The Cost of Milk: Environmental Deterioration vs. Profit in the New Zealand Dairy Industry

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Abstract

In the past two decades major increases in production have occurred in the New Zealand dairy industry. Between 1990 and 2012 the dairy cow population increased by 87%. Milk production increases were double this (195%) over the same period, while the land area used for dairy production increased by 46% between 1993 and 2012. This intensification of production has required the use of externally sourced inputs, particularly an increase in fertiliser, feed supplements, and irrigation.

Dairy intensification has been associated with increased environmental impacts. Water quality in lakes, rivers and streams is declining, particularly in catchments with a predominance of dairy farms. Soil physical properties are worse on dairy land than other farming types and soil contamination on dairy land is reaching concerning levels. Furthermore, dairy farms are responsible for about a quarter of New Zealand's greenhouse gas emissions. Additionally, there are a range of offshore impacts relating to the importation of products used for dairy farm production under this intensified regime. New Zealand's 'clean green' brand is important for the dairy industry as well as other primary producers and international tourism. New Zealand's environment must live up to this brand to provide creditability to its products.

These environmental impacts not paid for by the farmer are termed environmental externalities. Most of the pollution caused by dairy farming is not currently remedied or paid for by the dairy industry. Hence, the public is largely left to deal with these externalities, both regarding the economic costs and the environmental degradation that occurs. The aim of this thesis was to compare the cost of the environmental impacts of dairy farming in New Zealand with the economic value (export revenue) of dairy and thus establish a clearer position and understanding of the actualised value of this industry. A conservative estimate of the economic costs of some of the externalities and imports were over $19 billion, much higher than the 2012 dairy export revenue of $11.6 billion. It is likely that this is a severe underestimate of the actual value of the environmental externalities given all that was not measured.

Acknowledgments

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Mihi ki ā Papatuanuku, mihi ki ngā tangata, mihi ki ngā awa.

For Mother Earth, for the people and for the rivers.

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Chapter 1. INTRODUCTION

In order for us to maintain our way of living, we must, in a broad sense, tell lies to each other, and especially to ourselves…The lies act as barriers to truth. These barriers to truth are necessary because without them many deplorable acts would become impossibilities.

(Derrick Jensen – A Language Older Than Words, 2000)

1.1. Problem Statement

The New Zealand dairy industry has experienced a period of rapid growth and intensification in the last two decades. It has reached the point where dairy farming is now New Zealand's largest export industry and contributes to over 20% of gross export revenue. However, the export revenue is only one facet to dairy's value to New Zealand and does not factor in environmental and social impacts of dairy farming. Quantifying the environmental consequences of dairying is challenging, both in monetary and ecological terms. Additionally, the dairy industry has other benefits not included in many valuations, such as employment and purchase of New Zealand products required for production.

There is evidence that agricultural intensification in the past few decades has contributed to environmental deterioration. For example, 96% of rivers in pastoral catchments are too polluted to swim in (Larned et al., 2004); half of the lakes in pastoral areas have poor ecological conditions (Verburg et al., 2010); soil resources are deteriorating; and agricultural greenhouse gas emissions are increasing (Ministry for the Environment, 2012c). Costly and challenging remediation solutions are often needed because pollution is not reduced at the source. Impacts damage New Zealand's 'clean green' image, threaten future food production, and lead to biodiversity loss and degradation of recreational areas.

The economic benefits must be weighed against the environmental costs of things such as declining water quality and loss of native biodiversity. Impacts not paid for are termed externalities and are often not counted in the dominant economic valuing system (Daly, 2005); hence, society as a whole is left to pay for the costs. Famers bear little of the costs, and instead are effectively receiving subsidies to pollute.

1.2. Aim of Research

The aim of this thesis was to compare the costs of the environmental impacts of dairy farming in New Zealand with the economic value (export revenue) of dairy and thus establish a clearer (and more complete) value of this industry.

1.3. Objectives

1. Describe dairy farming production and input trends from 1990 to 2012;
2. Identify and describe the environmental impacts of dairy farming on water, land and soil and its contribution to greenhouse gas emissions in New Zealand;
3. Identify a selection of environmental impacts from major imports used in dairy farming;
4. Identify the impacts from dairy farming on New Zealand's 'clean green' image and the potential loss in value of the image from these impacts;
5. Quantify in monetary terms, where possible, the costs of the environmental effects of dairy farming. Costs could be to restore the environment, to mitigate effects or to prevent effects in the first place.

1.4. Methods

This study involved a meta-analysis to obtain an overview of the current situation and changes that have occurred over the past two decades in the dairy farming industry. Due to the enormous scale, impacts outlined in this study were broad with several regional or catchment examples used to illustrate effects. The value of New Zealand's 'clean green' image is discussed as well as associated losses that could occur if the image were to be tainted.

The term environmental degradation was broadly used to describe an environmental system that is negatively affected, for example poor water quality that cannot be used for drinking or to support aquatic life, reduced habitat quality and quantity, and reduced biodiversity. Contamination was used to refer to specific environmental qualities exceeding standards or levels, for example nitrate in water exceeding specific levels, and cadmium in soil exceeding guideline levels. The terms ‘effects’ and ‘impacts’ are also used to describe things that have been caused by or associated with an activity.

Exploring the costs associated with the identified environmental impacts involved several methods in an attempt to either: a) determine the costs associated with remedying or mitigating environmental impacts; b) estimate the costs associated with cleaning up a degraded environment (caused by dairy farming practices); and/or c) evaluate the potential costs involved in paying for environmental contamination, for example through reduction agreements. Costs of avoiding contamination by using on-farm technologies were briefly considered to compare costs of not contaminating in the first place with costs of cleaning up. Examples of costs were obtained from previous studies. More information on these methods is provided in Chapter 9.

1.5. Contribution to Knowledge

Analysis of the environmental impacts from dairy farming in New Zealand has not previously been attempted on a national scale. Where reports on intensive farming and sustainability have been carried out, economic analysis has not been incorporated. Likewise, where the economic benefit of dairy farming has been reported upon, little consideration has been given to environmental impacts.

Although this is not new research, it is novel in that it brings together a large extent of information to obtain a more realistic view on dairy’s value, rather than solely the export revenue. However, many knowledge gaps still exist. Many other benefits related to dairy farming have been excluded, such as social benefits of farming communities and employment opportunities. This was because the focus was on environmental impacts. Accordingly, this research is not intended to be the last word on valuing dairy positives and negatives, but a contribution towards the discussion in finding solutions to reducing environmental effects associated with dairy farming, and thus, the frequent bad reputation this industry receives.

1.6. Limitations

Inevitably, there are many limitations in this study; some of the major ones are outlined. Other limitations relating to particular sub-topics are discussed in their respective section.

Due to the large scale of the dairy industry, only impacts on a farm scale (impacts from the dairy farm land itself) involved in milk production are described. Impacts associated with manufacturing, processing and transport are not included. While a complete overview of the whole industry is not provided, export revenue is based on the whole industry, including processing and transport. Furthermore, economic valuations on environmental impacts are likely to be conservative. This study is a broad overview incorporating only the main impacts and costs of dairy farming, considering only the New Zealand dairy industry.

Information was compiled from numerous sources and time periods. Consequently, elements measured are based on differing scales and assumptions. This made it difficult to compare and consolidate information. However, the focus was on the overall depiction of impacts, rather than small discrepancies. Furthermore, effort was made to obtain a recent synopsis but not all information was recent.

Deficiencies in information are due to a range of factors including: data reporting changing between years (i.e. between years criteria had changed to how data were reported – often not mentioned in data sources); incomparable data between things measured (i.e. different environmental impacts are hard to compare in importance and severity); differing methods between studies utilised (making consolidating information difficult); and deficient data in some years. Where possible, trends were established from the last two decades but these may not be accurate due to the aforementioned data issues. Where longitudinal studies were deficient, data from single years were used.

Identifying the precise environmental impacts from dairy farming is difficult as many of the associated impacts are hard to quantify or identify at source. One of the major problems is separating the impacts between dairy and other land uses as the impacts are cumulative and often take many years or decades to become visible. To accentuate this, many studies only report impacts from pasture or agriculture as a whole making it difficult to isolate those from dairy farming alone. Furthermore, like most human activities, cumulative impacts to the whole catchment are often overlooked with impacts often only being assessed within individual properties (Jay & Morad, 2007).

1.7. Outline of Thesis

Chapter 2 provides an overview of New Zealand dairy farming and intensification trends over the past few decades, including some of the major inputs increasingly used on dairy farms. Chapter 3 describes how conventional economics has failed to incorporate the environment into analyses, providing a false impression to the true value of an industry or product. Chapter 4 provides an overview of environmental management processes in New Zealand. Background information provided in Chapter 3 and 4 is important to understand how the dominant economic system and management regimes have inadequately regulated the environmental effects of production activities, in this case: New Zealand dairy farming.

The following three chapters describe the impacts from dairy farming associated with water (Chapter 5), land (Chapter 6) and greenhouse gas emissions (Chapter 7) in New Zealand. The consequential impacts in the environment from these issues are described. Chapter 8 describes some of the offshore impacts associated with international imports (which are major inputs) used for dairy farming.

Valuations of some environmental impacts are provided in Chapter 9. A national monetary analysis is often not possible for many of the impacts but examples are provided. Also included in this chapter is the value of the clean green image to New Zealand. Finally, Chapter 10 broadly discusses the environmental costs from the New Zealand dairy industry and some additional issues associated with the industry. Farm management practices that could be implemented to alleviate some environmental effects are also described.

Chapter 2. SETTING THE SCENE

Because we don't think about future generations, they will never forget us.

(Henrik Tikkanen)

Being an agricultural nation, the physical environment is vital to New Zealand's economic well-being. Primary production has been the dominant land use in New Zealand since European settlement. Pastoral farming (e.g. sheep, beef and dairy) is the main land-use in New Zealand; in 2004 it covered just over 37% of New Zealand's total land area (Ministry for the Environment, 2007a). Major increases in productivity have occurred in New Zealand over the last 25 years. In the 1970s, Central Government provided a number of agricultural subsidies, providing some farmers with up to 40% of their income for agricultural uses. During this time, sheep and beef farming were the dominant land-use in the farming sector.[1]

However, the removal of subsidies in the mid-1980s saw a move from low-intensity to high-intensity agricultural land uses mainly covering flat areas (i.e. from sheep and beef to dairying), accelerating sharply in the 1990s (Willis, 2001). Sheep numbers peaked in 1982 with a total of 70.3 million sheep and since the mid-1980s sheep numbers have been gradually declining (Statistics New Zealand, 2012b). By 2012, sheep numbers had decreased by more than half their peak numbers, down to 31 million (Bascand, 2012). Beef cattle numbers have also been slowly decreasing from a peak of 6.2 million in the mid-1970s to 3.7 million in 2012. Deer numbers peaked in 2003 with 1.76 million and decreased to just over 1 million in 2012 (Statistics New Zealand, 2012a). Conversely, dairy farming expanded and intensified in the last two decades with dairy cattle numbers doubling since 1990, reaching 6.5 million in 2012 (Bascand, 2012). Increasing cow numbers have been instrumental in dairy intensification.

Intensification describes the process that increases outputs per unit area (MacLeod & Moller, 2006). The transformation and intensification of dairy farming has increased the need for external inputs, such as synthetic fertilisers, irrigation, and additional stock feed (such as maize silage and palm kernel) (Clark et al., 2007; Densley et al., 2010; Ledgard et al., 1998; Ledgard et

[1] A number of articles and reports provide a more detailed overview of agricultural trends in New Zealand, including MacLeod & Moller (2006); Parliamentary Commissioner for the Environment (2004); and Willis (2001). For the dairy farming situation in particular, Jay (2007) provides a discussion on the rising dairy pressures in New Zealand.

al., 1996; MacLeod & Moller, 2006; Ministry for the Environment, 2004; Parliamentary Commissioner for the Environment, 2004). These inputs are required to enable higher stocking rates and increase on-farm productivity.

An overview of typical New Zealand dairy farm production is shown in Figure 2.1. Farms receive inputs from the outside environment, either from natural systems or human created ones. Outputs (milk) and externalities are produced from the farm. Additionally, some inputs will produce externalities during manufacturing before entering the farm system. Externalities are divided into four sections: water (Chapter 5), land (Chapter 6), atmosphere (Chapter 7), and offshore impacts (from some of the imported products used in dairy production) (Chapter 8). Often, the only parts of the dairy production process that are acknowledged are the product outputs (i.e. milk) and revenue. Although Government subsidies were removed, the public are now unofficially subsidising agriculture by paying to clean up the impacts (externalities) and through the loss of recreational areas.

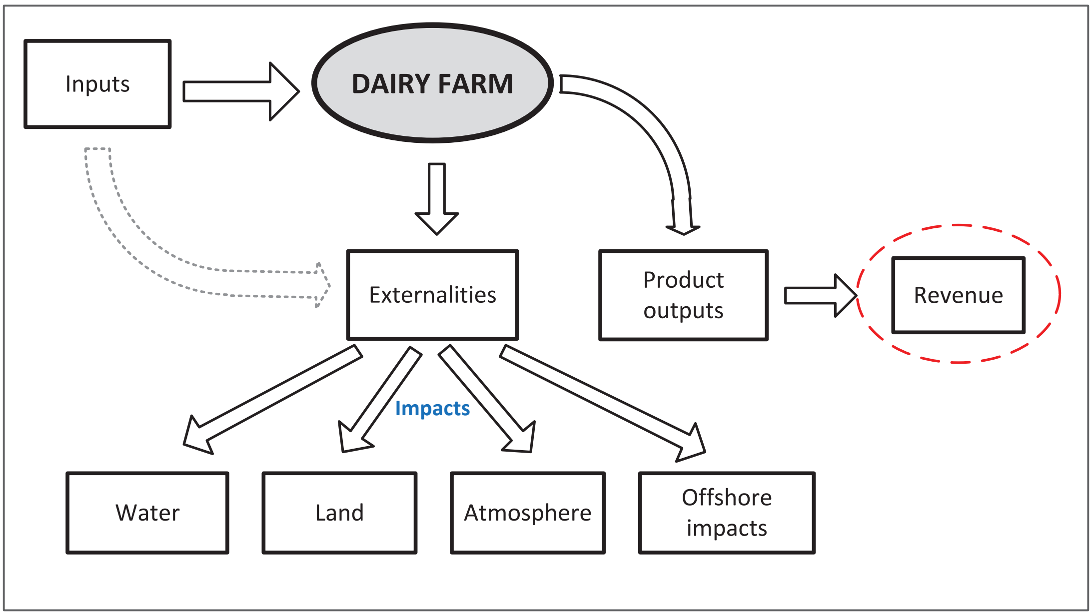


Figure 2.1: An overview of dairy farm production in New Zealand.
Notes: Most accounts on the value of dairy farming are based only on the revenue generated (denoted by the red dashed oval), and neglect to include externalities.

2.1. Export Revenue

New Zealand is the largest exporter of dairy products in the world, accounting for more than a third of the international market. In the year ending 2009, New Zealand exported dairy products to 151 countries, with about 72% of the total export value being derived from developing countries (Ministry for Primary Industries, 2012a). Although the largest exporter, New Zealand is only the 7th (Index Mundi, 2013c) or 8th (Marshall et al., 2012) largest milk producer globally, accounting for around 2% of world milk production (Ministry for Primary Industries, 2012a). This is because only 5% of world milk production is traded across borders. Most countries produce milk for their own domestic supply, whereas New Zealand exports more than 95% of milk produced (DairyNZ et al., 2009). Consequently, New Zealand's export prices are highly dependent on international market fluctuations. Milk powder is the main dairy commodity exported by New Zealand and New Zealand is the largest global exporter and producer of milk powder, surpassing the next largest exporter by almost three and a half times (Index Mundi, 2013a, 2013b).

The export value of New Zealand dairy products increased from $2 billion in 1990 to $11.6 billion in 2012 (Figure 2.2), an increase of 460% over two decades (Statistics New Zealand, 2012a). This increased the contribution of dairy to the New Zealand's export market from 13% to almost 25% from 1990 to 2012. The next two largest export goods after dairy in 2012 were meat and forest products, valued at $5.1 and $4.2 billion respectively (Statistics New Zealand, 2012a) (Figure 2.2). New Zealand export value trends from 1990 to 2012 for selected products are presented in Figure 2.2.

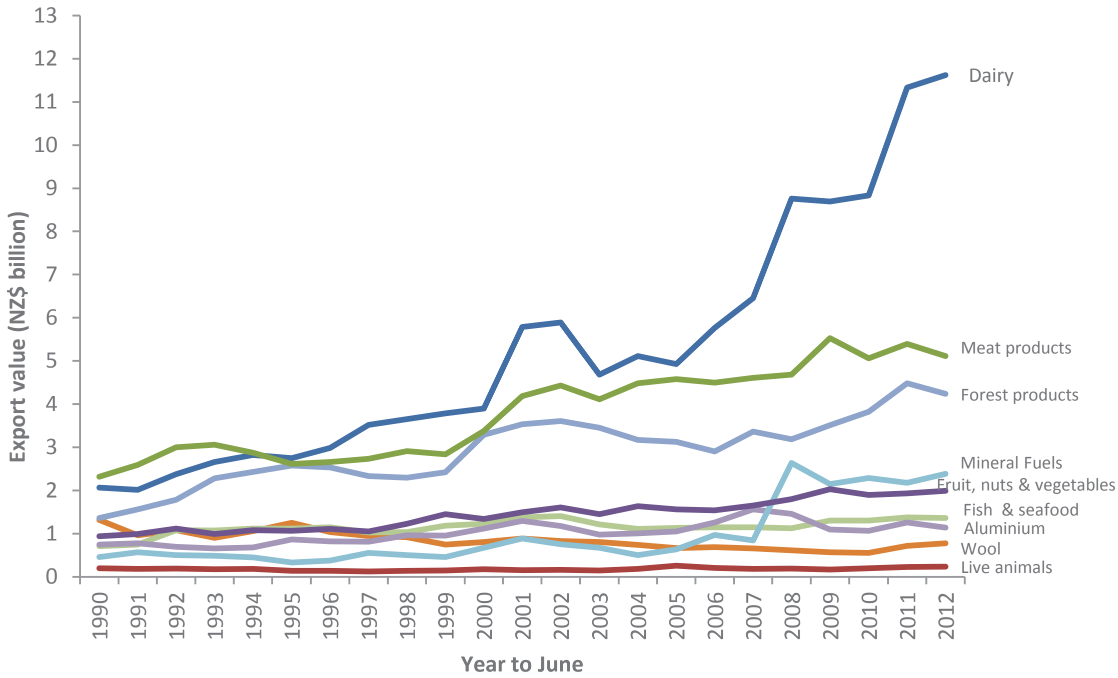


Figure 2.2: Export revenue trends of some sectors in New Zealand between 1990 and 2012.
Data source: Statistics New Zealand (2012a).

2.2. Production Trends

An overview of trends in the dairy industry relating to the area in dairy farming, numbers of dairy cows and milk production for the last two decades is outlined in this section.

2.2.1. Dairying area

Sheep and beef farming dominate agricultural land area in New Zealand (Figure 2.3), although estimations about the area of land in dairy farming vary, even when reported by the same organisation (Table 2.1). Over half of New Zealand's land area has been reported as being used for farming (Parliamentary Commissioner for the Environment, 2004) and around 37% was estimated under pasture in 2004 (Ministry for the Environment, 2007a). In 2012, dairy farming was estimated to cover about 9% of New Zealand's land area and around 17% of the total agricultural area (Figure 2.3) (Statistics New Zealand, 2013a). In the last five years (2007-2012) dairying area increased by 23%, while sheep and beef farming land area decreased. Most other agricultural sectors have also decreased in land area in the past five years (Statistics New Zealand, 2013a).

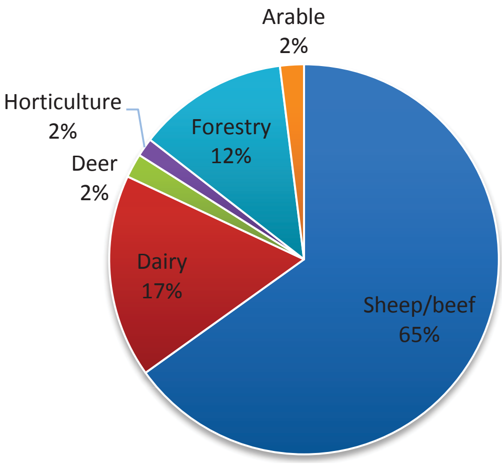


Figure 2.3: Proportion of New Zealand's agricultural and forestry land area in different land-uses in 2012.
Data source: Statistics New Zealand (2013a).

Table 2.1: Estimations of land area in dairy farming

Year of estimate	Hectares	Percentage of NZ in dairy
1985	1,378,607[1]	5.1
1990	1,023,545[2]	3.8
1993	1,653,137[1]	6.2
1995	1,864,302[1]	6.9
1996	1.728.537[3]	6.4
	1,276,551[2]	4.8
2002	2,048,211[4]	7.6
2004	1,879,600[5]	6.9
2007	1,962,724[6]	7.0
2012	2,414,769[7]	9.0
	1,638,546[2]	6.1

Data source: [1]Willis (2001); [2]NZ Dairy Statistics (LIC & DairyNZ, 2012); [3]Agricultural Production Census (Statistics New Zealand, 1998); [4]Agricultural Production Census (Statistics New Zealand, 2002); [5]Environment NZ 2007 (Ministry for the Environment, 2007a); [6]Agricultural Production Census (Statistics New Zealand, 2007); [7]Agricultural Production Census (Statistics New Zealand, 2013a). **Notes:** The NZ Agriculture Production Census classifies blocks of land based on main purposes. NZ Dairy Statistics do not include area for runoff pasture and winter grazing – only hectares used specifically as a milking platform are counted, explaining the lower area estimated.

2.2.2. Dairy cattle

The national dairy cattle population has been increasing steadily in the past two decades, particularly more rapidly in the South Island. Between 1990 and 2012, dairy cow numbers almost doubled, from 3.4 million to nearly 6.5 million (Figure 2.4) (Statistics New Zealand, 2012a). In the year from 2011 to 2012 dairy cow numbers increased by 23% (271,738) (Bascand, 2013), and in Canterbury alone, numbers rose by 19% (193,551), which was the biggest annual increase at a regional level for any type of livestock in the last two decades (Bascand, 2013). These increases brought the total South Island herd numbers to almost 2.5 million in 2012 (Bascand, 2013). In 2012, 28% of the national dairy cattle population was located in the Waikato region, followed by Canterbury (18.6%), Southland (10.4%), and Taranaki (9.4%) (Statistics New Zealand, 2012a).

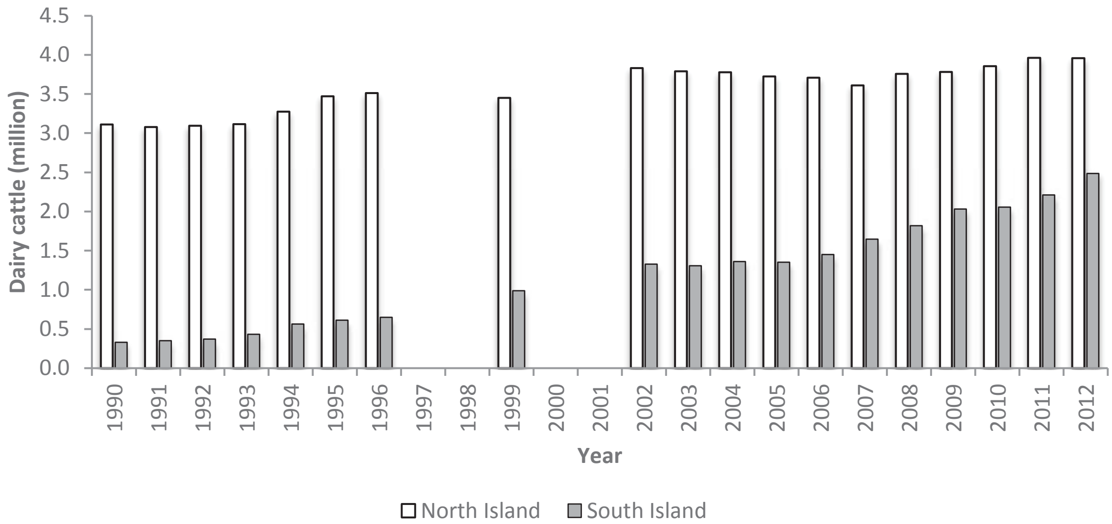


Figure 2.4: Total dairy cattle (including Bobby Calves) in the North and South Islands between 1990 and 2012.
Data source: Statistics New Zealand (2012a). **Notes:** No agricultural production survey relating to dairy was conducted in 1997, 1998, 2000 and 2001.

Stocking rates

In the last three decades average national dairy stocking rates increased by 37% from 2.0 cows per hectare in 1980 to 2.8 cows per hectare in 2011/12; in Canterbury average stocking rates are greater than 3 cows per hectare (LIC & DairyNZ, 2012). Recorded stocking rates are based on only lactating cows and the effective area in dairying and do not count dairy support areas used for winter grazing, rearing replacement of dairy heifers, and areas for growing silage crops for dairy feed.

Dairy herds

Dairy herd sizes have similarly grown over the past four decades; however, the number of herds has decreased. From the mid-1970s to 2012, the number of herds decreased from 18,540 to 11,798, while the average herd size increased from 112 cows to 393 during the same period (Figure 2.5) (LIC & DairyNZ, 2012). From 1990 to 2012 there was a 180% decrease in the number of herds and a 147% increase in the average herd size (LIC & DairyNZ, 2012).

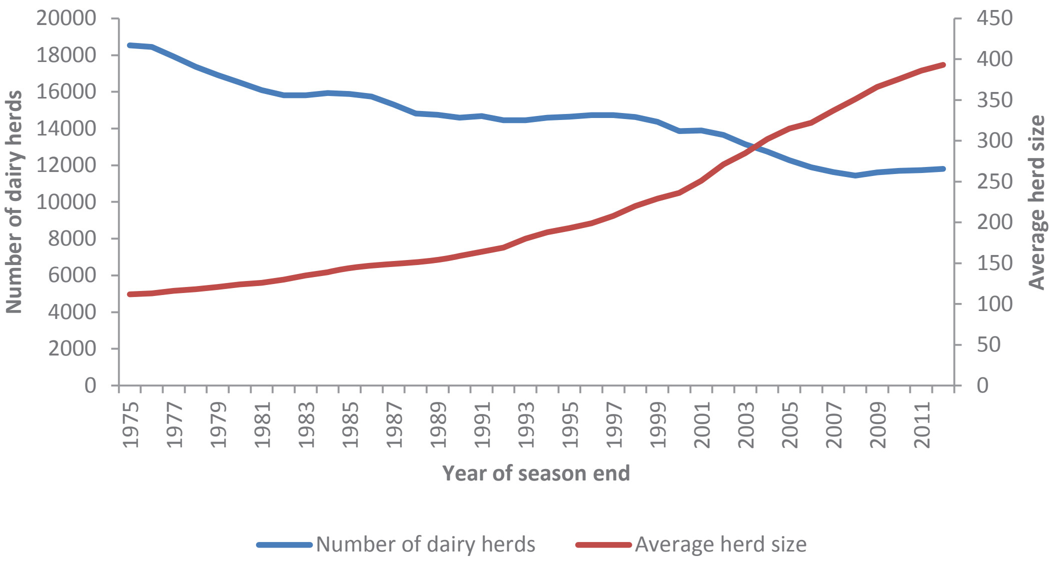


Figure 2.5: Number of dairy herds and average herd size of New Zealand dairy farms between 1975 and 2012.
Data source: Livestock Improvement Corporation and Dairy NZ (2012).

2.2.3. Milk production

Milk production has more than doubled over the last two decades. Between 1990 and 2012, milksolids (MS - milk fat plus protein) production in New Zealand increased by 195% from 572 tonnes to 1685 tonnes (Figure 2.6) (LIC & DairyNZ, 2012). Between 1993 and 2012, MS production per hectare and per cow increased by 60% and 40%, respectively[2] (Figure 2.6) (LIC & DairyNZ, 2012). These two measures followed very similar trends over this time period.

[2] Data on production per hectare and per cow were not available before 1993 and were not calculated in this study because of the uncertainty of land area in dairy.

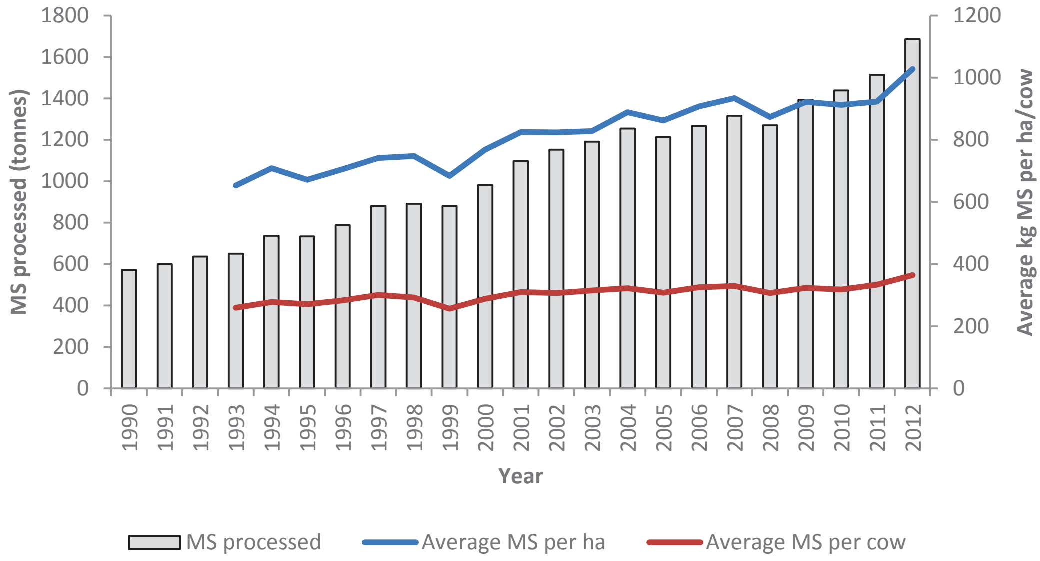


Figure 2.6: Total milksolids (MS) produced (1990-2012) and average kg MS produced per hectare (blue line) and per cow (red line) (1993-2012) in New Zealand.
Data source: LIC & Dairy NZ (2012).

2.2.4. Summary of production trends

Increases in milk production and dairy cow numbers have increased on a larger scale than dairy land area, indicating that large scale intensification has occurred (Figure 2.7). However, estimations for the change in land area in dairying were used from 1993, not 1990.

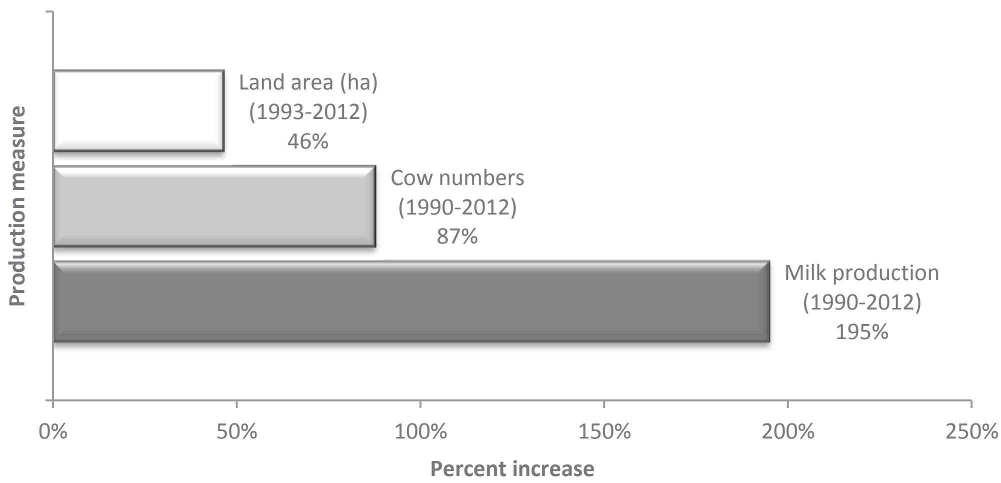


Figure 2.7: Summary of dairy production trends in last two decades
Note: Land area cannot be directly comparable to cow numbers and milk production as statistics start from 1993.

2.3. Input Trends

Dairy intensification has required increased inputs in order to increase production (Figure 2.8), such as greater amounts of fertiliser, supplementary feeds and often irrigation. Other agri-chemicals, such as pesticides, animal supplements, and animal remedies for infections/diseases, are also often needed. Some of these inputs are domestically produced in New Zealand, therefore carrying environmental implications, but do not affect imported costs. Others products are imported, so the cost of importing should be detracted from the export revenue gained from dairy farming. Major imported products used in dairy production that can be traced are palm kernel expeller (used as a feed supplement) and fertilisers. These two inputs are discussed here, along with water for irrigation. The externalities associated with imported resources are discussed in Chapter 8, while irrigation is covered in Chapter 5. Figure 2.8 lists some of the standard inputs into dairy farms and also those required to intensify; however, requirements generally extend further than this list.

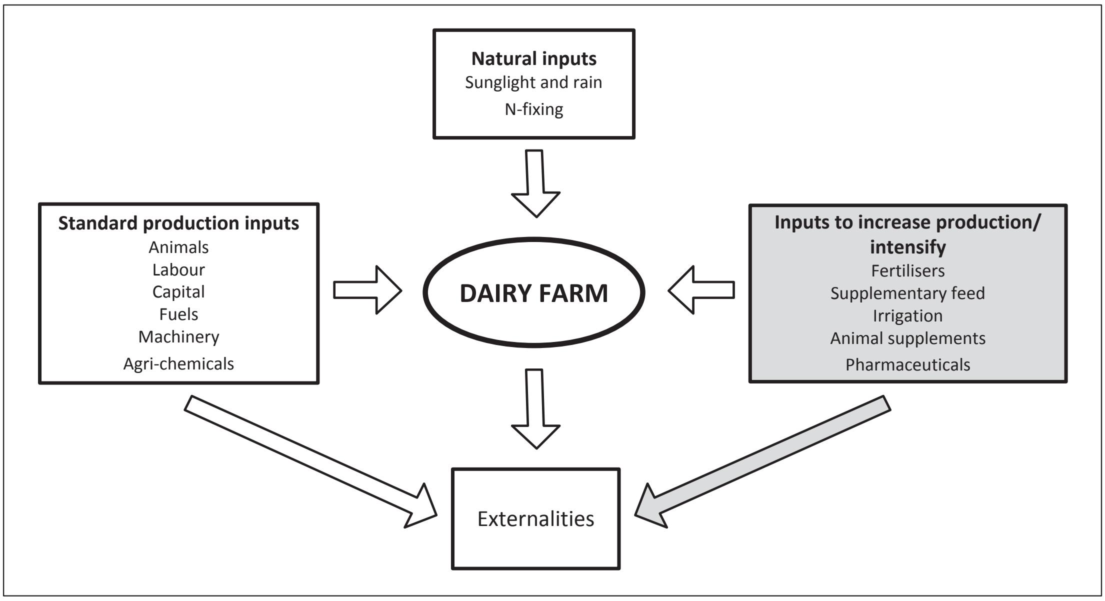


Figure 2.8: The main inputs used on New Zealand dairy farms.

2.3.1. Fertiliser use

Most soils in New Zealand are acidic and naturally low in nutrients due to their development under forests (Parliamentary Commissioner for the Environment, 2004). Therefore, adding nutrients to increase plant growth and lime (calcium oxide) to reduce acidity to soils is common for agricultural production. Natural fertilisation using clover[3] was common until synthetic fertilisers became heavily used globally in the last few decades (Parliamentary Commissioner for the Environment, 2004). In farming systems nutrients are needed in varying amounts depending on soil type, crop species (Statistics New Zealand, 2006), and nutrient loss – both from the soil and in agricultural products. Common fertilisers now used on New Zealand agricultural soils include lime; phosphatic (P) fertilisers such as Superphosphate; nitrogenous (N) fertilisers such as urea and ammonium sulphate; potassic (K) fertilisers; and compound fertilisers containing more than one nutrient, for example, di-ammonium phosphate (DAP).

Significant sources of nitrogen applied to dairy farms include nitrogen fertilisers, dung and urine from grazing animals, and farm dairy effluent discharges (Davies-Colley et al., 2003). Nitrogen fertiliser has been used to supplement (or completely replace) clover fixation in order to increase pasture production (Parliamentary Commissioner for the Environment, 2004; Roberts & Morton, 2009). In this way, N fertiliser can work as a form of supplementary feed when animal requirements exceed pasture growth (Roberts & Morton, 2009).

Applying effluent collected from the milking shed onto the land cycles nutrients back into the soil. This practice can decrease the amount of fertiliser application required, lowering fertiliser costs (Wang et al., 2004). However, farmers often over-apply fertilisers and effluent (Parliamentary Commissioner for the Environment, 2004). Not all farmers can store effluent to be applied when conditions are best; if there are no storage facilities, farmers have to apply effluent every day.

Quantity and cost of fertiliser application in New Zealand

The dairy sector is estimated to use 44% of the fertiliser applied on New Zealand agricultural land (Parliamentary Commissioner for the Environment, 2004). For some fertiliser types this proportion is much higher. Records show that more than 2.1 million tonnes of fertiliser is sold in New Zealand (Statistics New Zealand, 2012a). However, data on fertiliser use in New Zealand over the last three decades has been inconsistent. In particular, the categorisation of fertiliser statistics has changed since 1996, making it unviable to compare trends over time in some fertiliser types.

[3] Clover fixes nitrogen from the atmosphere and has been a significant feature in New Zealand's agricultural production.

Fertiliser use on agricultural land in New Zealand is reported from farm surveys and extrapolated for agricultural use in New Zealand (reported in Agricultural Census Data). The quantity and cost of fertiliser imported into New Zealand (reported by Statistics New Zealand) does not include fertiliser processed in New Zealand, nor does it forecast what land use the fertiliser is applied on. Hence, the proportion of fertiliser imported for dairying is based on the estimated amount used on dairy farms (in the Agricultural Census data), providing a basic indication of fertiliser use and imported cost trends for dairying.

2.3.1.1. Nitrogen fertiliser

New Zealand ranks amongst the highest per capita uses of nitrogen fertiliser globally. In an international study of 61 countries, New Zealand ranked second highest in territorial nitrogen use per capita (over 60 kg N/capita/year) after Ireland (Nykvist et al., 2013). Most countries included in the study used between less than 10 kg N/capita/year and 30 kg N/capita/year. In absolute terms, New Zealand's nitrogen use was much smaller, but still surpassed the environmental assimilation rate (rate the receiving environment can handle) (Nykvist et al., 2013). In New Zealand, about 90% of the cost of imported nitrogen fertiliser is for urea or ammonium sulphate - the two main fertilisers used for agriculture.

Urea

Urea is the most common form of nitrogen fertiliser and use has increased dramatically since the 1980s. According to 2012 farm surveys, dairy farms used 72% of the urea in New Zealand (Figure 2.9) (Statistics New Zealand, 2013a). Between 1996 and 2012, reported urea use in the dairy sector increased by 360% (Figure 2.9), and from 1990 to 2012 total agricultural urea use increased by 2600% (from 18,576 tonnes to 501,303 tonnes) (Statistics New Zealand, 1998, 2013a).

New Zealand urea is produced synthetically at the Kapuni Urea Plant in South Taranaki which produces approximately 260,000 tonnes annually. Natural gas is used from the Maui, Kupuni and Kupe gas fields for urea production. More than double the amount of urea produced domestically was imported in 2012 (526,000 tonnes) while almost 600,000 tonnes was imported in 2011 (Statistics New Zealand, 2013b). The quantity of imported urea is more than that reportedly used by farmers (501,303 tonnes in 2012) (Statistics New Zealand, 2012a), suggesting that surveys may be inaccurate or fertiliser is stored before purchase.

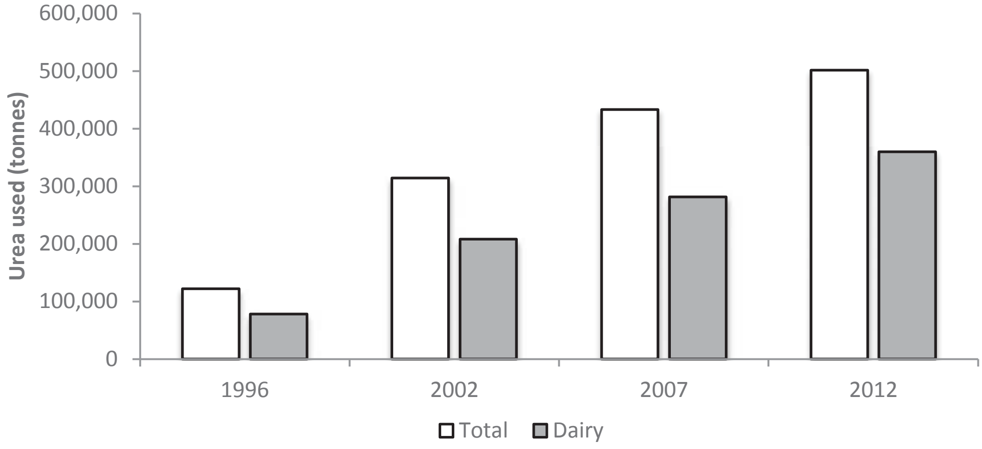


Figure 2.9: Tonnes of urea fertiliser used in New Zealand on all farms and only dairy farms in 1996, 2002, 2007 and 2012.
Data Source: Statistics New Zealand (1998, 2002, 2007, 2013a). **Notes:** Data for urea use on dairy farms was only available for years shown.

Imports

Over half of the total cost of imported fertiliser in 2012 was spent on nitrogen fertiliser ($374 million), of which 85% was urea (Statistics New Zealand, 2013b). In 2012, 80% of the imported urea was from just two countries: Saudi Arabia and Qatar (Statistics New Zealand, 2013b). Urea imports have increased substantially in the past two decades (Figure 2.10).

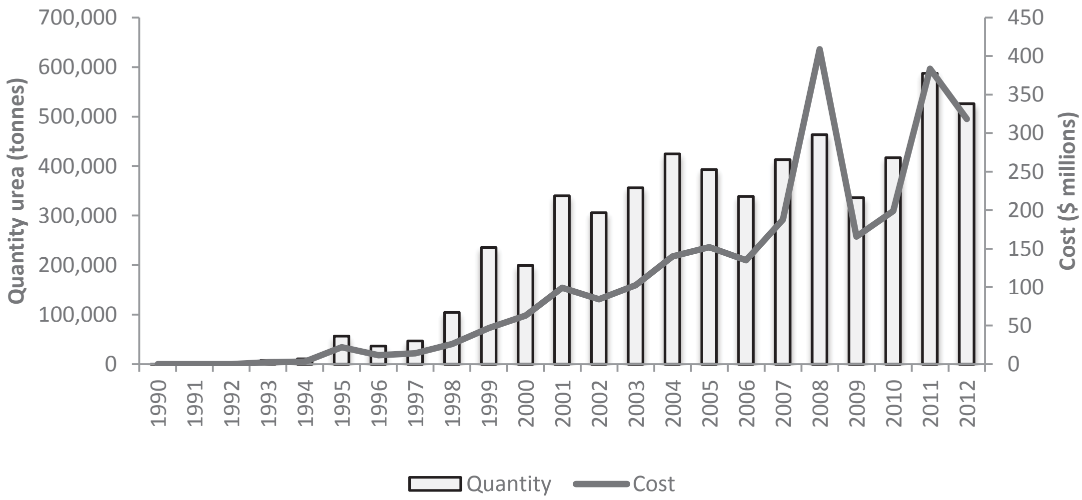


Figure 2.10: The quantity (bars) and cost (line) of urea imported into New Zealand between 1990 and 2012.
Data source: Statistics New Zealand (2013b).

Ammonium sulphate

Total ammonium sulphate use in agriculture has changed little since 2002; however, use on dairy farms has doubled from 2002 to 2012 with the proportion of the total used on dairy farms increasing from 33% in 2002 to 68% in 2012 (Figure 2.11).

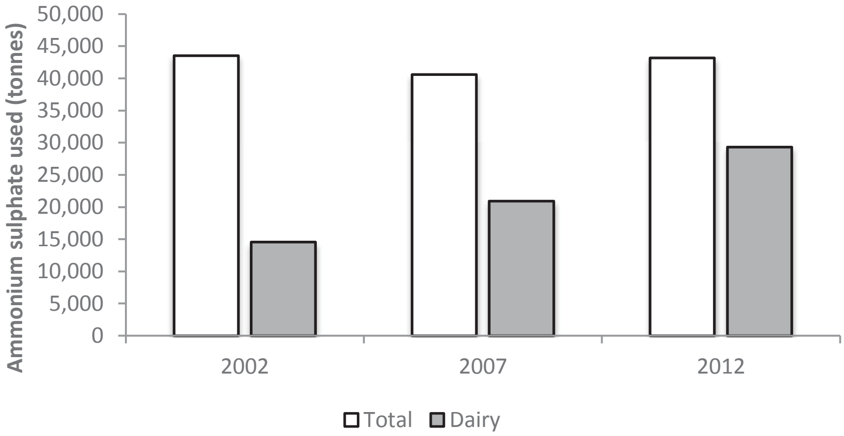


Figure 2.11: Tonnes of ammonium sulphate used in New Zealand on all farms and only dairy farms in 2002, 2007 and 2012.
Data source: Statistics New Zealand (2002, 2007, 2013a).

Imports

New Zealand imports of ammonium sulphate have been variable from 1990 with no upward trend as seen in urea imports, although recent spikes in price are apparent (Figure 2.12)

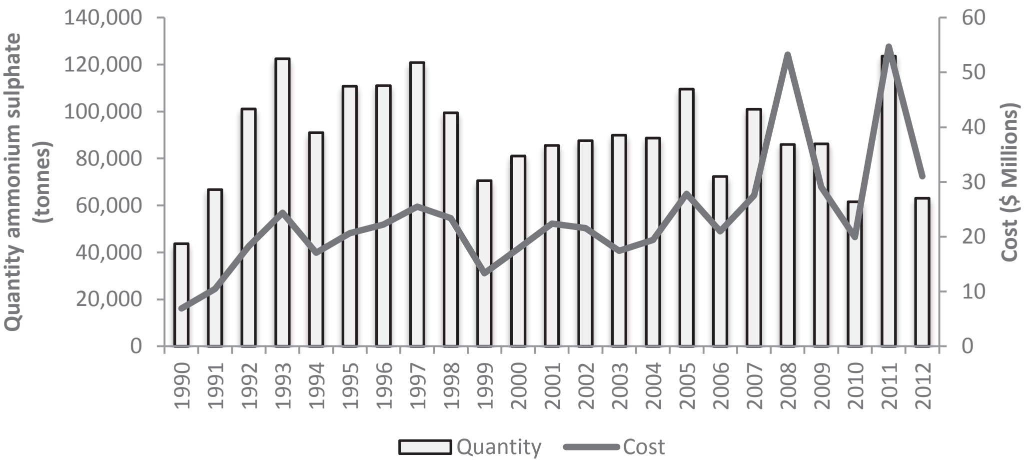


Figure 2.12: The quantity (bars) and cost (line) of ammonium sulphate imported into New Zealand between 1990 and 2012.
Data source: Statistics New Zealand (2013b).

2.3.1.2. Phosphorus fertiliser

Phosphorus is essential for all plants and animals and it one of the three key fertilisers in agricultural use. Global demand for phosphorus is heavily increasing as fertilisers are being excessively used (Rosemarin et al., 2009), primarily for industrialised agriculture to provide for increasing population growth (Ashley et al., 2011). As a result, a heavy increase in phosphorus mining has been observed (Ashley et al., 2011). Unlike nitrogen, there is no synthetic alternative for phosphorus.

Superphosphate fertiliser is the most commonly used form of phosphate fertiliser. First manufactured in 1839 in England, superphosphate is mainly made up of calcium, sulphur and phosphorus (Techhistory, n.d.). Aerial spreading of superphosphate in New Zealand began after WWII and superphosphate production in New Zealand peaked in the 1970s at over 3 million tonnes per year (Techhistory, n.d.). However, removal of government subsidies in the 1980s resulted in production halving, and it has since remained below peak levels.

Before 1950, the largest source of phosphorus for agricultural use was manure (Ashley et al., 2011). Bones and ancient guano deposits (from bird droppings) also provided a source of extracted phosphorus (Ashley et al., 2011). It is no surprise then that the first two fertiliser works in New Zealand were adjacent to freezing works (slaughterhouses or abattoirs) (Duncan, 2012). A major imported source was guano excavated from several islands, such as Nauru and Christmas Island where bird droppings had been accumulating for millions of years (Pearce, 2011). Now phosphate rock is the main source of phosphorus used in fertilisers.

Dairy farming is the second largest user of superphosphate after sheep and beef farms combined, albeit the latter two covering a much larger area. While dairy farms apply higher rates of superphosphate per hectare resulting in greater cadmium build-up (explained in Chapter 6), sheep/beef farms (which are usually situated on steeper slopes) have a greater potential for phosphorus surface runoff to waterways by soil erosion (explained in Chapter 5).

There is little data available on superphosphate use on farms. Data is lacking in some years on superphosphate or other phosphate use and the two are categorised interchangeably between years. In 2012, farm surveys determined 34% of superphosphate was used on dairy farms (Statistics New Zealand, 2013a), about the same proportion as in 2007 (Statistics New Zealand, 2007). Total phosphorus fertiliser use (including superphosphate) decreased between 1996 and 2012 (Figure 2.13).

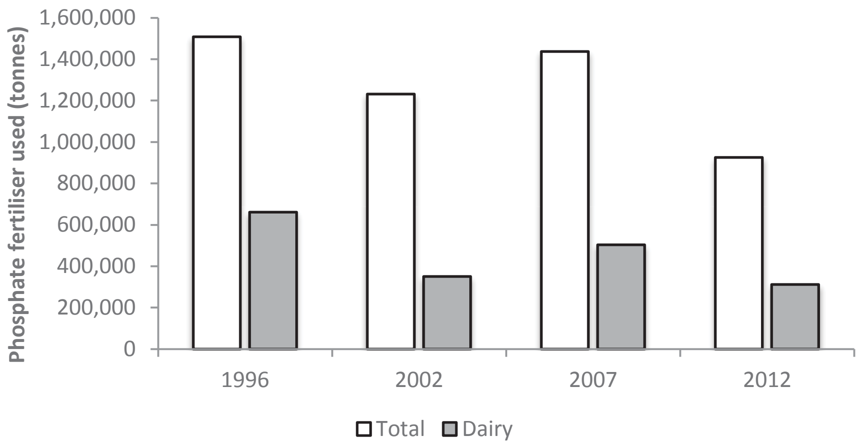


Figure 2.13: Tonnes of phosphate fertiliser used in New Zealand on all farms and only dairy farms in 1996, 2002, 2007 and 2012.
Data source: Statistics New Zealand (1998, 2002, 2007, 2013a). **Notes:** It is expected that most of phosphate use is superphosphate but data is categorised differently each year. In 2012, only estimates on superphosphate use were available; other phosphorus fertilisers could have been categorised under 'all other fertilisers', therefore making it difficult to determine how much was used.

A small amount of superphosphate is also imported ready-made, mainly from Australia, USA, China and parts of Northern Africa. However, in the past decade New Zealand has reduced its imports of superphosphate (Figure 2.14). This could be due to increases in domestic manufacturing or reductions in demand. The cost of imported superphosphate fertiliser and total imported phosphate fertiliser are very similar in most years, indicating New Zealand does not import many other types of phosphate fertiliser. However, a large notable exception occurred in 2006 (Figure 2.14).

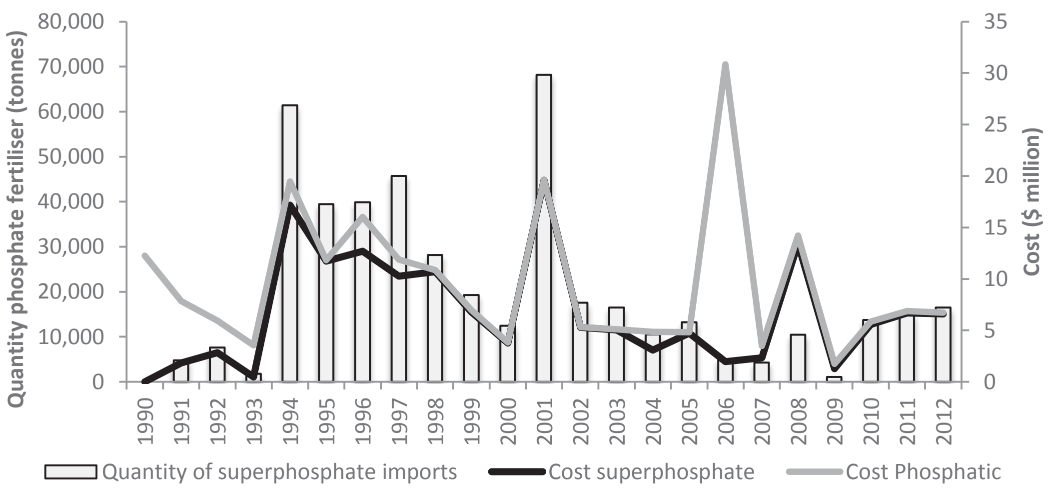


Figure 2.14: Quantity (bars) and cost (lines) of imported superphosphate and phosphatic fertilisers.
Data Source: Statistics New Zealand (2013b). **Notes:** Data on the quantity of phosphate fertiliser was not available.

Phosphate rock imports

New Zealand manufactures superphosphate from imported ingredients, mainly rock phosphate and sulphur. Additionally, some types of rock phosphate can also be applied directly to land without manufacturing. New Zealand is the 5th largest global importer of natural calcium phosphates by economic value (Index Mundi, 2013d). Imports of phosphate rock have decreased in the past seven years (Figure 2.15), perhaps due to decreasing demand.

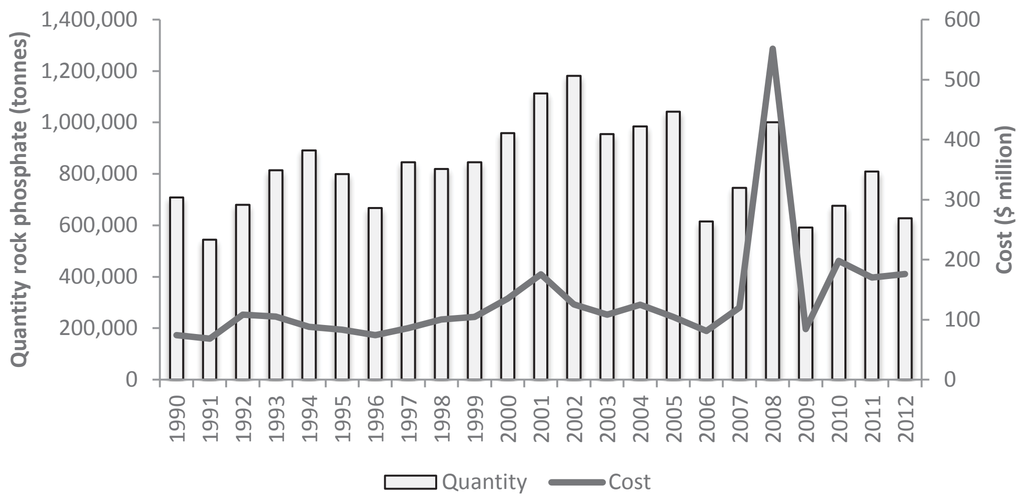


Figure 2.15: Quantity (bars) and cost (line) of rock phosphate imports between 1990 and 2012.
Data Source: Statistics New Zealand (2013b).

