone to confounding, where the potential risk factors and the outcome variables both vary as the result of some other unmeasured factor. Known potential confounders can be controlled or adjusted for statistically[3], but it is not possible to rule out an effect of unknown confounders. Unfortunately, the combination of ethical concerns and the nature of the nitrate-cancer relationship (long-term exposure to low-level nitrate contamination, combined with low incidence rates for most cancers) make it practically impossible to design a randomised trial to overcome the problem of confounding. As such, it is unlikely that any more reliable evidence will be forthcoming. Nevertheless, there are some other considerations which, when viewed as a whole, strengthen the case that high concentrations of drinking water nitrates increase the risk of developing thyroid cancer.

A potential biological mechanism for a carcinogenic effect of drinking water nitrates is well described. Ingested nitrates are endogenously nitrosated to form N-nitroso compounds, which are potent carcinogens associated with thyroid tumours in animal models (Bogovski & Bogovski, 1981; Lijinsky, 1991). Furthermore, most nitrate containing vegetables also contain compounds, such as vitamin C, which inhibit conversion of nitrate to N-nitroso compounds (Lundberg, Weitzberg, Cole, & Benjamin, 2004). These inhibitors could explain why, in contrast to consumption of nitrates in water, heavy consumption of nitrate containing vegetables has not been found to be associated with increased incidence of cancer.

Changes in drinking water nitrate exposure and rates of thyroid cancer have been shown to be parallel over time. Rates of thyroid cancer are rising worldwide (Kilfoy et al., 2011), with the United States (Kilfoy et al., 2009) and New Zealand (Ministry of Health, 2002) age adjusted rates of thyroid cancer having doubled since the 1970s. These rates are consistent with rising concentrations of nitrate in drinking water since the green revolution beginning in the 1940s, which for some

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[3] Kilfoy et al. (2011) adjust for: age, gender, BMI, level of education, smoking status, vitamin C intake, vitamin E intake, folate intake, beta-carotene intake and red meat intake.

Ward et al. (2010) control for gender (female only) and adjust for: age, BMI, level of education, smoking status, vitamin C intake, level of physical activity, and location of residence (farm, other rural, or town of population <1000, 1000–2499, 2500–9999, or ≥10,000).

Weyer et al. (2001) control for gender (female only) and adjust for: age, BMI, level of education, smoking status, vitamin C intake, vitamin E intake, level of physical activity, fruit and vegetable consumption, total energy intake, and waist hip ratio populations would have resulted in 30 years of exposure to elevated nitrate in drinking water by the 1970s.

Van Grinsven et al. (2010) conducted a social cost-benefit analysis of nitrogen fertiliser use in the EU in terms of additional food production and colon cancer attributable to nitrogen contamination of drinking water. They estimated a cost of 0.7 euro per kg of nitrate-N leaching (van Grinsven et al., 2010). They used an odds-ratio of 2.0 for high meat consumers (above the median for meat consumption) developing colon cancer in the face of prolonged exposure to drinking water nitrate concentration greater than half the MAV (based on previous research in the EU; De Roos, Ward, Lynch, & Cantor, 2003), and obtained information about exposure to nitrates from EU data. They found that overall nitrogen fertiliser use was beneficial in the EU, but that in some parts of Europe the social cost of colon cancer attributable to nitrogen fertiliser use approached the value of the additional production created. They argue that, as their analysis does not account for the full range of costs, it is likely that there would be a net benefit to society of reducing nitrogen fertiliser use in areas where the production increase from nitrogen fertiliser is low.

Other smaller studies have found positive associations between water supply nitrate concentration and increased risk of macular degeneration (Klein, McElroy, Klein, Howard, & Lee, 2013) non-Hodgkin lymphoma (when atrazine is also present; Rhoades et al., 2013), oesophageal cancer (Liao, Chen, Chiu, & Yang, 2013), gastric cancer (Chiu et al., 2012), colon cancer (Chiu, Tsai, Chen, Wu, & Yang, 2011) and fatal childhood brain tumour (Weng, Tsai, Wu, Sung, & Yang, 2011). However, many of these latter studies were post-hoc analyses and may suffer from confirmation and publication bias, so it would be inappropriate to draw any firm conclusions based on these data.

Effect of land use change

Agricultural fertiliser use is the primary source of nitrogen contamination of water, and a major source of phosphorus contamination, especially in developed countries where point source discharges are well controlled (Howarth, 2008). At a global scale, agricultural fertiliser use and concomitant environmental nitrogen and phosphorus losses have increased steadily since the 1960s as agriculture has intensified (Tilman et al., 2001). The same pattern is evident in New Zealand and Canterbury, where expanding irrigation infrastructure allows productive use of increasing amounts of fertiliser (Gabel, Wehr, & Truhn, 2012; Glubb et al., 2012; MacLeod & Moller, 2006; Parliamentary Commissioner for the Environment, 2004, 2013). Nitrogen inputs are further increased by the importation of supplementary feed such as palm kernel extract, which have increased dramatically since 2000 (Peel, 2013). These land use changes have led to the majority of the Canterbury plains being at risk of having groundwater which does not meet the drinking water standards for nitrates (Figure 10 and Box 4)(Bidwell, Lilburne, Thorley, & Scott, 2009; Parliamentary Commissioner for the Environment, 2013).

Concerns about declining water quality and availability in Canterbury led to the development of the Canterbury Water Management Strategy. An assessment of nitrate discharge to groundwater from agricultural land use on the Canterbury Plains was prepared for the Strategy (Bidwell et al., 2009). The key findings of this report were:

- Nitrate discharge from agricultural land use on the Canterbury Plains has the potential to cause levels of nitrate concentration in shallow groundwater (<20 metres below the groundwater surface), at some localities, that exceed the New Zealand MAV of 50 mg/l
- Groundwater quality generally improves with depth below the groundwater surface because of dispersive mixing with high quality groundwater from river recharge

- Reduction of nitrate discharge by improving existing practices has the most effect on the availability of drinking water from shallow groundwater (Bidwell et al., 2009)

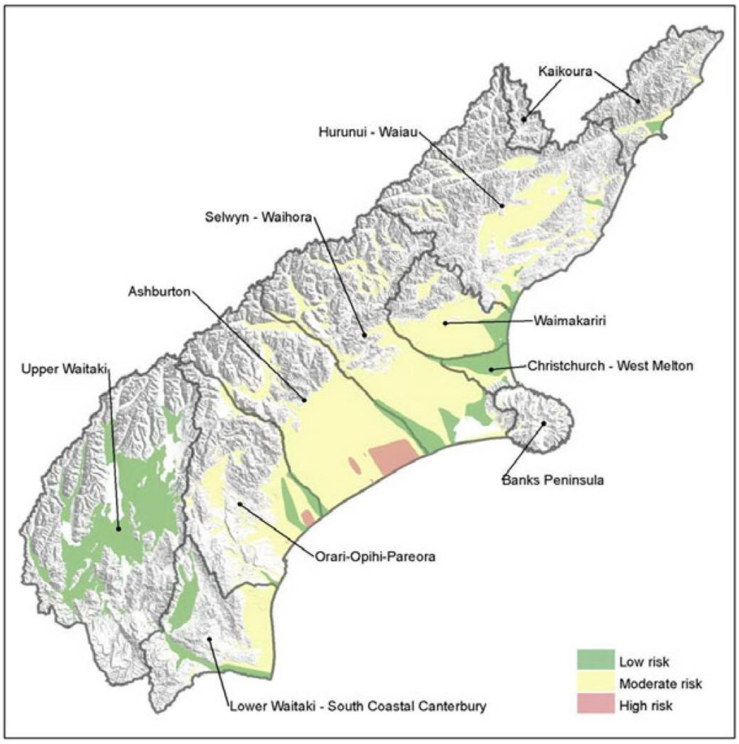

Figure 10. Map of the Canterbury region showing areas at low-, medium- and high-risk of nitrate concentrations exceeding the MAV in shallow groundwater. Risk was estimated based on the proportion of wells that had a nitrate reading above the MAV any time in the last 20 years. Areas where no wells had ever tested over the MAV were classified as low risk. Areas where most wells had tested over the MAV at least once were classified as high risk. Other areas, including areas with insufficient data (few wells or wells infrequently monitored), were classified as moderate risk. Low risk areas were associated with low-intensity agricultural use, predominantly river recharge of ground water, or very low dissolved oxygen in shallow groundwater (indicating good conditions for de-nitrification). From (Scott & Hanson, 2013)

In addition to groundwater, increasing nitrogen and phosphorus inputs have caused eutrophication of lowland Canterbury lakes such as Te Waihora / Lake Ellesmere (Box 5)(Kitto, 2010; Te Runanga o Ngai Tahu & Environment Canterbury, 2011). A report on lake water quality in New Zealand prepared by the National Institute of Water and Atmospheric Research (NIWA) for the Ministry for the Environment estimated that 32% of New Zealand lakes are eutrophic (NIWA, 2010). The report concluded that:

> *Pastoral land use in NZ is associated with eutrophication and ecological deterioration. Furthermore, the condition of some lakes currently in good condition is declining, likely as a result of nutrient enrichment from livestock farming (NIWA, 2010)*

Similarly, a very comprehensive study by Clapcott et al. (2012) used New Zealand nationwide databases of water quality, water ecology indicators, and land use to study the relationships between land use water quality throughout New Zealand. Land use and agricultural intensity explained 60% of the variation in nitrogen concentrations, but only 40% of variation in clarity and

Box 4. Nitrogen contamination of groundwater in Canterbury: Ashburton-Rakaia Plains

In the early 2000s ECAN identified groundwater nitrate concentrations approaching the maximum acceptable value (MAV) for groundwater in test wells in three areas near Ashburton (Hayward & Hanson, 2004). Detailed surveys of all wells (including private wells) near the contaminated wells revealed three nitrate plumes with nitrate concentrations exceeding the MAV. Two of the nitrate plumes were immediately down-gradient of meat-works effluent disposal areas, and affected a relatively small area (0.5 by 2km and 2 by 13km respectively). However, the plume in the Chertsey-Dorie area was not associated with any known point discharge and covered a large area (approximately 12 by 35km or 42,000 hectares). Overall, 155 wells were sampled in the three investigations, 39 wells (25%) had nitrate concentrations exceeding the MAV, and 124 (80%) had concentrations above half the MAV. Of the wells which exceeded the MAV, 24 were used for domestic drinking water supply and represented a direct health risk. Subsequent investigations south of the Ashburton river revealed another plume near Tinwald that also could not be explained by point source discharges (Hanson & Abraham, 2010).

