THE WATER FOOTPRINT OF AGRICULTURAL PRODUCTS IN NEW ZEALAND: THE IMPACT OF PRIMARY PRODUCTION ON WATER RESOURCES

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ABSTRACT

Protecting and sustaining global water resources are one of the most challenging issues facing the world. Future food security is threatened by the continued increase in demand for water. Agriculture is, by far, the largest consumer of global freshwater, with irrigation accounting for more than 70% of water withdrawals. As the dominant land use in New Zealand, agriculture has the most widespread impacts on freshwater quality and quantity. The water footprint (WF) is a metric that quantifies the environmental impacts related to water use. The WF is likely to form the basis of eco- verification or environmental product declarations related to water use, thereby communicating water use impacts associated with the production of goods and services to a range of stakeholders, including consumers. While the international standards for water footprinting are still being developed, a number of protocols have already been proposed for quantifying the WFs of a range of agricultural products. If New Zealand is to remain competitive within an increasingly discriminating market place, it will need to be able to demonstrate the impacts of resource use.

The objectives of this thesis are to quantify the impacts of the production of two of New Zealand's economically important agricultural products on water resources. These are wine-grape (*Vitis vinifera*), the top horticultural export product, and potatoes (*Solanum tuberosum*), the largest vegetable crop in terms of area under cultivation in New Zealand. In order to meet this objective, a new method, which is based on a full hydrological assessment, was developed. A further objective is to compare this method with three other WF methods, in relation to their usefulness to stakeholders. This study also aims to identify potential management options, which can be implemented to reduce the water-related impacts of these products.

Electricity is a major input into the supply chains of most primary products and, because hydropower is the major component of New Zealand's electricity mix, it was first decided to determine the WF of hydro-electricity. This WF value of electricity was then used in subsequent assessments of the WF of wine and potatoes. The hydrological water balance method has been used here to quantify the WF of wine production in Marlborough and Gisborne, which are two hydrologically different regions in New

Zealand. This assessment considered approximately 12,600 ha under grapes and 36 wineries across both the regions: and the vineyards were on 29 different soil types spread across 19 climatic regions. The functional unit (FU) is a 750-mL bottle of wine at the winery gate. The hydrological water balance method considers water inflows and outflows into and out of the system and it identifies two main water resources; namely, soil water (the green water resource), and groundwater (the blue water resource). The net uses of these two resources were quantified as the green and blue water WFs. The impact of wine production on water quality, the grey WF, was assessed by considering the average nitrate nitrogen (NO_3-N) concentration and the load of NO_3-N reaching groundwater. Subsequently, the WFs of the same wines were evaluated, by using three other WF methods: the 'consumptive water use method' of the Water Footprint Network (WFN); the stress-weighted WF; and a Life Cycle Assessment (LCA) based method that considers freshwater ecosystem impact and freshwater depletion. All these methods were evaluated for their ability to indicate local impacts on the water resources and their usefulness to key stakeholders. The hydrological water balance method was also used to assess the WF of a potato crop grown in the Manawatu region. This evaluation was supported by field measurements of the soil water content, drainage and leaching of NO_3-N below the root zone. Finally, the WF of a kilogram of potatoes at the packhouse gate was quantified, by using mechanistic modelling, which was robustly supported by these field measurements.

There was large variation in the WF of wine, both within and across the Marlborough and Gisborne regions. This variation reflects the large variability in regional rainfall and the large differences in local soil properties. At the grape-growing stage, the average blue-WFs were -81 L/FU and -415 L/FU for Marlborough and Gisborne, respectively. These negative values indicate that these water resources are being recharged on an annual timescale. The green-WFs were negligible, because the soils are returned to field capacity every year during winter. The average grey-WFs, that is, the water required to dilute the NO_3-N leached in the vineyard phase, were 40 and 188 L/FU for Marlborough and Gisborne, respectively. However, the average concentration of NO_3-N in the leachate was smaller than the New Zealand drinking water standard of 11.3mg /L. The comparison of different WF methods showed that the WFN method for the blue and green footprints does not represent impacts on the local water resources. The ability of the stress-weighted WF and the LCA based method to indicate the local impacts is limited due to the spatial constraints of the characterisation factors that have to be used. The hydrological water-balance method can indicate local impacts on water resources, and it does provide useful information to growers and resource regulators, which will enable them to set measurable targets, in order to reduce the WF.

In the potato study, high spatial and temporal variability in field measurements proved to be very challenging. Therefore, it was considered to be more accurate to account for the whole crop sequence and long term weather data, through mechanistic modelling. The average blue-WF of the potato growing phase was -72 L/kg, thus indicating that the rain-fed potato production system has no deleterious impacts on blue water quantity in the region. This indicates that, for every kilogram of potatoes harvested, 72 litres of water recharges the local aquifer. The average grey-WF was 61 L/kg, of which 56 L/kg is from the cropping phase. The use of the absolute value of the grey WF, in order to understand the impact on water quality, is not straightforward. This point notwithstanding, the average concentrations and loading of NO_3-N from the cultivation phase indicate that current practices are having some impact on water quality. The average concentration of NO_3-N leaching below the root zone was at 11.3 mg /L, which is just at the drinking water standard. The average loading rate of NO_3-N was 27.8 kg /ha/y. The potential to reduce the grey-WF was investigated by modelling three different nitrogen fertiliser application scenarios related to split applications and different timings. All three scenarios reduced the NO_3-N concentrations and loads from the production system. The simulated NO_3-N concentrations were reduced from 11.3 to 9.5 mg /L, and the loading rates were reduced from 27.8 to 24.3kg /ha/y, depending on the scenario. The WF is a useful tool to understand the impact of agricultural systems on water resources and also to derive improvement options. However, the robustness of current WF protocols for quantifying the impact of the product life cycle on water quality is dubious. These methods require further improvements, so that water footprinting can provide reasonable and rational metrics of the sustainable use of our water resources. The research described in this thesis has provided some new steps within this improvement process.

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LIST OF ABBREVIATIONS

Abbreviation	Meaning
WF	Water Footprint
LCA	Life Cycle Assessment
WFN	Water Footprint Network
FU	Functional Unit
CF	Characterization Factor
FEI	Freshwater Ecosystem Impact
FD	Freshwater Depletion
ADP	Abiotic Depletion Potential
WSI	Water Stress Index
WTA	Withdrawal to Availability
ET	Evapotranspiration
GIS	Geographic Information System
SPASMO	Soil Plant Atmosphere System Model
TDR	Time Domain Reflectometer

CHAPTER 1

General Introduction

1.1 Background and Justification

Freshwater is the most essential of natural resources. There are ominous hydrological signs, across the globe, around both the quantity and the quality of freshwater resources; these include groundwater depletion, lowering of river flows, and water pollution. These problems indicate that current levels of water use and the quality of water discharges exceed sustainable limits, in many parts of the world. One of the key challenges facing the world is meeting the rapidly increasing demands for food, water and material goods of an increasing global population. Meanwhile, there are simultaneous imperatives for protecting the ecosystem services provided by natural water ecosystems (Postel, 2000). Agriculture is, by far, the largest consumer of global freshwater, with irrigation accounting for more than 70% of water withdrawals (UNEP, 2007). It is widely recognised that agricultural production represents a significant proportion of the anthropogenic environmental impacts on water (Canals et al., 2010). Due to agriculture's huge water usage, and its contribution as a source of agrichemical emissions into freshwater, it is considered central that future attempts to address the global stress on water resources must focus on agricultural production. As the dominant land use in New Zealand, agriculture has the most widespread impacts on freshwater quality and quantity.

The water footprint (WF) has been proposed as a metric that indicates the magnitude of water use and the impacts of water use in the production of goods or services. The WF has been applied to a range of agricultural production systems around the world (Ridoutt et al., 2009; Hoekstra et al., 2011; Canals et al., 2010). It is considered to have the potential to underpin an environmental product declaration, and thereby act as a communication means of environmental performance to stakeholders (Ridoutt et al., 2009). The labels that are likely to appear on products in the future are expected to alert retailers and customers on the product's water use and highlight its impact on the environment (Segal and MacMillan, 2009). The world's largest retailers, such as Walmart, have announced plans to develop product indices of sustainable water use and metrics of the environmental impacts of this water use (Walmart, 2009). The multifarious issues surrounding the sustainability of water use are, therefore, of growing importance to primary industries, not just from a strictly environmental point of view, but also for the marketing of products through eco-verification, in order to respond to questions and concerns of informed consumers.

International standards for water footprinting are still being developed. A number of methodologies and protocols have been proposed for quantifying the water footprints of agricultural products. Some methods are based only on consumptive water use (Hoekstra et al., 2011). Other methods attempt to quantify the impact on water resources, by referencing the product's consumptive water use to the characterisation factors of regional water availability (Canals et al., 2009; Ridoutt and Pfister, 2010). Whether it is a volumetric measure, or an impact-orientated indicator, the outcome of the WF process should be meaningful to all stakeholders, including growers, resource regulators, retailers and consumers. It is argued that an aggregated impact indicator is not useful to the formulation of a specific response. Therefore, it is timely and important to consider the comparative advantages of the different approaches that have been proposed for the calculation of WFs; and to consider the usefulness of their outcomes to key stakeholders. Irrespective of the differences in quantification, it is of great importance that a water footprinting method captures the inherent variability in the regional hydrology and that it indicates local impacts on freshwater resources.
Since New Zealand is a major exporter of water through agricultural products (Clothier et al., 2009), with its touted 'clean and green' to a wide range of destinations around the world, it will increasingly be necessary to eco-verify its products in terms of resource use, not only for greenhouse gas emissions (as is topical) but also for water, which will be increasingly important. Furthermore, producers are keen to know the impact of their production system on local water resources and thereby the long-term sustainability and prosperity of their industry. In particular, they will be interested in the potential management and improvement options that are available, in order to reduce their WF.

It is well established that environment assessment tools, such as life cycle assessment (LCA), are based on a supply-chain perspective (PAS:2050, 2008), which considers resource use and impacts from all inputs utilised throughout the production process and usage phase. From this point of view, it is important to quantify the WF of the key inputs from the background system. New Zealand's electricity generation is dominated by hydropower, which is deemed to have a larger water footprint on global average than other energy carriers (Gerbens-Leenes et al., 2009). In this study, it was considered that such an input needs better quantification, in detail and under local conditions.

As given in the objectives below, this study aims to assess the WF, in order to understand the hydrological impacts of the lifecycles of two key agricultural products, which are economically important for New Zealand: that is, wine, a $1.1 billion export earner, and potatoes, which earn $81 million through exports (Fresh Facts, 2011).

1.2 Research objectives

The main objective of this study was to quantify the impacts of water use through the life cycle of two important New Zealand agricultural-products, by using the concept of water footprinting. The two products studied were a 750-mL bottle of wine at the winery gate and a kilogram of fresh potatoes at the packhouse gate. Since this study was being conducted at the time of the development of international standards for water footprinting, it also aimed to evaluate existing water footprinting protocols and to contribute towards the improvement of methodologies.

The specific objectives of this study were as follows:

1. *To quantify the hydrological impact of New Zealand hydropower that accounts for the major part of New Zealand's electricity supply, and which is a major input of both product life cycles considered in this study.*

2. *To quantify the hydrological impact of the life cycle of New Zealand wine production on water resources in two hydrologically different major wine-regions in New Zealand: Marlborough and Gisborne.*

3. *To assess a number of different water footprinting protocols currently proposed, and to evaluate the usefulness of their outcomes to the key stakeholders by considering New Zealand wine production as a case study. This also includes identifying the challenges facing water footprinting, and the improvements needed for the development of protocols that are meaningful for all stakeholders.*

4. *To assess the impact of potato production on water resources, by using measurements and modelling to calculate the water footprint of a case study crop grown in the Manawatu region (one of the main potato-growing areas in New Zealand).*

5. *To formulate specific management or improvement options, in order to reduce the water footprint of potatoes.*

1.3 Thesis structure

This thesis is comprised of eight chapters, including this introductory chapter. The foci of the subsequent chapters are outlined below.

Chapter 2 provides a review of the pertinent literature, which highlights the impacts of agriculture and its increasing pressure on water resources, from both global and New Zealand perspectives. Subsequently, there is a discussion on the water footprinting concept and methodological issues surrounding it. This review of literature emphasises the importance of assessing a water footprint at the local scale, and it proposes that it is timely to assess the different methodological frameworks and to compare their abilities to capture the multiple impacts of agricultural production systems on water resources. This review necessarily provides an overview of the literature whereas, in the individual chapters, the literature reviews are more detailed and specific to the research in that particular chapter.

Chapter 3 highlights the importance of an accurate assessment of the water footprint of hydroelectricity, since it is a key input to the lifecycle of agricultural products in New Zealand. In this chapter, the results of a detailed assessment of the water footprint of New Zealand's hydropower are presented. In order to accurately estimate the WF of New Zealand electricity, all the major hydroelectric power plants, which account for more than 95% of hydropower generated in the country, were considered. This chapter examines two existing methods and introduces a new method of water footprinting, by using the full water-balance of the water storage system. The chapter concludes that, among current water footprinting protocols, the water-balance method provides a better understanding of hydrological impacts. Thus, this theoretical framework is then used to assess the product water footprint within the agricultural systems presented in Chapter 4 on wine and Chapter 6 on potatoes. The content of Chapter 3 has been published in a peer-reviewed international journal (Herath et al., 2011): (Publication no. 1 in the Appendix B). Furthermore, these findings have also been presented and discussed at different forums, which have led to publications 10, 19, 20 and 21 in Appendix B and Activity no. 9 and 11 in Appendix C.

Chapter 4 presents the results of a detailed assessment of the impact of wine production on the quality and quantity of water resources of two regions, by using the hydrological water-balance method of water footprinting. The water footprint is presented for a bottle of wine produced in the two hydrologically different regions of Marlborough and Gisborne. This work has been published in a peer reviewed journal as Herath et al. (2013) (Publication 2, Appendix B). Furthermore, these findings have also been discussed at different forums, which have led to publications 7, 8 and 16 in Appendix B.

Chapter 5 presents a comparative evaluation of the results from different water footprinting protocols, by considering a bottle of wine produced in the Marlborough and Gisborne regions, as a case study. Three other WF methods were considered, together with the hydrological water-balance method used in Chapter 4. These three approaches were the volumetric water footprint method of the Water Footprint Network (WFN); the stress-weighted water footprint method proposed by Ridoutt and Pfister (2010); and the LCA-based method proposed by Canals et al. (2009). The Canals et al. (2009) method requires a water scarcity assessment, in order to use a characterisation factor for the freshwater ecosystem impacts. For this purpose, the water scarcity of all New Zealand's regions was assessed, and these results are provided in Appendix A**.** This water scarcity assessment study was presented at the workshop on 'water footprinting' organised by New Zealand Life Cycle Management Centre (NZLCMC), and it led to Publication 9, Appendix B. The outcomes of the three different WF methods were evaluated, based on their ability to represent the local impacts on water resources, and their usefulness to key stakeholders. The content of this chapter has been published in a peer reviewed journal (Publication 3 in Appendix B). It was also presented and discussed at two conferences (Publication 8 and 15 in Appendix B).

Following the assessment of WF for perennial crop-based wine grape in Chapters 4 and 5, the hydrological water balance method principles were then used to quantify the water footprint, by using measurements under field conditions of an annual crop – potatoes, in Chapter 6.

Chapter 6 describes the experimental results of quantifying the soil-water dynamics and the leaching of nitrate under a potato crop. This assessment was carried out using measurements from drainage fluxmeters over one growing season. Mechanistic modelling was used to extend the experimental results, and to account for spatial and temporal variability, in order to quantify both the blue and grey WFs of potatoes. Preliminary results of this work were presented and discussed in two workshops. This work has led to Publications 4 and 9 in Appendix B and Activity 12 and 13 in Appendix C.

In **Chapter 7,** the results from the previous chapter (Chapter 6), using measured soil water dynamics and measured drainage and leachate for just one potato growing season are combined with long-term weather patterns and the cropping sequence of a potato crop and green cover-crop rotation. This assessment covers the water footprint of the full lifecycle, and the system boundary extends from field cultivation through to the packhouse gate. Furthermore, the modelling is used to establish field fertiliser-management practices, which would reduce the grey water footprint, as a result of the leakage of nutrients from the root zone of the potatoes. This work has led to Publication 5 in Appendix B.

Finally, **Chapter 8** provides a review of the major findings from Chapters 3 to 7 and brings these together with some concluding assessments. Furthermore, an outline for further research in this area of water footprinting is presented

1.4 Highlights of this thesis

- In this thesis, a framework for a hydrologically based water footprinting method is provided. It has been applied to hydropower, plus the life cycles of wine and potatoes.
- Subsequently, this method was compared with other water footprinting methods, in terms of quantification, outcome and utility of the results to all stakeholders.
- It was considered that the hydrological method provides a better understanding of the hydrological impacts of agricultural production systems.
- This thesis is presented in the form of chapters that have (or soon will be) published as stand-alone scientific papers. Therefore, Chapters 3-7 are, to a certain degree, stand-alone.
- Chapters 1 and 8, therefore, provide the link, in order that the thesis is a coherent record of academic research.

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CHAPTER 2

Review of Literature

This chapter provides a generic review of the literature regarding the issues surrounding freshwater resources, and the protocols being developed for water footprinting. A specific review of pertinent literature is given in the introduction of each chapter. In order to address the issues that were raised from this review, the research plan had the aim of contributing to the developing water footprinting standards, in addition to gaining an understanding of the environmental impacts of two primary production processes of wine-grape and potatoes on local water resources.

2.1 Introduction

Emerging freshwater scarcity has been recognised as a global issue of utmost importance, since the world is witnessing a steadily worsening situation, due to rapidly decreasing freshwater resource availability and utility (Postel, 2000). There is a growing awareness that increased water use by humans does not only reduce the amount of water available for future industrial and agricultural development, but it also has a profound effect on aquatic ecosystems and their dependent species (Koehler, 2008). Due to this pressure on limited water resources (Postel, 2000), metrics, which are used to indicate water consumption and the environmental impacts of water consumption, are becoming important. Among these metrics, the water footprint is becoming a widely used tool. While international standards for water footprinting are in the process of development, there are a number of different concepts and protocols that have already been proposed and discussed.

2.2 Freshwater resource and its availability

Freshwater is one of the planet's most valuable resources, it being a crucial life-sustaining element which cannot be substituted (Koehler, 2008). The intimate involvement of water in all life processes makes all living matter highly vulnerable to changes in the quality and quantity of water stocks (Falkenmark, 1990). As the source of drinking water and the basis for hygiene and food supply, freshwater is indispensable for humans, while at the same time it sustains biodiversity and supplies the pivotal ecosystem functions on which ultimately we all depend (Koehler, 2008). Freshwater makes up only 0.01% of the world's water, and covers only approximately 0.8% of the Earth's surface. Yet, this tiny fraction of global water supports at least 100,000 species (Dudgeon et al., 2005). Since this miniscule fraction of freshwater is unevenly distributed around the globe, a fifth of the world’s population (more than 1.2 billion people) live in areas of physical water scarcity (Fig. 2.1)(CAWMA, 2007). In some aspects, freshwater scarcity is a global phenomenon, but in other ways it is distinctly a local or regional problem (Ridoutt et al., 2009).

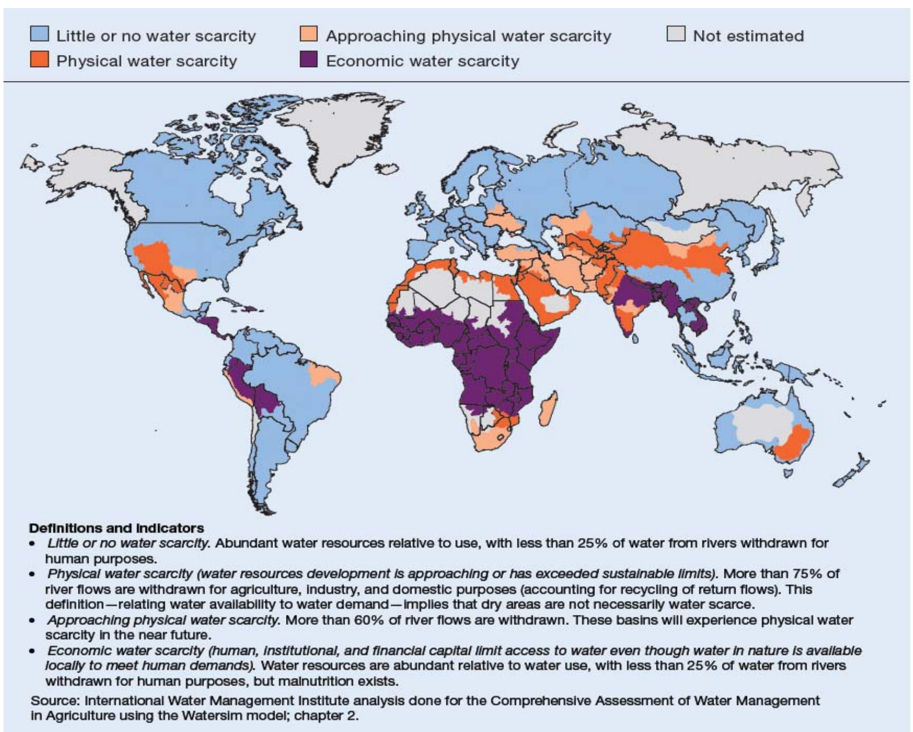


Figure 2.1 World map indicating the degree of water scarcity (CAWMA, 2007). The red colours indicate scarcity and the blue reflect greater water availability.

2.3 Agriculture and water-use impacts

The production of biomass for food, fuel and fibre, by agriculture, accounts for 86% of global fresh water use (Hoekstra and Chapagain, 2007a). Food production is a highly water-consumptive activity. Approximately 40% of the world's food is produced within 17% of the world's irrigated cropland. This is, in principle, a good thing and water enables greater disproportionate advantage to be taken from the world's finite land resources. However, the area under irrigation is expected to increase in the future, due to limited opportunities to expand rain-fed crop production (Postel, 2000). Furthermore, in order to feed the global population in 2025, it has been estimated an additional 500 km^3 of irrigation water will be required (Shikomanov, 1996). An even larger volume of additional water will be needed, if the water that is delivered and applied to farms is used inefficiently (Postel, 2000). In addition, as urban water demands expand, cities are beginning to draw water away from agriculture. By 2025, nearly five billion people are expected to live in cities, approximately twice as many as in 1995 (Postel, 2000). Almost certainly, a portion of these greater urban and industrial demands will be met by transfers of water out of agriculture. This relocation will threaten food security. If these water-use trends eventuate, it will be difficult to supply this amount of additional irrigation water on a sustainable and ecologically sound basis.

The over-exploitation of surface water bodies and consumption of 'fossil' groundwater for agricultural production might jeopardise the freshwater needs of future generations. Irrigation and the damming of rivers can cause fragmentation of river basins, thereby drastically reducing downstream freshwater availability and threatening the riparian aquatic and terrestrial ecosystems (Koehler, 2008). Inappropriate water resource management endangers ecological functioning and biodiversity: it disturbs water cycling and it has led to the desiccation of rivers streams and lands.

Globally, agriculture has also had a significant impact on water quality. It contributes to non-point source water pollution through excess nutrients, pesticides and other contaminants. Agricultural nutrient losses are a major contributor to water pollution around the world, and in New Zealand in particular (Marsh, 2012). The pollutants that come from diffuse, non-point sources are much more difficult to quantify, compared to point-source discharges. Therefore, the impact of primary production on the quality of receiving water resources is challenging to quantify, and contentious to manage.

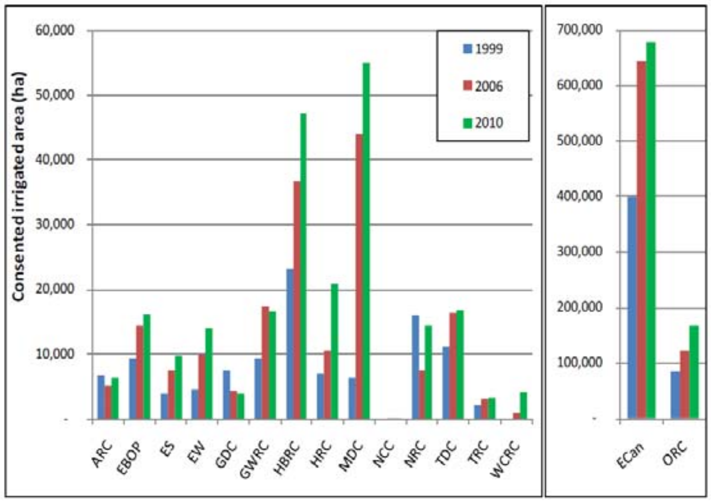
2.4 Freshwater availability and demand in New Zealand

2.4.1 Water scarcity in New Zealand

As a preliminary assessment for this research, the freshwater scarcity in different regions of New Zealand was assessed, by using three water scarcity indicators (Herath et al., 2010a, presented here as Appendix A).These indices are the Falkenmark Index (per capita water availability) (Falkenmark, 1986); the Water Scarcity Index (Alcamo et al., 2003); and the Water Stress Indicator(Smakhtin et al., 2004). According to the Falkenmark indicator, all regions in New Zealand are well above the threshold value of 1,700 m^3 per capita, per year. The Water Scarcity Index shows that none of the regions in New Zealand are freshwater stressed, but all regions of New Zealand fall into the 'slightly exploited' category (Herath et al., 2010b; Herath et al., 2010a). It is important to note that all these indicators only consider the quantity of water. However, water quality governs the utility of available water. Therefore, it is important that water assessment metrics and protocols take into account both water quantity *and* water quality.

2.4.2 Water demand in New Zealand

The major water use in New Zealand is by agriculture, particularly through irrigation. Nearly 80% of New Zealand's water abstractions are destined for agriculture and horticulture. Intensification of agriculture has led to a rise in the area under irrigation over the past few years, in all regions (Fig. 2.2) (MfE, 2010). It is predicted that demand for irrigation water could double again over the next 20 years (Davie, 2009). Water scarcity already limits the growth of irrigation schemes in some parts of New Zealand, despite the economic potential of irrigable land (Clothier et al., 2009). Canterbury includes 450,000 ha of extra land that could be irrigated if water was available, in comparison to the 560,000 ha which is currently consented for irrigation (EC, 2009).