2.3.1.3. Combination fertilisers: DAP

Most of the combination fertiliser used in New Zealand is Di-ammonium phosphate (DAP). DAP is officially classed as a nitrogen fertiliser, although it contains about 18% nitrogen and 40% phosphorus. Total DAP use has decreased since 1996 (Figure 2.16). In 2012, dairy used about 38% of the DAP in New Zealand, down from 54% in 1996 (Figure 2.16).

Imports

In 2012, 80% of the imported fertiliser containing two or more elements was DAP (Statistics New Zealand, 2013b). Like phosphate imports, the quantity of DAP imports has decreased significantly in the past few years. However, notably the total cost of imported DAP has continued to increase (Figure 2.17).

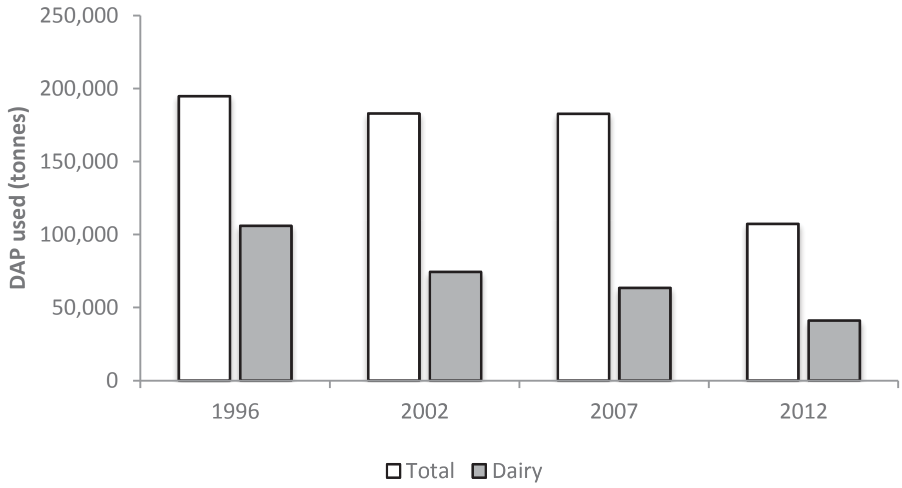


Figure 2.16: Tonnes of DAP fertiliser used in New Zealand on all farms and only dairy farms in 1996, 2002, 2007 and 2012.
Data source: Statistics New Zealand (1998, 2002, 2007, 2013a).

Figure 2.17: Quantity (bars) and cost (line) of DAP fertiliser imports to New Zealand between 1990 and 2012.
Data source: Statistics New Zealand (2013b).

2.3.1.4. Potassium fertiliser

Potassic fertiliser use in New Zealand decreased significantly from 2002 to 2007, but increased from 2007 to 2012 (Figure 2.18). This trend could be attributed to data deficiencies, e.g. fertiliser categorised differently between years. Dairy farms used 66% of potassium fertiliser in 2012 (Figure 2.18) (Statistics New Zealand, 2013a).

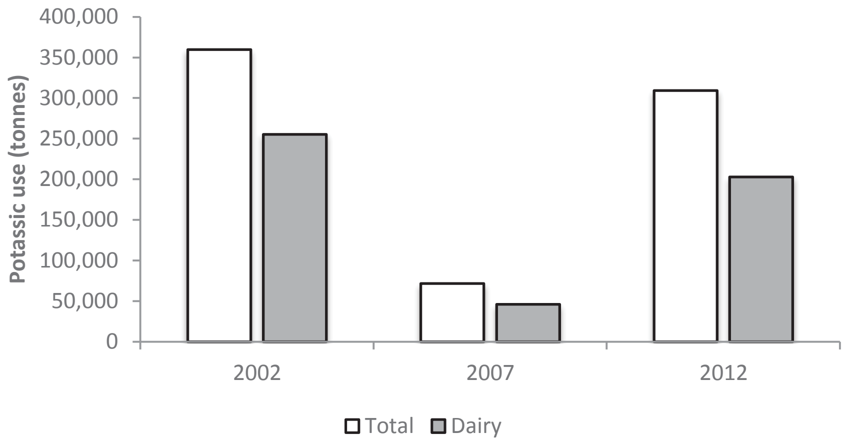


Figure 2.18: Tonnes of potassium fertiliser used in New Zealand on all farms and only dairy farms in 2002, 2007 and 2012.
Data source: Statistics New Zealand (2002, 2007, 2013a).

Imports

Approximately 70% of potassium is imported from Canada and about 20% from Europe (Statistics New Zealand, 2013b). Potassic fertiliser imports have been decreasing in recent years (Figure 2.19).

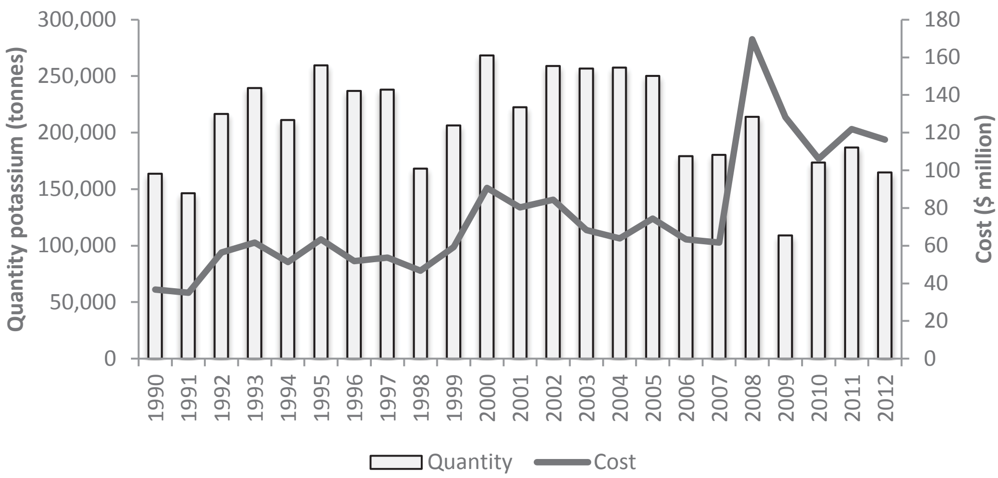


Figure 2.19: Quantity (bars) and cost (line) of potassium fertiliser imports to New Zealand from 1990-2012.
Data source: Statistics New Zealand (2013b).

2.3.1.5. Cost of fertiliser for the dairy industry

The cost of fertilisers for dairy farming from 2008-2012 was estimated using the proportions of fertiliser used on dairy farms from surveys in 2007 and 2012 (Table 2.2) (Statistics New

Zealand, 2007). During this time dairy increased their proportional use. Estimates show that dairy farmers spent about $503 million on fertiliser imports in 2012, including phosphate rock for superphosphate manufacturing (Table 2.3).

Table 2.2: Proportion of fertiliser used in New Zealand on dairy farms in 2007 and 2012

Fertiliser type	Proportion used for dairy	
	2007	2012
Urea	65	72
Ammonium sulphate	52	68
Phosphorus fertiliser and rock phosphate	35	34
DAP	35	38
Potassic	64	66

Table 2.3: Estimated cost of fertilisers imported into New Zealand for use on dairy farms from 2008-2012.

Fertiliser type imported	Cost per year NZ$ million				
	2008	2009	2010	2011	2012
Urea	265.9	110.4	135.9	268.8	228.7
Ammonium sulphate	27.7	16.3	12.0	35.0	21.1
Phosphate	5.0	6.2	2.0	2.3	2.3
Rock phosphate	193.0	29.5	67.4	57.8	59.9
DAP	124.1	97.4	130.3	130.6	114.8
Potassic	108.6	82.6	68.7	79.4	76.2
Total	**724.2**	**336.7**	**416.3**	**573.9**	**502.9**

2.3.2. Feed supplements

Imported feed supplements are now used by about 85% to 90% of dairy farms (DairyNZ, 2013a) with the purpose of grazing cows off the milking area and/or extending lactation periods and increasing stocking rates. Dairy farmers traditionally relied on off-farm grazing, import of silages (such as maize and cereal), or growing silage crops on the farm. Recently, there has been an increasing trend to buy in overseas stock feeds to increase or lengthen production phases, particularly when drought or other risk periods have reduced pasture growth (MacLeod & Moller, 2006). Imported ingredients include bran and pollard, cottonseed, palm kernel expeller, linseed (MacLeod & Moller, 2006), barley, maize, soyabean meal, tapioca, and canola meal.

2.3.2.1. Palm Kernel Expeller

Of particular importance in the use of imported feed supplements is palm kernel expeller (PKE), also known as palm kernel extract, palm kernel meal (PKM) and palm kernel cake (PKC). PKE is the easiest product to associate with dairy farming as all of the PKE imported into New Zealand is used for dairy feed. Other feed supplements are difficult to attribute solely to dairy as they are used in other agricultural sectors. Furthermore, the use of PKE has increased rapidly in the last decade and is perhaps the most controversial because of its deleterious offshore impacts (explained in Chapter 8). Thus, PKE will be the only imported feed supplement discussed here. Imports of PKE into New Zealand began in 1992 with 15 tonnes. Since then imports have increased substantially, from 408 tonnes in 1999 to nearly 1.4 million tonnes in 2012 (Figure 2.20) (quantity and cost figures are listed in Appendix A) (Statistics New Zealand, 2013b). In 2012, New Zealand spent $274 million on PKE imports (Statistics New Zealand, 2013b).

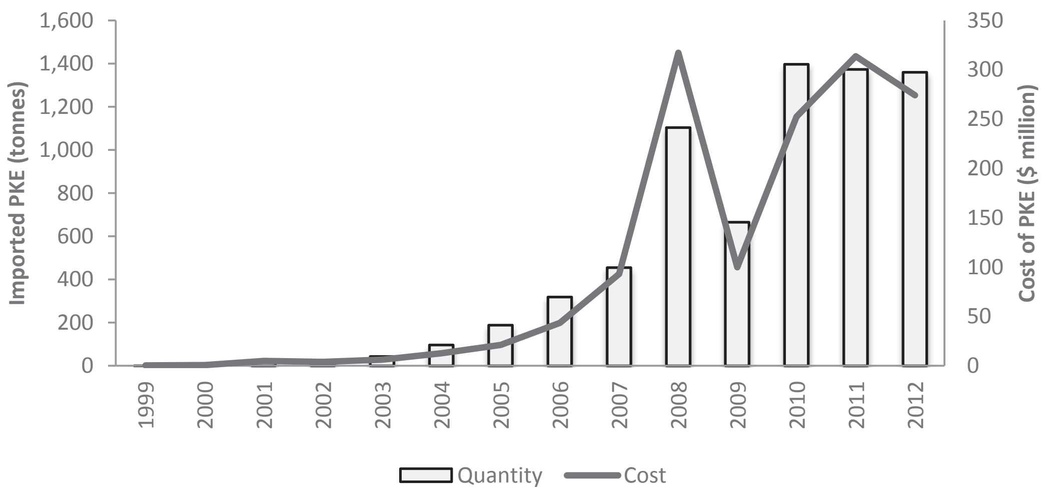


Figure 2.20: Quantity (bars) and cost (line) of palm kernel expeller (PKE) imports into New Zealand from 1999 to 2012.
Data source: Statistics New Zealand (2013b).

Demand for PKE in New Zealand escalates particularly in times of drought, flooding and volatile milk prices (Carlton, 2011). These demand peaks are likely to increase as drought events are predicted to amplify with climate change (Ministry for the Environment, 2009a). The impact of drought events were particularly evident in 2008, when PKE imports rose significantly, and early 2013, although statistics on 2013 PKE imports were not available at the time of data gathering.

What is palm kernel?

PKE is a product left after the oil is extracted from the palm seeds of the oil palm. After oil extraction and the flesh of the fruit has been stripped, the solid part of the seed kernel is mashed to make PKE. The main products from oil palm are crude palm oil, palm kernel oil (used mainly in foods, cosmetics and biofuels) and PKE, used predominantly for animal feed and power generation. PKE has been used as an animal feed, particularly for ruminants, for a long time (Nordin et al., 2005). Increasingly, PKE has been used as a standard input of production for dairy farming in New Zealand.

Where is it grown?

Palm plantations are mainly situated in South East Asia. Indonesia is the largest producer of palm kernel with over half of the total production and together with Malaysia produces 86% of the total PKE (Figure 2.21) (Index Mundi, 2012).

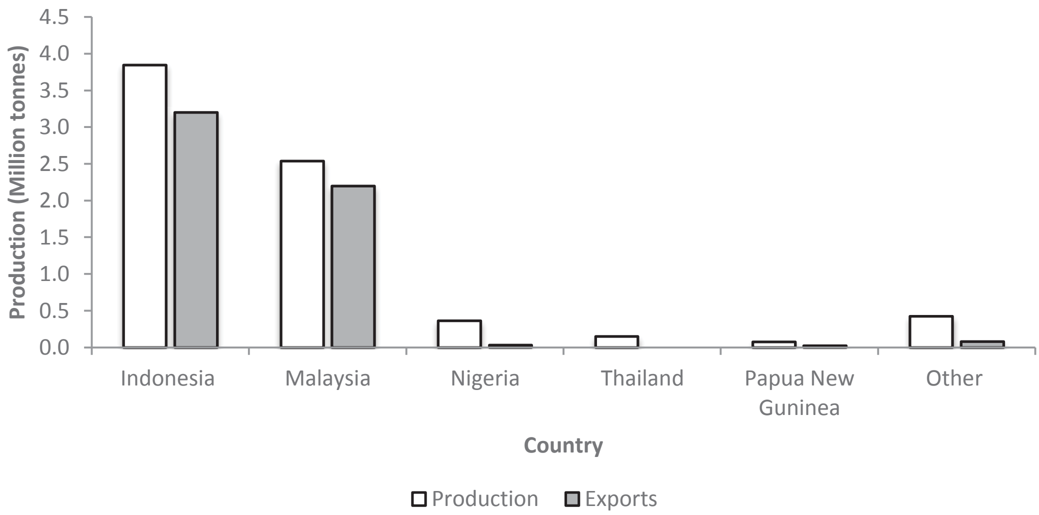


Figure 2.21: The total production and exports of PKE for the top five PKE producing countries and the rest of the world (other) in 2012.
Data source: Index Mundi (2012).

Importers and consumers of PKE

In 2012, New Zealand imported 30% of the total globally traded PKE and consumed 23% of the total produced (Figure 2.22) (Index Mundi, 2012). In New Zealand all of the imported PKE is used for dairy farming but in the European Union (EU) about 80% is used for animal feed (Index Mundi, 2012), while the remaining is used for other uses, such as energy production.

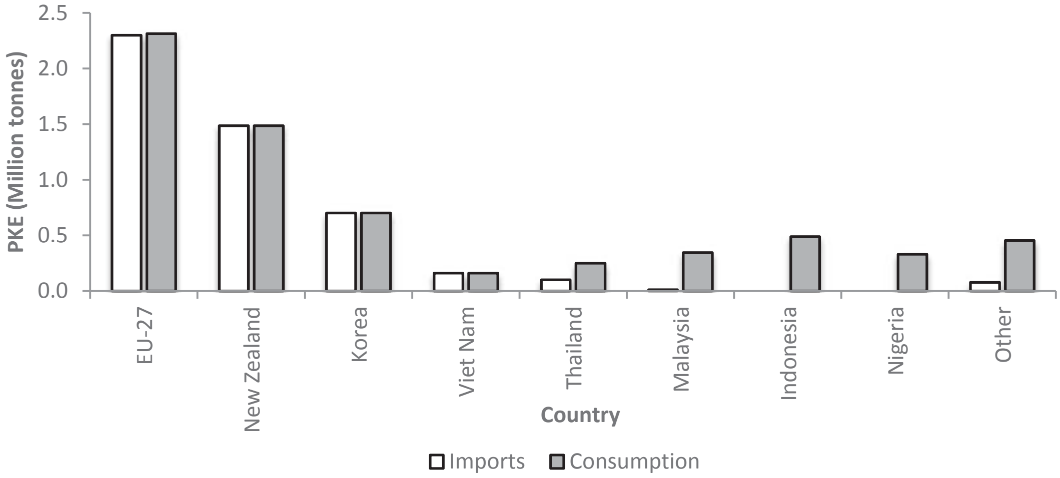


Figure 2.22: Global imports and consumption of PKE in 2012 for the top consumers.
Data source: Index Mundi (2012). **Notes:** EU-27 = 27 countries in the European Union

Economic value to palm oil producers

New Zealand spent $274 million on PKE in 2012 and over $300 million in 2008 and 2011 (Figure 2.20) (Statistics New Zealand, 2013b), associated with higher PKE prices these years. According to the Malaysian Palm Oil Board, PKE is “an important product from the oil palm industry” which generates substantial export earnings for Malaysia (Nordin et al., 2005, p. 37). In 2012, 52% of the value of PKE imported into New Zealand was from Malaysia ($143 million) (Statistics New Zealand, 2013b). Thus, New Zealand imports of PKE are an important part of the palm industry, as New Zealand is the biggest single country importer of PKE.

2.3.3. Irrigation

Water for irrigation is another input heavily increasing in the dairy industry, particularly as intensification has expanded into drier areas of Canterbury and Otago. Resource consents (explained in Chapter 4) are generally required to remove water for irrigation, drinking water supply, industrial and manufacturing works, and other activities (Ministry for the Environment, 2007a). Irrigation accounted for 75% of national freshwater consents in 2010, followed by 9% each for industrial and drinking uses, 6% for stock and 1% for hydro (Rajanayaka et al., 2010). Of the volume of water allocated (rather than number of consents), over half was allocated for irrigation in 2010 (Figure 2.23) (Rajanayaka et al., 2010). In this analysis, allocations for stock water represent only those requiring consent. However, the majority of stock takes are permitted activities.

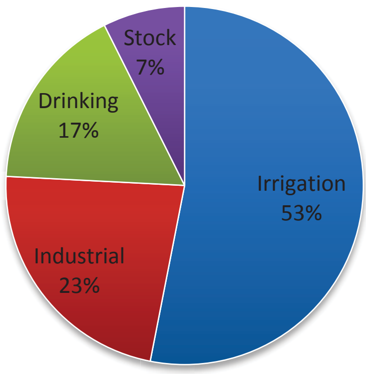


Figure 2.23: Distribution of annual consumptive water allocation by use excluding hydro generation takes in New Zealand in 2010.
Data source: Rajanayaka et al. (2010).

The amount of water takes consented by Regional Councils has increased substantially since 1999 as well as the amount of land that is under irrigation. Excluding a Southland hydro take,[4] annual allocated water use increased by 86% between 1999 and 2010 (Lincoln Environmental, 2000; Rajanayaka et al., 2010). Demand for freshwater is particularly noticeable in Canterbury, which used over 60% of the national allocated irrigation water in 2010 (Figure 2.24) (Rajanayaka et al., 2010). Furthermore, almost 90% of the allocated water within Canterbury was used for irrigation, mainly for pastoral agriculture.

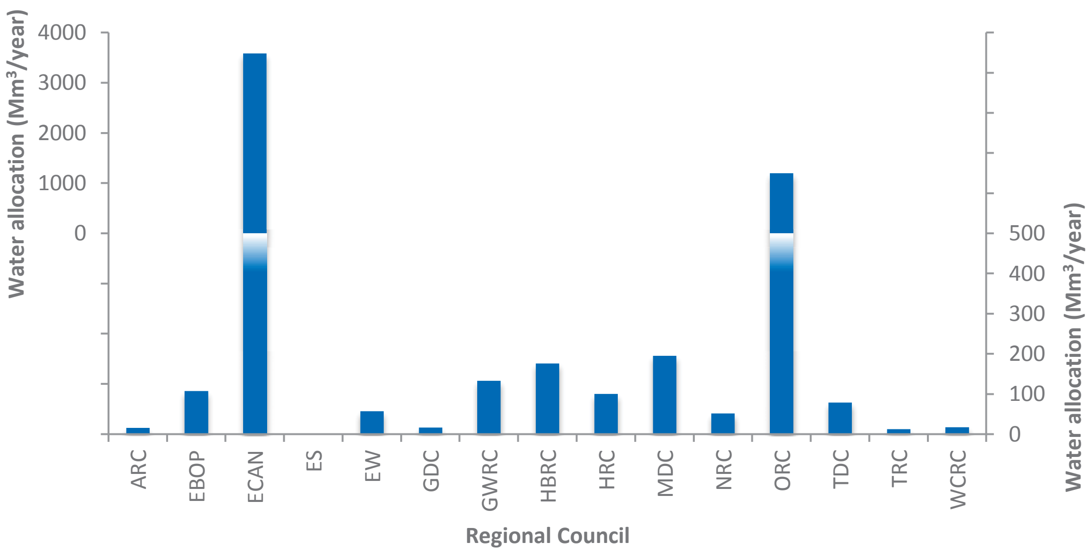


Figure 2.24: Regional Council annual water allocation (Mm^3/year) for irrigation in 2010.
Data source: Rajanayaka et al. (2010). **Notes:** Mm^3/year = Million cubic metres per year. The bottom panel shows water allocation by Regional Council on the right axis and the top panel shows ECAN and

[4] A hydro generation plant in Southland discharges water out to sea and accounts for 59% (16 billion m^3) of the total national consumptive allocation (27 billion m^3) (Rajanayaka et al., 2010).

ORC allocation on the left axis. Faded columns extend to the top panel due to the much higher allocations.
Abbreviations: ARC: Auckland Regional Council; EBOP: Environment Bay of Plenty; ECAN: Environment Canterbury; ES: Environment Southland; EW: Environment Waikato; GDC: Gisborne District Council; GWRC: Greater Wellington Regional Council; HBRC: Hawkes Bay Regional Council; HRC: Horizons Regional Council; MDC: Marlborough District Council; NRC: Northland Regional Council; ORC: Otago Regional Council; TDC: Tasman District Council; TRC: Taranaki Regional Council; WCRC: West Coast Regional Council. EBOP now changed to Bay of Plenty Regional Council.

Increasing amounts of land are being irrigated in New Zealand; however, data on the actual area that is irrigated is not available. Instead only the area of land that has resource consent to irrigate or is equipped to irrigate is reported on. From 1965 to 1999, the area consented for irrigation increased by 55% per decade, totalling 600,000 hectares (ha) in 1999 (Lincoln Environmental, 2000). This increased to 972,000 ha in 2006 (Aqualinc, 2006) and to 1,076,502 ha in 2010 (Rajanayaka et al., 2010). In 2010, two-thirds of irrigated land was in Canterbury (680,000 ha) and another 16% in Otago (168,000 ha) (Rajanayaka et al., 2010). In 1999, 31% of the irrigated area was used for dairy farming (Lincoln Environmental, 2000). Irrigated dairy land was not measured in 2006 and 2010. In 2006, pasture accounted for 31% of the irrigated area and 76% in 2010 (Figure 2.25). Differences may be attributed to the classification of data between years. Estimations of land area with irrigation systems were considerably less than those with irrigation consents. In 2007, the reported area equipped for irrigation was 619,293 ha, of which 37% (230,555 ha) was under dairy farming and 62% (385,271 ha) was in the Canterbury region (Statistics New Zealand, 2007). In 2012, this increased to 721,740 ha nationally, with 49% in dairy farming (352,414 ha) (Statistics New Zealand, 2013a).

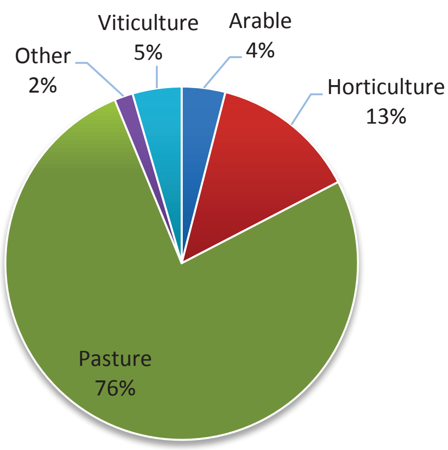


Figure 2.25: Proportion of consented irrigated land area in different agricultural uses in 2010.
Data source: Rajanayaka et al. (2010).

2.3.4. Summary of inputs

Dairy inputs have substantially increased in the past two decades. Irrigation increases were measured by the change in the reported area equipped for irrigation, as data was non-existent for water use on dairy farms for most years. The area reported under irrigation increased by 123% between 1999 and 2012. Changes in urea and PKE imports and costs are shown in Figure 2.26. Urea use was used to represent nitrogen fertiliser as most of the imported N fertiliser is urea. PKE and urea increases from 1999 to 2012 are shown in Appendix B revealing much smaller increases in urea imports during this period.

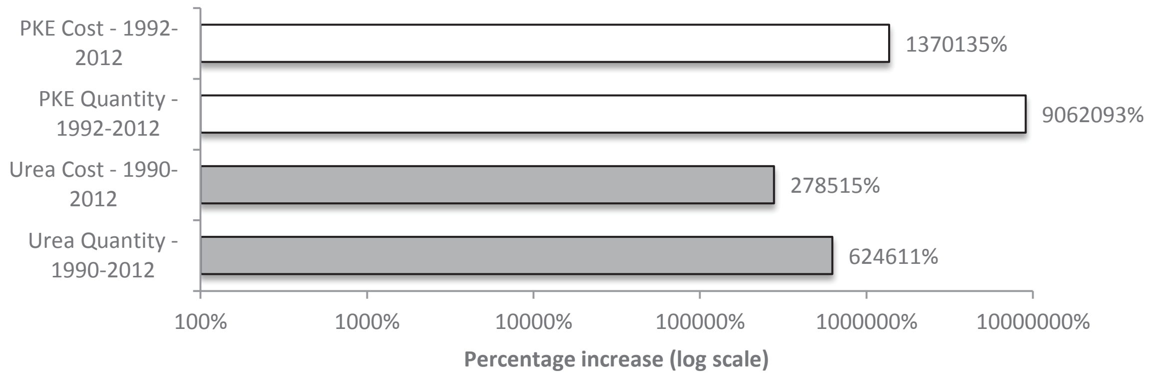


Figure 2.26: Percentage increase in the imports (quantity and cost) of PKE (white bars) and urea (grey bars) for dairy farming.
Note: Graph is presented on a logarithmic scale.

2.4. Borrowing for dairy

Dairy farming has required large bank loans in order to pay for the increasing price of land and use of infrastructure. In early 2012, New Zealand's agricultural debt was $47 billion (Table 2.4). The growth of agricultural debt has been increasing while the growth of debt in other sectors has slowed (Reserve Bank of New Zealand, 2012). Agricultural debt reached almost $50 billion in early 2013 (Reserve Bank of New Zealand, 2013). Dairy farming represents about 65% of total agricultural debt ($30 billion in 2012). However, debt levels are not evenly spread over all farms; for example, 20% of dairy farms have 73% of the total debt while 60% of farms have less than 10% of the total (Greig, 2010). Obviously debt requires interest to be paid; an estimated $3 billion of interest was paid on dairy loans in 2012 (Table 2.4).

Table 2.4: Agricultural and dairy farm debt in New Zealand

Year	Agriculture debt by registered banks ($billion)		Dairy debt proportion of total agriculture	Estimated interest payments ($billion)	
	Total agriculture	Dairy		Total agriculture	Dairy
1990	5.1				
2003	18.7	11.2	60%	1.8	1.1
2004	21.2	12.3	58%	2.1	1.2
2012	47.6	30.5	64%	4.6	3.0
2013 - Jan	49.8				

Data source: Reserve Bank of New Zealand (2012, 2013) **Notes:** An interest rate of 9.7% was used which was an approximate interest rate of banks in July 2013. Estimates of dairy debt were not available for 1990 or 2013.

2.5. Conclusion

Dairy production and input trends over the past two decades indicate large-scale intensification has occurred, particularly in areas not historically under dairy production, such as Canterbury. Environmental conditions (for example, lower rainfall) in Canterbury require irrigation to maintain production. The Canterbury region is the biggest water user in the country, mostly for irrigation. Although dairy export revenue has accelerated, dairy farm debt has also dramatically increased more rapidly than other agricultural sectors. Interest payments on debt are significant and are detached from the equation when considering economic benefits. Impact trends associated with production are described in further chapters.

Chapter 3. EXTERNALITIES: ENVIRONMENTAL TRADE-OFFS

Conventional economics is a form of brain damage. Economics is so ***fundamentally*** *disconnected from the real world it is destructive....If you ask the economists, in [an economic equation] where do you put the ozone layer? Where do you put the deep underground aquifers of fossil water? Where do you put topsoil or biodiversity? Their [conventional economist] answer is, 'oh they are externalities.' Well then you might as well be on Mars, that economy is not based on anything like the real world…Nature performs all kinds of services [which] …are vital to the health of the planet. Economists call these externalities, that's* ***nuts!***

(David Suzuki in Sustainable Man, 2012).

The current dominant economic system of valuation (using the measurement of GDP) and decision-making does not adequately incorporate the costs of environmental impacts or the true value of natural resources and services (Bertram, 2013; Costanza et al., 1997; Daily et al., 1997; Daly, 2005). This chapter briefly describes how the contemporary economic system fails to account for environmental impacts. The concept of environmental externalities is discussed as well as methods that can incorporate externalities into economic analyses. These concepts can be applied to the New Zealand dairy farming industry.

3.1. Conventional economics

Conventional economics, also known as mainstream or neoclassical economics, portrays the economy as a closed and linear system, with economists in this field committed to the growth agenda (in order to develop, the economy must grow) (Daly, 2005). In this view, the economy is seen as operating outside of the environment rather than within the bounds of the natural world (Turner et al., 1994). However, the economy works as an open system within other areas: it extracts resources from the environment, processes them into goods, and disposes or discards wastes back into the environment (Daly, 2005; Prugh et al., 1999; Turner et al., 1994). Waste is generated in all parts of the economy (extraction, processing and consumption). If the receiving environment cannot assimilate the waste, the environment will become contaminated and cause harm or damage to ecosystems. Absolute efficiency in this conventional system of economic production process cannot be achieved; pollution is inevitable and, generally, more economic production means more pollution (Czech, 2013).

Despite the view of traditional economic theories, the economic system is physically constrained by the environment (Turner et al., 1994). It cannot operate without the support of the ecological system and its interrelationships; in other words, the economy is a “subsystem of the finite biosphere that supports it” (Daly, 2005, no page numbers). As the economy grows, the economic system gets bigger with respect to its ‘fuel system’ (the ecosystem that supports it). However, the ecosystem is finite and cannot expand (Daly, 2005). Hence, if the economy grows too large, the ecosystem will be consumed and will result in the collapse of both the economy and the ecosystem (Prugh et al., 1999). For this reason, there is a physical limit to economic growth (Czech, 2013; Daly, 2005). Due to these inconsistencies in the system, resource use must be balanced and work in a circular closed system (one which produces no external products).

Often the response to environmental issues and major economic ills is to maximise economic growth (Daly, 2005). However, this has not necessary benefited society. Ecological economist Herman Daly notes that “economic growth already has become uneconomic” (Daly, 2008, p. 2). The expansion of the economic system “increases environmental and social costs faster than production benefits, making us poorer not richer” (Daly, 2008, p. 2). Conventional economics does not differentiate between beneficial and harmful environmental outputs. For instance, cleaning up pollution from an oil spill would generate economic activity (assuming it would cost money), and therefore increase a nation’s gross domestic product (GDP), which is good in a conventional economist’s view. However, the oil spill would generate ecosystem impacts (harmful environmental outputs or costs), of which could be valued as the cost to clean up. Similarly, spending on doctor’s visits and medication increases GDP. In this system, unhealthy societies and environmental disasters are good for the economy provided money is spent to remedy or mitigate them. However, these outputs are generally not beneficial to society or the environment. Costs and benefits are not separated when counting GDP and are all counted as ‘activity’ (Daly, 2008). Hence, GDP is flawed as a measure of progress and wellbeing in a society as it ignores many of the costs.[5] There are alternative indicators that can be used to more thoroughly include spheres of natural and social capital, such as the Genuine Progress Indicator (GPI), ecological footprint, and Human Development Index (HDI).

3.2. The production of externalities

At present, polluters only pay a small share of the costs associated with the pollution they produce. They generally only pay for the input and production costs of a product, i.e. the internal private costs or economic costs. The full social costs of production or consumption activities are made up of internal private costs and external costs (public or social cost) (Pigou, 1920, cited in Turner et al., 1994). External costs include the pollution and harm caused from activities, termed externalities.

The profit motive of competitive market systems gives companies' an incentive to pollute if it allows costs to be reduced and profits to increase (Hackett, 1998). If companies' can avoid paying for pollution, they will generally choose not to pay in order to minimise costs. This avoided cost of pollution is effectively a producer subsidy and it understates the true social costs of production (Hackett, 1998). This leads to overproduction, as companies can produce more when their costs are lower. If externalities were incorporated into production costs, production may become uneconomic or items would be much more expensive, thus reflecting the true social cost of the product.

Most environmental externalities are public in nature (Perman et al., 1996). What this means is that externalities affect the utility or wellbeing of a third party (the public) and are uncompensated by the producer of the effect (Hackett, 1998; Perman et al., 1996; Turner et al., 1994). Thus, the wider community (the public) is left to deal with the costs of the effects (Turner et al., 1994), whether these involve, for example, the cost of cleaning up pollution or the cost of having a degraded environment. These costs could be in the form of government remediation funded by public taxes (Abell et al., 2011), or public health costs associated with an unhealthy environment or contamination, among many others. If remediation is not carried out, resources are left to degrade, and future generations will eventually have to pay the price.

[5] For a more detailed discussion on the flaws of GDP and an alternative of steady state economics see Daly (2005, 2008).

3.3. Ecological economics

An alternative to conventional economics is ecological economics, which is a trans-disciplinary approach that brings natural capital (oceans, rivers, land, forests, wetlands, agricultural land etc.) into the frame of economics (Costanza & van der Belt, 2013). Ecological economics views resource degradation in terms of a reduction to the value of ecosystem services (Abell et al., 2011). Ecological economics aims to make ecosystem services visible. Ecosystem services are the products, benefits and functions that the ecosystem provides to humanity including nutrient cycling, air and water purification, flood mitigation, climate regulation, pollination of crops, seed dispersal, biodiversity, and food provision, among many others (Daily et al., 1997). The value of these services would be immense and to replace them would be very costly, if even possible. Costanza et al. (1997) valued the world's ecosystem services and natural capital at US$33 trillion per year in 1997, almost twice the global GDP at the time (US$18 trillion). A tool used in ecological economics is total economic valuation (TEV), which can be used to value things not included in traditional market economies.

3.3.1. Total economic valuation

The valuation of environmental functions and goods is important to weigh up the costs of harmful activities to obtain a more holistic value of production and address environmental externalities. Unfortunately, identifying and assessing the significance of pollution externalities is often a difficult task. Thus, economic valuation has limitations: there are many things that cannot be ascribed a money value (Andrew Steer, cited in Dresner, 2008). Reasons for this could include lack of information on the object in question, the object or product not been counted in the traditional market economy (e.g. GDP), or it may be unfeasible to determine the value of impacts. Conversely, the non-measurable values may be the most important for humans and ecosystems. Hence, there needs to be a form of valuation.

To derive at a total economic value (TEV), cost-benefit analyses can be used to measure changes in environmental conditions on a single monetary scale, thus helping decision-makers incorporate all aspects of an activity (Edwards-Jones et al., 2000). Traditionally, only direct use values are included in cost benefit analyses as there are usually market prices that can be used to estimate changes in value (NIWA, 2010a). TEV is made up of use values (active) and non-use values (passive), and incorporates both market and non-market values (Figure 3.1). Non-market values are not included in the formal economy (e.g. GDP) because there are no markets where they are regularly brought and sold, and hence their market price cannot be easily determined (NIWA, 2010a).

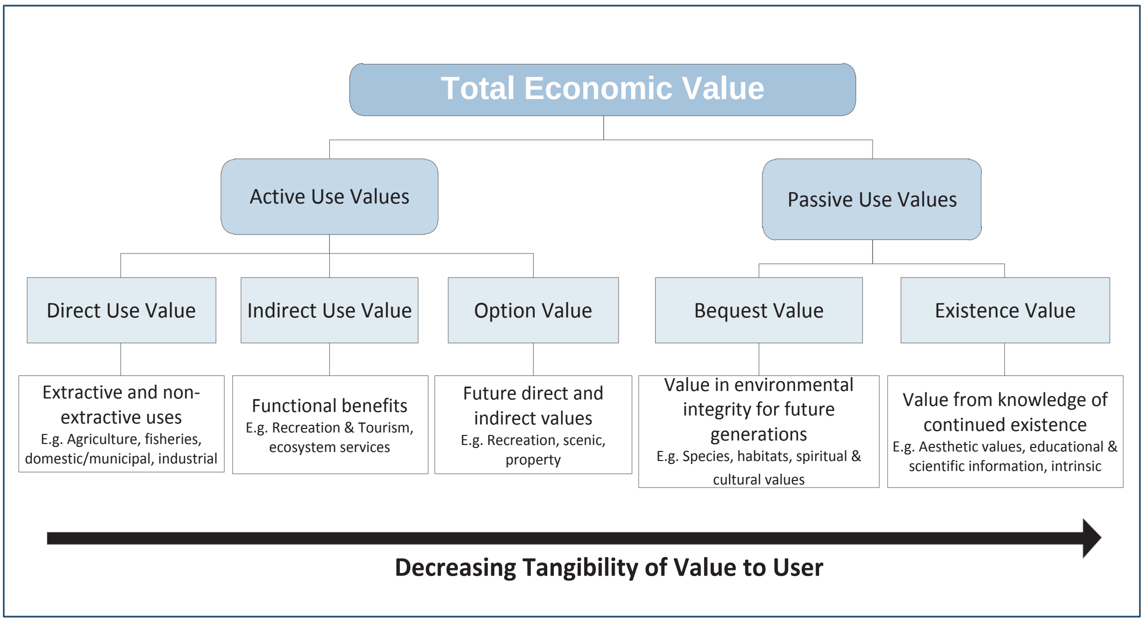


Figure 3.1: Overview of the components of Total Economic Value
Adapted from: NIWA (2010a).

Active use values are associated with the benefits derived directly or indirectly from a natural resource. These could be from consuming the resource directly or from secondary or non-consumptive uses including benefits from recreation, amenity and ecosystem services the environment provides (Edwards-Jones et al., 2000; Hatton MacDonald et al., 2004). Active use values are separated into direct use, indirect use and option value (Figure 3.1). Direct use values involve actual consumption of the resource and are usually production related (e.g. agriculture, water supply, fisheries) (NIWA, 2010a). Many direct use values are incorporated in market values and therefore have monetary values, or if not, are easier to measure from market and survey data compared with indirect values (Awatere, 2005). Indirect use values are functions that support or protect direct use activities. They include ecosystem services which the environment provides that benefit human use or welfare, and recreation provided by the environment that does not include actual consumption of a resource (Awatere, 2005; NIWA, 2010a). Option value refers to the value received for preserving a resource for future use (Awatere, 2005; Hatton MacDonald et al., 2004; NIWA, 2010a). These are also often categorised as non-use values.

Passive use values include bequest and existence/intrinsic values (Figure 3.1). Bequest values describe the future value of environmental and social goods, such as the value of species survival, and spiritual and cultural values. Existence or intrinsic values covers aesthetic, educational and scientific values – or fundamentally, the knowledge that these values exist, irrespective of human use and benefit. Passive values may be particularly hard to value in a monetary sense because of the ethical reasons in doing so (particularly in regards to cultural values) (Awatere, 2005), and because values exist irrelative of human use.

Non-market valuation techniques

Non-market valuation (NMV) techniques can be used to estimate the community wide value of changes in environmental quality or to measure the benefits and costs of developing or protecting natural resources. In essence, NMV can measure parts of TEV. Existing market data may be used to determine parts of TEV but many values derived from resources can only be estimated by using related or hypothetical market data. Revealed or stated preference valuation can be used to estimate non-market values (Awatere, 2005; Hatton MacDonald et al., 2004).

Revealed preference techniques use existing information from related markets to determine a value of non-market goods (Hatton MacDonald et al., 2004). Essentially, values for the non-market items are 'revealed' from the consumer's behaviour in a related market (Awatere, 2005; Hatton MacDonald et al., 2004). Two commonly used revealed preference techniques are the hedonic price method and travel cost method[6]. Revealed preference techniques require the use of existing related market data and, therefore, can only be used in limited situations. Consequently, only use values can be estimated.

Stated preference techniques estimate non-market values based on the indicated preferences of individuals. Surveys are used to determine the non-market benefits of resource use by people. Stated preference methods can estimate use values where no related market data are available and can also estimate non-use values, while revealed preference techniques cannot (Hatton MacDonald et al., 2004). The Contingent Valuation Method (CVM) is the most widely used stated preference technique (Awatere, 2005; Hatton MacDonald et al., 2004). In this method, respondents willingness to pay (WTP) for a given project or to avoid environmental

[6] The hedonic price method quantifies the effects of changes in environmental quality on property prices or wages as a proxy for the value of change in environmental quality. The travel cost method uses information on travel costs or the change in site visits as the distance from the site increases as a proxy to determine the access value of recreational resources.

changes is estimated, which can potentially embody non-use values. There are, however, limitations to this method, including: the high costs involved to carry out, limited information provided about people's preferences, and biases from respondents and researchers (Hatton MacDonald et al., 2004). Another stated preference technique is choice modelling (CM). Similar to CVM but respondents are asked to choose their preferred alternative between a series of hypothetical choice sets, each containing usually three or more alternatives (Baskaran, Cullen, & Colombo, 2009; Hatton MacDonald et al., 2004).

Economic tools cannot value all parts of a natural resource (Awatere, 2005). The natural environment has intrinsic value that recognises a natural resource exists within its own right. Intrinsic value is covered under TEV but, in practice, it often very difficult to ascribe a value to. Furthermore, some of the values derived from people's preferences and behaviours do not actually measure the worth of a resource, the cost associated with replacing a resource, or the cost of remedying or mitigating environmental impacts. Thus, all the measures explained above undervalue environmental degradation. Instead, non-market valuation techniques can help inform decision-makers to design policies and incentives, such as charges and subsides, which could, for example, encourage farmers to provide ecosystem services from agriculture (Baskaran, Cullen, & Takatsuka, 2009).

3.3.2. Application to New Zealand

The study of ecological economics is growing in popularity and use and has been applied in New Zealand, most notably in the Waikato River Independent Scoping Study (WRISS) for the management and valuation of restoring the Waikato River. However, most ecological economics projects receive little recognition from Government or industry institutions when considering the costs and benefits of activities. In the Manawatu-Whanganui region ecosystem services have been valued at $6 billion per year (2006 dollars) (van den Belt et al., 2009). This is considered a conservative estimate as it does not include some direct and indirect use values for some ecosystem types as well as passive values, due to lack of primary valuation studies. Of note in this study is the value per hectare attributed to dairy ($1,831/ha), compared to the highest value ascribed to wetlands ($43,320/ha). Although this work was never published, it was referred to in the WRISS report (discussed above).

3.4. Conclusion

The purpose of this chapter was to show how the traditional economic system has failed to incorporate environmental pollution and ecosystem benefits into accounting, and thus deemed them worthless. This has led to the production of environmental externalities. New Zealand is not exempt from this system and the dairy industry is a classic example of producing externalities. The dairy industry does not pay for its environmental pollution; furthermore, the intensive production model it follows relies on externalising the environmental costs. The growth of dairy farming has occurred without any balanced economic evaluation and awareness of the full environmental impacts and costs. It is likely that the lack of any cost on externalities has facilitated dairy expansion and intensification.

The examples of different valuing systems were provided to show that there are methods to incorporate nature and environmental pollution into economic frameworks to minimise environmental impacts. This can be done using regulation (e.g. through taxes and subsidies), voluntary initiatives, or economic initiatives (explained in Chapter 4). For further information on concepts of ecological economics, steady state economics, and ecosystem services see: (Bertram, 2013; Costanza et al., 1997, 1998; Daily et al., 1997; Daily & Matson, 2008; Daly, 2005, 2008; Daly, 1968; Howarth & Farber, 2002).

Chapter 4. Environmental Management or Economic Development?

We abuse land because we regard it as a commodity belonging to us. When we see land as a community to which we belong, we may begin to use it with love and respect.

(Aldo Leopold - A Sand County Almanac)

The environment sustains New Zealand's most important industries and provides a sense of belonging and heritage for many people. Thus, to be sustainable, management should provide for the environments' multiple uses and values, while sustaining future use. Environmental management can be implemented through regulatory, voluntary or economic measures. In New Zealand mainly regulatory and voluntary processes are used. This section provides an overview of some of the major processes and policies of environmental management in New Zealand in order to better understand the inadequacy in environmental protection that has occurred in some areas.

4.1. Regulatory Management

Regulation is often imposed by government departments by placing limits or targets on pollutants to restrict environmentally harmful activities. This is usually implemented through legislation. Legislation in New Zealand has been world leading in the area of sustainable management of the environment, but this is changing as the current New Zealand Government aims to amend some of the underlying principles of this legislation. In New Zealand legislation should allow Māori to be actively involved in environmental management (New Zealand Government, 1991); however, this rarely occurs except in a few cases where some iwi (Māori tribes) have fought for more management rights and arrived at agreements for their involvement (Awatere, 2005; NIWA, 2010b; Taiepa et al., 1997).

4.1.1. Resource Management Act 1991

The principle government legislation in New Zealand relating to the regulatory management of land, water, air and the coastal environment is the Resource Management Act 1991 (RMA). The RMA replaced over 20 major statutes and 50 laws relating to town planning and environmental management, such as the Town and Country Planning Act 1972, the Water and Soil Conservation Act 1967, and the Clean Air Act 1972. Previously, a single activity could involve rules under a number of different Acts, requiring proposers to go through multiple different consent processes (Palmer, 2013b). At the time of its implementation, sustainable development was coming into the forefront internationally with the World Commission on Environment and Development in 1987, and the concept of the RMA was leading worldwide. The overriding purpose of the RMA is to “promote the sustainable management of natural and physical resources” (New Zealand Government, 1991, Part 2, s5(1)). Sustainable management was described as management which sustains natural and physical resources for future generations; protects the life-supporting capacity of air, water, soil and ecosystems; and aims to avoid, remedy or mitigate adverse effects on the environment (New Zealand Government, 1991, Part 2, s5(2)).

Controls in the RMA are intended to be effects-based rather than activity-based; i.e. the effects of activities are managed rather than regulating individual activities. The problem with this approach is that it can result in solutions to environmental issues being reactive rather than proactive. Management responsibilities under the RMA are delegated to regional councils so the RMA is primarily implemented by local government. An overview of the processes in the RMA is shown in Figure 4.1. Environmental quality and management is achieved through:

- National and regional policy statements
- National environmental standards
- Regional and district plans, and
- Resource consents

National policy statements (NPS) are developed by central government on matters of national significance and these were ideally supposed to be developed in the 1990s after the implementation of the RMA. Despite this, there are currently only four national policy statements regarding: electricity transmission, renewable electricity generation, coastal policy and freshwater management (Ministry for the Environment, 2011a) with the NPS on freshwater management being developed 20 years after the RMA. The lack of national policy statements severely impedes the ability of the RMA to carry out adequate environmental protection. National environmental standards are set by the government and prescribe technical standards, methods or other requirements for environmental matters. The same standards must be enforced by each regional, city or district council but councils can impose stricter standards in some circumstances.

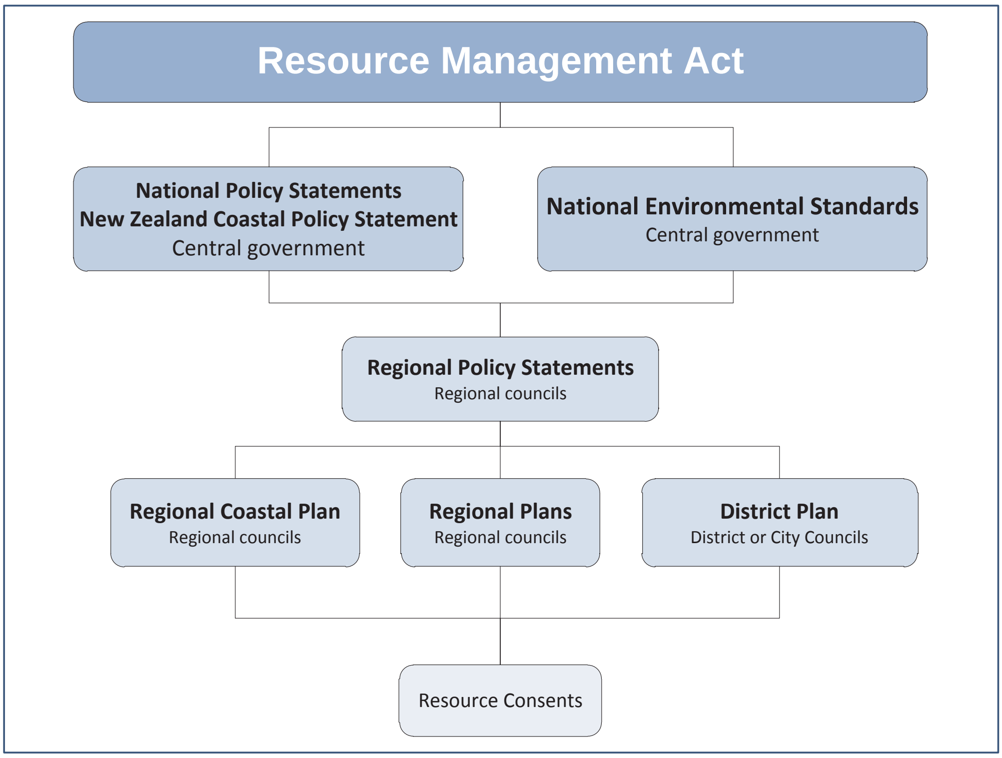


Figure 4.1: Key processes of the Resource Management Act (1991)
Adapted from: Ministry for the Environment (2013c)

Regional policy statements must be prepared for each region to provide broad direction on resource management. They set the direction for regional environmental management but must give effect to national policy statements. Regional plans assist the council in carrying out RMA functions and they concentrate on discharges of contaminants to land, air or water; water quality or quantity; the coastal marine area; soil conservation; and land use to avoid natural hazards. Regional councils must prepare a regional coastal plan but others are optional (Environmental Defence Society, 2013b). Territorial authorities (city or district councils) must prepare district plans that cover issues related to: effects of land use, noise, activities on the surfaces of rivers and lakes, and subdivision (Environmental Defence Society, 2013a; Palmer, 2013b). District plans must not be inconsistent with regional plans when regarding matters of regional significance.

Resource consents are issued by regional and district councils to control the effects of activities. Land use does not require consent unless it contravenes a rule in a plan or proposed plan. Local authorities monitor the implementation of consents and consent holders may also be required to monitor the conditions imposed on their consent. The conditions of consents can usually only be reviewed if provided for in the consent. When required to meet conditions in a national environmental standard or newly operative regional plan, conditions of water, coastal or discharge permits may be reviewed by consent authorities; however, the duration of consents cannot be changed. Consents and their associated rights are not tradable in New Zealand; however, land use consents are attached to property, as opposed to a specific person.

4.1.2. Māori values and the RMA

Information and implementation on whole community ideals about environmental management, in particular water, is lacking. Industrial and farming needs are usually given priority while broad community expectations are overlooked. This is especially pertinent to Māori cultural values (Cullen et al., 2006). A requirement under the RMA under matters of national importance is to "recognise and provide for…the relationship of Maori and their culture and traditions with their ancestral lands, water, sites, waahi tapu, and other taonga" (New Zealand Government, 1991, Part 2, s6(e)). Despite this legislative requirement, Māori are seldom involved in decision-making beyond a token inclusion (Awatere, 2005; Taiepa et al., 1997).

How decisions are made and whose values are included in regards to resource management is important to Māori. Concepts such as mauri (life principle), kaitiaki (guardian), tapu (sacred) and rāhui (temporary prohibition or restriction on an area or resource) have importance for the relationship between Māori and the natural environment (James, 1993). Resource management agencies in New Zealand do not adequately incorporate Māori values into resource management decisions, resulting in low participation by iwi and hapū in processes and decisions (Awatere, 2005). Consequently, due to low participation and the degradation of the environment, Māori cannot effectively carry out their kaitiaki role. Understanding indigenous knowledge can result in resource management strategies that identify with the values of the community as a whole and can empower communities to become more active in local government decision-making (Awatere, 2005; Cullen et al., 2006).

Other strategies have been more successful than the RMA in implementing Māori into decision-making. One is the Waikato River co-management agreement between the crown, the four Waikato River iwi and Waipa River iwi, titled the Waikato-Tainui Raupatu Claims Settlement Act 2010. This agreement has more specific objectives to restore and protect the health and wellbeing of the Waikato River for future generations in an integrated, holistic and coordinated approach (NIWA, 2010b). People's relationship with the river is acknowledged and it ensures river iwi can exercise kaitiakitanga (guardianship) according to their tikanga (customs). The river restoration study relating to the agreement incorporates both indigenous environmental knowledge (mātauranga Māori) and western science, making it unusual on a global scale (NIWA, 2010b). It is the first co-management agreement of its scale in New Zealand and it is looked at by other indigenous cultures around the world as a means for empowerment in their own communities (NIWA, 2010b). The Vision and Strategy under the Settlement Act prevails over the National Policy Statement on Freshwater Management (Smith, 2011).

Another example of incorporating Māori into environmental management is the Strategy for the Lakes of the Rotorua district. Although this is not a statutory document, it aims to identify and address the problems of lake management using a co-ordinated approach between the Te Arawa Maori Trust Board, Environment Bay of Plenty and Rotorua District Council. The Strategy identifies costs involved in cleaning up the lakes and meeting water quality goals so the greatest benefit can be achieved for money spent. The vision for the Strategy is to ensure:

> *The lakes of the Rotorua district and their catchments are preserved and protected for the use and enjoyment of present and future generations, while recognising and providing for the traditional relationship of Te Arawa with their ancestral lakes.* (Rotorua Te Arawa Lakes programme, 2000).

4.1.3. Horizons One Plan

The problem of diffuse agricultural pollution[7] has been ignored by many Regional Councils, but Horizons Regional Council (HRC) has attempted to use regulatory measures to control diffuse pollution in the One Plan (proposed regional plan for HRC) with much objection, debate and controversy. Appeals to the One Plan were heard in 2010, with an ongoing appeal and decision process with the Environment and High Court going into 2013. The One Plan was first introduced to stakeholders in 2004 and to the public in 2005. Almost 10 years to reach a decision is an extremely long process for a 10 year management plan.