The lack of a point discharge for the Chertsey-Dorie and Tinwald plumes suggest that the nitrogen contamination is the result of non-point source agricultural discharge. A non-point source of nitrogen is also supported by the profile of concentrations, gradually increasing towards the coast and decreasing at greater depths (Figure 11).

Excessive nitrogen contamination from agricultural non-point sources is particularly concerning because, due to groundwater lag of almost 50 years, there has been substantial agricultural intensification and corresponding increase in nitrogen leaching since water in the Chertsey-Dorie wells entered the groundwater. These concerns are supported by ongoing investigation south of the Ashburton river which shows that elevated groundwater nitrate concentrations in this area, including the Tinwald plume, have continued to increases since they were identified in 2004 (Hanson & Abraham, 2010). Hayward and Hanson write:

> *Estimated ages of groundwater from two wells located at Dorie, 35 m and 93 m deep, were 35 and 47 years respectively (Stewart et al., 2002). This indicates that nitrates in some of the groundwater may have originated from land activities occurring over 40 years ago, before extensive areas of irrigation were developed. Therefore, if current land uses are more intensive with greater potential to leach nitrates, concentrations in groundwater can be expected to increase further (Hayward & Hanson, 2004)*

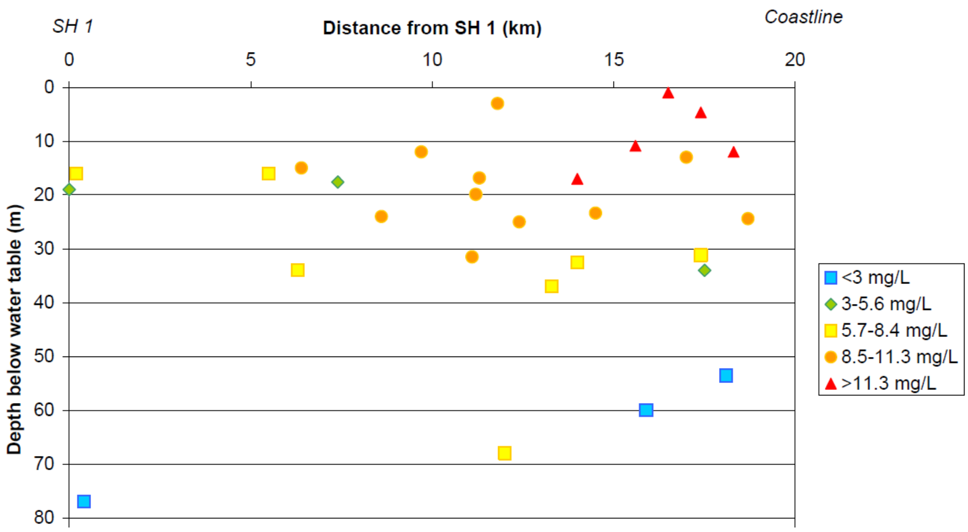

Figure 11. Range of nitrate nitrogen concentrations in a cross section of SH1 to the coast in the Chertsey-dorie area compared to the depth of well screens below the water table. Figure excludes wells immediately adjacent to the Rakaia River, which receive recharge from river water. From (Hayward & Hanson, 2004).

20% of variation of dissolved reactive phosphorus (DRP)[4], reinforcing the primary role of agricultural intensity in determining nitrogen contamination.

The most prominent land use change in Canterbury in recent years has been the replacement of low intensity sheep farming and forestry with intensive dairying, leading to a massive increase in cattle populations (see the *History of land use in Canterbury* section on page 10 of this report). Cattle populations are closely associated with increasing nitrogen losses to surface and groundwater (Johnes, Moss, & Phillips, 1996). This is dramatically illustrated in a study of Xiangyang province in China where a population of 600,000 cattle causes more nitrogen pollution than the population of 6 million humans (Chen & Hongjuan, 2012). In contrast, Canterbury has 1,800,000 cattle and only 566,000 humans (Statistics New Zealand). Continued growth in the number of cattle without additional mitigation measures will inevitably increase the rate of nutrient entry into waterways and groundwater.

The Parliamentary Commissioner for the Environment (2013) published a report summarising the results of an extensive modelling exercise which predicts the changes in overall nutrient outputs from diffuse sources in response to changing market forces, land uses, and mitigation practices. The modelling exercise demonstrated that although mitigation may be able to prevent an increase in nutrient losses from the predicted intensified dairying (i.e. more cows on the same land), it cannot compensate for land use change to dairying from other uses (i.e. cows on new land). As such, the nutrient quality of fresh water is predicted to decline as long as dairy conversions continue to occur.

Trends such as increasing intensification and growth of cattle numbers have led Journeaux (2003) to conclude that agriculture will struggle to achieve the New Zealand environmental threshold value for nitrate (3.2mg/l NO3) even with full uptake of best management practices. He suggests that maintaining production at or above current levels could necessitate, in some instances, the classification of some water bodies as being “for agricultural purposes”.

These conclusions are supported by Daigneault et al. (Daigneault, Greenhalgh, & Lennox, 2011) who used economic equilibrium modelling to predict the effects of increasing the water supply in the Hurunui District to be able to supply up to 44.3 thousand hectares of irrigation (up from 22.3 thousand hectares). They considered a range of fertiliser application strategies, nutrient management practices, and environmental regulation regimes to achieve nutrient pollution mitigation, but found that if environmental outputs were constrained to current levels, then there would only be about two thousand additional hectares of irrigation rather than the 20,000 hectares available, because most landowners would be unable to change their land use to make use of irrigation without also increasing nutrient leaching and greenhouse gas output.

Although nitrogen is the nutrient of primary concern in Canterbury, many surface waterways are permanently high in nitrogen so productivity, and eutrophication, of the waterways are phosphorus limited (Meredith, Croucher, Lavender, & Smith, 2006). There are documented cases of phosphorus contamination of groundwater in China, Brazil, and the USA as a result of importation of phosphorus containing stock feed (Kleinman et al., 2011). The 12-fold increase in the importation of palm kernel expeller for stock feed in New Zealand between 2004 and 2010 (Peel, 2013) further increases the nitrogen loading on waterways but also raises the possibility of increased phosphorus losses and eutrophication.

[4] Variablility of DRP was better explained (60%) when environmental factors were added to the model, especially rainfall and flow variability. Addition of water temperature, air temperature, and flow variability also improve the explanation of clarity variability.

Box 5. Nitrogen enrichment of Te Waihora.

Te Waihora – also known as Lake Ellesmere – is a large shallow lake just south of Banks Peninsula. Sediment records demonstrate that the lake formed c. 7500 years ago when the Kaitorere Spit joined with Banks Peninsula, trapping the flow of the Selwyn and other lowland rivers to form a freshwater lake with low nutrient concentration. Since that time, the lake has been predominantly freshwater, but has experienced two brackish episodes when the Waimakariri river has changed course to drain in to the lake, scoured a large lake outlet, then reverted to an outlet north of Banks Peninsula leaving a large outlet through which seawater enters the lake (Kitto, 2010). The lake has historically been an important source of food for Ngai Tahu, and was originally known as Te Kete Ika o Rākaihautū, or the fish basket of Rākaihautū, the chief who landed the first Waitaha waka on the South Island (Prudham, 2004; Te Runanga o Ngai Tahu & Environment Canterbury, 2011).

Today the lake is brackish, shallow, eutrophic, and turbid. As the lake has been highly modified since European settlement, it is likely that its current state is the result of numerous factors, many related to land use change. The agricultural and residential use of the land surrounding the lake has led to the need for artificial control of the lake level by digging a channel, which allows ingress of sea water in a similar fashion to historic scouring by the Waimakariri. Intensifying agriculture since the 1970s has led to a turbid, hyper-eutrophic state. Additionally, the 1968 'Wahine' storm destroyed the lake's macrophyte beds, and high turbidity and low water clarity have prevented their regeneration, leading to a phytoplankton dominated ecology with little macrophyte growth. Restoration of the lake would require less frequent openings to the sea and reduced nutrient input (Kitto, 2010).