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	Hawke's Bay Regional Council
HRC	Horizons Regional Council
MDC	Marlborough District Council
MfE	Ministry for the Environment
NCC	Nelson City Council
NRC	Northland Regional Council
ORC	Otago Regional Council
TDC	Tasman District Council
TRC	Taranaki Regional Council
WCRC	West Coast Regional Council

Figure 2.2 Consented irrigated areas (ha) by region in New Zealand. Source: MfE (2010)

In most catchments, in regions like Marlborough, the quantum of water available for allocation is now approaching the sustainable limit (MDC, 2012). Given this increasing demand for irrigation water, it will be important to ensure wise water stewardship. Since freshwater resources are finite, increasing demand will eventually be limited by supply, unless significant reductions or new storages can be found.

2.4.3 Water quality of New Zealand's water resources

Agriculture plays a prime role in New Zealand's economy. As New Zealand's agriculture sector expands, more pressure is being put on its ecological infrastructures. Water pollution is now considered to be one of the most important environmental issues facing New Zealand (Marsh, 2012). Intensive agricultural production has significantly influenced the quality of surface and groundwater resources. For example, in the vegetable-growing areas of Pukekohe to the south of Auckland, the shallow groundwater is now undrinkable in many places, due to high nitrate levels resulting from decades of high fertiliser use. However, cleaner water can be taken from the deeper Kaawa aquifer in the same area. Water from this aquifer is protected from the water-resistant ground layers that lie above it (PCE, 2012). This type of impact is complex to quantify and difficult to manage. However, measures are being implemented by local government authorities around New Zealand, in order to regulate levels of nutrient loads, by establishing limits for agricultural land uses (Horizons Regional Council, 2008).

Such limits are needed, in order to protect the water quality of surface and groundwater resources.

At the present time, most point-source discharges are strongly regulated by the Resource Management Act (RMA, 1991). This advent of the Resource Management Act, in 1991, has been able to significantly limit water pollution through industrial waste-water discharges (PCE, 2012).

Therefore, it is now vital to understand the impacts of water use in agricultural production systems on water resources, both in terms of water quantity and water quality.

2.5 Measuring water use and assessing its impacts in agricultural systems

As the pressures on limited water resources increase, there is a growing interest in measuring water use in production systems, and the environmental impacts of that water use (Hoekstra et al., 2011; Ridoutt et al., 2009; Segal and MacMillan, 2009). The first step to understanding consequences, or impacts of water use, is to measure the water use.

Assessment of impacts associated with water use is complicated, due to the dynamic nature of the water cycle, with its enormous temporal and spatial variations. In addition, the disparity in water quantity and water quality governs its value to people and ecosystems (Rijsberman, 2005). Unlike many other natural resources, water circulates naturally and dynamically through the atmosphere, pedosphere and lithosphere, while also changing its physical state between its vapour, liquid and solid forms. Within this circulation, water is continuously going through various processes, such as evaporation, transpiration, condensation and freezing, plus running off the ground surface, in addition to draining through the soil profile (Oki and Kanae, 2006). When water is abstracted for use, its properties can change, such as its solute concentration, biological status, and temperature. However, there are also attendant natural processes by which the properties of water are changed. Sometimes, this change is for the better, as with the attenuation of soluble and reactive agrichemicals when leaching through the soil profile. The rate at which water moves through the hydrological cycle, and its residence times in various storages systems, are determined by local weather, in particular the local rates of rainfall and evapotranspiration (Oki and Kanae, 2006). Therefore, any assessment of water resource availability, or impacts on water resources, needs careful consideration of the dynamic nature of the hydrological cycle, and the alterations of water state and status during that cycle.

2.6 Indicators of measuring water use and environmental impact of water use in production systems

Assessing water use along the supply chain of primary production, and understanding its impacts on water availability, is an integral part of assessing the sustainability of water use. Not surprisingly, therefore, the world's largest retailers, such as Walmart (Walmart, 2009), have announced plans to develop sustainable product indices of water use and metrics of water impacts . Furthermore, the Food Ethics Council of the United Kingdom has looked at the value of labels on food products for promoting sustainable water use. They have assessed the effectiveness of these labels for communicating with consumers, and to inform them about the impact of food production on the world's water resources (Segal and MacMillan, 2009). The multifarious issues surrounding the environmental sustainability of water use are, therefore, of growing importance for primary industries, not just from a strictly environmental point of view, but also for product marketing, in order to respond to consumers' questions and concerns (Sinha and Akoorie, 2010).

There are a number of methods that have been proposed to quantify total water use and the environmental impacts of that water use. These different methodologies display a lack of consistency in terminology. However, in this review, every attempt is made to distinguish the commonly used terms, by referencing the definitions.

2.6.1 The concept of virtual water and water footprint given by Hoekstra et al. (2009)

Traditionally, most of the water use analysis has focused on water withdrawals from rivers, lakes or aquifers and it is expressed on the basis of an enterprise, or facility. These types of assessments are merely production driven, rather than simply consumptive (Ridoutt et al., 2009). These methods lack a systems approach or life cycle focus. In an attempt to account more fully for water use in the system, the concept of 'virtual water' has evolved. The generally accepted definition of virtual water is that given by Allan in

1998, when he coined the term. It is the amount of water consumed in the production process of a product, which is also known as the water embodied in the product (Allan, 1998). This includes the water consumed in the production process, in addition to the water physically present in the product. It is also referred to as direct and indirect water use (Figure 2.3)

The water footprint concept introduced by Hoekstra and others (Hoekstra and Chapagain, 2007b; Hoekstra et al., 2011) is very similar to the virtual water concept, but it also considers the spatial and temporal dimension of where and when the water is appropriated. This spatial aspect is considered to be very important, since the potential environmental impacts related to water use are different from one location to another (Ridoutt et al., 2009).

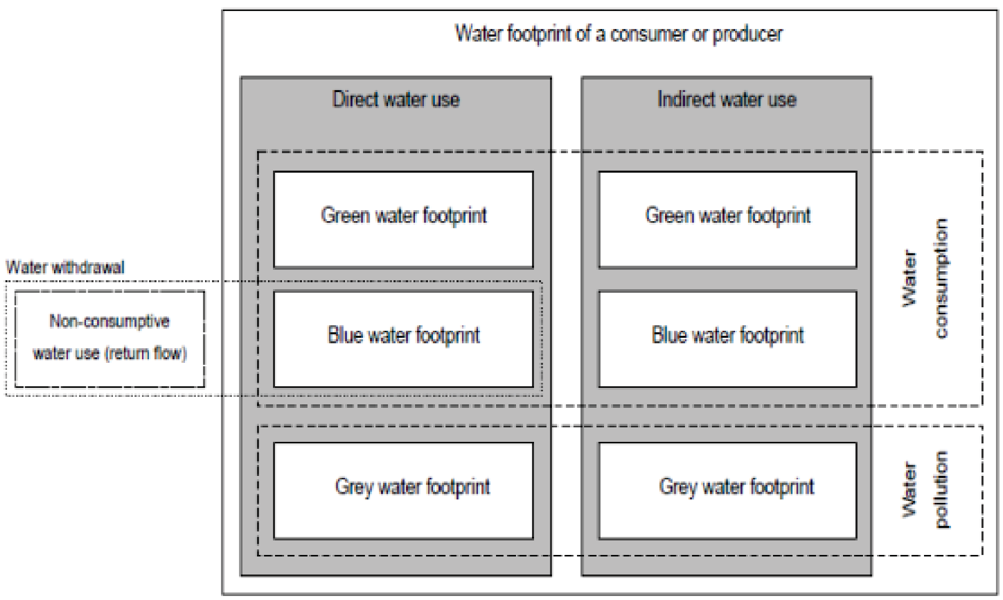


Figure 2.3 Schematic representations of the components of a consumptive water footprint. Source: (Hoekstra et al., 2009a)

The term 'water footprint' is used here to indicate the total water consumed during the production. However, some other authors have used the water footprint term as an indicator of the environmental impact of water use (Deurer et al., 2011; Ridoutt and Pfister, 2010; Ridoutt and Pfister, 2012). Therefore, for clarity, in this review the water footprint of consumptive use given by Hoekstra et al., (2011) is considered as a 'consumptive water footprint'.

The water footprint concept also distinguishes the different types of water used: the blue, the green and the grey (Hoekstra et al., 2011). This colouring step is vital, because different types of water fulfil different ecological functions, and different water qualities enable different uses.

The Blue Water

Blue water is the water resident in surface and groundwater resources. The blue water consumed, as a result of primary production, is considered as the blue (consumptive) water footprint (Hoekstra et al., 2011). Consumption refers to the volume of freshwater used, and then evaporated or incorporated into a product (WFN, 2010).

The opportunity costs of blue water use are generally higher, because blue water has a number of alternative uses. In addition, blue water has a supply cost, since it has to be pumped and transported through pipes or irrigation equipment before being applied to the crops. If a grower is paying the correct price for water, then their choice of crop would need to reflect a higher added-value, in order to cover the costs of using this water (Chapagain and Orr, 2009). However, this rarely happens.

The Green Water

The rainwater stored in the soil profile, as soil moisture, is considered as green water. The total green water consumed through evaporation and transpiration during production, plus the water incorporated into the harvested crop or wood, is considered as the green (consumptive) water footprint (Hoekstra et al., 2011).

Green water can only be used through land occupation. Therefore, the opportunity cost of green water is less, compared to that of blue water. The distinction between blue water and green water is important, since green water is only available for use by plants at the precise location where it occurs. Blue water is available generally for use in a wide range of systems which are managed by humans including, but not limited to, water use by plants (Canals et al., 2009).

The Grey Water

The term grey water is used to indicate water pollution. The grey water footprint of a product is an indicator of freshwater pollution that can be associated with the production of a product over its production chain.

It is calculated as the volume of freshwater that is required to assimilate the load of pollutants, based on existing ambient water quality standards (Hoekstra et al., 2011). According to Hoekstra et al. (2011), the grey WF can be calculated as given in the equation below.

$$WF_{Grey} = \left[L/\left(C_m - C_n\right)\right]/Y$$

Here, WF_{Grey} is the freshwater required [L/kg] to dilute the pollutant to an accepted water quality standard; L is the net-load of pollutants from the system [mg/ha]; and C_m is the maximum acceptable concentration [mg/L] for the pollutant given by the appropriate water quality standard. Here, C_n is the natural concentration [mg/L] of the pollutant in the receiving water body and Y is the yield [kg/ha]. The grey WF is expressed per functional unit (FU) of output within the system boundary.

According to this equation, the grey WF can be different within the same system, based on the water quality standard used and the pollutant considered. For example, the trigger value of nitrate-nitrogen (NO_3-N) for ecosystem protection suggested by the Australia and New Zealand Environmental Conservation Council is 7.2 mg/L (ANZECC, 2000), but the New Zealand Ministry of Health’s drinking water standard is 11.3 mg/L (MoH, 2008). Therefore, the grey WF for the same pollutant in the system, considering these different standards, will be different. It is, therefore, important to mention the standards used and the authority with which these standards are imposed. Hoekstra et al. (2011) suggest considering the most highly concentrated pollutant, in order to estimate the maximum water required to dilute all the pollutants. Ridoutt and Pfister (2012) mention that this critical dilution volume method becomes problematic, when assessing the compounds that do not have any documented acceptable concentrations. They also highlight the inability of this form of assessment to make use of advances in

Life Cycle Assessment (LCA) relating to the fate, exposure and effect modelling of emissions.

According to the Water Footprint Network (WFN) method (Hoekstra et al., 2011), the total water footprint is the sum of blue, green and grey water footprints. The appropriateness of aggregating different water colours has been strongly argued (Berger and Finkbeiner, 2012; Deurer et al., 2011; Zonderland-Thomassen and Ledgard, 2012), because the utility, impacts and opportunity costs of these various 'waters' can be quite different.

Environmental relevance of consumptive water footprint

The same amount of water consumption at locations of different water availability will have different environment, economic and social impacts. Therefore, the consumptive water footprint of different products are not readily comparable (Ridoutt and Pfister, 2010). In other words, a product with a lower consumptive WF could be more damaging to the environment than one with a higher consumptive WF, depending on from where and when the water is sourced. Furthermore, due to the differing proportions of blue, green and grey waters in the aggregated values, it is not possible to conclude that the lower water footprint is better, since the utility, impacts and opportunity costs of these various 'waters' can be quite different.

Generally, the term 'footprint' is used to indicate the potential environmental burden or impact of a certain process. From this point of view, the consumptive WF shows no clear relationship to the potential environmental and social harm from that process or product, unless the water stress or water availability at the location of the water source is considered. The LCA-based approaches take this aspect into account, in order to understand the environmental impact of water consumption (Canals et al., 2009; Pfister et al., 2009; Ridoutt and Pfister, 2010).

Regarding the consumptive WF, Hoekstra et al. (2011) have also added the concept of calculating water footprint impact indices, in order to reflect the local environmental impacts. This concept follows a similar framework of the LCA-based WF methodologies. According to this framework, an impact indicator for each water colour can be calculated by multiplying a blue and green water footprint by a blue and green-water scarcity measure, in a particular catchment for a particular time period. In regards to the grey WF impact indicator, it is proposed to multiply the grey WF by the water pollution level. However, the method is unclear as to how the scarcity and pollution levels are to be quantified.

2.6.2 Life Cycle Assessments (LCA) based water footprinting methods

Life cycle assessment is a widely accepted environmental management tool that can be used to quantify the various environmental impacts of a particular production process (Berger and Finkbeiner, 2010). When assessing the environmental performances of a product, by means of LCA, attention has generally been given to the emission of greenhouse gases, toxic substances, or the energy consumed throughout the life cycle of the product. The use of freshwater during the product life cycle had often been neglected, until recently. This is perhaps because LCA was traditionally used to assess industrial products, which require little water in their production (Berger and Finkbeiner, 2010), compared to agricultural products. In recognition of this limitation (Koehler, 2008), the LCA community has recently given considerable attention to including water use in the life cycle of products, and to assessing the related environmental impacts (Canals et al., 2010; Canals et al., 2009; Pfister et al., 2009). The LCA practitioners have also recognised the methodological challenges in both the life cycle inventories, which account for the quantity of water use, in addition to the life cycle impact assessments used to quantify the environmental impacts of water use.

LCA approaches recognise the need to consider the impacts of freshwater use on the environment, throughout the entire product life cycle (Canals et al., 2009; Koehler, 2008; PAS:2050, 2008). By identifying limitations to an understanding of the local impacts, through using the consumptive water-use measure of WF, a number of different methods have been proposed, in order to quantify impact orientated WFs within the standard framework of LCA (Canals et al., 2009; Pfister et al., 2009; Ridoutt and Pfister, 2010).

2.6.2.1 The LCA method proposed by Canals and others (2009, 2010)

The LCA approach by Canals et al. (2009) highlights the need to distinguish and quantify both the evaporative and non-evaporative uses of water within a system boundary

(Figure 2.4). They also consider the role of land-use changes that can lead to changes in the availability of freshwater in inventory modelling (Canals et al., 2009).

This method recognises two main pathways by which the use of freshwater can impact on water availability. They are: 1) the fresh water ecosystem impact (FEI); and 2) the freshwater depletion (FD).

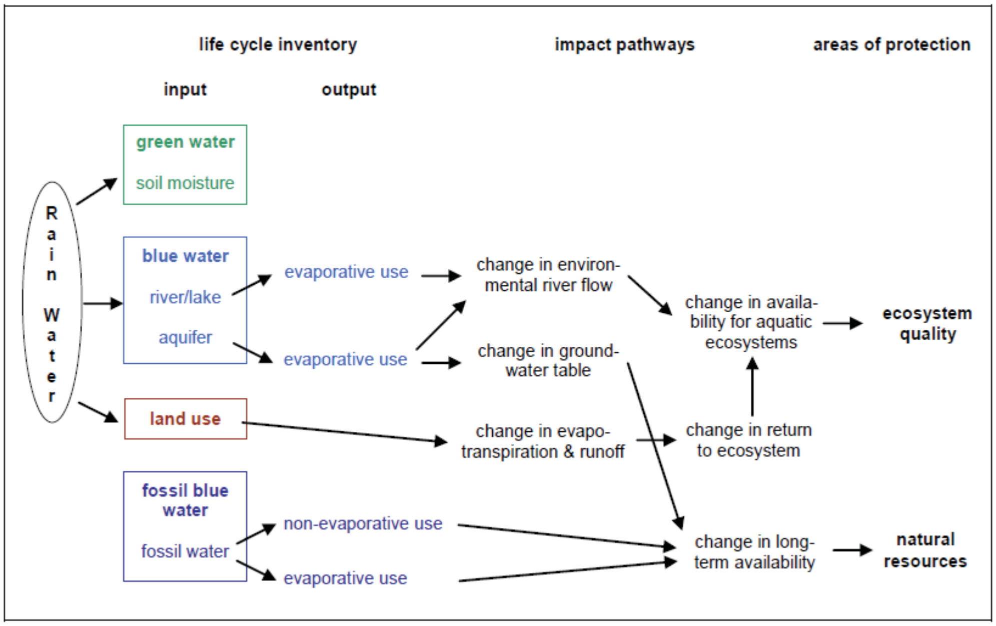


Figure 2.4 Inventory requirements and impact pathways resulting from different types of water use as proposed by Canals et al. (2009). Source: Berger and Finkbeiner (2010).

Freshwater ecosystem impact (FEI)

The FEI is proposed as a means to indicate the impact of blue-water consumption on the reduction in the availability of water for ecosystem functioning that affects ecosystem health (Canals et al., 2009). In regards to the FEI, the authors explored possible characterisation factors (CF), and then proposed a 'water stress indicator'. This is an indicator of the water resources available for further human use, after 'reserving' certain resources that are necessary for the ecosystem (Smakhtin et al., 2004). When the water stress indicator is calculated at a river basin level, the available water for human use is the difference between the total amount of water available in the basin, and the estimated environmental water demand needed to maintain the basin's ecosystem functions. However, even though it is calculated at a river basin level, it does not fully reflect the level of stress on the local water resources. This issue will be demonstrated in the case study of New Zealand wine presented in Chapter 5 of this thesis.

Freshwater depletion (FD)

The FD indicator quantifies the impact of the extraction of groundwater that results in a potential reduction of the long-term availability of freshwater for future generations. In regards to this category of impact, the proposed characterisation factor is the abiotic depletion potential (ADP). However, it has been recognised that the calculation of ADP for groundwater is still challenging, since most groundwater resources are rarely quantified in terms of their relative abundance and in relation to their potential uses (Jeswani and Azapagic, 2011).

In an LCA framework, Canals et al. (2009) point out that green water is used by natural ecosystems, regardless of the production system being considered. Therefore, they propose that the impact of green-water use should be excluded.

2.6.2.2 The LCA method of Pfister et al. (2009)

The approach of Pfister et al. (2009) enables a comprehensive impact assessment of freshwater consumption, at both the mid-point and end-point level of LCA. A water stress index (WSI), which indicates the impacts of water consumption in relation to water scarcity, is considered as the mid-point characterisation factor. It should be noted that this WSI is different from the water stress indicator; that is, the CF for FEI suggested by Canals et al. (2009). The WSI here is based on the withdrawal-to-availability (WTA) ratio of water. It can be applied at any spatial scale. However, Pfister et al. (2009) recommend that water use impacts be assessed at the watershed level.

The end-point impact category focuses on three areas of protection related to water consumption. These are: 1) human health; 2) ecosystem quality; and 3) the depletion of freshwater resources. All three areas of protection are quantified through long cause-and-effect-chains. For example, the impact of water consumption on human health is modelled through a cause-and-effect chain starting from the WSI, and then the percentage of agricultural water use, in relation to the total water used to quantify the water deprivation, due to agricultural purposes. Subsequently, this approach incorporates the per-capita water requirement needed to prevent malnutrition, in order to quantify the annual number of malnourished people. Thus, the overall human health effects resulting from a certain number of malnourished people can be quantified.

However, this impact modelling via long 'cause-and-effect' chains involves a number of assumptions. This results in a large degree of uncertainty. This incertitude might eventually mask links to the main impacts and thereby limit the usefulness of the outcome for enacting improvements through WF reductions. This situation can arise because these assumptions are often based on global-scale values, rather than those at the local scale, which are of paramount importance, especially for water related impacts. This is a common limitation with most LCA-based assessments, and this will necessitate further improvements.

2.6.2.3 The Ridoutt and Pfister (2010) LCA method

The LCA method of Riddout and Pfister (2010) seeks to understand the impact of water consumption in relation to the degree of water scarcity at the location where the water is sourced. According to this WF method, the impact of green-water use is also considered negligible, since green water is used in natural ecosystems, regardless of the flows within the system boundary. Therefore, this stress-weighted water footprint is expressed as only an indicator of the impact of blue-water consumptive use. This is calculated by multiplying blue-water use by the regional water stress index (WSI), as proposed by Pfister et al. (2009)(Figure 2.5).

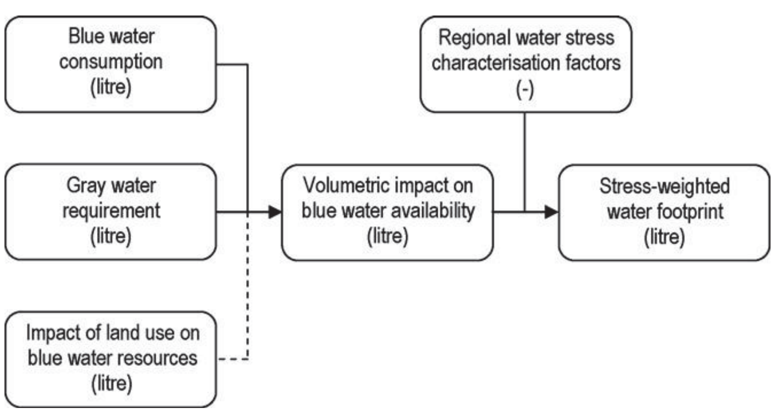


Figure 2.5 Method of calculating product water footprints incorporating water stress characterisation factors. Source: Ridoutt and Pfister (2010)

The characterisation factor WSI, used in this method to convert the volumetric water use into a weighted indicator, is estimated from water withdrawal, relative to the availability (WTA) ratio at that location. As Berger and Finkbeiner (2012) have pointed out, the WTA can be misleading, since not all water withdrawals are actually consumed. Hoekstra et al. (2012) note that approximately 40% of agricultural water withdrawals are returned to local water resources.

Recently, Ridoutt and Pfister (2012) presented a new framework for a single-score indicator, by combining consumptive and degradative water use. This new method from Ridoutt and Pfister (2012) is useful only for comparative purposes, since it references the WF to a global average, and forms a stand-alone, single-score indicator. Therefore, it is less useful for understanding the impact on local water resources.

Although LCA-based water footprinting methods have been developed rapidly over the recent past, currently there are few data in the scientific literature, or in key databases such as Ecoinvent 2.0 (Ecoinvent, 2012), which are commonly used in LCA. There is a mismatch with these and many of the methodologies recently proposed for water footprinting.

One widely understood limitation of LCA-based water footprinting assessments is their reliance on LCA databases for inventory modelling. Often these are lacking, or incomplete, especially in terms of the key aspects of water-use assessments (Berger and Finkbeiner, 2010). These aspects are listed below.

1. It is unclear whether all relevant inflows and outflows are included in the inventories. This is especially so when abstractions are used in inventories without considering return flows. It is the net water-balance result that is critical for the local hydrology.
2. There is a dearth of geographically relevant data. It is very important to use inventory data with a high spatial resolution in terms of water-related assessments. Water use and its impacts are highly localised and temporally variable.

3. Water quality information about inflows and outflows is scarce. This aspect is critical, since the utility of water resources is highly dependent on their quality.

Consequently, there is a lack of consistency between the various databases currently available. The accuracy of extant inventories is in doubt, with respect to both water quantity and quality.

In summary, the LCA-based methods do suggest indicators to represent the potential environmental impacts associated with water. They can facilitate somewhat meaningful comparisons to be made between different products from various locations of water availability. However, accurate inventories and localised characterisation factors are needed, in order to ensure the veracity of these assessments.

Both the WFN and the LCA-based methods are based on consumptive water use. Methods of water footprinting, based only on water consumption through evapotranspiration (ET) and considering ET as a 'loss', are questionable in understanding local hydrological impacts . Certainly, evaporation and transpiration play a vital role as drivers of the hydrological cycle. The major portion of the water that leaves the system through evapotranspiration returns locally as precipitation (Van der Ent et al., 2010). In a recent study on the length and time scales of atmospheric moisture recycling, van der Ent and Savenije (2011) found that evaporation recycling into the same ecosystem occurs within a short time (ranging from 3 to 20 days in temperate climates) and distance scale (500-5000 km). In addition, water consumption through transpiration plays a key role in crop production. It enables absorption and transportation of mineral nutrients and water from the soil to roots and shoots. Therefore, crop yield is well correlated with transpiration (Nahle and Kunz, 2012). However, it is emphasised here that assessing and quantifying ET provides useful information for water use and management from the orchard to catchment and beyond, but only when considered together with other water flows and storages.

2.7 Use of different methods and data availability in New Zealand

While there is much interest in the development of international standards for water footprinting, there are number of established methods. These methods can basically be categorised into two groups: consumptive-based water footprinting and LCA-based water footprinting.

There are a few elements common to both, such as consideration of the contribution from supply-chain inputs and the identification of different sources of water used, namely blue and green water. From New Zealand's perspective, the databases on water use and consumption, or impacts for major inputs used in the supply chain of the products, are poorly developed and at best sketchy (Clothier et al., 2009). For example, electricity is a dominant input in almost all product supply-chains. Furthermore, more than half New Zealand's electricity is sourced from hydropower (EDF, 2010). It has been claimed that hydropower has a large water footprint relative to other sources of energy (Gerbens-Leenes et al., 2009). These claims are based on global scale assessments, and there have not been any detailed systematic studies on the water footprint of hydroelectricity, especially at the power-station scale.

In both consumptive and LCA-based water footprinting methods there are difficulties in identifying the local impacts of the system under consideration. This is simply because these methods only consider the water loss through evaporation, without considering the supply side of the equation. The statement above notwithstanding, the LCA-based impact assessment has a limited ability to capture local impacts through consideration of the local water scarcity and availability within a region or area. However, at the present time, there are not any water stress assessments based on local water-resource availability for regions within New Zealand. This is a major impediment to the use of this LCA-based method.

2.8 Summary

2.8.1 Summary and highlights

1. Assessing the water use impact of agricultural products is important, both from a global and New Zealand local perspective.
2. Water footprinting is becoming an emerging tool for assessing environmental impacts related to water use by supply chains, but there is not any agreed methodological framework, as yet. However, it has been well established that

assessments need to consider a life cycle approach, through consideration of direct (foreground) and indirect uses, and impacts through secondary inputs (background).