The One Plan addresses four key issues: declining water quality, increasing demand for water, unsustainable hill country land use and threatened native habitats. It is the first regional plan in New Zealand that aims to deal with nutrient management on a catchment basis, so the process may be closely watched by other councils. The One Plan will attempt to manage agricultural diffuse pollution by regulating land use activities that are high diffuse polluters (particularly intensive farming), to improve water quality in rivers and lakes in the region. Farmers will have to follow nutrient management plans to reduce nitrogen loads entering waterways. The Plan could cause farmers to reduce stocking numbers and could determine whether farms can be converted from sheep and beef to dairy (Horsley & Galloway, 2012). This strategy has been met with a high amount of criticism and objection from the dairy farming industry (Hoggard, 2012; Horsley & Galloway, 2012; Wolfe, 2012). Fonterra, Federated Farmers and primary production organisations argue that the One Plan is too stringent and will affect the profitability of their products (Olsen, 2012; Stowell, 2012; Wolfe, 2012). Conversely, farm modelling has shown that farms can decrease nitrogen leaching by up to 30% without adversely affecting overall farm profitability (Dewes, 2012).

4.1.4. RMA reforms

Presently there is uncertainty to the future regulatory management of New Zealand's environment under the RMA as the New Zealand Government is undertaking reforms of the RMA. The first phase of reforms was completed in 2009 with the Resource Management (Simplifying and Streaming) Amendment Act 2009. Driving the changes to the RMA is concern

[7] Pollution that is derived straight from the land as opposed to point source pollution that comes from a pipe or discharge point (discussed in Chapter 5).

that resource management processes are costly and time consuming and that "the system is uncertain, difficult to predict and highly litigious" (Ministry for the Environment, 2013c, p. 6).

Proposed changes to the RMA would fundamentally rewrite Part 2, which drives the decisions made under the Act. Part 2 contains the purpose of the act in section 5. Sections 6, 7 and 8 outline principles intended to give guidance to the way in which the purpose is to be achieved. Matters of national importance in section 6 must be recognised and provided for by decision makers to achieve the purpose of the Act and decision makers must have particular regard to other matters outlined in section 7. Proposed changes to the Part 2 of the RMA will fundamentally change it by (Ministry for the Environment, 2013c; Palmer, 2013a):

- Combining section 6 and 7 into one set of 'principles';
- Adding new development-focused principles into the RMA;
- Removing several existing environment-focussed matters from the RMA;
- Weakening the majority of the remaining environment-focused elements; and
- Inserting a 'methods' section for councils to follow to perform functions and exercise powers under the RMA.

Combining sections 6 and 7 are based on the notion that too much emphasis is placed on environmental matters, overshadowing economic and social activities (Ministry for the Environment, 2013c). In other words, it is believed environmental matters have halted development, particularly the development of major infrastructure and the provision of land for housing. However, in 2010/11, less than one per cent (0.56%, 203 applications) of resource consent applications were declined (Ministry for the Environment, 2011c).

In the changes, new principles have been added including providing for: "the benefits of the efficient use and development of natural and physical resources"; "the effective functioning of the built environment including the availability of land for urban expansion, use and development"; and "the efficient provision of infrastructure" (Ministry for the Environment, 2013c, section 3.1.1), thus giving more weight to development. Existing matters protected under the Act that will be removed are amenity values, the intrinsic value of ecosystems and the maintenance and enhancement of the quality of the environment. Environmental principles that remain will be weakened by removing directive words such as "enhancement", "maintenance" and "protection" and limiting protection to "specified" natural features and landscapes and areas of indigenous vegetation.

Other proposed changes to the RMA include amendment to Section 32 which requires a consideration of alternatives, benefits and costs when developing a plan, policy or regulation. Clause 69 proposes two economic effects that must be included in an assessment: economic growth opportunities that are anticipated to become unavailable; and employment opportunities that are anticipated to be provided or reduced (Wright, 2013). Specifying these two economic effects while failing to specify other effects will make an evaluation under section 32 unbalanced, resulting in economic effects been given more weight than environmental, social, and cultural effects (Wright, 2013).

Additionally, under Clause 69, benefits and costs in an evaluation under section 32 must be quantified where practicable. This will further accentuate the dominance of economic effects. This is because many aspects will not actually be valued as some of the environmental, social and cultural values are difficult, if not impossible, to quantity (as discussed in Chapter 3). If values can be quantified, they will be seldom expressed in dollars. In contrast, estimates of economic and employment effects are commonly, if not always, quantifiable in dollar terms. Monetary valuations are likely to be given more weight to decision-makers than more intangible benefits and costs (Wright, 2013).

Changes will also allow more resource consents to be processed without public notification. This will substantially limit the ability of submitters to put forward evidence regarding the environmental impacts of proposals (Palmer, 2013a). Already 96% of consents are processed without public notification (Ministry for the Environment, 2011c). Additionally, central Government will have the power to take individual consent decisions away from local councils as well as insert provisions in local council plans without any consultation (Oliver, 2013; Salmond, 2013).

Response to the changes

There have been differing views on the RMA changes. The dairy industry and the current Government support the changes while environmental groups and those working to protect or manage the environment generally oppose. Proposed amendments will affect the ability of the RMA to achieve its stated purpose. Sir Geoffrey Palmer, former Prime Minister and architect of the RMA, notes "the [unnecessary] proposed changes to Part 2 will significantly and seriously undermine environmental protection", and "will lead to greater uncertainty and cost in the application and interpretation of the RMA" (Palmer, 2013a, p. 2). Palmer argues that proper consultation and consideration is required and changes should be made only if they are needed after consultation decides that is so (Palmer, 2013a).

Fish and Game believe the proposed changes will "fundamentally change the context of the RMA" by destroying "its environmental protections" and pave "the way for further rampant exploitation of finite water resources" (Fish & Game, 2013b). Dame Anne Salmond has described the changes as "an attack on participatory democracy" (Oliver, 2013). Resource use with fewer protections will be promoted – "unsustainable and unchecked development at the expense of our environment" (Fish & Game, 2013b). Many groups support the move to shorten timeframes, improve planning processes and simplify planning documents to save time and money (Fish & Game, 2013b; Palmer, 2013a). However, it is argued that these processes can be rectified without changes to environmental safeguards (Palmer, 2013a). Parliamentary Commissioner for the Environment Dr Jan Wright says the RMA "should not become an economic development act" (Parliamentary Commissioner for the Environment, 2013).

According to Environment Minister Amy Adams (current in 2013/pre-election 2014), the RMA reforms will "deliver both economic and environmental benefits for future generations" (Ministry for the Environment, 2013c, p. 5). Instead, it is argued the changes are trying to get better access to natural resources, particularly water for industry (Oliver, 2013). In support of the RMA reforms are Fonterra, Federated Farmers, Irrigation NZ, Horticulture New Zealand and Business NZ. Federated Farmers believes that the RMA currently "leans too far towards protection" (Federated Farmers, 2013a) and is "unfair, unpredictable and overly bureaucratic" (Federated Farmers, 2008).

4.1.5. National Policy Statement for Freshwater Management

In 2011, two decades after the implementation of the RMA, the National Policy Statement for Freshwater Management (NPSFM) was implemented (New Zealand Government, 2011a). This was the first step to improve freshwater management at a national level, even though it was two decades late (i.e. two decades after the implementation of the RMA which requires NPS to function adequately). The NPSFM sets out eight objectives for targets on water quality, water quantity, integrated management and tangata whenua roles and interests. The NPSFM provides a framework and direction for councils on the objectives they must meet and types of policies to use to achieve these objectives (Sinner, 2011), but it is up to councils to set rules and regulations to ensure the objectives and policies are fulfilled in their region (Smith, 2011). A minimum standard for all water bodies was recommended by the Board of Inquiry for the NPSFM; however, it was not implemented. Instead, councils are in charge of determining the standards of each lake, river, aquifer and wetland within its region (Sinner, 2011). Therefore, changes in current policy will only be made if a council decides that the objectives or standards are not currently fulfilled (Sinner, 2011). The uptake of the NPSFM may take some time, as councils will need to set limits on water quality and quantity and change regional plans although parts of the NPSFM can be implemented into plans progressively. Plan changes typically take three to five years to develop, propose and finalise, and possibly more if there are multiple appeals (Sinner, 2011), such as the case with the Horizons One Plan.

The NPSFM aims to maintain or improve the "overall quality of freshwater within a region" (New Zealand Government, 2011a, objective A2), and therefore does not protect every freshwater body. This objective allows offsets within a region: as long as the overall water quality is maintained, some water bodies can be further degraded (Smith, 2011). Furthermore, the policy does not apply to consent applications made before the NPSFM took effect on 1 July, 2011. Regional management allows contamination to occur in some areas and does not halt serious degradation and contamination of certain waterbodies and catchments. As an alternative, catchment-based management may be more effective at improving water quality. Catchment-based management acknowledges land/water interactions, focusing management on a whole drainage level rather than individual properties, activities or streams/rivers.[8]

Recommendations were made for the NPSFM to ensure any new discharges or increases in land use intensity would require resource consent. However, this was not implemented in the final NPSFM and instead only activities that already require consent under the RMA are included (Sinner, 2011). Adverse effects are to be taken into account in regards to activities that do require consents, a policy already outlined in the RMA (Smith, 2011). The main source of the problem, diffuse pollution, is still not nationally controlled. Furthermore, objectives set out for tangata whenua roles and interests only cover what is already in the RMA (Smith, 2011).

Moreover, any improvements in water quality are not likely to be seen in the near future as councils have until 2030 to implement the NPSFM policies into regional plans. In the

[8] For more information on catchment-based research refer to Dodd, Wilcock & Parminter (2009) and Feeney, Allen, Lees and Drury (2010).

meantime, intensification is likely to continue. As the NPSFM does not enforce national standards or even controls on diffuse pollution, water quality is likely to worsen. It is up to Regional Councils to implement rigorous standards on pollution from intensive land use to maintain or improve the state of freshwater ecosystems, and therefore, Regional Councils determine the effectiveness of the NPSFM, something they have only previously done with financial assistance from Central Government (Sinner, 2011). Additionally, there is likely to be a wide variety of interpretations of the NPSFM from councils, and hence a mixed variety of results.

4.1.5.1. Land and Water Forum

The Land and Water Forum is a group of industry representatives, environmental and recreational NGOs, iwi, scientists, and other organisations with an interest in freshwater and land management. The Forum is a stakeholder-led collaborative process to develop a shared vision for management of water in New Zealand. This process led to the Fresh Start for Fresh Water initiative and provided recommendations for the National Policy Statement for Freshwater Management. The Forum has released three reports providing a national framework to set freshwater objectives and limits and how these should be managed. However, many of their recommendations have not been taken up by recent freshwater reforms.

Fresh Start for Fresh Water Clean-up Fund

The Clean-up Fund is part of a package of water reforms, including the NPSFM and the Irrigation Acceleration Fund, in response to the Land and Water Forum's recommendations. The Clean-Up fund provided $15 million over two years (2011/12 and 2012/13) to help projects restore waterways affected by historical water quality issues. Five projects were allocated funds (Ministry for the Environment, 2012b). Regardless, this initiative does not address the issue of land use intensification; it is pointless to clean up while still polluting.

Irrigation Acceleration Fund

The Irrigation Acceleration Fund (IAF) allocated $35 million over five years to support irrigation development (Ministry for Primary Industries, 2013f), with an additional $400 million expected to be added (Tarrant, 2011), despite evidence that irrigation intensification has environmental effects. A NZIER (New Zealand Institute of Economic Research) report estimates that the irrigation fund could support over 300,000 ha of irrigated land, predominantly in the

Canterbury and Hawkes Bay regions (Kaye-Blake et al., 2010). With this increased irrigation, exports are expected to increase by $4 billion by 2035 (Kaye-Blake et al., 2010). However, in this analysis the environmental, social or regional impacts of the irrigation schemes were not quantified.

4.1.6. Management of agricultural GHG emissions

Methods used to manage greenhouse gas (GHG) emissions in New Zealand do little to reduce agricultural emissions. New Zealand is committed to reducing emissions through international agreements and has developed an emissions trading scheme in an attempt to reduce emissions. Despite these efforts, GHG emissions in New Zealand are increasing.

International emission commitments

In 1992, New Zealand signed and ratified the Framework Convention on Climate Change (FCCC) that emerged from the United Nations Earth Summit. The FCCC placed legal obligations on parties to set up the legal structure for international cooperation on reducing global GHG emissions. Following from this was the Kyoto Protocol, which aimed to reduce emissions below 1990 levels. Countries who ratified this are legally bound to emissions commitments, either by reducing their emissions to targets or purchasing credits from others who have achieved reductions. New Zealand ratified the Kyoto Protocol in 2002. Commitments under the Kyoto Protocol mean that New Zealand needs to reduce its emissions to 1990 levels in the first commitment period (CP1), 2008-2012, or pay for the excess emissions by obtaining carbon credits.

New Zealand did not commit to a second period of the Kyoto Protocol, being one of the few countries amongst the 200 signatories to vote against extending it. This decision will lock New Zealand out of the carbon trading market from 2015, effectively increasing the price of carbon in New Zealand as buyers will not be able to access cheap credits from overseas. This decision is disappointing, particularly for New Zealand's 'clean green' image.

Emissions Trading Scheme

The Emissions Trading Scheme (ETS) is part of New Zealand's climate change policy to meet international obligations. A price is put on GHG emissions to provide an incentive for polluters to reduce their emissions. However, the agriculture sector is now exempt from paying for their emissions under the ETS until technologies allow GHG emissions to be reduced (Ministry for Primary Industries, 2013c), and until international competitors are taking sufficient actions to reduce their emissions (Ministry for the Environment, 2012d). This provides no incentive for farmers to reduce their emissions.

Terry (2012a) argues that the ETS will reduce gross emissions by less than 1% during its first five years. Instead of requiring agriculture to reduce emissions, New Zealand is seeking to cut emissions by using forestry credits. However, Terry (2007) suggests agriculture has the greatest potential to substantially reduce New Zealand's GHG emissions. Over 40% of the growth in emissions above 1990 levels is expected to be from livestock emissions from the rise in the dairy industry (Terry, 2007). Nitrous oxide emissions account for a third of livestock emissions and can be substantially reduced very cost effectively (for more information on this see Terry, 2007). The other two thirds of emissions are from methane and the costs of abatement are less well understood.

4.2. Voluntary Initiatives

Voluntary management usually involves individuals, firms, or landholders taking action to improve the state of a resource. Agreements are sometimes made between Government and individual businesses or industries. In New Zealand, voluntary initiatives have largely been developed by industry, such as the Dairying and Clean Streams Accord, but have failed to lesson or mitigate environmental impacts and instead may act as an excuse for the Government to not enforce more regulatory or economic incentives to reduce pollution.

4.2.1. Dairying and Clean Streams Accord

The Dairying and Clean Streams Accord (the Accord) was signed in 2003 between Fonterra, Ministers for the Environment and Agriculture and Forestry and local Government New Zealand as a response to the 'dirty dairying campaign'. Dirty dairying was a slogan instigated by Fish and Game to address the environmental effects of dairy farming. However, in the Accord's development stages there was no input from organisations such as Fish and Game, Forest and Bird, or other environmental interests. The Accord attempted to bring voluntary initiatives to address the decline in the quality of lowland waterways. It addresses dairy effluent and nutrients directly entering rivers and streams[9].

Despite its efforts, the Accord failed to achieve its major goal of reducing “the impacts of dairying on the quality of New Zealand’s streams, rivers, lakes, groundwater and wetlands” (Deans & Hackwell, 2008, p. 4; Fonterra et al., 2003, p. 1). Water quality has continued to fall in monitored dairying areas. For example, in five closely monitored ‘best practice’ catchments that have been managed above Accord standards, water quality has not improved or has declined during the period of the Accord (Deans & Hackwell, 2008). A number of the principal targets in the Accord have not been met. For example, a target of the Accord is that dairy farm effluent discharges comply with their resource consents and regional plans. However, 18-24% of all dairy farms have been found to be in ‘serious non-compliance’ with their consent obligations in some major dairying regions (Deans & Hackwell, 2008). Furthermore, prosecution of dairy farmers who regularly do not comply with their effluent discharge consent conditions has been rare. The Accord should not be a substitute for farmer compliance, and enforcement and monitoring by Regional Councils (Deans & Hackwell, 2008).

Another shortfall of the Accord has been reporting: in many cases this has differed from targets in the Accord. The Accord sets a target of all dairy farms to have systems in place to manage nutrient inputs and outputs. However, the Accord only reports on how many farms have a nutrient budget, not an operational nutrient management system (Deans & Hackwell, 2008).

In 2013, the Clean Streams Accord was changed to the Sustainable Dairying Water Accord and with it has come other changes. In the Clean Streams Accord dairy cattle were to be excluded from 90% of streams, rivers and lakes by 2012; however, the new Accord has pushed this target back to 2014 (DairyNZ, 2013b). Only 50% of dairy farms with waterways are required to have a riparian management plan by 31 May 2016 and of these only half need to complete half of their commitments by 2020 (i.e. a quarter of dairy farms with waterways have to complete half of their commitments), and all by 2030. Guidelines for riparian management have already been well documented and researched, so this timeframe is unnecessary (Fish & Game, 2013a). Furthermore, nitrogen management still only involves modelling. Targets to reduce nitrogen loss (as appropriate) only apply to catchments that have nutrient terms or caps as recorded in a regional plan. The new Accord still fails to address the main impacts on water

[9] The Dairying and Clean Streams Accord (Fonterra et al., 2003) and Snapshots of progress for the Accord can be found at: http://www.mfe.govt.nz/issues/land/rural/dairying.html

quality on small streams (less than a metre wide and 30 cm deep). Small streams provide important breeding and habitat areas for native and valued introduced freshwater fish species (Fish & Game, 2013a). Additionally, pollution entering these streams ends up in larger river systems (Fish & Game, 2013a).

Some good points of the Accord are that new dairy conversions are required to comply with Accord targets from the beginning and offsite grazing areas have also been brought into the Accord. However, the most significant environmental issue from dairy farms, nutrient loss, is still not adequately managed. Additionally, there are no requirements in the Accord to actually measure changes in water quality and there does not appear to be compliance monitoring of accord targets. While some provisions in the Accord could help to reduce dairying's impact on water quality, they need to be implemented much sooner.

As the Accord has failed to meet its goal after nearly a decade, it may be time to move to more stronger and enforceable conditions. Fonterra has an annual production growth goal of 4% (Deans & Hackwell, 2008), and it is likely that greater intensification is needed to be able to meet this growth. The effects of continued intensification are cumulative, even when combined with farm technologies to reduce pollution (Deans & Hackwell, 2008).

Voluntary initiatives aiming to increase water quality are unlikely to be successful without effective monitoring of indicators. Measurement on whether these initiatives are actually improving water quality should be undertaken (Cullen et al., 2006). Having targets in place will not mean anything if water quality is not improving.

4.3. Economic Initiatives

Economic measures involve persuading polluters and landowners to change their behaviour to avoid or lessen pollution by monetary incentives, such as taxes and charges, as well as subsidies to encourage more sustainable practices. There has been little implementation of economic initiatives in environmental policy in New Zealand. Examples are generally limited to nutrient cap and trade systems in a few lake catchments and the ETS (Emissions Trading Scheme).

Cap and trade

A nutrient trading scheme encourages those who can most cheaply mitigate nutrients to do so, and provides flexibility – participants can operate as they see fit as long as they hold enough allowances to cover their nutrient leaching (Kerr et al., 2012). Nutrient trading could reduce mitigation costs by allowing mitigation to occur in places and at times with the least cost. For instance, support could be given to those who have significant potential to reduce nutrient loss at relatively low cost. This could reduce the burden on those who may find it more difficult and reduces the demand for allowances, thus, lowering the cost for all (Kerr & Lock, 2009). The total amount of leaching allowances is equal to an annual cap on leaching that will achieve the desired water quality (Kerr et al., 2012). Landowners may also be encouraged to seek more profitable and less damaging land uses (Kerr & Lock, 2009).

Costs to reduce nitrogen leaching were estimated to be higher with a uniform cap on all farms with no trading allowed. For example, reducing average leaching from the Karapiro catchment from 31 to 30 kg/ha would cost around $23/ha under a uniform cap but less than $1 with trading (Table 4.1) (Marsh, 2012a). This is because with trading, farms that can reduce leaching with the lowest cost will do so. Estimated costs associated with nutrient reduction scenarios are presented in Table 4.1.

In the Horizons Region, it was estimated that a 20% reduction in N leaching would cost $25 per hectare, while a reduction of 30% leaching would cost $62 per hectare (Marsh, 2012a). Based on these estimates, the total cost of a nutrient cap to the Horizons region for the One Plan would be in the range $1.8-$4.4 million per year (Marsh, 2012a). This cost could, however, be reduced by using trading.

Table 4.1: Abatement quantity and cost for simulated polices to reduce nitrogen leaching from 31kg N/ha to a specified amount.

Measure	**Leaching target (kg/ha)**		
	30	**26**	**22**
Reduction in N leaching (kg/ha)	1	5	9
Cap emissions – no trade (cost $/ha)	22.9	49.47	96.6
Cap emissions – trade (cost $/ha)	0.69	14.79	54.39

Data source: Marsh (2012a). **Notes**: Farmers are assumed to make use of Currently Recommended Mitigation Practices (CRMPs) (Marsh, 2012a).

In 2011, a nutrient cap and trade scheme was implemented in the Lake Taupo catchment, the first instance attempting to control non-point source nutrient pollution in New Zealand using market policies (Barns & Young, 2012). The policy aimed to maintain water quality in Lake Taupo (Young & Kaine, 2009). The scheme limits the amount of nitrogen from agricultural land that reaches Lake Taupo to achieve the desired water quality. Farmers are able to make their own decisions in how to reduce nitrogen leaching, thus allowing flexibility (Barns & Young, 2012). Desired water quality is predicted to be achieved at least cost with this scheme (Barns & Young, 2012). Being the first of its kind in New Zealand, this policy approach set a framework for further diffuse source management schemes.

4.4. Conclusion

Regulatory management in New Zealand has failed to mitigate the most significant freshwater contaminant source – diffuse pollution. Until this is adequately managed, freshwaters in New Zealand will continue to deteriorate. Economic initiatives could step in and alleviate this downfall, but have been largely absent from policy approaches in New Zealand, with the exception of a few catchments. Voluntary programmes have appeared to do nothing more than raising awareness of the problems. This is beneficial but must be accompanied with actions to reduce impacts. Additionally, unfair distribution of funds for projects occurs, providing more money to use environmental resources than to clean up pollution. For example, only $15 million was awarded for clean-up projects but $400 million is proposed for irrigation development. This incentivises intensification over environmental protection.

Co-management agreements with iwi may provide holistic examples of how environmental resources can be managed. They usually have high expectations and visions for what restoration should include. The Waikato co-management agreement has been one of the few cases to incorporate costs to restore ecosystems in management frameworks, along with the Rotorua Lakes Protection and Restoration Action Programme. The Waikato co-management agreement not only collaborates with iwi but involves the integration of iwi into the management of natural resources by acknowledging and incorporating tikanga into agreements.

Chapter 5. WATER

Tuatahi ko te wai, Tuarua whanau mai te tamaiti, ka puta ko te whenua.
At birth water comes first, then the child, followed by the placenta known as whenua (land).
(Māori proverb)

New Zealand has an abundance of freshwater; rivers, lakes, snow and ice cover 7.8% of New Zealand's land area. There are about 425,000 kilometres of rivers and streams, almost 4,000 lakes larger than 1 hectare, and about 200 aquifers (Ministry for the Environment, 2007a). New Zealand's aquatic plant and animal communities need clean water and humans require clean water for domestic, industrial, agricultural and recreational purposes. Despite the wealth of freshwater, water quality in New Zealand is on a declining trend, driven by agricultural intensification, particularly dairy farming (Ballantine & Davies-Colley, 2009; Cullen et al., 2006; Larned et al., 2004; Ministry for the Environment, 2009b; Parliamentary Commissioner for the Environment, 2004).

Approximately 40% of both streams and lakes are in catchments that have been modified by agriculture or where pasture is the predominant land cover (Ministry for the Environment, 2007a). Rivers and streams in agricultural or urban catchments generally have poorer water quality than those with little or no farming and urban development (Larned et al., 2005; Ledgard et al., 1996; Ministry for the Environment, 2007a; Rodda et al., 1999). Particularly, dairying has been responsible for some of the poorest water quality in rural areas (Davies-Colley & Nagels, 2002; Ministry for the Environment, 2009b; Perrie et al., 2012), mainly in Waikato, and more recently in Southland and Canterbury where intensification and conversion from dry stock farming to dairy farming has contributed to declining water quality (Proffitt, 2010).

Inputs pumped into farm systems are not all used in milk production; some flow to lakes, rivers, streams and groundwater. Intensive dairying practices that apply pressure on the land and eventually impact freshwater include: freshwater demand, increased stocking rates, fertiliser application, the removal and destruction of vegetation (particularly riparian), and the drainage of wetlands. These processes have detrimental and widespread consequences to freshwater that arise from: faecal contamination of surface waters; nutrient pollution of freshwater; and an increase of sediment entering water (Figure 5.1) (Ledgard et al., 1996; Willis, 2001). These impacts have led to a loss of native biodiversity, through habitat destruction, nutrient overload and flow reduction; loss of recreational areas; and a reduction in the buffering capacity of floods from the removal of vegetation and wetlands and increase of sediment; among many other impacts (Figure 5.1) (Clark et al., 2007; Davies-Colley et al., 2003; Ministry for the Environment, 2007a; Moller et al., 2008; OECD, 2007; Parliamentary Commissioner for the Environment, 2004; Wilcock et al., 1999). Furthermore, many of these impacts lead to harm to New Zealand's 'clean green' image (discussed in Chapter 9).

Concerns associated with dairy farming covered in this chapter include water quality problems associated with faecal matter, nutrients (nitrogen and phosphorus) and sediment; and issues with water abstraction (Figure 5.1). A broad overview in relation to these concerns is provided using case studies and examples, followed by a description of the consequences on freshwater. Examples of costs associated with these problems are provided in Chapter 9.

5.1. Water Quality

Pollutants reach water bodies in main two ways: out of pipes (point sources) and through and over the land (non-point source (NPS) or diffuse sources) (Figure 5.2). Usually resource consents are required to discharge point sources to water and for the past two decades stricter controls have been enacted for point sources with the implementation of the Resource Management Act 1991 (discussed in Chapter 4). Point sources can generally be measured and accounted for. Conversely, non-point source pollution is relatively uncontrolled and is now the main cause of freshwater pollution (Ballantine & Davies-Colley, 2009; Blackwell et al., 2006; Davies-Colley et al., 2003; Hill Young Copper, 2006; Monaghan et al., 2007), particularly because of the rapid expansion of dairy farming (OECD, 2007). Although controlling point sources is important for to improve water quality, the focus on point source discharge limits has proved inadequate to protect water quality as it totally ignores the importance of diffuse inputs. This is despite reports as far back as the 1990s by the Ministry of the Environment (1997) highlighting the importance of managing and reducing the impacts of diffuse pollution.

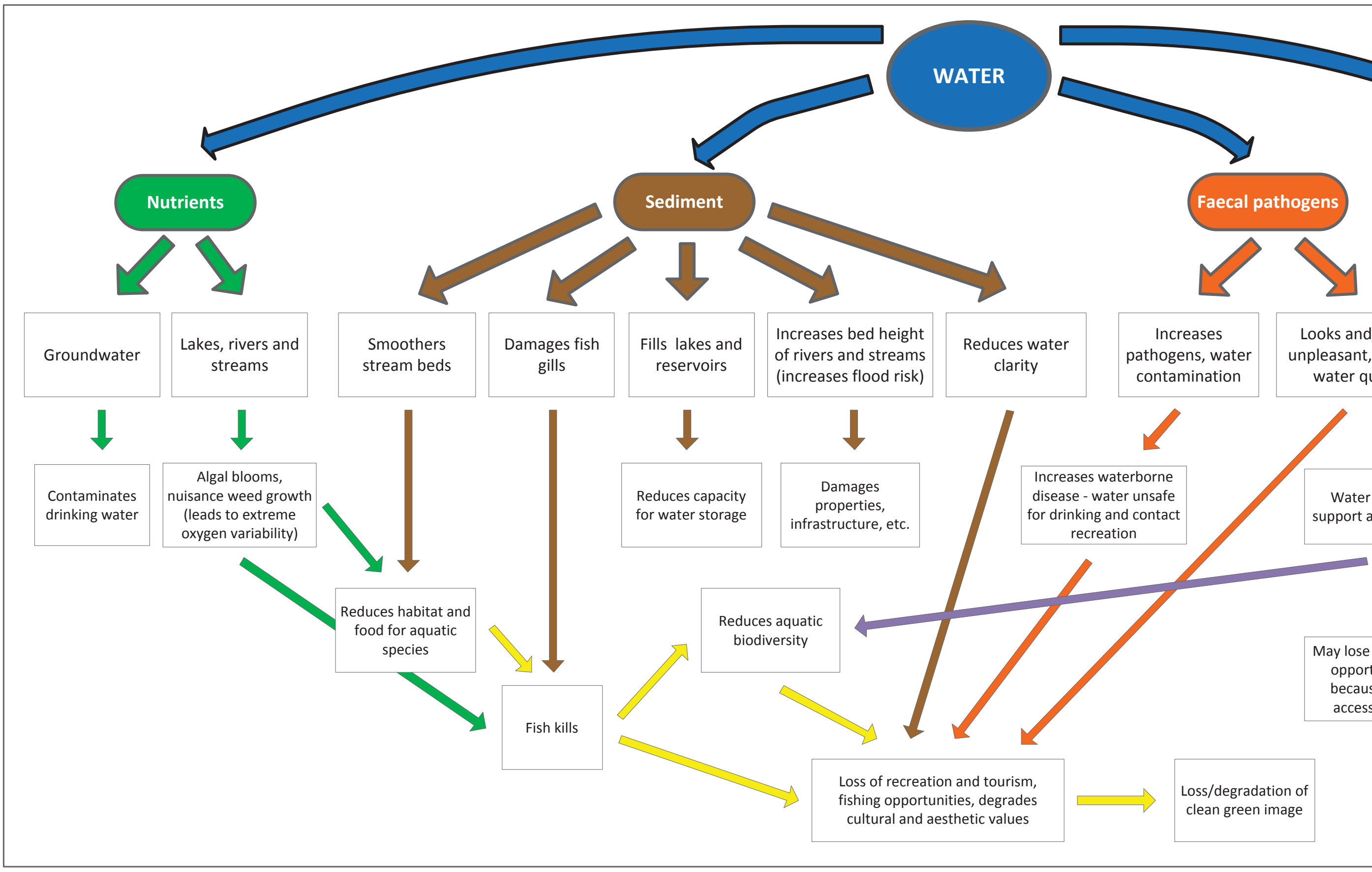


Figure 5.1: Major impacts on water from externalities of dairy farming in New Zealand.
Notes: Irrigation impacts link back to impacts of nutrients, sediment and faecal pathogens. Yellow arrows depict the flow from co

Water

Figure 5.2: Common sources of freshwater water pollution: livestock pollution; crops; industry; cities, towns and roads; septic tanks; and human effluent.
Source: Ministry for the Environment (2007a)

The only major point source discharge associated with dairy farming is effluent from the milking shed, although this has now largely been converted to a diffuse source due to the movement to land application (Cameron & Trenouth, 1999; Hawke & Summers, 2003). Historically, dairy shed effluent was mainly discharged to water, leading to high influxes of faecal contamination and nutrients, as well as the unpleasant appearance of water, at times of discharge (Cameron & Trenouth, 1999). The movement to land application of dairy shed effluent has reduced the input of nutrients and pathogens to surface water from point discharges (Roach et al., 2001); on the other hand, it has moved some of the problem elsewhere (Cameron & Trenouth, 1999; Hawke & Summers, 2006; Wang et al., 2004). Moreover, an increase in stocking rates has counterbalanced any effect of reducing nutrients to waterways via point sources. The main diffuse pollutants from dairy farming are: sediment, nitrogen and phosphorus, and microbiological contaminants (faecal pathogens) (Blackwell et al., 2006; Davies-Colley et al., 2003; Ministry for the Environment, 2007a). Diffuse runoff and leaching occurs particularly when land is over-grazed or when soils are saturated as the infiltration capacity of the soil is compromised. Overgrazing can lead to soil compaction (further increasing runoff), and saturated soils are unable to soak up excess nutrient particles (explained in Chapter 6) (Davies-Colley et al., 2003).

5.1.1. Faecal pathogens

Faecal matter from human sewage or animal waste contains pathogenic micro-organisms. The presence of faecal matter as pathogenic bacteria, viruses and protozoa in freshwater can be revealed by measuring faecal indicators, such as faecal coliforms, *Escherichia coli* (*E.coli*), faecal streptococci and enterococci (Davies-Colley et al., 2003). *E.coli* concentrations are used to measure drinking and recreational water standards. Adequate drinking water should have next to no *E.coli* present, but acceptable levels are higher for contact recreation (see Appendix C for guidelines). Faecal material can enter waterways through point source discharges from storm-water or sewage treatment (Ministry for the Environment, 2011b), animal processing plants (Donnison & Ross, 1999; Ferguson et al., 2003), and effluent pond discharges (Davies-Colley et al., 2003; Smith et al., 1993; Wilcock et al., 1999); as well as surface runoff from the land (Collins et al., 2007; Davies-Colley et al., 2003; Donnison & Ross, 1999; Environment Waikato, 2008) or subsurface flow (Ritchie & Donnison, 2010).

Pathogen levels in freshwater

Previous national studies have shown that faecal bacterial levels have been high throughout the country. In the period 2003-2007, *E.coli* concentrations exceeded the MfE/MoH contact recreation guidelines (550 *E.coli*/100 ml – see Appendix C) at about 75% of sites (Ballantine et al., 2010). The latest Ministry for the Environment indictor data (Ministry for the Environment, 2013f) for the suitability of swimming at 221 coastal and 204 freshwater sites nationwide showed that almost 70% of freshwater sites were unsafe for contact recreation at some point (Figure 5.3). Fifty per cent of sites are classed as poor or very poor and therefore should not be used for contact recreation, while 20% are classed as fair – described as sites that have potential sources of faecal material and water may be unsuitable for contact recreation after rainfall. These monitored sites are dedicated swimming spots (Ministry for the Environment, 2013f), situated in areas that are likely to have better water quality, such as forested streams. The poor performance of these areas does not give a promising outlook for streams in the rest of New Zealand.

Not surprisingly, when considering pastoral catchments *E.coli* concentrations are even higher. *E.coli* concentrations frequently exceed guidelines for contact recreation in pastoral catchments and are typically between two and 20 times higher than those in forested catchments (Davies-Colley et al., 2004; Larned et al., 2004). However, mixed results have been reported. Larned et al. (2004) found that *E.coli* guidelines for contact recreation were exceeded at 96% of 259 pastoral sites from 1998-2000, while Donnison et al. (2004) reported only 28% of pastoral sites exceeded guideline levels.

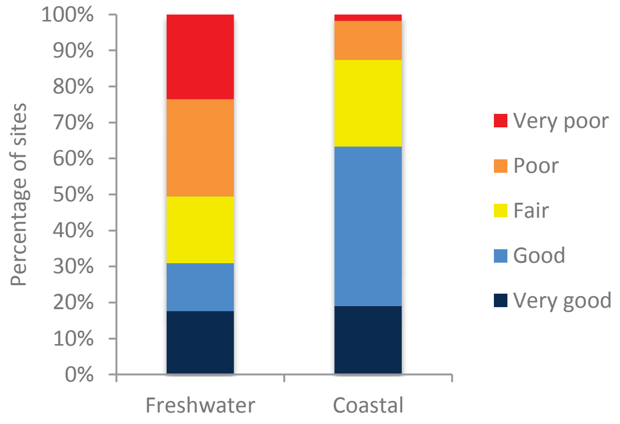


Figure 5.3: Suitability for recreation grades at freshwater and coastal sites used for recreation around New Zealand assessed in 2013.
Data source: Ministry for the Environment (2013f). **Note:** No data was provided for sites in Auckland, Northland, Waikato or the West Coast regions.

Faecal matter from dairy farms

Dairy farms are likely to contribute a higher proportion of faecal contamination to waterways compared to other livestock types. Sheep faeces do have high concentrations of microbes but sheep do not enter water like cattle do (Ritchie & Donnison, 2010). Studies show dairy catchments have higher *E.coli* concentrations than average for lowland pastoral farming catchments (Collins, 2002; Davies-Colley & Nagels, 2002; Ministry for the Environment, 2009b). In Waikato intensive dairying areas, *E.coli* concentrations were exceeded at over 70% of sites tested for contact recreation (Collins, 2002). Likewise, median *E.coli* concentrations in five dairying catchment streams around New Zealand ranged from 290-1250 most probable number (MPN)/100 mL (Wilcock et al., 2007), 2-10 times the ANZECC guideline for contact recreation (median <126 E.coli/100 mL) (ANZECC, 2000). The five streams also had extreme (95 percentile) concentrations of *E.coli* that were 5-17 times the ANZECC 'Red Alert' value (>550/100 mL) (Wilcock et al., 2007).

Dairy farms in particular produce a large amount of faecal material. Cattle excrete 11-16 times a day and produce an average of 28-30 kg of faeces a day (Wilcock, 2006) One dairy cow is estimated to excrete faecal bacteria equivalent to between 14 (Environment Waikato, 2008) and 33 people (Figure 5.4) (Fleming & Ford, 2001). Given New Zealand's dairy herd population of 6.5 million, this represents faecal bacteria concentrations equivalent to over 90 million and up to 215 million people. Solid waste from one dairy cow contains the equivalent amount of nitrogen, phosphorus and total solids as 29, 30 and 44 humans respectively, while beef cow ratios are much smaller (Figure 5.4) (Fleming & Ford, 2001). Extending the waste measures to New Zealand's entire dairy (6.5 million) and beef cattle (3.7 million) populations gives very high human population equivalents (Figure 5.5), compared to the relatively small human population of New Zealand.

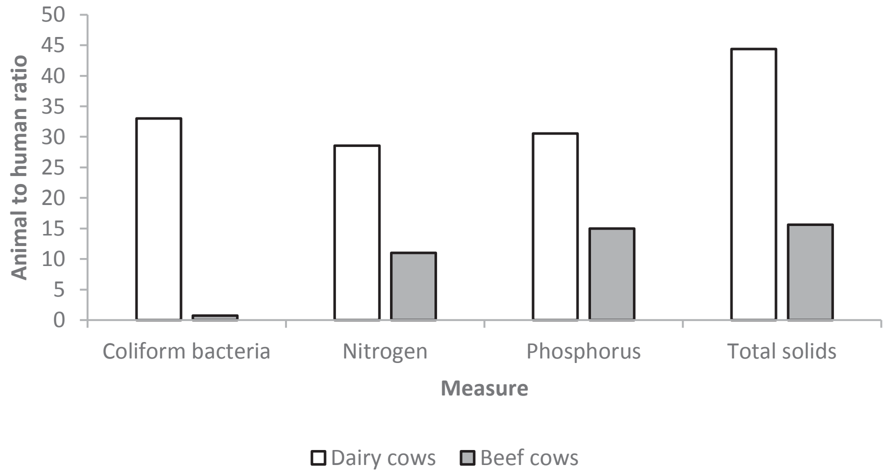


Figure 5.4: Ratios of coliform bacteria, nitrogen, phosphorus and total solids in waste from dairy and beef cows to human equivalents.
Data source: Fleming and Ford (2001). **Notes:** Bars represent waste from one animal. Comparative data from sheep were not available from this data source.

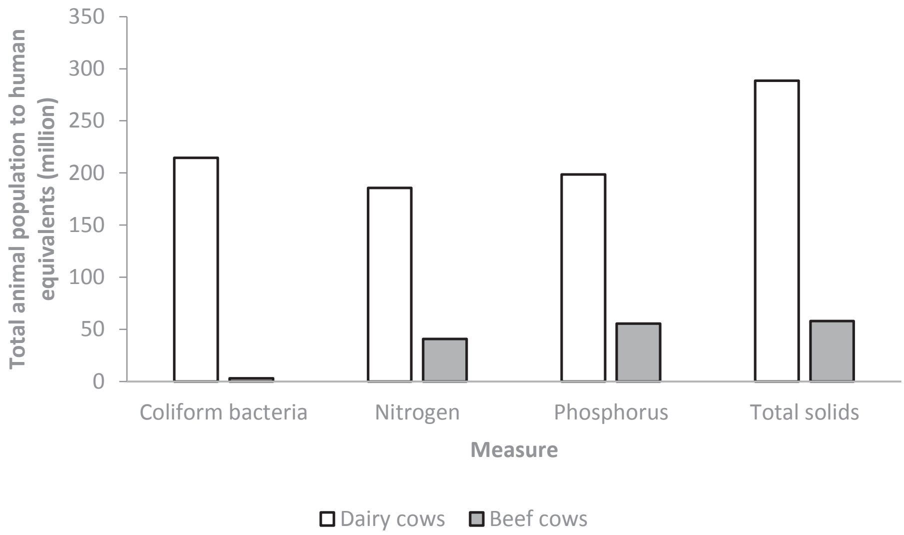


Figure 5.5: Ratios of coliform bacteria, nitrogen, phosphorus and total solids in waste from the entire New Zealand dairy and beef cattle populations to human equivalents.

Obviously due to the large amount of dairy excrement, large quantities of dairy shed effluent water are produced from dairy farms. Dairy effluent contains mainly faeces, urine and wash-down water, but also comprises storm-water, spilled milk, soil and feed residue, detergents and other chemicals (Cameron & Trenouth, 1999). Estimations made over a decade ago (1997-2000) of effluent water produced from New Zealand dairy farms were around 950 million cubic metres annually, of which about 59% was predicted to go to surface water (Flemmer & Flemmer, 2008). This amount would have undoubtedly increased since then and at the time was even deemed by the authors to be very imprecise. Furthermore, it also only accounts for the effluent from dairy shed operations, which is only a small proportion (5-15%) of the total waste produced by dairy cattle (Cameron & Trenouth, 1999). However, the total volume of dairy shed waste is likely to be larger than waste deposited on paddocks because of the large volume of water used to wash down dairy sheds.

Prior to the 1970s, dairy shed effluent was mainly discharged untreated into waterways. The wide spread adoption of on-farm oxidation treatment ponds saw effluent treated before discharge (Longhurst et al., 2000; Parliamentary Commissioner for the Environment, 2004). Regardless, pathogens often survived this process (Parliamentary Commissioner for the Environment, 2012). For this reason, land application of effluent became encouraged by regional councils (Cameron & Trenouth, 1999). However, some effluent is sprayed directly onto pastures without treatment (Cameron & Trenouth, 1999; Parliamentary Commissioner for the Environment, 2004). Even if two-pond treatment systems are used prior to discharge, pathogens can still be washed to water if the storage pond overflows, the effluent irrigator breaks down, or by irrigating on poorly drained and saturated soils (Parliamentary Commissioner for the Environment, 2012).

Regardless, the bulk of manure produced in New Zealand dairy systems is deposited directly to pasture from grazing cattle, of which is a significant source of faecal contaminants to freshwater (Davies-Colley et al., 2003; Davies-Colley et al., 2004; Donnison et al., 2004; Parliamentary Commissioner for the Environment, 2012; Wilcock, 2006; Wilcock et al., 2006; Wilcock et al., 1999). Overland flow and stock crossings have been found to generate the highest *E.coli* loadings to waterways in some instances (Wilcock, 2006). Concentrations are also high after rainfall as overland flow increases (Ritchie & Donnison, 2010). Direct deposition of faecal material to freshwater is common from stock grazing in or near water bodies or passing through streams and drains to and from the milking shed. Davies-Colley et al. (2004) found that dairy cows were 50 times more likely to defecate during stream crossings.

Similarly, high *E.coli* counts were associated with dairy cow crossings in a Wellington region stream (Perrie et al., 2012). Once faecal pathogens are excreted, they begin to die; therefore, concentrations are highest in fresh dung, and dark, moist, cool conditions are best for pathogen survival (Davies-Colley et al., 2003). In this regard, dung excreted directly in or near water-bodies poses the greatest threat of pathogen contamination.

Overland flow may be responsible for the largest proportion of annual catchment yields of faecal contamination. For example, modelling in the Toenepi catchment estimated that 80% of the faecal contamination occurred by overland flow (Ritchie & Donnison, 2010). Although, this modelling assumed that 90% of streams were fully fenced and effluent was treated and discharged from a two-pond system, so direct deposition into streams and contamination from dairy shed effluent were less likely to occur. Conversely, in the Bog Burn catchment (Southland), 78% of the total stream *E.coli* load from dairy farms was from direct drainage of irrigated dairy shed effluent onto mole-pipe drained land[10], with overland flow, subsurface drainage and direct deposition of dung contributing 16%, 6% and 0.1% respectively (Monaghan et al., 2007). Within the whole catchment considering other land uses, irrigation of dairy shed effluent contributed approximately 65% of the *E.coli* generated and discharged to the Bog Burn Stream (Monaghan et al., 2007). These studies emphasise the importance of the management system in place in contributing to overall loads.

Stock exclusion and effluent management changes in some dairying catchments have not achieved contact recreation standards in waterways (Ritchie & Donnison, 2010). Riparian vegetation strips can prevent access to waterways as well as trap microbes being washed down-slope into streams (Collins et al., 2007; Monaghan et al., 2007). However, indicator bacteria can remain in farm run-off trapped in buffer strips and can be released to surface water in high flow events (Ritchie & Donnison, 2010). In addition, once in streams, faecal indicator bacteria can survive in bed sediments, and re-suspend in the water during flood events (Ritchie & Donnison, 2010).

5.1.2. Nutrients

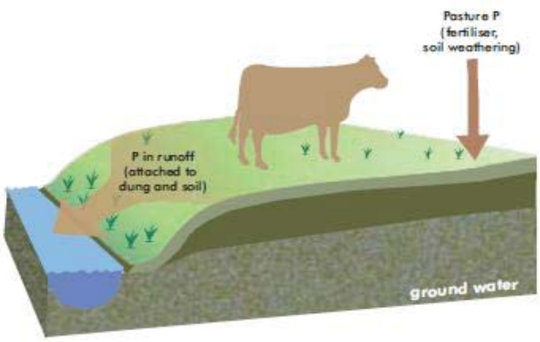
Nutrients circulate within and between ecosystems. Plants take up nutrients such as nitrogen and phosphorus which are cycled back into the soil when plants die. Nutrients are also added

[10] The mole-pipe drainage system provides rapid drainage of surplus rainfall and discharges it into a central pipe drain that carries drainage to surface water streams and ditches.

into dairy systems by fertilisers, brought-in feed, and atmospheric nitrogen fixing by clover root nodules, and are returned to the soil in the form of urine and dung. In dairy systems, some nutrients leave the farm in milk. Between 75% and 90% of the nitrogen (N) ingested by dairy cows is excreted due to the high N inputs in dairy systems; thus the N intake by grazing animals far exceeds the N off-take in products (de Klein et al., 2010). Nutrient surpluses occur when crops are unable to take up all the available nutrients. Nutrients not taken up are lost to the environment through leaching and runoff into ground and surface waters, and nitrification into the atmosphere.

Nutrient pathways to freshwater

Nitrogen and phosphorus enter water in different ways (Figure 5.6). Phosphorus is mobilised by soil or dung particles that wash off the land into streams or by direct input of fertiliser, effluent, sewage or industrial discharges into water (Davies-Colley et al., 2003; Environment Waikato, 2008). Although soil loss typically occurs on steep slopes on sheep and beef farms, it is also an issue in more intensive farming sectors (Parliamentary Commissioner for the Environment, 2004). Rainfall or intense surface runoff can exacerbate soil erosion. Phosphorus also enters waterways through farm runoff if the rainfall or irrigation intensity exceeds the soil infiltration capacity. Infiltration is influenced by soil physical properties such as texture and structure (Davies-Colley et al., 2003), and can be affected by compaction (see Chapter 6). On the other hand, nitrogen leaches through the soil to groundwater as nitrate when concentrations exceed pasture requirements. Once in groundwater, nitrate flows into streams, rivers and lakes. Nitrogen also enters surface water as surface runoff in saturated soils and can be released into the atmosphere as nitrous oxide (N_2O) in anaerobic soil conditions (Beukes et al., 2012; Environment Waikato, 2008). Both nitrogen and phosphorus can enter waterways through direct application, for instance by air application of fertilisers.



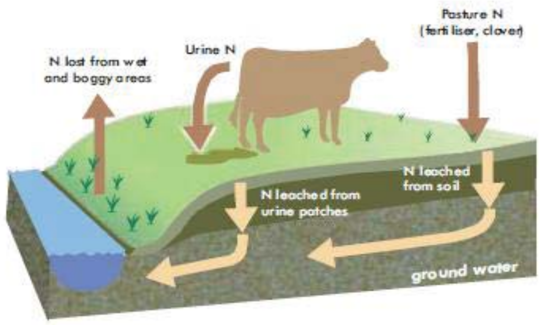


Figure 5.6: Pathways of phosphorus (left) and nitrogen (right) movement to surface and ground water.
Source: Environment Waikato (2008).

Nutrients and land use

Evidence suggests that nutrient levels in rivers increase in proportion to the levels of agricultural activity in river catchments (Ballantine et al., 2010; Ballantine & Davies-Colley, 2009; Environment Waikato, 2008; Hamill & McBride, 2003; Larned et al., 2004). Nitrogen and phosphorus concentrations are generally 2-7 times higher in pastoral catchments than forested catchments (Larned et al., 2004). Some of the most nutrient enriched rivers are located in lowland areas surrounded predominantly by pastoral farmland (Environment Waikato, 2008). This trend is clearly seen in Figure 5.7: as the proportion of catchments in pasture increases so does the median total N found in Waikato rivers (Environment Waikato, 2008). Over 90% of intensively farmed Waikato catchments had moderate to high levels of nitrogen concentrations (Environment Waikato, 2008).

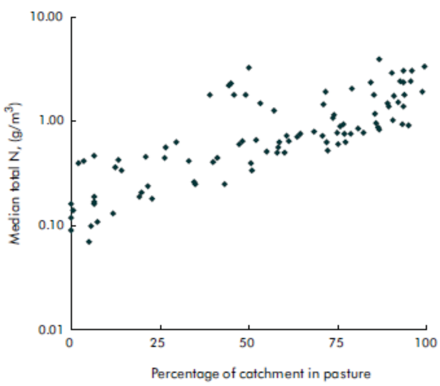


Figure 5.7: Median total nitrogen measured at 100 Waikato river and stream sites during 2000-2004 versus proportion of each catchment in pasture.
Source: Environment Waikato (2008).

Nutrient losses

Nationwide, agriculture contributes 70% of total nitrogen to the coast, of which half is from dairying, despite dairy farms only occupying around 7% of the country's land area (Environment Waikato, 2008). In the Upper Manawatu Catchment, dairy farms contribute around half of the nutrient load in the catchment but only occupy approximately 17% of the catchment, while sheep and beef farms contribute the other 50% of the nutrient load and occupy around 77% of the catchment (Dewes, 2012). In the Waikato region, dairying is responsible for 68% of the nitrogen and 42% of phosphorus entering waterways (Figure 5.8) from only 22% of the land area (Environment Waikato, 2008). Between 1992 and 2002 dairy cow numbers increased by 37% in the Waikato region while nitrogen and phosphorus levels in streams increased by 40% and 25% respectively, highlighting correlations between dairy farming and stream nutrient levels (Proffitt, 2010). Additionally, between 1990 to 2007 nitrogen fertiliser use on Waikato dairy farms increased 7 fold (Environment Waikato, 2008). These increases in intensification have clearly led to increased nutrients entering freshwater. Furthermore, in spite of management largely focused on point sources, only 3% of nitrogen and 7% of phosphorus is derived from point sources in the Waikato region (Figure 5.8), likely to be representative of trends in other predominant dairying areas.

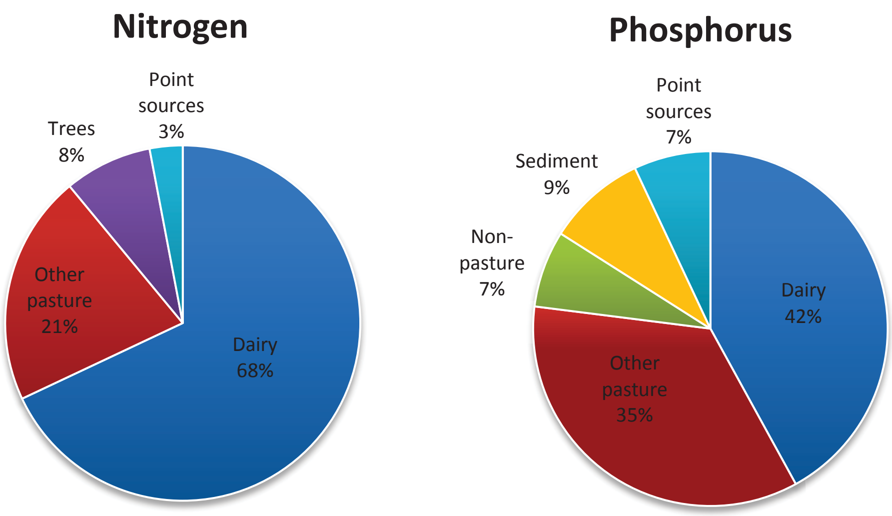


Figure 5.8: Sources of nitrogen and phosphorus entering streams in the Waikato region
Adapted from: Environment Waikato (2008).

Land use changes can significantly increase nutrient inputs into streams and rivers. For example, converting a 36,500 hectare pine forest in the upper Waikato catchment to dairy and drystock farming was estimated to increase nitrogen loss by more than 17 fold (Environment Waikato, 2008). The pine forest was estimated to leach 1.3 kg nitrogen/ha/yr (48 tonnes in total per year) and 0.08 kg phosphate/ha/yr (3 tonnes per year). If converted to dairy, leaching would increase to 23.8 kg N/ha/yr (total of 870 tonnes per year) and 1.3 kg P/ha/yr (total of 49 tonnes per year). Consequently, average summer nitrogen levels in the Waikato River hydro lakes would increase by 18-23% and would more than counteract the benefit of the $15 million upgrade to Hamilton's sewage treatment plant which decreased N inputs into the river by 460 tonnes per year (Environment Waikato, 2008).

Nitrogen leaching and land use

Dairy farms may have greater nitrogen losses than other pastoral farming systems because of the greater stocking intensity and higher soil fertility than other farming types (Ledgard et al., 1996). In the Waikato region, dairy farms leach on average 12 times more nitrate per hectare than land in forest (Figure 5.9) (Environment Waikato, 2008). Land used for crop and vegetable growing leaches higher rates of nitrogen than dairy land but only covers a small land area in the region (Environment Waikato, 2008). Moreover, a dairy wintering area was found to leach 42 times the amount of N than land in forestry (Monaghan et al., 2007). The greatest contributors to nitrogen loss on dairy farms are stocking rate and the amount of N fertiliser application (Longhurst & Smeaton, 2008). A strong relationship is apparent between stocking rate and nitrogen loss from dairy land (Figure 5.10) (Environment Waikato, 2008). This leads to higher nitrogen levels in rivers, driving the relationship between dairy stock numbers and decreasing water quality (Larned et al., 2004).