Availability of water

Importance for health

Increased abstraction primarily affects health via reducing ground water levels and surface water flow rates. Both these effects can reduce water quality, especially in surface water where reduced flow can concentrate contaminants, increase water temperature, decrease the frequency of flushing flows, and increase saltwater intrusion (Nilsson & Renöfält, 2008). They also have the potential to reduce availability of water for other uses, such as downstream abstractions for domestic, agricultural, or industrial supply (Taylor et al., 2013). This effect could decrease health by imposing additional costs on downstream users, which is likely to have an adverse effect on the health of those unable to afford these costs.

Effect of land use change

Within Canterbury, the primary effects of increased abstraction of groundwater will be reduced flows in spring-fed streams and increased abstraction costs for all water users. However, increased abstraction from surface waterways, especially lowland and foothill fed streams, is likely to have greater effects, with reduced downstream flows and lower water supply reliability, especially during low-flow periods. For both surface and ground water, abstraction limits must be carefully managed to avoid depleting the resource (Morgan, Bidwell, Bright, McIndoe, & Robb, 2002).

Internationally, there are many examples of increased abstraction depleting aquifers and either increasing the expense of obtaining groundwater, or requiring utilisation of a completely different water resource (e.g. Ahmed, Sultan, Wahr, & Yan, 2013; Steward et al., 2013; Voss et al., 2013). However, in these examples the aquifers have very slow recharge so are effectively a finite resource, often referred to as fossil water. In contrast, the Canterbury plains aquifers have relatively small volume and rapid recharge, so have greater capacity to recover from poor management (Morgan et al., 2002).

Climate change may also affect groundwater availability, either by affecting recharge or by altering demand for groundwater. In Canterbury, climate change is likely to cause a slight decrease in groundwater recharge as the climate of eastern New Zealand dries, although this may be offset by an increase in recharge due to increased frequency of heavy rain events and increased irrigation using surface water (NIWA National Climate Centre, 2008; Taylor et al., 2013). However, modelling studies suggest that local scale effects are less predictable than regional effects, with the effect of climate change on water availability predicted to vary markedly from watershed to watershed, even for apparently similar watersheds in close proximity to each other (Van Liew et al., 2012). Similarly, climate change is likely to make rainfall less predictable (NIWA National Climate Centre, 2008; Taylor et al., 2013). Although the effects of increased abstraction of groundwater are expected to far outweigh any changes in recharge due to climate change, decreased predictability will increase demand for irrigation water, so the primary effect of climate change on groundwater availability will likely be to increase abstraction thereby reducing groundwater levels (Taylor et al., 2013).

Ways to support water quality

An assessment of nitrate discharge to groundwater from agricultural land use on the Canterbury Plains prepared for the Steering Group of the Canterbury Water Management Strategy (Bidwell et al., 2009) found that reduction of nitrate discharge by improving existing practices would be the most effective way to improve the availability of drinking water from shallow groundwater in Canterbury. Various measures to reduce the impact of farming practices on water quality, and especially nutrient concentrations, have been proposed in the literature. Most are more or less effective for treating surface runoff or point discharges, so improve the quality of surface water (Gabel et al., 2012; Laitos & Ruckriegle, 2013), but may be less effective at reducing the leaching of nitrates into groundwater. There has also been considerable research attention devoted to identifying the most appropriate regulatory or management structures to encourage farmers to implement mitigation measures (e.g. incentives, penalties, and markets). However, even the best (and most expensive) mitigation will not be sufficient to prevent further deterioration of water quality in the face of the expected rates of dairy conversions and dairy intensification; the Parliamentary Commissioner for the Environment in her recent report (Parliamentary Commissioner for the Environment, 2013) found that implementation of the most effective mitigation measures available today would be sufficient only to offset the effects of intensification of existing dairy farms, not the effects of converting land to dairying from other uses.

Treating effluent and contaminated runoff

There is an abundance of research examining the effectiveness of various designs of constructed wetland or vegetative filter strips for removing nitrogen, phosphorus, and pathogens from water directed though them (e.g. Barber & Quinn, 2012; Carrer et al., 2011; Douglas-Mankin & Okoren, 2011; Lee, Maniquiz, Choi, Jeong, & Kim, 2013; Morand et al., 2011; Winston, Hunt Iii, Osmond, Lord, & Woodward, 2011). When left as unmanaged wetlands or vegetated areas, these systems are generally only moderately effective. In some cases net sequestration of nutrients occurs during low flows and summer weather, but net release of nutrients occurs during high flows and winter (Barber & Quinn, 2012; Lee et al., 2013; Winston et al., 2011). In other cases removal of nutrients is dominated by leaching to groundwater (Douglas-Mankin & Okoren, 2011). This release of nutrients has led some authors to conclude that constructed wetlands would have to be prohibitively large to be effective (Barber & Quinn, 2012). However, other research has demonstrated that the nutrient removal performance of constructed wetland can be greatly improved if some mechanism is included for harvesting biomass, which effectively removes sequestered nutrients from the system and can be re-cycled as either fertiliser or stock feed (Carrer et al., 2011; Lee et al., 2013; Morand et al., 2011). As such, the design of constructed wetlands for nutrient removal should consider incorporating biomass removal into the ongoing management of the wetland.

Constructed wetland management could potentially be improved if the wetland manager was able to continuously monitor the performance of the wetland. Owen et al. (2012) used automated real time monitoring to continuously record water quality in demonstration catchments in Britain. Such monitoring could also be used to improve the understanding of contamination processes, but could also be used for active monitoring to identify and address breaches as they occur.

Although constructed wetlands may be effective at removing nutrients from water which passes through them, they can have no effect on nutrients which leach directly into the groundwater through the pasture where fertiliser or effluent is applied. However, IFAP facilities, in contrast to their potential negative effects on disease transmission and antimicrobial resistance, do have the positive effect of allowing easy collection of effluent for treatment before discharge or application to fields. Such systems, if properly regulated, have the potential to allow very intensive agriculture with little effect on water quality (Macfie, 2014; Morand et al., 2011).

Conventional systems for effluent collection and disposal, usually employed in Canterbury dairy sheds, involve collecting effluent during milking and spreading on pasture via spray irrigation. Unfortunately, such systems can only collect the small fraction of effluent as the cows spend most of their time on pasture. Furthermore, if more effluent were collected, there would be potential for over-application of effluent on to land causing excessive nutrient leaching, especially when animals are fed with imported supplementary feed (Beck, 2013). However, Morand et al. (2011) demonstrated a high-throughput treatment system that uses screening, vermifiltration, and macrophyte digestion to remove 95% of suspended solids, chemical oxygen demand, and nitrogen; 75% of other nutrients; and 99.99% of microbial indicators. Nutrients were able to be removed as vermicast and macrophyte biomass, so could easily be transported to distant fields to avoid over-application in nearby fields. Biomass from the macrophyte *Azolla caroliniana* was successfully used as animal feed without further treatment, and microbial removal was sufficient to allow treated water to be re-used directly for flushing effluent from the animal shed. The ability to re-use water for flushing allowed frequent flushing (6 flushes/day), which in turn reduced ponding of effluent beneath the shed, and reduced atmospheric ammonia within the shed from 25ppm to 8ppm, reducing odour and indicating reduced atmospheric nitrogen emissions.

Preventing nitrogen leaching

The most effective means to avoid excessive nitrogen leaching is to avoid excessive nitrogen and water application. The use of either resin- or sulphur-coated controlled release nitrogen fertiliser compared to standard nitrogen fertiliser for US maize crops (Zhao, Dong, Zhang, & Liu, 2013) or New

Zealand rye grass dairy pasture (Bishop, 2010) results in lower peak nitrogen release, increased nitrogen uptake, increased photosynthesis, increased grain yields, and decreased nitrogen leaching. When combined with carefully controlled irrigation, careful application of controlled release fertiliser is an effective means of achieving adequate nitrogen application to achieve high production without excessive nitrogen leaching. However, Zhao et al. (2013) note that there is poor uptake of controlled release fertiliser amongst American maize farmers due to its high cost. The literature search conducted for this review found no information on the uptake of these fertilisers in New Zealand. Furthermore, careful fertiliser use cannot help to mitigate nitrogen leaching from cow urine patches, which are responsible for over half of total nitrogen leaching even with high rates of fertiliser application (Silva, Cameron, Di, & Hendry, 1999; Williams & Haynes, 1994), especially when cattle are fed imported supplementary feed (Beck, 2013; Peel, 2013).

One technology which shows promise for reducing nitrogen leaching from urine patches is chemical nitrification inhibitors, such as dicyandiamide (DCD, AgResearch, 2012). When urea in urine enters the soil it is quickly converted into ammonium (NH_4^+), which is positively charged so binds to soil particles and is not easily leached. However, over time nitrification by soil bacteria converts the ammonium to nitrate (NO_3^-), which is negatively charged so does not bind to soil and leaches rapidly (Williams & Haynes, 1994). DCD slows the conversion of ammonia to nitrate so helps to retain nitrogen in the top layer of soil where it can be utilised by plants. DCD is an effective mitigation technology (AgResearch, 2012), but a recent controversy about traces of DCD found in milk is likely to make its use unfeasible on dairy farms (Shuttleworth, 2013).