3. There has been a great deal of debate about whether the water footprint should be a volumetric indicator of consumptive water use, or a concept that can indicate the environmental impacts arising from water use. However, according to the working draft of the ISO water footprinting methodology, in order to be comparable with other internationally standardised footprinting concepts (i.e carbon footprint), the WF should indicate potential environmental impacts related to water use.
4. There have been a number of methods proposed for water footprinting. These methods have different foci. However, water footprinting based only on consumptive water use are limited in understanding the impact on water resources. Therefore, methodological improvements are needed, in order to assess the net water use.
5. It is well understood that water related impacts are highly local. This implies that these assessments need to consider local inventories. This situation is challenging and resource intensive. Therefore, most of the current water footprinting assessments have been based on global or regional databases.
6. There have been several studies that have discussed the advantages and limitations of proposed methods. However, there have not been any studies that focussed on evaluating these methods for their usefulness to inform the key stakeholders.

2.8.2 Focus for the study

1. As discussed above, agricultural production has widespread impacts on New Zealand’s water resources, both in terms of water quantity and quality. Therefore, assessing and quantifying these impacts is vital, in order to derive potential improvement options (Chapters 4, 5, 6 and 7).
2. Water footprinting is increasingly becoming a widely used tool, but there is a lack of consistency and agreement regarding the methodology. Comparative methodological assessments are needed to further develop international standards (Chapters 3 and 5).

3. In order to serve the purpose of understanding local impacts, assessment based on local hydrological status is imperative, especially for agricultural products (Chapters 4, 5, 6 and 7).
4. The WF analysis needs to have a life-cycle focus, in order to provide a contribution from the appropriately quantified inputs which are needed for an accurate assessment (Chapters 3, 4, 5 and7).
5. There are well understood limitations of extant water inventory databases. Therefore, there is a need to develop locally based inventories, especially for the commonly used inputs (Chapter 3, 4 and 7). This is especially so for electricity, which is a major input into the primary-production supply chain, and because hydropower is a major component of New Zealand's electricity mix (Chapter 3).
6. There is a need to evaluate the outcome of the various methods based on their usefulness to the key stakeholders, such as producers and growers, plus resource regulators and the wider community, due to their role in demanding, or achieving, reductions in the impact through improvement options. This is the key to reducing the overall environmental impacts of water use throughout the primary production supply chain (Chapter 5).
7. This thesis has provided new knowledge and ideas in relation to the six points above, and it also contributes to a critical review of the various methodologies and outlined current exigencies for the water footprinting community.

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CHAPTER 3

The water footprint of hydroelectricity: A methodological comparison from a case study in New Zealand

As discussed in Chapter 2, it has been well established that product footprinting needs to consider the full life cycle of the product, including the impacts on water resources and their scarcity from the inputs used in the production. Electricity is a key input for major parts of the lifecycle stages of agricultural products grown in New Zealand. Furthermore, the lakes impounded directly, or even indirectly, by New Zealand's hydroelectricity dams form a significant part of New Zealand's natural capital stocks of water. Within this Chapter are presented the results of a detailed assessment of the water footprint of New Zealand's hydropower, which is the major source of New Zealand's electricity mix. A new method of water footprinting using net water balance is introduced, along with the comparison of two other existing water footprinting methods. The hydrological method we describe here for the first time, in this thesis, will then be applied in later Chapters to quantify the water footprint of a bottle of wine at the winery gate (Chapters 4 and 5) and to assess the water footprint of fresh ware potatoes at the packhouse gate (Chapters 6 and 7).

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3.1 Abstract

Hydroelectricity has been rated to have a large water footprint (WF) on global average. We assessed the WF of hydroelectricity by three different methods using New Zealand as a case study. The first (WF_1) and second (WF_2) methods only consider the consumptive water use of the hydroelectricity generation system, while our third method (WF_3) accounts for the net water balance. Irrespective of the method, the WF of

New Zealand's hydroelectricity was found much smaller than the commonly cited international value of 22 m^3/GJ. Depending on the method, the national WF ranged from 1.55 m^3/GJ (WF_3) to 6.05 m^3/GJ (WF_1). The WF_3 considers the net water balance including rainfall, which is the key driver for replenishing water resources. It provides meaningful information that helps our understanding of the differences of the WF in locations, which are diverse in terms of water resource availability. We highlight the effects of local climatic differences and the structural specifics of a hydroelectricity scheme on the WF. The large variation in the WF of hydropower across New Zealand illustrates the inappropriateness of using global average values. Local values, calculated using our hydrologically rational method, must be used.

Keywords: **Water-energy nexus, Hydropower, Hydroelectric dams, Renewable energy, Virtual water, Ecosystem services, Hydrology**

3.2 Introduction

Water and energy are two critical necessities for modern civilizations. Freshwater is one of the planet's most valuable resources, being an essential life-sustaining element that cannot be substituted for (Koehler, 2008). At the same time, freshwater is increasingly becoming a scarce resource. Across the globe, there are ominous hydrological signs, such as groundwater depletion, lowering of river flows, and the deterioration of water quality. This indicates that current levels of water use exceed sustainable limits in many parts of the world (Postel, 2000). Furthermore, companies which produce water-intensive products and services around the world are facing significant water-related risks (Lambooy, 2011). Energy is considered to be the life-blood of technology and development (Khan and Hanjra, 2009). As the world's population grows, the demand for both freshwater and energy is increasing faster than ever. Competition for freshwater and energy will become one of the defining issues of this century (IEEE, 2010). According to Beddington (2009), by 2030 we will need to be producing 50% more food. At the same time, we will need 50% more energy, and 30% more freshwater (Beddington, 2009). The challenge is to meet these additional food, energy and freshwater demands in a way that does not affect natural capital stocks and the ecosystem services that flow from them.

Energy and freshwater resources are intricately and intimately connected. Energy is required to operate modern water-supply systems and purification facilities. Without the input of substantial amounts of energy, shifting large quantities of water from water-rich to water-poor regions, desalinization of brackish or seawater, and the pumping of groundwater aquifers and surface water for irrigation would all be impossible (Gleick, 1994). On the other hand, the production and use of energy often requires significant quantities of water. In almost every type of power plant, water is a major hidden input. Water cools the hot steam of thermal plants, and it turns the hydroelectric turbines. It is a vital ingredient in biofuel crops, and brings geothermal energy as steam from the depths of the earth (IEEE, 2010). With increasing frequency, we will need to assess energy production with reference to water protection, whilst also considering our urgent need to reduce greenhouse gas emissions.

Environmental footprints have been widely used in recent years as indicators of resource consumption and waste creation (Hammond, 2006), or in other words, they provide measures of the impacts of human activity on the environment. The water footprint has attracted interest as a metric that indicates the use of freshwater resources and its impacts. In the current methodology, the water footprint is defined as the volume of freshwater used directly, or indirectly, in the production of a good or service (Hoekstra and Chapagain, 2007b). The term ‘used’ considers two facets: the water consumed (evaporated) and the water polluted throughout the production.

Among different sources of energy, hydropower is very attractive because of its low CO_2 emissions (Herpaasen et al., 2001), and its renewable nature. Sims (2004) has shown that hydropower can save 229 g C/kWh (63.61 kg C/GJ) carbon emissions compared with a conventional coal-fired power. But hydropower has been claimed to have a large water footprint per unit energy, relative to other sources of energy (Gerbens-Leenes et al., 2009). However, there has not been a detailed systematic assessment of the water footprint of hydroelectricity to substantiate this claim.

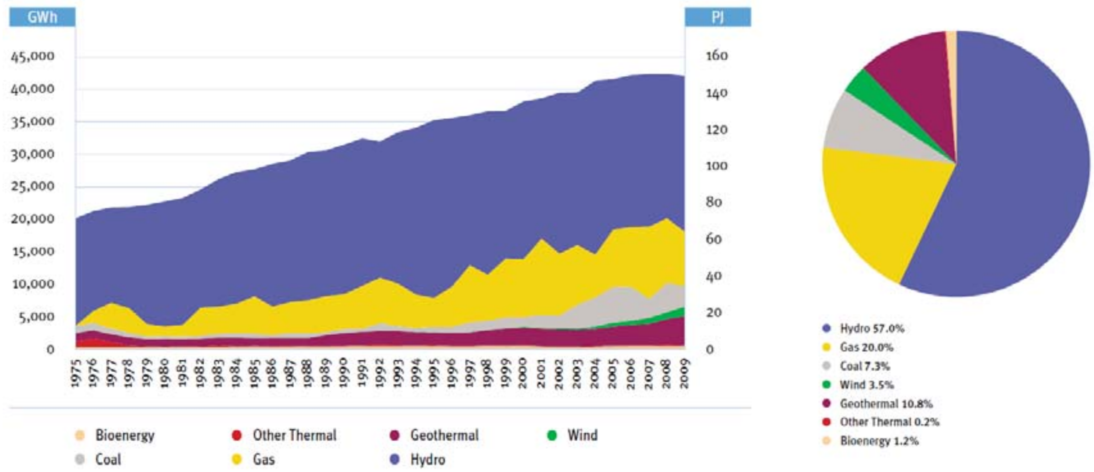

Figure 3.1 New Zealand's electricity generation by fuel type: for past 34 years (left) for 2009 (right) Source: New Zealand Energy Data File, 2010

In New Zealand, the major portion of electricity is generated by hydropower (Figure 3.1, left), which has been the mainstay of New Zealand's energy system for over 100 years. In 2009, 57% of total energy generation in New Zealand (Figure 3.1 right) was from hydroelectricity (EDF, 2010). The current recommendation is to include the water footprint of energy in the assessment of the total water footprint of products and services if they are sourced from bio-fuel, or hydropower (Hoekstra et al., 2009a). For accurate water-footprint assessments of many of New Zealand products and services, accounting for the water footprint of hydropower is very important. As many of New Zealand's export products are marketed using a 'clean green' image, the water footprint of hydroelectricity will have ramifications for the competitive advantage of New Zealand's export products.

The objectives of this study were threefold. Firstly, we aimed to assess the impact of hydroelectricity generation on water resources by using a water footprint concept considering New Zealand as a case study. In addition to the water footprint assessment based on consumptive water-use (WF_1 and WF_2), we attempted to develop a hydrologically rational water-footprint assessment for hydroelectricity generation (WF_3). Secondly, we sought to quantify the influences of regional climatic conditions and structural variables (e.g. reservoir surface area) on the WF of hydroelectricity. Thirdly, we attempted to estimate the WF of a unit of New Zealand hydroelectricity as delivered to the national grid and compare this with reported values.

3.3. Methodology

3.3.1 Description of New Zealand hydroelectric power plants

In this study, all major hydroelectric power plants in New Zealand (Fig 3.2) were considered. These power plants account for more than 95% of hydropower generated in the country (EDF, 2010). Figure 3.2 shows the geographical locations of the plants. The hydropower stations in the North Island are clustered together in the central part of the island, while the plants in the South Island are more widely scattered.

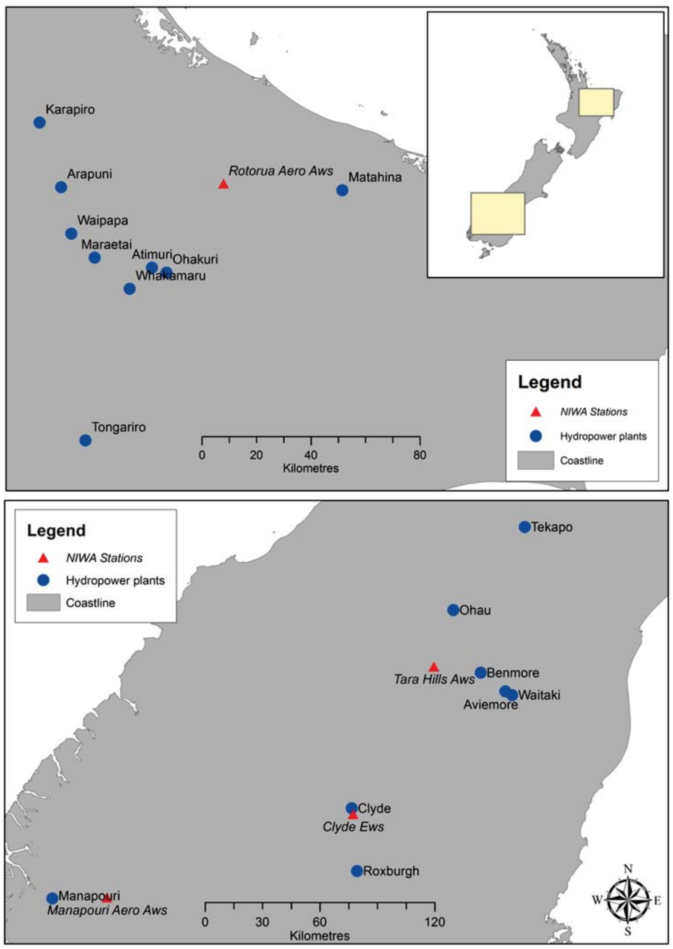

Figure 3.2 Locations of hydropower plants and NIWA weather stations considered in the study.

3.3.2 Three methods to quantify the water footprint of hydroelectricity

The science of water footprinting is still in its infancy, and methodologies are still being developed and revised. There is no well-documented and accepted methodology yet to quantify the WF of hydroelectricity. In this study, we considered three different methods to assess and discuss the water footprint of hydroelectricity.

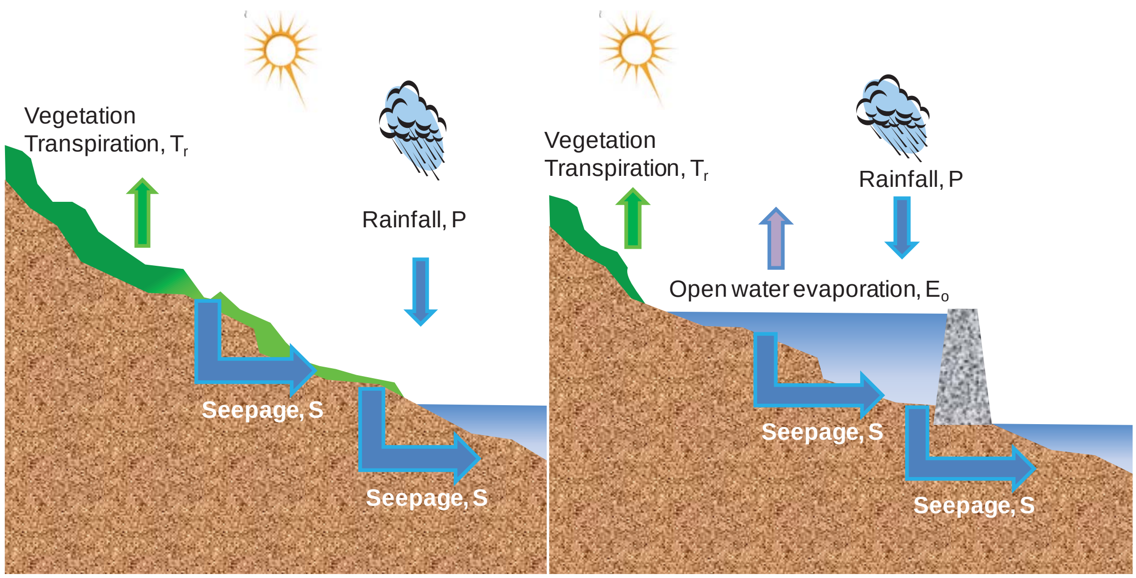

Figure 3.3 Schematic diagram showing different hydrological components and landscape features before (left) and after (right) a hydroelectric dam. Through flow is ignored.

3.3.2.1 WF_1: Consumptive water use

In the first method, we follow the definition of the water footprint given by Hoekstra and Chapagain (2007). This essentially accounts for the water consumed in the process under consideration. For hydropower generation, the water footprint (WF_1) (m^3/GJ) can be calculated as the evaporative water loss from the surface of the reservoir divided by the energy produced by that hydropower plant,

$$WF_1 = E_0 / P_w$$

Here, E_0 is the annual open-water evaporative loss from the reservoir (m^3) and P_w is the annual energy production of the power plant (GJ). This definition has been used by Gerbens-Leenes *et al.* (2009) to estimate the WF of hydropower on a global average basis.

3.3.2.2 WF_2 : Net consumptive use

The second approach also considers consumptive water use, but it compares the consequences of land use changes created by the dam. Building of a dam results in the replacement of vegetation by a free-water surface (Fig 3.3). Thus, evapotranspiration from the vegetation is replaced by open-water evaporation from the reservoir. Taking this into account, the WF_2 (m^3/GJ) considers the net evaporative water loss from the area occupied by the reservoir,

$$WF_2 = (E_0 - T_r)/P_w$$

Here, T_r is the amount of water lost by transpiration (m^3/ year from the antecedent vegetation that would have occurred in the absence of the dam). The vegetation was considered full-cover, so soil water evaporation was ignored.

Evapotranspiration was determined using a soil-water balance calculated using daily rainfall, runoff and soil-water deficit data from the 'National Institute of Water and Atmospheric Research' (NIWA) database (NIWA, 2012). It was assumed that pasture was the vegetation before the dam was constructed. Even though pasture would have been the dominant vegetation type for most of the areas covered by the reservoirs, for some of the locations in the North Island the vegetation would have been of mixed types. However, we found that the vegetation type does not make much difference to the calculations. Our evapotranspiration estimations are based on daily soil-water deficit, rainfall and runoff data.

3.3.2.3 WF_3 : Net water balance

In the third method, we moved beyond the simply consumptive-use definition of the water footprint. A simple water balance was used to estimate the water footprint considering both water inputs and outputs from the reservoir.

Factors considered in the water balance

Evaporation is the most obvious consumptive use of water from the hydroelectric reservoirs. Seepage losses through the porous geology underlying hydroelectric reservoirs may also be considered as a consumptive use of water. However, seepage and evaporative losses have important qualitative differences. Water loss by evaporation usually leaves the hydraulic basin and, thus, is a true loss. But water loss through seepage generally remains within the basin, and is highly likely to become available again downstream, or it may recharge underlying ground water resources (Gleick, 1994). Accordingly, we did not consider the seepage to be a true loss from the reservoirs (Fig 3.3). Likewise, the water used to turn the turbines is not considered as it returned to the river, accordingly it was not shown in the Figure 3.3. It is considered as simply a through-flow.

In the WF_3 method, we analysed the net water balance taking into account the water leaving and entering the surface of the reservoir, namely evaporation as the output and rainfall as the input. Therefore, the net water balance was calculated as,

Output- Input= Evaporation- Rainfall.

The WF_3 (m^3/GJ) for hydroelectricity therefore is the net loss of water from the reservoir per unit energy produced in the hydroelectric plant estimated as,

$$WF_3 = (E_0 - P)/ P_w$$

Here, P is the annual volume of rainfall falling on the reservoir (m^3).

The E_0, T_r and P volumes were calculated on an annual basis by multiplying the annual average open-water evaporation, evapotranspiration and rainfall by the respective reservoir surface area. The three WF methods (WF_1, WF_2 and WF_3) were used to calculate the WF values of hydroelectricity generated by different hydropower plants in the North Island and the South Island of New Zealand (Table 3.1 and 3.2).

3.3.3 Data collection

The water loss through evaporation and water input through rainfall, are directly related to the surface area of the reservoir. The areas of the reservoirs associated with hydropower plants were determined using results from a Geographic Information Systems (GIS) analysis. We assumed that the surface area of the water stored in the reservoir was constant throughout the year. In this analysis, we only considered the water storage area of the reservoir, and not the whole catchment area that drains into the reservoir. Also, in this study, we considered each reservoir as a single unit.

Therefore, their origin, occurrence in series, and their interdependencies were not affected in our calculations of the WF.

Evaporation of water also varies with temperature, wind speed and the humidity of the air above the reservoir (Gleick, 1994). To capture the climate-induced variability due to highly variable climates across the country, we sourced open-water evaporation and evapotranspiration data from the nearest official meteorological stations to the reservoirs in the network maintained by the NIWA. Four NIWA stations were selected (Figure 3.2): Rotorua Aero Aws (Automatic Weather Station) from the North Island; and Tara Hills, Clyde Ews (Electronic Weather Station), and Manapouri Aero Aws, from the South Island. These stations record, amongst other things, rainfall, estimated open-water evaporation, and reference evapotranspiration. For this analysis, daily open-water evaporation and evapotranspiration rates were considered for the ten-year period from 2000 to 2009 (Figure 3.4). There were seasonal differences in evaporation and evapotranspiration rates. However, when annual averages were considered, the seasonal differences cancel out.

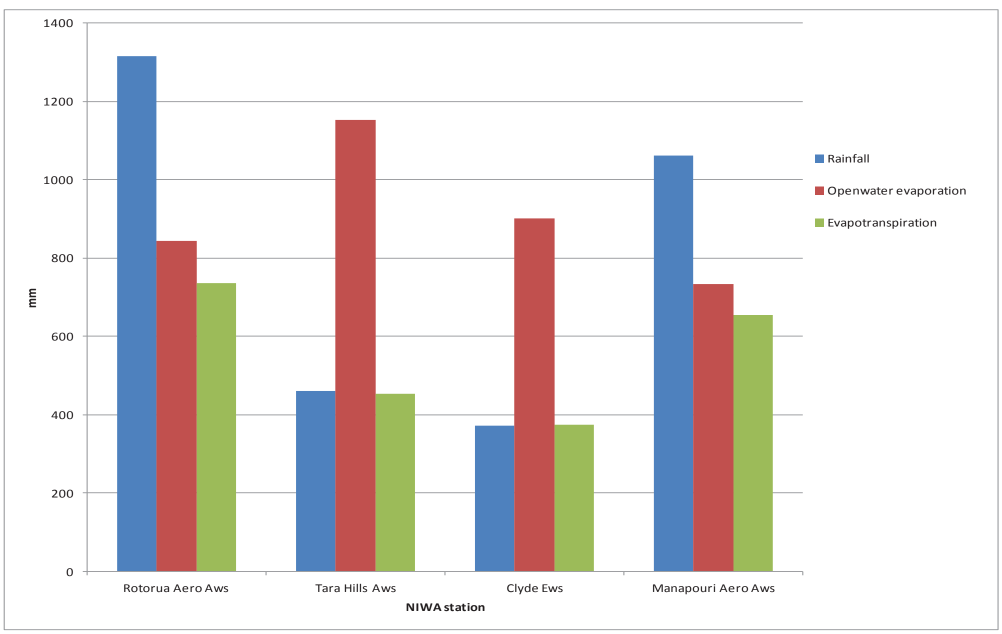

Figure 3.4 Annual average (2000-2009) rainfall, open water evaporation and calculated evapotranspiration at selected NIWA weather stations.

The water use per unit energy output also depends on the energy-generation efficiency of the plant. Therefore, annual average energy generation from each hydropower plant was collected and considered separately. The water footprint of hydroelectricity generated by each plant was determined separately using the three methods described above, and then the weighted-average water footprints for the North and the South Islands were calculated. Finally, a single water footprint value for New Zealand was determined based on relative contribution of power generation in both the Islands to the national grid (Table 3.4).

3.4 Results and Discussion

3.4.1 Effect of hydroelectricity generation on freshwater resource availability

In all three methods used to quantify the water footprint, evaporation was considered as a loss of water from the water storage reservoirs. But it is also important to note that evaporation and evapotranspiration are major driving forces of the functioning of the hydrological cycle. However, in the calculation of water footprints, these two are considered as consumptive uses of water, and therefore, they are considered as losses from the ecosystem under consideration.

The three water-footprint values for the different hydropower plants in the North Island and the South Island of New Zealand are shown in Tables 3.1 and 3.2. The method WF-1 follows the definition used by Gerbens-Leenes *et al.* (2009) to estimate the WF of hydropower on a global average basis, where global evaporation from artificial surface water reservoirs was divided by the global hydroelectric generation for the year 1990 (Gerbens-Leenes et al., 2009). The WF_1 values estimated using local climate and energy production values in this study were small compared with the value of 22 m^3/GJ given by Gerbens-Leenes *et al.* (2009). The Tekapo power plant in the South Island has the highest WF_1 value of 32.48 m^3/GJ (Table 3.1). This is because of its larger surface area and higher evaporation rate in relation to power generation (Table 3.1). In contrast, the Waipapa hydropower plant in the North Island showed the lowest WF_1 value of 0.75 m^3/GJ due to its smaller surface area and lower evaporation rate (Table 3.2). In response to highly variable climatic conditions and structural configurations, there was high variability in the WF_1 values within the South Island power stations (CV of 115%) (Table 3.1), as compared to those in the North Island (CV of 53%) (Table 3.2).

Table 3.1 The water footprint of hydropower generated in the stations in the South Island of New Zealand

Hydro power plant	NIWA station	Annual energy output (GWh)	Surface Area (ha)	Rainfall (mm/y)	Evapotranspiration (mm/y)	Open-water evaporation (mm/y)	WF -1 (m^3/GJ)	WF-2 (m^3/GJ)	WF-3 (m^3/GJ)
Tekapo	Tara Hills	960	9733.5	460	454	1153	32.48	19.69	19.51
Ohau	Tara Hills	3090	5898.9	460	454	1153	6.11	3.71	3.67
Benmore	Tara Hills	2200	7523.9	460	454	1153	10.95	6.64	6.58
Aviemore	Tara Hills	940	2804.0	460	454	1153	9.55	5.79	5.74
Waitaki	Tara Hills	500	622.0	460	454	1153	3.98	2.42	2.39
Clyde	Clyde Ews[a]	2100	983.7	371	374	902	1.17	0.69	0.69
Roxburgh	Clyde Ews[a]	1650	528.8	371	374	902	0.80	0.47	0.47
Manapouri	Manapouri Aero Aws[b]	4800	13840.5	1061	653	733	5.87	0.64	-2.63
CV (%)							115	127	148

[a] Ews-Electronic weather station [b]Aws- Automatic weather station

Table 3.2 The water footprint of hydropower generated in the stations in the North Island of New Zealand

Hydropower plant	NIWA station	Annual energy output (GWh)	Surface Area (ha)	Rainfall (mm/y)	Evapotranspiration (mm/y)	Open-water evaporation (mm/y)	WF -1 (m^3/GJ)	WF-2 (m^3/GJ)	WF-3 (m^3/GJ)
Ohakuri	Rotorua Aero Aws[a]	400	854.2	1315	737	844	5.01	0.64	-2.80
Atimuri	Rotorua Aero Aws	305	193.9	1315	737	844	1.49	0.19	-0.83
Whakamaru	Rotorua Aero Aws	486	667.6	1315	737	844	3.22	0.41	-1.80
Maraetai	Rotorua Aero Aws	855	438.6	1315	737	844	1.20	0.15	-0.67
Waipapa	Rotorua Aero Aws	330	104.8	1315	737	844	0.75	0.09	-0.42
Arapuni	Rotorua Aero Aws	805	883.2	1315	737	844	2.57	0.33	-1.44
Karapiro	Rotorua Aero Aws	490	725.8	1315	737	844	3.47	0.44	-1.94
Tongariro	Rotorua Aero Aws	1350	1601.2	1315	737	844	2.78	0.35	-1.55
Matahina	Rotorua Aero Aws	300	235.8	1315	737	844	1.84	0.23	-1.03
CV (%)							53	53	53

[a]Aws- Automatic weather station

The WF_2 considers the land use before and after the construction of a dam. A similar approach has recently been discussed in the water footprinting assessment of beer, and this was termed the 'net green water' loss (SABMiller and WWF-UK, 2009). In the case of beer, a net approach considered the evaporative water loss from the cultivated crop as compared to the antecedent natural vegetation. However, this approach raises the question as to what is 'natural'? This question would also apply here. We have considered the antecedent vegetation to be shallow-rooted pasture which is prone to drought. This is what would have covered these areas immediately prior to dam construction. But many centuries ago, prior to human colonization, the natural vegetation would have been deep-rooted, evergreen, subtropical forest.