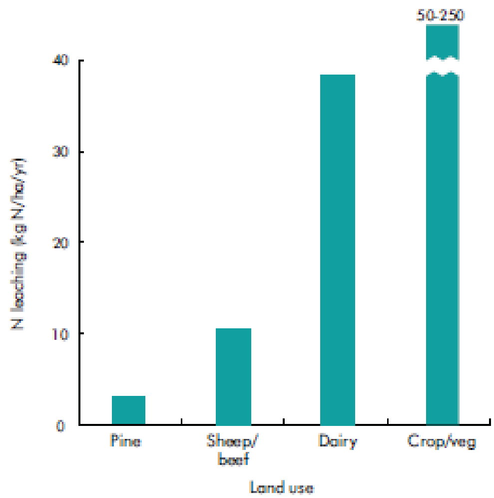


Figure 5.9: Nitrogen loss from different land uses in the Waikato Region.
Source: Environment Waikato (2008).

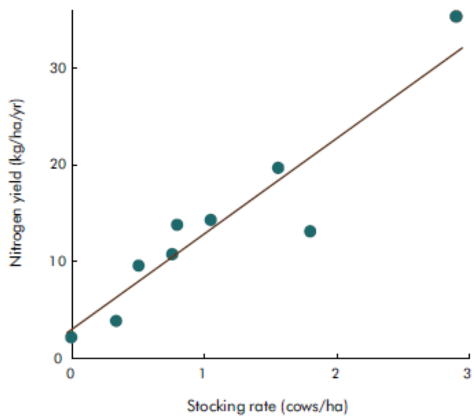


Figure 5.10: Relationship between stocking rate and nitrogen yield from dairy farms
Source: Environment Waikato (2008). Based on Environmental Waikato monitoring data and Agribase.

Livestock urine is the largest source of nitrogen leaching from dairy farms (Figure 5.11) (Davies-Colley et al., 2003; de Klein & Ledgard, 2001; Environment Waikato, 2008; Ledgard et al., 2009; Longhurst & Smeaton, 2008; Menneer et al., 2004; Parliamentary Commissioner for the Environment, 2004). Each dairy cow discharges about 23 litres of urine a day (NIWA, 2012), amounting to nearly 150 million litres of urine per day nationally. Urine contains high concentrations of nitrogen in the form of urea and is rapidly converted to nitrate to concentrations considerably in excess of pasture requirements (Ministry for the Environment, 2001a). It has been estimated that as much as 90% of the total N leaching loss is from urine due to the high localised N-concentrations in urine patches (de Klein & Ledgard, 2001; Environment Waikato, 2008; Ledgard et al., 2009). The N loading under a cow urine patch is approximately 1000 kg N/ha, compared with a sheep urine patch of about 500 kg N/ha (Davies-Colley et al., 2003; Ledgard et al., 2009; Menneer et al., 2004). Moreover, N concentrations under urine patches can be up to 10 times greater than under dung patches, and more than 30 times greater than unaffected areas (Ledgard et al., 2009). The very high N levels are unable to be taken up by pasture, thus lead to N leaching (Clark & Harris, 1995; Ledgard & Thorrold, n.d.). N present in dung is mostly in organic forms that are slowly available and hence far less susceptible to leaching (Menneer et al., 2004).

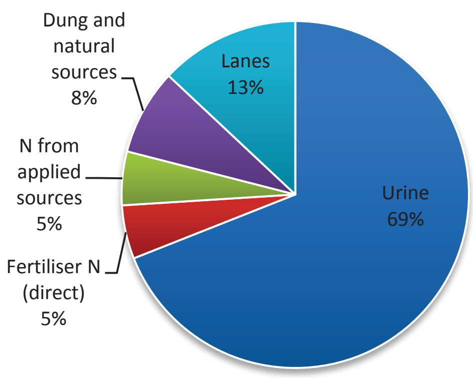


Figure 5.11: Source of nitrogen loss from a typical Waikato dairy farm
Data source: Environment Waikato (2008).

Most urine leaching occurs in autumn-early winter (de Klein & Ledgard, 2001; Longhurst & Smeaton, 2008; Menneer et al., 2004; Roberts & Morton, 2009; Rodda et al., 1999). At these times, N uptake by pasture is slower due to the cooler soil conditions. Nitrate accumulates in the topsoil and cannot be utilised fast enough so is prone to leaching to groundwater (Beukes et al., 2012; Longhurst & Smeaton, 2008). Previous dry summer conditions with associated reduction of plant growth leading to nitrate build-up in soils can further exacerbate leaching when rainfall occurs (Menneer et al., 2004).

Dairy farm nitrogen leaching estimates

Various estimates of nitrogen leaching from dairy farms have been made, ranging from 12-200 kg N/ha/yr (for examples refer to Appendix D), depending on soil type, amount of fertiliser applied, supplementary feed given, stocking rate and irrigation application. Moreover, it is likely that higher rates of leaching occur than those recorded. OVERSEER (a nutrient budget computer model) estimated average N leaching on dairy land of 28 kg N/ha/yr, while the New Zealand average from agricultural land (including dairy land) is 8 kg N/ha/yr (Ledgard et al., 2000). Nitrogen leaching yields from dairy land in the Waikato region estimated in the late 1990s were around 20-35 kg N/ha/yr associated with stocking rates of 2-3 cows/ha (Vant & Huser, 2000; Wilcock et al., 1999). Studies of leaching on irrigated dairy farms in Canterbury have recorded rates as high as 180 kg N/ha/yr (Lilburne et al., 2010), while in the Horizons Regional Council area, nitrogen losses are estimated at 15-115 kg N/ha/yr (Dewes, 2012). On heavily grazed pasture, nitrogen leaching has been estimated as high as 100-200 kg N/ha/yr, mostly draining to groundwater (Ministry for the Environment, 2001a). Dairy intensification (with associated increases in stocking rate) may increase N leaching by up to 100% (Power et al., 2002). For example, a 20% increase in productivity was estimated to increase N leaching from between 23 and 71 kg N/ha/yr (Power et al., 2002).

Higher leaching losses have been associated with higher intensity farms. For instance, nitrogen leaching was found to be lower with lower stocking rates (2.5 cows/ha), large inputs of supplementary feed (2.2 t DM/cow) and having sufficient effluent storage area and a large effluent block (to spread effluent over pasture) (Longhurst & Smeaton, 2008). In contrast, other studies found that leaching increased with more brought-in supplements (Beukes et al., 2012; Ledgard et al., 2006). The disparities between studies may arise due to the inclusion of leaching on land to grow supplementary feed, resulting in higher leaching estimations. Supplementary feed increased milk productivity by 35% to 150% but reduced environmental efficiency (kg N leached/kg MS), as leaching losses were relatively high in areas used to produce supplementary feed. For example, the production of one tonne of maize silage was associated with about 3.3 kg of N leaching (Ledgard et al., 2006).

Nutrient loss and fertiliser use

The timing of N fertiliser application is important for uptake by plants and leaching losses. Direct leaching losses are likely to be higher during applications in winter and lower with applications around rapid pasture growth periods (Ledgard et al., 2009; Ledgard et al., 1996; Roberts & Morton, 2009). Direct leaching of N fertiliser has a greater effect on nitrate leaching when N applications are excessive (>400 kg N/ha/yr) or untimely (e.g. >50 kg/ha in winter) (Davies-Colley et al., 2003; Menneer et al., 2004) (refer to table in Appendix E for leaching rates from different fertiliser applications). In one study, an increase in fertiliser application rates from 200 kg N/ha/yr to 400 kg N/ha/yr increased direct fertiliser leaching losses by up to about 20 times (Ledgard et al., 1996). In another study, even at applications of 170 kg N/ha/year, fertiliser leaching per hectare increased by at least two-fold compared with no fertiliser input while milksolids production increased by approximately 20% (Ledgard et al., 2006). An increase in N fertiliser use can also reduce the amount of N fixing by clover (Ledgard & Thorrold, n.d.), thus requiring even more inputs to maintain pasture growth.

Furthermore, indirect losses were affected even more by increased N in urine. Increased leaching is not necessarily a result of higher rates of fertiliser application, but rather the associated increases in stocking rates that more fertiliser application allows (Davies-Colley et al., 2003; Ledgard et al., 2009; Ledgard et al., 1996; Roberts & Morton, 2009). Fertiliser N is generally used efficiently by pastures but it increases pasture N uptake by cows, increasing N excretion in urine (Ledgard et al., 2009; Roberts & Morton, 2009).

Phosphorus losses

Phosphorus is mainly lost from the land bound to soil particles. For instance, in one study Menneer et al. (2004) found that up to 80% of P lost was run-off in the form of particle-bound P while less than 20% was lost as dissolved P. Phosphorus losses from more intensive land uses are likely to vary substantially with differences in animal stocking rate, soil type, topography, cultivation, fallow periods, cover crop and P fertiliser management (Menneer et al., 2004). Phosphorus surpluses on New Zealand dairy farms typically range between 20 and 50 kg P/ha/yr (Longhurst & Smeaton, 2008). Phosphorus inputs from fertiliser and feed into farms tested by Longhurst and Smeaton (2008) averaged 56 kg P/ha/yr (range 43-75) with surpluses averaging 45 kg P/ha/yr (range 33-60). Losses from these farms averaged 2.2 kg P/ha/yr (range 0.7-4.3 kg P/ha/yr) (Longhurst & Smeaton, 2008). Average losses from dairy areas in Southland were 1.3 kg P/ha/yr (Monaghan et al., 2007), and from the Horizons region

0.2-1.0 kg P/ha/yr (Dewes, 2012). Other studies have recorded much higher losses. For example, a dairy catchment in an extremely high rainfall area of Westland recorded P losses of 10 kg P/ha/yr (Davies-Colley & Nagels, 2002). Examples of P losses from dairy land are listed in Appendix F.

5.1.3. Sediment

Soil erosion is an issue in many agricultural areas in New Zealand due to the physical nature of New Zealand's terrain and the climatic conditions. However, it is accelerated by land clearance and poor land management practices. Sediment loads from pasture catchments have been measured at 1.5 to 5 times more than catchments with forest cover (Ritchie, 2011). Soil loss is also associated with phosphorus losses as P binds to soil particles (as previously explained). In some Otago dairy farming catchments, Matthaei et al. (2006) found streams had 47% cover of fine sediment. In an intensive dairying catchment, peaks of suspended sediment (SS) and total phosphorus (TP) in streams occurred in winter-spring, associated with stream bank erosion (McDowell & Wilcock, 2007). McDowell and Wilcock (2007) attribute this to stock trampling and destabilisation, channel straightening and sediment removal, as well as the removal of riparian vegetation. During summer and autumn, overland flow contributed most of the P-enriched sediment into streams (McDowell & Wilcock, 2007).

Livestock can be particularly damaging to stream banks and wetland vegetation when allowed access to stream channels or lake margins by causing the mobilisation of sediment into these areas (Davies-Colley et al., 2003). Riparian areas are important determinants of sediment entering waterways as stream banks distribute significant quantities of sediment to waterways (Ritchie, 2011). Riparian vegetation can slow run-off and allow large particles to settle out, however, vegetation will not protect against run-off as sub-surface flow or through tile drains (Ritchie, 2011).

5.2. Water Quantity

New Zealand has a wealth of water resources; it has more freshwater per person than more than 90% of almost 200 countries (Ministry for the Environment, 2007a). However, New Zealand's water usage per person equates to two to three times more than most other OECD (Organisation for Economic Co-operation and Development) countries (OECD, 2007). In 2010, New Zealand was the 4th highest water abstractor per capita in the OECD (OECD, 2013). Water use has been increasing in the past few decades, particularly in response to increased use for irrigation for dairy farming intensification in drier areas.

Regional Councils issue consents for irrigation water and are responsible for monitoring and controlling access to water once it has been allocated.[11] Generally, water is allocated based on minimum flow river conditions (Clark et al., 2007); however, water is over-allocated in some catchments (Lincoln Environmental, 2000). The allocation is the amount that is allowed to be used from a water resource and does not include most stock takes as these are generally classed as permitted activities under the RMA and do not require a consent (Aqualinc, 2006). Permitted uses include water for stock and dairy shed activities but the RMA does not specify a permitted amount so these may differ by Regional Council.

Water for permitted uses

Water for permitted uses may be significant relative to other water uses. For example, in the Waikato Regional Plan additional takes of up to 15 m^3/day of surface water for permitted activities are allowed, provided there are no adverse effects on the environment (Brown et al., 2007). However, adverse effects may not be detected as there is little information available about the number of, location and amount of water used by permitted activities (Brown et al., 2007). Dairy farms with more than 215 cows are likely to use more than the permitted amount of 15 m^3/day (Brown et al., 2007). In 2012, the average herd size in New Zealand consisted of 393 cows and there were over 9200 herds with more than 200 cows (78% of total herds)[12] (LIC & DairyNZ, 2012). It is likely that these herds are using more than 15 m^3/day. However, in 2010 there were only 1204 consents issued by Regional Councils nationally for stock water (Rajanayaka et al., 2010). Although other regions may permit taking more than 15 m^3/day without a consent, the total amount taken may represent much more than considered when resource managers allocate water for minimum river flows.

Brown et al. (2007) determined that total water use in a catchment is highly influenced by parameters relating to dairy cows, people and leakage. Dairy cows require the most water out of all stock types. Lactating dairy cows are expected to use around 70 litre/cow/day, but when water for dairy shed wash-down is included the total may be as high as 140 l/cow/d (0.140 m^3/cow/d) (Brown et al., 2007). Additionally, if feed pads are used, more water may be

[11] The amount allocated in 2010 for irrigation by region is shown in Figure 2.24 in Chapter 2.
[12] Herd statistics were similar in 2009/10 (average herd size: 376; 9054 herds with more than 200 cows (74% of total herds)) (LIC & DairyNZ, 2010).

required (Brown et al., 2007). About 64-70% of permitted water used in the Waikato region is used for dairy cow drinking water and dairy shed operations (125,700 m^3/day out of a total permitted water use of 196,600 m^3/day) (Brown et al., 2007). This includes water to cool milk, clean equipment and wash down bails and other areas. The remaining third is for drinking requirements of other animals, public domestic use and other activities including leakage (Brown et al., 2007).

5.3. Impacts of Contaminants in Water

Freshwater ecosystems have probably been the most affected by intensive agriculture. Evidence shows that dairy farming practices have contributed substantially to the degradation of freshwater. These impacts reduce plant and animal diversity, threaten public health, reduce productivity and animal health, and diminish aesthetic, cultural and recreational values of waterways (Blackwell et al., 2006).

5.3.1. Pathogen contamination

Water contaminated with faecal pathogens affects recreational uses, drinking water quality (Davies-Colley et al., 2003; Ministry for the Environment, 2009b; Parliamentary Commissioner for the Environment, 2004) and shellfish in estuaries and lakes (Collins et al., 2007; Donnison & Ross, 1999) because of the health effects involved. Contaminated water can also effect livestock, causing reduced growth, morbidity, or mortality (Smith et al., 1993). This is not a new issue; in 2001 the Ministry for the Environment (2001a) noted that freshwater was becoming so polluted that some lowland streams were not only unsuitable for humans to drink or swim in, but they were also unsafe for livestock to drink. Donnison and Ross (1999) also suggested that all river water in New Zealand is at risk of containing pathogens and hence is unsafe to ingest. Although it has been 15 years since this statement was made, it still has relevance today.

Health effects

The most common human health effects from contaminated water are gastro-intestinal diseases (GID) caused by pathogens, and result in symptoms like diarrhoea or vomiting, and infections of the eye, ear, nose and throat. Harmful forms of GID include giardiasis, cryptosporidiosis, campylobacteriosis, and salmonellosis, and infection with *E.coli* 0157; viruses such as hepatitis A can also be contracted from contaminated water (Ministry for the Environment, 2011b). Thirty per cent of the total cases of cryptosporidiosis were attributed to waterborne transmission, the highest out of all GIDs (Ball, 2006). However, there are far more cases of waterborne campylobacteriosis than other GID.

Estimates of endemic waterborne GID are around 18,000 and 34,000 cases per annum, although this is predicted to be an under-estimate (Ball, 2006). Using these assessments, an annual rate of GID from drinking-water was measured at 878 cases/100,000 using the 2000 population statistics (Ball, 2006). However, the *notified* rate of GID in 2000 was only 378 cases/100,000, of which only around 15.5 – 140 cases/100,000 were estimated to be from waterborne sources (Ball, 2006). Certainly, notified underestimations are likely because not all cases are reported or diagnosed by medical practitioners. It is difficult to attribute the source of waterborne diseases so it is unclear the proportion that would be caused by dairy farming.

5.3.2. Nutrient impacts

5.3.2.1. Groundwater

Elevated nitrate (NO_3) levels are found in many shallow groundwater aquifers (down to 60 metres), especially in high stocking areas and below dairy land (Ministry for the Environment, 2007b; Parliamentary Commissioner for the Environment, 2004). The Ministry of Health's (MoH) Maximum Acceptable Value (MAV) for nitrate in drinking water is 50 mg/L nitrate (Ministry of Health, 2008), equivalent to 11.3 mg/litre nitrogen as nitrate-N (Cassells & Meister, 2000; Ford & Taylor, 2006). Leaching of one kg nitrate N will pollute 88.5 cubic metres of water (88,496 litres) from a level of no nitrate to a level 11.3 mg/L nitrate (personal communication with Peter Robinson, Hill Laboratories, June 2013).

In 2008, around 30% of groundwater sites under dairy land in Waikato did not meet the MoH drinking water guidelines, compared with only 5% from drystock farms and urban wells (Figure 5.12) (Environment Waikato, 2008). Likewise, groundwater testing in the 1980s on Taranaki dairy farms showed that over 40% exceeded nitrate drinking water standards (Ledgard et al., 1996). It is likely these levels have increased since this testing. Testing in Canterbury groundwater wells in 2012 showed that 33 out of the 289 wells (11%) tested did not meet the MAV for nitrate drinking water standards (Environment Canterbury, 2013b). This was up from

7% in 2011 (Environment Canterbury, 2013a). Most of these sites are in Ashburton (Environment Canterbury, 2013a, 2013b), located on the free-draining alluvial Canterbury Plains, and an area with a large predominance of dairy farming.

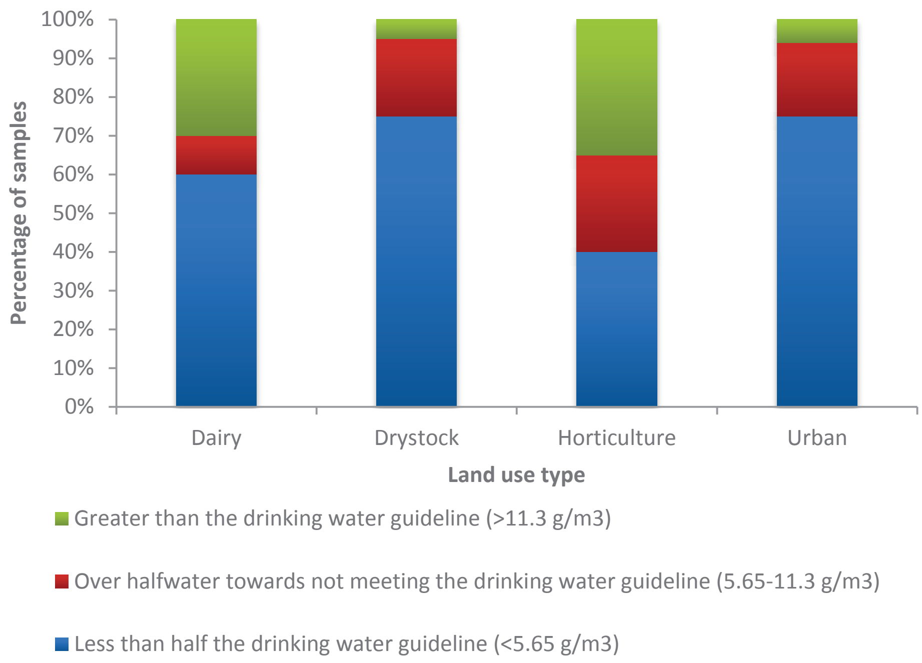


Figure 5.12: Proportion of Waikato groundwater sites under different land uses with nitrate levels exceeding or below drinking water guidelines.
Data source: Environment Waikato (2008).

Heavy fertiliser applications further exacerbate consequences in groundwater. For example, groundwater under farms receiving 400 N fertiliser loadings had nitrate concentrations averaging about twice the recommended maximum for drinking water and were close to the recommended maximum for livestock (30 mg nitrate-N/litre) (Ledgard et al., 1996).

Elevated nitrate levels in groundwater are an issue because about 40% of New Zealand's population relies on groundwater as a source for drinking water (Rajanayaka et al., 2010). Drinking water contamination from nitrogen can lead to certain types of cancers and has been linked with blood disease in infants, known as the blue baby syndrome (Ministry for the Environment, 2007a; Parliamentary Commissioner for the Environment, 2004). Some Canterbury groundwater wells have level so high that pregnant women and infants who rely on wells for drinking water are urged to check nitrate levels before consuming (Scott & Hanson, 2013).

5.3.2.2. Surface water

Nitrate level thresholds for ecological protection are much lower than for drinking water. Ecological effects in surface water have been shown to occur at nitrate levels lower than 1 mg/L (Davies-Colley, 2000). The construction of an irrigation dam has been proposed in the Hawkes Bay region – the Ruataniwha Water Storage Scheme. The Board of Inquiry for the proposed dam ruled that dissolved inorganic nitrogen (DIN) levels in the Tukituki River would not be allowed to reach above 0.8 mg/L (Science Media Centre, 2014). This decision could limit intensification that would likely occur from the irrigation scheme. A predictive model of levels of DIN above 0.8 mg/L in New Zealand rivers reaches showed levels were strongly associated with the proportion of upstream catchments in heavy pastoral landcover (Figure 5.13) (Curtis, 2014). Areas above DIN levels of 0.8 mg/L are associated with dairying land use.

Meanwhile, the Government's proposal for managing freshwater has an upper limit for nitrate of 6.9 mg/L (Ministry for the Environment, 2013e). Impacts from nitrogen are very unclear, particularly when several things are affecting a river. The limit of 6.9 mg/L is associated with the point when nitrogen is toxic to 20% of species (i.e. gives an 80% protection rate to aquatic species) (Hickey, 2013). Environments where these levels of nitrogen exist are described as 'highly disturbed systems' and are 'measurably degraded' (Hickey, 2013). Toxic nitrogen levels to 10-20% of aquatic species can be even lower, at 2.4 – 3.6 mg/L (Hickey, 2013; Hickey & Martin, 2009). However, far below this level (more like 0.1 mg/L) is where effects start to occur (Davies-Colley, 2000). The Board of Inquiry for the Ruataniwha set its level based on instream ecological health, rather than nitrogen toxicity.

In surface water excessive levels of nitrogen and phosphorus increase plant growth, and can cause algal blooms and an over-abundance of aquatic weeds, leading to eutrophication (enhanced phytoplankton growth) (Marsh, 2012b; Ministry for the Environment, 2007a; Smith et al., 1993; Tilman, 1999). Eutrophication causes highly fluctuating oxygen levels in water (harmful and deadly for fish) as well as poor water clarity, rending water bodies unsuitable for swimming and degrading the aesthetic appeal of freshwater (Smith et al., 1993). Ecological effects of eutrophication in aquatic ecosystems include "loss of biodiversity, outbreaks of nuisance species, shifts in structure of food chains, and impairment of fisheries" (Tilman, 1999, p. 5995). An additional problem associated with surplus nutrients is the time it takes between nutrients being applied to the land and reaching groundwater, lakes and rivers (lag time). This can cause difficulties in predicting nutrient inputs to freshwater and severely hinder the success of management programmes for controlling nutrients.

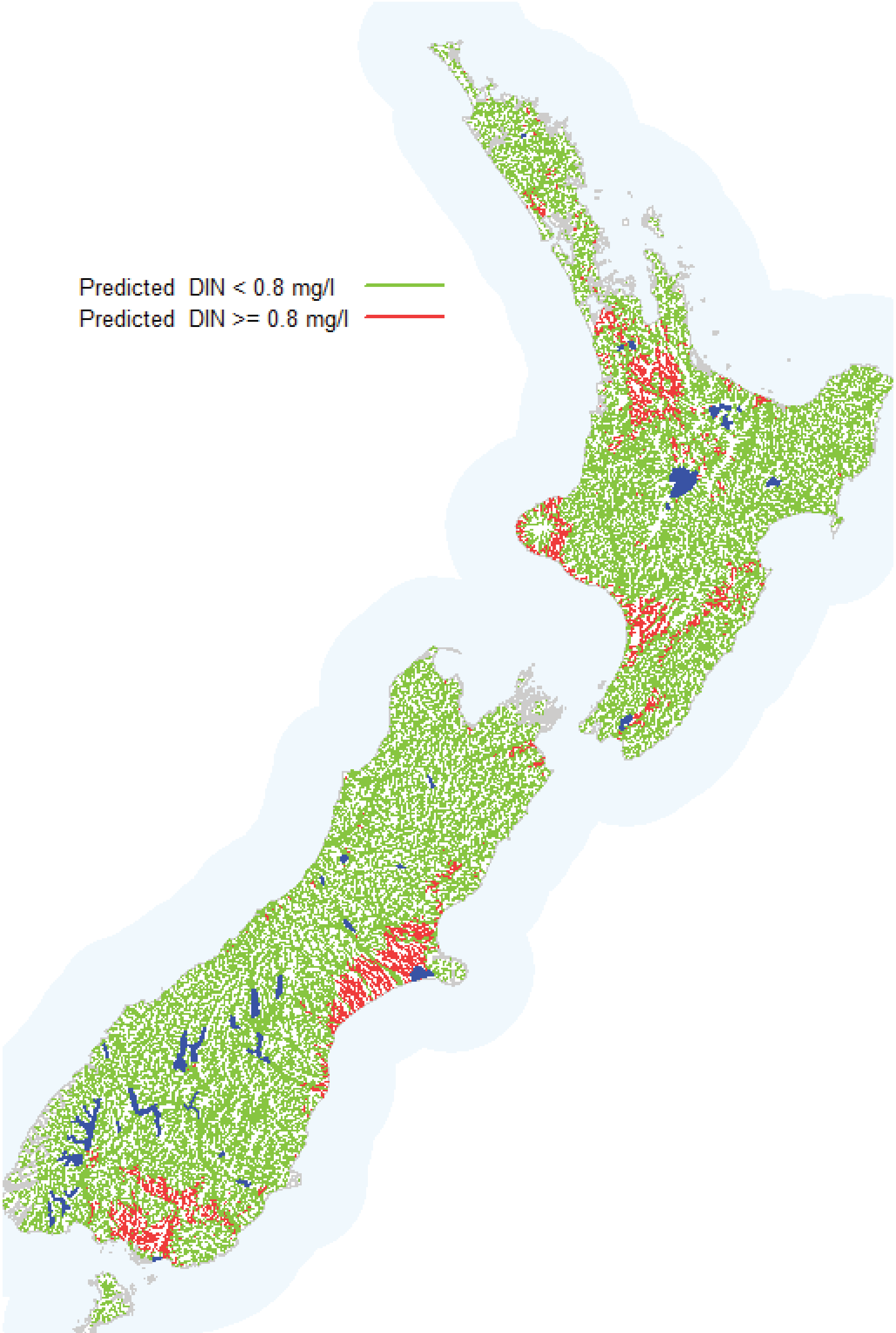


Figure 5.13: Predicted median Dissolved Inorganic Nitrogen (DIN) in NZ rivers relative to a threshold concentration of 0.8 mg/L based on observations at 622 sites, 2006-2012.
Source: Curtis (2014).

Nutrient losses from agricultural land have led to increased incidence of algal blooms in rivers and lakes (Marsh, 2012b). Algal and cyanobacteria blooms (blue-green algae) are likely to occur at sites with high levels of GPP (Gross Primary Productivity - algal photosynthesis). Sites where there is plenty of light and nutrients available to support plant growth will have higher rates of GPP (Young, 2009). Problems with this are degradation of aesthetic and recreational values, large fluctuations in pH and DO levels, smothering of habitat for invertebrates, and taste and odour problems for water supplies. Cyanobacteria blooms occur frequently in the Waikato River catchment as a result of high nutrient concentrations (from farmland) and long residence times in the hydro lakes (NIWA, 2010b). Cyanobacteria can affect public water supplies and cause adverse health effects to recreational water users, stock and other domestic animals. Fish and other food (such as freshwater mussels) may also accumulate these toxins, posing a "health risk for people consuming them" (NIWA, 2010b, p. 49).

Agricultural streams are often degraded from having wide diurnal changes in pH, temperature and dissolved oxygen (DO), and poor visual clarity (Davies-Colley & Nagels, 2002; Wilcock et al., 2007; Wilcock et al., 1999). Continuous monitoring of pH, temperature and DO in dairy streams have revealed extreme values not normally observed in general monitoring programmes (where parameters are measured once or twice a day if that) (Wilcock et al., 2007), highlighting the significance of measuring both during the day and at night.[13] Minimum DO levels measured in five dairy catchments were almost all below the guideline for the protection of aquatic ecosystems, fisheries and fish spawning of >80% saturation; healthy ranges for biodiversity are between 98 and 105% saturation (Wilcock et al., 2007). In autumn, DO levels for the Toenepi Stream (predominantly in dairy) were all below 40% saturation and some studies have reported levels of less than 10% DO in this stream (Wilcock et al., 2007). Wide DO ranges were found during winter and spring in the Toenepi. In the Manawatu River, DO levels ranged between 40% and 332% (Clapcott & Young, 2009). DO levels of 10-50% were measured in the Whangamaire Stream, near Pukekohe, which is predominantly in market gardening with pastoral agriculture in lower regions (Wilcock et al., 1995). Concentrations of nitrate nitrogen were 18.3 mg/L in spring waters feeding the stream and nitrate concentrations were typically 7-10 mg/L in the lower reaches (Wilcock et al., 1995). Five point source agricultural discharges were entering the study reach, all from dairy farms (Wilcock et al., 1995). Highly fluctuating DO levels with both low and high levels are deadly to fish (Young, 2009).

[13] Some problems with the measurement of freshwater quality are outlined in Chapter 10.

Lakes can be useful in determining the impact of land use on water quality as nutrient levels in lakes tend to reflect land-use within the catchment (Vant & Huser, 2000). However, the impacts from current land use may not be apparent in lakes for years because of lag times for nutrients to reach lakes. The time lag between an action being taken and the consequential effects on water quality will vary depending on location, catchment size and activity in the catchment. Often lags are often difficult to quantify. Time lags can span many years or even decades, which can have significant implications for monitoring programmes (Ministry for the Environment, 2009b). Because of this lag time, even if changes are made now to reduce nutrient losses from land, nitrogen reaching groundwater, streams, rivers and lakes is likely to increase from past activities and lake water quality may continue to slowly decline in response to current intensification (Parliamentary Commissioner for the Environment, 2004; Vant & Huser, 2000).

Many shallow lakes in New Zealand are nutrient enriched (eutrophic) (Smith et al., 1993) and most of these are in lowland pasture dominated catchments in the North Island (Parliamentary Commissioner for the Environment, 2004; Verburg et al., 2010). Deeper lakes are also at risk of becoming eutrophic from nutrient inputs. Forty four per cent of monitored lakes in New Zealand are in a eutrophic state or worse (Verburg et al., 2010). Monitoring in lakes has also found that half of the lakes in pastoral land had poor ecological conditions and bottom water oxygen concentrations decreased significantly with increased percentage of pastoral land cover (Verburg et al., 2010).

Monitoring in Lake Taupo indicates that the condition of the lake is gradually worsening from the addition of nitrogen, resulting in increased plant growth. Additionally, the lag time between nitrogen entering the lake is several decades (between 30 and 80 years), resulting in worse conditions to come (Parliamentary Commissioner for the Environment, 2004; Vant & Huser, 2000). Effects of land clearance around Lake Taupo in the 1950s is only now beginning to emerge in increased nitrate levels in streams around the lake; it is estimated that nitrate levels in streams around Lake Taupo will increase from 30-40% above current levels due to this time lag (Environment Waikato, 2008). It was estimated that conversion of 100-250 km^2 of sheep and beef pasture to dairying in the Lake Taupo catchment would increase nitrogen levels and hence phytoplankton levels in the lake by about 20-60% causing water clarity to fall by about 20-40% (Vant & Huser, 2000). In sheltered and near-shore areas these changes may become apparent reasonably quickly but in deeper parts it may take longer. Similar issues of declining water quality due to time delays are also occurring in the Rotorua Lakes (Bay of

Plenty Regional Council et al., 2010; NIWA, 2012; Parliamentary Commissioner for the Environment, 2004).

5.3.1. Sediment effects

When eroded soil enters waterways it causes an influx of suspended sediment (suspended in the water flow) and deposited sediment (on the bed), causing many water quality issues such as poor water clarity, smothering of food and habitat, and damage to aquatic organisms (McEwan & Joy, 2011, 2013). Suspended sediment causes turbidity (cloudy water due to sediment particles interfering with light) (Davies-Colley et al., 2003), reducing light transmission (light attenuation) through water, thus causing reduced visual clarity and light availability for photosynthesis (Davies-Colley & Smith, 2001). A reduction in photosynthesis results in reduced food availability and plant biomass. Reduced visual clarity will affect vision for fish as well visual predators such as aquatic birds (Davies-Colley et al., 2003). Suspended sediment can cause other detrimental effects for fish, such as death, a reduction of growth rates and disease resistance, diminished egg and larval development, reduced food availability, modification of movements and migrations (Alabaster and Lloyd, 1980 cited in Smith et al., 1993), and can clog fish gills (Bruton, 1985).

Benthic dwelling animals are particularly harmed by deposited sediment. Wood and Armitage (1997) explain that deposited sediment affects benthic invertebrates by altering substrate composition and changing habitats; damaging respiration, either by affecting respiratory structures or reducing oxygen concentrations in the substrate; and degrading feeding habits, by damaging filter feeding structures or reducing the periphyton food quality. By clogging the interstitial spaces within the stream bed crucial to fish and invertebrate life, sediment can smother habitat and degrade food availability for fish (Bruton, 1985; McEwan & Joy, 2011; Smith et al., 1993; Wood & Armitage, 1997). Fish spawning habitat can be degraded and reduced, which can have a major impact on spawning success (Wood & Armitage, 1997). As most of New Zealand's native freshwater fish are benthic, this can have major implications for the longevity of many fish species.

Suspended sediment also has impacts on the recreational use of waters and can hinder safety (Davies-Colley et al., 2003). Standards for visual clarity are imposed by the RMA to protect visual habitat and amenity value of water (ANZECC, 2000). For contact recreation, a minimum black disc visibility of 1.6m is recommended by ANZECC (2000). Measuring visual clarity (using black disk visibility) provides an instant synopsis on aesthetics, contact recreation, and fish habitat (Davies-Colley & Smith, 2001). Over 75% of pasture streams measured from 2003-2007 fell below guidelines for water clarity of 1.6 m (Ballantine et al., 2010). In a national review, Larned et al. (2004) found that the median water clarity of streams draining pastoral and urban land cover was 40-70% lower than streams in native and plantation forest classes. Correspondingly, temperature and conductivity were trending upwards in low-elevation rivers in the period 1996-2002 (Larned et al., 2004). Additionally, Smith et al (1993) found that base flows in streams draining agriculture catchments in the Wellington region were four times more turbid than those in native bush catchments.

Sedimentation rates in estuaries and lakes suggest that yields are about 10 times greater than when land was forested (Parliamentary Commissioner for the Environment, 2004). This is causing infilling of lakes, estuaries and reservoirs which can lead to a reduction in water storage; shoaling of channels in estuaries; and the siltation of water supply intakes (Clark et al., 1985, cited in Davies-Colley et al., 2003). Fine sediment tends to flocculate and settle at the border of rivers and estuaries (Davies-Colley et al., 2003). Fine sediment deposition can smother shellfish (Environment Waikato, 2008), affect food supplies of suspension-feeding shellfish (Davies-Colley et al., 2003) and eels, and destroy spawning habitat for ocean fish like snapper (Morrison et al., 2008).

5.3.1. Impacts in dairying catchments

Poor water quality has been recorded in catchments dominated by dairy farming. High concentrations of N, P and faecal bacteria were measured in five New Zealand dairying catchment streams (Wilcock et al., 2007). These are attributes correlated with poor water quality. Other studies revealed that mean concentrations of N and P were well above typical levels for low elevation streams in New Zealand in four of the streams (Toenepi, Waiokura, Waikakahi and Bog Burn) (Larned et al., 2004; Wilcock et al., 2007).

High nutrient levels in the Toenepi Stream attributed to intensive dairy farming were coupled with riparian shade loss and reduced flows. These factors caused low levels of dissolved oxygen (DO) and sluggish, warm conditions in the stream (Wilcock et al., 1999). Water quality in the Bog Burn catchment in Southland deteriorated in response to a change from forestry to dairy farming down the catchment. Measurements of faecal indicators, nutrients and turbidity in the Bog Burn catchment were well above the ANZECC trigger values for lowland streams and were similar to other dairy catchments (Monaghan et al., 2007). Additionally, nitrogen concentrations increased down the catchment with a change to dairy farming. Milking areas covered 27% of the catchment and contributed 35% of the N loads, while dairy wintering areas occupied only 10% of the catchment but were responsible for over 40% of the N loss (Monaghan et al., 2007). Dairy land emitted over twice the amount of phosphorus to water per hectare than dry stock areas and more than 6 times the amount of forestry (leaching losses for different land uses are shown in Appendix G) (Monaghan et al., 2007). Monitoring from 14 dairy farming catchments in New Zealand showed that 13 exceeded the soluble inorganic nitrogen and half the dissolved reactive phosphorus (DRP) guidelines for the prevention of nuisance periphyton growth (Ministry for the Environment, 2009b). Additionally, in seven out of nine catchments measured, excessive weed and/or algal growth was found.

5.3.2. Cultural impacts

Water is very culturally and spiritually significant to Māori; it is believed to be the lifeblood of Papatūānuku (earth mother) and the tears of Ranginui (sky father). Māori have strong cultural, traditional and historic links with freshwater including wetlands, lakes, rivers, streams and springs (Ministry for the Environment, 2001a). The quality of freshwater is an important aspect of their lives and responsibilities as kaitiaki (care-takers of the land). Water quality is a reflection of the state of the land and thus the health of tangata whenua (people of the land) (James, 1993). Rivers are often considered a taonga (treasure) encompassing the entire river system, not just one part (James, 1993) and each waterway is regarded as having its own mauri (life-force) and mana (authority, prestige or power). Water with a healthy mauri will sustain healthy ecosystems, support cultural activities, provide mahinga kai (food sources) and thus, be a “source of pride and identity to the people” (Ministry for the Environment, 2001a).

The mixing of water is of particular concern to Māori. This includes mixing of waters from different sources with separate mauri and discharges of ‘used’ waters or wastes to living waters that supply food (Ministry for the Environment, 2001a). Waters diverted from one river to another (e.g. for irrigation) is also an unwanted activity (Ministry for the Environment, 2001a). Diffuse and point-source discharges that enter water contribute significantly to the degradation of the mauri of streams. In particular, discharging effluent to waterways is culturally offensive to Māori and depletes the mana of the waterway as well as the mana of the local iwi. Many Māori have a strong cultural belief that waste should be cleansed through contact with land before returning to water bodies (NIWA, 2010b). In this regard, land disposal is seen as the only acceptable option for wastewater and effluent treatment.

In the Waikato River catchment, recreational and cultural activities have been affected by many land practices, including agriculture. Using the river for swimming is seen by iwi as reconnecting themselves to their awa tupuna (ancestral river). However, contact recreation is unsafe in some parts of the river due to faecal contamination (NIWA, 2010b). The safety of drinking water and food taken from the river is affected by faecal pathogens and cyanotoxins (at times of some algal blooms). Foods that are eaten raw, such as watercress, are especially at risk. Donnison et al. (2009, cited in NIWA, 2010b) found that watercress sourced from unfenced small streams on a sheep-beef and a dairy farm typically had *E.coli* levels at marginal or unsatisfactory levels according to New Zealand guidelines for ready to eat food (FSANZ, 2001, cited in NIWA, 2010b). Average *E.coli* concentrations in these streams were 461 and 710 *E.coli*/100 mL for sheep/beef and dairy respectively, typical of unfenced headwater tributary streams on pastoral farms in the Waikato. On the other hand 90% of watercress taken from a bush reserve was satisfactory (NIWA, 2010b).

A Cultural Health Index (CHI) for streams has been developed, linking Western scientific methods and cultural knowledge about stream health, to integrate iwi participation into land and water management processes and decision making (Tipa & Teirney, 2003). Results show that the CHI is a credible measure of stream health that correlates well with Western scientific methods of stream health. It also is a good indicator of land use within a catchment (Tipa & Teirney, 2003). Cultural indicators address cultural values, "assess the state of the environment from a cultural perspective", and involve Māori in environmental monitoring (Harmsworth et al., 2011, p. 423), management, and decision making (Tipa & Teirney, 2003). Scientific and cultural river health indicators can be complementary, reflecting different knowledge systems and perspectives (Harmsworth et al., 2011). Other studies have identified cultural preferences for stream flows, identifying flows which cater for Māori cultural values (Tipa & Nelson, 2012). Results suggest that current flow regimes, which only set minimum flows, are unsatisfactory (Tipa & Nelson, 2012).

5.3.3. Irrigation effects

Irrigation is a key component of intensification in drier regions and it makes some types of farming possible in areas that were previously difficult or impossible to carry out

(Parliamentary Commissioner for the Environment, 2004). Most of the land use change in New Zealand associated with irrigation has been towards dairying or dairy support land. Two major environmental effects correspond with irrigation: the effects of reducing the amount of water in water bodies and water quality effects caused by the use of allocated water (Parliamentary Commissioner for the Environment, 2004). Irrigation reduces natural water flow, thereby increasing temperatures and altering sediment movement. These effects could render rivers unsuitable for breeding grounds or habitat for some fish (Parliamentary Commissioner for the Environment, 2004).

Secondly, irrigation permits farmers to grow more pasture, allowing them to have higher stocking rates. Higher stocking rates result in more urine deposition on pasture, increasing N leaching. Associated with this intensification is the application of fertilisers to assist with pasture growth and increase milk production; hence irrigation increases the risk of N-leaching losses (Green et al., 2012).

Potential for runoff from irrigation application as either surface runoff or sub-surface flow occurs when excessive volumes of irrigation are applied. Runoff feeds streams and can account for a good proportion of summer flows in very dry regions (McDowell et al., 2011). McDowell et al. (2011) estimated that irrigation may increase N and P concentrations in receiving streams by 30-400% if dilution via direct discharge did not occur (nutrient diluted with irrigation water).

Increasing irrigation is having an effect on nitrate levels in groundwater. Under natural conditions in groundwater, nitrate-N concentrations are low, generally less than 3 mg/L (Ford & Taylor, 2006). In shallow, unconfined aquifers recharged primarily by soil drainage, nitrate levels were significantly higher than in coastal aquifers and groundwater where streams and rivers were the dominant source of recharge (Ford & Taylor, 2006). It is estimated that nitrate concentrations in these shallow aquifers could exceed the limit for drinking water standards within 30 years (Ford & Taylor, 2006). Remediation of aquifers is likely to take considerable time and resources; thus, preventing a decline in groundwater quality is the only effective and economic way of managing groundwater resources (Ford & Taylor, 2006).

Irrigation in Canterbury

Land use trends from dry pastoral and arable farming to heavily irrigated dairy farming is occurring in Canterbury with dairy cows numbers increasing at a faster rate than the rest of the country, putting pressure on water resources. Approximately 75% of the Canterbury area is in some type of agricultural production, with the most intensive farming occurring on the down lands and plains (Ford & Taylor, 2006). Eighty-seven per cent of irrigated land in Canterbury is in pasture (Rajanayaka et al., 2010). Irrigated pastoral land increased by 110% between 1999 and 2010 and about 70% of consumptive water use in the region was used for irrigation in 2010 (Rajanayaka et al., 2010).

In an irrigated pastoral catchment in Canterbury (Waikakahi Catchment) nitrogen losses from dairy farms varied between 27 kg N/ha/yr (poorly drained soils) to 52 kg N/ha/yr (free-draining soils with high N inputs via clover fixation, N fertiliser and imported feed) (Monaghan et al., 2009). P losses from dairy land were between 0.6 and 1 kg P/ha/yr and were greater from poorly drained soils. In the Waikakahi Stream, water quality guidelines for lowland streams (ANZECC, 2000) were exceeded by four times for *E.coli*, two times for turbidity, three times for P concentrations, and six times for nitrate N (NO_{x}-N) (Monaghan et al., 2009). Irrigation runoff was the major cause of most of the elevated contaminant concentrations and loads in the stream. Irrigation losses of up to 50% were measured. High loadings of N, P and faecal matter are a direct result of dairying, which is only possible because of irrigation (Monaghan et al., 2009). Dairy farms contributed more than 70% of nutrient loads to the stream, despite only occupying 40% of the total catchment area.

The Waikakahi flats are now covered almost entirely in dairy farms. Irrigation was once allowed on 63% of any property but now is allowed over the total area of each property (Ministry for the Environment, 2001a). The Waikakahi stream is totally unsuitable for any contact recreation and drinking water and also frequently unsuitable as a stock water source due to high bacterial counts. On an annual basis, the highest faecal coliform levels were recorded in the period October to December, when maximum pasture growth occurs and when irrigation is generally used for the first time in the season. Nitrate levels in the stream are routinely high throughout the year (0.5-2.3 g/m^3 NO_3-N), likely to be predominantly from urine leaching (Ministry for the Environment, 2001a).

5.3.4. Impacts on biodiversity

Ecosystem interactions are complex and influenced by multiple effects (stressors) (Matthaei et al., 2010; Piggott et al., 2012). Addressing the effects of only one stressor in isolation may produce unrealistic results. Multiple stressors often combine to cause biodiversity effects in rivers, common in streams impacted by agricultural land-use practices (Matthaei et al., 2010).

For instance, sediment levels and stream flow affected invertebrate abundance in experimental stream channels (Matthaei et al., 2010). Matthaei et al. (2010) found that reduced flows combined with rising sediment levels decreased total invertebrate abundance in river channels. Water abstraction was expected to have greater effect on reducing invertebrate abundance in streams with high fine sediment inputs than similar streams with lower sediment levels (Matthaei et al., 2010).

Native fish in New Zealand are facing the impacts of land use intensification, among other pressures. Sixty-eight per cent of New Zealand's native freshwater fish were considered threatened or at risk in 2009, compared with 53% in 2005 (Allibone et al., 2010). In 2013, this proportion increased to 74% (Goodman et al., 2014). Freshwater fish decline is particularly prominent in pasture catchments, reflecting poor ecosystem health (Joy, 2009; Joy & Death, 2004). Threats and pressures identified for freshwater fish include habitat loss and degradation (particularly due to land use intensification); competition and/or predation by introduced species; migration barriers; contaminants in water, such as sediment; declining water quality and water abstraction; and fishing pressures (Department of Conservation, 2006; Goodman, 2014). Lowland streams, where agricultural impacts mainly occur, are important migratory pathways for many native fish species (McDowell, 1990).

5.4. Conclusion

Water-bodies in dairying catchments experience poorer water quality than those in other land-uses, due to generally higher loads of nutrients, faecal pathogens and sediment. Recreation activities are affected with many streams and rivers being un-swimmable. Lakes in dairying catchments are also affected with a high incidence of algal blooms. Additionally, high concentrations of nitrate in groundwater under dairy land render water unsafe for drinking. Aquatic biodiversity in many lowland streams is severely affected. Cultural values are affected by poor water quality as water is extremely important for spiritual and nourishment values for Māori. Irrigation exacerbates water quality issues and creates low flow problems in rivers and streams. As more irrigation projects are being implemented, these impacts will worsen.

Chapter 6. LAND

You must teach your children that the ground beneath their feet is the ashes of your grandfathers. So that they will respect the land, tell your children that the earth is rich with the lives of our kin. Teach your children what we have taught our children, that the earth is our mother. Whatever befalls the earth befalls the sons of the earth. If men spit upon the ground, they spit upon themselves.

(Native American Wisdom)

Historical impacts on land from dairy farming include native forest removal (particularly lowland forest) and wetland drainage. These changes resulted in large scale habitat removal and biodiversity loss, and occurred mainly during land transformations to agriculture decades ago. Much of this land may have not been directly converted to dairy but to sheep and beef, and then later to dairy; however, much of the productive lowlands were probably originally converted to dairy (Willis, 2001). Some of the landscape alterations that have occurred with increased agricultural use are briefly discussed including the associated biodiversity loss. However, the main focus in this chapter is on the current impacts of dairy farming on land, mainly regarding the effects on soil.

Intensification of dairy farming has direct impacts on soil with associated impacts on production, thus affecting potential future land uses. Fertilisers and other agricultural chemicals applied to dairy land often contain heavy metals that can reach high levels in soil, consequently risking the potential to export produce and grow certain crops on contaminated land. Overstocking cows and using heavy machinery, among other management practices, causes soil compaction. Compaction has many severe physical impacts on soil that may limit production and increase runoff of contaminants.

Five key issues have been identified as threatening the loss and damage of soil resources. These are soil compaction, loss of soil organic matter, excessively high fertility levels, erosion risk, and accumulation of contaminants (Taylor, 2011). These issues are particularly problematic on dairy land. Soil compaction, excessive fertility, and accumulation of contaminants are considered in more depth here. Major impacts on land and soil from dairy farming are presented in Figure 6.1.

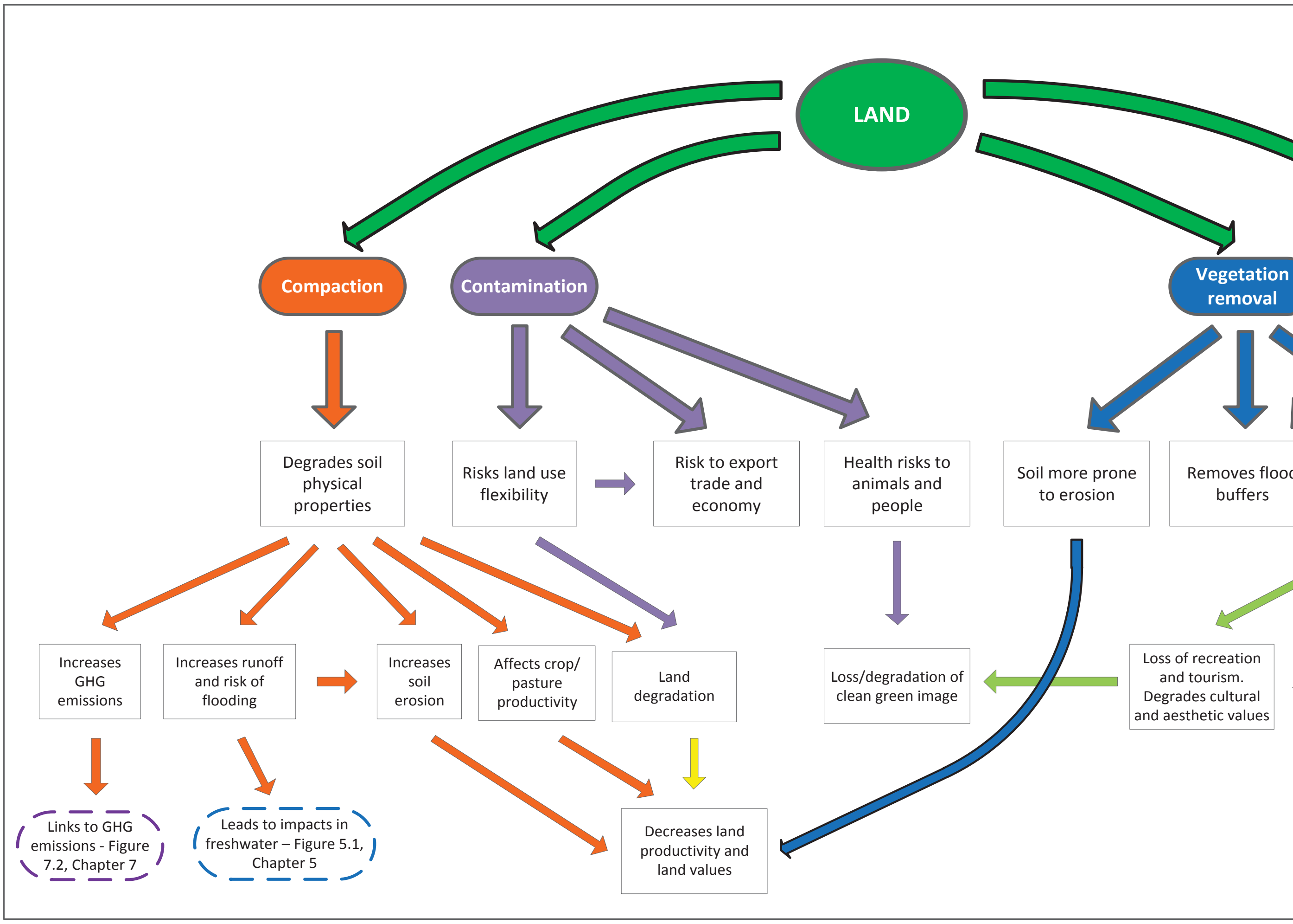


Figure 6.1: Major impacts on land and soil from dairy farming.
Notes: Yellow arrows depict the flow from combined inputs.

6.1. Biodiversity Loss

Agricultural systems have transformed natural landscapes into "highly simplified, disturbed, and nutrient-rich" systems (Tilman, 1999, p. 5995). Modern agricultural systems apply external inputs to control crops, soil fertility and pests (Tilman, 1999). Natural ecosystems that once contained thousands of plant and insect species, as well as many species of vertebrates, have been replaced by monocultures (one crop or species) (Tilman, 1999). Furthermore, adding exotic species into the local species pool does not assist global, or even national, biodiversity if the abundance of indigenous species is being reduced (Lee et al., 2008).

Since 1960, New Zealand agricultural landscapes have declined in structural complexity and area and diversity of indigenous vegetation (Moller et al., 2008). Lee, Meurk & Clarkson (2008) argue that agricultural intensification is causing indigenous species' habitat loss and homogenisation of landscapes. Decline of indigenous biodiversity is primarily caused by habitat loss. Lee et al. (2008) note that private land has incurred the greatest loss of indigenous habitats as landowners have developed these areas primarily for economic return.

In New Zealand, the amount of native habitat area within agricultural lands decreased from 53% (9 million ha) in 1950 to 8% (1.3 million ha) in 2002 (Moller et al., 2008). Most of New Zealand's accessible and productive land has been cleared or modified for a range of uses. In New Zealand, almost 2,500 land-based and freshwater species are classified as threatened, while about 3,300 species are data-deficient, meaning there is not enough data on them to determine if they are threatened (Ministry for the Environment, 2007a). Many of these species could be threatened and consequently no management plans are being put in place for recovery due to their data deficient status.

Only 10% of New Zealand's original wetlands remain, covering less than 2,500 km^2 (Ministry for the Environment, 2007a). Lowland areas where wetlands have been drained for agricultural land have incurred the greatest losses (Campbell et al., 2003) perhaps due to the natural predominance of wetlands in lowland areas. A key risk of wetlands in Southland has been identified as dairy farming expansion causing nutrient enrichment, land clearance and stock trampling (Campbell et al., 2003).