A biological approach to removing nitrogen at the soil level was developed by Fujiwara et al. (2012) who argue that, as it is not possible to collect diffuse pollutants such as nitrates and pharmaceuticals in their original form, research should instead focus on diffuse countermeasures which transform pollutants into a form which is easier to capture. Their team has developed special nitrate sequestering catch crop solutions for agriculture in Japan, which allow the removal of excess nitrates from the soil following harvest of nitrate intensive crops (Fujiwara, 2012). However, such solutions may be impractical for pastoral agriculture in New Zealand, where animals graze perennial pasture directly rather than crops being harvested and subsequently used as animal feed. Furthermore, they rely on using the soil as a temporary nitrogen reservoir, which may be unrealistic in the porous gravel soils of the Canterbury Plains. Without the use of chemical nitrification inhibitors, or a practical biological solution, it may be that the only way to reduce nitrogen leaching from cow urine patches is to reduce production.

Whole farm management plans

Environmental whole farm management plans have been suggested as a way to ensure farmers and financiers are aware of the environmental and financial benefits of implementing mitigation measures. These plans are the result of a whole farm modelling exercise which builds on nutrient management plans to optimise nutrient inputs, nutrient management practices, stocking rates, and farming practices, and capital constraints to minimise environmental load and maximise profit. Preparing a plan requires specialist consultant expertise, and is often supported by a regional council. The benefit of whole farm plans compared to implementing predefined best management practices, a nutrient management plan or an environmental farm plan is that other systems, beyond fertiliser and effluent management, are also optimised to achieve environmental and economic benefits. The inclusion of economic optimisation is essential as corporate farmers and financiers often require a financial return on any investment they make, including in mitigation measures (Landcare Trust, 2013). A nationwide survey of environmental farm plan programmes in New Zealand found that environmental farm planning is “widespread and diverse, but fragmented”

(Manderson, Mackay, & Palmer, 2007). In Canterbury, and the South Island as a whole, farms receiving water from irrigation schemes are often required to complete environmental farm management plans as part of their resource consent (e.g. Hunter Downs Irrigation, 2007) but there is no formal support for whole farm planning. Furthermore, there has been no assessment of the likely compliance with whole farm management plans should they be carried out for all farms.

Nutrient emissions limits and markets

In a review of the effectiveness of the United States' Clean Water Act, Laitos et al. (Laitos & Ruckriegle, 2013) state "best management practices (BMPs) can, if implemented, begin to control the extent and gravity of the water pollution problem caused by agricultural nonpoint sources. But, if BMPs continue to be optional and voluntary practices, agricultural sources likely will not install them". Even when regulation requires management practices to be implemented, compliance rates may not be high. An Environment Canterbury monitoring report on the disposal of dairy shed effluent in the Canterbury region (Burns, 2013) found that for the 2012-13 season only 72% of dairy farms were fully compliant with their effluent disposal conditions, with 7% having major non-compliance issues and causing substantial pollution. Compliance rates are improving; they have increased for seven years running from 39% in 2006 to 72% in 2013. However, if 28% of farms are unable to comply with their effluent disposal consent, it seems unlikely that a higher proportion could successfully implement a much more complicated whole farm management plan.

Winsten et al. (2011) argue that although enforcing the use of a defined set of nutrient best management practices will reduce nutrient runoff and leaching, blanket policies will also mean that some possible mitigation techniques will be missed because they are not included in the defined best practice, and overall spending on mitigation will not be targeted to the areas where it could have best effect. Instead, they suggest that performance incentives (i.e. payments or penalties for farmers based on actual nutrient loss) would allow farmers to choose the most appropriate mitigation measures for their situation and would target spending to the most effective areas. However, they identify barriers to performance incentives including difficulty calculating actual nutrient loss and resultant high administration costs, choice of denominator for nutrient loss (land area or productivity), and political pressure from farmers to avoid a penalty system. In addition to performance incentives at an individual farm level, they suggest that whole watershed level incentives could help increase uptake through peer pressure (or perhaps creating a watershed wide culture where farmers encourage each other to adopt mitigation practices).

While various forms of performance incentives have been proposed, the options most frequently discussed in the literature are financial incentives such as nutrient emissions limits (with financial penalties) or nutrient trading schemes (Parsons, 2012).

Market based incentives (MBIs) allow the trading of nutrient emissions risk, which theoretically allows the most efficient mitigation of risk because the required mitigation measures will be

Box 6. OVERSEER Nutrient Modelling Software.

OVERSEER is a software tool which allows the calculation of nutrient runoff and leaching and greenhouse gas emissions for a variety of land uses in New Zealand. The OVERSEER model was developed by AgResearch with funding from the Ministry of Agriculture and Forestry, the Fertiliser Association. OVERSEER is free to use for all New Zealanders, and is commonly used by farmers and consultants to optimise their farming practices, and by regional councils as the basis of their nutrient management policies (AgResearch, 2013) undertaken by those for whom mitigation is cheapest (Parsons, 2012). Four conditions must be met for MBI to represent an improvement over simple nutrient emissions limits:

- Nutrient emissions can be accurately quantified at the emitter level (e.g. individual farm losses)
- Nutrient emissions can be controlled by each emitter
- The location of the nutrient loss does not affect the environmental outcome (although this condition can be managed through careful market design)
- It must be cheaper for some emitters to reduce their emissions compared to other emitters. Without this heterogeneity, MBI can still cause reductions in emissions as all emitters will have to reduce their emissions the same amount. However, a simple emissions cap could achieve the same reductions for the same cost will less administrative complexity

(Parsons, 2012)

In the case of non-point source nutrient emission, the actual emissions from an individual farm cannot be directly measured nor precisely controlled. Instead, nutrient emissions and the effectiveness of mitigation measures must be estimated using a model such as OVERSEER (AgResearch, 2013). Use of estimation means that the market will optimise mitigation measures to the model, which will differ from reality to some extent, so is unlikely to be as efficient as a market based more reliable measurement. However, use of a model has the advantage that it encourages mitigation practices without penalising farmers for events beyond their control, such as random variation in actual emissions dependent on weather conditions (Parsons, 2012).

Most of the discussion of MBI in the literature has focussed on baseline and credit schemes, whereas within New Zealand most discussion has focussed on cap and trade schemes. Baseline and credit schemes usually only impose a cap on point source emissions, but allow capped emitters to offset emissions above the cap by buying credits. Credits can be generated by capped emitters coming in under-cap, or by diffuse emitters by implementing additional mitigation measures beyond some baseline of good practice. Baseline and credit is preferred in most countries because it avoids the need to measure emissions from non-point sources. However, as there is no hard and fast limit on emissions, economic growth can result in increasing emissions with no additional incentive to implement mitigation measures. Furthermore, in many regions of New Zealand there are insufficient point source emitters to create demand for credits generated by non-point source emitters, so it is unlikely that a baseline and credit programme would create substantial incentive to implement mitigation measures.

Cap and trade has a set overall emissions cap so enforces emissions targets – but is unpopular with business because it exposes them to a potentially volatile emissions market, and may be less effective at stimulating innovation as business may choose to move investment to other sectors instead of taking on risk (Parsons, 2012).

Taupo trading scheme case study

Lake Taupo has excellent water clarity and quality, but monitoring of water quality indicators demonstrates a gradual but steady decline over the past 30 years (Kerr, 2013). Figure 12 illustrates the changes in land cover since European settlement with shift from native scrubland and forest to pastoral farms and exotic plantation forest (Waikato Regional Council, 2004). These changes have resulted in increased nutrients leaching in to the lake, most importantly nitrogen, which is limiting for plant growth in the lake (Edgar, 1999). Recently, there has been further land use change with a shift from forestry and extensive sheep and beef farming towards more intensive dairy farming and increased nitrogen leaching. Should current trends continue, with no additional mitigation measures, water clarity could be substantially reduced (Waikato Regional Council, 2004).

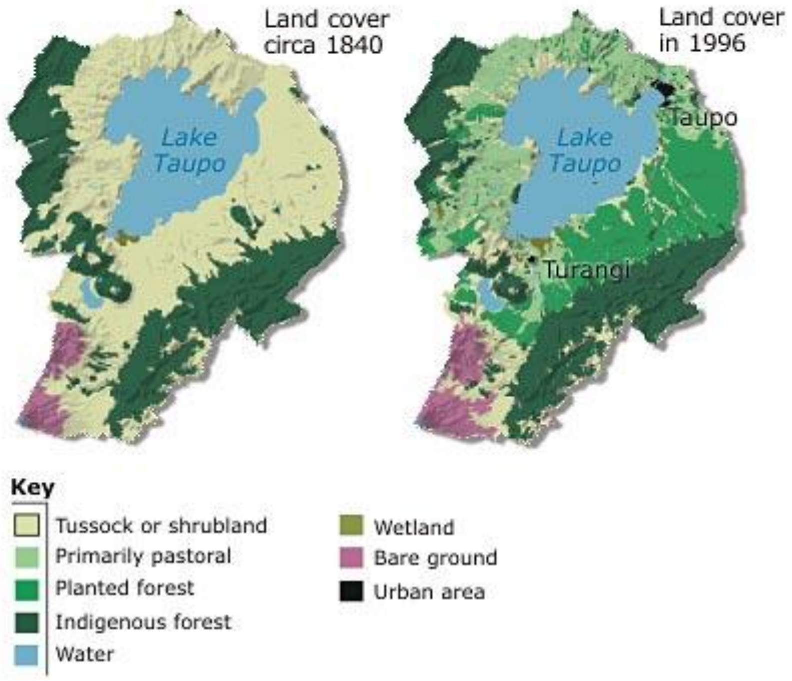

Figure 12. Change in land cover in the Lake Taupo catchment 1840-1996. 1996 land cover data supplied by Terralink NZ Limited. 1840's vegetative information derived from Environment Waikato's Native Vegetation Inventory. (Waikato Regional Council, 2004)

In 2001, Tuwharetoa Māori Trust Board and Environment Waikato signed a contract with the Ministry for the Environment to develop a strategy to protect Lake Taupo and community cultural, environmental, and commercial values. This partnership led to the release in 2004 of the 2020 Taupo-nui-a-Tia Action Plan (Taupo-nui-a-Tia 2020 Joint Management Group, 2004), which lays out the values that are important to the Taupo community, the priority threats to these values, what actions need to be taken to address these threats, and which agencies are responsible for each action. One action required the Tuwharetoa Māori Trust Board, Environment Waikato, and the Taupo District Council to reduce nitrogen inputs into the lake by 20% through a variation to the Waikato Regional Plan to include "a new regime for controlling diffuse run-off of nutrients from all land" (Taupo-nui-a-Tia 2020 Joint Management Group, 2004).