The WF values estimated using the WF_2 method ranged from 19.69 to 0.09 m^3/GJ and they were lower than the WF_1 values (Table 3.1 and 3.2). The WF_2 also does not provide meaningful information that helps us to understand the impact of WF of hydropower generation in different locations.

Arguably, a better hydrological understanding of the impacts of hydropower generation on water resources is gained by considering the net balance of the water inputs and outputs of the reservoirs. The method WF_3 considers the water input through rainfall, as well as water loss through evaporation from the reservoirs. When the water input is greater than the output, this results in a negative value for the water footprint indicating net water surplus in the reservoir, while WF_1 and WF_2 always yield positive values, as they only account for the water loss. Arguably, the net water surplus in the reservoir is of high natural capital value as, downstream, it could be used for agricultural, industrial, domestic, cultural and riparian services that would otherwise not have been possible.

By considering continental precipitation-recycling ratios, Van der Ent *et al.*, (2010) found that on average 57% of terrestrial evaporation returns as precipitation over land. In many areas this can exceed 80%, such as in southern Amazonia, the Congo and Eurasia (Van der Ent et al., 2010). Precipitation is therefore a key driver for replenishing water resources. Recently, Mauro *et al.*, (2010) presented a new concept, the "CO_2-Water" connection in order to illustrate the impact of emitted carbon to atmosphere, and the water consumed to sequester that amount of carbon through forests. They considered the life-cycle of ethanol and gasoline. This study showed that the embedded water (m^3/TJ), including the CO_2-Water linkage, is greater in gasoline from tar sands than in ethanol from sugarcane (Chavez-Rodiguez and Silvia, 2010). This challenges the widely accepted perspective of embedded water in fossil fuels by linking the mitigation needed to sequester carbon in trees and then accounting for the additional water use that this afforestation would appear to create. However, in this calculation of the water needed to sequester carbon by forests, we consider that rainfall should also be included, like we have included here in WF_3 for hydropower generation. In linking CO_2 and water, it is also important to consider the location specificity, because, unlike carbon, water-related issues are local and driven by the rainfall and the local hydrology.

The WF_3 values were the smallest of the three methods considered for all of the hydropower plants in New Zealand (Table 3.1 and Table 3.2), since it includes the hydrologically important input of rainfall. The Manapouri hydropower plant in the South Island and all the hydropower plants in the North Island showed negative footprint values for WF_3. This is because these reservoirs receive more water as rain than they lose through evaporation. This also reflects the sensitivity of WF_3 to regional climatic differences, or local weather, in terms of rainfall and evaporation rates. The definition of the water footprint we present here in the WF_3 method moves beyond the consumptive water-use based description of water footprint (WF_1), and considers also the net water balance of the system including rainfall as an input, which is the key driver for replenishing water resources. This definition can be used to assess the WF of any other hydropower generation system, or product and service elsewhere in the world. It provides meaningful information to understand the differences in the hydrological impacts of the WF of different locations which are diverse in terms of water resource availability.

From a water footprint perspective, it is an advantage for hydroelectric dams to be located in wet regions and/or areas of low open-water evaporation rates, such as the Manapouri reservoir in the South Island and all the reservoirs in the North Island. These reservoirs harvest much more water through rainfall than they evaporate (Table 3.1 and Table 3.2). This net water surplus can provide a number of other riparian ecosystem services, or possibly even dis-services, downstream of the dam. We ignore, however, any positive or deleterious impacts that the dam might have on the local ecosystems, for we are focussing purely on changes in water quantity. Therefore, assessment of WF alone is inadequate to describe the sustainability of hydropower generation in a given location.

3.4.2 Impact of local climate and structural specifics on the water footprint

Climatic parameters like rainfall, temperature, relative humidity and wind velocity are highly variable across New Zealand. Therefore, the local climates above the reservoirs are different. With respect to this variability, evaporation and evapotranspiration rates were also different, thereby resulting in high variability in the water footprints for electricity generated by different hydroelectric power plants (Table 3.1 and Table 3.2). Figure 3.4 shows the differences in annual average rainfall, open-water evaporation and calculated evapotranspiration for various meteorological stations that we used to calculate the water footprints of different hydroelectric power plants. At the Rotorua and Manapouri weather stations there was more rainfall than either open water evaporation, or evapotranspiration. As a consequence, North Island hydropower stations, and the Manapouri hydropower station in the South Island, showed negative values for WF-3 (Table 3.1 and Table 3.2).

The contribution of local climatic conditions and the reservoir surface area to the variability of the water footprint estimated by different methods was quantified using regression analysis (Table 3.3).

Table 3.3 Summary of the regression analysis: Effect of local climate and reservoir surface area on the variability of the water footprint of New Zealand's hydropower

Water footprinting Method	Local climate [a] R^2	Reservoir surface area R^2
WF-1	0.228*	0.386**
WF-2	0.311*	0.158^{NS}
WF-3	0.474**	0.227*

* $P \leq 0.05$, ** $P \leq 0.01$, *NS* not significant

[a] The difference between annual average open water evaporation and rainfall was considered as the local climate

For this analysis, local climate was defined as the difference between open-water evaporation and rainfall. The results indicate that the order of the contribution of local climate to the variability of WF is WF_1(23%) < WF_2(31%) < WF_3 (47%) (Table 3.3). The WF_3 shows higher sensitivity to the local climatic conditions than the other two methods. For any method used to quantify a WF as an indicator of the impact on freshwater resources, the sensitivity to local climate should be a key criterion, as water scarcity is a highly local issue (Ridoutt et al., 2009). In this regard we consider WF_3 to be the hydrologically most appropriate metric.

We also considered the impact of the surface area of the reservoirs on the water footprints. Reservoir surface area accounted for 38.6% of the variability of WF_1, and this was higher than in the other two methods (Table 3.3). The surface area of the reservoir is directly related to the main source of water loss (evaporation) from the storage of the hydroelectric plant. But WF_2 is not significantly sensitive to this factor. The reservoir surface area was also significant for WF_3 as this relates to the effectiveness of the dam to collect rainfall.

The major part of the variability in WF_1 (62%) and WF_3 (70%) is explained by the reservoir surface area and local climate. We anticipate that the major part of the unexplained variability is due to differences in the efficiency of power generation, namely the amount of energy produced per unit volume of water by the different power plants.

3.4.3 The water footprint of a unit of New Zealand hydroelectricity

The weighted-average values of the water footprint for New Zealand's electricity using the three different methods ranged between 1.55 and 6.05 m^3/GJ (Table 3.4). Irrespective of the method, the WF of New Zealand's hydropower is low compared with the value of 22 m^3/GJ estimated by Gerbens-Leenes *et al.* (2009), and also the value of 68 m^3/ MWh (18.9 m^3/GJ) reported for United States of America (UNCCC, 2009). However, Fthenakis and Kim (2010) reported a value of 17000 L/MWh (4.72 m^3/GJ) for the United States of America using average water consumption in hydropower generation. This compares with our WF-1 value of 6.05 m^3/GJ for New Zealand.

Table 3.4 Average water footprint of hydroelectricity generated in New Zealand, calculated using three different methods

Method	Water Footprint (m^3/GJ)
Consumptive water use $WF_1 = E_0/P_w$	6.05
Net consumptive water use $WF_2 = (E_0 - T_r)/P_w$	2.72
Net water balance $WF_3 = (E_0 - P)/P_w$	1.55

Another interesting point to note is the very high variability in the WF across all the hydropower plants within New Zealand (Table 3.1 and Table 3.2). The combined coefficient of variation (CV) was 134% for WF-1, 183% for WF-2 and 332% for WF_3. This highlights the degree of variability that could be expected around the world. Therefore, it is considered imperative that local values be used in international water footprinting protocols.

3.4.4 Ecosystem service or dis-service?

We have simply explored water-footprint quantity metrics of hydropower generation. The water collected, stored, and discharged by hydroelectric dam reservoirs can provide both ecosystem services and dis-services (Zhang et al., 2007) in the riparian ecosystem. In the case of hydroelectric dams, environmental analysts have proposed quite different interpretations of both the magnitude and the extent of the environmental costs of hydroelectric facilities (Gleick, 1994). Some believe that hydropower is a benign source of electricity generation, while others have concluded that new large dams may be the worst electricity option in terms of damage to ecosystems per unit of electricity generated (OECD, 1988). Humbert and Manedly (2008) have shown that the damage to aquatic biodiversity caused by water use, especially from the dams used for hydropower, is not negligible. But this study did not consider any of the benefits, or draw backs, associated with hydroelectric dams, such as the increase, or decrease, in water quantity available to the aquatic ecosystem due to the reservoir. Hydropower facilities can have large environmental impacts by changing land use, relocated homes, altered natural habitats, and displacing some parts of the river ecosystem. However, they do create new standing water ecosystems. In addition, water storage reservoirs tend to attenuate flood peaks and change the annual distribution of flows by storing water at one time, and releasing it at another. This can perform either an ecosystem service, or dis-service, depending on what ecological communities are assessed. In this study, we have not considered any of those environmental and social aspects of the impacts of hydroelectric projects. Rather we have focussed on the single parameter of water quantity. Nonetheless, with water and energy supplies likely to reach crisis proportions in the coming decades (Beddington, 2009), some tough decision will need to be made.

3.4.5 Gaps in the methods

In this study, we have not considered the dilution water requirement for the polluted water (grey water) from hydropower generation. The grey water could be due to changed temperature, turbidity, or chemical status. However, we anticipate this to be very low, as there is minimal pollution from hydroelectricity. Furthermore, no fuel is burned. Hydropower is widely accepted as a clean and climate-friendly source of energy (Huang and Yan, 2009; Sims, 2004; USGS, 2010). This is also considered to be the most important advantage of hydropower, as compared to other power sources such as coal, petroleum, and natural gas.

We have not considered the water consumption, or environmental impact of the full life-cycle of hydroelectricity generation, namely the water use during the construction of dam, or power plant, and nor have we considered the end-of-life decommissioning of the dam. This study was limited to a quantity assessment of the operational water consumption, plus the inputs and their impact on the quantity of water resource. In a study of life-cycle water use in USA electricity generation, Fthenakis and Kim (2010) also considered the "upstream" (indirect) water withdrawal by a hydroelectric power plant to be zero. They also accounted only for on-site, "instream", or direct water use (Fthenakis and Kim, 2010).

3.4.6 Looking to the future

Hydropower represents 19% of total global electricity production (Sims, 2004; USGS, 2010). Approximately two-thirds of the economically feasible hydroelectric-power potential remain under-developed throughout the world. Untapped hydro-resources are still abundant in Latin America, Central Africa, India and China. In this study, we have explored the role of local climatic conditions and reservoir surface area on the net water balance of functioning of a dam, and we have considered its claims on changing freshwater resource availability. This could be used in understanding the hydrological advantage of locating hydroelectric dams in wetter regions, should an increase in downstream water availability be sought.

3.5 Conclusions

Three methods (WF_1, WF_2 and WF_3) have been used to quantify the water footprint of hydropower considering New Zealand as a case study. The first method (WF_1) simply considers the consumptive water use of the hydroelectricity generation system, while the second method (WF_2) accounts for the net water consumption by including the effect of the land-use changes created by a dam. In the third method (WF_3), the metric is linked to the net water balance at the local scale of the hydroelectricity generation system. The WF_1 and WF_2 values which are based only on the consumptive and net consumptive water use were higher than the values from the hydrologically rational WF-3 method based on the net water balance approach.

In addition to direct consumptive water use (WF_1), the WF_2 considers the differences due to land-use changes from dam construction to assess a net consumptive water use. However, this approach has a methodological challenge regarding the time scale over which the antecedent comparison is made. The WF_1 and WF_2 metrics only strictly consider consumptive use, and not the supply side of the hydrology. Therefore, they are inadequate to understand the impact of hydroelectric power generation on freshwater resource availability.

We have revealed the importance of considering both the water inputs and outputs from reservoirs in the WF_3 method which considers the net water balance. These have not been considered in previous WF assessments. The definition of water footprint we presented here in WF_3 method moves beyond the consumptive water-use based description of water footprint (WF_1), and considers the net water balance of the system including rainfall as input, which is the key driver for replenishing water resources. This theoretical framework can be used to assess the WF of any other hydropower generation system, or production and service system elsewhere in the world. It provides meaningful information to understand the differences in the impacts of the WF in different locations, which are diverse in terms of water resource availability.

The negative water footprint values which resulted from WF_3 indicate that reservoirs in high rainfall areas can collect more water than they lose through evaporation. This net water surplus could be utilized for many other riparian purposes downstream, other ecosystem dis-services notwithstanding.

This study also shows the effect of local climatic differences and the surface area of the reservoirs on the water footprint calculated using different methods. The surface area of the reservoir is directly related to the main source of water loss (evaporation) from the reservoir of the hydroelectric plant. Both WF_1 and WF_3 are significantly sensitive to this factor while WF_2 is not. The WF_3 values showed a high sensitivity to local climate, which is an important criterion, because water-related issues are highly local. This study also suggests that hydropower plants with reservoirs located in wet regions having a smaller reservoir surface area and efficient energy generation techniques are, from a hydrological water-quantity perspective, advantageous.

The weighted-average WF of New Zealand's hydroelectricity using our approach of the net water balance (WF_3) is smaller than other reported values internationally. Large variation in the water footprints of hydropower from different locations within New Zealand shows, however, the degree of variability that could be expected around the world. Therefore, it is considered imperative that local values be used in international water footprinting protocols.

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CHAPTER 4

Water footprinting of agricultural products: A hydrological assessment for the water footprint of New Zealand's wines

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In this Chapter is quantified the of water footprint of wine produced in New Zealand using hydrological water balance method of water footprinting. The functional unit for the water footprint is a 750-ml bottle of wine. The system boundary extends from the vineyard to the winery gate and the life cycle assessment considers the footprints from both the foreground and background systems. The analysis was performed for two hydrologically different regions: the Marlborough region in which vineyards rely on irrigation and the Gisborne region where the grapes are grown under rain-fed conditions. The hydrological water balance method that was used in Chapter 3 for the water footprint of hydroelectricity is now used here to quantify the water footprint of wine. The results of this method will in the next Chapter (Chapter 5) be compared with the outcome for the water footprint of a bottle of wine using three other commonly used water footprinting methods. Water footprinting protocols are being developed, as discussed in Chapter 2, so it is timely and pertinent to establish the value of the various footprinting methodologies to the stakeholders of viticulturalists, winemakers, consumers, and regulatory agencies. This will be discussed in Chapter 5 for wine, and in Chapter 7 for ware potatoes.

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4.1 Abstract

Agriculture plays a key role in relation to global water stresses. Increasingly, water footprints (WF) are being used to indicate the impacts of the water use by production systems. International standards for WF are being developed and this paper contributes to these from a hydrological perspective. The impacts of water use through the life cycle of grape-wine production on water resources were assessed for two regions in New Zealand: Marlborough and Gisborne. The functional unit (FU) was a 750-mL bottle of wine at the winery gate. The WF was assessed using a full water-balance calculated by subtracting inflows from outflows. The net usage from groundwater and soil moisture storage were quantified as blue and green water footprints respectively. We found a large variability of blue-WF even within a region. For the grape-growing stage, the average blue-WF was negative, at -81L/FU for Marlborough and -415L/FU for Gisborne indicating the water resources are being recharged on an annual timescale. The green-WF was negligible. The grey-WF, water required to dilute NO_3-N leached in the vineyard phase, was 40 for Marlborough and 188L/FU for Gisborne. However, the average concentration of NO_3-N in the leachate was well within the drinking water standard of 11.3mg/L (5.01mg/L and 8.7mg/L for Marlborough and Gisborne). The impacts of the winery phase were very small compared with that of the vineyard. The variability we have found indicates the importance of considering water issues at the local scale. Locale is the essence of terroir for wine.

Key words: hydrological impacts, water footprints, wine production, groundwater, water quality, water quantity

4.2 Introduction

One of the key challenges facing the world is meeting the rapidly increasing demands for food, water, and material goods of an increasing global population, while simultaneously protecting the ecosystem services provided by natural water ecosystems (Postel, 2000). Freshwater is the most essential of natural resources, yet freshwater systems are directly threatened by human activities and will be further impacted by anthropogenic climate change. Agriculture is, by far, the largest freshwater consumer, accounting for more than 70% of world's water withdrawals (UNEP, 2007). Because of agriculture's huge water usage and its significant contribution as a source of chemical emissions into freshwater, it is considered central to future attempts to address global stress on water quantity and quality.

The wine industry is a global sector which places for a significant demand on the world's water resources (Cichelli et al., 2010). New Zealand produces many premium quality wines (Imre and Mauk, 2009) and its wine industry has expanded significantly over the past decade. The grape-producing area has tripled from just 10,000ha in 2000, to over 33,400ha in 2010 (Annual Report, 2012), reflecting the industry's reputation as a global provider of super-premium, cool-climate wines. As consumers of super-premium wines become increasingly environmentally conscious, demonstration of environmental credentials and evidence of continuous environmental improvement will probably become vital from the viewpoint of producers in order to secure eco-premium prices. In a study of consumer attitudes regarding environmentally sustainable wine, Forbes et al. (2009) have mentioned that over 70% of respondents had indicated that they would prefer to buy and even prepare to pay more for an environmentally sustainable wine. Suppliers will increasingly be obliged to provide quantitative information on their resource use and its impact on the environment so that the retailers can provide environmental assurances to their customers. Environmental values are the most important drivers in New Zealand wine industry (Gabzdylova et al., 2009). Furthermore, in the long run, growers or producers will need to evaluate and improve their practices on resource use to sustain their production and to sustain the natural capital stocks of their vineyards. The environmental and economic sustainability of wine making are inextricably linked.

The water footprint (WF) has been proposed as a metric that indicates the water use and impacts of the production system on water resources. It considered to have the potential to underpin an environmental product declaration and act as a communication of environmental performance to stakeholders (Ridoutt et al., 2009). However, it depends on the methodological scope being considered. International standards for water footprinting are still being developed. It is an area of burgeoning scientific developments. There are a number of methodologies that are being proposed for water footprinting. Some of these methods consider hydrological inflows, outflows and storage changes (Deurer et al., 2011; Herath et al., 2011). Some methods indicate the impact on water resources by referencing to a characterization factor (Canals et al., 2009; Ridoutt and Pfister, 2010) while other methods are based only on consumptive water use (Hoekstra et al., 2011). Irrespective of the differences in quantification, the outcome of the WF process should be meaningful to the stakeholders, including growers, resource regulators, retailers, and consumers.

The objective of our study is to assess the impacts of the wine-grape supply chain on both the quantity and quality of local water resources. For this, we use a hydrological water-balance method of water footprinting (Deurer et al., 2011; Herath et al., 2011). We compare the WFs of two wine-producing regions in New Zealand. The usefulness of the WF results to the key stakeholders has been evaluated by applying three other WF methods and this will be presented in a future paper.

4.3 Methodology

A life-cycle based approach was used to assess the freshwater use and its impacts along the grape-wine supply chain. Life cycle assessment (LCA) principles were used in inventory modeling. However, our interpretation of the impact of freshwater use on water resources is quite different from that used in other LCA-based WF methods (Canals et al., 2009; Pfister et al., 2009). We have used a hydrological water-balance method following the theoretical framework presented in Deurer et al. (2011) and Herath et al. (2011) to assess the hydrological impact of freshwater use on local water resources. The system boundary was established from raw-material acquisition through to the winery gate ready for dispatch (Figure 4.1). The functional unit (FU) was defined as a 750-mL bottle of wine at the winery gate in a 'ready to distribute' condition.

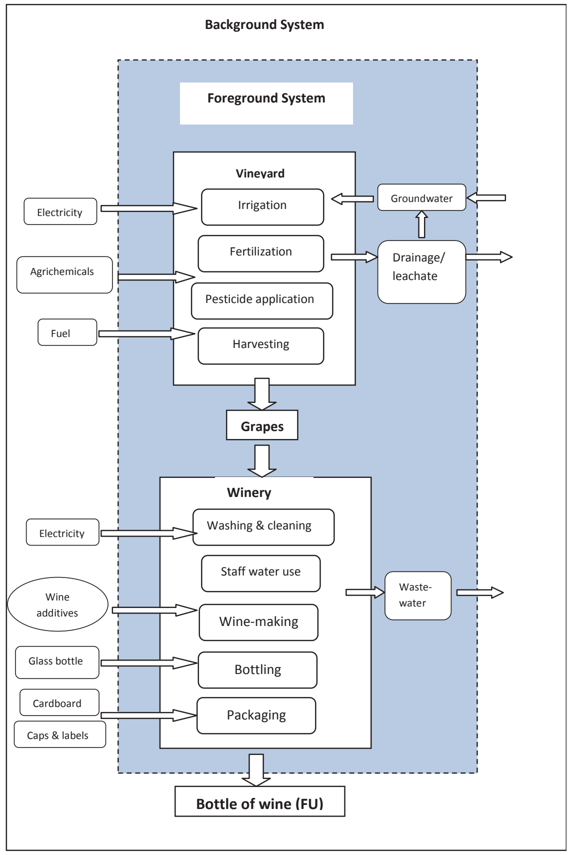

Figure 4.1 Flowchart for the wine production system studied. Shaded area marked with dashed line indicates the foreground system. Square boxes with rounded corners represent activities that are included and round shapes show the activities excluded.

4.3.1 Data and assumptions

Data for the foreground system (Figure 4.1) were sourced directly from a survey of grape growers and wine producers. In our data collection, we made every possible effort to capture the variability and assure the completeness of information on direct water use, and other inputs of embedded water. Data were sourced from 36 wineries that process grapes from the two regions. For this, we worked with 'Sustainable Wine Growing New Zealand' (SWNZ), which is a sustainability initiative of 'New Zealand Winegrowers', the industry body of the New Zealand wine growers. At present, the SWNZ program covers 94% of New Zealand's vineyard area and 90% of its wineries (NZWA, 2011). Each season, SWNZ members complete a scorecard which is used to record their vineyard and winery activities, and this includes water use and other inputs, along with waste management. Data from the background system were also collected from the producers wherever possible. For example, the direct water use, electricity use and waste water disposal associate with the manufacture of the wine bottles were ascertained from the major bottle manufacturing plant which is located in Australia.

Therefore, data used in this analysis are of high quality in terms of traceability, reliability, completeness, and their temporal and geographic representativeness. We only used LCA databases for inventory data for a very limited number of inputs from the background system (e.g. agrichemicals) when direct information of water use was not available. This is mainly because at present, life cycle inventory databases only contain limited information about water use (Berger and Finkbeiner, 2010; Jeswani and Azapagic, 2011; Pfister et al., 2009) and this data are insensitive to geographic location.

We considered a three-year average (2008-2010) of grape-production data. This enables us to smooth out any deviations, or variations, in the normal operations of a facility. This also captures the fluctuations in the level of local production. The construction and the maintenance of built capital, that is machinery and buildings, plus emissions from wine additives, were excluded from this study because it is assumed that their contribution would be insignificant to the total WF on an annual basis. Initial basic operations in the establishment of vineyard like land preparation, planting, establishing vine support structures that normally happen once in about 50 years of grape growing were not taken into account. In addition, the bottling process consumes subsidiary products such as glue, and ink, but these were not included partly because of the absence of reliable information, but primarily because of the small amounts used.

4.3.2 Area considered, regional dynamics, climate and soil variability

Two major and contrasting grape growing regions in New Zealand were considered (Figure 4.2): Marlborough and Gisborne. Marlborough is the largest grape-wine region accounting for 57% of the vineyard area in New Zealand while Gisborne is the third largest and its vineyards cover 6% at present (Annual Report, 2012; Hayward and Lewis, 2008).

Figure 4.2 A map of New Zealand showing the two regions considered in the study.

The two regions are different in terms of climate, water resource availability and soil types. The climate is highly variable across Marlborough, and the soil types are quite variable within both the regions. The average rainfall in the Marlborough region was 620mm/y and that of Gisborne was 1029mm/y over the 38-year period from 1972 to 2010 (NIWA, 2012). In Marlborough, all the vineyards were irrigated with water sourced from groundwater resources, while Gisborne vineyards were not irrigated. In this assessment, the vineyards were on 29 different soil types spread across 19 climatic regions. There were approximately 12,600ha under grapes across both the regions.

4.3.3 Water consumption and use along the wine supply chain

In this study, as in other previous water footprinting assessments, different water colors (blue, green and grey) were distinguished depending on the source of water incorporated into the product life cycle. The blue water refers to groundwaters and surface waters; the green water is the rainwater stored in the soil profile as soil moisture; and the grey water refers to water pollution, and is defined as the volume of freshwater required to assimilate the load of pollutants based on the accepted water quality standard (Hoekstra et al., 2011). We also tracked and recorded the geographical location of where this water was sourced from, and in case of grey water where the emissions occurred at the different stages of the product life cycle. In water footprinting, water consumption through evaporation and transpiration and direct water use for processing and indirect water use through background inputs are often reported separately from the impact assessment. In our study also, we first present the water consumption and then assess the impacts. This was done through modeling hydrological inflow and outflow components as described in the section below.

4.3.4 Water footprint assessment using water-balance approach: The water footprint as the impact of production on water resources

A detailed hydrological water-balance analysis was performed to assess the impacts of grape production, following the principles used by Deurer et al. (2011) as explained below. We illustrate the assessment of impacts of grape-growing stage by calculating the water-balance for 1kg of grapes, which is the input amount for a bottle of wine processed in the winery.

4.3.4.1 Impact on water quantity: The blue and green water footprints

Water use by agricultural production systems is sourced from two water resources.

1. The blue-water resource: The surface waters and/or groundwater used in irrigation and/or other direct applications
2. The green-water resource: The use of water stored in soil profile as soil moisture.