6.2. Soil Quality Parameters

Soil assessments in New Zealand have measured soil quality parameters to determine the status of soil. Organic matter, fertility, acidity and physical condition are often used to determine soil quality. Nationwide soil assessments found around 70% of monitored sites did not meet at least one of the soil quality targets (Sparling & Schipper, 2004). Sixty-six per cent of land not meeting the soil quality target range was under pasture, accounting for nearly 20,000 km^2 of the pasture land measured (20% of the pasture land measured) (Sparling & Schipper, 2004). Sampling in the Waikato region in 2009 revealed over 80% of dairy pasture sites not meeting at least one soil quality target, and over 30% failing to meet two or more targets (Taylor, 2011). This has increased from earlier surveys of soil quality in the Waikato region from 1998 to 2004, showing 73% of sampled dairy sites not meeting national soil quality targets, a higher proportion than sheep and beef (64%), plantation forest (56%) and horticulture and cropping sites (46%) (Environment Waikato, 2008). Furthermore, 55% of surveyed Waikato dairy farms from 2003-2007 did not meet soil structure targets due to compaction (Environment Waikato, 2008).

The percentage of monitored Waikato dairy pasture sites not meeting soil quality targets (marcoporosity, and upper targets for Olsen P and total N) measured by Taylor (2011) are shown in Figure 6.2. Macroporosity measures the extent of compaction and Olsen P and total N measure phosphorus and nitrogen soil fertility respectively. The macroporosity target (-10 kPa) indicates the level at which lower production occurs. The upper Olsen P target assesses excessive phosphorus, while low Olsen P targets identify production limitations by phosphorus deficiency. All dairy sites met the low targets. Likewise, the upper total N target assesses excessive nitrogen, and the low total N target indicates nitrogen deficiency, causing production limitations. At all sites, there was no nitrogen shortages measured. Dairy and other pasture sites had the highest proportion of sites not meeting upper total N targets, indicating excessive nitrogen (Taylor, 2011).

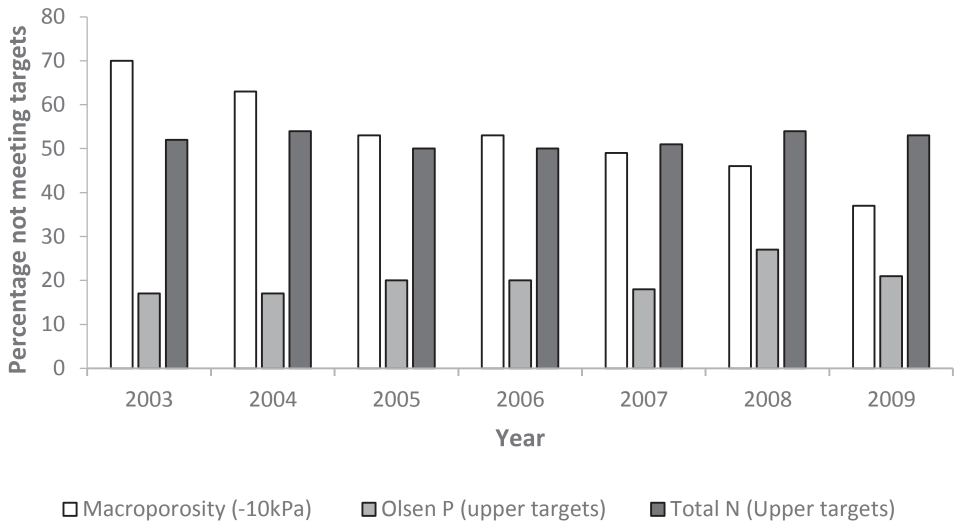


Figure 6.2: Percentage of monitored Waikato dairy soil quality sites not meeting soil quality targets from 2003-2009.
Data source: Taylor (2011). Data in Appendix H.

6.3. Soil Compaction

Soil compaction has been identified as a significant issue due to the large area of land affected and associated potential effects (Taylor, 2011). When soil cannot support the weight forced upon it, compaction will occur (Ledgard et al., 1996). Compaction intensifies when soils are wetter, at higher stocking rates, and when animals are grazed during long winter rotations (Ledgard et al., 1996; Mackay, 2008; Pande, 2002; Russell et al., 2001; Sparling & Schipper, 2004). Winter grazing commonly involves confining dairy herds to small areas with stocking rates of between 250-300 ha/day (Menneer et al., 2001), leading to serious compaction effects.

Effects of compaction

Compaction reduces plant cover, exposes soils and affects soil physical properties (Ledgard et al., 1996; Nguyen et al., 1998; Pande, 2002). Soil properties are affected by a reduction in the amount of macropores (air pockets) in soil, resulting in reduced aeration and drainage (Mackay, 2008). This is because soil bulk density is increased and infiltration rates reduced (Ledgard et al., 2009; Nguyen et al., 1998; Taylor, 2011). The resulting reduction in water storage (Drewry, 2006; Russell et al., 2001) can lead to increased runoff into surface waters, soil erosion (Ledgard et al., 1996; Nguyen et al., 1998; Pande, 2002), and surface ponding of water on land (Mackay, 2008). Consequently, these effects can cause flooding and sedimentation on both land and in waterways (Taylor, 2011). Furthermore, damaged soil structure may restrict root growth and nutrient uptake by plants, negatively affecting plant productivity (Ledgard et al., 2009; Ledgard et al., 1996; Mackay, 2008; Menneer et al., 2001). Soil physical characteristics are also important for biological activity in soil. Lower biological activity has been found on conventional dairy farms compared with organic dairy farms, linked to poor physical conditions caused by soil compaction (Schon et al., 2012). Another problem associated with soil compaction is the risk of increased greenhouse gas emissions from soils (Ledgard et al., 1996; Mackay, 2008).

Compaction on dairy land

Compaction intensity is measured by soil macroporosity (the volume of large pores within soil). Land under dairying has been the worst affected by compaction in New Zealand. For example, on half the dairying sites they tested In New Zealand, Sparling and Schipper (2004) found a macroporosity of less than 10%: rates at which pasture production can be adversely affected. Likewise, data reported by the Ministry of the Environment (2011d) showed that 53% of dairy sites failed to meet macroporosity targets, while Taylor (2011) reported that 37% of tested sites on dairy land did not meet macroporosity targets in 2009, decreasing from 70% in 2003 (Figure 6.2).

Dairy cow treading causing compaction has been reported to reduce pasture production by 20-80% (Ledgard et al., 1996; Mackay, 2008; Menneer et al., 2001; Pande, 2002), due to the reduction in infiltration. Soil type influences infiltration rates, so pasture production losses will depend on soil type (Ledgard et al., 1996; Thorrold, 2000). For every 1% unit decrease in soil macro-porosity, pasture yields have been predicted to decrease by 1-2% (Clark et al., 2007). Menneer et al. (2001) found pugging effects from dairy cows over 10 years lowered milk production by 21% and 54% for moderate and severe pugging, respectively. They predicted that moderate and severe pugging on 50% and 10% of the farm, respectively, would decrease milk production by 16% (e.g. from 1000 to 840 kg milksolids (MS) ha/yr) (Menneer et al., 2001).

Compaction affects soil infiltration, and thus soil drainage. Ledgard et al. (1996) measured infiltration rates that were ten times slower on compacted soils (compared to non-compacted), due a reduction in small soil pores (<0.3 mm diameter), reducing aeration and water storage (Ledgard et al., 1996). On steep zones, Nguyen et al. (1998) found compaction from cattle treading lowered infiltration rates up to 46% and runoff from compacted areas contained on average 87% more sediment, 89% more nitrogen and 94% more phosphorus compared with undamaged areas. Menneer, Ledgard, McLay & Silvester (2005) reported that dairy cow treading caused a short-term increase in denitrification and a significant reduction in soil aeration. This led to reduced nitrogen utilisation by soil, thus an increase in N leaching potential. Severe pugging by dairy cattle has also been reported to decrease annual nitrogen (N_2) fixation by up to 70% from 151 kg N/ha to 45 kg N/ha (Menneer et al., 2001).

Solutions to compaction

Soils can naturally regenerate through drying and wetting cycles, and with root and earthworm activity. However, intensive management of stock and high stocking rates may slow down the natural regenerative processes so soils are not able to recover from compaction, resulting in soil compaction up to 20 cm below the soil surface (Greenwood and McNamara, 1992 cited in Ledgard et al., 1996). Recommended management options to reduce compaction include restricted grazing on saturated soils (Pande, 2002) by having shorter grazing periods and moving stock to a loafing pad, yard, race or sacrifice paddock until the soil is no longer wet; using adequate drainage; using wintering barns; and to avoid using heavy vehicles and machinery on wet soils (Ledgard et al., 1996; Thorrold, 2000).

6.4. Soil Fertility

Plants need nutrients to grow. The major nutrients required are nitrogen (N), phosphorus (P) and potassium (K). Nitrogen can be fixed from the atmosphere by plants but P must be added to increase production. For agricultural uses, New Zealand soils are most commonly deficient in P and sulphur (S) (Carey, 2006) and fertilisers have been used to overcome deficiencies. Phosphatic fertilisers have been used extensively in New Zealand, either as superphosphate or via direct application of reactive phosphate rock (explained in Chapter 2) (Longhurst et al., 2004).

Dairy farms use the most phosphate fertiliser per hectare, mostly in the form of superphosphate (Cadmium Working Group, 2008). About 87% of phosphate fertiliser used is superphosphate and application rates are typically 200-600 kg/ha/yr, with dairy farms at the upper end of this range. However, many farms exceed these rates; surveys in 1992 found that 22% of pastoral farms were applying more than 600 kg/ha/year (Cadmium Working Group, 2008). These rates could still be indicative of current application rates. Sparling and Schipper (2004) report that considerable N build-up and high levels of available P were found under dairy pastures, increasing the potential for leaching and runoff.

Heavy phosphate fertiliser use leads many farms to exceed the optimal limit of P of 30-40 mg P/L, despite evidence showing production is unlikely to increase (Carey, 2006). Carey (2006) reported that average phosphorus levels on conventional and organic dairy farms were 75% and 57% above the top optimal guideline respectively. Taylor (2011) found that nitrogen and phosphorus levels in agricultural land uses are increasing in the Waikato region and were higher on dairy pasture than other land uses. The upper Olsen P target was exceeded at approximately 20% of dairy sites and the upper total N targets were exceeded at about 50% of dairy sites (Figure 6.2) (Taylor, 2011). Increasing levels of nitrogen and phosphorus in soils is likely to be contributing to increasing nutrient levels in freshwater causing detrimental ecosystem effects (explained in Chapter 5). Nitrification inhibitors were used on some dairy farms to decrease nitrogen leaching, although, their use has led to other problems.

DCD

In 2012, the nitrification inhibitor dicyandiamide (DCD) was found in milk samples, leading to the ban of DCD for sale in New Zealand. Nitrification inhibitors had been used since 2004 by farmers for reducing nitrous oxide (N_2O) emissions and nitrate (NO_3^-) leaching mainly from urinary-N deposits (Beukes et al., 2012; Smith & Schallenberg, 2013). Nitrification inhibitors keep more N in a less mobile state (ammonium NH_4^+) for longer in the soil (de Klein et al., 2010). This allows it to be available for pasture growth and soil requires lower fertilisation rates (Beukes et al., 2012). DCD was only able to be used by about 500 (4.1%) of the country's 12,000 dairy farms (Federated Farmers, 2013b), as it requires certain soil properties to work effectively. Testing of dairy products for DCD was carried out in 2012 in nearly 2000 samples with DCD detected in 371 samples (Ministry for Primary Industries, 2013a, 2013b). Sampling was targeted where DCD was applied to land, focusing on product manufactured during, and shortly following, the DCD application period of 1 June 2012 to 28 September 2012. DCD was for sale for seven years before any testing for the chemical in milk products was carried out, so has potentially been in milk longer without being recognised. The industry say that the presence of DCD in milk "creates absolutely no food safety risk whatsoever" (Ministry for Primary Industries, 2013b). There is no internationally agreed acceptable level, but countries may halt imports with higher levels.

Furthermore, DCD commonly reached concentrations of 1000-3000 µg/L in leachate when applied as recommended, suggesting that DCD could have leached into drainage waters and downstream aquatic ecosystems (Smith & Schallenberg, 2013). Measureable concentrations of DCD were found in many surface waters in a lowland agricultural catchment where DCD had been used, with a maximum measured concentration of 946 µg/L (Smith & Schallenberg, 2013). An effect threshold in freshwaters has been suggested at concentrations of around 200-500 µg DCD/litre (Smith & Schallenberg, 2013). DCD is a problem in aquatic systems because it can block in situ nitrification and affect nitrogen cycling in these systems, resulting in elevated ammonium concentrations and reduced nitrate concentrations. This could cause ammonia toxicity, eutrophication and change algal community composition (Smith & Schallenberg, 2013).

6.5. Soil Contamination

Soil contamination is an issue rising from the increasing amount of inputs applied to land. Cadmium, uranium, copper and zinc have been found in significantly higher concentrations in soils under dairy pasture and other land uses than in soils under native and plantation forest (Taylor et al., 2011). High concentrations of cadmium, uranium and fluorine are consistent with phosphate fertiliser applications (Gray et al., 1999; Longhurst et al., 2004; Roberts et al., 1994; Taylor, 1997, 2007), while zinc is used for the treatment of facial eczema in cows. Cadmium is of particular concern as it is a toxic cumulative heavy metal and concentrations in soil are increasing on dairy land. More importantly, there is little awareness or management of the problem, thus leaving little hope for the problem to subdue. Although awareness has recently been increasing, there still appears to be little remediation.

6.5.1. Cadmium

Phosphate rock is used to make phosphate fertiliser and many sources contain high levels of cadmium. Cadmium naturally occurs at low concentrations in air, water and soils, but phosphate fertiliser contributes a more significant source (Roberts et al., 1994). The cadmium content of phosphate rock varies depending on the origin of the phosphate rock deposit and even within single deposits (Cadmium Working Group, 2008; Longhurst et al., 2004; Roberts et al., 1994). Historically, New Zealand obtained phosphate from the avian guano deposits of

Christmas and Nauru Islands which contained some of the highest cadmium concentrations in the world (Gray et al., 1999; Taylor, 1997), averaging about 450 mg Cd/kg P for Nauru phosphate rock (compared to present averages of 180 mg Cd/kg P) (Cadmium Working Group, 2008). Nauru phosphate rock was relatively inexpensive compared to phosphate with less cadmium content.

In 1995, voluntary limits for cadmium in fertilisers were put in place in the fertiliser industry. In the early 1990s, New Zealand superphosphate contained an average of 550 mg Cd/kg P (48 mg Cd/kg fertiliser) (Roberts et al., 1994). When limits were put in place in the mid-1990s, they were initially 340 mg Cd/kg P (29.7 mg Cd/kg fertiliser), dropping to an upper limit of 280 mg Cd/kg P (24.4 mg/kg fertiliser) from 1997 onwards. Phosphate now averages about 180 mg Cd/kg P (15.71 mg/kg fertiliser). Currently, there is no cost-effective or practical way to remove cadmium from phosphate rock. Phosphate rock containing low cadmium content is either unavailable or difficult and more expensive to source. Once limits were in place, the source of phosphate rock switched mainly to China, Morocco and Togo. Morocco is now the dominant source of phosphate rock used in New Zealand as other sources have become unavailable to New Zealand. For example, China imposed limits on exports of phosphate rock (Cadmium Working Group, 2008), as well as the United States.

Cadmium addition to New Zealand soil

Soil samples show that cadmium levels in New Zealand's soils are increasing which poses a risk for food products grown on soils with elevated cadmium levels (Cadmium Working Group, 2008). Excessive cadmium levels in food can have implications for human health, access to markets and trade, and threaten the ability to change land uses (Cadmium Working Group, 2008). There are currently no national-level standards for the permissible amount of cadmium in agricultural or residential soils or for the discharge of cadmium onto soil. There is a guideline for cadmium concentration in agricultural soils of 1 mg/kg, but this is not legally binding unless councils wish to enforce it or include as a condition for resource consents (Cadmium Working Group, 2008)

Taylor et al. (2007) measured a national average level of cadmium of 0.35 mg/kg on tested sites, with a range of 0 - 2.52 mg/kg. The background level was 0.16 mg/kg. Regions with high average levels of cadmium were Taranaki (0.69), Waikato (0.55) and Bay of Plenty (0.53) (Figure 6.3). These regions have historically been large dairy farming areas with a higher use of phosphate fertiliser, coupled with soils that have a high tendency to accumulate cadmium

(Taylor et al., 2007). However, the authors note the results may actually underestimate the real situation and averaging cadmium levels does not give an accurate view of sites with high levels of cadmium. Cadmium accumulation rates are higher on dairy and horticultural land; however, rates are often averaged over all land uses. This gives a false representation of the actual cadmium levels on dairy land, thus, underestimating the cadmium problem.

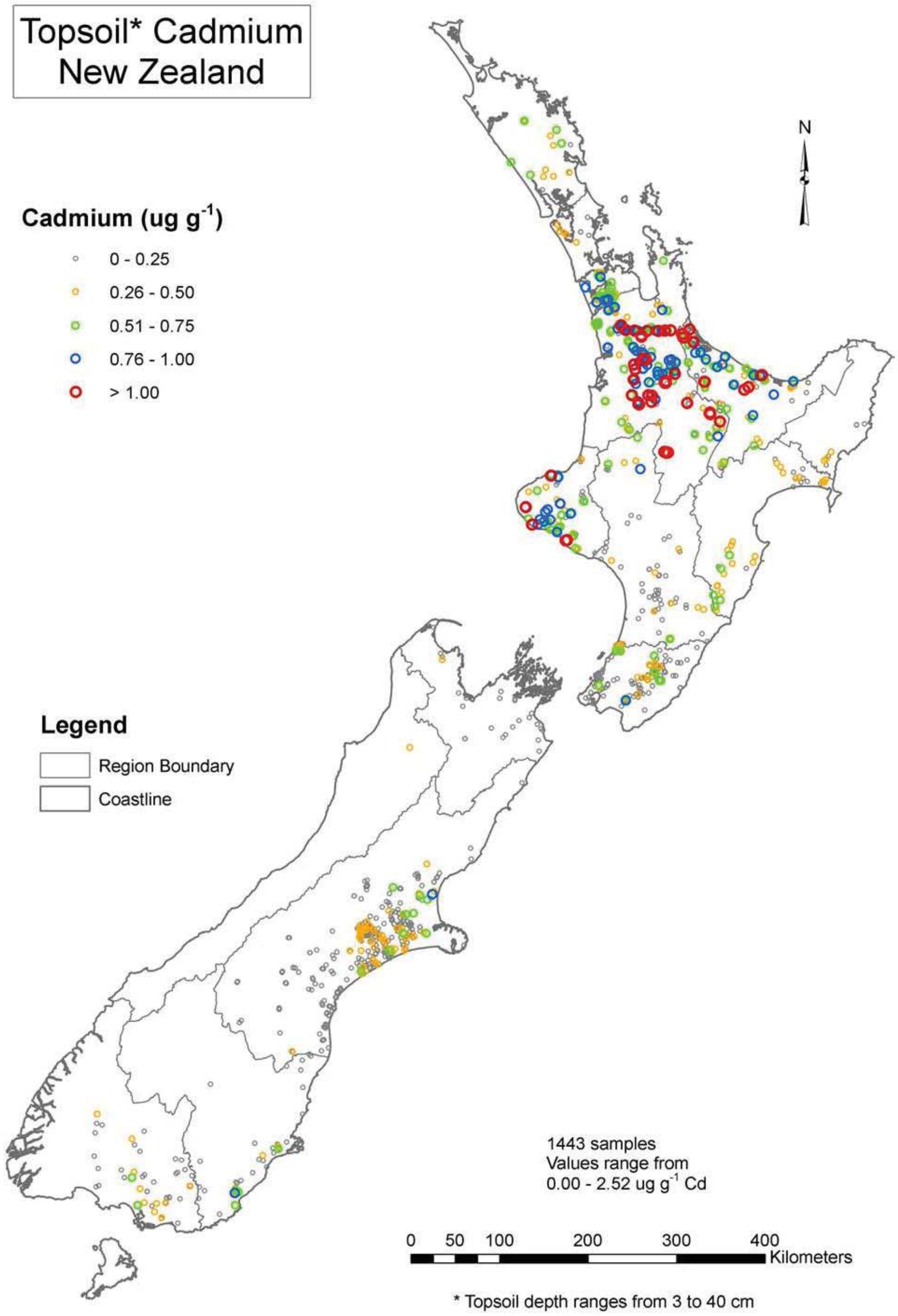


Figure 6.3: Topsoil cadmium levels measured in New Zealand.
Source: Taylor et al. (2007).

When land uses are broken down, topsoil sampling has shown higher cadmium concentrations on dairy land compared to other land uses (Figure 6.4) (Taylor et al., 2007; Taylor et al., 2011). Dairying land had the highest national average cadmium concentration (0.73 mg/kg) and the highest level measured of 2.52 mg/kg (Figure 6.4) (Taylor et al., 2007). Kim (2005) measured an average cadmium level on Waikato dairy farms of 0.83 mg/kg. Nationwide natural background levels were 0.16 mg/kg (Taylor et al., 2007), similar to that found by Roberts et al. (1994) for non-farmed soils (0.20 mg/kg), while average concentrations across agricultural land were 0.35 mg/kg (Taylor et al., 2007) (similar to 0.44 mg/kg for pastoral soils from Roberts et al. (1994)).

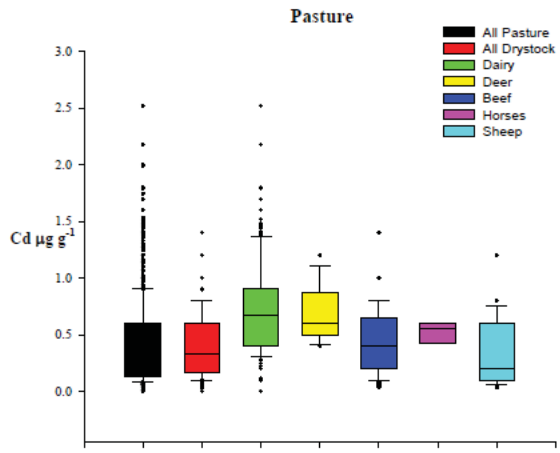


Figure 6.4: Boxplots of cadmium in pasture soils by farm type in New Zealand
Source: Taylor et al. (2007).

In the five years from 2001-2005, approximately 150 tonnes (30 tonnes annually) of cadmium was added to New Zealand's agricultural soils by phosphate fertiliser (Cadmium Working Group, 2008). This equates to approximately 2380 mg Cd/ha/year. The estimated average concentration increase in surface soils (0-7.5 cm) over this five year period would be 24 µg/kg, approximately 5 µg/kg/year assuming no leaching (Cadmium Working Group, 2008). However, Kim (2005) has estimated higher cadmium loadings on New Zealand soils of 48 tonnes per year with accumulation rates on Waikato dairy land estimated at 14.5 µg/kg/year. Cadmium concentrations have been increasing in the Waikato since the 1940s (Environment Waikato, 2008). Around 8.3 tonnes of cadmium is thought to be applied to Waikato soils each year, leading to Waikato's agricultural land having average cadmium levels five times higher than background levels (Kim, 2005). Additionally, the Canterbury and Manawatu-Whanganui regions are estimated to receive about 6.4 tonnes annually, followed by Southland (5.8 t/yr), Otago (4.3 t/yr), Hawkes Bay (3.5 t/yr) and Northland (3.3 t/yr) (Kim, 2005).

Half of the cadmium added to soils in the Waikato region is applied to dairy land, covering 25% of the region. Kim (2005) estimated that 11% of Waikato pastoral soils exceeded the recommended guideline of 1 mg/kg, all on dairy land and covering approximately 157,000 ha. If accumulation continued at rates measured by the surveys, all Waikato dairy land would reach guideline levels (1 mg/kg) by 2021, and productive pastoral land (comprising mainly of sheep, beef and dairy farms), would reach guideline rates by 2043 (Table 6.1). However, Kim (2005) noted that it is a conservative estimate and levels would be much higher on some dairy land. Taylor et al. (2007) reported that over 5% of samples in Taranaki, Waikato and Bay of Plenty have already exceeded 1 mg/kg of cadmium. Additional sampling estimated the proportion of tested dairy sites in the Waikato not meeting the cadmium recommended levels increased from 2007 to 2009 from 13% to 19%, respectively (Taylor, 2011). Preventing further cadmium accumulation in New Zealand soils was estimated to require an 80% reduction in the cadmium content of superphosphate, from 24.4 mg Cd/kg fertiliser to 4.8 mg/kg (Kim, 2005).

Impacts associated with soils exceeding 1 mg/kg cadmium are the inability to subdivide land without some form of assessment or remediation of cadmium levels, and possible hindrances to market access for products grown on contaminated land. The current agricultural soil guideline is set partly with reference to expectations of international trading partners and may take on a more significant trade role in the future (Kim, 2005). Additionally, food grown on land with excessive cadmium levels may be unsafe to consume.

Table 6.1: Cadmium loading and year to reach Cd soil guidelines on dairy and pastoral land in the Waikato region.

Land use	Land area (ha)	Proportion of Waikato's land area (%)	Cadmium loading (t/yr)	Proportion of Cd loading for total Waikato (%)	Year for land area to reach 1 mg Cd/kg
Dairy	623,000	25	4.3	51.4	2021
All pastoral land (includes dairy)	1,427,800	57	8.13	97.9	2043

Data source: (Kim, 2005)

Cadmium in animals

As heavy metals are not particularly mobile, they accumulate in agricultural soils and can be taken up by plants and passed on in the food chain (Longhurst et al., 2004; Taylor, 1997). Although heavy metal levels in soils are generally small, any contamination is potentially serious because heavy metals accumulate in plants and animals (Longhurst et al., 2004).

Cadmium contamination is an increasing problem in New Zealand; in the early 1990s the kidneys from 14-20% of sheep or cattle tested had a Cd concentration above the maximum allowable concentrations of 1 mg/kg fresh weight set by the New Zealand Department of Health (Roberts et al., 1994, cited in Condron et al., 2000; Marshall, 1993, cited in Mackay, 2008). Consequently, from 1991, kidneys from sheep and cattle over 2.5 years of age were banned from human consumption (Cadmium Working Group, 2011; Roberts et al., 1994).

Human health risk from cadmium

Cadmium is the one contaminant in the diet that comes closest to its Provisional Tolerable Weekly Intake (PTWI), reaching almost 50% for children and toddlers under 6 (Ministry of Agriculture and Forestry, 2011). The average non-smoking person absorbs over 90% of cadmium from food (Kim, 2005). Evidence suggests that the current food standard is routinely being exceeded in some New Zealand crops, particularly some varieties of wheat. In 2004, approximately 1.5% of potatoes purchased in the Waikato region likely exceeded standards (Kim, 2005). Acute cadmium poisoning (exposure to a high concentration over a short period) can cause death, while continued, low level exposure (chronic) leads to accumulation in the liver and kidneys and can also affect the lungs and bones (Kim, 2005). Cadmium levels in the body increase with age, from an estimated 1 µg at birth to between 15,000 and 80,000 µg at 50 years (Kim, 2005). Continued cadmium accumulation in soils is associated with increased food risks. Although not directly an issue for dairy products, if cadmium levels on dairy farms exceed standards, landowners may be unable to convert to other uses.

Risk to exports and land use

If food produced on soils that has accumulated cadmium breaches food safety standards, domestic and export sales of these food products could be compromised. Risks fall on mainly leafy vegetables and offal from animals (Cadmium Working Group, 2008). New Zealand's standards for cadmium in soil or fertiliser may also fall behind those of our trading partners, damaging our clean green reputation. Food grown for export must meet domestic and trading partner's food standards. Some countries may have lower regulatory thresholds for cadmium in food than New Zealand (Cadmium Working Group, 2008). Normally, compliance testing with standards is carried out at the port of entry. If standards are exceeded, products may not be accepted, resulting in an economic loss for New Zealand producers (Cadmium Working Group, 2008).

Accumulation of cadmium in agricultural land could affect the future ability to subdivide the land for residential purposes (Cadmium Working Group, 2008; Kim, 2005). Acceptable concentrations for one type of land may be too high for a more sensitive use. This is concerning as a substantial portion of residential housing development takes place over previous agricultural land. This problem would mostly affect land that had received on-going applications of phosphate fertiliser (e.g. dairy) that was close to the perimeter of an urban area (Cadmium Working Group, 2008). Subdividing the land may become an issue. Kim (2005) estimated that on-going cadmium accumulation may reduce the range of uses of approximately 58% of the total Waikato land area in the short-to-medium term (between 10-60 years depending on land use), covering pastoral agricultural, arable, cropping and horticulture. This could be further increased by land conversions from plantation forests to dairy farms (Kim, 2005).

High cadmium levels in soils affect the ability of landholders to grow certain types of crops if the cadmium levels in these products exceeded food standards or requirements set by overseas markets. This could affect landowners using phosphate fertiliser applications wanting to convert to growing a horticultural crop which is sensitive to soil cadmium levels. Sheep, beef or dairy farms may have significant accumulated cadmium in the soils to cause food standards to be exceeded in some leafy vegetables or grain crops grown after land use has changed (Cadmium Working Group, 2008). This issue may not be apparent until testing is carried out either in New Zealand or at an export destination. However, soil assessments are usually only requested on agricultural properties when a significant change is proposed for a property which requires regulatory oversight and possibly involves soil contamination issues. Many agricultural soils may actually exceed guidelines for cadmium but are not tested so the issue is overlooked (Cadmium Working Group, 2008).

6.5.2. Uranium

Uranium may be of greater concern than cadmium as there has been little research carried out regarding concentrations in New Zealand soils. Uranium is another contaminant derived from phosphate fertiliser. Phosphate rock from Christmas Island and Nauru is particularly high in uranium (U). Concentrations in these deposits ranged from 31-56 mg U/kg for Christmas Island and from 64-121 mg U/kg for Nauru (Taylor, 2007). Superphosphate measured in 1995 contained 8-37 mg U/kg (Taylor, 2007). Phosphate rock is now obtained from sources containing lower levels of Cd and U, albeit they are more expensive.

Uranium levels in soil samples collected in 1992 increased compared with samples 40 years previously (Taylor, 2007). Concentrations of total U ranged from 0.62 to 2.34 µg/g for the historical soil samples and from 1.69 to 3.54 µg/g for the 1992 samples (Taylor, 2007). The largest increase measured was from 0.79 to 2.48 µg/g (annual increase of 0.046 µg/g/yr), reflecting high inputs of superphosphate fertiliser (600 kg/ha/yr). Additionally, Sampling from 2007 to 2009 showed average uranium levels in soil on dairy land in the Waikato region were around 2-3 times higher than on native and forestry land (Taylor, 2011).

Risks of high uranium concentrations

Uranium can accumulate in the soil where it can be taken up by plants or it can be leached into ground and surface waters. Although overseas studies have found uranium levels in surface waters, linked to leaching from land, there has been little testing carried out in New Zealand. In one New Zealand study, Taylor (2007) noted that nearly all of the uranium applied appeared to have remained in the soil and had not leached to groundwater or taken up by plants. Similarly, other testing on pastures applied with phosphate fertiliser found most of the uranium applied remained in the surface soil (Schipper et al., 2011). Uranium can be taken up by plants but when ingested by humans, most of the uranium does not remain in the body (Taylor, 2007).

There is little information on safe levels of uranium in the soil, and hence no guidelines. Uranium is the only trace element with no target range in soil. This poses a problem because uranium is radioactive and classified as “very toxic” “causing skin, lung, intestinal and bone marrow disorders” (Schipper et al., 2011, p. 96). Uranium decays to radioactive products such as radon^{222} gas. Radon^{222} is considered a human carcinogen by the World Health Organization (WHO) and the US Department of Health as it causes lung cancer. All radon^{222} originates from uranium (Taylor et al., 2011). The greatest concern is thought to be with direct contact and inhalation of dust containing uranium during mining of phosphate rock and fertiliser manufacturing (Schipper et al., 2011).

6.5.3. Other contaminants

Sampling of soils in the Waikato region across a range of land uses showed all farmed sites had soil copper and zinc levels that were significantly higher than soils under native land and forestry (Taylor et al., 2011). This widespread distribution reflects the wide range of agricultural products these metals are used in. Copper is used as a livestock supplement and zinc is commonly fed to animals to prevent facial eczema. Remedies for facial eczema are widely used at annual zinc loadings of 5-7 kg/ha/yr and much of zinc is excreted by animals where it builds up in soils (Taylor et al., 2011). This could explain the significant increase in average zinc levels in Waikato pastoral soils from a background level of 28 mg/kg to a level of 62 mg/kg in farmed soils (Taylor et al., 2011). While not as toxic as cadmium to humans, zinc can pose environmental effects at lower levels in water bodies. As zinc builds up in the sediments of water bodies, it can have a toxic effect on sediment-dwelling animals and aquatic plants (Taylor et al., 2007).

Other chemical elements building up in soil include fluoride and arsenic. Soil assessments carried out in 2009 showed that 40% of monitored dairy sites failed to meet the soil fluoride target of 500 mg/kg (Taylor, 2011). Dairy pasture had the highest fluoride measurements but horticulture had the highest average (Taylor, 2011). Fluoride is found in superphosphate and di-ammonium phosphate in appreciable amounts (up to 3%). High concentrations of soil fluoride may restrict the versatility of land use (Taylor, 2011). Additionally, dairy land had the highest levels of arsenic among all land uses. Five per cent of measured dairy sites did not meet targets (20 mg/kg) for arsenic in soil in 2009, with dairy sites being the only sites above the target (Taylor, 2011).

6.6. Conclusion

Soil compaction and contamination from dairy farming practices could have severe consequences for future productively, land use change and agricultural exports. Compaction can seriously impede productivity and exacerbate other environmental issues associated with diffuse pollution. Future concentrations of cadmium and uranium will increase with ongoing phosphorus application, which may limit flexibility for other land uses and cause barriers for future agricultural exports. Already, cadmium contamination is affecting large areas of farmland in the Waikato region and is set to worsen in the next few decades. There has been no adequate management of these issues. Weak or no standards have been applied on soil contamination issues and often contamination is overlooked.

Chapter 7. ATMOSPHERE

What makes global warming so serious and so urgent is that the great Earth system, Gaia, is trapped in a vicious circle of positive feedback. Extra heat from any source, whether from greenhouse gases, the disappearance of Arctic ice or the Amazon forest, is amplified, and its effects are more than additive. It is almost as if we had lit a fire to keep warm, and failed to notice, as we piled on fuel, that the fire was out of control and the furniture had ignited. When that happens, little time is left to put out the fire before it consumes the house. Global warming, like a fire, is accelerating and almost no time is left to act.

(James Lovelock, 2004)

Atmospheric impacts from dairy farming result from the large volumes of greenhouse gas (GHG) emissions emitted. On a global scale, New Zealand's overall GHG emissions are miniscule; in 2005 New Zealand contributed approximately 0.2% to global emissions (Ministry for the Environment, 2012d; OECD, 2007). However, on a per-capita basis, New Zealand ranks among some of the highest emitting countries in the world. In 2005, New Zealand's GHG emissions, at 18.3 tonnes carbon dioxide equivalents (CO_2-e)[14] per person, were 13th highest in the world and 5th largest in the OECD (World Resources Institute, 2012). In 2010, New Zealand dropped to 24th highest in the world (at 18.72 tonnes CO_2-e per person), likely due to large scale development and environmental pollution occurring in many developing countries as New Zealand still ranked 5th in the OECD (World Resource Institute, 2013). Based only on carbon dioxide (CO_2) emissions however, with 8.2 tonnes CO_2 per person in 2005, New Zealand ranked 39th highest in the world (Ministry for the Environment, 2012d). This increased slightly to 8.61 tonnes CO_2 per person in 2010 at 38th highest (World Resource Institute, 2013).

New Zealand's GHG emissions are primarily made up of three gases: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Carbon dioxide is mostly derived from energy generation and transport whereas methane and nitrous oxide are mostly from agriculture. Most of New Zealand's GHG emissions are from the agriculture and energy sectors (Figure 7.1)

[14] CO_2-e = carbon dioxide equivalent is a measure used to compare the emissions from various greenhouse gases based on their global warming potential in the equivalent concentration of carbon dioxide. Most estimates of total GHG emissions are calculated in terms of CO_2-e.

and about a quarter of New Zealand's GHG emissions are from dairy farming alone. Consequences of rising GHG in New Zealand involve paying for emissions under international agreements, such as the Kyoto Protocol, which New Zealand has obligations under; paying forest credits back when production forests are cut down; or more importantly from the effects of global climate change and the threats it poses for the survival of future generations and biodiversity (Figure 7.2).

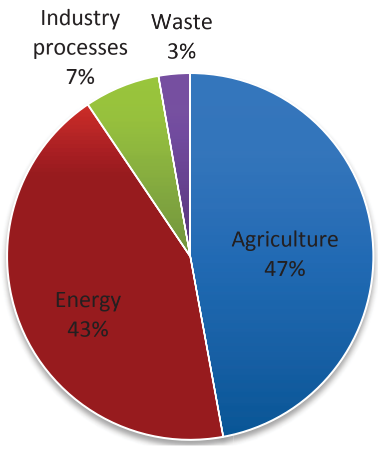


Figure 7.1: New Zealand greenhouse gas emissions by sector in 2010
Data source: Ministry for the Environment (2012c).

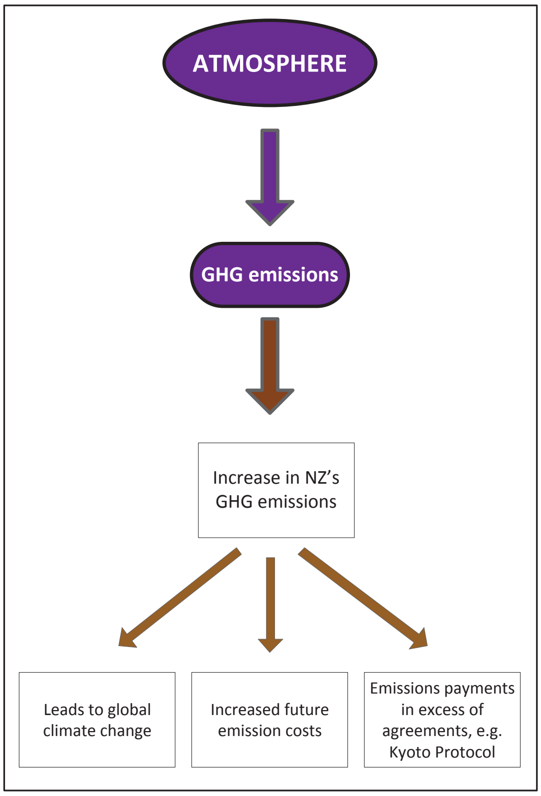


Figure 7.2: Impacts from New Zealand greenhouse gas emissions

Reporting of GHG emissions in New Zealand has not been consistent (Table 7.1). According to the Ministry for the Environment (2012d), emissions peaked in 2005. Despite this, total GHG emissions increased by about 20% from 1990 to 2010 (Ministry for the Environment, 2012c). The increase has been attributed to an increase in: road transport, methane emissions from dairy livestock, emissions from agricultural soils, and public electricity and heat production (Ministry for the Environment, 2012c). Terry (2007) estimated that over 40% of the growth above 1990 levels is expected to have come from agriculture, mainly resulting from dairy industry growth.

Table 7.1: Total reported greenhouse gas emissions in New Zealand

Source	Total emissions Mt CO_2-e			Percentage change	
	1990	2005	2010	1990-2005	1990-2010
Environment NZ (Ministry for the Environment, 2007a)	61.9	77.5	-	25.2	-
NZ's GHG Inventory (Ministry for the Environment, 2012d)	59.8	76.5	71.7	27.9	19.9

Note: Mt CO_2-e = million tonnes of carbon dioxide equivalents.

7.1. Agricultural Emissions

Agriculture is the largest source of GHG emissions in New Zealand. In 2010, agriculture contributed to 47% of total emissions (Figure 7.1), of which about half was from dairy farming. This is in contrast to most other developed countries where agricultural emissions make up about 10% or less of total emissions. Globally, agriculture has been estimated to account for between 10% (Eckard et al., 2010) and 35% of total GHG emissions (Monteny et al., 2006). From 1990 to 2005 global agricultural emissions increased by 17% (Intergovernmental Panel on Climate Change, 2007), while New Zealand's increased by 21%.

New Zealand's agricultural emissions peaked in 2005 (37.4 Mt CO_2-e) (Figure 7.3) (Ministry for the Environment, 2007a). This was followed by a steep drop, but from 2008 emissions started rising again. It is unknown what caused this large drop in emissions over such a short period of time. Emissions were projected to increase to 35.3 Mt CO_2-e in 2012 (Ministry for the Environment, 2012d). According to the Ministry for the Environment (2012c), the increase in agricultural emissions in 2010 was due to increases in dairy cattle numbers as well as a favourable milk price. Nitrogen fertiliser sales also increased (of which the main user is the dairy industry) (Ministry for the Environment, 2012c), contributing to increased nitrous oxide emissions. Between 1990 and 2012 reported agricultural emissions were only available for a few years and projections were made for 2011 and 2012 (Figure 7.3).

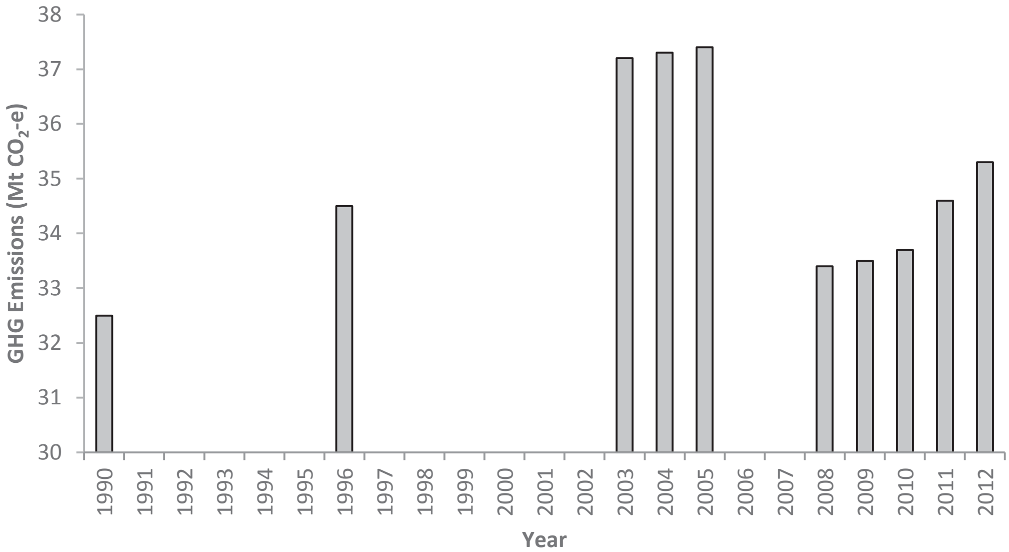


Figure 7.3: New Zealand agricultural GHG emissions – 1990-2012
Data sources: Ministry for the Environment (2007a, 2012c) and OECD (2007).

The importance of agricultural greenhouse gas emissions has been largely downplayed. Agricultural emissions are particularly important in New Zealand where they make up such a large proportion of the emissions profile. Methane and nitrous oxide are the two main agricultural GHGs, with agriculture contributing 96% of total nitrous oxide emissions and 91% of methane emissions in New Zealand (Ministry for the Environment, 2007a). Of the total GHG emissions in 2010, methane accounted for 37.5% and nitrous oxide for 15% (Ministry for the Environment, 2012c). Methane and nitrous oxide are much more potent than carbon dioxide. In relation to warming potential, methane is 21 times more damaging than carbon dioxide (1 kg of CH_4 has 21 times as much warming potential as 1 kg of CO_2 over a period of 100 years) and nitrous oxide is 310 times stronger (Ministry of Agriculture and Forestry, 2010b). Although methane and nitrous oxide have a stronger short term impact in the atmosphere, they do not persist as long as carbon dioxide, which can last for thousands of years in the atmosphere (Ministry for the Environment, 2012d). Nitrous oxide can persist in the atmosphere for 120 years (Li, 2005) while methane has a much shorter span (10-12 years). Hence, reducing methane emissions may have a quicker effect on mitigating climate change than focusing mainly on carbon dioxide.

Methane is produced by the digestive processes (enteric fermentation) of ruminant animals such as sheep, cows, goats and deer, and also from animal waste. Nitrous oxide is mainly produced from dung, urine and excessive nitrogen fertiliser application to soil (Ministry of Agriculture and Forestry, 2010b; Pinares-Patino et al., 2009). Enteric fermentation accounts for about 64% of agricultural emissions, while emissions from agricultural soils (mainly from fertiliser) contribute 34%. Almost 60% of enteric fermentation emissions are from cattle (Ministry for the Environment, 2007a).

7.2. New Zealand Dairy Farming Emissions

According to the UN Food and Agriculture Organisation (FAO), dairy farming produces 4% of the world's total GHG emissions (Levitt, 2010). Dairy farming is particularly emissions intensive due to the requirement of many inputs, such as fertiliser, agriculture-chemicals and supplementary feeds, which all produce significant emissions in their production. Although emissions from New Zealand dairy farms are minuscule globally, they make up a large proportion of New Zealand's emissions. Usually the estimates of GHG emissions are grouped together when reporting for all agricultural land uses. This does not give an accurate picture of different land uses within agriculture as they have different relative contributions.

The most recent estimates of animal emissions from agriculture are reported by the Ministry of Agriculture and Forestry (2010a) where emissions have been broken down for different land use categories (e.g. dairy, sheep/beef). For dairy emissions, estimates were calculated for the population of only milking cows and not all dairy cows. In 2012, the milking population was about 4.6 million (LIC & DairyNZ, 2012), compared to a total dairy population of 6.5 million (Bascand, 2013). Thus, official calculations for emissions from dairy cows do not count about 30% of emissions.

7.2.1. Carbon dioxide emissions from energy use

There has been little information complied on the energy use from agriculture and on the resulting carbon dioxide emissions. The most thorough study on energy used on dairy farms was conducted between 1997 and 1999 by Wells (2001). In this assessment, all forms of energy required for farm operation were estimated, including direct energy (fuel and electricity), indirect energy (for the production of inputs such as fertiliser and supplementary feeds), and capital energy (for the manufacture of vehicles and buildings). Although these are the best estimates available for energy use on dairy farms, they rely on information of inputs and production over a decade ago (when production was less intensive), and hence would likely under-estimate emissions.

Energy use was obviously higher on irrigated farms. A survey of 150 dairy farms throughout New Zealand revealed that the gross CO_2 emission intensity from energy use was about 1.1 and 2.0 tonnes CO_2/effective milking hectare/year for non-irrigated farms and irrigated farms respectively, or 1.4 tonnes and 1.8 tonnes CO_2/tonne MS/year for non-irrigated farms and irrigated farms respectively (Wells, 2001). The survey revealed regional differences in energy emissions with average energy intensities significantly higher in Canterbury due to the high use of pumped irrigation (Wells, 2001). The most significant contributions to energy requirements on non-irrigated farms was fertiliser (38%) followed by fuel (21%) and electricity (20%) while irrigated dairy farms used more energy on electricity (40%), followed by fertilisers (34%) (Figure 7.4) (Wells, 2001). Saunders and Barber (2007) extended the survey completed by Wells (2001) to include emissions associated with methane and nitrous oxide and used updated energy requirements for farms using the same survey data. The total CO_2 emissions calculated for dairy production was 1.25 tonnes CO_2/tonne MS (Saunders & Barber, 2007), close to Well's (2001) prediction of 1.4 tonnes CO_2/tonne MS.

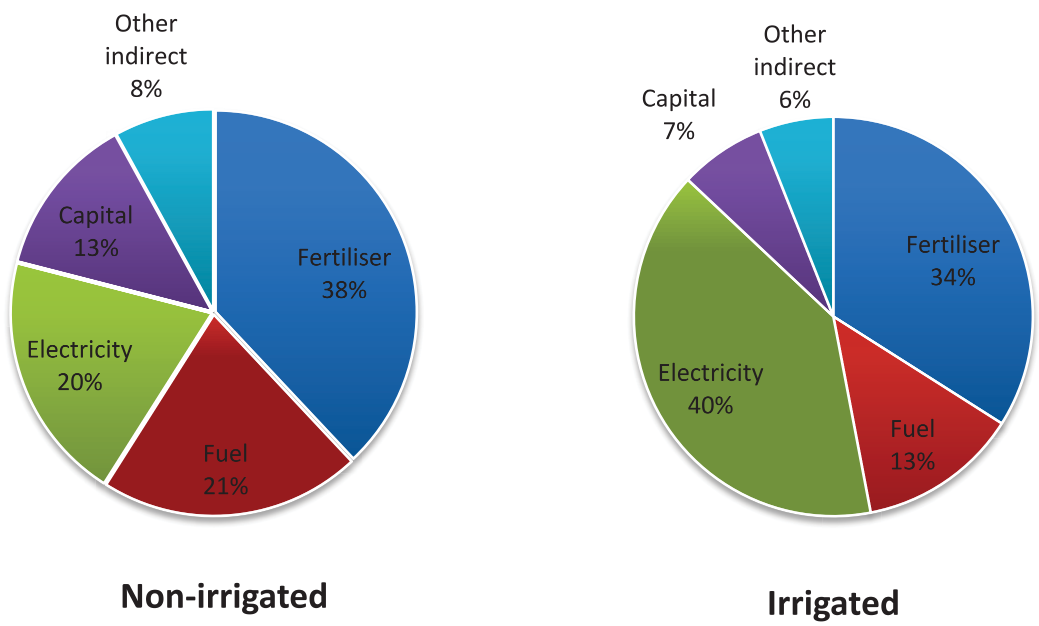


Figure 7.4: Proportion of energy inputs on the average non-irrigated and irrigated dairy farm

Using the energy coefficients from Wells (2001) and Saunders and Barber (2007) the total energy related emissions attributed to farm inputs and production (excluding emissions from animals) were calculated for milksolids processed from the first commitment period of the Kyoto Protocol (2008-2012)[15], and in 1990 for a comparison (Table 7.2). Not surprisingly, the two energy coefficients yielded similar emission estimates; using the Wells coefficient, emissions averaged 2.0 Mt CO_2 per year over the five years while Saunders and Barber's estimate resulted in average emissions of 1.82 Mt CO_2 per year.

Table 7.2: Total energy emissions from dairy farms by milksolids processed

Year to June	MS processed (million tonnes)	Total energy emissions Mt CO_2	
		1.4 CO_2/tonne MS	1.25 CO_2/tonne MS
1990	0.572	0.801	0.713
2008	1.270	1.778	1.583
2009	1.396	1.954	1.740
2010	1.438	2.013	1.792
2011	1.513	2.118	1.886
2012	1.685	2.359	2.100
Total CP1		10.223	9.101

Notes: CP1 = Totals for the first commitment period for the Kyoto Protocol (2008-2012).

7.2.2. Emissions from dairy cows

Enteric methane and soil nitrous oxide are the main agricultural GHG sources. Other sources include those from animal waste in ponds and dairy sheds. The estimated GHG emissions from dairy and proportion of dairy's emissions to agriculture are presented in Table 7.3. Emissions from dairy cows have been reported by the Ministry of Agriculture and Forestry (2010a).

Enteric methane emissions

Methane emissions from enteric fermentation occur during the digestion of feed in ruminant animals. Most of the methane is belched by animals and the more an animal eats the more emissions it produces. Different animals also emit different amounts of methane from enteric fermentation. For example, dairy cows emit higher amounts of enteric methane than other farmed animals and enteric dairy cow emissions per cow have been increasing in the last two

[15] The first Kyoto commitment period is the period to which New Zealad was signed to the Kyoto Protocol, requiring a reduction of emissions below 1990 levels. If reductions were not made, New Zealand would have to pay for its excess emissions.

decades. As a comparison, methane emissions from dairy cows were estimated at 81.4 CH_4/head/year in 2012 (increased from 70.8 kg CH_4 in 1990), while those for beef cattle, sheep and deer were 59.3, 11.4 and 22.8 CH_4/head/year respectively in 2012 (Ministry of Agriculture and Forestry, 2010a). In 1990, emissions from enteric fermentation from dairy farms totalled 5.03 Mt CO_2-e (23% of the total agricultural enteric methane emissions). These emissions increased to 10.77 Mt CO_2-e in 2012 (44% of the total) (Table 7.3).

Table 7.3: Estimated animal emissions from dairy farming and total agricultural emissions in 1990 and 2008-2012.

Year	Summary of dairy emissions projections (Mt CO_2-e)					Dairy proportion of total ag (%)
	Enteric	Animal waste	Soils	Total dairy emissions	Total agriculture emissions	
1990	5.03	0.21	2.44	7.68	31.9	24.1
2008	9.02	0.39	5.08	14.49	34.5	42.0
2009	9.47	0.41	5.08	14.96	34.5	43.4
2010	10.11	0.44	5.15	15.70	35.5	44.2
2011	10.51	0.46	5.31	16.28	36.2	45.0
2012	10.77	0.46	5.61	16.84	36.9	45.6
Total CP1	49.88	2.16	26.23	78.27	177.6	

Emissions from animal waste

Emissions from animal waste include methane from material deposited on pasture, and methane and nitrous oxide emissions from animal faecal material collected and treated in waste management systems (e.g. pond systems). Estimations of GHG emissions from livestock effluent have ranged from 2% (Ministry for the Environment, 2012c) to 10% of total agricultural emissions (Chung et al., 2013). Methane emissions from dairy cattle waste increased from 2.9 kg CH_4/cow/year in 1990 to 3.5 kg CH_4/cow/year in 2012. In 2012, methane emissions from dairy manure were 0.46 Mt CO_2-e (Table 7.3), (56% of total agricultural manure methane emissions) (Ministry of Agriculture and Forestry, 2010a).

The New Zealand GHG Inventory estimated that emissions from anaerobic ponds on dairy farms were 0.315 Mt CO_2-e in 2009, representing approximately 80% of total dairy manure CH_4 emissions (0.405 Mt CO_2-e). The remaining 20% is from manure deposited on pastures (Chung et al., 2013). However, only about 5% of the excreta from dairy cows are stored in anaerobic ponds, the remainder is deposited directly on pasture. These proportions relate to the time dairy cattle spend on pasture compared with the milking shed (Ministry for the Environment, 2012c). Hence, emissions reported in Table 7.3 from animal waste are probably under-estimated in light of recent research by Chung et al. (2013).

Accordingly, Chung et al. (2013) argue that the methods used in the New Zealand GHG Inventory for estimating emissions from anaerobic ponds are inaccurate. Data from only one farm was used to quantify emissions and the study used for this data concluded that "further work is required to assess the general applicability of the method to anaerobic ponds" (McGrath & Mason, 2004, p. 471); regardless it was still used for the national New Zealand GHG Inventory. Wastes that enter anaerobic ponds in addition to milking shed manure, such as waste milk, feed residues, and manure from feed and stand-off pads were not included in the Inventory. Recalculations by Chung et al. (2013) of methane emissions from dairy shed waste in anaerobic ponds ranged between 0.579 Mt CO_2-e and 0.918 Mt CO_2-e, about 2-3 times higher than the estimate of 0.315 Mt CO_2-e used in the Inventory.