The Taupo catchment, like many in New Zealand, presents a difficult regulatory challenge as the great majority of nitrogen discharges are from non-point sources. In the Taupo catchment pastoral farming accounts for 92% of the anthropogenic nitrogen load on the lake, with the remainder coming from urban runoff, sewage, and nitrogen fixing plants (Duhon, Young, & Kerr, 2011). As such, any attempts to limit the nitrogen inputs into the lake must address non-point source pollution.

To respond to this challenge, Waikato Regional Council developed Regional Plan Variation Five (RPV5), which establishes a cap on nitrogen emissions and allows for trading of emission rights between landowners. RPV5 is unique globally in that it is the first cap and trade programme to be applied to non-point source polluters. The overall cap ensures that nitrogen emissions are kept at a sustainable level, but the trading scheme allows individual landowners to exceed their cap provided the emissions are offset by reductions in emissions elsewhere in the catchment (Duhon et al., 2011). The programme is enabled by the use of the OVERSEER software to determine landowners emissions (AgResearch, 2013).

The trading programme is kept simple by treating all nitrogen emissions within the catchment as equal, regardless of where they occur. Because lake water quality is the primary concern, it does not matter where in the catchment nitrogen emissions occur, they will all end up in the lake sooner or later. There are differences in the groundwater lags from different areas of the catchment, with some nitrogen losses reaching the lake in only a short time, and others taking over half a century, but there is little difference in nitrogen attenuation throughout the catchment. In the interests of simplicity, RPV5 ignores the differences in groundwater lags and treats all nitrogen losses as equal. This system will ensure that total nitrogen emissions remain at a sustainable level, but over-cap emissions from before implementation of RPV5 will continue to enter the lake for many years yet (Parsons, 2012).

Some substantial challenges were encountered in the development of the RPV5 scheme. The choice of the nitrogen cap at 20% below 2004 levels is a compromise between environmental goals and economic impact and is informed by science which has considerable uncertainty. As such, it is possible that the chosen cap is too loose so stricter controls will have to be imposed in the future if environmental goals are to be achieved, or that the cap is too strict so economic activity is unnecessarily curtailed (Duhon et al., 2011).

In addition to setting the overall cap, the council had to perform an initial allocation of emissions rights to existing landowners. This was done by a grandparenting system, where landowners were allocated a cap equal to their greatest annual emissions in the five year period from 1999-2004. However, this system had the effect of increasing the property values of landowners who had already intensified their farming systems, especially dairy farms with very high nitrogen outputs, while also decreasing the property values of landowners who had low nitrogen outputs, such as forest owners and many Māori landowners. This last point was particularly problematic as, by disadvantaging Māori, many felt that the Waikato Regional Council was failing to meet its obligations under the Treaty of Waitangi. Disadvantaged parties instead suggested that allocation use a land area basis or a delayed land area basis where grandparented allocations revert to land area over time. However, these systems would have eliminated much of the value of the infrastructure investment carried out on intensively farmed land. These equity problems led to an allowance for low-nitrogen leaching activities on Māori and non-Māori land to increase their nitrogen outputs by up to 2kg/ha/yr up to a total of 11,000kg/yr from Māori land and 3,100kg/yr from non-Māori land. However, this allowance could result in the overall cap being exceeded (Duhon et al., 2011).

Rotorua trading scheme case study

Water quality in Lake Rotorua has declined steadily over the last 30 years due to increasing inputs of nitrogen, primarily from agricultural intensification but also with a significant contribution from urban development (Kerr, McDonald, & Rutherford, 2012). To combat these changes the Rotorua Nutrient Trading Study Group was set up to examine how a nutrient trading programme for the Rotorua catchment might work. They proposed a programme where nutrient losses are estimated using OVERSEER and average climate data. All nutrient allowances would be treated as equal, and trading would be allowed between any landholders in the catchment.

Like Taupo, there is little nitrogen attenuation in the Rotorua catchment, so all nitrogen losses eventually end up in the lake. However, there are very large differences in groundwater lag from different parts of the Rotorua catchment, with some nitrogen losses taking over 200 years to reach the lake. Because of these very long delays, the Rotorua Nutrient Trading Study Group (Kerr et al., 2012) considered the use of a nitrogen "vintage" market, where nutrient loss reductions occurring today would be worth more than nutrient loss reductions in the future. However, simulations indicated that such a market would achieve little mitigation cost reduction or emissions reduction despite adding considerable complexity and administrative costs. Instead, they recommended incentivising mitigation now by allowing emitters to save their nutrient allowances year-on-year for future use or trading, which would reward emitters for early mitigation but adds little administrative complexity (Kerr et al., 2012).

The study group (Kerr et al., 2012) also addressed the problem of allocation of discharge allowances. They suggested that, to avoid a sudden economic shock, it would be necessary to grandparent allowances for the first few years of the scheme, similar to the Taupo allocation. However, as this allocation would effectively reward the worst polluters, the study group suggested that future allocation should gradually transition to a system based on the land's capacity to absorb nutrients, as estimated using OVERSEER.

In addition to changing allocation between landowners, the study group recommended that the overall nutrient cap should fall over time to eventually reach the level required for environmental goals to be achieved. The choice of mechanism to reduce the cap is essentially a choice of who should pay the cost of emissions reductions. If the cap is simply set to a lower point then the cost to reduce emissions will be borne by the emitters. Conversely, if the cap is reduced by government buying and retiring allowances, as occurs in the Taupo programme, then the cost will largely be borne by taxpayers or ratepayers. It is likely that a mixture of falling cap and retiring allowances will be used to spread the cost amongst stakeholders.

It is proposed that when landowners exceed their nutrient allowance, they will have to pay a hefty fine and will surrender future emissions credits to make up the difference. However, there is currently no legal framework for the regional council to enforce a fine. There are provisions for fines under the Resource Management Act, but these fines are small enough to be considered a routine business expense (Kerr et al., 2012).

Nutrient trading in Canterbury

The situation on the Canterbury plains would require a more complex trading system than in Taupo or Rotorua. The Taupo and Rotorua schemes are focussed on improving lake water quality and are in catchments with low nutrient attenuation, so emissions anywhere in the lake catchment will have similar effect on the outcome. In Canterbury however, the quality of groundwater and river water is just as important as lake water, so emissions will affect downgradient and downstream outcomes, but not upgradient or upstream outcomes. For a nutrient trading programme to successfully protect water in Canterbury, the region would have to be divided into many small catchments with trading allowed only within each catchment. Careful consideration would be required to strike a balance between catchments small enough to adequately protect ground water quality, and catchments large enough to allow the nutrient trading market to function.

Putting a price on water

Giving water use an explicit monetary value could help to limit overall abstraction and preserve water quality. Two options for putting a price on water include a simple charge per volume used, or a water rights trading market (Lange, Wood, & Winstanley, 2006).

Water rights in New Zealand are currently allocated on a first-come first-served basis. This arrangement has a number of negative effects including:

- Water may be allocated to very low value activities at the expense of high value activities
- Some water allocations are not utilised, which encourages water authorities to over-allocate water resources, potentially creating environmental and reliability of supply issues in the future
- There is no incentive for existing rights holders to use water more efficiently (Dridi & Khanna, 2005; Kiem, 2013; Lange et al., 2006)

A simple price on the volume of water used could incentivise the efficient use of water, but will not help re-allocate water from low-value to high-value uses, and will create an even greater incentive for authorities to over-allocate water resources (Lange et al., 2006). As such, most discussion has focussed on creating a trading market for water use rights.

Trading of water rights has the potential to limit water abstraction whilst also allowing the flexibility to change the way water is used in the face of changing economic, climatic, social, and environmental drivers (Dridi & Khanna, 2005; Gohar & Ward, 2010; Lange et al., 2006). Water trading is an attractive option in New Zealand because trading is already allowed for within the existing Resource Management Act legal framework. Indeed, informal trading already occurs within New Zealand (e.g. Hydrotrader, 2014). However, the lack of any formal trading mechanism means that transaction costs are very high and the market provides no protection for environmental values. To create a market which protects environmental and societal values whilst also maximising economic productivity requires consideration of a number of issues including equity and moral concerns, the nature of water use property rights, potential unexpected third party effects including environmental effects, the potential for monopoly formation, and prioritising pure economic efficiency relative to other social goals (Lange et al., 2006).