The impact on water quantity was assessed as the net water usage by wine production from those two resources using a full water-balance considering all hydrological inflows and outflows. The net water-balance was quantified by subtracting inflows from the outflows to account for the net usage, and this is expressed as blue and green water footprints. Groundwater is the main blue-water resource immediately connected to viticulture in the study regions. The blue-WF was calculated as the net usage of groundwater being the difference between groundwater extraction for irrigation (outflow) and the drainage plus runoff (inflow) as follows (Eq. 4.1 and Figure 4.3):

$$WF_{Blue} = \frac{10\left\{ I - \left(D + R\right)\right\}}{Y} \qquad \textbf{(Eq. 4.1)}$$

Here, WF_{Blue} is the net use of blue water [L/kg of grapes] from groundwater, *I* is the amount of water [mm/year] extracted from the aquifer for irrigation, *D* is the drainage [mm/year] from the root zone under irrigated grapes, and R is the surface runoff [mm/year] from the irrigated vineyards that ends up either in groundwater or surface water. As the viticultural landscape is flat to undulating, we have assumed that any surface runoff eventually recharge groundwater resources. Here, *Y* is the grape yield under irrigated conditions [ton/ha/year]. The factor 10 is for conversion of units.

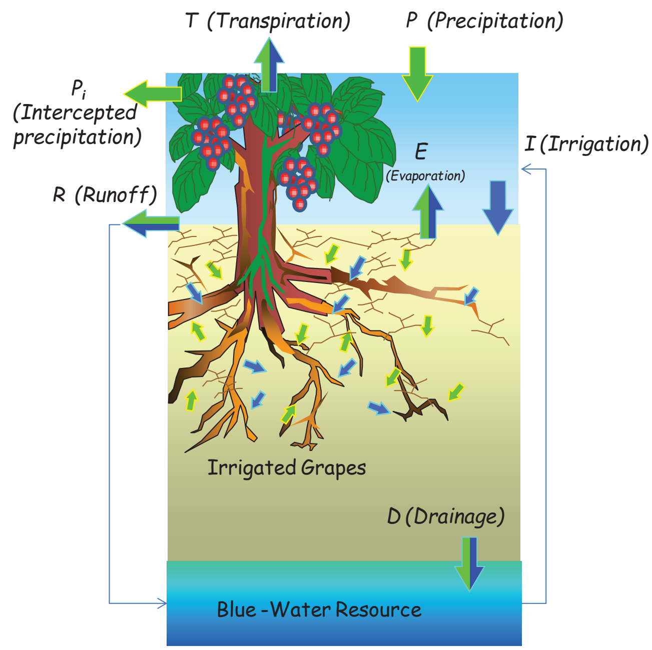

Figure 4.3 Components of the full hydrological assessment of the water balance of irrigated wine grapes where precipitation (*P*) is considered, as is the hydrological connectedness of surface runoff (*R*) and drainage (*D*) with the blue-water (groundwater) resource.

We followed the same hydrological principles as Deurer et al. (2011) to assess the net water-balance for the soil-moisture store by subtracting effective rainfall, the inflow from the summation of the outflow components, namely, transpiration from the vegetation, evaporation from soil, drainage and runoff under rain-fed conditions to assess the green-WF (Eq. 4.2 and Figure 4.4).

$$WF_{Green} = \frac{10\left\{\left(T_r + E_r + D_r + R_r\right) - \left(P - P_i\right)\right\}}{Y_r}, \qquad \textbf{(Eq. 4.2)}$$

where, WF_{Green} is the net green-water consumption [L/kg of grapes] from soil-moisture store under rain-fed conditions, T_r is transpiration from the vegetation [mm/year] of the vineyard under rain-fed conditions (both from the grape-vines and the grass cover), E_r is the evaporation from soil [mm/year] under rain-fed conditions, D_r is the drainage [mm/year] from the root zone in rain-fed conditions, and R_r is the surface runoff [mm/year] under rain-fed conditions, P is the precipitation [mm/year], and P_i is precipitation intercepted [mm/year] by the canopy. Here, Y_r is the grape yield under rain-fed conditions [ton/ha/year]. The factor 10 is for conversion of units.

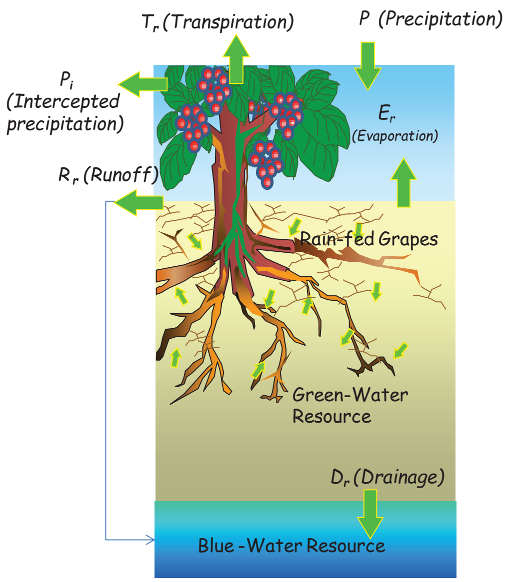

Figure 4.4 Components of the full hydrological assessment of the water balance of vineyards under rain-fed conditions. The effective precipitation (P- Pi) is the inflow for soil moisture store, and Er (evaporation), Tr (transpiration), Dr (drainage) and Rr (runoff) are the outflow components.

4.3.4.2 Impact on water quality: The grey-water footprint

The impact of wine production on the quality of local water resources was calculated using the grey-WF as defined by Hoekstra et al. (2011). It is calculated as the volume of freshwater that is required to dilute a pollutant so that its concentration meets the prevailing water quality standards. The following equations, Eq. 3a and Eq. 3b were used to assess the grey-WF for the vineyard.

$$WF_{Grey} = \left[L/\left(C_m - C_n\right)\right]/Y \qquad \textbf{(Eq. 4.3a)}$$

$$L = 10^4\left(D \times C_d + R \times C_r\right) \qquad \textbf{(Eq. 4.3b)}$$

Here, WF_{Grey} is the freshwater required [L/kg of grapes] to dilute the runoff and leachate down to an accepted water quality standard, L is the net-load of pollutants from the system [mg/ha], D is the drainage [mm/year] from the root zone under irrigated grapes, and R is the surface runoff [mm/year]. Here, C_d is the concentration [mg/L] of the pollutant in drainage, C_r is the concentration [mg/L] of the pollutant in surface runoff, C_n is the natural concentration [mg/L] of the pollutant in the receiving water body, and C_m is the maximum acceptable concentration [mg/L] for the pollutant given by the local and appropriate water quality standard. Here, Y is the grape yield [kg/ha/year]. The factor 10^4 is for conversion of units.

Eq. 4.3c was used to calculate the grey WF of winery waste water and effluent from the background system.

$$WF_{Grey} = \frac{E\left(C_e - C_b\right)}{\left(C_m - C_n\right)} \qquad \textbf{(Eq. 4.3c)}$$

Here, E is the effluent volume [L] per FU, C_e is the concentration of the pollutant considered in the effluent, and C_b is the background concentration of the pollutant.

4.3.5 Modeling water and solute dynamics of vineyards

The soil-water dynamics and solute transport in the vineyards were assessed using the Soil Plant Atmosphere System Model (SPASMO) (Green and Clothier, 1999; Green and Clothier, 1995; Green et al., 1999). This is a mechanistic model which considers water, solute (e.g. nitrogen and phosphorus), and pesticide transport through a one-dimensional soil profile to the base of the root zone. The SPASMO model includes components that predict the carbon and nitrogen budgets of the soil, which enable calculation of plant nutrient uptake, plus the various exchange and transformation processes that occur in the soil and aerial environment, along with the recycling of nutrients and organic material to the soil biomass, and the addition of surface-applied fertilizer and/or effluent to the land (Green et al., 2008). This model has been validated for a range of New Zealand soils under various land uses across a wide range of climatic conditions and management practices (Green and Clothier, 1999; Green and Clothier, 1995; Green et al., 1999; Green et al., 2008).

We used weather data (NIWA, 2012) for a 38-year period (1972-2010) to simulate soil-water dynamics and grape production in the regions using SPASMO, so as to explore the range in annual values for the components of the water-balance. The model used local climate data to simulate the water-balance for 29 different soil types within the two regions. Climate data were sourced from the Virtual Climate Network Stations (VCNS) of the National Institute of Water and Atmospheric Research (NIWA), New Zealand (NIWA, 2012). The soil physical and hydraulic properties were deduced using data from the New Zealand National Soils Database (NSD, 2012). The net-water consumption, recharge and runoff for each soil type/climate combination were then weighted according to their relative area to calculate a regional value.

In the background system (Figure 4.1), for both the water quantity and quality impact assessments, we have used data from the literature when information is not available to enable full hydrological water-balance analyses. In brief, for the water used in paper and cardboard production, manufacture of pesticides, extraction and refining of petroleum fuel we used reported values. The grey-WF of the background system was only partially assessed because of the unavailability of reliable information and also because their impacts are considered to be insignificant.

4.4 Results and Discussion

Firstly, we present the results of water consumption and use along the wine-grape supply chain by distinguishing the blue and green-water components. This is essentially the water evaporated (*E*) and, transpired (*T*) which is commonly referred as evapotranspiration; *ET* or that water which is incorporated during the production. Secondly, we discuss the impacts of the wine-grape supply chain on the quantity and quality of water resources. In the viticultural stage, the water use and its impacts are reported for 1kg of grapes which is the input amount for making a bottle of wine in the winery.

4.4.1 Water consumption in wine production supply chain

The largest water consumption and greatest impacts occur during grape cultivation, and we found large variation within the regions. These variations are mostly induced by the heterogeneous nature of soils across the landscape, plus the differences in local climate which is dominated by local rainfall. The variability in water consumption for Marlborough grapes is illustrated in Figure 4.5.

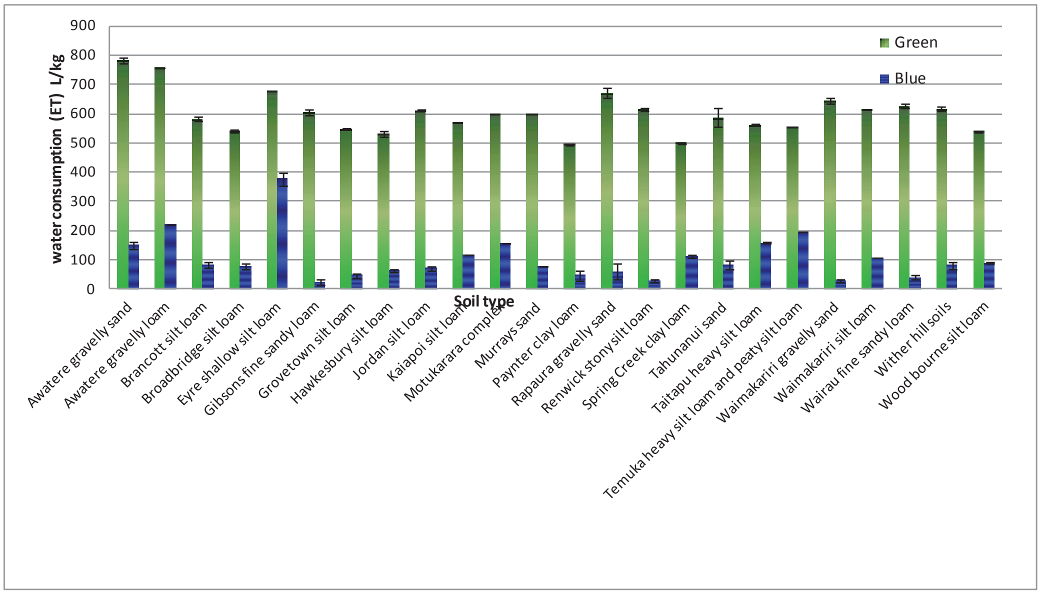

Figure 4.5 Green and blue water consumption through evapotranspiration (*ET*) from vineyards (normalized to grape yield) categorized by different soil types in the Marlborough region. The bars indicate the variability that is mainly due to local climatic differences.

The green-water consumption is essentially the evapotranspiration that is E_r plus T_r of Eq 2 and Figure 4. The blue-water consumption is the additional evapotranspiration due to irrigation; in other words, the difference between *ET* under irrigated and rain-fed conditions. The variability in blue-water consumption in the Marlborough region (Figure 4.5) is mainly due to the irrigation water use differences, and reflects the variation in local rainfall and soil hydraulic properties. In Gisborne, no irrigation is used generally and therefore, the evapotranspiration losses are sourced from green water (Figure 4.6). The weighted average green-water consumption through evapotranspiration in the vineyards for the Gisborne region was 601L/kg, and it was 611L/kg for the Marlborough region (Table 4.1). The average blue-water consumption in the vineyards for the Marlborough region was 71L/kg, while that of the Gisborne was zero, as no irrigation is used. These numbers represent the blue and green volumetric water footprints (Hoekstra et al., 2011) of grape production for the two regions.

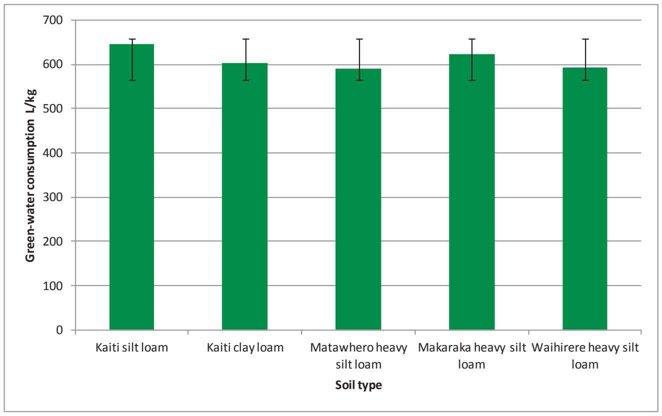

Figure 4.6 Green-water consumption through evapotranspiration (ET) from vineyards (normalized to grape yield) categorized by different soil types in Gisborne region. The bars reflect the variation found by modeling over the 38-year period of simulation (from 1972 to 2010).

Water use in the winery stage was very similar for both the regions due to similarity in the operations and inputs in the winery stage. The direct use of water in the winery includes washing, cleaning of the wine barrels and the floor and staff water use. The background system water use in the winery operations is mainly from packaging materials, such as cardboard production and bottle manufacturing. This will be discussed later on in the section. The background system water use in the vineyard is primarily from the energy and the fuel for the operations, and from agrichemical manufacturing. However, altogether this was in total less than 2L/FU in both regions. Total water use, both blue and green water, along the wine supply-chain was at 742.5L/FU for Marlborough, and 667L/FU for Gisborne (Table 4.1). Winery water use is only 8% and 9% of the total (blue plus green) water use in the Marlborough and Gisborne regions (Table 4.1).

Table 4.1 Water consumption (green and blue), plus direct and indirect use (blue) in the wine supply chain for the two regions of Gisborne and Marlborough (Units: L/FU, where the FU is a 750-mL bottle of wine at the winery gate).

Type of water used	Marlborough				Gisborne			
	Vineyard		Winery		Vineyard		Winery	
	Foreground system	Background system	Foreground system	Background system	Foreground system	Background system	Foreground system	Background system
Green	611	nil	nil	41.9	601.2	nil	nil	41.9
Blue	70.6	1.8	2.7	14.5	3.6	1.5	4.3	14.5
Blue +Green	681.6	1.8	2.7	56.4	604.7	1.7	4.3	56.4
Total (regional)	683.4		59.1		606.4		60.7	

For this information on volumetric water consumption to be meaningful, we carried out a detailed analysis of its impact on local water resources using a hydrological water-balance as indicated in the methodology section above (see Section 2.4). Results from this methodology are presented here, and a comparative assessment using different WF methods will be discussed in a subsequent paper.

4.4.2 The water footprints as indicators of impacts of water use on water resources

We quantified the impacts on local water quantity by calculating a hydrological water-balance for groundwater store (Figure 4.3 and Eq. 4.1) and the soil moisture store (Figure 4.4 and Eq. 4.2). The impact on local water quality was assessed by means of the grey-WF, as defined in Eq. 4.3a, 4.3b and 4.3c.

4.4.2.1 Viticultural phase - The foreground system

Impacts of water use on blue-water resources: The blue-water footprint

The groundwater was considered as the primary blue-water source in the study regions. The water-balance components quantified to estimate the net usage from groundwater (blue-WF) by the viticultural system (Figure 4.3 and Eq. 4.1). Not surprisingly, given the variability in local climate and soil hydraulic properties, the blue-WF varied from -250 L/kg to 248 L/kg for the different soil types in the Marlborough region (Figure 4.7). Even though all vineyards considered here were irrigated in Marlborough, under some of them the groundwater is still recharged by rainfall on an annual basis (Figure 4.7). When the blue-WF is ranked according to the local rainfall rates (Figure 4.8), the dominant role of rainfall becomes clear. The variability in the blue-WF corresponds to the differences in the rainfall across the range of 600 mm to 1200 mm/y in Marlborough. In the areas with annual rainfall <670 mm/y net groundwater usage is positive, and contributes to the depletion of groundwater resources. The weighted average blue-WF based on area of soil types and local climate was negative at 85L/kg grapes, which indicates that groundwater is replenished on average, on an annual basis across the Marlborough region. However, from a resource-management perspective, these vineyards of different soil types and local climate have to be considered separately. There are 16 different aquifers below the vineyards considered in the Marlborough region, and they are different in terms of their recharge rates, ages and depths (Davidson and Wilson, 2011).

Therefore, water resource availability for grape cultivation, and the rate at which this resource can be sustainably used, is highly variable within the region.

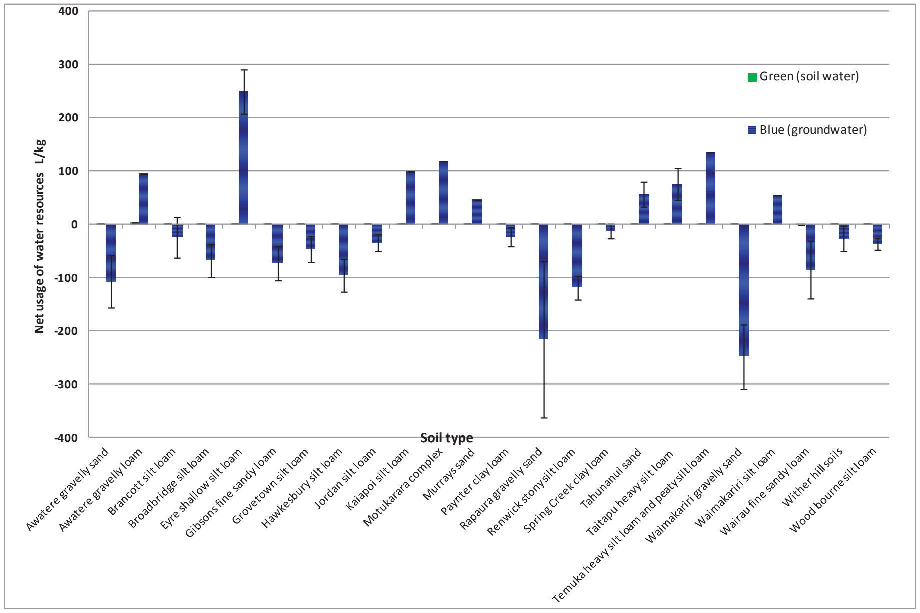

Figure 4.7 Net usage of groundwater resource (blue-WF) and soil-water store (green-WF) (normalized to grape yield) below the vineyards across the different climatic locations in the Marlborough region, as ranked by soil types. The green-WF is barely perceptible in the figure since it is near-zero as described in the text. The bars reflect the variability, mainly induced by the local climatic differences.

In the Gisborne region, irrigation is generally not used; therefore, the direct blue-water consumption was minimal, being limited to the water used for herbicide and pesticide applications. We accounted for this in our analyses (Table 4.1). In Gisborne, because of the ample amount of rainfall, the blue-WF, net use of groundwater was negative, varying from -369 to -427 L/kg grapes. The weighted average blue-WF was -418 L/kg, indicating that the groundwater resource is well recharged under viticultural soils in the Gisborne region.

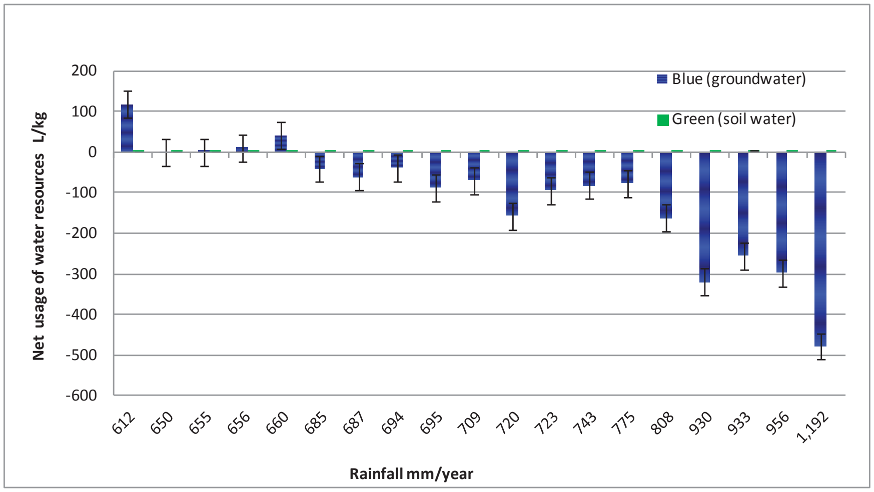

Figure 4.8 Net usage of groundwater resource (blue-WF) and soil-water store (green-WF) (normalized to grape yield) below the vineyards in different climatic locations in the Marlborough region, as ranked by annual rainfall. The green-WF is barely perceptible in the figure since it is near-zero as described in the text. The bars reflect the variability, mainly due to the differences in soil types.

Impacts of water use on green-water resource: The green-water footprint

To understand the impact of green-water consumption, the net change in soil-water storage (green-WF) was quantified (Figure 4.3, 4.4 and Eq. 4.2). In our analysis, as expected, we found that the green-WF was negligible (Figure 4.7 and 4.8) on an annual basis, as did Deurer et al. (2011) for kiwifruit cultivation across different regions in New Zealand. Therefore, the consumption of green water is considered to have an insignificant impact on the soil-water storage. This corresponds to the LCA-based water footprinting studies where the impact of green-water consumption is also considered negligible (Canals et al., 2009; Ridoutt et al., 2009; Ridoutt and Pfister, 2010), although the LCA-based studies are not based on the same hydrological water-balance principles presented here.

Impacts on water quality: The grey-water footprint

The impact of grape cultivation on water quality was assessed by considering the leaching associated with the main agrichemicals used in the vineyards. As most of the grape-growing areas in both regions are on flat to undulating terrain, potential contamination of surface waters by runoff of agrichemicals was considered to be minimal. Therefore, we considered only the pollution of the groundwater underlying the vineyards in the assessment of impacts on water quality. Nitrate-nitrogen (NO_3-N) was considered the dominant pollutant leaving the root zone of vineyards. We also modeled the transport and fate of glyphosate, which is the most widely used herbicide. When glyphosate was applied at 2.4 kg a.i./ha, the concentration at the bottom of the root zone was almost zero, being on average 1.4 x 10^{-4} µg/L, as it is rapidly degraded in the soil profile. Most of other modern pesticides will probably return similar results (Deurer et al., 2011).

The grey-WF was therefore calculated for NO_3-N according to Eq. 4.3a and 4.3b (Hoekstra et al., 2011). In a study with New Zealand kiwifruit, Deurer et al. (2011) discussed how the outcomes of this method are, affected by the different water-quality standards and the natural concentrations used. In our assessment, we have used New Zealand's drinking water standard of 11.3 mg NO_3-N/L (MoH, 2008) for the C_m. For the background concentration C_b, we used 0.31 mg NO_3-N/L for the Marlborough region, which was the average NO_3-N concentration measured in the regional groundwater over the past five-year period (Davidson and Wilson, 2011). For the Gisborne we used 1.7 mg/L, which is the national median concentration of NO_3-N for New Zealand groundwaters during 1995-2008 (MfE, 2009). The natural concentration C_n is the NO_3-N concentration in the water body if there has not been any human intervention (Hoekstra et al., 2011). We assumed this to be zero for New Zealand's groundwaters.

There is some input load of nitrate ($I*C_b$) from the use of irrigation water. We calculated this based on the amount of irrigation water use and concentration of NO_3-N in the groundwater used in irrigation (C_b). We found this to be very small, with a weighted average load of just 0.2 kg/ha. This is negligible. Therefore, this was not accounted for in our modeling and for the grey-WF calculations. The outflow of NO_3^- with runoff was also considered, as it ends up in the groundwater. The concentration of NO_3^- in the surface runoff from vineyards has been found to be minimal (Barlow et al., 2009) and runoff mainly infiltrates to groundwater through the preferential flow (Clothier et al., 2008). Therefore, we have assumed that the nitrate loading through runoff is negligible. Hence, the grey-WF is calculated by considering drainage as the main carrier of nitrate to the groundwater.

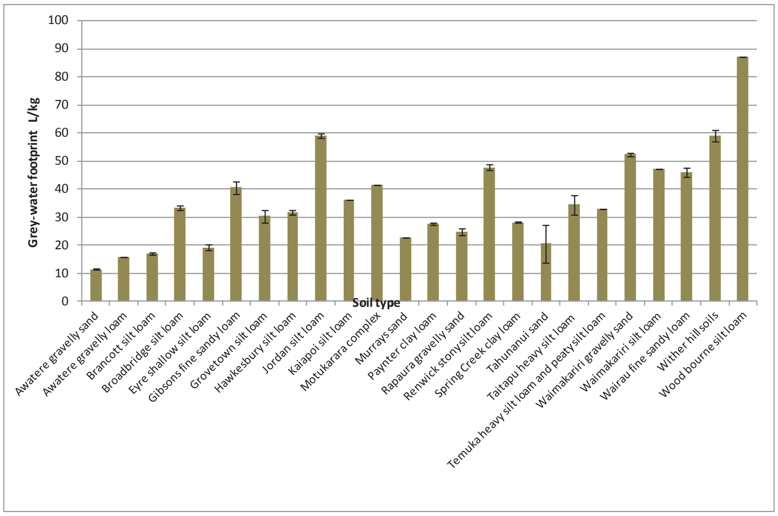

Figure 4.9 Grey-water footprint as calculated by considering the freshwater required to dilute the leached nitrate down to the New Zealand drinking water standard of 11.3mgNO_3-N/L under the different soil types and local weather zones of the Marlborough region. The bars reflect the variability, mainly induced by the local climatic differences.