Thus, it appears that Inventory is underestimating methane emissions from anaerobic ponds. This is a significant problem considering that manure collected in milking sheds is not the only source of dairy waste, but it is the only one used by the NZ Inventory. When additional sources of methane in anaerobic ponds are added in from manure from feed and stand-off pads, waste milk, and supplementary feed waste, emissions increase to between 1.029 and 1.744 Mt CO_2-e annually, about three to six times higher than currently reported (Chung et al., 2013). This is not just a problem in New Zealand, but other countries could be under-reporting as well.

Nitrous oxide from agricultural soils

This category covers N_2O emissions derived from animal nitrogen outputs (excreted urine and dung), synthetic nitrogen fertiliser use and crop residues. Animal nitrogen output is a function of animal feed intake and the nitrogen content of the diet minus any nitrogen stored in animal products (e.g. meat, milk etc.). Emissions from N-fertiliser are affected by the rate and timing of application, with emissions increasing exponentially with the amount of N applied (Pinares-Patino et al., 2009).

The Ministry for Agriculture and Forestry (2010a) did not calculate fertiliser emissions by land use so emissions from fertiliser use on dairy farms have been estimated. The main nitrogen fertiliser used in New Zealand is urea and 65% is used on dairy farms. Therefore, as a rough estimate, 65% of the emissions from fertiliser use were attributed to dairy. Emissions in this category have been grouped under soils in Table 7.3.

Total dairy emissions

The estimated animal and soil emissions from dairy farming in 2012 were 16.84 Mt CO_2-e (46% of total agricultural emissions) (Table 7.3). Including energy emissions, estimated GHG emissions from dairy farms in 2012 were 19.20 Mt CO^2-e. Dairy emissions have more than doubled from 1990 to 2012 (Figure 7.5). Dairy emissions (excluding energy emissions) as a proportion of agricultural and total New Zealand emissions are shown in Figure 7.6 and total dairy, agricultural and New Zealand's emissions are shown in Figure 7.7.

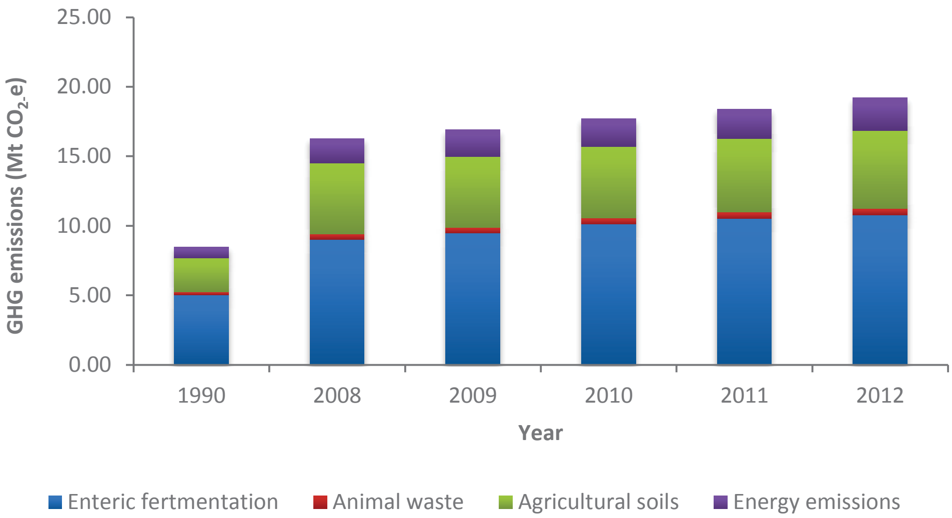


Figure 7.5: Total estimated GHG emissions from dairy farms in 1990 and 2008-2012 by different categories of emissions.

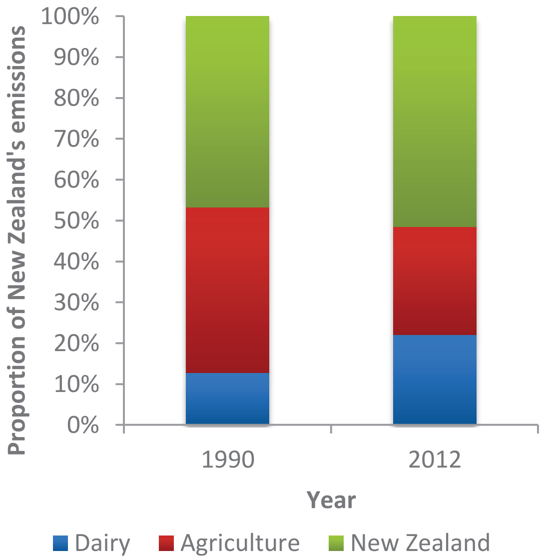


Figure 7.6: The proportion of GHG emissions from dairy (blue), and all agriculture including dairy (red), out of New Zealand's total GHG emissions (green) in 1990 and 2012.
Note: Energy emissions on dairy farms are not included in the dairy or agriculture total.

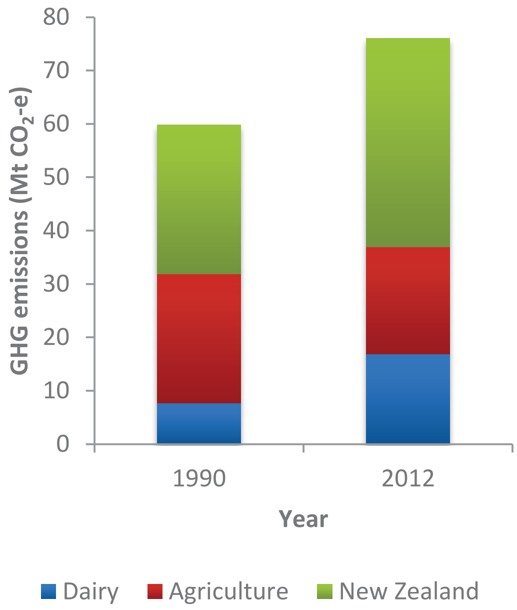


Figure 7.7: Total GHG emissions (CO_2-e) from dairy farming, all of agriculture and New Zealand in 1990 and 2012.

7.3. Conclusion

Methane and nitrous oxide are the main agricultural GHG emissions and are much more powerful in terms of atmospheric warming than carbon dioxide. Additionally these gases make up nearly half of New Zealand's GHG emissions profile. Despite this, agricultural emissions are left out of emission reduction policies. Dairy farming produces about a quarter of New Zealand's GHG emissions but the industry does not have to pay for their emissions, while other industries, such as energy and transport, do pay under the Emissions Trading Scheme. GHG emissions from dairy are rising due to intensification while emissions from the rest of agricultural are relatively stagnant or decreasing. Due to the serious and harmful consequences of climate change, New Zealand should be taking greater leadership in aiming to reduce its per capita GHG emissions.

Chapter 8. Offshore Impacts

Nature shrinks as capital grows. The growth of the market cannot solve the very crisis it creates.

(Vandana Shiva - Soil Not Oil: Environmental Justice in an Age of Climate Crisis)

As discussed in Chapter 2, New Zealand dairy farming imports products to intensify production. These products have their own environmental implications at the source of manufacture or production, of which are not considered in the New Zealand dairy farming system (Figure 8.1). These environmental impacts also carry economic costs, although they are not considered here. Only the imported cost to New Zealand is included. Environmental impacts discussed in this section are related to imported fertilisers, mainly rock phosphate, and imported palm kernel as a feed supplement.

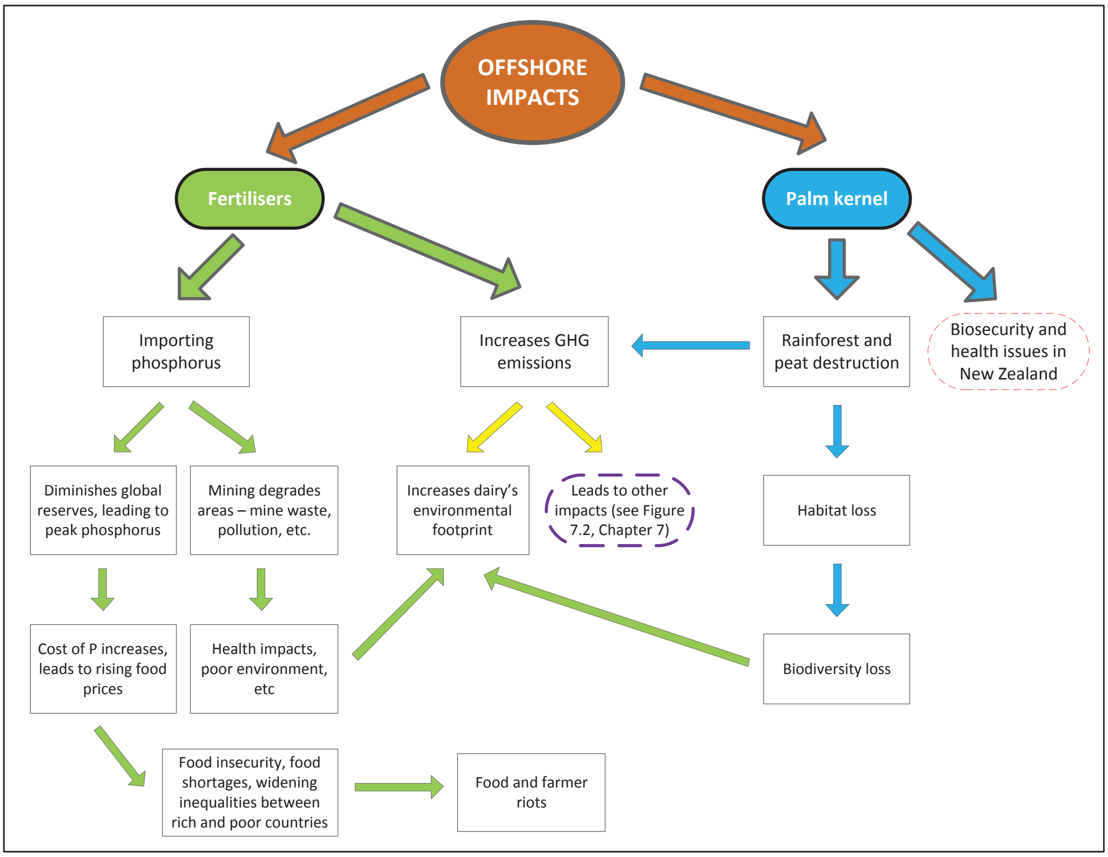


Figure 8.1: Some of the offshore impacts of imported fertiliser and palm kernel for use in the New Zealand dairy industry.
Notes: Biosecurity and health issues (red dashed line) are experienced in New Zealand and not offshore.

8.1. Fertilisers

The only conventionally used fertilisers New Zealand manufactures domestically are urea and superphosphate. Urea is made from New Zealand natural gas and superphosphate is made from imported ingredients of phosphorus, sulphur and calcium. Some of the environmental impacts associated with imported fertilisers and ingredients include GHG emissions, land degradation from mining, and impacts associated with depleting non-renewable resources.

Greenhouse gas emissions

While the GHG emissions associated with using fertiliser in New Zealand are included in New Zealand's GHG Inventory, emissions associated with manufacturing fertilisers offshore, mining, and transport to New Zealand, are not included. Nor are the emissions associated with producing fertilisers domestically included in agricultural farm emissions. Fertilisers and lime were estimated to contribute about 15% of farm-related total GHG emissions or over 50% of farm-related CO_2 emissions on dairy farms (Ledgard et al., 2011). Proportions of GHG emissions from N fertiliser, non-N fertilisers (P, K, S) and lime from the average New Zealand dairy farm were estimated to contribute 11.6%, 2.1%, and 1%, respectively to overall emissions (Ledgard et al., 2011).

As part of the energy emissions estimated in Chapter 7, fertiliser manufacture and transport were included but were based on fertiliser use estimates nearly 15 years ago (Wells, 2001). However, emissions attributed solely to fertilisers are not known as they were incorporated into the energy emissions as a whole. Fertiliser emissions have already been included in Chapter 7 under energy emissions; therefore, they will not be incorporated into the emissions total but will be estimated for comparison with New Zealand emissions. Ledgard et al. (2011) provides a summary of GHG emissions for various fertilisers from overseas production and shipping to New Zealand (Table 8.1). Using data of the quantity of fertilisers imported into New Zealand and estimations of how much dairy uses, total GHG emissions were estimated for some major fertiliser types used on dairy farms (Table 8.2). The last five years have been estimated as well as a comparison with 1990. Emissions from manufacturing of imported fertiliser have increased by 354% from 1990 to 2012 (Table 8.2). Using these calculations, GHG emissions from imported fertilisers in 2012 represent about 3% of total dairy farm emissions, much less than the 15% estimated by Ledgard et al. (2011).

Table 8.1: Summary of GHG emissions (kg CO_2-e/kg fertiliser) for various imported fertilisers.

Emission type	Fertiliser type					
	Urea (middle East)	Urea (China)	Ammonium sulphate	TSP	DAP	KCl
Production	0.732	2.140	0.473	0.350	0.910	0.365
Local transport and shipping	0.201	1.130	0.135	0.246	0.206	0.218
Total	**0.933**	**2.270**	**0.607**	**0.596**	**1.117**	**0.583**

Data source: Ledgard et al. (2011). **Notes:** Abbreviations: TSP = triple superphosphate; DAP = diammonium phosphate; KC1 = muriate of potash (potassium chloride). For more details on emissions see Ledgard et al. (2011). Emissions for local transport for some fertilisers were zero due to some fertilisers being produced close to ports and some estimates included emissions from transport with production.

Nitrogen fertiliser is the largest contributor to GHG emissions out of all the fertiliser nutrient types (Ledgard et al., 2011). While New Zealand urea is produced using natural gas, urea from China is produced using 80% coal and 20% natural gas; therefore, it has about three times the GHG emissions per kg produced compared to urea produced entirely from natural gas (Ledgard et al., 2011). However, 80% of imported urea into New Zealand is from Saudi Arabia and Qatar with less than 5% coming from China (Statistics New Zealand, 2013b), so urea emissions were based on these proportions. Sixty-eight per cent of the total GHG emissions associated with manufacturing superphosphate in New Zealand were produced from shipping raw materials to New Zealand, while 22% was from mining phosphate rock (Ledgard et al., 2011).

Table 8.2: GHG emissions from imported fertilisers for dairy farming in New Zealand for 1990 and 2008-2012.

Year	GHG emissions Mt CO_2-e by fertiliser type						
	Urea	Ammonium sulphate	TSP	Phosphate rock	DAP	KCl	Total
1990	0.000	0.014	0.000	0.049	0.004	0.059	**0.125**
2008	0.301	0.027	0.002	0.069	0.028	0.078	**0.506**
2009	0.225	0.029	0.000	0.041	0.069	0.037	**0.401**
2010	0.285	0.022	0.003	0.045	0.069	0.060	**0.485**
2011	0.412	0.048	0.003	0.054	0.062	0.065	**0.645**
2012	0.378	0.026	0.003	0.042	0.063	0.058	**0.569**

Notes: Estimations for urea imports were based on 95% of the urea coming from places with production similar to the Middle East and 5% from China. See fertiliser type abbreviations in Table 8.1.

8.1.1. Phosphorus impacts

New Zealand imports rock phosphate for production of superphosphate fertiliser. In the 1990s, about 30-40% of the rock phosphate imported into New Zealand was from Nauru (Statistics New Zealand, 2013b), but these were stopped as most of Nauru's reserves were depleted after decades of mining (Pearce, 2011). Other large imports of rock phosphate came from the United States, Morocco and Israel (Statistics New Zealand, 2013b). In the late 1990s, imports from Morocco increased as the Nauru supply dwindled and imports from the United States stopped altogether. China was a significant supplier to New Zealand in the early 2000s, but it too has decreased exports. Togo provided imports in the early-mid 2000's but they have also stopped (Statistics New Zealand, 2013b).

Phosphate imports from different countries fluctuate from year to year, so in some years imports from a certain country may be high, while the next year may be miniscule. However, the majority of imports have been sourced from Morocco since the late 1990s with over half of the rock phosphate imports derived from Morocco in 2012 (Figure 8.2) (Statistics New Zealand, 2013b). Viet Nam and Peru have been other significant importers in recent years. Surprisingly, from 2009 imports were coming into New Zealand from Nauru again after halting for six years and in 2012 imports from Nauru accounted for 13% of imports (Figure 8.2) (Statistics New Zealand, 2013b). Other imports in recent years include from other parts of Northern Africa, such as Egypt and Tunisia, as well as sporadic imports from Christmas Island (Statistics New Zealand, 2013b).

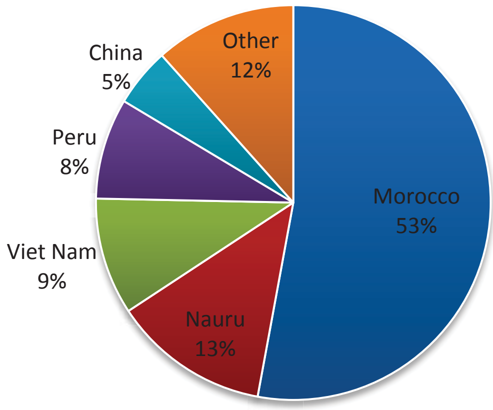


Figure 8.2: Proportion of main imports of rock phosphate into New Zealand in 2012.
Data source: Statistics New Zealand (2013b).

Rock phosphates are usually unable to be used for direct application as a fertiliser because of their insolubility under most soil conditions and therefore unavailability to plants. These forms have thus been converted to superphosphate using sulphuric acid (Techhistory, n.d.). Early sources of sulphuric acid came from volcanic deposits in Italy and White Island, then from Frash sulphur in Texas and Louisiana (Duncan, 2012). From the 1960s, it mainly came from natural gas from Canada (Duncan, 2012). However, another form of phosphate: reactive phosphate rock (RPR), can be applied directly to pasture without chemical processing and is an effective source of sustained-released P (Chatham Rock Phosphate, 2012; Roberts, 2012). RPR is estimated to reduce run-off P losses into waterways by 89-90% (Chatham Rock Phosphate, 2012). RPR is currently imported but is not utilised effectively as a fertiliser by itself. Mining phosphate from the Chatham Rise has been advocated as an alternative for New Zealand to importing phosphate rock as it can be applied directly to pasture without chemical manufacture, thus reducing the need for superphosphate fertiliser (see Chapter 10 for more information on Chatham rock phosphate).

Sustainability of phosphorus/peak phosphorus

Phosphate rock is a non-renewable resource that takes 10-15 million years to cycle naturally (Cordell et al., 2009). The largest commercially recoverable reserves are located in just three countries: China, the United States and Morocco/Western Sahara and five countries control almost 90% of the world's reserves of phosphate (Rosemarin et al., 2009). China and the US largely keep their supplies for their own use; China has imposed a 135% tariff on exports and extraction in the US has now peaked with reserves estimated to be depleted within 30 years. Morocco is the biggest international trader (Rosemarin et al., 2009), controlling more than half the total trade (Pearce, 2011).

There are now reports emerging that global peak phosphorus production levels are looming, however, this has attracted little attention in the agriculture industry or in the media. It has been estimated that phosphate extraction will peak around 2030 (time at which the maximum global phosphorus production rate is reached) (Cordell et al., 2009; Rosemarin et al., 2009). At current rates of extraction, global reserves will start to run out within 50-100 years (Cordell et al., 2009; Rosemarin et al., 2009). Phosphate rock reserves that are economically recoverable are currently estimated at 15 billion tonnes and about 167 million tonnes are extracted each year (Cordell et al., 2009). Reserves are estimated to last about 50 years at the current rate of extraction, which is increasing by 2% per year. However, if extraction increased to 3% per year, reserves will last less than 45 years (USGS, cited in Rosemarin et al., 2009). If rates were to exceed this amount, as they have done in the past, reserves could be depleted much earlier than estimated. In 2007-8, extraction increased by 7%, driven mainly by China, where output rose by 10%, and the US and Morocco by 4% (Rosemarin et al., 2009). If all the reserves were exploited, including those that are at this time not economically recoverable, then they would be depleted within 75 years if the rate of extraction were to rise to 3% per year (Rosemarin et al., 2009).

Impacts of peak phosphorus

Diminishing phosphorus reserves are likely to have severe impacts in terms of rising food prices, growing food insecurity and widening inequalities between rich and poor countries, putting constraints on global economic development (Rosemarin et al., 2009). In 2008, there was a spike in the price of food, oil, fertilisers and other raw materials. This spike was evident in many of the agricultural fertiliser imports into New Zealand in 2008. World phosphate rock prices increased by 800%, affecting farmers and leading to China imposing a tariff on phosphates, effectively halting exports from one of the largest phosphate producing countries (Ashley et al., 2011).

Action is needed to conserve the remaining stocks of phosphate rock, reduce the demand for fertilisers and recycle phosphorus wherever possible (Rosemarin et al., 2009). Phosphorus cannot be manufactured from alternative sources, but it can be recovered and reused. However, at present there are few initiatives to promote recycling. The tariff imposed by China could result in high prices elsewhere and encourage more efficient use of fertilisers within the agricultural sector. Other than this, none of the major suppliers have taken proactive steps to conserve or manage their reserves sustainability (Rosemarin et al., 2009).

Phosphorus inefficiencies and waste

Only a fifth of the phosphorus mined to produce food actually reaches the food we eat (Cordell et al., 2009). The remainder is lost during mining and fertilizer production, application and harvest, livestock rearing, food processing and retail, and finally during consumption and excretion (Ashley et al., 2011; Cordell et al., 2009). Phosphorus leaves the land through harvested agricultural crops and products, and when consumed most phosphorus leaves the body in urine (70%) and faeces (30%) (Ashley et al., 2011). The change in sanitation from around the mid-1800s which involved water-based disposal of human waste now sees phosphorus discharged to oceans, lakes and rivers instead of land, and thus it is permanently lost from the human food system (Ashley et al., 2011).

Additionally, fertiliser overuse has resulted in runoff and led to the accumulation of phosphorus in terrestrial and freshwater ecosystems. An estimated 37 million tons of phosphorus leaks to the environment globally each year, flowing into rivers and lakes where it promotes the growth of toxic cyanobacteria, consuming oxygen and creating eutrophication and 'dead zones' (Pearce, 2011). Nutrient levels are now about 75% above those in pre-industrial times and are threatening lakes, rivers and coastal zones (Rosemarin et al., 2009).

Impacts from global phosphorus mining

Phosphate strip mines produce around 150 million tonnes of toxic waste a year. Surrounding the world's largest mine in Florida, a million tons of mine waste is piled up at dump sites containing low levels of radioactivity (Pearce, 2011). Disputes have been raised over promised mine clean-ups, rivers have dried up, and settling ponds have leaked (Pearce, 2011).

Nauru and pacific islands

Perhaps the most devastating effects of phosphate mining and exploitation have occurred in Nauru. Nauru was once a phosphorus rich island but over 70 years of open-cut phosphate mining has depleted resources and turned the island into a wasteland. Much of the island is now covered in stone pinnacles left behind by mining and about 40% of the surrounding marine life has been killed (Colt, 2011). Rehabilitation has not occurred and much of the middle of the island is useless for farming or even living on (Colt, 2011). Because of this, the island lacks fresh food; most people live off low-cost fried food. Diabetes has increased to the point where about half of the population suffers from the disease (The Economist, 2001).

Phosphate production was crippled over disputes between phosphate mining companies and landowners over royalties paid (Colt, 2011). This virtually stopped export revenue. Nauruan's experienced a brief period of wealth not seen in other Pacific Island nations. They once had the second highest GDP in the world but now about 90% of residents are unemployed (Colt, 2011). Phosphate corporations have conducted numerous environmental and economic violations to the Nauruan people (The Economist, 2001). Nauru has been exploited for this critical natural resource.

Morocco

About half of the world's phosphate reserves are located in Morocco and it has been suggested that Morocco aim's to drive the commodity's price higher (Bloomberg Businessweek, 2010). This means the cost of everything requiring the use of phosphate will increase (Bloomberg Businessweek, 2010), particularly evident when prices climbed significantly in 2008. Morocco's phosphate reserves have changed significantly in recent years; in an update of global phosphorus resources in 2010, Morocco's portion went from 5.7 billion metric tons to 50 billion metric tons, 85% of the world's total phosphorus reserves (Bloomberg Businessweek, 2010). If this is true, Morocco has considerable influence over the world's phosphate prices, and hence over global food production by current production methods. If Morocco decide to stockpile their resources, as China and the United States have done, considerable ramification for global food production could occur. Likewise, if they significantly increase the price, many poorer nations relying on phosphorus for food production could be harmed.

8.2. Palm Kernel

Palm kernel expeller (PKE) is imported into New Zealand for a dairy cow feed supplement. PKE is derived from palm oil production. Therefore, the environmental effects of palm oil, and thus PKE, are discussed here. Palm oil production is rapidly increasing; the UN Food and Agricultural Organisation (FAO) estimates global oil palm cultivation increased from 3.6 million ha in 1961 to 13.2 million ha in 2006 (Koh & Wilcove, 2008). The production of palm oil generates numerous environmental impacts, including deforestation, often by extremely destructive methods; biodiversity loss; and GHG emissions.

Rainforest and peat destruction

Palm oil production is a major driver of deforestation in South East Asia as areas of rainforest and peat lands are converted to palm plantations as the industry expands (Greenpeace, 2010). Rapid increases in palm oil plantations in Malaysia and Indonesia in the last two decades have occurred (Table 8.3). At current deforestation rates, the United Nations Environment Programme (UNEP) warned that 98% of Indonesia's rainforest could be destroyed by 2022 and lowland forest much sooner (Nellemann et al., 2007). It is estimated that over half of the planting of oil palm between 1990-2005 in Malaysia and Indonesia involved clearing native forests, despite producers asserting that forests are not being cleared to grow oil palm (Koh & Wilcove, 2008). The Indonesian Ministry of Agriculture estimated an area of 7.8 million ha was under cultivated palm oil in Indonesia in 2010 (Directorate General of Plantations, 2011 cited in Carlton, 2011).

Table 8.3: Areas of palm oil plantation harvested in Indonesia and Malaysia in 1990 and 2008.

Country	Area of palm oil planation (million ha)		Percent change
	1990	2008	
Malaysia	1.75	3.90	123
Indonesia	0.67	5.0	646
Total	**2.42**	**8.90**	**268**

Data source: FAOSTAT (2010, cited in Carlton (2011)).

Palm oil producing companies often breach conditions required by law and destructive techniques often go unreported, thus the situation is probably worse than reported. For example, one particular company, Sinar Mas, was found to be operating illegally without carrying out Environmental Impact Assessments before clearing in eight out of eleven concessions examined (BSI-CUC, 2010). Additionally, deep peat forest was cleared for palm plantations on two Sinar Mas concessions in breach of Indonesian law (BSI-CUC, 2010).

Biodiversity loss

Conversion to oil palm plantations from forests results in significant biodiversity losses. Palm plantations (like all monocultures) are less complex than natural forests and support far less biodiversity. For example, the conversion of primary and logged forests to oil palm plantations in Malaysia was found to decrease forest birds species richness by over 70% and butterflies species by around 80% (Koh & Wilcove, 2008).

Palm production is also one of the leading threats to the orang-utan and the Sumatran tiger, among other species, due to the loss of natural forest habitat (Nellemann et al., 2007). The Borneo Orang-utan is classified as endangered, with estimates of between 45,000 and 69,000 left in the wild (Ancrenaz et al., 2008) while the Sumatran tiger is listed as critically endangered with a population possibly less than 500 (Linkie et al., 2008). Areas cleared for palm oil across the Island of Borneo are part of the last refuges for the critically endangered orang-utan.

PKE and New Zealand dairy

Fonterra's farm food supply subsidiary, RD1, obtains its PKE from Wilmar International, the largest trader of palm oil and kernel with a reputation as one of the most environmentally destructive palm companies. For example, Wilmar has been implicated in cases of rainforest destruction, illegal burning, and social conflicts over community lands (ECO, 2009). To meet New Zealand's 2008 imports of PKE, about 2.7 million hectares of palm plantations would have been required[16]. However, the Ministry for Primary Industries claims New Zealand plays no part in the destruction in South-East Asia, claiming that the expansion of the industry and deforestation is driven by the demand for palm oil alone, not other palm products. Claims are made that palm kernel is only a low-value by-product of palm oil (Ministry for Primary Industries, 2013g) that is "otherwise burnt or left to rot on the ground" (New Zealand Parliament, 2009). Further ignoring the implications of importing PKE, New Zealand's finance Minister, Bill English, turns the argument to users of palm oil (New Zealand Parliament, 2009), of which New Zealand imports about 0.1% of the world's supply. In contrast, New Zealand imports 30% of the world's PKE supply. In 2008, New Zealand's palm oil imports were valued at $35 million (22 million litres), whereas palm kernel imports were valued at $317 million (around 1.4 million tonnes) (Statistics New Zealand, 2013b). Although palm oil is used in many more imported products than just the raw product, it is not difficult to see which New Zealand is implicated more in.

Sustainable palm plantations?

The Roundtable on Sustainable Palm Oil (RSPO) is a voluntary certification scheme set up in 2002 to obtain palm oil from sustainable sources. However, the RSPO has failed to deliver major changes on the ground. Members of the RSPO, which include companies such as Unilever, Cadbury and Nestle, are dependent on suppliers that are involved in deforestation and peatland conversion (Greenpeace, 2007). The RSPO is required to carry out assessments before plantations are certified sustainable. However, assessments are not carried out in many plantations that are certified and there is little traceability (Greenpeace, 2010) so consumers do not know the origin of their palm products. It is suggested many companies use the RSPO certificate as a way of 'green washing' while not changing business practices (Greenpeace, 2010). With weak standards on GHG emissions, and poor monitoring and enforcement of standards, the RSPO is justifying further expansion of palm plantations and increasing demand for palm products by creating a deception of sustainable palm oil.

Currently, only 14% of global palm oil is certified by the RSPO (Roundtable on Sustainable Palm Oil, 2012). However, the proportion of PKE that is certified may be much smaller. Fonterra claims New Zealand imported PKE comes from sustainable sources (Mediapeople NZ, 2010), but according to Dr Vengata Rao, secretary-general of the RSPO, "very little expeller cake coming into New Zealand [in 2008] would have been RSPO certified at all" (Knight, 2009). In

[16] This is based on the premise that one hectare of oil palm produces about 500 kg of palm kernel (Wikipedia, 2013).

total, it was estimated that only about 80,000 tonnes of RSPO certified PKE was on the market between August 2008 and August 2009 (Knight, 2009) and in 2008, only about 15,300 tonnes of PKE was produced in Wilmar RPSO certified mills in Malaysia (Greenpeace, 2009), representing about 5% of New Zealand imports from Malaysia in 2009, or 1% of total PKE imported into New Zealand (Statistics New Zealand, 2013b). Even if all of it was brought by New Zealand (which is unlikely), the other 99% of PKE imported into New Zealand came from unknown sources (Greenpeace, 2009).

The Malaysian palm oil industry argues that sustainability will increase production costs and believes they are unfairly targeted compared to other oil crops (Chandran, 2010). If tropical rainforest destruction and clearing of peatlands were to be avoided for palm cultivation, production costs would increase, which may threaten the economic viability of the palm oil industry. The Malaysian palm industry admits they are focussing on markets in developing countries that will not place sustainability demands on their product and that will "impose fewer protectionist barriers" (Chandran, 2010, p. 10). In these countries, the need for cheap food usually outweighs "attitudes against development of third world countries, changing tree-cover scenarios, possible loss of biodiversity and the actual causes of climate change" (Chandran, 2010, p. 10).

Greenhouse gas emissions associated with PKE

GHG emissions from tropical forest destruction have variously been estimated at 11.3% (Herzog, 2009) and 20% (Greenpeace, 2007) of global emissions. Indonesia is the highest emitter of GHG's from deforestation, accounting for about a quarter of global deforestation emissions (Greenpeace, 2007). Moreover, Indonesia has the fastest deforestation rate of any major forested country, losing 2% of its remaining forest every year (Greenpeace, 2007). Indonesia was ranked the eighth largest global emitter of GHG emissions in 2010 (World Resource Institute, 2013).

The main GHGs emitted by palm oil plantations are carbon dioxide, methane and nitrous oxide. Deforestation and the clearing of peatlands releases carbon dioxide from biomass destruction, which can continue over many decades after deforestation has occurred (Carlton, 2011). Carbon dioxide is also emitted from energy for production operations and inorganic fertiliser manufacture. Methane is released from palm oil mill effluent and nitrous oxide is released from soil from the application of process residues and fertilisers as well as the depletion of soil organic matter (Carlton, 2011).

Peatlands store between a fifth and a third of the total carbon contained in terrestrial biosphere, but cover just 3% of earth's land surfaces. Stored carbon is estimated at 528 billion tonnes (Greenpeace, 2007). Deforestation and burning of peatlands in Indonesia releases about 1.8 billion tonnes (Gt) of GHG emissions every year (Greenpeace, 2007), equalling around 4% of global GHG emissions (World Resource Institute, 2013).

Conversion to palm oil plantations results in differing GHG emissions depending on the previous land use. The highest emissions were associated with peatland conversion, while the conversion of grasslands to oil palm was expected to result in carbon sequestration (emission estimates from palm plantation conversion from different land uses are shown in Appendix I) (Carlton, 2011). Emissions associated with PKE production in reference to New Zealand imports for dairy production were also calculated (Carlton, 2011). The production of one tonne of PKE was estimated to emit up to 18.2 t CO_2-e, depending on prior land use (see Appendix I) (Carlton, 2011). Two methods were used to estimate emissions: economic allocation which is based on the economic value of PKE; and mass allocated based on the physical mass of PKE.

Based on the economic value of PKE, emissions ranged from 0.55 to 1.02 t CO_2-e/t PKE and based on the physical quantity of PKE, emissions were between 3.40 and 6.33 t CO_2-e/t PKE (Table 8.4) (Carlton, 2011). Using these two methods, emissions associated with the 2012 imports of PKE ranged from 0.74 to 8.61 Mt CO_2-e (Table 8.4). These emissions were equivalent to between 1% and 12% of New Zealand's total emissions, or between 4% and 45% of New Zealand's estimated dairy emissions. For just the import of one product for the dairy industry, these emissions proportions are high. Thus, by importing PKE, Fonterra is significantly increasing the carbon footprint of dairy products.

Table 8.4: Emissions associated with PKE imported into New Zealand

	Emissions t CO_2-e t/PKE		Emissions associated with 2012 imports of PKE (000 t CO_2-e)	
	Low case	High case	Low case	High case
Economic allocation	0.55	1.02	748	1387
Mass allocation	3.40	6.33	4622	8605

Data source: Emissions associated with 2012 imports of PKE based on methods from Carlton (2011).

8.3. Conclusion

Not only does dairy farming have internal environmental impacts experienced in New Zealand, but there are also offshore impacts associated with products imported for dairy production. These imports are often hidden and not acknowledged as part of dairy's impact. Although the dairy industry is not directly responsible for the practices used during manufacturing and production, nor the management of associated impacts, dairying should be careful in choosing inputs that are more sustainably produced. By importing products that are associated with environmental destruction and high GHG emissions, the dairy industry are risking the reputation of not only themselves but other export industries and New Zealand's 'clean green' image. This will harm New Zealand's export revenue.

Chapter 9. VALUATION OF IMPACTS

We use nature because she is valuable. We lose nature because she is free.
(Pavan Sukhdev, Founder and CEO of GIST (Green Indian States Trust) Advisory)

Unrestrained intensification of dairy has clearly led to widespread environmental effects, both in New Zealand from farming itself, and offshore from the imported products used by the dairy industry. The costs of some mitigation options to reduce the harms associated with dairy farming have only been evaluated in rare instances. However, many of the impacts remain un-valued (or under-valued). To fill gaps, some research has delved into assessing the public perception of environmental degradation. These studies have the benefit of a deeper understanding of what healthy ecosystems may be worth in the market-place, and what value is placed on a healthy and aesthetically pleasing environment. On the other hand, perception studies usually severely under-estimate the true value of an ecosystem, the costs of replacing an ecosystem, and the provision of ecosystem services. Thus, the available valuations are a gross under-estimate of the true costs of dairy impacts if the impacted areas were to be reinstated to a healthy (or even less-degraded) ecosystem.

Despite a lack of impact valuation assessments and knowledge regarding production effects, the New Zealand dairy industry, led by Fonterra and Dairy NZ, are powering ahead with intensification. Economic assessments focus on: the export revenue of dairy farming, boasting that is it the highest in New Zealand; high milk production; increasing cow numbers; and the economic and employment opportunities offered from dairy farming (Ballingall & Lattimore, 2004; Schilling et al., 2010). Economic appraisals propose dairy makes a strong contribution to New Zealand's economy (Ballingall & Lattimore, 2004; Schilling et al., 2010), and without it, New Zealand would be economically depressed. However, these assessments leave out key facets of production. For example, the NZIER report titled: '*Dairy's role in sustaining New Zealand – The sector's contribution to the economy*', failed to mention any environmental impacts or costs of imports (Schilling et al., 2010). Reports like this give a false indication of dairy's true value. Furthermore, most Regional Councils and the Government push environmental remedying costs onto the public. Additionally, they do more to assist intensification by providing irrigation subsidies and implementing weak environmental regulations that lead to further exploitation and degradation. On the other hand, environmental assessment studies are typically small-scale, usually only focus on a selection of impacts, and most importantly do not economically value the loss or degradation of a resource or ecosystem. Essentially, there are two sides that need to be amalgamated. This has occurred only in rare instances on regional, catchment or river/stream scales[17]. Albeit, there has been no national evaluation of the major impacts of dairy farming: many ignore the effects and continue to exploit the excuse that 'dairy sustains New Zealand's economic well-being' (Schilling et al., 2010). In other words, we count the gains and ignore the costs.

Costs are involved in: remedying impacts, environmental degradation, and legitimising environmental externalities that are largely publically subsidised. Farmers often find it's cheaper to avoid cleaning up and increase production than attempting to mitigate the effects. Whereas collectively, more cost-effective solutions may involve management practices to minimise the impacts in the first place, rather than paying for clean-up projects. Some of the costs associated with environmental effects from the dairy industry are discussed in this chapter. Only a small proportion has been valued thus far so this estimate is incomplete. However, even this preliminary investigation reveals an indication of the costly remediation practices New Zealand could be facing.

9.1. Water

Major impacts in water from dairy farming are derived from faecal pathogens, nutrients and sediment (discussed in Chapter 5). Remediation options that have been priced in water include the cost to remove nitrate to reach drinking standards and lake mitigation techniques to remove nutrients from lakes. Other values considered are related to the public perception of the degradation of water resources and the value of water as an irrigation source.

9.1.1. Faecal pathogens

Health effects

Waterborne diseases in New Zealand are estimated to cost about $15 million per year in health related costs, while the total cost of pathogenic illnesses is nearly $114 million (see

[17] For example, in the Te Arawa Lakes, Rotorua; the Waikato River catchment; and for some measures of water quality in other stream catchments.

Appendix J) (Ministry for the Environment, 2007c). However, without knowing where the contaminated areas are, it is difficult to estimate the proportion of responsibility from different land uses. Therefore, the proportion of costs associated with dairy on a national scale is difficult to determine. Moreover, impact sources will differ greatly between regions, with dairy farming having a higher influence in the more prominent dairy regions of Waikato, Taranaki and Canterbury for instance. Health costs will be borne mainly on individuals who get sick (along with the Government who subsidies health-care), so it is important to realise that the costs are externalised from the problem.

Recreational and aesthetics

Freshwater areas contaminated with faecal matter will have a public loss/cost of being unavailable for contact recreation. Odour problems may also arise from the discharge of treatment ponds and effluent (Cameron & Trenouth, 1999). Although this does not involve a direct health cost, social costs of having degraded recreation areas are endured. These costs have not been quantified in this study.

9.1.2. Remediation of nitrate in water

Leaching of one kg of nitrate N (NO_3N) from soil will contaminate about 88.5 m^3 of water from 0 mg NO_3N/L to 11.3 mg NO_3N/L[18]: the MAV (Maximum Allowable Value) for drinking water standards. Dairy leaching rates per hectare can be used to estimate the volume of water that dairy farming could contaminate from having zero nitrate concentrations. To reduce nitrate concentrations by 85 to 95% in water costs at a minimum between $0.30 and $1.80 per 1000 litres (Jensen et al., 2012); however, costs vary considerably depending on the system type used and can be much higher.

To estimate the amount of water that could exceed nitrate drinking water standards caused by dairy farming in New Zealand, leaching rates of 12, 28, 130 and 200 kg N/ha/yr were used (Table 9.1) based on leaching ranges measured throughout New Zealand (detailed in Chapter 5). The average dairy leaching rate of 28 kg N/ha/yr may be heavily conservative for some regions, particularly irrigated dairy land, and it is likely that some dairy land leaches more than this. The volume of water contaminated was estimated for one hectare, the average dairy

[18] Based on the following calculations: 1 kg NO_3N = 1,000,000 mg NO_3N; 1,000,000/11.3 = 88496 L = 88.5 m^3.

farm size (140 ha[19]), and the total land area in dairy farming nationally (about 2.4 million ha) (Table 9.1). Although leaching rates up to 200 kg N/ha/yr are unrealistic for all dairy land, they could indicate possible future scenarios under continued dairy intensification. Scenarios estimate the cost to treat all water contaminated from dairy farming that would exceed nitrate drinking water standards. In reality, all of the contaminated water would not be used for drinking. Nevertheless, it represents a degraded natural resource and an externality from dairy farming. Additionally, estimates of nitrate contamination are based on water initially containing no nitrate; however, many groundwater reservoirs already contain nitrate with some areas currently exceeding drinking water standards. For example, 11% of monitored wells in the Canterbury region exceeded nitrate drinking water standards in 2012 (Environment Canterbury, 2013b). Therefore, some estimates in Table 9.1 are likely to be more conservative than reality. For instance, if water previously contained 50% of the nitrate levels for drinking water standards, then 1 hectare would contaminate between 2,124 and 35,400 m^3 of water per year depending on N leaching rates (compared to 1,062 to 17,700 m^3).

Table 9.1: Estimations of water exceeding nitrate drinking water standards from various dairy farm leaching rates for one hectare, the average farm size, and national dairy land.

	Leaching rates (kg N/ha/yr)			
	12	**28**	**130**	**200**
1 ha dairy land				
Nitrate leached (kg)	12	28	130	200
Water polluted (cubic metres - m^3)	1,062	2,478	11,505	17,700
Cost to remediate ($/ha)	$319 – $1,912	$743 - $4,460	$3,452 - $20,709	$5,310 - $31,860
Average farm size – 140 ha				
Nitrate leached (kg)	1,680	3,920	18,200	28,000
Water polluted (m^3)	148,680	346,920	1,610,700	2,478,000
Cost to remediate	$44,604 - $267,624	$104,076 - $624,456	$483,210 – $2.9 million	$743,400 – $4.46 million
National dairy land – 2.4 million ha				
Nitrate leached (kg)	28,800,000	67,200,000	312,000,000	480,000,000
Water polluted (Mm^3)	2,549	5,947	27,612	42,480
Cost to remediate ($ billion)	$0.76 - $4.59	$1.78 - $10.70	$8.28 - $49.70	$12.74 - $76.46

Notes: Mm^3 = Million cubic metres. Costs are based on a range of $0.30 - $1.80 per 1000 litres.

[19] Average effective hectares on dairy farms; however, this does not include wintering areas (LIC & DairyNZ, 2012).

Water contaminated with nitrate above drinking water standards in three predominant dairying catchments (Waikakahi, Canterbury; Bog Burn, Southland; and Toenepi, Waikato) were estimated using measured leaching rates from dairy farms in these respective catchments (estimates are outlined in Appendix K). The volume of water that could reach nitrate drinking water standards from proposed irrigation schemes in Canterbury was also estimated (Appendix K). Proposed irrigation schemes were estimated to equip an additional 270,000 hectares of land with irrigation in Canterbury (Kaye-Blake et al., 2010). Associated with this is an expected 180,000 hectares of dairy support land, estimated to leach 70 kg N/ha/yr (personal communication with Angus Robson, 11 June 2013). This projection assumes all newly irrigated land would be converted to dairy, leaching around 130 kg N/ha/yr; however, other land-use sectors may also use the irrigated land.

Waikakahi catchment, Canterbury

The Waikakahi catchment has attracted considerable attention regarding the impacts of intensive irrigated dairy farming on stream water quality (Monaghan et al., 2009). About 40% of the catchment is occupied by dairy farming and irrigation is required to maintain farm production[20] (Monaghan et al., 2009). Around 2000 ha was reported to be in dairy farming. Monaghan et al. (2009) determined leaching rates by soil type, with 52 kg N/ha/yr on free draining soil and 27 kg N/ha/yr on poorly drained soil (Appendix K). In addition, Power et al. (2002) measured average leaching rates in the catchment and estimated leaching would almost double if dairying intensity was to increase by 23%, from an average leaching rate of 39 kg N/ha/yr to 64 kg N/ha/yr.

Bog Burn catchment, Southland

About 37% of the Bog Burn catchment is in dairy farming with dry stock farming and forestry predominant in the headwaters (Monaghan et al., 2007). Monaghan et al. (2007) differentiated between leaching rates from dairy milking areas (16 kg N/ha/yr) and dairy wintering areas (55 kg N/ha/yr) (Appendix K). Power et al. (2002) reported average current leaching rates (18 kg N/ha/yr) from dairy farms in the catchment and estimated leaching rates from increased dairying intensity (23 kg N/ha/yr) (Appendix K).

[20] Border dyke irrigation is used in the catchment, where “water is applied to the top of an irrigation bay at flow rates that exceed soil infiltration rates” (Monaghan et al., 2009, p. 201). Excess water flows down the bay into drainage channels where captured water may be re-applied for irrigation elsewhere or discharged to a stream.

Toenepi catchment, Waikato

Nearly 90% of the Toenepi catchment is under dairying with no irrigation used on farms (Power et al., 2002). Average leaching from dairy land was 41 kg N/ha and scenarios estimate leaching associated with a 20% and 50% increase in intensity (Appendix K). Nitrate contamination from the average farm and total dairy land in the catchment are estimated (Appendix K).

Conclusion

To remedy water polluted with nitrate associated with leaching from one hectare of dairy land is estimated to cost between \$319 and \$31,860, depending on leaching rates and the treatment system used (Table 9.1). For an average sized dairy farm of 140 ha, costs range between \$44,600 and \$4.5 million. National water remediation costs for average leaching of 28 kg N/ha/yr are estimated between \$1.78 billion and \$10.70 billion (Figure 9.1 and Table 9.1). On the one hand, estimates are conservative because they assume contaminated water initially has zero nitrate levels. On the other, not all water would be used for drinking so would not require treatment. Therefore, it may be more realistic to only value the amount of water that is used for drinking water in New Zealand, estimated at 1832 Mm^3/year (Rajanayaka et al., 2010). To treat this amount of water for nitrate contamination would cost between \$549 million - \$3.30 billion. It is important to realise that this is not an actual fee that the dairy industry has to, or may have to, pay in the future. Rather, these costs represent an externality that dairy farming is not compensating, by being allowed to discharge nitrate to water generally unrestricted.

To remediate contaminated water from dairy land for the three dairy catchments would cost between \$500,000 and almost \$14 million per catchment (Appendix K). At estimated increased intensity scenarios, remediation would cost up to \$25 million for one of the catchments (Toenepi) for a 50% increase in intensity. For an average sized farm in the Toenepi catchment, remediation would cost between \$82,000 and \$496,000, but if intensity were to increase by 50% the costs would increase to between \$200,000 and \$1.2 million per farm (Appendix K). Even more deplorable is that proposed irrigation areas in Canterbury have the potential to cost up to \$8.2 billion for water remediation.

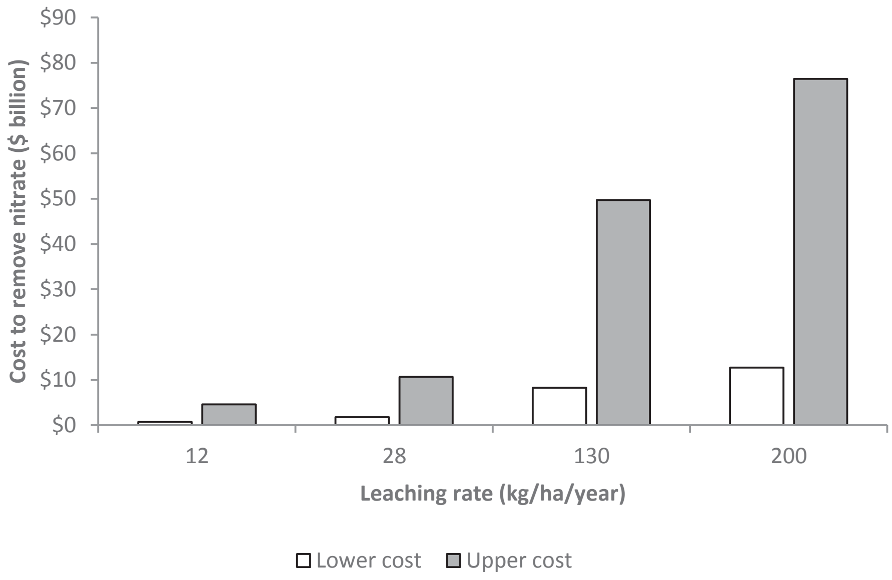


Figure 9.1: Variation of costs to remove nitrate in water that exceeds nitrate drinking water standards (11.3 mg N/L) estimated to leach from New Zealand's national dairy land under various leaching rates.

9.1.3. Lake water quality: Rotorua Lakes

Rotorua is one of New Zealand's main tourism destinations; almost a third of all international visitors to New Zealand spend at least one night in Rotorua (Parliamentary Commissioner for the Environment, 2006) and in 2012, tourism contributed $209.8 million towards GDP (Infometrics, 2012). With 14 freshwater lakes in the region, water based activities are important and good water quality is essential for these activities to thrive. A survey found that 94% of Rotorua residents and 86% of residents from the rest of the Bay of Plenty used the Rotorua Lakes (Environment Bay of Plenty et al., 2007). Around 45% of the Rotorua District is in pasture and the main agricultural activities are dairy, beef, sheep and deer farming (Parliamentary Commissioner for the Environment, 2006).

Twelve lakes make up the Rotorua (Te Arawa) Lakes, four of which will be discussed here: Lakes Rotorua, Rotoiti, Rotoehu and Okaro. Nutrient loads into the lakes have led to declining water quality for at least 30 to 40 years. Sources of nutrients are from farming, erosion, septic tank effluent, stormwater, community sewerage schemes, springs, geothermal sources and internal loads from lakebed sediments (Bay of Plenty Regional Council et al., 2010). Some catchments have high internal stores of nutrients and are in a degraded state (Abell et al., 2011; Bay of Plenty Regional Council et al., 2010; Parliamentary Commissioner for the Environment, 2006). Therefore, to achieve improvements in water quality in the short-median term, actions to remove nutrients are essential, combined with catchment-based strategies. Sediments in Lake Rotorua, for example, have high internal stores of nutrients that even if nutrient inputs were stopped, desired improvements in water quality would not be achieved (Abell et al., 2011). Time lags from groundwater entering the lakes delay the effects of nutrient loads from present-day land use, proving a major complication for lake restoration efforts. For instance, in some catchments water entering groundwater will take up to 170 years to reach lakes (GNS, 2005, cited in Abell et al., 2011), meaning that groundwater from the 1840s may only be reaching some lakes now. In eight out of 12 samples the mean residence time for water from groundwater to enter Lakes Rotorua and Okareka was over 60 years (Parliamentary Commissioner for the Environment, 2006).

A range of mechanical, chemical and biological methods are being used to improve water quality in the lakes but many will take years or decades to have an effect (Bay of Plenty Regional Council et al., 2010). Complex interactions occur in the lakes which help determine water quality (Bay of Plenty Regional Council et al., 2010). One of these is stratification (affecting dissolved oxygen levels), which occurs when oxygen in the bottom waters is not replenished adequately and may cause nutrients such as phosphorus to be released from lake sediments. Remixing can bring these nutrients to the surface where they can stimulate algal blooms, fuelling the cycle further (Bay of Plenty Regional Council et al., 2010).

9.1.3.1. Effects of degraded water quality

Excessive nutrient levels in lakes cause algal growth and toxic algal blooms, leading to eutrophic lakes. Toxic blue-green algae become dominant and evasive and other phytoplankton taxa do less well (Environment Bay of Plenty et al., 2007). Between 2000 and 2007, there were on average four and ten weeks of blooms each year in Lake Rotorua and Lake Rotoiti, respectively (Environment Bay of Plenty et al., 2007).

Algal blooms in the lakes affected recreational uses for 69% of Rotorua residents and 62% of the rest of the Bay of Plenty (Environment Bay of Plenty et al., 2007). Algal blooms affect recreation and swimming activities in the lake and health warnings have been issued for both Lakes Rotorua and Rotoiti. Frequent cyanobacteria blooms over the summer months in Lake Rotoiti have shifted recreational activity to other, cleaner lakes (Environment Bay of Plenty et al., 2007). This could be detrimental for activities as Lakes Rotorua and Rotoiti are popular lakes for anglers from around New Zealand and overseas. For instance, around $25 to $32 million is spent on angling-related activities and the trout fishery is valued at $70 million

(Environment Bay of Plenty et al., 2007). During cyanobacteria blooms on Lake Rotoiti in 2003, surveys estimated that peak summer angling usage declined by about 50%. Furthermore, water-based activities at the Rotorua Lakes Water Sports Trust were halted for almost four months in the summer of 2003/2004 because of toxic bloom warnings, having major impacts on revenues, participation, and out-of-town perceptions of facilities (Environment Bay of Plenty et al., 2007).

Habitat and food availability for aquatic species are also affected by increased algal growth as algal consume oxygen in the water (Environment Bay of Plenty et al., 2007). Macrophytes decrease when algal mass increase as they compete for nutrients and light in aquatic systems. A decline in macrophytes can change food webs in some systems (Environment Bay of Plenty et al., 2007) Declining water quality can affect fish, particularly lakebed dwellers such as koaro, as habitat reduces (Environment Bay of Plenty et al., 2007).

9.1.3.1. Nutrient inputs

Nutrients from past activities and land uses are dispersed in the bottom sediments of both Lakes Rotorua and Rotoiti (internal loading) and can be released into the lake water. Nutrients also enter the lakes from land use. The current inputs are the ones entering the lake now and exports will enter the lake in the future from current land use (nutrient inputs detailed in Appendix L). Nitrogen exports from land use are higher than inputs for Lake Rotorua due to the time it takes nitrogen to pass through the land and groundwater system to the lake (lag time). There is no groundwater lag time in the Rotoiti catchment, hence nutrient inputs are not expected to increase over time. While nitrogen leaches through the land, most of the sediment and phosphorus is transported to freshwaters during short periods after heavy rainfall; for example, half of the phosphorus and sediment loading from two major stream inflows to Lake Rotorua occurred about 15% and 1% of the time, respectively (Hamilton et al., 2013). Nutrient input targets have been established to improve water quality in the lakes (targets in Appendix L) (Environment Bay of Plenty et al., 2009).