Most discussion of water rights trading in the literature is theoretical, or examines ways to promote the implementation of a water trading scheme. The literature search for this review found only three articles (Heaney, Dwyer, Beare, Peterson, & Pechey, 2006; Qureshi, Grafton, Kirby, & Hanjra, 2011; Turral et al., 2005) describing a water trading scheme in practice: the Murray-Darling Basin in Australia.

By 2003, the simple cap-and-trade water rights trading put in place in the Murray-Darling basin in the 1980's had resulted in only very small increases in economic efficiency due to low trading volumes, although substantial gains had been made in some localised areas due to within-scheme trading (Heaney et al., 2006; Turral et al., 2005). However, a substantial increase in trading was stimulated by the drought of the mid 2000's, leading to much higher volumes of trading by 2009 (Wheeler, Bjornlund, Zuo, & Shanahan, 2010). Turral et al. (2005) and Wheeler et al. (2010) argue that the low trading volumes are primarily the result of limits on the trading of water entitlements out of catchment (typically 2%-4% of total entitlements per year), which are intended to allow communities time to adapt to changes in their economies caused by a shift in water use. On the other hand, Heaney et al. (2006) argue that trading is limited because the market fails to account for conveyance and storage losses, conveyance capacity, and different environmental effects of water use in different areas or for different activities (Heaney et al., 2006). For example, transferring water rights from an area with low pollution susceptibility to an area with high susceptibility could increase pollution, or transferring rights to a user with a more efficient irrigation system could reduce groundwater recharge, which in turn could increase groundwater extraction costs for other users. These market limitations mean that regulators may not approve many apparently economically beneficial trades because of environmental or reliability effects on third-parties. Furthermore, other trades which would create a net real benefit are uneconomic in the incomplete market structure (Heaney et al., 2006).

Beyond limiting the market effectiveness, failure to account for all externalities in the market instrument can result in substantial unintended negative outcomes (Kiem, 2013; Turral et al., 2005). In a “water only” trading market the highest value users are often also the greatest polluters (e.g. dairying and cropping in Canterbury, mining in the Murray-Darling basin), and a lower market value is assigned to other uses such as urban water supply (drinking water excepted, Kiem, 2013). Unrestricted trading in these examples could substantially increase the rates of water pollution and greenhouse gas emissions, while reducing local food security and availability of water for fire-fighting. Water markets may also reduce security of supply if they enable the trade and subsequent use of existing but unused water allocations (Heaney et al., 2006; Turral et al., 2005).

A market that more accurately reflects the true value of water rights may help to overcome these problems, and Heaney et al. (2006) present a number of options to help improve the water market in the Murray-Darling basin. The most promising options are to create additional markets for pollution and for conveyance capacity. For conveyance, an alternative with lower administrative burden is a congestion charge imposed on all water drawn when total delivery for a region is above the congestion limit (Heaney et al., 2006).

There are many unused or partially used water allocations in Canterbury (ref), so implementation of a water right trading scheme would require careful consideration of the implications for overall water abstraction. If a goal of the scheme is to limit or reduce overall water consumption then additional measures will be required, such as government buying and retiring water allocations, or a proportional reduction in the size of all allocations (Qureshi et al., 2011), the choice of which largely determines who bears the financial burden of environmental impact mitigation. These considerations are more fully discussed in relation to nitrogen trading schemes.

The hydrogeology of Canterbury is substantially different from the Murray-Darling basin. Most importantly, Canterbury has many catchments each with their own outlets to the sea, and a large proportion of abstracted water is drawn from relatively rapidly replenished groundwater. The implementation of parallel markets for water, water pollution (probably nitrogen) and water conveyance is likely to also be the optimal solution for Canterbury, but different market structures are likely to produce optimal results.

Raffensperger (2011) proposes a more advanced method of water trading –smart markets – that incur minimal transaction costs and can make use of the best hydrological knowledge to maximise production for a given environmental impact. Smart markets are particularly suited to catchments with large groundwater abstraction because they can allow for the complex interaction between abstractions in different parts of an aquifer. Smart markets require all transactions to be made through a central clearinghouse, controlled by the regulator, which allows the regulator to optimise the price of water at each well. The price is automatically optimised based on the demand for water at that well and throughout the catchment, predicted externalities throughout the catchment, and how the water will be used (which affects return flows). Because a computer model is used to automatically generate prices within environmental constraints, water traders are not required to seek out and negotiate with trading partners and transaction costs are minimised. Furthermore, it ensures that appropriate abstraction occurs at each well, rather than just limiting total abstraction from the catchment regardless of location, so has the potential to produce a more economically and environmentally efficient solution than simpler models (Raffensperger, 2011). Although not discussed by Raffensperger, (2011), a smart market model could be developed which simultaneously accounts for both water abstraction and other third-party effects such as nitrogen emissions and congestion of water conveyance infrastructure. On the other hand, smart markets rely on complex hydrological models to determine environmental impacts, so if these models are not available or not accurate, smart markets are infeasible or could produce worse results than traditional markets. Furthermore, the complex model used to calculate prices is likely to be unintelligible to many traders, so may be treated with suspicion, making it politically difficult to implement.

Provided adequate hydrological models are available and stakeholder buy-in is achieved, smart markets offer the best solution for maximising productivity while avoiding undue environmental impact due to water abstraction. However, a simple cap-and-trade market is also likely to provide substantial improvement over the current over-allocated free-trade situation. In either case, it is important that abstraction and other externalities (especially water pollution) are considered together.

Climate change

Importance for health

Vermeulen et al. (2012) provide a comprehensive review of the anticipated global effects of climate change on food systems. On the Canterbury plains, the projected effect is for less rainfall and more droughts on the plains, although possibly with more frequent heavy rainfall events. In contrast, there is expected to be increased rainfall in the headwaters of the alpine rivers. Substantial sea level rise would also threaten coastal areas (NIWA National Climate Centre, 2008).

The literature search for this review found no information about the direct impacts of these changes for Cantabrians, but it could be reasonably expected that increased alpine river flow with decreased plains rainfall would increase the reliance on surface-water irrigation for agriculture and decrease the viability of dryland agriculture. Similarly, sea level rise is likely to disadvantage costal dwellers to a greater extent than others. This situation is likely to create winners and losers – improving the wealth and health of those who are able to irrigate while decreasing the wealth and health of others. Furthermore, decreased water availability could result in the need to make use of lower quality water sources with high treatment costs or high risk of water borne illness, such as the recycled wastewater irrigation schemes in Melbourne (Barker-Reid et al., 2010). In addition to direct health impacts, the 5th Assessment Report from the Intergovernmental Panel on Climate Change highlights many indirect ways in which climate change could decrease health, ranging from increased disease transmission to increased global conflict (IPCC, 2013)

Effect of land use change

Food systems contribute between 19 and 29% of global greenhouse emissions (100 year carbon dioxide equivalent), predominantly from animal agriculture (Vermeulen et al., 2012). Within New Zealand, agriculture contributes almost half of the country's greenhouse emissions, and further increases in production will further increase emissions. However, as New Zealand has a low ratio of emissions to food production, an increase in food production in New Zealand could lead to a decrease in net global emissions provided increased production in New Zealand is offset by reductions in less efficient production overseas (Ministry for the Environment, 2012).

Global climate change is expected to decrease food production in tropical and other water constrained regions, such as dryland farming areas on the Canterbury plains, but increase potential and demand for food production in colder regions where water is not limiting to growth, such as irrigated areas of the Canterbury plains (Vermeulen et al., 2012). Such changes could drive further intensification on the plains, which in turn could lead to increased emissions from New Zealand, but again may reduce net global emissions (Ministry for the Environment, 2012).

Ecosystem dysfunction and loss of biodiversity

Importance for health

Agricultural landscapes provide, in addition to crop and livestock production, a range of other ecosystem services. Levy et al. (2012) state that ecosystem services provide

> *Goods (such as seafood and timber), life-support functions (water purification and flood control), and life-fulfilling conditions (beauty and inspiration), as well as the preservation of options (such as genetic diversity for future use; Levy et al., 2012)*

The epidemiological importance of biodiversity in reducing disease risk, with particular reference to species that break the chain of pathogen transmission to humans, is evident in the case of malaria and domestic livestock in India. Active zooprophylaxis has also been utilised, with cattle being used as a barrier between human settlements and known mosquito breeding sites, in Soviet collective agriculture and currently in Tanzania (Dobson, Cattadori, Holt, Ostfeld, & Keesing, 2006).

Effect of land use change

Landscapes "are made up of a number of poorly understood interlinked biophysical systems – so tinkering with one element can cause vast changes throughout the connected systems" (Falkenmark & Galaz, 2007). An ecosystem will not necessarily respond in a linear manner to gradual change and disturbances. These systems can undergo damaging and irreversible shifts. It is essential therefore to be aware of critical thresholds which exist within these systems for, once they have been crossed, degradation quickly follows.