The grey-WF showed a large variation between the different soil types and weather zones of both the regions (Figures 4.9 and 4.10). The weighted average grey-WF for the Gisborne region was 188 L/kg, which was higher than that of the Marlborough region, at 41 L/kg. We investigated this further and found that there was no significant difference in relation to nitrogen fertilizer application or other amendments, but rather the Gisborne soils have a higher N-mineralization rate as a result of the higher soil organic matter contents (Pullar, 1962; Rae and Tozer, 1990). High rainfall and consequently greater drainage also contributed to higher leaching of NO_3-N in the Gisborne region.

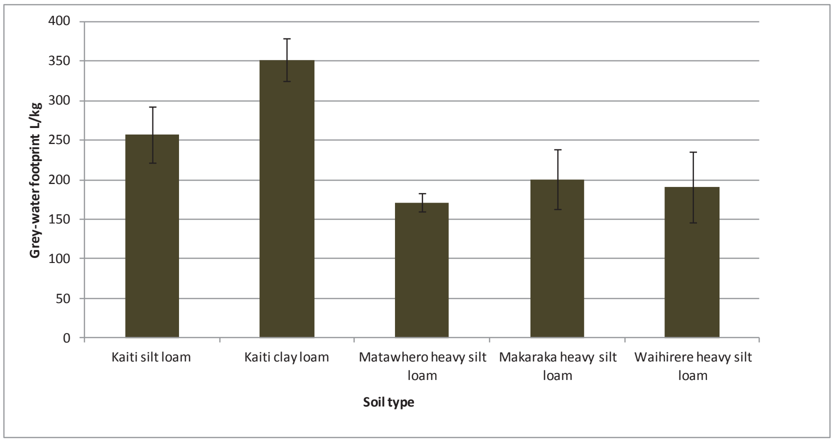

Figure 4.10. Grey-water footprint as calculated by considering the freshwater required to dilute the leached nitrate down to the New Zealand drinking water standard of 11.3 mgNO3-N/L under the different soil types and local weather zones of the Gisborne region. The bars reflect the variability due to local climate.

This method of quantifying grey-WF as the impact on water quality could be used to make comparisons between products and regions. However, from a resource management perspective, it is doubtful if this grey-WF provides sufficient information for regulatory or policy decisions. The loading rates and average concentration of NO_3-N in the drainage from vineyards are likely to be more useful for these purposes. Based on the modeling results, we predicted weighted average loading rates as 4.9 and 29.3 kg of NO_3-N/ha/y for the Marlborough and Gisborne regions, respectively. However, the pollutant loading needs to be considered together with that of other land uses for it to be meaningful and useful for policy implications.

The weighted average concentration of NO_3-N in the drainage was 5.01 mg/L in Marlborough, but with a high variability ranging from 0.51 to 17.9 mg/L for different soil types. This for the Gisborne region was 8.7mg/L, with a range from 6.9 to 17 mg/L. On a regional average basis, the NO_3-N concentration in the drainage was below the critical level for the drinking water standard of 11.3 mg/L (MoH, 2008). The variability found here indicates the importance of considering the water quality issues at the local scale. The loading rates and concentrations both need to be considered, together with geohydrological characteristics of the local aquifers, for their volumetric flow rates might well provide sufficient dilution of these loadings.

4.4.2.2 Viticultural phase: The background system

Electricity is a common input in both the vineyard for pumping irrigation water, and in the winery for refrigeration and other operations. The WF of New Zealand electricity was calculated using the WF of hydropower by Herath et al. (2011) and the consumptive water use of the other energy carriers (Scown et al., 2011) in the New Zealand electricity mix. This analysis realized a weighted average of 13.57 L/kWh based on consumptive use, and just 4.34 L/kWh for the hydrological water-balance approach (Table 4.2).

Table 4.2 The water footprint values for energy carriers used in a vineyard and winery in New Zealand. The values for the consumptive water use and the hydrological water balance approach are given separately.

Energy carrier	**Water footprint**	
	Consumptive water use approach	Hydrological water balance method
New Zealand electricity mix	13.57 L/kWh [a]	4.34 L/kWh [a]
Diesel (by weight)	5.04 L/kg [b]	5.04 L/kg [b]
Diesel (by volume)	4.22 L/L [b]	4.22 L/L [b]
LPG	3.92 L/kg [b]	3.92 L/kg [b]

[a] Herath et al. (2011) for hydropower, and Scown et al. (2011) for water consumption for gas, coal, geothermal and bioenergy sources

[b] Gleick (1994)

The other vineyard operations such as fertilizer use, herbicide and pesticide applications, mowing, mulching and trimming were also accounted for. The WF of fuel use (diesel and LPG) was assessed considering the water consumption in energy production (Gleick, 1994). The manufacturing energy of agrichemicals used in the vineyard phase was obtained from reported literature (Greenhaigh et al., 2008), and the water consumption from each energy source was then quantified based on the country of production (Pfister et al., 2011).

4.4.3 Winery phase

In the winery, a quantitative impact is due to the direct use of water for washing, cleaning and staff use. The water-use impacts from the background system are dominated by the WF of packaging materials and these are mainly the paper and cardboard used in boxes, dividers and labels, as well as in glass-bottle manufacturing.

We carried out a detailed assessment of these two packaging products. The impact on water quality is mainly due to emissions from winery wastewater, and the manufacture of the packaging materials.

Glass-bottle production

Glass bottles for New Zealand wine are mainly sourced from Australia. Data were obtained directly from the bottle production plant. We found that direct water use in glass manufacture was predominantly from the evaporative losses of cooling systems. Evaporative water loss was found to be 265 L/tonne of glass, on average. The grey-WF was calculated based on the composition of wastewater from the plant as 2.04 L/bottle of wine (4.69 L/kg of glass), with ammonium nitrogen (NH_4-N) being the pollutant of concern. The water quality standard (C_m) used was NH_4-N concentration for aquatic ecosystem protection of ANZECC (2000) (0.02 mg/L) and we assumed C_b and C_n to be 0.02 mg NH_4-N /L (Eq. 4.3c)

Cardboard and paper production

We have used the reported values from the literature for the assessment of the WF of cardboard. McDevitt et al. (2012) found that green-water consumption through the forest production phase of the wood to be 1042 L/kg of paper. In addition, it takes 113L/kg of additional (blue) water for the pulping and 76.6 L/kg for the paper making (McDevitt et al., 2012). Considering the water consumption due to electricity use (Herath et al., 2011) during paper making, which is estimated at 1.6 L/kg of paper, we estimate that the blue-WF of paper is 191.2 L/kg. We use the grey-water volume generated in paper production at 14.1 L/kg (Li and Nwokoli, 2011). We combine this information with the wastewater disposal procedure used in New Zealand paper production and given by McDevitt et al. (2012). They mentioned that "…..the release of the water back into the environment from paper production is heavily regulated by an external regional governmental body and therefore it is estimated that no grey water is associated with wastewater release …". However, according to Hoekstra et al. (2011), if the waste water is treated to the acceptable standard, the grey-WF is equal to the waste water volume, if C_n and C_b is the same. Following this, we estimate the grey-WF to be 14.1 L/kg.

Winery waste water

The wastewater volume is highly variable, and fluctuates markedly with size (production level) of the winery and with the season. For the period 2008-2010, it varied from 1.1 to 5.7 L/FU. We calculated the weighted average based on the production level of the winery. The values for Marlborough and Gisborne were 1.2 and 2.4 L/FU. The composition and disposal of winery wastewater for the two regions were considered. We found that 45% of the waste water is added to the local wastewater system, wherein it is treated to bring it to the accepted standard before it reaches, or is discharged to, a water body. This system is strictly regulated in New Zealand by local government bodies under the Resource Management Act (RMA, 1991). We calculated the grey-WF for this fraction of wastewater considering the water is treated up to the standard of aquatic ecosystem protection given by the Australian and New Zealand Environment Conservation Council (ANZECC, 2000). The pollutant considered was total nitrogen. Background concentration and standards used were as in the illustration below. The grey-WF was assessed at 0.54 and 1.1 L/bottle of wine for Marlborough and Gisborne, respectively.

Some 39% of the winery waste water is directly applied to land, and this application is also strongly regulated by Regional Councils. For example, in Marlborough the consented limit has been set so as not to exceed a loading of 200 kg-N/ha/y (MDC, 2009). According to our analysis based on soil properties and the local climate, given the concentration of wastewater, the possibility of contamination of groundwater under this regulation is minimal. Therefore, the grey-WF associated with winery waste water applied to land is considered in our analysis to be zero. However, by way of illustration, we estimated the grey-WF for the highly unlikely scenario of discharging the *untreated* winery wastewater to a surface water body. This would result in a grey-WF of 137.9 L/bottle of wine, according to Eq. 3c (section 2.4.2). Total nitrogen was considered to be the dominant pollutant. Here, E is the volume of wastewater discharge (L/bottle of wine), C_e is the total N concentration in the wastewater (25 mg-N/L), C_m is the trigger value (total N: 0.6 mg-N/L) for aquatic ecosystem protection given by the Australian and New Zealand Environment Conservation Council (ANZECC, 2000), C_b is the background concentration of total N in surface waters, which we considered as 0.4 mg-N/L (Larned et al., 2004) and we assumed C_n , the natural concentration, to be the same.

This illustration emphasizes the importance of treating the wastewater, and of regulating discharges to water bodies in such a way that they have a minimal impact on the receiving environment, which reduces the impact on water quality of the resources. As well, it is imperative to make every possible effort to reduce the quantity of effluent generates and also emissions associates with its treatment.

4.4.4 Impacts of water use and emissions of wine production supply-chain: Summary

The impacts of wine-grape production and winemaking on water quantity and water quality are presented distinguishing the direct impacts and indirect impacts from the foreground and background systems of the supply chain (Table 4.3).

In both regions, the overall blue-WF (Eq. 4.1) is negative; indicating groundwater is replenished on an annual timescale under these vineyards. However, interpretation of the magnitude of the blue-WF needs further consideration taking into account the impact of other land uses in the catchment, and the potential contribution of viticulture on seasonal water-table fluctuations and groundwater- surface water interactions.

Table 4.3. The impacts of water use in wine production on quantity (blue-WF) and quality (grey-WF) of the water resources in two regions of New Zealand (Units: L/FU, where the FU is a 750-mL bottle of wine at the winery gate).

Region	System		Impact on water quantity (Blue-WF) L/FU	Impact on water quality (Grey-WF) L/FU
Marlborough	Vineyard	foreground	-81.3	40.6
		background	1.1	NA[a]
	Winery	foreground	2.7	0.5
		background	10.7	3.3
Gisborne	Vineyard	foreground	-414.9	187.8
		background	1.0	NA[a]
	Winery	foreground	3.9	1.1
		background	10.7	3.3

[a]NA - not accounted for

The grey-WF of Hoekstra et al. (2011) indicates that the impact on the quality is greater in Gisborne than in Marlborough. However, the average concentration of NO_3-N is below

its critical limit for drinking water in both regions. The absolute impacts of pollutant loads reaching the water body depend on the dynamics and properties of the receiving water bodies. For example, if the receiving water body has a large recharge rate from clean surface waters, as is the case of the Wairau River feeding the large and fast-moving Wairau Aquifer in Marlborough (Davidson and Wilson, 2011), the impact on the receiving resource will be even less than that predicted by the above calculated grey-WF. For a better understanding of the impact on the aquifer, loading of agrichemicals from all the land uses across the aquifer recharge area would need to be considered, together with the specific hydrological and biochemical conditions of the aquifer.
In exploring improvement options to reduce the impacts on water resources, an aggregated indicator of the different life-cycle stages is less than meaningful; for example, the impacts of the vineyard production phase and the winery processing stage need to be managed separately. Therefore, we recommend that they are presented separately based on the water quantity and water quality impacts (Table 4.3). This way of presenting the WF matrix, rather than a single aggregate number, should be useful to guide effective management and policy developments.

4.4.5 Practical implication of the results

Our results indicate that the main impact of wine production on water resources occurs in the wine-grape production stage. Therefore, efforts to reduce the WF and hydrological impacts should focus mainly on vineyard management. However, this does not mean that winery operations do not need some consideration. The main reason for the lesser impact from the winery stage is that the strong resource management regulation operating in New Zealand means wastewater disposal is heavily regulated and enforced. This was clearly seen in this case study with the winery wastewater discharge, as well as in the treatment of the grey-water component of cardboard and paper manufacturing by McDevitt et al. (2012).

The blue-WF indicates that in both the regions, on average, viticulture tends to maintain the recharge to the groundwater. This recharge is higher in the Gisborne region than in the Marlborough region, because of the difference in rainfall. It is seen that in humid regions like Gisborne, the impact on water quality is more prominent, while in Marlborough, water quantity aspects could be of greater concern.

However, for the allocation of irrigation consents by local authorities, both water quantity and water quality aspects need to be considered, as they are under New Zealand's Resource Management Act (RMA, 1991). These need to be informed by taking into account the land use type, as well as consideration of the local soil type and weather patterns. The hydrological water-balance method of footprinting we discuss in this paper enables this.

From the growers' perspective, the timing and amount of nitrogen fertilizer applications need to consider the pattern of crop demand and the ongoing nitrogen mineralization from soil organic material, as this will offer a means of reducing leaching losses. Managing winery wastewater through recycling and land treatment to meet local government regulations reduces the impact of wine making on water quality.

The variability encountered in vineyard water use, and its impacts, indicates the high degree of detail required for an accurate assessment of the WF of agricultural products. This is because of the very local nature of water issues, heterogeneous soils and the variability in local weather conditions. It is imperative to capture this local variability, if the WF is to be meaningful.

In addition, it needs to be understood that the impact on water quality and water quantity are not simply additive, even if the definitions of blue, green and grey-WFs can be made to be dimensionally homogeneous. Therefore, it is not straightforward to simplify water use-related impacts into a single WF value that could be used for environmental labeling. This becomes more complicated and further improvements are needed when combining with carbon footprint to understand the overall environmental impacts (Page et al., 2012).

4.5 Gaps and limitations

In our assessment, we have only considered water-use related impacts and we have not considered emissions that do not have an immediate impact on water, but may have an impact on water resources at a different timescale. For example, we have not considered the potential contribution of SO_2 emissions from the system for its impacts on water resources, or how carbon emissions might affect water resources through climate change.

Furthermore, where information is not available to enable calculation of the full hydrological water balance, especially for the supply-chain inputs, we have only been able to consider consumptive water use data, which is more easily available.

4.6 Conclusions

The impacts of wine supply chain on the water resources of two different wine-producing regions in New Zealand were assessed considering all hydrological inflows, outflows of the system. The major impact on both water quality and quantity occurs in the grape-growing stage. On average, the blue-WF, the net usage of blue water was negative in both regions. This indicates that grape growing as a land use, and wine production as an industry, do not have a deleterious impact on depletion of water resources in either region. The green-WF, the net use from the soil moisture storage on an annual time scale was negligible; therefore, its impact is insignificant.

The grey-WF, the water required to dilute the pollutants indicates that the impact on water is higher in Gisborne than in Marlborough, because of the differences in soil properties, and the higher rainfall in Gisborne. However, the average NO_3-N concentrations in the drainage under vineyards were within the drinking water quality standards. The water use and the impacts from the wine-making stage were found to be very small compared with those from the vineyard phase. Strong resource management and environmental regulations in New Zealand already contribute significantly to reducing the grey-WF of the winery production phase.

For agricultural-product WFs to be meaningful, the natural variability in the production phase needs to be well accounted for. Given the variability we have found in the impacts of water use on the local water resources within two regions of New Zealand, we recommend WFs be assessed at a local level, as they were in this study at the vineyard level.

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CHAPTER 5

Water footprinting of agricultural products: Evaluation of different protocols using a case study of New Zealand wine

At this time when international standards for water footprinting are being developed (Chapter 2), it is timely and important to consider the advantages and limitations of the different approaches that have been proposed to quantify the water footprint of products. In this chapter, the hydrological method that was set out in Chapter 4 is compared with three other recently proposed water footprinting protocols. The water footprint of a bottle of wine produced in Marlborough and Gisborne of New Zealand is assessed using all four methods. The methods were evaluated in terms of their ability to provide an unequivocal understanding of the hydrological impacts of wine-grape production and winemaking, especially at the local scale, and their usefulness to key stakeholders such as consumers, growers, winemakers and resource regulators.

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5.1 Abstract

Globally, food industries are confronting the serious challenges associated with freshwater use. The water footprint (WF) can be used as an indicator of the impacts of food production systems on freshwater resources. A number of methods for water footprinting have been proposed, while international standards are in the process of being developed. The water footprints of bottles of wine produced in two different regions of New Zealand were assessed using four water-footprinting methods: the consumptive use method of the Water Footprint Network (WFN), plus two recently proposed life cycle assessment (LCA) based methods and hydrological water balance method. The outcomes of these methods were evaluated for their ability to indicate the local impacts on water resources and their usefulness to key stakeholders.

The WF method of WFN quantifies the blue and green-water consumption, and further information on water resource availability is required to assess the impacts on local water resources. The grey-WF indicates the impact on water quality, and comparisons can be made between products, but the absolute values are less meaningful. The LCA based methods of the stress-weighted WF, freshwater ecosystem impact and freshwater depletion do indicate impacts. Also products from different locations can be compared. However, these indicators are limited in their ability to show local impacts because of constraints in their characterization factors. The WFN method and two LCA methods are based on consumptive water use. The water consumed through evapotranspiration, however, returns as precipitation within a reasonably short time and often at a local scale. So the water footprinting based on water consumption alone is limited in terms of assessing the local impact of primary production on water resources. Furthermore, the applicability of the outcomes of these three methods to growers and resource regulators is not straightforward. The hydrological water-balance method indicates the impact on water quantity as a volumetric measure, so that it can be understood by the non-technical community. For the growers, it provides a sensible measure of the impact of their production, as well as useful information for setting measurable targets to reduce the WF. It can also be helpful for resource regulators to manage their water resources by matching water demand to water availability or replenishment.

The robustness of the WF protocols for measuring the impact of product life cycle on water quality needs further improvement so that water footprinting can provide metrics of the sustainable use of our water resources.

Key words: life cycle assessment; groundwater quality; water scarcity; hydrology; environmental impacts; eco-verification

5.2. Introduction

Freshwater is an essential resource for the survival of all organisms, including humans. Although our planet is often called the 'blue planet' because of the large inventory size of its capital stocks of water, only 2.5% of this capital stock is freshwater (Oki et al., 2003). Most of that water is stored in glaciers and deep groundwater, and only a small amount is easily available in accessible reserves of groundwater, lakes, rivers and the soil-water store. Because of the disparities in availability, plus anthropogenic pressures on these limited water resources, water scarcity is becoming a widespread concern in many parts of the world (Oki et al., 2003).

Agricultural production is the major user of water globally (Hoekstra and Chapagain, 2007b; Postel, 2000). Assessing water use along the supply chain of primary production, and understanding its impacts on water availability, are an integral part of assessing the sustainability of water use. Not surprisingly then, the world's largest retailers such as Walmart (Walmart, 2009), have announced plans to develop sustainable product indices of water use and metrics of water impacts . Furthermore, the Food Ethics Council of the United Kingdom has looked at the value of labels on food products for promoting sustainable water use. They have assessed the effectiveness of these labels for communicating with consumers, and to inform them about the impact of food production on the world's water resources (Segal and MacMillan, 2009). The multifarious issues surrounding the environmental sustainability of water use are therefore of growing importance for primary industries, not just from a strictly environmental point of view, but also for product marketing to respond to consumers' questions and concerns (Sinha and Akoorie, 2010).

The concept of water footprinting has gained momentum as a metric that quantifies the potential environmental impacts related water (ISO, 2011). Water footprinting is also considered to have potential to underpin the development of environmental product declarations to communicate the performance of production systems to stakeholders (Ridoutt et al., 2009). In water footprinting, three water colours are distinguished: blue, green and grey (Hoekstra et al., 2011). The 'blue water' refers to the surface and/or groundwater used by the production system. The 'green water' refers to the rain water used by plants that had previously been stored in the root zone soil as soil moisture. The term 'grey water' is used to indicate water pollution. There are many methodologies and protocols proposed for quantifying the water footprints of goods and services. Some of these methods highlight the importance of an aggregated single-score indicator which can easily be used in communications and product-by-product comparisons (Ridoutt and Pfister, 2012). Others argue that water related impacts are difficult to combine and that a single score indicator can be misleading (Deurer et al., 2011; Hoekstra et al., 2011). However, international standards for water footprinting are being developed, so it is timely and important to consider the advantages and limitations of different approaches that have been proposed to quantify the WF of products.

There have been several attempts at evaluating different water footprinting protocols (Berger and Finkbeiner, 2012; Herath et al., 2011; Jeswani and Azapagic, 2011). Some of these have emphasised the inappropriateness of considering evapotranspiration as being 'lost' from the system, and the limitation of using the water withdrawal-to-availability ratio to characterise the water scarcity of regions (Berger and Finkbeiner, 2012). These studies do highlight the advantages and limitations of different methods. However, none of them has evaluated the outcome of different approaches in terms of their usefulness to key stakeholders such as producers, resource regulators and consumers.

In a companion paper, we have assessed the water consumption and impact of the wine supply-chain on local water resources by considering the local hydrological water balance (Herath et al., 2013). This is the precursor to the work presented here. The objective of this paper is to assess the WF of the same product, a bottle of wine produced in each of two regions of New Zealand, using the three other WF methods. We then evaluate the applicability of the four different methods for their ability to provide understanding of hydrological impacts, especially at the local scale, and their usefulness to the key stakeholders such as consumers, growers and regulators.

Three WF methods are considered in this paper, along with the hydrological water balance method for water footprinting as presented in Herath et al.(2013). The three WF methods are: the consumptive water-use based volumetric WF proposed by the Water Footprint Network (WFN) (Hoekstra et al., 2011) ; and two Life Cycle Assessment (LCA)-

based approaches, being the stress-weighted WF of Ridoutt and Pfister (2010), and the LCA-based method proposed by Canals et al. (2009).

5.3. Different water footprinting protocols and their estimation methods

As described in Herath et al.(2013), the system boundary for the New Zealand wine production starts at the raw-material acquisition and extends through to the winery gate. The functional unit (FU) was defined as a 750-ml bottle of wine at the winery gate ready for dispatch. The WF was assessed for two wine-producing regions in New Zealand. The Marlborough region, which receives an average rainfall of just 620 mm/y, is dry and relies on irrigation from groundwater for grape cultivation. The Gisborne region, which receives 1030 mm/y rainfall on average, uses no irrigation. Thus the two regions have very different hydrologies.

In the wine supply-chain, most of the water is used in the vineyard phase. The soil-water dynamics and solute transport in the vineyards were assessed using the Soil-Plant-Water-Atmosphere System Model (SPASMO) of Green et al. (2008). Different hydrological components of vineyard water use such as evapotranspiration, drainage and runoff were predicted using this model. The SPASMO model has been validated for a range of New Zealand soils under various land uses across a wide range of climatic conditions and management practices (Green and Clothier, 1999; Green and Clothier, 1995; Green et al., 1999). Specifically, this validated model has been used for wine-grape production (Green et al., 2008).The water fluxes and storage changes for all methods were modelled using SPASMO for vineyards on 29 different soil types spread across 19 climatic regions over the 12600 ha under grapes across both the regions. The actual irrigation water-use in vineyards and wineries was collected from growers and producers through the Sustainable Winegrowing New Zealand (SWNZ) initiative of Winegrowers New Zealand. Unlike carbon footprinting where impacts occur on a global scale, the impacts related to freshwater use have to be assessed and addressed with local specificity, for that is the intrinsic nature of the hydrological cycle (Deurer et al., 2011; Herath et al., 2012; Herath et al., 2011; Ridoutt et al., 2009). Therefore, this assessment was performed considering local climatic and soil conditions.

In this study, we considered four widely used water footprinting methods:

1. The method of the Water Footprint Network (WFN)
2. The stress-weighted WF of Ridoutt and Pfister (2010)
3. The method proposed by Canals et al. (2009)
4. The hydrological water balance method (Deurer et al., 2011; Herath et al., 2013)

5.3.1 Method of Water Footprint Network (WFN method)

According to the WFN method, the WF of a product is defined as the volume of freshwater used to produce the product over the full supply-chain (Hoekstra et al., 2011) . It identifies the importance of specifying water appropriation geographically and temporally. It also distinguishes the volumes of water consumed by different 'water-colors' depending on the type of water sourced and polluted (Hoekstra et al., 2011). In this approach, the blue-water component refers to the consumption of groundwaters and surface waters. The green-water element refers the consumption of the rainwater that is stored in the soil as moisture. The grey-water term indicates water pollution, and it is defined as the volume of freshwater that is required to assimilate or dilute the load of pollutants based on some appropriate water quality standard, given the natural and background concentration(Hoekstra et al., 2011) . The WFs of a wide range of products have been calculated by summing up the total volume of water consumed across the colours over the supply chain (WFN, 2012).

The various components of water use were quantified using SPASMO by distinguishing the different colours of water as follows:

- Green water: This comprises the soil-water originated from the rainfall then used as soil evaporation and vegetation transpiration from vineyards (Hoekstra et al., 2011). It also includes the green-water use from the background system, eg. through timber production for the cardboard and paper packaging materials used in the winery.
- Blue water: This is represented as the additional soil-water evaporation and crop transpiration due to irrigation in the vineyard (Hoekstra et al., 2011). As well it includes the direct water-use in the winery and the water consumption from

electricity, other fuels, plus agrichemicals and the packaging materials of mainly the glass bottles and the cardboard from the background system.

- Grey water: This component is estimated as the water required to dilute, or assimilate, the pollutants (Hoekstra et al., 2011) that reach groundwater in the drainage from vineyards. It also includes the wastewater discharge from the winery, along with the wastewater from the background system, which includes the manufacture of the glass bottles and cardboard and other packaging materials.

These different 'colours' of water use were quantified to realize the total water consumption throughout the wine supply chains of the Marlborough and Gisborne regions.

5.3.2 LCA-based water footprinting methods

The LCA approaches recognize the need to consider the impacts of fresh-water use on the environment throughout the product life cycle (Canals et al., 2009; Koehler, 2008). By identifying the limitations to understanding the local impacts with the consumptive water use based measure of WF, a number of different methods have been proposed to quantify impact oriented WFs within the standard framework of LCA (Canals et al., 2009; Pfister et al., 2009; Ridoutt and Pfister, 2010). Two of these methods are used in this paper.

5.3.2.1 Stress-weighted water footprint: The Ridoutt and Pfister (2010) method

In order to indicate the contribution of water consumption to environmental impacts related to the availability of water resources, Ridoutt and Pfister (2010) proposed a stress-weighted WF. This method focussed on assessing water use as a function of the 'water stress' on local water resources. This method followed the concept presented by Pfister et al. (2009).