Lake Rotoiti is connected to Lake Rotorua by the Ohau Channel where almost 70% of the nitrogen and 86% of the phosphorus entered Lake Rotoiti from Lake Rotorua (Environment Bay of Plenty et al., 2009). Hence, water quality in Lake Rotorua previously affected Lake Rotoiti. This was before the Ohau Channel wall was completed in 2008. The wall was built to prevent nutrient rich water flowing to Lake Rotoiti. Instead nutrients have been diverted down the Kaituna River (Bay of Plenty Regional Council et al., 2010). Accordingly, inputs into Rotoiti are expected to drop considerably when nutrient reductions from the wall are included. The diversion wall cost around $10 million (Bay of Plenty Regional Council et al., 2010), mostly funded by Bay of Plenty Regional Council, with additional finances coming from central Government (Abell et al., 2011) (i.e. from ratepayers and taxpayers). The short and long term effects of the diversion on Lake Rotoiti and the Kaituna River are uncertain (Parliamentary Commissioner for the Environment, 2006). Nutrient problems in Lake Rotoiti may be remediated, but consequently shifted downstream (Ford-Robertson, 2013a).

Lake Okaro is the smallest of the publically managed Rotorua lakes and its trophic state is supertrophic (very high nutrients and algal productivity) (Bay of Plenty Regional Council et al., 2010). Pastoral farming comprises 90% of the catchment with 10% in dairy farming (38 ha), although about 20% of the nutrient load into Lake Okaro is from dairying (Figure 9.2) (Environment Bay of Plenty et al., 2006). There is less infiltration to groundwater in the Lake Okaro catchment as it has a small, elevated groundwater catchment so a long lag time is unlikely. This means that the lake water quality generally reflects the current catchment land use (Environment Bay of Plenty et al., 2006).

Lake Rotoehu is a shallow lake (average depth is 8.2 m). Dairy farming covers about 5% of the catchment but contributes to 25% of the nitrogen load and 7.5% of the phosphorus load into the lake (Figure 9.2). Algal production in the lake has been a problem as early as the 1960s. In the 1990s nutrient levels in the lake doubled from the 1970s and algal mass in the lake quadrupled (Bay of Plenty Regional Council et al., 2007). Subsequently, from 1993/94 cyanobacteria blooms have occurred in the lake every summer except one (Bay of Plenty Regional Council et al., 2007). Increased nutrients have contributed to de-oxygenation of bottom waters, triggering nutrient releases from sediments in the lakebed (Bay of Plenty Regional Council et al., 2007).

Nutrient loads in catchments

In all four of aforementioned lake catchments dairy farming contributes a greater amount of nitrogen loading than its proportional land area, with the greatest impact in Lake Rotorua (38% of total N from 11% of the catchment) (Figure 9.2). Phosphorus loading from dairy farming is relatively similar to the proportion of land in dairy farming, except for Lake Okaro which contributes almost double the proportion of P than its land area. Nutrient inputs from dairy and some other land uses are shown in Appendix M.

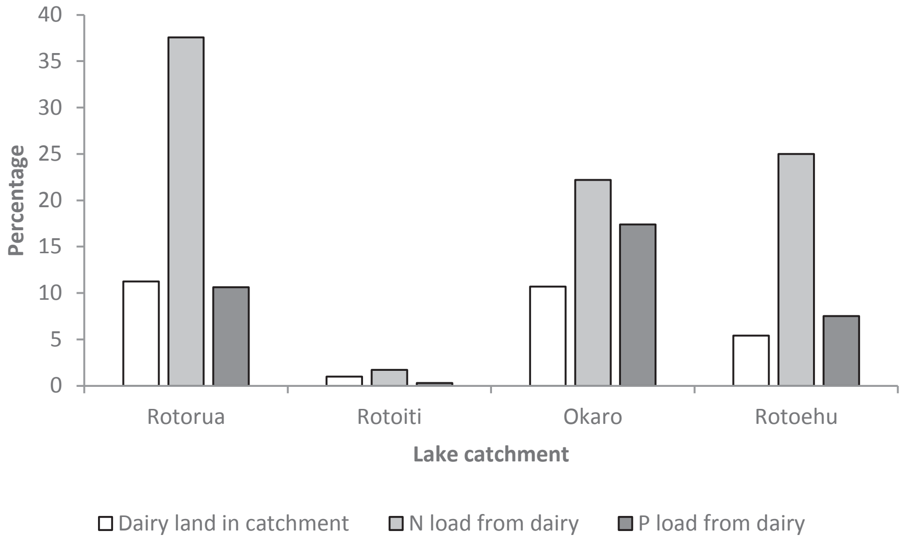


Figure 9.2: The proportion of land in dairy farming and proportion of nutrient loads (N and P) from dairy land in four Rotorua Lake catchments.
Data source: Environment Bay of Plenty et al. (2007).

Limiting nutrient inputs in the Lakes

The Rotorua Lakes Protection and Restoration Action Programme was established to identify issues and propose actions to address water quality problems. The Government has committed $72.1 million towards the programme which Environment Bay of Plenty (EBOP) and Rotorua District Council (RDC) will match ($144.2 million in total). Four Rotorua lakes will receive this funding: Lakes Rotorua, Rotoiti, Rotoehu and Okareka (Environment Bay of Plenty et al., 2009).

RDC proposes to spend $98 million over 10 years on new community sewerage schemes and other initiates that will benefit the lakes, including water conservation strategies and engineering expertise (Parliamentary Commissioner for the Environment, 2006). Waikato University has been awarded $10 million to research lake restoration over 10 years, focusing on the Rotorua Lakes (Parliamentary Commissioner for the Environment, 2006). The costs of the Proposed Action Plan for Lakes Rotorua and Rotoiti are expected to be around $10 million per year, excluding reductions from land use and management changes. This would achieve a reduction of around 59 tonnes N and 16 tonnes P per year (Kerr & Lock, 2009), costing about $169,500 per tonne of N ($170 per kg N).

9.1.3.2. Mitigation actions

A range of actions have been implemented to reduce nutrient loads from the internal (lake bed sediments) and external (land-use) sources in the lakes. Lake-based actions are reactive or 'end of pipe' techniques because they aim to reduce nutrient loads once nutrients have already entered streams or lakes, rather than reducing nutrients at the source. Although these actions are required for already degraded ecosystems, experience shows that techniques designed to reduce internal loads, without addressing external loads, often have limited success with short-lived benefits or delayed improvements (Abell et al., 2011). Consequently, repeated actions may be needed and failures can frequently occur, potentially causing additional environmental impacts, not to mention the high economic cost often involved for these actions. A summary of some lake-based actions and costs that have been investigated or carried out in the Rotorua Lakes are listed in Appendix N. More detailed mitigation costs for some actions are presented in Table 9.2. Options related to the Rotorua Lakes are expanded upon here, while more general examples of land management (i.e. riparian zones) are discussed in section 10.8 of Chapter 10.

Table 9.2: Actions carried out in the Rotorua Lakes and associated costs

Action	Nutrient reduction kg/year		Cost $	Cost $/kg nutrients	
	N	P		N	P
Wetlands					
- Constructed – Rotoehu 3.2 ha	1650 (516/ha)		$1 million $125,000/year	$76/year	
- Constructed - Okaro	348	16	$520,000	$1494 ($176/year)	
- Floating – Rotoehu (2800 m^2)	220 - 340		$690,000	$2,029 - $3,136	
- Floating – Proposed Rotorua	220 - 3650	2-12	$900,000	$246 - $4,090	
Riparian					
- Okaro – fencing, planting, restoration	423	37	$200,000[1]	$473	$5,405
- Rotoehu – protection and environmental programmes	542	249	$100,513	$185	$404
Weed harvesting – hornwort (Rotoehu)	2,400	320	$52,800	$22	$165
Sediment capping / phosphorus inactivation					
- Okaro – Phosphorus absorbent lakebed cap	240	380	$225,000 over 3 years (lasts 7 years)	$134	$84.6
- Rotorua - Alum dosing			$1 million		
- Okareka - phoslock			$300,000 capital		
- P flocculation – in 3 streams Lake Rotorua	0	6,000	$1.260 million/year		$210
Ohau channel diversion wall	150,000	15,000	$10 million $1,240,000/year	$8	$82

Data source: Environment Bay of Plenty et al. (2006); Bay of Plenty Regional Council et al. (2007); Environment Bay of Plenty et al. (2009). **Notes:** [1]Does not include all the costs; costs listed in Environment Programmes are given on a per-property basis (Environment Bay of Plenty et al., 2006). Differing cost estimates were available; hence some costs are available per year whereas others are per project.

Nutrient reduction using wetlands

Wetlands can remove dissolved nitrogen, sediment and toxins from water flows (Environment Bay of Plenty et al., 2006, 2007; Hamill et al., 2010). The process relies on the build-up of organic rich sediments and abundant denitrifying bacteria. Wetlands will take up high amounts of nutrients to start with, but over time if the plants are not harvested, nutrient uptake will cease (Hamill et al., 2010). Therefore, wetlands require maintenance and renewal over time. Wetlands remove some small amounts of phosphorus. For example, five Waikato wetlands receiving treated effluent were assimilating between 2% and 14% of the total phosphorus (Environment Bay of Plenty et al., 2006). However, eventually wetlands will build-up phosphorus and may end up releasing it; therefore, phosphorus removal is required (Environment Bay of Plenty et al., 2006). Floating wetlands remove internal lake stores of nutrients but do not target nutrient outputs from the land. Thus, they are only useful if nutrient inputs into lakes have stopped or decreased. Wetlands also have other ecological benefits – habitat for fish, amenity values, biodiversity, aesthetic and cultural benefits. Ecosystem benefits from wetlands can be substantial; for example in the Manawatu/Wanganui region, wetland ecosystems were valued at $43,320/ha, compared with $1,831/ha for dairy land (van den Belt et al., 2009).

Cost-effectiveness has been estimated over the life of different types of wetlands (Table 9.3), accounting for on-going maintenance, renewal and lease costs. Protecting natural wetlands was estimated to be the most cost-effective, while floating wetlands were the least cost effective (Hamill et al., 2010). Despite these findings, floating wetlands have been used in several instances for restoration of the Rotorua Lakes.

Table 9.3: Average cost-effectiveness of nutrient removal for different types of wetlands

Wetland option	Nutrient reduction kg/ha		Cost effectiveness - $/kg (range)	
	N	P	N	P
Protecting natural wetlands	-	-	$14 (11-18)	$431 (260-870)
Seepage wetlands	323	2	$20 (14-29)	$2,739 (1600-4720)
Restoring natural surface flow wetlands	289	10	$60 (47-85)	$1,714 (1110-3190)
Constructed wetlands	368	11	$79 (64-97)	$2,548 (1650-4600)
Floating wetlands	714	13	$437 (330-570)	$24,271 (17000-35900)

Data source: Hamill et al. (2010). Cost effectiveness = amount of nutrients removed for monetary cost.

Measures to target phosphorus

A phosphorus cap is an in-lake absorbent cap that will absorb the phosphorus released from lakebed sediments, thus limiting the release of phosphorus into lakes. In-lake actions like lakebed capping speed up the restoration process by decreasing algal biomass and improving lake water quality (Environment Bay of Plenty et al., 2006). A lakebed cap was proposed to cover the 20 ha of Lake Okaro that is deep enough to turn anoxic, with a total cost of $225,000 ($75,000/year) (Environment Bay of Plenty et al., 2006).

Alum was also applied to Lake Okaro in 2003 to remove dissolved phosphorus in the lake. It binds to phosphorus in the water column and settles on the lakebed. The alum mixed throughout the surface waters within five days and lowered phosphorus concentrations by 20% (Environment Bay of Plenty et al., 2006). However, the effectiveness of the Alum application in Lake Okaro appeared to be only temporary and phosphorus concentrations increased in 2006-2007. This could be due to the low Alum concentrations or the pH in surface waters exceeding 9 as pH outside the recommended range of 6-8 can significantly reduce the P sorption capacity of Alum (Ozkundakci et al., 2010).

9.1.3.3. Summary Lake Rotorua restoration

Dairy farming in the four discussed Lake catchments is estimated to leach around 320 tonnes N per year and 4.5 tonnes P per year. Costs for the removal of nitrogen from the Lakes range from $14,000/tonne N (Hamill et al., 2010)to $4 million/tonne N and around $250,000/tonne P (Ford-Robertson, 2013a, 2013b), depending on the removal method. This yields a cost between $4.48 million and $1.28 billion for N and $1.125 million for P.

For the most part, lake restoration attempts fall short of expectations of water quality improvements. Internal loading from lakebed sediments have significantly reduced the effectiveness of restoration efforts and delayed water quality improvements (Ozkundakci et al., 2010). Furthermore, despite lake restoration efforts, evidence suggests that nutrient reduction targets in Lake Rotorua and Lake Okaro would mean they will both remain eutrophic (Ford-Robertson, 2013a; Ozkundakci et al., 2010). The actions in the Lake Okaro Action Plan will not reduce all the nitrogen required for the target; 149 kg/yr still needs to be reduced which may be achieved by best management practices (Environment Bay of Plenty et al., 2006).

Despite the efforts of lake-based actions, focus should be on catchment-based actions to reduce external nutrient inputs into the lake and avoid costly and lengthy remediation programmes (Abell et al., 2011). Catchment based actions could include changes to land management practices (examples in Chapter 10) and education on eutrophication issues, along with stricter environmental standards and market-based instruments to correct the problem of environmental externalities (Abell et al., 2011).

9.1.4. Lake Taupo

The health of Lake Taupo is also deteriorating as a result of increased nitrogen inputs. Intensification of land use over the last 50 years has increased nitrogen loads entering the lake (Environment Waikato, n.d.). Preserving Lake Taupo's water quality was thought to be the most important issue for around 90% of the Taupo community in 1999 (Environment Waikato, n.d.). To maintain water quality, a 20% reduction in manageable nitrogen loads was required (Lake Taupo Protection Trust, n.d.; Samarasinghe et al., 2011). Approximately 50% of the nutrient load into Lake Taupo is derived from pastoral agriculture, covering 22% of the catchment (Cullen et al., 2006).

The Lake Taupo Protection Trust was given $81.5 million, funded by national taxpayers and local ratepayers (Ministry for the Environment and Regional and District councils), to help protect Lake Taupo's water quality by reducing N lake inputs by 20% (170 tonnes N) (McLay, 2012), yielding a cost of $479,000/tonne N. Potential methods considered are purchasing land and converting land use to low N uses (Lake Taupo Protection Trust, n.d.). A nitrogen trading scheme has also been set up in the catchment, with nitrogen credits reaching as high as $650 per kg N but later dropping to around $400 to $420 (MacGibbon, n.d.). There are six dairy farms in the catchment, producing a load of 68 tonnes N/year among them and contributing to 12% of the manageable N reaching Lake Taupo (MacGibbon, n.d.). Following a nitrogen trading scheme and purchasing credits for the entire N load, dairy farmers could face costs of up to $44.2 million. Based on the cost of $479,000/tonne N to change land use, evenly spread for all nitrogen inputs, the cost to convert all dairy land would be $32.6 million. However, both of these costs would be unrealistic in this situation, as the entire dairy load would not need to be stopped. Based on dairy farms contributing 12% of the manageable N load, 12% of the total costs could be allocated to reducing dairy N loads, totalling $9.8 million. Again this could be unrealistic because dairy land may be more expensive to purchase than other land types, or mitigation measures to reduce N loads from dairy land may be more expensive to implement. This range of costs could be used to value the externality of nitrogen leaching from dairy farms in the Lake Taupo catchment.

9.1.5. Waikato River restoration costs

As part of the Waikato co-management agreement between the Crown and Waikato River iwi[21], the Waikato catchment is undergoing a transformation to restore the River catchment to a vision aspired by the iwi and the community. A number of scenarios have been priced that will improve the health of the river. To improve the state of the river but not fully meeting the vision expressed for a healthy and well river is expected to have a net cost of $1.66 billion over 30 years (2011-2040) (net present value of $0.9 billion) (NIWA, 2010b). However, to go further and meet almost all the vision goals would cost around $4.02 billion over 30 years (net present value of $3.18 billion), while implementing priority actions that are the most cost-effective have an estimated net cost of $2.24 billion (net present value of $1.4 billion) (NIWA, 2010b).

A number of actions have been proposed on dairy farms to contribute to improving the state of the catchment, but they must be combined with other actions to achieve overall catchment benefits. Actions included the use of nitrification inhibitors ($138 million) to limit nitrogen leaching, and fencing and planting five metre wide riparian buffers along all streams and drains on dairy land ($263 million) to prevent stock access (NIWA, 2010b). Riparian buffers will also have added benefits of intercepting runoff, supplying leaf litter and stream wood, and providing shade and overhang cover for fish, as well as improving aesthetics. However,

[21] Refer to NIWA (2010b) for more information

nitrification inhibitors only work on certain soil types (around 5-10% of dairy farms), and are now banned from use (Federated Farmers, 2013b); hence they can no longer be considered an effective strategy. Better nutrient management (with nutrient plans) would cost $10.5 million but would be recovered by savings on fertiliser ($36.1 million) (NIWA, 2010b). Diverting runoff from dairy farms is estimated to cost $5.4 million and creating wetlands on dairy farms would cost $45 million (NIWA, 2010b). Excluding the cost of nitrification inhibitors, the total cost to implement the above mentioned actions on dairy farms (riparian buffers, nutrient plans, nutrient diversion and wetlands), taking into account the savings on fertiliser, would be $287.8 million.

9.1.6. Costs of a degraded environment

Contingent valuation methods have been used to measure the monetary value of environmental impacts on natural systems that are not usually measured (more detailed description in Chapter 3). Instead of measuring how much impacts cost to remediate, these methods estimate the value humans attribute to ecosystem degradation and ecosystem services. As a result, studies frequently under-estimate ecosystem values, as they rely on the knowledge of respondents' who are generally unaware of the full extent of ecological impacts. Various water quality valuation studies have been carried out in New Zealand relating to effects of dairy farming (Table 9.4). Valuations from these studies are not included in the costs from dairy farming impacts in this analysis as they represent a perceived value rather than a real value or cost.

Many surveys use the 'willingness to pay' (WTP) and 'willingness to accept' (WTA) measures. WTP is generally the amount people would pay for improvements in environmental condition and WTA is the value people would find acceptable for a degraded environment. WTP significantly underestimates the benefits and Marsh (2012a) argues that the appropriate measure of the benefits of improved environmental quality is provided by WTA. WTA is always larger than WTP, suggesting perhaps that people are not willing to have a degraded environment but a clean environment is a given. For example, Marsh (2012a) measured a WTP of around $6 million for improved water quality versus a WTA of over $26 million for deteriorating water quality in the Manawatu/Whanganui (Table 9.4). The difference between WTA and WTP is usually based on income. In terms of WTP, the amount of disposable income available constrains how much can be demanded, whereas compensation (determining WTA) is not limited by income (Marsh, 2012a).

Table 9.4: Summary of water quality valuation studies undertaken in New Zealand

Measurement	Location/ catchment	Value	Reference
Value of water quality improvement	Canterbury	5 years – $81- $185 million	(Tait et al., 2011)
Recreational users and residents WTP for better water quality	Manawatu / Whanganui	>$6 million/year	(Marsh, 2012a)
WTA for deteriorating water quality	Manawatu / Whanganui	>$26 million/year	(Marsh, 2012a)
WTA for loss in quality of 'suitability for recreation' from good to not satisfactory	Canterbury household	$410/household/year	(Marsh, 2012a)
WTA for deterioration in tributary water quality from 'not satisfactory' to 'poor'	Hurunui catchment, Canterbury	$282/household/year	(Marsh, 2012a)
Improving water quality to a standard	Lower Waimakariri River	$10.1 - $17.2 million over 10 years, depends on interpretation of water quality standards	(Sheppard et al., 1993)
WTP for improvement in water quality to swimming standard	Lower Waimakariri River	$102/household For Canterbury residents - $94.4 million	(Sheppard et al., 1993)
Lake water quality improvements	Lake Karapiro, Waikato	Median WTP - $26-86/household – no job losses; $-4-30/household – dairy job losses. Catchment $0-$0.7 million	(Marsh, 2012b)
WTP for 10 or 30% reduction in nitrate leaching	Canterbury; aggregated to New Zealand	$11.83 (10%) - $38.55 (30%) / household. $17.75 million - $57.83 million (NZ)	(Baskaran, Cullen, & Takatsuka, 2009)
WTP for 10 or 30% reduction in water use for irrigation	Canterbury	$8.33 (10%) - $8.90 (30%) / household. Canterbury $1.8 million for 30% reduction	(Baskaran, Cullen, & Takatsuka, 2009)
WTP for 30% reduction in nitrate concentrations	Canterbury	$5.5 million	(Baskaran, Cullen, & Colombo, 2009)
Gains in improved water quality (increased water clarity in metres) in lakes with poor and average water quality	Rotorua Lakes	1 m increase - $1.18 million; ($865 to $572,392 per lake). 3 m increase - $3.68 million over all the lakes ($2,902 to $1.85 million per lake)	(Mkwara & Marsh, 2011)
Losses to possible lake closure due to poor water quality	Rotorua Lakes	$0.57 - $276 per angler per fishing season. Losses ranged from $7,565 to $6 million per lake	(Mkwara & Marsh, 2011)
Water quality effect on house prices	Rotorua Lakes	1 m improvement in water clarity resulting in average house price increase of 7%	(Woodham & Marsh, 2011)

WTP = willingness to pay; WTA = Willingness to accept

9.1.7. Value of water resources

There is a common assumption while assessing productivity that the value of inputs into production are assumed to be equal to their total costs (York, 2011). This theory assumes that free inputs, such as many natural resources, do not add measurable value to farmers. Although water in New Zealand is free, it obviously has substantial benefits for many farmers. There are costs associated with accessing water, but the water itself is not priced. When valuing natural resources, selecting a proxy price for a free good is contentious. York (2011) suggests the good's production cost or using an existing market price from within the country are the most available and least controversial options. However, production costs generally value the cost to provide water and not the value of water itself so may significantly under-estimate the value of water. For example, Lynch & Weber (1992) found the public value of water resources was much higher than the value of water to farmers. They found the value of in-stream flows of the Ashburton River to Canterbury residents averaged between \$2.47 and \$5.15 million; however, the value of water to farmers in the Ashburton catchment was only \$0.62 million.

The economic value of groundwater resources to users in the Waimea Plains is estimated at \$250 million and represents around 1% of New Zealand's total consumptive water allocation and 3% of New Zealand's groundwater allocation (White et al., 2001). This equates to \$38 to \$42 million for irrigators on the Waimea Plains (marginal value of \$240 to \$300/m^3). Applying the economic value of Waimea Plain's groundwater to New Zealand's total consumptive water allocation would attain an economic value of around \$24 to \$25 billion (White et al., 2001). Additionally, if non-consumptive uses were included the value would increase significantly (White et al., 2001).

Irrigation in Canterbury – Hurunui

Infrastructure improvement projects have been proposed for the Hurunui Catchment in North Canterbury to increase water availability for irrigation. The current area of irrigated land in the catchment is about 22,300 ha, of a total catchment area of 246,600 ha. Over 99% of the base irrigation occurs in the plains area (comprising over 76,000 ha), which has the highest productivity and revenue potential (Daigneault et al., 2011). The plains area produces a high proportion of the catchment's nutrient loads and GHG emissions and also has the greatest potential to alter its environmental outputs through changes in farm inputs and land use (Samarasinghe et al., 2011).

Total catchment income before irrigation development was estimated at $224.4 million (Daigneault et al., 2011). There is a large difference in farm incomes for farms with and without irrigation and there is little additional water available in the region to be allocated (Daigneault et al., 2011). Increasing water availability by as much as 86% will only increase total catchment income by 1.7% after additional capital and operation costs from the project are taken into account (Daigneault et al., 2011). Increased production will also increase environmental outputs such as N and P, as well as CO_2 emissions from additional farm operations and energy used for irrigation (Daigneault et al., 2011). Additionally, costs to other sectors of the local economy that are reliant on good water quality could increase (Daigneault et al., 2011); however, these have not been accounted for in productivity models.

Furthermore, it is estimated that less than 4,000 of the nearly 20,000 ha of newly irrigated water available would be used on a yearly basis. This is because most land owners would be unable to economically change their land use and inputs while meeting environmental limits; thus catchment income would only increase by 0.3% (Daigneault et al., 2011). The net present value of the Hurunui Water Project was estimated to be in the order of millions to irrigators. In contrast, Kerr (n.d.) argues that the proposed irrigation scheme not only has the potential to impose net costs on society as a whole, but may also be unviable for irrigators. This is because external costs were not included in assessments.

Value of irrigation to dairy

Assessments on an accurate value of water are not possible for the scope of this thesis as water does not have a fixed price. Furthermore, the volume of water that dairy farms use is unknown. A rough estimate of water use could substitute the proportion of irrigated dairy land area as the proportion of water use. However, this is likely to underestimate the volume of water used by dairy if dairy land uses more water per unit of land than other land uses. Of the area equipped with irrigation, dairy covered 49% in 2012. In 2010, 5791.2 million m^3 was allocated for irrigation; thus, approximately 2802.9 million m^3 per year may be used on dairy land. Water has been priced for the proposed Ruataniwha water storage and irrigation scheme at $0.23/m^3 (Hawke's Bay Regional Investment Company, 2014), likely to be used mainly for dairy. Using this water price, estimated water used for dairy nationally is valued at $644.7 million per year. Although this price may not be representative of water prices for agriculture in many catchments, this represents another externality of dairy farming, as farmers largely do not pay for water.

9.2. Land

Very little cost estimates exist for environmental impacts experienced on land from dairy farming. Those explained here are mainly experienced by farmers and effect dairy productivity, rather than impacts and costs endured by the public.

9.2.1. Soil properties

Samarasinghe and Greenhalgh (2009) found that the farmland value is reflected by the value of the inherent soil characteristics. Many soil characteristics and soil itself cannot easily be replaced or substituted. Therefore, soil is a critical natural stock. Soil characteristics such as particle size drainage, potential rooting depth, and available water are valued and used by regional councils to determine property rates for rural land (Samarasinghe & Greenhalgh, 2009).

A hedonic pricing technique (explained in Chapter 3) was used to value soil characteristics and specific land areas within the Manawatu catchment (Samarasinghe & Greenhalgh, 2009). On average, farmland with higher potential rooting depth and total available water were valued higher. Estimations suggest farmers are willing to pay a premium of approximately NZ$1,017 per hectare (5% of the average per hectare farmland value) to avoid reducing potential rooting depth by 25 cm (Samarasinghe & Greenhalgh, 2009). As on-going compaction can reduce rooting depth by up to 20 cm, these costs could be associated with treading damage.

Treading damage costs can be significant. To increase milk production on highly stocked farms there is a move to supply more feed. The cheapest option to increase feed supply may be by reducing pasture damage (Thorrold, 2000). Thorrold (2000) reviewed the costs of pasture damage on a typical Southland farm growing 12000 kg DM/ha and milking 2.5 cows/ha (Table 9.5). A 10% decrease in yield over the farm was modelled, representing 1200 kg DM/ha. The cost will depend on whether the feed is replaced or if production is reduced (Table 9.5). Compaction has been estimated to occur on 37% (Taylor, 2011) to 53% (Ministry for the Environment, 2011d) of dairy land in New Zealand (discussed in Chapter 6). Based on the individual farm costs (Table 9.5), national costs have been estimated at $75 million to over $600 million over effected land in dairy farming (Table 9.6).

Table 9.5: Actions and costs of pasture damage from compaction on dairy farms

Action	Cost ($/ha)	Details
Replacing feed	84	Urea, assuming a response rate of 10 kg DM/kg N and 70c/kg N
	180	Off-farm grazing, assuming a cost of 15c/kg DM
Reduced production	288	For lost annual production, assuming a conversion rate of 60g MS/kg DM and a payout of $4/kg MS
	480	For lost autumn production, assuming a conversion rate of 100 g MS/kg DM and a payout of $4 kg/MS.

Data source: Costs estimates by Thorrold (2000). **Notes:** Milk pay-outs have increased since this valuation was carried out. Other costs could also vary.

Table 9.6: Cost of compaction damage for effected national dairy farming area

Action	Cost ($million)	
	37% effected	**53% effected**
Replacing feed	$75 - $160	$107 - $229
Reduced production	$256 - $426	$366 - $611

9.2.2. Soil contamination

Financial impacts associated with cadmium accumulation (see section 6.5, Chapter 6) for landowners wishing to subdivide could be substantial. These costs could include those associated with a site investigation to determine whether guideline values are likely to be met; remediation if guidelines are not met (usually involving soil mixing or removal); a site validation report; and additional costs associated with a more elaborate resource consent assessment (Cadmium Working Group, 2008). Furthermore, cadmium accumulation could influence the market value of a property (Cadmium Working Group, 2008). Despite the likely high costs associated with soil contamination, particularly from cadmium, cost assessments have not been carried out.

9.3. Atmosphere

New Zealanders have been assured that New Zealand is on track to meet its Kyoto reduction commitments (Groser, 2012). The National Net Position estimates show a surplus of 29.6 million tonnes (Mt) for the period 2008-2012 (Ministry for the Environment, 2013d). While actual emissions are 18% over the Kyoto target, they are offset by counting carbon stored in forests. New Zealand has forest credits of 86 Mt, thus contributing to a surplus. This leads to the illusion the taxpayer is in good shape for the first commitment period. However, Terry (2012a) explained gross emissions for this period are 18% above 1990 levels (56 Mt in excess of target) while net emissions are 31% above 1990 levels. Future taxpayers will have to pay when stored carbon is released from the harvesting of production forests. Further information on the ETS and tax-payer position is discussed in Appendix O.

If polluters had to pay for their GHG emissions based on a carbon price, the dairy industry would have to pay for 19.20 Mt CO_2-e for their on-farm emissions. How much they pay depends on the carbon price. Currently the carbon price is very low so the cost would be relatively small ($12.7 million); however, at a price of $25/tonne CO_2-e (former international carbon price) the cost would increase to $480 million (Table 9.7). Furthermore, potential future prices may potentially increase substantially, generating a cost of almost $2 billion. A range of prices were used to show the annual cost variability for emissions in 2012 and to predict potential future scenarios (Table 9.7). The total emissions for the Kyoto commitment period were also priced.

Table 9.7: Potential cost of dairy farming emissions with different carbon prices

Carbon price (NZ$ per tonne CO_2-e)	Cost for on-farm dairy emissions (2012)	Cost for emissions (2008-2012 – 88.5 Mt)
March, 2013 - $0.09	$1.73 million	$7.97 million
May 2013 - $0.66	$12.67 million	$58.41 million
$1.00	$19.20 million	$88.50 million
$15.00	$288 million	$1.33 billion
International price - $25.00	$480 million	$2.21 billion
$50.00	$960 million	$4.43 billion
Future price? - $100.00	$1.92 billion	$8.85 billion

Notes: Future carbon prices are expected to be costly in the 2020s, rising from $50/t to $140/t (Terry, 2012a).

9.4. New Zealand's Clean Green Image

Initially used for tourism marketing, the 'clean green' image has been picked up by many other industries and is now fundamental to many of New Zealand's export industries, particularly agriculture. It is likely to be worth hundreds of millions, if not billions, of dollars per year

(Ministry for the Environment, 2001b). Linked to this, is the '100% Pure New Zealand' campaign launched in 1999 by Tourism New Zealand and based on the same assumption that New Zealand's environment is pristine and among the best in the world. Although, this slogan was changed to '100% Pure You' in 2011, perhaps to avoid association with the environment. Whether or not New Zealand can (or should) claim to be clean and green is discussed further in Chapter 10.

More than 80% of New Zealand exporters in a 2008 survey believed that the clean green image was vital to their export profile and in 2005 the '100% Pure' brand was valued at $20.17 billion per year (Stewart, 2012). A study on the value of New Zealand's 'clean green' image found surveyed international consumers would purchase 54% less dairy products if New Zealand's environment was perceived as being degraded (Ministry for the Environment, 2001b). Due to this, the loss in revenue of dairy products was estimated at between $241 and $569 million (Ministry for the Environment, 2001b). Additionally, losses in tourism revenue from a degraded image were estimated to be between $530 million and $938 million (Ministry for the Environment, 2001b). Other sectors could also be affected without even branding themselves as a 'green' product. This study was conducted over a decade ago, so current losses are likely to be higher than those initially estimated. Additionally, the brand is probably worth more than estimated as most of New Zealand's primary products benefit from it and the study was only carried out for three export sectors – dairy products, tourism and organic foods. If New Zealand were to lose its clean green image, it is suggested it would be very hard to regain – harder, perhaps, than restoring the environment itself (Ministry for the Environment, 2001b).

9.5. Summary of Environmental Costs

After assessing some of the costs involved in mitigating or remedying impacts, costs incurred on a lower and upper scale have been estimated (Table 9.8 and Table 9.9). This is a conservative estimate as many of the impacts cannot be valued and some are only valued on a local or regional scale. For example, costs to remove nutrients from lakes have only been valued for a handful of lakes. Likewise, the costs of reducing impacts from dairy land on river ecosystems have only been valued for the Waikato River catchment. Furthermore, health costs from water contaminated with pathogens are not included, nor are potential costs associated with contaminated land or costs of biodiversity loss. Additionally, non-market values are not included, which include the value of environmental degradation to the public, the value of recreation, and the value of aesthetics and cultural values. Costs from offshore impacts are also not included in this total, but imported costs of some products have been incorporated in Chapter 10 (Table 10.2).

As a result, this is only a preliminary and conservative estimate to determine some of the costs of mitigating or remedying effects, or the value of some environmental externalities produced by dairy farming in New Zealand. From available assessments, total costs (incorporating both available national and local costs) are estimated between $3 billion and $16 billion. Those only estimated on a national scale are between $2.76 and $14.4 billion (Table 9.8). However, costs estimated on a local scale (Table 9.9) would be significantly larger if they were extrapolated to the rest of New Zealand. Thus, this analysis reveals that it is likely that the environmental externalities from dairy farming exceed dairy's export revenue. It is important to note that this is not a present cost to the dairy industry, but a valuation of some of the externalities that dairy farming produces. Nevertheless, it should be questioned whether dairy is beneficial for the country, economically, environmentally and socially, given the many costs and impacts that have not been measured. This analysis highlights the urgent need for more research to assess the true value. Independent reviews of the dairy industry are required, involving cost/benefit analyses on all aspects of the industry.

Table 9.8: Summary of the available upper and lower national costs from some of the impacts from dairy farming in New Zealand

Measure	Scale of measure	Cost ($ million) Min Lower	Cost ($ million) Min Upper[1]
Removing nitrate from drinking water	National water surpassing nitrate drinking water standards from dairy leaching annually (Section 9.1.2)	1784	10705
Value of water	Value of water for annual dairy irrigation nationally (Section 9.1.7)	644.7	644.7
Cost of compaction	National dairy land affected by compaction (Section 9.2.1)	75	611
GHG emissions	Potential annual cost of national dairy GHG emissions $0.66 - $100 (Section 9.3)	12.7	1920
Clean green image	Loss of annual value for dairy products if image was degraded (Section 9.4)	241	569
	National subtotal	2757.4	14449.7

Notes: [1]The minimum upper limit is the upper value for which value assessments were undertaken. It is still regarded as conservative considering the valuations that were not included.

Table 9.9: Costs associated with restoring some local/regional areas from dairy farming impacts

Measure	Scale of measure	Cost ($ Million) Min lower	Cost ($ Million) Min upper
Lake remediation for nutrient inputs (Sections 9.1.3 and 9.1.4)	Rotorua Lakes - nutrient outputs from dairy farming for four lakes - nitrogen	4.48	1280
	Rotorua Lakes - nutrient outputs from dairy farming for four lakes - phosphorus	1.125	1.125
	Lake Taupo – costs for nitrogen leaching from dairy farms in catchment	9.78	44.2
Restoring river catchment ecosystems (Section 9.1.5)	Waikato River - dairy land Riparian ($263 million); nutrient management and fertiliser savings (-$25.6 million); diverting runoff ($5.4 million); wetlands ($45 million)	287.8	287.8
	Local subtotal	303.2	1613.1

Chapter 10. DISCUSSION: THE TRUE VALUE OF MILK

The only measure by which we will be judged by the people who come after is the health of the land base, because that is what is going to support them. They are not going to…[care] whether or not we were pacifists; they are not going to…[care] if we supported Israel or we didn't support Israel; whether we voted green or…[labour or national] or not at all. What they are going to care about is whether they can drink the water, whether they can breathe the air, whether the land can support them. One of the important questions is to ask what does the land need from you.

(Derrick Jensen, 2006)

The drive for growth via intensifying dairying has occurred without any balanced economic evaluation and awareness of the full environmental impacts. There are various examples of environmental degradation in New Zealand, significantly fuelled by intensive dairying. Unfortunately, the government response to this crisis to date will only exacerbate the environmental problems through intensification from a $35 million investment in irrigation (Ministry for Primary Industries, 2012b) and an additional $400 million proposed for irrigation (Tarrant, 2011). Preliminary valuations involving only some of the impacts reveal costs in billions of dollars and it is likely that the cost of impacts could exceed the export revenue received from dairy farming. Hence, the dairy industry is receiving a subsidy for their externalities. An overview of impacts from dairy farming is provided in Table 10.1. Many of the consequences listed in Table 10.1 will impact both domestic and international tourism in New Zealand.

Table 10.1: Overview of impacts from dairy farming

Output	Direct impact	Consequences of impacts
Nutrients	Increased GHG emissions – Nitrogen	Climate change; increased emissions costs
	Nitrate in drinking water	Drinking water contamination - health risks
	Increase in plant growth, algal blooms and weeds in water - eutrophication	Fluctuating oxygen levels, increased temperature – leads to biodiversity loss and degraded recreation areas
Pathogens	Soil contamination	Could cause cattle or humans to become sick

	Water contamination	Waterborne diseases, illnesses and infections – loss of recreation areas; impacts cultural values
Sediment	Turbidity in water	Reduced visual clarity and light, affecting food availability and predator visibility
	Affects growth rates and disease resistance of fish; and clogging of fish gills, damaging respiration	Fish kills and reduction of biodiversity
	Smothering of benthic food and habitat in aquatic systems	Reduction of biodiversity
	Reduced clarity in water	Degradation of recreation areas; affects safety of contact recreation
	Infilling of lakes, rivers, estuaries and reservoirs	Reduced water storage, siltation of water supply intakes, channel shoaling, raises stock banks, increases flooding risk
	Loss of productive land	Affects economic returns
GHG emissions	Contributes to climate change	Widespread global environmental impacts
	Poor environmental image for NZ	May affect tourism and export products
	Increased costs for the ETS and Kyoto Protocol	Future generations may face heavy economic costs
Heavy metals	Land contamination	Unable to convert land, reduction in land value
	Water contamination	Harms human health and biodiversity
	Food contamination	Food safety implications, may cause health problems, unable to export contaminated food
Dairy shed effluent	Increases nutrients and pathogens in soil and water	Possible run-off to water, leading to pathogen contamination issues
Vegetation removal and wetland drainage	Increase in erosion	Increased sedimentation in water; reduction in land productivity
	Reduced buffering capacity for floods	Increased risk of flooding
	Reduction in nutrient absorption	
Soil compaction	Increases sediment loss from land	See sediment impacts
	Increase in erosion	Loss in productive capacity of NZ; sediment impacts
	Decrease in land productivity	Economic loss
	Increase in flooding and water ponding on land	Requires more irrigation
	Increase in GHG emissions, nutrient loss	See impacts of GHG emissions and nutrients in water

10.1. Production and Inputs

Up until 2003, New Zealand was one of the lowest production cost dairy producing nations in the world (DairyNZ et al., 2009), in the sense that land was cheap and all farming was pasture based. Hence, there were no expenses of housing cows, and fewer inputs were used. This low-cost system generated a competitive advantage (DairyNZ et al., 2009). However, New Zealand dairying is moving away from low cost and input production by increasing the use of feed supplements (van der Nagel et al., 2003), fertilisers, and irrigation. Additionally, since that time, production costs and land prices in New Zealand have been increasing, while in other parts of the world, such as South and North American nations, as well as regions of Europe, Asia and Africa, transformation to lower cost dairy systems have occurred. This has resulted in a narrowing of this competitive advantage. Capital, skills and expertise from New Zealand are moving to other countries where production costs and land prices are lower. Uruguay is an example of this: NZ Farming Systems Uruguay has attained 36,300 hectares for dairy conversion (DairyNZ et al., 2009). Similar projects have also been implemented in other South America countries, Eastern Europe and parts of the United States (DairyNZ et al., 2009).

Although other countries are catching up to New Zealand's production, New Zealand is still the largest exporter of dairy products globally. In the past three decades dairy cow numbers in New Zealand have doubled and milk production has increased by almost 200% since 1990. On the other hand, dairying land area has only increased by 46%, indicating increased production per unit of land.

Notably, in 2008 the cost of many imports into New Zealand rose dramatically, most likely attributed to the Global Financial Crisis of 2007-2008. The estimated cost of fertiliser and palm kernel imports for use on dairy farms in 2008 was over $1 billion. Following this, imports of many fertilisers and palm kernel decreased in 2009, perhaps due to reduced demand because of a weakened economy. In 2012, the estimated cost of fertiliser and palm kernel imports for use on dairy farms were $500 million and $274 million respectively, totalling $774 million, a decrease from 2011 costs of $888 million.

10.1.1. Fertilisers

New Zealand dairy farmers use a proportionately greater amount of urea than other agricultural sectors. Seventy-two per cent of urea fertiliser and almost 70% of ammonium sulphate and potassic fertilisers are used on dairy farms. This substantial amount of fertiliser is being applied to around 6-10% of New Zealand's land area. The implications of this in dairy farming's contribution to nitrate leaching are, therefore, not surprising. Furthermore, in an international study on global environmental pressures, New Zealand ranked second highest out of 61 countries for nitrogen use per capita and the highest for phosphorus use per capita (Nykvist et al., 2013).

Phosphate

At present, phosphate is imported from the Middle East but recently proposals have been submitted for the mining of phosphate on the Chatham Rise to the east of New Zealand. The Chatham Rock Phosphate company currently holds an exploration licence and hopes to start production in 2016 to excavate 1.5 million tonnes of phosphate annually from the seabed from estimated reserves of 25 million tonnes (Chatham Rock Phosphate, 2013). The mine life is currently expected at 15 years but the company believes investigations may identify additional areas for mining that could extend mining up to 35 years (Chatham Rock Phosphate, 2013). Chatham Rock Phosphate is targeting its phosphate to supply New Zealand as well as potential export markets (Hartley, 2013). Reserves are estimated to be valued around NZ$6 billion (Moore, 2013).

Extractive costs for Chatham phosphate are expected to be lower than importing rock phosphate (Chatham Rock Phosphate, 2013). Rock phosphate has increased dramatically in price over recent years, from around US$40/million tonnes (Mt) in 2006 to a peak of US$430/Mt in 2008 (Mongabay, 2013). Prices then fell back to $90/Mt and they have been lingering around $US150-200/Mt over the past couple of years (Mongabay, 2013). According to the company, Chatham rock phosphate does not need chemical manufacturing before being applied to pasture and is more readily available to crops (Chatham Rock Phosphate, 2012). Another benefit is that cadmium levels of Chatham rock phosphates are among the lowest in the world (around 2 mg Cd/kg), while Moroccan phosphate has around 38 mg Cd/kg and Nauruan phosphate has cadmium levels higher than 100 mg/kg (Clovertone, 2013).

Mining more phosphate is a short term solution and involves depleting an ancient resource. Deposits of Chatham phosphate were formed 7 to 12 million years ago (Chatham Rock Phosphate, 2013). Furthermore, deep sea mining required for Chatham phosphate could have devastating effects on the ocean. About a metre deep of sand and silt will be excavated from the bottom of the sea, creating a large sediment plume (Moore, 2013). The fishing industry fears that mining will cause damage to fish nursery and breeding areas (Moore, 2013).

Phosphate reserves in New Zealand could influence future phosphorus imports. New Zealand would no longer be dependent on the global supply of phosphorus and likely rising prices. This could cause New Zealand to over-use the resource, increasing use past levels that are beneficial to production and lead to increasingly devastating effects in freshwater and coastal zones. Furthermore, New Zealand would be implicated in its own destruction of the sea bed to mine a resource that is globally wasted. Although importing phosphorus from other areas is unacceptable given the environmental effects and devastation that has occurred, New Zealand should aim to secure its future in agricultural uses less reliant on phosphorus inputs. The impacts of phosphorus mining, manufacturing, and use are externalities from the dairy industry. Mining impacts are usually at the expense of other nations, while the effects of phosphorus use are seen in New Zealand's lakes, rivers, streams and coastal areas.

Due to the threat of peak phosphorus, recovering phosphorus from the soil for re-use should have precedence over mining phosphorus. Biological solutions involve mobilising P bound to soils to make it available for plant uptake (Stutter et al., 2012), and removing phosphorus from waste streams for reuse (Shilton et al., 2012), among others (Shilton & Blank, 2012). These practices would see phosphorus used more efficiency and perhaps reduce the demand for mined phosphorus.

10.1.2. Palm kernel

The New Zealand dairy industry is a key player in purchasing palm products, accounting for a quarter of worldwide palm kernel expeller consumption. Other companies have responded to public pressure over concerns about palm production, including Cadbury and Nestle[22] (Greenpeace, 2010). Meanwhile, Fonterra and MPI continue to ignore concerns about PKE, justifying the increasing PKE imports to support dairy intensification.

The cost of palm kernel extract (PKE) has been increasing. International PKE prices have dramatically increased as demand has increased faster than expected. This demand is mainly from New Zealand. Prices have increased from US$80/tonne to nearly US$200/t (MacKinnon

[22] Cadbury stopped using palm oil in its chocolate in 2009 after consumer pressure. Nestle obtain palm oil from certified sustainable sources, although the reliability of the sustainability of certification has been questioned (see Chapter 8).

& Clark, 2012). In 2008, the increase in PKE imports was influenced by a major drought in New Zealand, estimated to cost the New Zealand economy $2.8 billion (Carter, 2009). Palm kernel is heavily used in drought conditions for emergency relief when normal supplies of supplementary feed are disrupted (Federated Farmers, 2011). The dairy industry argues that PKE is the only alternative (in intensive systems) during drought conditions (Greenpeace, 2009). Although local alternatives such as maize can be used, farmers are looking for cheap feeds to boast milk production and New Zealand crops are being undercut as they have had to compete with cheap palm kernel imports (Greenpeace, 2009). Consequences of PKE use experienced in New Zealand include biosecurity and health risks.

Importing PKE poses a severe biosecurity risk, perhaps as risky as the kiwifruit sector importing pollen (the pathway which PSA[23] entered the country) (Radio New Zealand, 2013). Biosecurity risks from importing palm kernel include imported insects, risks from soil contamination and foot and mouth, as well as food safety issues (Knight, 2009; Rural News, 2013). Two Federated Farmers members (Colin MacKinnon and David Clark) visited Malaysian palm factories in 2012 and observed birds, rodents, monkeys and cattle near processed PKE in an area of known foot and mouth outbreaks (MacKinnon & Clark, 2012). They report that post-production handling and storage of PKE poses severe risks and breaches New Zealand's Health Import Standards (HIS) (MacKinnon & Clark, 2012). HIS set out requirements that must be met to mitigate the risk associated with importing PKE. To eliminate risk from foot and mouth and for New Zealand biosecurity protection, products are required to be heat treated and stored in bird-proof facilities to avoid contamination (MacKinnon & Clark, 2012; Ministry for Primary Industries, 2013g). However, representatives from Federated Farmers were told that biosecurity was a not a matter for the Malaysian Government or palm industry and that it would be the responsibility of the New Zealand Government to ensure the product was not contaminated when it arrived at the border (MacKinnon & Clark, 2012). Furthermore, MacKinnon and Clark (2012) argued that contamination occurred during stages of storage and processes after PKE production, and heat treatment (which is carried out as part of the crushing process, not for biosecurity purposes) occurs weeks or even months prior to export. Therefore, heat treatment cannot be relied on for biosecurity protection as it is too early in the process.

[23] *Pseudomonas syringae pv actinidiae* (bacterial kiwifruit vine disease) – first found in NZ in 2010; the government contributed $25 million in its initial stages of outbreak to manage and continues to support the industry in research and management of the disease (Ministry for Primary Industries, 2012c).

MacKinnon and Clark (2012) explained that the majority of PKE is sold to commodity traders who consolidate products at the port for bulk export so there is little traceability to the factory where the products came from. This is inconsistent with industry claims that imported PKE is sustainably sourced and health standard approved as it is not known which plants imports come from. Vastly contrasting levels of contamination risk have been reported between palm kernel facilities (MacKinnon & Clark, 2012). While researching their report, the two Federated Farmers representatives visited two palm kernel facilities. First, they were escorted to Malaysia's most modern facility which was clean and capable of meeting New Zealand's HIS (MacKinnon & Clark, 2012). This plant was the only RSPO certified kernel crusher in Malaysia and products were not on-sold to commodity traders as the plant sold its own supply; however, the plant manager claimed the plant did not sell to New Zealand (MacKinnon & Clark, 2012). MacKinnon and Clark also made a visit to another crushing plant, chosen because it was in an area where foot and mouth disease had been officially reported. The second plant did not have rodent control and animals were observed in the immediate vicinity of the crushing plant. Decaying PKE was present inside and outside the building and fungal infections were routinely experienced. This plant would not have met New Zealand HIS but PKE from this plant could be exported to New Zealand (MacKinnon & Clark, 2012). Biosecurity and food safety requirements were the responsibility of the commodity trader and not of the plant itself (MacKinnon & Clark, 2012).

In response to MacKinnon and Clark's report, MPI met with Malaysian officials in 2013 to ensure there is a full understanding of New Zealand's HIS (Rural News, 2013). Malaysian officials confirmed with MPI that no PKE had been exported to New Zealand from the inadequate processing mill identified by MacKinnon and Clark (Rural News, 2013). Additionally, MPI recommended that some facilities needed to improve systems for the manufacturing and storage of PKE to adhere to New Zealand biosecurity requirements, and agricultural departments in Malaysia and Indonesia must ensure that PKE from unapproved facilities cannot be exported to New Zealand (Ministry for Primary Industries, 2013d, 2013e). A strengthening of the HIS for animal feeds regarding PKE was also recommended (Ministry for Primary Industries, 2013e). Judging by the MPI recommendations, there were facilities not up to New Zealand's standards.

Health risks have also been identified in feeding PKE to dairy cows. Firstly, overfeeding palm kernel to pre-calving cows can cause milk fever, a disorder where cows lack sufficient calcium to maintain milk production and induces muscle functioning loss. This is caused by the high phosphorus content of palm kernel (Country TV, 2010). High levels of phosphorus reduce the effectiveness of vitamin D in assisting the absorption of calcium (Country TV, 2010). Spraying effluent on paddocks can also be a potential risk source for milk fever as well as feeding silage or hay crops made from paddocks that receive large amounts of effluent (Country TV, 2010). Another health risk from PKE is the change in milk fat content that occurs when cows are fed PKE or other grain foods (Morgan, 2013a). Omega 3 that is found in milk from grass fed cows changes to less healthy palmitic fatty acid and even trans fats (which are banned in many European countries) (Morgan, 2013a).

A confidential report from AgResearch in 2006 warned that deadly toxins growing in PKE could pass into the food chain, presenting significant risks to animal and human health (Ball, 2013). The hazardous mycotoxin-producing fungi and mycotoxins are invisible, so farmers cannot identify if PKE is contaminated. However, the risks of feeding visibly mouldy material are known. The report identified inadequate storage methods were responsible for creating mouldy and spoiled feed from moisture. Many storage facilities in New Zealand are still exposed to moisture, and facilities in Malaysia are even worse (MacKinnon & Clark, 2012). Aflatoxin blood and fungal infection was identified as a problem from storage problems (Ball, 2013). Although at the time of the AgResearch report the toxin hadn't yet been detected in food safety tests, since then, PKE imports have increased substantially – from less than 250,000 tonnes to around 1.4 million tonnes annually. Little information of this has reached the public and there are conflicting opinions on whether a real risk is posed. If milk were to be contaminated, export restrictions on dairy products could be imposed.

The dairy industry argues that PKE is only one per cent of what dairy cows are fed (Greenpeace, 2010). For such a small amount, it begs the question why New Zealand imports a product with such high environmental, social, biosecurity and health implications. One postulation is that farmers are pressured to produce more milk just to maintain profits (Morgan, 2013a). New Zealand farmers have to complete globally to produce increasing amounts of milk powder for growing developing nations. Farmers must also meet Fonterra's growth production goal of 4% per annum (Deans & Hackwell, 2008). Conversely, the more dependent on milk volumes the dairy industry becomes, the more vulnerable it becomes to risks and the quality and quantity of milk volumes (Morgan, 2013b). To produce more from the same amount of land, farmers become dependent on inputs such as supplementary feed and fertilisers. If the impacts of the inputs are not questioned or acknowledged, and hence not implicated in the products image, use of inputs will continue.

10.2. Externalities and Management

The failure of the current dominant global economic system to incorporate all factors of production, manufacturing and consumption has resulted in externalities. Currently, in this system polluters generally do not have to pay for most of the pollution they emit. If they are able to increase production while escaping responsibility of the outside effects, they will do so. Thus, the current process of economic accounting inevitably leads to environmental degradation. Management practices aimed to aid development in a sustainable way do little to alleviate market failures and are often politically charged. Processes that could help, such as effective regional plans; national standards and policy statements; stronger regulation; and economic measures to encourage less pollution; are slow to occur and often extremely weakened when finally implemented. Co-management agreements have had more progress at forming strategies and visions for environmental clean-ups, but have too been slow to implement and report on successful change. A mix of different policy instruments is required to effectively control diffuse pollution. Policy examples have been provided for controlling diffuse pollution in the Rotorua Lake catchments (see Appendix P).

Dairy intensification has occurred without adequate control of the impacts. Non-point source pollution is still largely ignored by management regimes, despite the knowledge that most nutrients that end up in waterways are from diffuse sources. Since regulation fails to manage impacts, economic or voluntary management initiatives should be stronger to address the problems that regulation omits. However, this is generally not the case. The scope of voluntary schemes in New Zealand aiming to moderate dairy impacts is limited to the Dairying and Cleans Streams Accord (now the Sustainable Dairying Water Accord), which has largely failed in its aims and barely monitors results, so successes are unknown. Economic initiatives, such as emissions trading and nutrient cap and trade schemes, have been slow to start and struggled to get farmers to take up. Nutrient caps have only been implemented in nationally iconic catchments such as Lake Taupo and the Rotorua Lakes and the success of these is not yet known. Furthermore, targeted reductions may not even be enough to reduce nutrients to levels required for healthy ecosystems.

Another failure is that the New Zealand Emissions Trading Scheme (ETS) for GHG emissions excludes agriculture. Thus, it provides no incentive to reduce emissions. There is currently no date for when agricultural will pay for emissions under the ETS (Ministry for Primary Industries, 2013c), after been pushed back from previous implementation dates. When, or if, agricultural emissions are brought into the ETS, 90% of costs will be provided by government assistance (Ministry for the Environment, 2012a). Dairy farming emits around 25% of New Zealand's GHG emissions, but is exempt from paying for emissions, effectively receiving a tax-payer subsidy to pollute. Emissions or nutrient reduction schemes should be targeted to the sectors responsible, so costs are not detached from polluters.