In south-western Australia, for example, soils in low-lying areas have become unproductive due to rising water tables and highly saline groundwater. Water tables have risen due to increased infiltration of irrigation water drawn from surface water, and up-gradient farmers replacing deep-rooted native plants with shallow-rooted agricultural plants which take up less water. Low-lying areas are now waterlogged and the soils are unproductive due to salination, and will remain saline for many years even if the water table is lowered (Greiner & Cacho, 2001).

Modification of a landscape on a large scale changes the plant and animal diversity within the area. The associated risks include the loss of organisms that perform important activities such as recycling nutrients and those able to suppress undesirable organisms. Loss of biodiversity can also make an ecological system more fragile. Any disturbances in the system, that once may have been accommodated, result instead in a possibly irreversible shift (Falkenmark & Galaz, 2007).

The increasing demand for dairy products, which require large quantities of water to produce, poses an enormous challenge to natural ecosystems with increases in water usage anticipated to have a severe impact on ecosystems. Both coastal and freshwater aquatic ecosystems will suffer through reduced flows, changing flow variability and potential chemical loadings. Gabel et al. (2012) found that indices of the health of diatom ecosystems (Diatom Model Affinity and the Trophic Diatom Index) was strongly correlated with chemical measures of surface water quality (specific conductance, pH, total dissolved phosphorus, $NH4^+$-N, and $NO3^-$-N), whereas macroinvertebrate ecosystem health was not. They suggest that indices of diatom ecosystem health may be a practical means to assess overall water quality in agricultural landscapes.

In addition to water quality changes, some ecosystems will be cut off due to the building of water infrastructures for irrigation (Environment Canterbury, 2010b; James, 2011). Terrestrial ecosystems could also suffer with changing land use and land cover changes altering hydro-climatic moisture feedbacks (Falkenmark & Galaz, 2007).

> *A critical aspect not to be forgotten is that increased water demands affect not just ecosystems, but also the important services they provide. For example, landscape alterations lead to increased eutrophication and potential loss of marine diversity and fish stock. Or in another example, drained or altered water bodies or wetlands can greatly affect the cultural, religious, aesthetic, ethical or recreational services provided in different societies around the world (Falkenmark & Galaz, 2007)*

Dobson et al. (2006) identify a number of human processes that contribute to a loss of biological diversity including 'habitat loss, habitat fragmentation and the overexploitation of populations for food or other economic uses, the introduction of invasive species and diseases, climate change and pollution'. Of these habitat loss is the primary cause, cited in 70% of the species listed as threatened or endangered by the Red List. Species requiring larger land area requirements tend to be lost first, with the impact of fragmentation compounding this. In smaller remaining areas only smaller species or those with particularly effective dispersal capabilities are able to persist. As the predators and competitors that determine the numbers of the prey species disappear from the smaller patches then the numbers of prey individuals increase. If these prey individuals happen to be reservoirs for zoonotic pathogens, there will be a consequent increase in the abundance of the particular pathogen.

Within Canterbury, habitat fragmentation has already occurred at a large scale with the clearing of the Canterbury plains. However, a new, if less diverse, ecosystem has emerged which relies in part on anthropogenic habitats such as shelter-belts and water races (Environment Canterbury, 2010b; James, 2011). Destruction of these habitats to facilitate installation of new irrigation systems is likely to further reduce biodiversity on the plains.

Strong rural communities

Importance for health

Equity has a large impact on the overall health of communities. Increasing the personal wealth of the most disadvantaged people in developed world communities will help to reduce their major burdens of disease: non-communicable disease and violent or accidental deaths and injuries (Marmot, 2005). Conversely, more well-off people are already at low risk of these health problems, so they are likely to experience a smaller increase in health for the same increase in their personal wealth (Blas & Kurup, 2010).

Effect of land use change

Donham et al. (2007) focus in their paper on the community health and socioeconomic issues associated with intensification of agriculture and industrial farm animal production (IFAP) specifically. The authors represent a workgroup that was convened following a scientific conference and workshop held in Iowa in March 2004. The consensus of this workgroup was that 'improving and sustaining healthy rural communities depends on integrating socioeconomic development and environmental protection'. Rural and agricultural communities in the United States have experienced dramatic changes over the last fifty years. There has been an overall reduction in the number of farms, but an increase in their size. Many farms are managed by corporate producers and upper management as IFAP facilities and many of the workers often do not come from or live within the vicinity of the IFAP. Although there has been little IFAP in Canterbury to date, there have been attempts to establish aspects of IFAP (Gorman, 2009), and these systems are established in other parts of the country (Macfie, 2014). Furthermore, many of the socioeconomic effects of IAFP in the United States are the result of an increased size of farms with fewer owner-operators and more corporate farmers (Donham et al., 2007). These changes are analogous to those in Canterbury where there are increased numbers of very large corporate farms and very small farms, but fewer medium sized owner-operated farms (Mulet-Marquis & Fairweather, 2008).

There is evidence that the economic concentration of agricultural operations results in a greater proportion of money being removed from rural communities compared with an industry which is dominated by smaller farm operations which instead keep money circulating within the local community. Donham et al. (2007) mention a number of studies that have found that both the concentration and industrialization of agriculture were associated with economic and community decline, locally and regionally (Chism & Levins, 1994; Goldschmidt & Nelson, 1978; Gómez & Zhang, 2000; Osterberg & Wallinga, 2004). Donham et al. (2007) conclude that high social and economic well-being in rural communities is associated with a high number of farmers rather than a high volume of commodity produced.

Agricultural intensification may also reduce equity by imposing additional costs on all community members, some of whom may already be struggling to afford access to health-improving services. Tait and Cullen (2006) estimated the external costs that dairy farming in Canterbury imposed on the wider Canterbury community in 2003. They used established methods to estimate the cost of lost angler values, greenhouse gas emissions, loss of shelterbelts, and zoonotic illness. For some costs, there was no established method, so for loss of surface water, loss of groundwater, sediment in surface water, and bovine TB they substituted an estimate of the amount of public money actually spent mitigating the costs of dairying, which is likely to be an underestimate of a true external cost. Some factors were not considered at all, such as loss of ecosystem services or non-angling amenity value. The final estimate was that dairying in Canterbury created between $28 and $45 million of external costs annually, which were predominantly due to greenhouse gas emissions. Even if greenhouse emissions were not considered, $1.43 million of public money was actually spent on mitigating the external effects of dairying, and a great deal of loss of amenity value and ecosystem services was not considered. Furthermore, the population of dairy cattle in Canterbury has continued to increase since 2003, so the external costs of dairying are likely to be greater now.

External costs are often predominantly borne by those who can least afford to pay. Balazs et al. (Balazs, Morello-Frosch, Hubbard, & Ray, 2011) demonstrated that, in the San Joaquin Valley in California, the percentages of water supply users of Latino ethnicity and who did not own their home were each predictive of greater nitrate concentration. That is, the burden of drinking water nitrates was disproportionately borne by those who could least afford to implement mitigation measures. Although no similar analysis has been completed for Canterbury, the plains are similar to the San Joaquin Valley in that they are agricultural areas where intensive agriculture is facilitated by irrigation, they derive the majority of their drinking water from groundwater sources, and the lowest average incomes occur in small urban centres serving the surrounding agricultural community (Balazs et al., 2011; Goodyear, 2004)

Very intensive agriculture may also reduce quality of life for neighbouring residents through air pollution. Smell can be than an unpleasant odour, and can have dramatic consequences for rural communities where lives are rooted in enjoying the outdoors (Thu, 2002). Situating large-scale livestock facilities near homes disrupts rural living and can result in “highly cherished values of freedom and independence associated with life oriented toward the outdoors give[ing] way to feelings of isolation and infringement". Homes cease to be a base from which to enjoy the natural world but instead become a barrier against conditions outdoors that must be avoided (Donham et al., 2007).

Studies that have evaluated the impacts of IFAP on communities suggest that much local controversy arises and that rifts can occur between community members that often prove to be "deep and long-standing" (Wright et al., 2001). An in-depth study across six counties in southern Minnesota, USA, found three patterns reflecting a decline in social capital as a result of the siting of IFAP in each of the rural communities studied:

- *Widening gaps between IFAP and non-IFAP producers*
- *Harassment of vocal opponents of IFAP*
- *Perceptions by both supporters and opponents of hostility, neglect or inattention of public institutions that resulted in perpetuation of an adversarial and inequitable community climate*

(Wright et al., 2001)

Research has also found a disproportionate number of IFAP facilities in the United States are located in non-white areas and areas of low-income, and near low-income and non-white schools. The IFAP facilities and their associated hazards are unwanted by communities and are often thrust upon sectors with the least political influence. These communities are often already experiencing the impacts of poor housing, low income, a lack of access to medical care and poor health status (Wright et al., 2001).

A study by Slemp et al. (2012) examined the effects of urban expansion on aspects of healthy rural communities other than equity. Researchers performed in-depth qualitative interviews with 32 residents from two small Illinois communities undergoing rapid urbanisation about their perceptions of the changes brought by urbanisation. They identified four main themes amongst residents' interview data, each of which influence health in the community:

1 - Loss of "natural" environments

The change of landscape was felt by most residents as a personal loss. They identified the farmed landscape as "natural" and felt that losing this landscape reduced ecosystem health and the ecosystem's capacity to provide ecosystem services, and therefore also reduced their own health. However, some residents also felt the rapid growth was necessary to attract enough funds to afford environmental protection.