According to this WF method, the impact of green-water use is considered negligible. Therefore, the stress-weighted water footprint is expressed as only an indicator of the impact of blue-water consumptive use. This was calculated by multiplying blue-water use by the regional water stress index (WSI) as proposed by Pfister et al. (2009). In this study, blue water use was calculated as the sum of r evaporative blue-water use and non-evaporative blue-water use when there is no clear evidence that the return flows can reach a water-storage body, for example, water used for pesticide applications in the vineyard phase, and the water use for cleaning in the winery. The WSI values for Marlborough and Gisborne are both 0.0101 (Pfister et al*.,* 2009) despite their quite different climatic and hydrological conditions.

In the background system (Herath et al., 2013), when products are sourced from another region or country, the relevant WSI value for blue-water WF for that particular location is used. When the specific location is not known, especially as occurs with agrichemicals and fuels sourced from other countries, the average WSI value for that country is used. Ridoutt and Pfister (2010) also presented the stress-weighted water footprint by considering the blue-water and grey-water dilution volumes together as a function of the regional water stress characterised by the WSI. However, this study only considered blue water as the source of water required to 'dilute' the pollutant, for the grey-water quantification is unknown.

5.3.2.2 The method of Canals et al. (2009)

The LCA approach by Canals et al. (2009) highlights the need to distinguish and quantify both the evaporative and non-evaporative uses of water. They also considered the role of land use changes that can lead to changes in the availability of freshwater in inventory modelling (Canals et al., 2009). The inventory accounting used here was the same as for the other methods presented above.

For the impact assessment, indicators are suggested for two main impact pathways: The freshwater ecosystem impact (FEI), and the freshwater depletion (FD). The FEI is proposed as a means to indicate the impact of blue-water consumption on the reduction in the availability of water for ecosystem functioning that affects ecosystem health (Canals et al., 2009). They explored possible characterization factors (CF) and then proposed the 'water stress indicator', which is an indicator of the water resources available for further human use, after 'reserving' resources that are necessary for the ecosystem (Smakhtin et al., 2004). This serves as the CF for the FEI. The method also considers the impact of land-use change on water availability in the calculation of FEI.

The midpoint indicator FD is used to assess the impact of the direct use of groundwater which can cause reduced availability of water in the long term. They did not present any impact pathways to characterize impacts on water quality, recognizing the need for future research to assess water-quality related impacts in an LCA framework.

Freshwater ecosystem impact (FEI)

Following Canals et al. (2009) we derived the evaporative component from blue-water use along the supply chain. To quantify the evaporative proportion of water as a result of uses other than crop evapotranspiration, we followed the guidelines of Canals et al. (2010; 2009). Additionally, Canals et al. (2009) proposed the inclusion of changes in the availability of blue water due to land-use change. According to their protocol, the land-occupation type in which vineyards are included is 'permanent crop, fruit, intensive'. Considering forest as the reference land use, Canals et al. (2009) have estimated the 'lost precipitation' due to conversion to this land use type to be zero. Ridoutt and Pfister (2010) also pointed out that most agricultural production-systems can be assumed to have no negative impact on blue water resource availability as a result of land occupation. Therefore, no change to blue-water availability was assumed as a result of land use change to viticulture in Marlborough and Gisborne.

In an LCA framework, Canals et al. (2009) pointed out that green water is used by natural ecosystems regardless of the production system being considered. Therefore, they proposed that the impact of green-water use should be excluded.

For the calculation of the FEI, we considered the 'water stress indicator' as the characterization factor, as proposed by Canals et al. (2009). We used the regionally-specified water stress indicators for New Zealand (Appendix A), which are indicators of the water resource availability for human use after 'reserving' requisite resources for ecosystem functioning (Smakhtin et al., 2004). The characterization factors for Marlborough and Gisborne regions were 0.011 and 0.017 respectively, as calculated by Herath et al. (2010), who considered regional water-use and the local availability of resources. The water stress indicator of the relevant country was used for water sourced from another country through the supply chain inputs, as given in Canals et al. (2009).

Freshwater depletion (FD)

The FD indicator quantifies the impact of the extraction of groundwater, which results in a potential reduction of the long-term availability of freshwater for future generations. For assessment of this impact category the evaporative water use from the groundwater abstraction for irrigation was quantified using the SPASMO model. As suggested by Canals et al. (2009), the Abiotic Depletion Potential (ADP) was used as the characterization factor to assess FD impacts (Guinée et al., 2002). The ADP was derived for two wine-growing regions following the method of Guinée et al. (2002) which considers antimony (Sb) as the reference resource (Eq 5.1):

$$ADP_i = \frac{ER_i - RR_i}{(R_i)^2} \times \frac{(R_{sb})^2}{DR_{sb}} \qquad \textbf{(Eq. 5.1).}$$

Here, ADP_i is the abiotic depletion potential of resource i [dimensionless]; ER_i is the extraction rate of resource i [kg/y]; RR_i [kg/y]is the regeneration rate of resource i; R_i is the ultimate reserve of the resource i [kg]; DR_{sb} is the de-accumulation rate of the reference resource (antimony); and R_{sb} is the size of the ultimate reserves of the antimony [kg]. The groundwater is the resource *i* here.

The difference of the water extraction rate and the regeneration rate of aquifers was calculated using the data from the national groundwater stock volume assessment by the Institute of Geological and Nuclear Sciences (GNS) for Statistics New Zealand for the period of 1995-2010 (Statistics New Zealand, 2012). This assessment distinguishes groundwater stocks of the confined and unconfined aquifers of different regions in New Zealand. The ADP for Marlborough and Gisborne groundwaters were calculated as 1.54 x10^{10} kg Sb equivalent/kg H_2O and 1.11 x10^{10} kg Sb equivalent/kg H_2O, respectively.

5.3.3 Hydrological water-balance approach of water footprinting

The detailed methods and analyses of the hydrological water-balance assessments are given in Herath et al. (2013). Briefly, this method assesses the impact of the production system on local water resources, separately in terms of water quantity and quality.The impact on water quantity was assessed as the sum of the net use (i.e. outflow minus inflow) from blue (groundwater) and green (soil-moisture store) water resources. This provides the blue and green-water footprints respectively.

Groundwater is the main blue-water resource immediately connected with viticulture in the study regions. The blue-water footprint was therefore calculated as the net use of groundwater, being the difference between groundwater extraction for irrigation and the recharge due to drainage plus runoff.

The net water-balance for the soil-moisture store was calculated by subtracting effective rainfall from the summation of transpiration from the vegetation, evaporation from soil, plus drainage and runoff. This was considered as the green water footprint.

The impact on water quality was quantified considering the nitrate-nitrogen in the leachate leaving the root zone of vineyards. Pesticide leaching was found to be negligible by the SPASMO modelling of the fate of the commonly used pesticides. The water quality impact was given as the average concentration and the annual loading of nitrate being received by the groundwater.

5.4. Results and Discussion

5.4.1 Consumptive-based volumetric water footprint (WFN method)

As outlined in Section 2, the WFN method represents the blue and green water consumptive uses and the estimated freshwater volume needed to assimilate pollutants. These three components are named as the blue, green and grey WFs according to this WFN method (Hoekstra et al., 2011). In Table 5.1, these components are presented for the wine supply chains of Marlborough and Gisborne. The blue WF shows a large difference between the two regions because of differences in irrigation water-use. The Marlborough vineyards are all irrigated, whereas Gisborne vineyards are all rain-fed. This method does indicate which product consumes more blue water. In this case, production of wines from Marlborough consumes more blue water than that of Gisborne wines (Table 5.1). However, the hydrological impact of this blue-water consumption depends on the local availability of water resources in the region. It cannot be deduced from these values.

Table 5.1 Water footprints calculated using the Water Footprint Network (WFN) method. Direct and indirect water use in the winery and vineyards are given separately. The units are in litres of water per functional unit (FU), where the FU is a bottle of wine at the winery gate

Stage		Marlborough			Gisborne		
		Blue (L/FU)	Green (L/FU)	Grey (L/FU)	Blue (L/FU)	Green (L/FU)	Grey (L/FU)
Vineyard	Direct	70.6	611	40.6	3.6	601.2	187.8
	Indirect	1.8	nil	NA[a]	1.5	nil	NA[a]
Winery	Direct	2.7	nil	0.5	4.7	nil	1.1
	Indirect	14.5	41.9	3.3	14.5	41.9	3.3
Total		89.6	652.9	44.4	24.3	643.1	192.2

[a]NA- not accounted for

This way of assessing the WF has been criticized for its inability to enable meaningful comparisons between the WF of products that are made in locations of differing water-resource availability. Furthermore, it is considered that it does not provide an indication of the environmental impact of the calculated blue water consumption (Ridoutt *et al*., 2009). The WFN has responded by noting that an index resulting from multiplying water volumes by certain characterization factors is somewhat meaningless from a water-resource management perspective (Hoekstra et al., 2009b). As the authors mentioned, this water consumption-based volumetric WF could be useful in water management perspective and irrigation water allocation, but it needs to be considered with other relevant information such as the size of the water resources available, and their renewability.

Hoekstra et al. (2011) have also mentioned the concept of calculating water footprint impact indices to reflect the local environmental impacts. This concept follows the similar framework of LCA-based WF methodologies. According to this, an impact indicator for each water colour can be calculated by multiplying blue and green water footprint by blue and green-water scarcity in a particular catchment for a particular time period. For the grey WF impact indicator, it is proposed to multiply the grey WF by the water pollution level. However, the method is unclear as to how the scarcity and pollution levels are quantified. At the same time, the authors argue that this way of assessing impacts and aggregating them to synthesize an overall index is not useful in terms of a specific response formulation (Hoekstra et al., 2011).

Nevertheless, the WFN concept of identifying the different colours of the water used (viz. blue, green, and grey) throughout the production cycle has been adopted as a basis by other WF methods. For example, it has been suggested that life cycle inventory modelling needs to distinguish and quantify the blue and green-water components that are associated with production and processing (Berger and Finkbeiner, 2010; Canals et al., 2009).

There is only a small difference in the green WF between the two regions. The evaporative water demand of Marlborough grapes is higher compared to Gisborne because of climatic differences, mainly the lower relative humidity. However, the grey WF for Gisborne is over four times higher than that of Marlborough, primarily because of the higher N-mineralization rate as a result of the higher soil organic matter contents in the Gisborne soils (Herath et al., 2013). This might indicate that the impact of wine production on water quality in the Gisborne region is higher than that in Marlborough. But the absolute numerical values do not indicate how critical the impacts are, because the dilution water requirement (L/FU) cannot be directly related to water quality impacts. Moreover, it is unclear how an individual grower or producer can improve, or set targets to reduce the footprints.

If the blue-, green- and grey-water footprints are aggregated, the total consumptive water footprint is 787 L/FU for Marlborough, and 860 L/FU for Gisborne. Because of the differing proportions of blue, green and grey waters in the aggregated values, it is not possible to conclude that the lower water footprint is better. This is because the utility, impacts, and opportunity costs of these various 'waters' are quite different. Others also have recognized the inappropriateness of aggregating water quality and quantity metrics, and they have suggested presenting them separately (Deurer et al., 2011; Ridoutt and Pfister, 2010; Zonderland-Thomassen and Ledgard, 2012). However, Ridoutt and Pfister (2012) comment that a profile of indicator results is less appropriate than a stand-alone single score indicator for communication to a lay audience.

The outcome of this method of assessing water footprint needs further information on water resource availability and the rate of resource renewability, for growers or producers to be able to understand the impact of their production on water resources, and how they might reduce those impacts. Likewise, this information is only partially useful for resource regulators who need to plan the sustainable management of water resources, and for consumers who might wish to know the size of the environmental burden of the product they are about to purchase.

Methods of water footprinting based only on water consumption through evapotranspiration (ET) and considering ET as a 'loss' are questionable in understanding local hydrological impacts . Certainly, evaporation and transpiration play a vital role as drivers of the hydrological cycle. The major portion of the water that leaves the system through evapotranspiration returns locally as precipitation (Van der Ent et al., 2010). In a recent study on the length and time scales of atmospheric moisture recycling, van der Ent and Savenije (2011) found that evaporation recycling into the same ecosystem occurs within a short time (ranging from 3 to 20 days in temperate climates) and distance scale (500-5000km). In addition, water consumption through transpiration plays a key role in crop production. It enables absorption and transportation of mineral nutrients and water from the soil to roots and shoots. Therefore, crop yield is well correlated with transpiration (Nahle and Kunz, 2012). However, it is emphasised here that assessing and quantifying ET provides useful information for water use and management from the orchard to catchment and beyond, but only when considered along with the other water flows and storages.

5.4.2 Stress-weighted water footprint: The Ridoutt and Pfister (2010) method

This method seeks to understand the impact of water consumption in relation to the degree of water scarcity at the location where the water is sourced. The impacts of blue-water consumption along the wine supply-chains for the two regions are given in Table 5.2. The impact of green-water consumption is ignored. According to the Ridoutt and Pfister (2010) method, the WF of wine produced in Marlborough has a higher impact than that of Gisborne (Table 5.2). The WF follows the same trend as blue-water use in the vineyard phases of the two regions. This is mainly because the WSI for both Marlborough and Gisborne are the same, at 0.0101 (Pfister et al*.,* 2009), despite their different local hydrologies. If the impact of blue-water use alone is considered, without considering water quality impacts, the major part of the impact occurs in the vineyard phase in Marlborough, while for Gisborne it comes from the background system of the winery (Table 5.2), which is mainly derived from cardboard and wine bottle production.

This method can be used to make comparisons between products from locations with different hydrologies, depending on the ability of WSI to represent the status of water scarcity of the region. However, from individual growers' point of view, the usefulness of the absolute values normalised to WSI is limited at present. With this WF, the growers would have little control over their own product WF, as the WSI value depends on other water users across the region as well. In other words, the water footprint of the production system of an individual grower could change irrespective of what that grower does. Therefore, from a practical point of view, this could discourage producers from embarking on sustainable efforts to reduce their own product's footprint.

Table 5.2 The blue-water use in different life cycle stages along the wine supply chain and the impact of water use as indicated by the stress-weighted water footprint (WF). The percentage contributions from each phase are given in parentheses.

Life cycle phase		Marlborough		Gisborne	
		Blue-water use (L/FU)	Stress-weighted WF (water equivalent L/FU)	Blue water use (L/FU)	Stress-weighted WF (water equivalent L/FU)
Vineyard	Foreground system	70.6 (78.8%)	0.707 (43.8%)	3.6 (14.8%)	0.037 (3.9%)
	Background system	1.8 (2%)	0.236 (14.6%)	1.5 (6.2%)	0.229 (23.9%)
Winery	Foreground system	2.7 (3%)	0.027 (1.7%)	4.7 (19.3%)	0.047 (4.9%)
	Background system	14.5 (16.2%)	0.646 (40%)	14.5 (59.7%)	0.646 (67.4%)
Total		89.6	1.615	24.3	0.958

The characterization factor, WSI, used in this method to convert volumetric water use into a weighted indicator, is estimated from water withdrawal relative to the availability (WTA) ratio of that location. As Berger and Finkbeiner (2012) have pointed out, WTA can be misleading, as not all water withdrawals are consumed. Hoekstra et al. (2012) noted that about 40% of agricultural water withdrawals are returned to the local water resources. When the published WSI values for different regions of New Zealand are considered, there is little discrimination, even for regions with very different local hydrologies. The stress-weighted WF characterised by the WSI is primarily correlated with the size of water consumption, which raises the same set of concerns discussed above.

Ridoutt and Pfister (2010) used the same method given by Hoekstra et al. (2011) to assess the impact on water quality. This is the dilution-based, grey-water footprint. They recognized the inappropriateness of the aggregation of this dilution-water based grey-WF with blue and green consumptive water use.

5.4.3 The method of Canals et al., (2009)

As outlined in Section 2.2.2, the method of Canals et al. (2009) assesses the impact of blue-water consumption via two impact categories, and the impact of green-water consumption is considered negligible. The results indicate that FEI, the impact of water consumption on ecosystems, is higher for the wine produced in Marlborough than it is in Gisborne (Table 5.3). According to the FEI, the major part of the impact occurs in the foreground system of vineyard phase in Marlborough wines, while for the Gisborne wines, it derives from the background system of the winery (Table 5.3). This is mainly from the cardboard used in packaging and through wine-bottle production. This is similar to the results we have found for the stress-weighted water footprint, and the method of the WFN blue WF (Tables 5. 1 and 5.2).

Table 5.3 The impact of blue-water consumption along the wine supply chain of the two regions as indicated by Freshwater Ecosystem Impact (FEI). The percentage contributions from each phase are given in parentheses.

Life cycle phase		FEI L of ecosystem-equivalent water/FU	
		Marlborough	Gisborne
Vineyard	Foreground system	0.777 (66.5%)	0.037 (8.2%)
	Background system	0.130 (11.1%)	0.147 (32.5%)
Winery	Foreground system	0.004 (0.3%)	0.010 (2.2%)
	Background system	0.258 (22.1%)	0.258 (57.1%)
Total		1.169	0.452

Canals et al. (2009) suggested consideration of the impact of land-use change on water availability in the life cycle inventory for FEI. When land transformations are considered, most arable or horticultural land uses tend to increase the availability of blue water because of the effects of the changes in transpiration, surface runoff and groundwater recharge. Yet in our study, land transformation made no difference, as there was no change in water availability assumed for viticulture following Canals et al. (2009). However, in other cases, when land-use change leads to a change of the blue-water availability could lead to unexpected consequences such as raising the water table, or greater likelihood of flooding in surrounding areas (Berger and Finkbeiner, 2012; Herath et al., 2011).

Table 5.4 The impact of groundwater consumption along the wine supply-chain of the two regions as indicated by Freshwater Depletion (FD). The ADP is the Abiotic Depletion Potential for groundwater which is the characterization factor for FD.

Region	Groundwater consumption (L/FU)	ADP for groundwater	Freshwater Depletion (FD) (Kg Sb-equivalent/FU)
Marlborough	70.6	1.54×10^{10}	1.09×10^{12}
Gisborne	3.6	1.11×10^{10}	3.99×10^{10}

The freshwater depletion (FD) in Marlborough is higher than that for Gisborne (Table 5.4). The absolute numerical values, both for FEI and FD, result from an impact assessment that does not reveal how critical the impacts are, especially, in relation to this particular production system. The FD does not clearly indicate whether the production system itself contributes to net depletion of groundwater. Therefore, use of this information to guide the setting of limits for groundwater abstraction that should not be exceeded by any particular system under analysis is not straightforward from a practical point of view. This will become clearer when these results are compared with the blue WF of the hydrological method (Section 3.4). That analysis shows on average, that the groundwater is recharged on an annual basis under viticulture in both the regions. From an individual producer's point of view, as FEI and FD are also depend on how other users manage the water resources, traceability of their own improvements is not clear. Unlike in carbon footprinting, water consumption is unavoidable in agricultural production systems. Therefore, it is important that the water footprint indicates the degree of sustainability of water consumption.

One widely understood limitation of LCA-based water-footprinting assessments is their reliance on LCA databases for inventory modelling. Often these are lacking, or incomplete, in terms of key aspects of water-use assessments (Berger and Finkbeiner, 2010). These aspects are listed below.

1. It is unclear whether all relevant inflows and outflows are included in the inventories. This is especially so when abstractions are used in inventories without considering return flow. It is the net result that is critical for the hydrology.
2. There is a dearth of geographically relevant data. It is very important to use inventory data with a high spatial resolution in terms of water-related assessments. Water use and its impacts are highly localised and temporally variable.
3. Water quality information about inflows and outflows is scarce. This aspect is critical as the utility of water resources is highly dependent on their quality.

Consequently there is a lack of consistency between the various databases that are available. The accuracy of extant inventories is in doubt with respect to both water quantity and quality.

In summary, both the LCA-based methods suggest indicators to represent the potential environmental impacts associated with water. They can facilitate somewhat meaningful comparisons to be made between different products from various locations of water availability. However, accurate inventories and localised characterization factors with further improvements in methods are needed to ensure the veracity of the assessments.

5.4.4 Hydrological water-balance approach of water footprinting

Water footprinting using the water-balance approach indicates the hydrological impacts of the production system on the local water resources with which the system is intimately connected. The net use of water resources, being the difference between the hydrological inflows and outflows, is considered as the impact. The impact is given as a volumetric measure, so that it can be easily understood by the non-technical community.

The net use of green-water storage (soil moisture) as a result of water consumption by vineyards was insignificant on an annual basis, as winter rains recharge the soil-water store. Thus, the impact on green-water storage (green WF) was considered negligible. The details of this assessment, and the methodological aspects and the results are given in Herath et al. (2013). The results are summarised in Table 5.5 for comparative purposes. These principles of assessing the water balance to evaluate the impact on groundwater by considering recharge through drainage are supported by a recent study on the water balance of aquifers through the 'groundwater footprint' by Gleeson et al. (2012). They emphasised the need for the water footprint to be able to assess the effect of water consumption on natural stocks and flows, as we have considered here in this hydrological water-balance method of water footprinting.

The net usage of groundwater (the blue WF) (Table 5.5) indicates that the blue WF is negative for the vineyard foreground system. This means that the water resources of the groundwater below the vineyards of Marlborough and Gisborne are being recharged at the rate of 81.3 and 414.9 L/FU of grapes respectively. This negative footprint implies that the studied system has no deleterious impacts on the quantity of the blue-water resource of groundwater in the region. However, given the water credit of this recharge to the resource, there still needs to be an assessment of the environmental flow requirements and the water stocks that are needed to supply the ecosystem services from these groundwater resources. This aspect has also been recognised within the LCA context. There remain issues around accounting for change in blue water availability due to land use transformation (Berger and Finkbeiner, 2012). When the system has a positive blue WF this means a net loss. This may need to be further assessed, depending on the severity of the water stress on the water resource or aquifer.

Table 5.5 The impact of water consumption along the wine supply-chain on the water resources, according to the hydrological water-balance method. The impact on water quality was estimated by considering the contamination of groundwater through nitrate leaching. The water-quality impact is given only from the vineyard phase.

Life cycle phase		**Impact on water quantity: Blue-water footprint (L/FU)**		**Impact on water quality**			
		Marlborough	Gisborne	Marlborough		Gisborne	
				Average NO_3^- N concentration (mg/L)	Average NO_3^- N load (kg-N/ha/y)	Average NO_3^- N concentration (mg-N/L)	Average NO3-N load (kg-N/ha/y)
Vineyard	Foreground system	-81.3	-414.9	5.01	4.9	8.7	29.3
	Background system	1.1	1.0				
Winery	Foreground system	2.7	3.9	-		-	
	Background system	10.7	10.7				

Even though the contribution is small, the winery stage and the background system of vineyard have positive blue WF and thus some minor impact on the blue-water resources (Table 5.5).

The impact on water quality is given as the average NO_3^- N concentration of the leachate below the root zone of the vineyards. It can then be compared directly with the NO_3-N concentration given by the New Zealand Drinking Water Standard (NZDWS) of 11.3 mg NO_3N/L. As shown in Table 5.5, on average, the NO_3-N concentration in the leachate is well less than the NZDWS for both regions. Although the concentration of nitrate in the drainage can easily be referred to standards, using concentrations as guidelines can sometimes be misleading. For example, over-irrigation can reduce the concentration of nutrients in the leachate. Therefore, loading rates of pollutants can become more important when regulatory aspects are concerned. Growers and regulators are accustomed to addressing water quality issues using either concentrations or loadings.

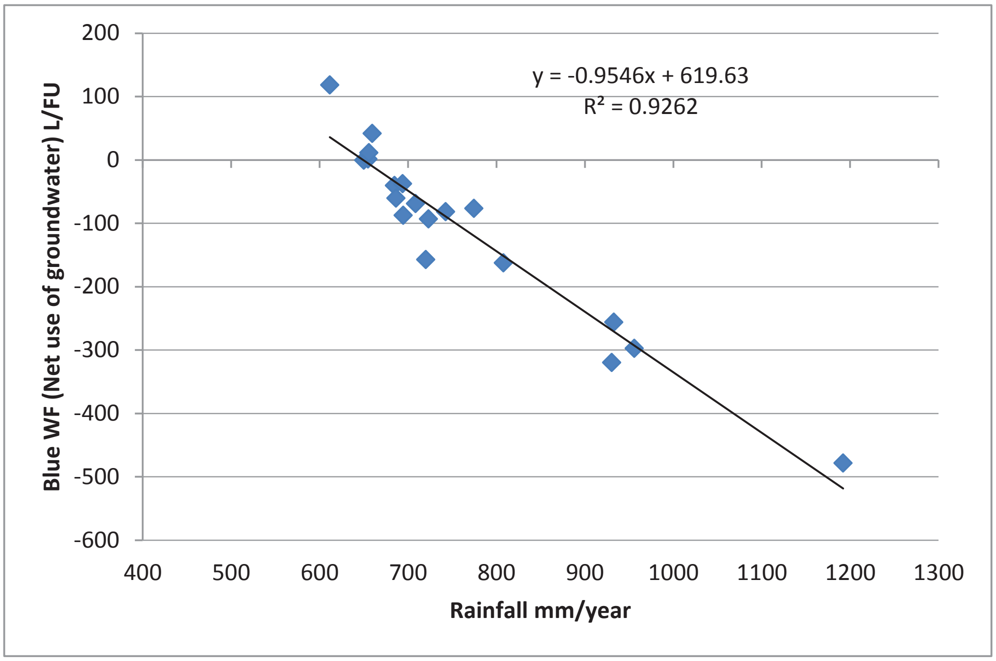

Figure 5.1 A scatter plot showing the relationship of the blue-water footprint calculated from the hydrological approach across the local climatic regions of vineyards within Marlborough, referenced to the local annual rainfall rates.

The load of nitrate reaching the groundwater resources (Table 5.5) is useful to resource managers for planning environmental regulation, as we discuss below.

As given in Herath et al.(2013), the hydrological method captures the inherent local variability of water use and the hydrological impacts of agricultural production systems on local water resources. Not surprisingly, this method is highly sensitive to rainfall, which strongly controls the local hydrology (Figure 5.1). It is essential to perform water footprinting assessments at the local scale, especially in the case of viticulture, as wine is characterised by local terroir. As shown in Figure 5.1, some locations have positive water footprints which indicate their impact on the depletion of local water resources, but the weighted average for the region is negative. So on regional average under vineyards, there is a net annual replenishment of groundwater. This contrast indicates that inappropriateness of regionalising the results and stresses the importance of quantifying footprints at the local scale. The sensitivity to the local climate should be a key consideration of any water footprinting protocol (Herath et al., 2011; Herath et al., 2013).