Subsidies often encourage activities that damage the environment. Official agricultural subsidies were removed in New Zealand in the 1980s (MacLeod & Moller, 2006), albeit unofficial subsidies now exist. Essentially, every environmental externality emitted by an industry (such as dairy farming) is a subsidy received by the industry for polluting or for failing to mitigate or remedy impacts. The subsidies are paid for by the rest of New Zealand, in the form of lost recreational, aesthetic and cultural areas; lack of water resources for other land-uses; loss of biodiversity and ecological integrity; or in the form of compensating remediation programmes, irrigation schemes, and research for farm mitigation technologies.

Other management failures involve the methods used to monitor the state of the environment and the varying parameters used between regions to assess environmental health, many of which are inadequate at accurately evaluating environmental condition. Furthermore, the drivers and causes of environmental degradation are not always identified or are attributed to broad-scale practices. For example, dairy farming land is categorised with pasture, comprising largely of sheep and beef farms. Hence, pressures from pasture land are averaged over the entire area of pasture, hiding the much larger influence in ecosystem degradation dairy has over sheep and beef farming. For instance, average national nitrogen leaching rates for dairy have been measured at 28 kg N/ha/yr, while average rates for sheep and beef farms are 11 kg N/ha/yr, and the national average is 8 kg N/ha/yr (Ledgard et al., 2000). In this regard, sheep and beef farming are taking a large part of the blame for pastoral impacts, when dairy is largely responsible.

Another issue is the amalgamation of impact and reference sites for water quality with the National River Water Quality Network (NRWQN). Reference sites are located in areas that have not been influenced by human land-use, usually native forest; whereas impact sites are situated in areas impacted by humans, such as urban and agricultural land. Water quality is correlated with surrounding land use, so obviously water quality in these two different areas are expected to differ remarkably. Regardless, sites are commonly averaged, providing an overall indication of water quality and masking any differences between land uses.

10.3. Water

Freshwater in New Zealand undoubtedly experiences the worst impacts from dairy farming. This in part is a failure of adequate environmental management and protection of freshwater (explained above). Furthermore, poor monitoring and measurement of the state of freshwater masks the severity of the problem. In terms of water use for dairy farming, stock water takes are classed as permitted activities so do not require consent and are not monitored. Hence actual water use is probably under-estimated. Irrigation is increasing dairying in drier areas and further exacerbates water quality impacts.

10.3.1. Measuring freshwater in New Zealand

National freshwater State of Environment (SOE) monitoring uses a limited number of parameters to define “water quality”. Rivers, lakes and groundwater are measured as well as recreational sites for the suitability for swimming (Ministry for the Environment, 2013b). The river condition indicator measures nutrients (nitrogen and phosphorus), bacteria (*E.coli*) and very rarely macroinvertebrate community condition. The indicator for lake water quality is based on two indices: the Lake Tropic Level Index (TLI) and the Lake Submerged Plant Indicators (LakeSPI). The TLI measures the nutrient (trophic) status of lakes, visual clarity and algal biomass, while Lake SPI looks at the native and invasive character of vegetation in a lake. In groundwater, nitrate and *E.coli* concentrations are measured, and finally the suitability for swimming indicator measures bacteria levels at recreational sites.

In reality, this suite of assessment measures does not give a reliable measure of waterway condition and seems more related to the simplicity of sampling. Firstly, the freshwater monitoring tools used do not measure functional impairment or habitat quality and only macroinvertebrates are measured for biodiversity (if they are measured at all). Functional indicators measure ecosystem processes and evidence suggests they respond to land-use pressures (Collier et al., 2009). Functional indicators that could be measured in river ecosystems to provide a useful measure of ecosystem health are ecosystem metabolism and organic matter breakdown (Collier et al., 2009; Young, 2009). Ecosystem metabolism measures the life-supporting capacity of a river by measuring changes in dissolved oxygen (DO) concentrations over at least a 24-hour period (Clapcott & Young, 2009; Collier et al., 2009; Young, 2009). Ecosystem metabolism is influenced by factors such as nutrient inputs, organic waste discharges, shading, water temperature and river flow (Clapcott & Young, 2009; Young, 2009). Conversely, Death, Dewson and James (2009) found that structural (benthic invertebrates) measures of ecosystem integrity gave a better indication of physiochemical changes caused by water abstraction than functional measures. Additionally, measuring functional indicators requires greater time and monetary investment (Death et al., 2009). However, not all functional measures were considered in this analysis. Another measure that could be included is the alteration of habitat from land use activities affecting biological characteristics of aquatic life (Joy, In Press)[24].

Another failure of MfE monitoring is that parameters are measured as a one-off 'snap-shot' sample, when in actual fact attributes become more variable with the accumulation of impacts in freshwater systems (Joy, In Press). For example, oxygen levels typically fluctuate diurnally due to algal photosynthesis, and these fluctuations become greater as nutrient levels increase (Clapcott & Young, 2009). One off snap-shots of DO levels do not give an indication of the extremes between night and day (Clapcott & Young, 2009; Joy, In Press). For example below an intensively farmed dairy catchment on the Manawatu River (Hopelands), DO levels varied from less than 40% in the morning to more than 140% in the late afternoon on a single day (Joy, In Press). Healthy DO levels for ecosystems are around 98-105% saturation (Wilcock et al., 2007) and the lowest levels should be around 60-95% saturation (Wilcock et al., 2011). The guideline for the protection of aquatic ecosystems, fisheries and fish spawning is >80% saturation (Wilcock et al., 2007).

Extreme fluctuations in oxygen availability are potentially lethal for all stream life, but variability is missed by 'snap-shot' sampling and so these conditions are not apparent to resource managers (Joy, In Press). Other measures, such as nutrient levels, temperature, pH and suspended sediment, also vary with flow and biological in-stream processes (Joy, In Press). Regardless, recent freshwater changes mean that many of these aren't even measured on a national scale anymore (Ministry for the Environment, 2013a). Moreover, Regional Councils measure different things in their region so comparisons between regions become difficult.

10.3.2. Water use

In some areas the total water takes may exceed the allocated amount, since water metering is not mandatory in many places (Parliamentary Commissioner for the Environment, 2004). Furthermore, councils do not usually have adequate knowledge and information about groundwater and surface water resources, generating problems for setting rules in plans and

[24] For more information on measures of ecosystem health and water quality see Collier (2009).

granting resource consents. If long term records are not available, it can be difficult to set minimum flows for rivers and determine what takes are sustainable for healthy ecosystem functioning (Parliamentary Commissioner for the Environment, 2004). Consents are usually issued for a long period of time and once a water body has been fully allocated, it is difficult for new users to gain access to that resource (McGregor, 2007). Additionally, consents are generally not tradable; therefore, in order to secure future rights in some high demand catchments, users over-consume water (OECD, 2007). Enhanced transferability between water uses could reduce pressure on water permit holders and the environment by reducing pressure to allocate access to water (Hawke, 2006). This would facilitate more efficient sharing of available water and increasing flexibility of water management (Hawke, 2006).

Water allocation in New Zealand is assigned by a first-in-first-served basis, providing an incentive to over use (McGregor, 2007; Tait et al., 2008). This can lead to situations where potentially more valuable uses are unable to gain access to a fully allocated resource (McGregor, 2007). Additionally, water use is generally not priced (York, 2011), so there is little encouragement to minimise water use, leading to inefficient use. Consents for water use are tied to land so effectively the price of land should capture the value of water (Tait et al., 2008).

Permitted uses

Permitted uses for dairy farming include drinking water for stock, and dairy shed wash down. In the Waikato region, permitted uses are allowed up to 15 m^3 per day; however, permitted water use is not monitored so farmers may be exceeding this take. Only two water meter records for stock takes are available in two regions; therefore, for the most part there are no records available of water takes (Rajanayaka et al., 2010). Dairy cows require about 70 litres per cow per day so farms with more than 215 cows would surpass the permitted volume.

Continued dairy intensification will increase the amount of permitted animal drinking water allowed due to the high priority it is afforded under the RMA. This could result in nearly all of the allocable flow being used for this purpose in some catchments (Brown et al., 2007). For instance, in the Piako catchment (Waikato), already 100% of the allocable flow is used as a result of the high density of dairy cows in this catchment and large water requirements. Whereas, high proportions of use in sub-catchments of the Waipa (Waikato) are used due to the very low level of allocable flow (Brown et al., 2007).

The high level of permitted and animal drinking water use in many catchments in the Waikato region pose a limitation of the RMA allowing animal drinking water as a permitted activity without consideration of the activities linked directly with it, such as dairy shed operations. The combination of permitted and consented takes increases water used in the region. For example, there were 70 catchments in the Waikato region with more than 50% of the allocable flow taken, of which 41 had more than 100% taken (Brown et al., 2007). Limiting cow numbers per catchment that can receive water under permitted allocations could be a solution to lower water abstractions, essentially limiting intensification in some areas, and could also help secure supply and protect minimum flows (Brown et al., 2007). Moreover, the permitted take threshold of 15 m^3 per day may have to be lowered to ensure the current volume of permitted water takes remain within the allocable flow. In some catchments it may even have to be zero. In this instance resource consents would be required for previously permitted activities. During critical times, conditions could stop water being taken for dairy shed operations to maintain minimum stream flows (Brown et al., 2007).

10.4. Land

Present land-based impacts from dairy are mainly soil compaction and contamination, affecting soil physical properties and land security. Impacts on soil threaten food safety, the production potential of land, future land-uses and land-use flexibility, and exports which could be potentially contaminated. Soil physical quality is most affected on dairy land; around 70% of monitored dairy sites do not meet at least one soil quality target – the highest among various land uses. Compaction affects around half of dairy land in New Zealand, having substantial potential to affect land productivity and involving costly remediation.

High nitrogen fertiliser use and increasing stocking rates are increasing the N content in soil with around half of dairy sites exceeding total N upper targets. Likewise, the use of phosphate fertiliser increases soil fertility, but is being over-applied and thus wasted. When too much phosphorus is applied, the surplus P can easily run-off to water. Additionally, the use of phosphate fertiliser has elevated contaminant levels in soils. Cadmium is of main concern due to the large amount of land affected. In the Waikato region alone, at least 157,000 ha of dairy land has cadmium levels above recommended guidelines. Moreover, most agricultural land is not tested for cadmium so the issue is supressed. Currently, there is little indication of the full extent of the cadmium problem, presenting huge implications for future land uses. Cadmium contamination of soil can limit future land use options e.g. growing cereals and vegetables for human consumption (Mackay, 2008), and changing to residential zones. This is because cadmium has a tendency to accumulate in plants, hence entering the food chain. Despite the risks of cadmium contamination, New Zealand has no specific regulations on allowable cadmium concentrations in soils. Already food safety issues are of concern with cadmium being the highest contaminant to reach its recommended daily intake. Other contaminants associated with phosphate fertiliser are uranium and fluorine. Very little testing has been done on these two so the area affected is unknown. Furthermore, uranium does not have a guideline value in soil, despite being a radioactive pollutant. The future costs of soil contamination may be substantial if not addressed. Export revenue may fall if food grown exceeds international standards for contaminant concentrations. There has been a lack of effective response to the issue of contamination on agricultural soils, particularly from cadmium. For instance, the National Environmental Standard for contaminants in soil excludes production land (agricultural land) as being classed as contaminated (New Zealand Government, 2011b). Hence, even if agricultural land surpassed contaminant standards, it would not be classed as contaminated, having implications for future land users.

10.5. Atmosphere

The increase in agricultural emissions from 1990 levels has been driven by several factors: GHG emitted per animal increasing as animals consume more supplements; N fertiliser use increasing dramatically over the past two decades (Leslie et al., 2008); and increasing dairy cow numbers. Nitrogenous fertiliser use in the dairy industry increased form 59,000 tonnes in 1990 to almost 390,000 tonnes in 2012, causing large increases in nitrous oxide emissions (Li, 2005; Statistics New Zealand, 2013a). Additionally, increased use of irrigation in drier areas has resulted in increasing energy emissions and environmental impacts driven by irrigation.

Feed supplements such as total mixed rations (TMR) doubles or triples cow milk production (van der Nagel et al., 2003). TMR comprise of grains, silages and protein or energy supplements, which require annual cultivation, high fertiliser inputs and substantial energy requirements for tillage, harvesting, transport and feeding. van der Nagel et al. (2003) found that methane emissions increased by 58% when cows were fed TMR (van der Nagel et al., 2003). The increasing use of PKE is also associated with increased emissions, through cows consuming more feed and emissions from PKE production.

The exclusion of agriculture from the ETS creates another externality of dairy farming. Academic institutions are researching methods to decrease biological emissions but neither farmers nor the dairy industry have to pay for emissions, incentivising increasing production. The Government invests over $20 million annually into research to reduce biological agricultural emissions and another $45 million (until June 2016) is used to support the Global Research Alliance, mostly used on research to reduce livestock emissions (Ministry for Primary Industries, 2013c). However, recent research has found methods to quantify the methane emissions from anaerobic ponds are incorrect, thus national emissions have been under-estimated (Chung et al., 2013). This is concerning as GHG mitigation strategies have largely focussed on enteric methane emissions, which are costly and difficult to reduce, while technologies to mitigate methane from ponds are reasonably easy to implement (Chung et al., 2013). Because agricultural sectors in other countries do not have to pay for their emissions, the Government does not wish to impose this cost onto New Zealand farmers (Ministry for Primary Industries, 2013c). However, with New Zealand's unique make-up of GHG emissions (mostly from agriculture), the industry most responsible should be paying.

Fonterra coal mine

Another environmental liability from dairy is coal mining. Out of eight Fonterra South Island processing plants, seven are powered by coal and one by fuel oil. Three North Island Fonterra plants burn coal and the remaining 16 rely on gas (Oram, 2013). Fonterra is proposing to open a new coal mine in Waikato to fuel its processing plants. Glencoal, a subsidiary of Fonterra lodged consents to develop an opencast coalmine at Mangatawhiri, in North Waikato (National Business Review, 2013). It is expected 120,000 tonnes of coal a year will be extracted if it goes ahead (National Business Review, 2013). The proposed mine would be close to schools, homes and roads and visible from state Highway 2 (Chisnall, 2013). Fonterra say it's cheaper to have their own coal mine than to buy coal (Chisnall, 2013).

10.6. Clean Green Image

Tourism New Zealand claims that "100% Pure is not an environmental statement or promise and never has been" (Linklater, 2013) but is more about the whole 'unique' experience of New Zealand (Anderson, 2012; Cumming, 2010; Linklater, 2013; Stewart, 2013). However, the "100% Pure" slogan has been widely interpreted differently; New Zealand is perceived as a clean green country and many overseas buyers purchase New Zealand's products based on this assumption (Ministry for the Environment, 2001b). Surveyed international visitors in 2011/12 rated New Zealand 9 out of 10 for its natural environment (Stewart, 2012). However, there is a case of two worlds in New Zealand; international tourists visit national parks, where the environment is pristine. As a result, visitors attain an impression that New Zealand is '100% Pure'. Conversely, visitors seemingly do not often visit the lowland rivers that are more accessible to locals but inaccessible for contact recreation because of degradation.

Moreover, some studies, such as Yale's Environmental Performance Index (EPI), show New Zealand performing well environmentally (Emerson et al., 2010; Emerson et al., 2012). However, what this study does not measure are some of the things New Zealand does not perform well in, such as non-CO_2 emissions, proportion of threatened species, and nutrient use. Water quality isn't even measured in the EPI anymore because of the failure to properly represent the state of freshwater in each country. If these things are measured, New Zealand ranks among the worst in the world environmentally for some indicators, (for example Bradshaw et al., 2010; Nykvist et al., 2013). The various results between studies show the danger in ranking countries on an environmental scale as performance is dependent on what is measured. However, if New Zealand was among the best in the world environmentally, it would surely perform well in all environmental indicators. Therefore, the reality of New Zealand's environment is far from the 'clean green' or '100% Pure' perception and dairy farming impacts all contribute to the failing of New Zealand's environmental image.

The deterioration of New Zealand's environment over the past few decades has challenged the 'clean green' image and led to New Zealand failing quite dismally in some international benchmarks. Although all land-use practices have led to this degradation, dairy farming contributes a significantly larger proportion. Measures that New Zealand is failing at include:

- Threatened species – in one study New Zealand was ranked the worst out of 179 countries for the proportion of threatened species (Bradshaw et al., 2010). In another New Zealand was the 6th highest out of 61 countries for species threats per capita (Nykvist et al., 2013).
- Fertiliser use – New Zealand was the 2nd highest N user on territorial performance on per capita boundary and the highest per capita user of phosphorus out of 61 countries (Nykvist et al., 2013).

- New Zealand was ranked 18th worst out of 189 countries by combined proportional environmental metrics, such as natural forest loss, natural habitat conversion, fertiliser use, water pollution, threatened species and carbon emissions (Bradshaw et al., 2010).
- New Zealand has some of the highest rates of campylobacteriosis in the developed world (OECD, 2007). By 2011, New Zealand had double the rate of Australia and 12 times the rate of US for waterborne diseases (Daily Mail, 2013).

Poor management and use of policy instruments in New Zealand are contributing to these failures by:

- Being the only OECD country that does not have a legislative requirement to produce a state of the environment report (Wright, 2011). Proposals have been announced to implement this requirement but would be implemented by the Government, not an independent entity like the Parliamentary Commissioner for the Environment (PCE) recommended.
- Not signing up to the 2nd commitment period of the Kyoto Protocol, being one of the few countries out of 200 signatories to pull out.
- Despite having world leading environmental legislation, there is a lack of regulation on diffuse pollution and intensive land uses do not require consents.

These deficiencies lead to a loss in the value of New Zealand's 'clean green' image and perception as an environmental leader, thus a reduction in value of New Zealand primary products and tourism.

10.7. Costs of Dairy Farming

The costs of dairy farming not only include direct financial implications for mitigating and remedying impacts, but also that of the loss of recreational areas, areas of cultural significance and aesthetic attributes of the environment. Not only do these affect present generations but impacts are likely to proliferate in the future, generating an uncertain environment for future generations. This could accelerate to the extent that future generations are unable to provide for their basic needs, a key requirement of the RMA (New Zealand Government, 1991) and the international Rio Declaration on Environment and Development initiated in 1992 (United Nations, 1992). Moreover, many of the economic costs are likely to fall on future generations, as costs are being currently off-set or deferred.

At a conservative estimate, costs of some environmental externalities from dairy (those included in Chapter 9), imported products and interest payments on debt for 2012 were between $6.8 billion and $19.8 billion. Omitting local costs that had not been estimated for all of New Zealand (such as costs for cleaning up the Rotorua Lakes and Waikato River), generates a cost between $6.53 billion and $18.23 billion (national annual cost) (Table 10.2). However, this estimate excludes valuations of many of the core impacts because they are unavailable.

Two valuations of the value of the dairy industry were estimated: one only included environmental costs that had been valued on a national scale annually and one included all environmental costs that had been estimated, whether they were on a national or local scale and whether they were annual costs, periodic costs or one-off costs. Import costs and interest payments on debt were included in both scenarios. Costs were detracted from the 2012 dairy export revenue. For the first scenario (national annual cost), the value of dairy products was estimated between $5.10 billion and a loss of $6.60 billion (Table 10.2). When all environmental costs were included, the value decreased to between $4.79 billion and a loss of $8.21 billion. This is a concern considering dairy farming could be costing the country billions of dollars through environmental degradation, with economic costs likely to fall on future generations.

Table 10.2: Summary of some national production costs excluded from dairy revenue

Measure	Scale	Cost ($ Million) Min Lower	Cost ($ Million) Min Upper[1]
Environmental remediation and mitigation (from Table 9.8)	Costs from national annual environmental impacts on water, land, GHG emissions and clean green image	2757	14450
Imports	Fertiliser imports for dairy in 2012	502.9	502.9
	Palm kernel imports in 2012	274	274
Interest payments on debt	Estimated payments on dairy farming debt in 2012	3000	3000
	National Total cost[2] ($ Billion)	**6.53**	**18.23**
	Revenue from 2012 dairy exports[3] ($ Billion)	**11.63**	**11.63**
	Difference ($ Billion)	**5.10**	**-6.60**

Notes: [1]The minimum upper limit is the upper value for which value assessments were undertaken. The total is still regarded as conservative considering all the valuations that were not included.
[2] Excludes local one-off costs that have not been extrapolated out to the rest of New Zealand.
[3]Other benefits that are not included with the revenue include jobs and direct and indirect contributions to other sectors in the economy.

10.8. Farm Management Practices

Dairy farm management practices can reduce pollution at the source. These solutions are undoubtedly more cost-effective than expensive remediation techniques which generally involve on-going costs and maintenance. Costs would be borne by farmers rather than the public but some funding could be provided by Government to encourage change in management practices. Many techniques involve reducing nutrient and pathogen loss to water, including using lower protein diets, wintering systems, stream fencing and riparian areas, and simply reducing the amount of nutrients applied to the land. Some of these will be briefly discussed here to provide an indication of costs. To reduce the effects of other impacts requires relatively straight-forward practices, such as techniques to avoid soil compaction and contamination (discussed in Chapter 6). Greenhouse gas emissions are harder to avoid; research has investigated reducing biological agricultural emissions but decreasing intensification may be more effective at reducing emissions.

10.8.1. Nutrient/effluent management

Reducing nutrient loads at source is ideal before water-quality starts to decline. Often mitigation measures are advocated to reduce leaching while continuing current production or intensifying. However, these reductions are unlikely to make up for the increases in losses that may occur under future intensification (de Klein et al., 2010). More likely, these measures only reduce the N losses per unit of milk produced (de Klein et al., 2010), making milk production more efficient, but not reducing environmental impacts. To minimise impacts on freshwaters, a reduction in the total N leaching losses is required, rather than reducing N leaching intensity (de Klein et al., 2010).

In the Rotorua catchment reducing N use fertiliser use had the biggest effect of reducing N leaching rates (Ledgard et al., 2010). Large reductions in GHG emissions per hectare (around 31%) and N leaching (47%) have been recorded from halting N fertiliser use (Ledgard et al., 2010). Cutting N fertiliser on 6 dairy farms in the Rotorua catchment was estimated to reduce returns by $46 to $428/ha/yr and the average loss in gross revenue per ha was $173 (Ledgard et al., 2010). This reduced N leaching by 4-57 kg N/ha/yr and GHG emissions by 0.2-2.1 kg CO_2-e/kg MS (Ledgard et al., 2010). The average reduction in N leaching over the 6 six farms was 26 kg N/ha/yr (Ledgard et al., 2010), yielding a reduction in gross margin of $6.62/kg N or

$6620/tonne N. Conversely, removing nitrogen from lakes was estimated between $14,000 - $4 million/tonne N (2-600 times as costly). Hence, it is obvious where focus should be concentrated. It is also much cheaper to purchase urea fertiliser, at around $641/tonne, than it is to remove once it has leached to water. Farmers could initially be subsidised for the lost revenue until production systems stabilise. Similarly, removing phosphorus from lakes has been estimated to cost around $165,000 - $250,000/tonne P and possibly much more, while phosphorus fertiliser costs around $250/tonne P ($0.25/kg). Obviously, it is substantially cheaper to apply phosphorus than to remove it; hence, focus should be on recycling phosphorus instead of wasting it.

Lower protein diets

New Zealand's pastural dairy cows consume around 26% crude protein annually while the requirement is only around 16% (Dewes, 2012). The surplus protein in the diet is excreted as urea in urine, which then leaches to groundwater. Low protein feeds can enhance rumen efficiency leading to lower urea production. Using cereals of about 8-9% crude protein, combined with pasture, will reduce the amount of urea in urine by up to around 40% (Dewes, 2012). Additionally, modelling suggests the use of low protein supplementary feeds combined with a range of mitigation techniques would not significantly harm profit, while reducing leaching (refer to Dewes (2012) for more information).

Wintering systems

Dairy cows are often wintered off-farm for a few months of the year, usually on neighbouring dry stock farms. Stocking rates are much higher in these areas and plant uptake of N is lower; hence, high rates of N leaching occur during this period (de Klein et al., 2010; Monaghan et al., 2007). For example, wintering areas in the Bog Burn catchment (Southland) produce 60% of the total dairy system N leaching, despite only representing 15% of the dairy area (Monaghan et al., 2007). Despite this, these areas are usually not targeted for nutrient reduction schemes or management, even though low-emission wintering systems such as feed-pad operations are currently available (Monaghan et al., 2007).

A loafing pad or standoff is an area with no feeding facilities in which seepage is captured and stored in an effluent pond. It is used to stand cows for part of the day to reduce trampling damage to wet paddocks and minimise the amount of excreta and urine being deposited onto pastures during critical times, such as late summer and autumn (Beukes et al., 2012; de Klein et al., 2010; Longhurst & Smeaton, 2008). Pads may reduce N leaching losses by up to 25%

(Ledgard et al., 2006). Feed-pads generally do the same thing but may be used for longer periods as they provide feeding facilities. Generally, pasture grazing is also used with these systems on a rotational basis. A downside to these structures is the high capital and operational costs required for construction (Longhurst & Smeaton, 2008). Animal welfare issues could also be an issue. Furthermore, this technique encourages maintaining production levels, i.e. they do not require a reduction in stocking rates or fertiliser use. Systems such as these may even encourage farmers to increase stocking rates in order to increase production. Additionally, more supplementary feed is used in feedlots, associated with increased leaching from areas used to grow feeds.

Restricted and nil grazing avoid deposition of urine during autumn/winter and require the use of a standoff pad, feed-pad or animal shelter. They have been found to reduce nitrate losses by 35-50% (restricted) to 55-65% (nil) (de Klein & Ledgard, 2001). A restricted grazing system for the average New Zealand dairy farm is likely to be economically viable on farms where an effluent application system or feed pad is already in place (de Klein, 2001). Costs of different wintering systems range between \$22/cow/week to \$38/cow/week (Table 10.3). For the average New Zealand herd size of 393 cows (LIC & DairyNZ, 2012), costs would be \$8646/week to \$14,934/week or between \$73,491 and \$126,969 for a wintering period of 60 days. Average stocking rates on dairy farms are 2.83 cows/ha so costs would equate to \$25,696 to \$44,865 per ha. Although expensive, costs can be reduced if effluent management systems are already in place (in the case of many farms). These costs are not comparable to removing nitrogen as they were estimated per ha rather than per tonne of nitrogen.

Table 10.3: Costs of wintering options on dairy farms

Per 200 cows – consuming 8 kg DM/cow/day	**Herd home**	**Feed pad**	**Standoff pad**	**Grazing off farm**
Construction capital costs (\$/cow)	1,350	319	184	0
Effluent management capital costs – (\$/cow)	0[1]	96	71	0
Total capital costs – (\$/cow)	1,350	415	255	0
Total operational costs (\$/cow)	35	88	69	0
Total costs – (Capital and operating) – wintering for 60 days (\$/cow/week)	38	33	27	22

Data source: Longhurst & Smeaton (2008, Table 9). **Notes:** [1]Effluent bunker system for herd home is part of capital construction costs. The total costs in the bottom row include the costs of the feed supplied to the herd home and self-feeding pad. The value for grazing off does not include a credit for time saved by the farmer not having to shift cows while they are grazed off.

Stream fencing and riparian areas

The riparian area is the zone between aquatic and terrestrial ecosystems and riparian vegetation is a distinct assemblage of plants uniquely suited to this zone. The health and conditions of waterways can be significantly affected by the plants and animals occurring along the riparian margin (Ministry for the Environment, 2001a). Riparian strips should be at least 10 metres wide as narrower areas are generally dominated by exotic brushweeds and require additional management (Ministry for the Environment, 2001a). Fencing is aimed to exclude all stock from waterways, to benefit the stream and the farm.

Riparian vegetation and wetlands reduce nutrient loads and faecal matter running to surface water. However, targeting source reduction rather than transport interception of farm water pollutants may be more successful (Monaghan et al., 2007). Furthermore, riparian areas generally only target phosphorus runoff and do not reduce nitrogen leaching. Vegetated functional riparian areas provide benefits to maintain or improve water quality by (Ministry for the Environment, 2001a):

- Reducing land-based impacts on waterways through: reducing erosion by slowing overland flow; filtering overland flow containing nutrients, soil, microbes and agriculture chemicals; and utilising nutrients for plant growth before they enter streams.
- Reducing or buffering the impact of water processes on adjacent land by: protecting banks from erosion; buffering the impacts of floods; and buffering channels from morphology changes.
- Promoting and sustaining in-stream plants and animals by: reducing fine sediment; maintaining water clarity; providing in-stream food supplies and habitat; preventing nuisance plant growths; maintaining lower summer temperatures; reducing light levels; and maintaining natural food webs.

These benefits are greatly reduced in agricultural waterways with little or no riparian vegetation (Ministry for the Environment, 2001a).

Fencing can be beneficial by restricting stock access to waterways. The costs of fencing stream banks in three prominent dairying regions have been estimated (Cullen et al., 2006). The length of stream banks in dairy farms in Taranaki is 16,000 km, Manawatu 2800 km and Waikato 583.8 km. The length remaining to be fenced is at least 10,512 km[25] (Cullen et al.,

[25] The length remaining may have reduced since the time of this estimate.

2006). Estimates suggest hot-wire fencing costs approximately $1550 per kilometre to erect. In these three regions fencing off the remaining stream banks would cost around $16.3 million (Cullen et al., 2006), which could be funded through clean-up funds or by rate-payers within a short time period.

10.8.2. Managing sediment loss

Erosion-prone areas can be retired for practical and economic benefits, such as reducing sediment loss from land. Fences can prevent stock access to certain areas. Problems that could be incurred by fencing are weed control, flood damage to fences, and fencing in steep country with meandering streams (Ritchie, 2011). Measuring costs of implementing actions to reduce sediment in the Waikato River catchment from dairy farms include: (Ritchie, 2011):

- Run-off diversion (from laneways): $5 million
- Creating wetlands over 1% of the catchment: $45 million
- Fencing and planting 5m buffers on all streams: $263 million
- Herd shelters for wintering stock: $1090 million.

Costs for retiring areas and riparian planting include the cost of fencing, installing alternative stock water and any additional planting. These practices have other benefits, such as reducing nutrient losses and effluent management.

10.9. Conclusion

Although detailed cost assessments have not been carried out for management techniques aimed to reduce dairy farming impacts (beyond the scope of this thesis), from preliminary investigation (and from evidence in other studies) it is likely the cost to clean up effects will be far more than the costs of not polluting in the first place. This analysis omits many of the costs involved, but the primary valuations here should act as a precaution for the dairy industry and environmental regulatory managers in New Zealand to put a halt to intensification before a full cost/benefit analysis has been carried out. If this is not done, costs could escalate to unaffordable rates, if they aren't already. At a conservative cost of up to $16 billion, the environmental impacts are not negligible, and should no longer be omitted in valuation accounts of the worth of the New Zealand dairy industry. Furthermore, adding dairy imports and interest payments on debt into this equation yields a total of $19 billion, $8.2 billion in excess of the 2012 dairy export revenue.

This investigation aims to spark action and further independent research into a large-scale environmental and economic analysis of the dairy industry. It is important to note that many dairy farmers are victims in this process rather than culprits given the prevailing regulatory and economic system. Often they are maximising production rather than profit as they are required to produce increasing volumes of milk for the industry. Farmers are simply doing what most businesses would do to increase returns. A different agricultural system is required to sustain New Zealand into the future, although deciding what this could be is beyond the scope of this thesis.

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Appendices

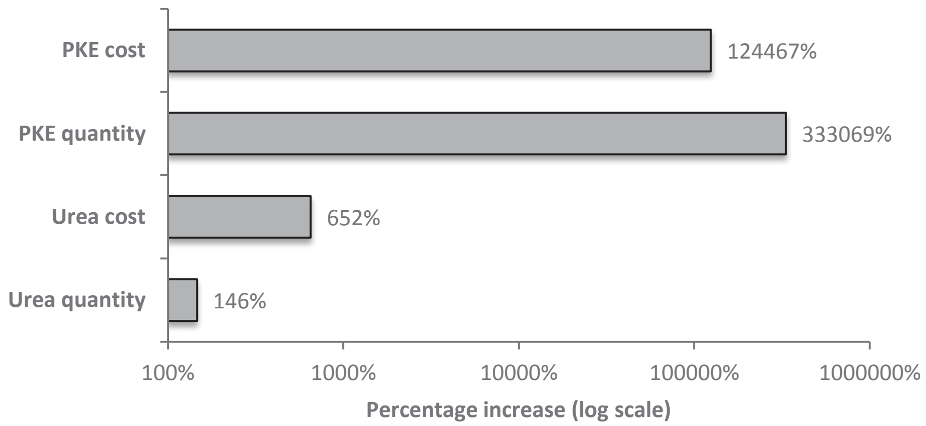
Appendix A: *Quantity and cost of palm kernel extract (PKE) imports*

Quantity and cost of PKE imports into New Zealand from 1992 to 2012

Year	PKE imports Quantity (tonnes)	Cost (NZ$ million)
1992	15	20
1993	15	23
1994	0	0
1995	0	0
1996	100	85
1997	106	138
1998	40	91
1999	408	220
2000	1,554	624
2001	25,877	4,758
2002	23,258	4,058
2003	43,322	6,176
2004	95,921	12,686
2005	188,262	21,007
2006	318,324	43,461
2007	455,314	92,870
2008	1,104,187	317,294
2009	665,382	99,685
2010	1,396,315	252,480
2011	1,373,566	313,764
2012	1,359,329	274,047

Data source: Statistics New Zealand (2013b).

Appendix B: *Increases in PKE and urea imports*



Percentage increase in imports of PKE and urea (quantity and cost) for dairy farms between 1999 and 2012

Appendix C: *Guidelines for faecal bacteria in water*

Guideline values for measuring faecal bacteria for different water uses.

Measure	Water use		
	Drinking water	Contact recreation	Shellfish gathering
E.coli/100 mL	Values should be <1	Median <126 Alert mode - >260 Action/Red mode - >550	
Faecal coliform – Most Probable Number (MPN)			Median shall not exceed 14/100 mL No more than 10% should exceed MPN of 43/100 mL

Data sources: Ministry for the Environment (2002); Ministry of Health (2008); ANZECC (2000).

Appendix D: *Nitrogen leaching estimates from dairy farms in New Zealand*

Estimates of nitrogen leaching from dairy farms in New Zealand

Average N loss (range) kg N/ha/yr	**Location**	**Reference**
45 (21-52)	Upper Waikato	Longhurst & Smeaton (2008)
25-30	Waikato	Beukes et al. (2012)
40-65 (12-115)	Throughout NZ	Menneer, Ledgard, & Gillingham (2004)
39	NZ	Reported in Proffitt (2010)
(18-41)	Bog Burn (Southland) Waikakahi (Canterbury) Toenepi (Waikato)	Power et al. (2002)
16 – milking platform; 55 – dairying wintering	Bog Burn (Southland)	Monaghan et al. (2007)
27-52	Canterbury	Monaghan et al. (2009)
Up to 180	Canterbury	Liburne, Webb & Bidwell (2010)
15-115	Manawatu (Horizons Regional Council area)	Dewes (2012)
24-39; 24-60 – incorporating rearing replacements and forage cropping	Waikato	Ledgard et al. (2006)

Appendix E: *Fertiliser and leaching losses*

N leaching losses under grazed dairy pasture systems with different fertiliser applications

Fertiliser N application (kg N/ha/y)	Leaching loss (kg N/ha/y)	Region	Reference
0	12-50	Waikato	Ledgard, Thom, Singleton, Thorrold & Edmeades (1996)
0	30	Southland	Monaghan et al. (2000)[1]
0	33	Canterbury	Silva et al. (1999)[1]
0	20-74	Waikato	Ledgard, Penno & Sprosen (1999)
0	15-25	Waikato	Ledgard et al. (2006)
100	34	Southland	Monaghan et al. (2000)[1]
170	40-110	Waikato	Ledgard et al. (1996)
200	54	Canterbury	Silva et al. (1999)[1]
200	12-101	Waikato	Ledgard et al. (1996)
200	59-78	Waikato	Ledgard et al. (1999)
200	46	Southland	Monaghan et al. (2000)[1]
400	40-180 (approx.)	Waikato	Ledgard et al. (1996)
400	51	Canterbury	Silva et al. (1999)[1]
400	100-204 (stocking rate 3.24)[2]	Waikato	Ledgard et al. (1999)
400	116-147 (stocking rate 4.48)[2]	Waikato	Ledgard et al. (1999)
400	56	Southland	Monaghan et al. (2000)[1]

Notes: [1] Monaghan et al. (2000) and Silva et al. (1999) cited in (Davies-Colley et al., 2003). [2]Average nitrate leaching per year over 3 years.

Appendix F: *Phosphorus losses from dairy farms*

Examples of phosphorus losses from dairy farms

Average surface P loss kg P/ha/yr	Location	Reference
2.2 (range 0.7 – 4.3)	Upper Waikato	Longhurst & Smeaton (Longhurst & Smeaton, 2008)
1.3 Best practice – 0.7*	Bog Burn catchment, Southland	Monaghan et al. (Monaghan et al., 2007)
0.6 – 1.0	Waikakahi – Irrigated Canterbury Catchment	Monaghan et al. (Monaghan et al., 2009)
0.2 – 1.0	Manawatu – Horizons Regional Council region	Dewes (Dewes, 2012)
10	Westland	Davies-Colley and Nagels (Davies-Colley & Nagels, 2002)

*Best practice is dairy land using best practice techniques to minimise nutrient losses

Appendix G: *Leaching losses from different land uses in the Bog Burn catchment*

Land use and nutrient loss in the Bog Burn catchment (Southland)

Land use	Per cent of catchment	Total kg loss per year		Leaching per ha (Kg/ha/yr)		Per cent of load	
		N	P	N	P	N	P
Dairy milking area	27	10,640	845	16	1.3	35	54
Dairy wintering area	10	13,310	24.2	55	0.1	40	1.6
Dry stock farming – sheep-beef	33	4908	491	6	0.6	16.5	31.5
Forestry	30	958.1	147.4	1.3	0.2	3.2	9.5

Data source: Monaghan et al. (2007).

Appendix H: *Waikato dairy pasture sites not meeting soil quality targets*

Percentage of monitored Waikato dairy pasture sites not meeting soil quality targets from 2003-2009

Measure	Percentage of sites not meeting targets (Year)						
	2003	2004	2005	2006	2007	2008	2009
Macroporosity (-10kPa)	70	63	53	53	49	46	37
Olsen P (upper targets)	17	17	20	20	18	27	21
Total N (upper targets)	52	54	50	50	51	54	53

Data source: Taylor (2011).

Appendix I: *Emissions associated with conversion to palm plantations*

Emissions associated with palm oil plantations per ha and per tonne of PKE showing the impact of land source before conversion.

	Emissions associated with land source (t CO_2-e/ha/yr)					
	Peatland forest high	Peatland forest low	Mineral soil forest high	Mineral soil forest low	Oil palm plantation	*Imerata* grassland
Total	96	49	10.4	6	4.2	-4.2
	Emissions associated with PKE by land source (emissions t CO_2-e/t PKE)					
Economic allocation	2.94	1.49	0.32	0.18	0.13	-0.13
Mass allocation	18.22	9.21	1.97	1.13	0.79	-0.79

Data source: Carlton (2011)

Appendix J: *Cost of waterborne disease in New Zealand*

Estimates of the annual cost of waterborne disease in New Zealand

Pathogen	Total incidence per 100,000 (corrected for under-reporting)	Reported incidence per 100,000 (ESR 2004)	% reported	Cases per annum	Cost per case (1999$)	Proportion waterborne (%)	Total $	Total waterborne $
Campylobacteriosis	3040	400	13	121600	533	10%	64,812,800	6,481,280
E.coli 0157 (VTEC)	9	3	35	343	60000	20%	20,580,000	4,116,000
Cryptosporidiosis	200	20	20	8000	978	30%	7,824,000	2,347,200
Giardiosis	250	25	10	10000	855	20%	8,550,000	1,710,000
Salmonellosis	112	35	31	4480	526	5%	2,356,480	117,824
Yersiniosis	62	12	20	2480	891	10%	2,209,680	220,968
Toxins	414	207	50	16560	221	5%	3,659,760	182,988
Virus (including Hep A)	478	72	15	19120	204	2%	3,900,480	78,010
						Total ($ million)	**113.89**	**15.25**

Adapted from: Ministry for the Environment (2007c, Appendix 3)

Appendix K: *Nitrate leaching and water pollution from dairy farming catchments and proposed irrigation schemes in Canterbury and estimated costs to remediate.*

The following tables detail the volume of water contaminated with nitrate from dairy farming in three predominant dairying catchments and proposed irrigated land in Canterbury. The costs of recovering nitrate (remediate) from contaminated water are estimated using ranges of minimal potential costs between $0.30 and $1.80 per 1000 litres.

Nitrate leaching and water pollution from dairy farming in the Waikakahi catchment, Canterbury, and estimated cost to remediate.

	Monaghan et al. (2009)			**Power et al. (2002)**	
	Free draining soils	**Poorly drained soils**	**Total**	**Average dairy farm**	**Increased intensity (+23%)**
Land area in dairy (ha)	1176	886	2062	1913	1913
Nitrate leached (kg N/ha/yr)	52	27		39	64
Total N leached (kg)	61,152	23,922	85,074	74,607	122,432
Water polluted (m^3)	5,411,952	2,117,097	7,529,049	6,602,720	10,835,232
Cost to remediate ($ million)	$1.62 - $9.74	$0.64 - $3.81	$2.26 - $13.55	$1.98 - $11.88	$3.25 - $19.50

Note: For details of increased intensity of dairy farming see Power et al. (2002)

Nitrate leaching and water pollution from dairy farming in the Bog Burn catchment, Southland, and estimated cost to remediate.

	Monaghan et al. (2007)			Power et al (2002)	
	Dairy milking	Wintering	Total	Average	Increased intensity
Land area in dairy (ha)	665	242	907	1110	1100
Nitrate leached (kg N/ha/yr)	16	55		18	23
Total N leached (kg)	10,640	13,310	23,950	19,980	25,530
Water polluted (m^3)	941,640	1,177,935	2,119,575	1,768,230	2,259,405
Cost to remediate ($)	$282,492 - $1.69 million	$353,381 - $2.12 million	$635,873 – $3.82 million	$530,469 - $3.18 million	$677,821 - $4.07 million

Note: For details of increased intensity of dairy farming see Power et al. (2002)

Nitrate leaching and water pollution from an average dairy farm and all dairy farming in the Toenepi catchment, Waikato, and the estimated cost to remediate.

	Average	Intensive dairy (+20%)	Intensive (+50%) + high N inputs
Nitrate leached (kg N/ha/yr)	41	71	99
Average farm in catchment - 76 ha			
Total N leached (kg)	3116	5396	7524
Water polluted (m^3)	275,766	477,546	665,874
Cost to remediate ($)	$82,730 - $496,379	$143,264 - $859,583	$199,762 - $1.2 million
Total dairy land area in catchment - 1598 ha			
Total N leached (kg)	65518	113458	158202
Water polluted (m^3)	5,798,343	10,041,033	14,000,877
Cost to remediate ($ million)	$1.74 - $10.44	$3.01 - $18.07	$4.20 - $25.20

Note: Leaching rates and land in dairy estimated by Power et al. (2002).

Nitrate leaching and water pollution from proposed irrigation land in Canterbury (270,000 ha), and the estimated cost to remediate.

	Dairying land	Dairy support	Total
Nitrate leached (kg N/ha/yr)	130	70	
Total N leached (kg)	35,100,000	12,600,000	51,600,000
Water polluted (m^3)	3,106,350,000	1,115,100,000	4,566,600,000
Cost to remediate ($ billion)	$0.93 - $5.59	$0.33 - $2.01	$1.27 - $7.60

Appendix L: *Nutrient statistics for Rotorua Lakes*

Nutrient inputs, internal loads and nutrient targets in Lakes Rotorua, Rotoiti, Okaro and Rotoehu.

	Nitrogen t/yr				Phosphorus t/yr			
	Rotorua	**Rotoiti**	**Okaro**	**Rotoehu**	**Rotorua**	**Rotoiti**	**Okaro**	**Rotoehu**
Internal load	360	50	2.4	5.6	36	20	0.38	1.4
Current inputs	556	364	5.3	53.3	39	29	0.35	2.4
Current exports	746				39			
Targets	435[1]	230	4.4	44.5	37	13.3	0.33	1.7
Difference: target - current inputs	121	134	0.9	8.8	2	15.7	0.02	0.7
Rotorua – difference in:								
2055	224							
2105	264							
2250	311							

Data source: Bay of Plenty Regional Council & Te Arawa Lakes Trust (2007); Environment Bay of Plenty, Rotorua District Council & Te Arawa Lakes Trust (2006, 2009)

Notes: Exports are only estimated for Lake Rotorua as the other lakes are not expected to have significant lagtimes in nutrient loss to water.

[1]Other lake modelling scenarios suggests nutrient inputs closer to 350 t N/yr would be needed to attain sustainable water quality in Lake Rotorua (Hamilton et al., 2013), requiring removal of an additional 85 tonnes N/yr.

The current nutrient inputs into Lake Rotoiti have been calculated prior to the completion of the Ohau Channel wall, completed in July 2008.

Appendix M: *Nutrient inflows to the Rotorua Lakes*

Nutrient inflows and proportions from selected land uses to Lakes Rotorua, Rotoiti, Okaro and Rotoehu

Land use	Area (ha)	% of catchment	N load (t/yr)	Kg N/ha/yr	% of total N	P load (t/yr)	Kg P/ha/yr	% of total P
Lake Rotorua								
Dairy	5,883	11 (29% of pasture)	294.1	50	37.6 (52.2% of pasture)	4.12	0.70	10.6 (24.3% of pasture)
Sheep / beef	11,464	22 (57% of pasture)	226.7	19.8	29.0 (40.2% of pasture)	10.33	0.90	26.7 (61% of pasture)
Total pasture	20,112	38	563	28	71.9	16.93	0.84	42.5
Native forest and scrub	10,588	20	42.1	4.0	5.4	1.31	0.12	3.3
Exotic forest	9,463	18	28.4	3.0	3.6	0.95	0.10	2.4
Total catchment inflows	52,347	100	783.1	15	100	39.80	0.74	100
Lake Rotoiti								
Dairy	126	1	6.3	50	1.7	0.09	0.71	0.3
Sheep/beef	847	6.8	17.4	18	4.8	0.76	0.9	2.6
Native forest and scrub	3,347	26.7	13.4	4	3.7	0.40	0.12	1.4
Exotic forest	4,281	34.2	12.8	3	3.5	0.43	0.1	1.5
Ohau channel			250		68.8	25		86.4
Total catchment inflows	12,519	100	363.6	9*	100	28.93	0.31*	100
Lake Okaro								
Dairy	38.3	10.7	0.575	15	22.2	0.0689	1.80	17.4
Sheep/beef	234.5	65.6	1.663	7	64.3	0.2613	1.10	66.0
Scrub	13.2	3.7	0.033	2.5	1.3	0.0005	0.04	0.1
Forestry	19.9	5.6	0.049	2.5	1.9	0.0008	0.04	0.2
Catchment total	357.4	100	2.588			0.3959		
Lake Rotoehu								
Dairy	266.4	5.4	13.320	50	25	0.1865	0.70	7.5
Sheep/beef	974.3	19.6	17.537	18	33	0.8769	0.90	35.4
Exotic Forestry	1292.9	26	3.8787	3	7.3	0.1293	0.1	5.2
Bush	1373.1	27.6	5.4924	4	10.3	0.1648	0.12	6.7
Total catchment inflows	4971.6		53.088	10.62	100	2.4454	0.49	100

Data source: Environment Bay of Plenty et al. (2007); Bay of Plenty Regional Council et al. (2007); Environment Bay of Plenty et al. (2006)

Notes: Additional land uses contribute to nutrient inflows, but these ones were selected to compare.

*Excludes Ohau Channel inputs

Appendix N: *Lake restoration procedures*

Examples of restoration procedures carried out in the Rotorua Lakes

Action	How it works	Estimated cost / example	Positives	Negatives
Dredging (description below)	Removes nutrients/weeds in bottom sediments.	Lake Rotorua: - To control weed infestations – NZ1991 \$1.6 – \$2 million for removing only weeds to \$27 million for root removal: permanent control of weeds. - To remove phosphorus - \$84-252 million (Analytical & Environmental Consultants, 2007). Lake Okaro: - To remove phosphorus - \$974,000	Proven effective overseas	- Disposal issues - Re-suspension of sediment - Adverse environmental impacts - Can fail - Expensive although costs vary considerably.
Inflow diversion	Diverts nutrient-rich lake inflows downstream	Ohau Channel wall in Lake Rotoiti diverts nutrients down the Kaituna River- \$10 million	Potential for immediate benefits	Can cause adverse effects downstream
Oxygenate hypolimination	Removes poor quality water at the bottom of stratified lakes?	Lake Rotoiti (Proposed) - \$1.5 million capital, \$1.2 million/year – no residues / western end of lake not treated (Analytical & Environmental Consultants, 2007)	No residues / may not be self-sustaining solution.	May have uncertain environmental impacts
Oxygenation / destratification	O_2 pumped to the bottom of lakes can reduce nutrient releases and increase lake health	Destratification trial in Lake Rotoehu (790 ha) \$524,000	No chemicals, low level of intervention; potential for rapid results;	On-going operation costs
Sediment capping / phosphorus inactivation	Chemicals like 'alum' can lock up nutrients in lakes	Lake Okaro (30 ha) modified zeolite application c. \$75,000/year over 3 years. Lake Rotorua alum dosing \$1 million/year. Lake Okareka - \$300,000 capital - phoslock	Proven effective overseas	- Culturally sensitive - Toxicity issues - Repeated applications required - Extensive consenting process - Phoslock – uncertain environmental impacts
Weed harvesting	Removes nutrients locked up in excess weed growth	Hornwort harvesting in Lake Rotoehu (790 ha) \$52,800/year = \$22/kg N and \$165/kg P	Co-benefits	On-going operation costs; limited effectiveness on its own
Wetlands		Costs vary depending on type of wetland		

References: A&E Consultants (2007); Hamilton et al. (2013); Abell et al. (2011).

Dredging

Dredging removes nutrients from the beds of lakes and reduces nutrient cycling through and from sediment (Abell et al., 2011; Analytical & Environmental Consultants, 2007). Problems associated with dredging relate to resuspension of sediment and disposing of the dredged material as well as other environmental impacts (Analytical & Environmental Consultants, 2007). Dredged material could be used as a resource rather than dumped. Dredging would only be practical for some of the Rotorua Lakes, i.e those experiencing severe water quality problems and which undergo considerable nutrient recycling from sediments during anoxic periods, which include all four lakes focused on here (Analytical & Environmental Consultants, 2007).

Because it is so expensive there is little point in carrying out if the benefits are not likely to last. Practices to decrease nutrient inputs into the lake will also need to be carried out alongside dredging to ensure it does not need to be repeatedly carried out.

Appendix O: *ETS and taxpayer position*

Terry (2012a) reported the ETS position is the flow of income and expenses associated with the ETS. The government issues credits to foresters for absorbing carbon and as subsidies and compensation to major industry, and credits are collected for industries emitting carbon. A document prepared by the New Zealand Treasury department detailing the ETS position in 2010 showed that 87 Mt of credits will be issued while only 47 Mt will be earned, resulting in a 40 Mt deficit. This deficit has increased to 74 Mt because 121 Mt of credits have now been issued while credits earned have stayed the same. These credits will have to be surrendered in the future, when carbon prices have most likely increased, putting an even greater cost on future taxpayers. The Taxpayer position is the sum of the National position and the ETS position, calculated as 51 Mt in deficit (Terry, 2012a).

At today's carbon price of NZ$0.66 (May 2013) (Ministry for the Environment, 2013d), this deficit would be $33.7 million. However, it will be paid back in the future when prices are expected to be much higher. At a carbon price of $25/tonne, this will be $1.3 billion. Additionally, between now and 2050, it is estimated that New Zealand's emissions will exceed government targets by 1.1 billion tonnes, costing $28 billion to buy carbon credits at $25/tonne – a price far below UK government forecasts for that period (Terry, 2012a). The government uses the carbon price of $25/tonnes to consider policy options under the ETS (Terry, 2012b). However, it is more appropriate to use a forecast future carbon price when assessing the deficit value as the deficit will be paid off in the future (Terry, 2012b). This price could be much higher than $25/tonne.

Government reports now show there is a 54 Mt overall deficit on the government's carbon accounts for the commitment period, essentially the same amount the Sustainability Council reported on which was previously denied by the Climate Change Minister (Terry, 2012b). This reality has been supressed by the Government, even to the extent that officials are asked to consider whether ETS credits given away by the Government should be counted as a cost to the taxpayer (Terry, 2012b).

The agricultural sector is exempt from a tax on non-carbon dioxide emissions for at least the first Commitment Period of the Kyoto Protocol (2008-2012). The excess emissions from agriculture above 1990 levels over the first commitment period have a value of $600 million at $15 per tonne of carbon dioxide equivalent (6 MAF, Agriculture: Briefing for incoming ministers, October 2005, p14) (Terry, 2007).

Appendix P: *Policy instruments to manage pollution*

Types of policy instruments for environmental managers to use when controlling diffuse pollution.

Instrument	Uses	Examples
Education and engagement	Can be instrumental for gaining support and prompting voluntary action. However, some people/groups are resistant to change so can be limited	- Workshops and meetings to provide a forum for lake managers to engage with stakeholders, e.g. the Lakes Water Quality Society (which seeks to protect and restore the region's lakes) regularly holds meetings including symposia attended by international water-quality experts - An active programme of research into lake science. Several research organisations, including Waikato University are involved in lake restoration - Education of the farming community about nutrient management issues - Education of the public about the causes of water-quality decline
Voluntary action	Limited potential in achieving better water quality, especially in the most polluted catchments were remedial actions are often extensive and costly	- Dairying and Clean Streams Accord – contains quantified performance targets to meet the goal of achieving 'clean, healthy water'. Doubt has arisen regarding the success of this

Regulation	Often the best option when an absolute change in behaviour is required. Limitations in controlling diffuse pollution as nutrients are hard to measure and attribute to specific individuals	- Limits on N loss from properties, e.g. Rule 11 of the regional Water and Land Plan (BOPRC) prohibits loss of nutrients from pastoral land at rates greater than an individual farm benchmark level. Can use farm-scale modelling software (Overseer) to estimate nutrient losses from properties. - NPS on Freshwater Management requires councils to set limits on freshwater quality
Economic instruments	Often regarded as more efficient than regulations, allowing environmental objectives to be achieved more cost effectively than with regulation alone	- Subsidies to encourage landowners to undertake fencing and planting in retirement areas and riparian margins - Nutrient trading – operating in Lake Taupo. Unlikely to be feasible to implement at the national scale due to the extensive amount of catchment information required and likely administrative work involved. Could be used as a tool to control pollution in the most sensitive and iconic catchments - Environmental taxes, e.g. on fertilisers - Eco-labelling schemes – consumers pay more for certified farms that adhere to an audited Environmental Management System, providing economic incentives for farmers to make positive changes - Carbon farming - Pay farmers for the provision of ecosystems services, e.g. nitrogen loss.

Adapted from: Abell et al. (2011).