Most participants identified the link between land use change and decreased water quality, reflecting the social importance of water quality and the well-established links between land use and water quality.

2- Economic development

Most residents saw economic development as the primary or only benefit of increased urbanisation. They felt urbanisation would attract wealthy people and expensive developments, which would increase the municipal tax base and provide economic opportunities for residents. However, dissenting voices claimed that rapid urbanisation led to planning problems, required expensive upgrades to existing infrastructure, and would still fail to bring any economic benefit. Residents who disliked the urban development for other reasons were also more likely to question the economic benefits.

3 - Sense of community

Most residents felt that community capital was threatened by in-migration. They said that urbanisation caused community fragmentation and reduced community identity, especially with the influx of commuters who worked in the city. Many participants cited their sense of community as an important factor in their quality of life, and were concerned this quality would be reduced. Conversely, a few participants, especially newer arrivals, said the in migration would create new opportunities to develop community spirit such as theatre, art, and music events.

4 - Cultural conflict

The researchers argued that differing expectations for the community between existing residents and new comers create social conflict. They cited examples such as recent immigrants complaining about the smell of a long-established pig farm. Farmers especially felt that this kind of cultural conflict could threaten their livelihood (Slemp et al., 2012).

Conclusions

Land use change has been occurring rapidly in recent years, both at a global scale and within Canterbury. While there is potential for this change to create wealth – indeed wealth creation is usually the purpose of land use change – there is also potential for unintended effects which may impair the health of communities.

The history of land use change in Canterbury is as long as the history of human occupation, and has allowed substantial increases in health and wellbeing. Recently, the dairy boom has driven rapid expansion of irrigation and agricultural intensification, while the 'lifestyle block' trend and the 2010-2011 Canterbury earthquakes have driven urban expansion of Christchurch's rural fringe and satellite towns.

The majority of literature examining the health effects of land use change is focussed on water quality. There is clear evidence linking agricultural intensification and urban expansion with decreases in water quality and increases in the incidence of waterborne disease. Furthermore, these effects are often most keenly felt by those who can least afford mitigation measures – people who are often already vulnerable and can easily be pushed in to poor health. Poor health anywhere in a community has a financial cost for the whole community, through the cost of treatment and the loss of productivity to the economy. Many mitigation technologies are available which, if implemented, could allow intensification of existing agricultural activities without the negative external effects on water quality. However, these mitigation technologies also add cost so may make more intensive agriculture uneconomic. Furthermore, currently available mitigation technologies are not sufficiently effective to offset conversions of land from low-intensity activities such as forestry and dryland sheep-beef farming to high-intensity activities such as dairying or cropping.

Apart from water quality, agricultural intensification and urban expansion have the potential to affect health through increased greenhouse gas emissions, loss of biodiversity and ecosystem services, and weaker rural communities. Furthermore, agricultural intensification could also affect health through increased risk of zoonotic disease and increased antimicrobial resistance. The evidence for the health implications of these effects is not specific to Canterbury, but national and international evidence demonstrates that the kind of changes occurring in Canterbury will be likely to have these effects. Furthermore, although the health effects of these changes are less immediate than those of water quality, they could have much more substantial effects in the long term so they cannot be ignored.

Land use in Canterbury will continue to change along with the global environment and economy. Care must be taken to ensure land use decisions support economic growth without creating adverse health outcomes. The information in this review provides decision makers with information to assist them to meet this challenge.

Glossary

100 year carbon dioxide equivalent	Emissions of greenhouse gases nitrous oxide (N2O) and methane (CH4) are often expressed as the equivalent units in CO2 in terms of their global warming potential in 100 years: N2O has 296 times the warming potential of CO2 and CH4 23 times the warming potential of CO2
Agricultural intensification	Increasing the inputs into an agricultural system (e.g. labour, fertiliser, irrigation, or pesticides) so as to increase production.
Antimicrobial resistance	Resistance of a microorganism to an antimicrobial drug that was originally effective for treatment of infections caused by it
Aquifer	An underground, permeable, and saturated layer of rock, gravel, sand, or silt. Water may travel through the aquifer or collect in an aquifer.
Baseline and credit	A nutrient trading model in which point source emitters are provided with a baseline emissions allowance, and may exceed the allowance by purchasing credits. Credits may be generated by point source emitters coming in under their allowance, or by non-point source emitters implementing nutrient management practices. Non-point source emissions are not monitored
Best management practices (BMPs)	A defined set of management practices which minimise nutrient emissions. All emitters must implement the same BMPs regardless of their baseline emissions
Biodiversity	The variety of life present in an ecosystem
Blue baby syndrome	See "Methaemoglobinaemia"
BMP	See "best management practices"
Cap and trade	A nutrient trading model in which a fixed limit for nutrient emissions is set, and emissions allowances are divided up among emitters. Emitters are free to trade allowances, but there is no way to create extra allowances. All emissions must be monitored and quantified
Determinants of health	All the factors which contribute to an individual's health. Most of these factors lie outside the health system. E.g. housing, air quality, water quality, personal income, social support networks, gender.
Dryland agriculture	Agriculture which does not make use of irrigation
Ecosystem	A community of organisms and their physical environment linked together by flows of nutrients and energy
Emissions	The release of potential pollutants from a human system. In the context of this document, emission refers to the release of nitrogen or phosphorous into waterways, or the release of greenhouse gasses into the atmosphere
Equity	The concept of equal opportunity for all individuals, regardless of their personal circumstances. E.g. children from wealthy and less-well-off families having the same access to healthcare and education
Greenhouse gas	A gas which absorbs and emits radiation within the atmosphere. Greenhouse gases create a "greenhouse effect", where heat is trapped near the surface of the Earth rather than being radiated in to space. The primary greenhouse gases in the Earth's

	atmosphere are water vapour, carbon dioxide, methane, nitrous oxide, and ozone
IFAP	See “Industrial farm animal production”
Industrial farm animal production (IFAP)	The practice of keeping animals at very high stocking rates through the use of imported feed and, frequently, prophylactic antibiotic use
Intensification	An increase in the productivity of a farming system through the use of new management practices, irrigation, fertilisers, or imported feed
Irrigated agriculture	Agriculture which uses artificial application of water to the land to promote crop growth (cf. dryland agriculture)
Irrigation	The artificial application of water to the land to promote crop growth
Land cover	The physical material covering the surface of the land. E.g. grass, crops, forest, concrete, asphalt, and buildings
Land use	Land use is the purpose for which humans use the land. E.g. for agriculture, transport, habitation, or recreation. One area of land can often support many uses simultaneously
Land use change	When humans change the way they use a particular area of land
Leaching	The loss of water soluble nutrients from the soil via transportation with rainwater or irrigation water
Market based incentives	Any system which uses a trading market to incentivise a reduction in nutrient emissions. See “baseline and credit” and “cap and trade”
MAV	See “maximum acceptable value”
Maximum acceptable value (MAV)	The maximum concentration defined by a jurisdiction of a specific contaminant allowed in water to be used for a specific purpose. E.g. the maximum concentration of nitrates allowed in drinking water. The MAV is usually defined at the maximum concentration which avoids known risks to public health
MBI	See “market based incentives”
Methaemoglobinaemia	A disorder characterised by an excessive proportion of blood haemoglobin existing as methaemoglobin (containing an Fe^{3+} ion rather than an Fe^{2+}). The disorder usually occurs in infants fed on formula prepared using water with high nitrate concentrations. Methaemoglobinaemia is often called “blue baby syndrome”, as symptoms include blue tinged skin.
Nitrate	An ionic form (NO_3^-) of nitrogen. Nitrate is water soluble and is an essential and often limiting plant nutrient
Nitrogen	A chemical element with atomic number 7. In the form of the nitrate ion (NO_3^-), nitrogen is an essential and often limiting plant nutrient
Non-point source	A source of nutrient emissions which is released over a wide area. E.g. fertiliser applied to a paddock. Cf. point source
Nutrient	A chemical substance which an organism uses to survive and grow
Nutrient emissions	The loss of soil nutrients beyond the confines of the system controlled by the emitter. E.g nitrogen losses into groundwater
Nutrient emissions limits	A maximum allowance for the amount of nutrients able to be released by an emitter. In the context of nutrient management, nutrient emissions limits often refers to non-tradable allowances.

Nutrient trading scheme	Any system which allows emitters to trade their nutrient emissions allowances. Most frequently "baseline and credit" or "cap and trade" schemes
OVERSEER	Software used to estimate the nutrient emissions of a farm system. OVERSEER is jointly owned by MPI, FANZ and AgResearch, and is provided free for use within New Zealand
Pathogen	A microorganism which is capable of causing disease
Phosphate	An ionic form (PO_4^{3-}) of phosphorus. Phosphate is insoluble in water and is an essential and often limiting plant nutrient
Phosphorus	A chemical element with atomic number 15. In the form of the phosphate ion (PO_4^{3-}), phosphorus is an essential and often limiting plant nutrient
Point source	A source of nutrient emissions which comes from a single point. E.g. a sewer pipe or a septic tank. Cf. non-point source
Water quality	The chemical, physical, biological, and radiological characteristics of water
Watershed	An area of land from which all water drains through a single outlet
Zoonosis	See "zoonotic disease"
Zoonotic disease	A contagious disease spread between animals and humans
Zooprophylaxis	The use of animals to divert insect disease vectors away from humans

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