The main advantage of this method is it takes the focus away from just evaporated water, and places it on the net recharge of water reserves. Furthermore, it enables analysis at the local scale of the vineyard or orchard so as to assess local performance. However, combining the water quality impacts of different phases across the different pollutants, as well aggregating water quality and quantity impacts using this method needs further development. This is especially so when a single-score indicator is preferred to communicate to consumers or for labelling (Ridoutt and Pfister, 2012; Segal and MacMillan, 2009).

5.4.4.1 How the results of the hydrological method can be useful to the stakeholders

The outcome of the hydrological method can inform stakeholders, notably, growers and producers, plus resource regulators. For example, the blue-water footprint provides the impact of wine production on water resources as a volumetric indicator. This can be easily understood by non-technical people, including consumers and retailers. This information can easily be coupled with water availability and geohydrological modelling to allow resource regulators to make regulatory decisions such as for irrigation consent allocations. For instance, other land uses in the catchment may have positive water footprints and thus contributing to depletion of groundwater in varying degrees. Moreover, for growers, the outcome of this method provides a sensible measure of the impact of their production, and it is useful for setting measurable targets to reduce the impacts, and therefore to minimize the footprints.

By measuring the impact on water quality as the concentration of pollutants reaching a water body the results can easily be referred to a water quality standard to assess the degree of impact. Then, the growers can use this for setting targets to reduce the concentrations of agrichemicals leaching into the water bodies. However, over-irrigation or higher drainage can provide a false indication of less pollution due to diluted concentrations of contaminants. Therefore, presenting impacts as a contaminant loading rate (kg/ha/y) is more helpful for the resource managers to plan regulatory measures based on soils, climate, land-use types, while also considering the assimilation capacity along the flow pathways. This will support the sustainable management of water resources. This concept has been implemented into management of water resources by many local government authorities in New Zealand, and the recent One Plan policy development and the Environment Court hearings of the Manawatu-Wanganui Regional Council highlight this (Horizons Regional Council, 2008).

The hydrological method requires, as is appropriate, details about the hydrological parameters relating to soil and climate, the biophysical functioning of crops, and land management practices. However, the primary goal of any footprint assessment should eventually be identifying possible improvements to reduce environmental impacts. In agricultural production systems, these need to focus on the crop-production phase. Thus, understanding and monitoring these hydrological and biophysical parameters becomes vital. Unlike carbon footprinting, water-related impacts are very localised both in terms of water quantity and quality. Therefore, biophysical and management-practice information is needed at high spatial resolution to ensure useful and sensible outcomes.

5.5 Conclusions

The water footprint of a bottle of wine produced in each of two different regions of New Zealand was assessed using four water-footprinting methods. These methods were evaluated based on their ability to indicate impacts on local water resources, and the usefulness of the metrics to the key stakeholders in terms of ease of understanding, ability to set targets to reduce the footprint, and the applicability in regulatory policy formulation.

The volumetric WF of water footprint network (WFN) quantifies the blue and green WF based on consumption along the product life cycle. When used in this way, because of its lack of correspondence to the impact on local water resources, the comparisons of impacts between products of different regions are limited. The grey WF quantifies the impact in terms of the water required to dilute the contaminants. Here, the impact of different products can be compared, but the absolute values are somewhat less meaningful for impacts on the local hydrology. The concept of the different types of water in this method is useful and instructive, and furthermore serves as an accounting basis for the impacts assessment of water use in a catchment or region.

The two LCA-based methodologies of Ridoutt and Pfister (2010) and Canals et al. (2009) indicate the impacts of water use by supply chains. Comparisons can be made between the WF of product from different locations using these methods. However, their ability to indicate the localised impacts is limited due to the spatial limitations of the characterization factors used in these methods. At the product water-footprinting level, it is not straight forward for growers and producers to understand the impact of their practices using these two methods. As well, resource regulators cannot readily use this information to manage water resources.

The WFN method, and the two LCA based methods are based on consumptive water use. However, hydrologists have found local evaporation is linked to local rainfall within short time period and spatial scales. So, assessing water use impacts based only on water consumption alone can lead to unintended consequences.

The hydrological water-balance method for water footprinting considers all relevant liquid and vapour flows to assess the net use of the resource. It indicates the hydrological impacts of the production system on the quantity of local water resources as a volumetric measure, so that it can be easily understood by the non-technical community. For the growers, it provides a sensible measure of the impact of their production, and it represents useful information for setting measurable targets to reduce the impacts through the WF itself. It is also helpful for resource regulators to manage resources by matching demand to availability via geohydrological modelling. The impact on water quality of the vineyard phase, which is the major impact, is given as a leachate concentration, or loading rate of pollutant in the drainage. It can easily be referred to a water quality standard to assess the degree of impact. Therefore, it is useful for the growers to set targets to reduce impacts, and to track the improvements. Resource regulators can readily use this information to manage better their water resources, and to formulate policy and establish monitoring priorities. However, further developments are needed to combine the water quality impact of the different lifecycle phases in order to inform consumers.

Robust protocols for the impact of product life cycles on water quantity and water quality are rapidly evolving. There needs to be further developments, especially in water quality impact assessments, so that rational and useable water footprint metrics become adopted and used by all stakeholders.

5.6 References

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CHAPTER 6

Measuring water and nutrient dynamics to quantify and reduce the water footprint of potato cultivation

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As found in the study comparing different water footprint protocols in Chapters 4 and 5, the hydrological water balance method is better able to represent the local impacts of a perennial crop such as grapes, in addition to clearly informing the key stakeholders. With the aid of field measurements, the hydrological water-balance method is used in this chapter, in order to quantify the water footprint of an annual crop – potatoes. The functional unit is a kilogram (kg) of fresh potatoes. This chapter presents data for soil-water dynamics, as measured in the field, and the leaching of nitrate monitored under a commercial potato crop. This assessment only considers the crop production phase of the potatoes. The extended assessment, including full life cycle considerations, is presented in the following chapter (Chapter 7).

The content of this Chapter has been submitted for publication in the international journal of *Agricultural Water Management*

6.1 Abstract

Agricultural production systems have a significant impact on freshwater quantity and quality all over the world. Water footprinting (WF) has been used to quantify such impacts. To date, WF has been assessed through modelling, and none of the assessments has been based on measuring WF at the field level. Measured water use and drainage were used here to quantify the WF of potatoes grown in the Manawatu region of New Zealand. The net uses of groundwater and soil water storage were considered as the blue and green WFs, respectively.

Six tension fluxmeters were installed below the root zone and drainage was measured after every significant rainfall event. The nitrate-nitrogen (NO_3-N) concentration of the

leachate was analysed. The WF was calculated, based on measurements over the cropping season, from October 2011 to April 2012. Irrigation was not sourced, and the net use of groundwater was negative and equal to drainage. The mean of the cumulative drainage from six fluxmeters was 263.8mm (SE ± 84). Thus, the blue WF was calculated at -58.6 L/kg. This indicates that potato cultivation does not contribute to groundwater depletion in the region. The green-WF that quantifies the change in soil-water storage over the growing season was calculated at 15.8 L/kg. However, this green-water deficit would certainly be replenished later on, by winter rainfall.

The assessment of the impact of potato cultivation on water quality indicates that current practices could pose certain risks to groundwater quality. The grey WF that the water needed to 'dilute' NO_3-N to drinking water standard was found to be 133.1 L/kg of potato. The calculated impact on water quality was based on the used-fraction of nitrate assimilation capacity of groundwater, which was 4.82×10^{-11} per kg of potatoes harvested. The quantified nitrate leaching during the growing season indicates that most leaching losses can potentially be minimised through changed fertiliser management practices.

Despite the intensity of the field measurements, which revealed critical insights, the data were bedevilled by the lack of spatial and temporal representativeness of the measurement devices. It was estimated that ten groups of fluxmeters, paired on the ridge and in the furrow, would be required to reduce the standard error to 12.5% of the mean. This means a total of 20 fluxmeters, rather than the six used in this study. Modelling will, therefore, always be the dominant and cost-effective means by which WFs will be quantified. Nonetheless, model validation by measurements will always be necessary.

Key words: Drainage; leaching; groundwater; fluxmeter; water quality

6.2 Introduction

Hydrological sustainability is one of the most challenging issues the world is currently facing. Future food security is threatened by continued increase in the demand for water (Hanjra and Qureshi, 2010). Current observations and climate projections provide clear evidence that freshwater resources are vulnerable, and that they have the potential to be strongly affected by climate change. This will have wide-ranging and deep consequences for both mankind and ecosystems (Bates et al., 2008). The challenge is to meet these additional food and freshwater demands in a way that does not affect natural capital stocks and the ecosystem services that flow from them. Higher crop productivity has traditionally involved intensive management and the use of high inputs, such as irrigation, fertiliser and agrichemicals that can have potentially detrimental impacts on the environment. Optimising water and nitrogen supply is a key to achieving higher and stable yields (Spiertz, 2012). Being the largest consumer of the world's water resources (UNEP, 2007), agriculture has widespread impacts on both freshwater quantity and quality. With increasing pressure on limited water resources, there is a rising interest in the metrics of the environmental impacts of water use by agricultural production systems.

The water footprint (WF) metric has been proposed as an indicator of the impacts of water use by agricultural production systems on freshwater resources (Deurer et al., 2011; Herath et al., 2013; Ridoutt and Pfister, 2010). In water footprinting, three water colours are distinguished: blue, green and grey (Hoekstra et al., 2011). The 'blue water' refers to the surface and/or groundwater used by the production system. The 'green water' refers to the rain water used by plants that had previously been stored in the soil as soil moisture. The term 'grey water' is used to indicate water pollution.

In agricultural production systems, water-related impacts are highly variable, due to the variability in local climate and the heterogeneous nature of soil properties across rural landscapes. Therefore, the accuracy of WF assessments are highly dependent on the ability of these methods to capture the variability of local impacts (Herath et al., 2013). This local aspect becomes vital when exploring improvement options to reduce water use impacts, because the major impacts and reduction options have to be primarily focused on the cultivation and growing phase of agricultural products.

While international standards for water footprinting are in the process of being developed, there have been a number of extant protocols proposed to quantify the water footprint of agricultural products (Deurer et al., 2011; Hoekstra et al., 2011; Ridoutt and Pfister, 2010). These different water-footprinting methods have their own advantages and disadvantages. For example, the 'stress-weighted' WF is aimed at communicating water use impacts to the consumers, or end users of the product (Ridoutt and Pfister, 2012). However, its ability to inform resource management is limited. The 'volumetric' WF is intended to be more useful in water resource management (Hoekstra et al., 2011). Among the various WF methods, the protocol for assessing the WF, based on the water balance of the production system, which identifies the major inflows and outflows of the blue and green waters to/from the production system, has been shown to provide an unequivocal understanding of the hydrological impacts (Deurer et al., 2011; Herath et al., 2011; Herath et al., 2013).

Thus far, WFs of agricultural production have generally been assessed by estimating the hydrological components, such as evaporation and transpiration of the system, through modelling. This is because of the difficulty of measuring the green and blue water fluxes, and the challenge of quantifying the leaching and runoff of agrichemicals to surface and/or groundwater bodies. Most grey-water footprint calculations have, therefore, been simply based on the crude assumption that a fixed fraction of the applied fertiliser is lost through leaching (Chapagain et al., 2006; Dabrowski et al., 2009). This is a rough approximation that excludes critical factors, such as different soil types, various agricultural practices, variable local soil hydrological conditions and interactions among the different chemicals within the soil.

Measuring water and agrichemical dynamics under field conditions is expensive and time consuming and, as a result, there has not been a focused assessment of water footprinting, especially for agricultural products, based on field measurement of water use and drainage. Therefore, the objective of this study was to attempt to quantify the WF of potatoes, by measuring the water use and leaching under field conditions. We assess the viability of using measurements, rather than modelling, to quantify the WF. The advantages and difficulties of an empirical approach are emphasised. The synergy between measurements and modelling is discussed, and it is shown how measurements are critically important to modelling schema. This study also aims to identify options to reduce the water footprint of potato cultivation.

6.3 Methodology

6.3.1 Measuring the impact of crop cultivation on the quantity of water resources: the blue and green water footprints

The impacts of potato growing in the Manawatu region of New Zealand, on local water resources, were assessed using the hydrological water balance method of WF (Deurer et al., 2011; Herath et al., 2013). The impacts were assessed by considering the two main and intimately connected water resources: namely the groundwater of the blue-water resource; and the soil-moisture store of the green-water resource. The net uses of these two resources are considered as the blue and green water-footprints, respectively (Deurer et al., 2011; Herath et al., 2013). The blue and green WFs indicate the impacts related to water quantity.

The green-water footprint is defined as the net use from the soil moisture storage (green water), and the blue-water footprint, as the net use from the groundwater storage (blue water) (Deurer et al., 2011; Herath et al., 2013). The net use of green and blue water was quantified by measuring the pattern of the daily change in the rootzone soil moisture content and drainage, under field conditions.

A commercial-scale potato production system was studied for the season 2011 to 2012 in the Manawatu region of New Zealand. The soil type was Manawatu fine sandy loam (a Dystric Fluentic Eutrochrept) and the annual average rainfall has been 940mm over the 40-year period, 1972 to 2012. Irrigation was not applied, since there was generally sufficient rainfall to meet the crop's water demand. The growing season began with the planting of the potatoes on 3 October 2011 and the crop was harvested on 15 April 2012. Measuring devices were installed at the start of the season immediately after planting, and the measurements were continued throughout the season.

Tension fluxmeters were used to measure drainage under the root zone (Deurer et al., 2008 ; Gee et al., 2009; Gee et al., 2002). The fluxmeters consisted of a convergence tube, a funnel, a hanging wick and a subterranean reservoir to collect the drainage (Fig.6.1). Six tension fluxmeters were installed. They were locally paired at three sites in the field, one in the ridge and the other close by in the furrow (Fig. 6.1). After installation of the fluxmeter assembly, the soil column above the convergence ring was repacked to the original sequence of the soil at relevant depths. Drainage was collected after every significant rainfall event, by connecting a vacuum pump to the outlet tube (Fig. 6.2). The volume of leachate was measured. Irrigation was not required for this potato production system. Therefore, there was no groundwater extraction by the production system. Drainage was considered equal to the net recharge of blue water resource and therefore, the blue-WF was negative (Eq. 6.1). Surface runoff was considered negligible, since the landscape was flat to undulating. We assumed that any runoff would eventually end up in groundwater.

$$WF_{Blue} = \frac{-D \times 10}{Y} \qquad \textbf{(Eq. 6.1)}$$

Here, WF_{Blue} is the net use of blue water [L/kg of potatoes]. D is the drainage [mm] from the root zone and Y is the yield of potatoes [tonne/ha]. The factor 10 is for balancing the units.

The soil water content θ was measured using eight, three-wire, Time Domain Reflectometer (TDR) probes of 30 cm length (model CS616, Campbell Scientific Instruments Inc., USA). These were installed in the root zone at depths of 0-30 cm, 30-60 cm and 60-90 cm from the ridge. Four probes were located in the first 30 cm with two in the ridge and two in the furrow. The other four probes were placed in two sets at two depth intervals. Soil moisture content was recorded at 1-hour intervals with a data logger (model CR10X, Campbell Scientific Instruments Inc., USA).

The green-WF was quantified as the difference in the stored soil water content in the soil profile between the start and end of the season (ΔS) (Eqs. 6.2- 6.4).

In regards to the water stored in the soil profile S (mm) at any time: *t,* is the integral down to depth, z, here 0.9 m, of the profile water content at this time $\theta(z,t)$ (m^3/m^3) (Eq. 6.2). We have used our measurement of θ from surface to 0.9m to calculate $S(t)$.

$$S(t) = \int_0^{0.9} \theta(z,t)\, dz \qquad \textbf{(Eq. 6.2)}$$

Thus, the seasonal change in the stored soil water Δ*S* (mm) in the profile is the difference between the water content at the end of the cropping season, $S(t_1)$ minus the water content at the at the beginning of the season, $S(t_0)$ (Eq. 6.2).

$$\Delta S = S(t_1) - S(t_0) \qquad \textbf{(Eq 6.3)}$$

Therefore, the green-WF is,

$$WF_{Green} = \frac{\Delta S}{Y} \qquad \textbf{(Eq 6.4)}$$

Here, WF_{Green} is the net green-water consumption [L/kg of potato] from the soil-moisture store, and *Y* is the marketable yield of potatoes [tonne/ha]. In Eq. 6.4, a factor of 10 is needed to balance the units.

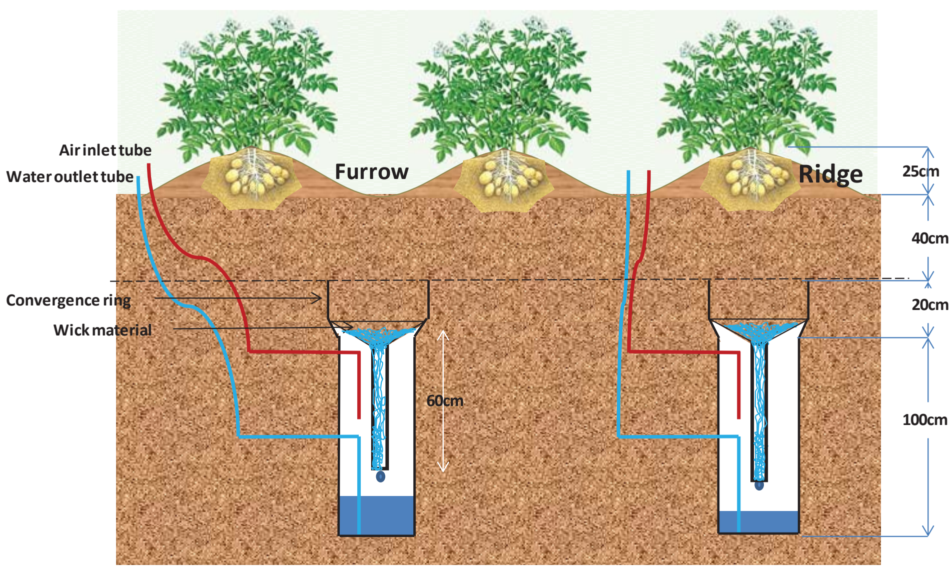

Figure 6.1 Schematic diagram showing tension fluxmeters installed under ridge (right) and furrow (left) in potato field. There were three sites with these paired fluxmeter set-ups.

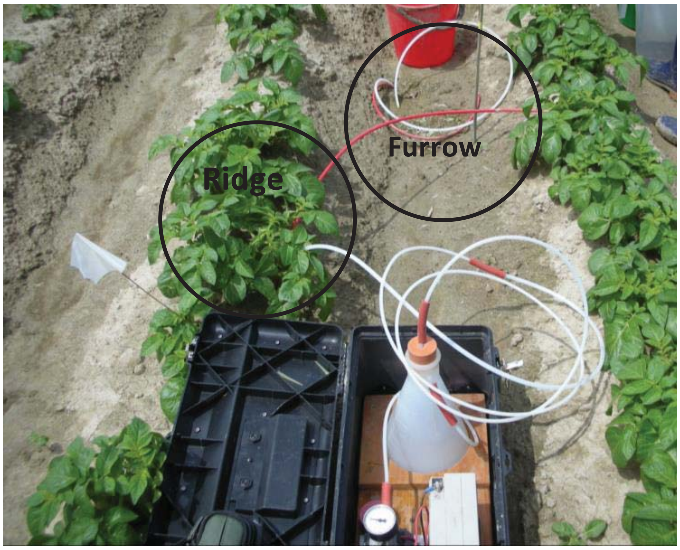

Figure 6.2 Extraction of drainage from fluxmeters by connecting the tube to a vacuum pump. Each fluxmeter has two tubes to the surface: the white one to collect water and the red for air entry and escape.

6.3.2 Measuring the impact of crop cultivation on the quality of water resources

The impact on the quality of water resources was assessed using the grey-WF, the commonly used method (Hoekstra et al., 2011). This study also presents another approach to indicate the impact on water quality, namely the impact fraction or the used-fraction of assimilation capacity of water resources by the system.

We sought to quantify the impact on water quality by measuring the drainage and leaching of nitrogen below the root zone under field conditions. After each drainage water extraction from the tension fluxmeters (Figure 6.1 and 6.2), the total drainage volume from each fluxmeter was measured and a representative sample was collected for chemical analysis. Each collected drainage water sample was analysed for the nitrate-N and ammonium-N concentrations, by using a Foss FIAStar 5000 flow injection analyzer (Foss Tecator AB, Höganäs, Sweden). Since the ammonium-N concentrations were found to be negligible, the grey WF was attributed only to NO_3-N.

The load of nitrate in kg-N/ha was quantified by multiplying the drainage volume [m^3/ha] by the concentration of NO_3-N [mg/L] in the drainage. Given the flat landscape, we assumed that extensive runoff was minimal. However, we do note (as we discuss later) that there would be local surface runoff on a small scale, as water moves across the surface before entering the soil nearby. A local surface redistribution of free-water is expected, due to micro-topography (Fig. 6.1), heterogeneity in soil properties, and the impact of cultivation practices on soil water repellency processes. These processes would probably lead to variability in the local pattern of measured drainage. It was also assumed that the drainage collected bellow the root zone reaches the groundwater at the same concentration of NO_3-N, without further denitrification, or attenuation. Two methods were used to assess the impact on water quality, by considering nitrate-N as the main pollutant.

6.3.2.1 Approach 1: the grey water footprint

Firstly, the impact on water quality was assessed, by using the grey WF of the volume of freshwater needed to 'dilute' the nitrate reaching the blue-water resource to an acceptable water quality standard (Hoekstra et al., 2011) (Eq 6.5).

$$WF_{Grey} = \left[L/\left(C_m - C_n\right)\right]/Y \qquad \textbf{(Eq 6.5)}$$

Here, WF_{Grey} is the freshwater required [L/kg of potatoes] to 'dilute' the runoff and leachate to an accepted water quality standard; L is the net-load of pollutants from the system [mg-NO_3-N/ha]; and C_m is the maximum acceptable concentration of nitrate [mg-NO_3-N/L] given by the local authorities. The standard we used here was the New Zealand drinking water standard of 11.3 mg NO_3-N/L (MoH, 2008). The natural concentration C_n is the NO_3-N concentration in the receiving water body, if there has been no human intervention (Hoekstra et al., 2011). We considered this to be zero, after considering the NO_3-N concentration found in the groundwater of this region (Daughney and Randall, 2009). Here, Y is the potato yield [kg/ha].

6.3.2.2 Approach 2: The grey water impact fraction

The critical dilution method given in Approach 1 above does not provide a clear indication on how much of the assimilation capacity of the water resource has already been used by the production system. Here, we used an approach to directly quantify the proportion of used assimilation capacity of the groundwater of the region, in order that the result has local relevance and therefore helps the growers and producers to understand the impact of their production system on local water quality.

This alternative method (Eq. 6.6) determines the fraction of assimilation capacity of the water resource that has been used by the production system. The assimilation capacity is defined as the critical load of pollutants that can be accommodated into the water body, without significant environmental damage. This critical load was calculated by multiplying the total discharge volume to the receiving water resource by the difference between the maximum allowable concentration and the background concentration, which is the prevailing concentration of the pollutant currently in the water body.

$$IF_{Grey} = \left[L / V\left(C_m - C_b\right)\right] / Y \qquad \textbf{(Eq 6.6)}$$

Here, the IF_{Grey} is the 'grey water impact fraction' which quantifies the fraction of local assimilation capacity that has been used by a kilogram of potato harvested. The volume V [L] of the resource is, here, the quantity of the receiving groundwater system of the region, and C_b is the background concentration, being the prevailing nitrate-N concentration in the water body [mg/L]. Here L, C_m and Y are the same as in Eq. 6.5. We considered C_b to be 3.03 mg/L, which is the mean nitrate-N concentration of the groundwater systems of the Manawatu region (Daughney and Randall, 2009).

6.3.2.3 Measuring plant N uptake to understand nitrogen dynamics to reduce leaching losses

Plant samples were taken randomly at two-week time intervals, four plants at a time, and they were separated into different plant parts and oven dried at 60^0C. The dry weight was used to calculate dry-matter production. The samples were ground and the total nitrogen was determined by an automated dry combustion method using a Leco TruSpec CN analyser (Leco Corporation, MI, USA). These plant N measurements, together with drainage and N leaching measurements, are used to derive potential improvement options in N fertiliser management, in order to reduce nitrate leaching in potato production in the studied system.

6.4 Results and Discussion

This growing season, 3 October 2011 to 15 April 2012, was wet compared to the average year (Table 6.1). This was especially so for the first three months of the season (October, November and December), and in particular for the first month of October after planting. However, towards the end of the season, the March and April months were drier than average.

Table 6.1 Rainfall received during the 2011/12 crop season and average rainfall for the 40-year period 1972 to 2012.

Time period	Monthly rainfall (mm) for the season 2011/12*	Average monthly rainfall (mm)*	Standard deviation
October	155.8	88.3	43.1
November	126.9	76.6	37.7
December	105.3	85.4	37.8
January	94.3	60.3	33.3
February	64.1	65.2	56.9
March	47.5	64.9	39.7
April	29.7	69.5	35.2
May-September	390.7	429.3	98.3
Annual total	1014.0	939.5	143.6

*Weather data were sourced from a meteorological station within a 5km grid of the experimental site within the Virtual Climate Station Network which is operated by the National Institute of Water and Atmospheric Research of New Zealand.

6.4.1 The impact of crop production on quantity of water resources

The impact of water use for potato production on blue and green water resources were considered separately. The impact on the blue-water resource of the local groundwater system was assessed by considering two aspects: water quantity and water quality.

6.4.1.1 The net use of groundwater (blue-WF)

Extensive surface runoff was observed to be minimal and therefore it was assumed to be negligible, since the landscape is almost flat. However, we did observe local runoff in the field during rainfall, with the spatial scale of the flows being 1-5 metres. Re-entry of this runoff into the soil was observed within this spatial scale.

There was large variability in the measured drainage among the fluxmeters (Fig. 6.3). However, there was a temporal consistency in the spatial pattern of drainage volume measured by the fluxmeters over the season. The fluxmeter FM-3F (FM-fluxmeter; 3-site 3; F-furrow) always showed the highest drainage and FM-1R (FM-fluxmeter; 1-site 1; R-ridge) the lowest. In each pair at a given site, the fluxmeter in the furrow (F) always returned a higher drainage volume than that obtained from the ridge (R). Given time-to-ponding considerations during rainfall, and the potential impacts due to water repellency, the local relief of the mounds would make this observation expected, even if the magnitude of the ridge and furrow differences are surprisingly large.

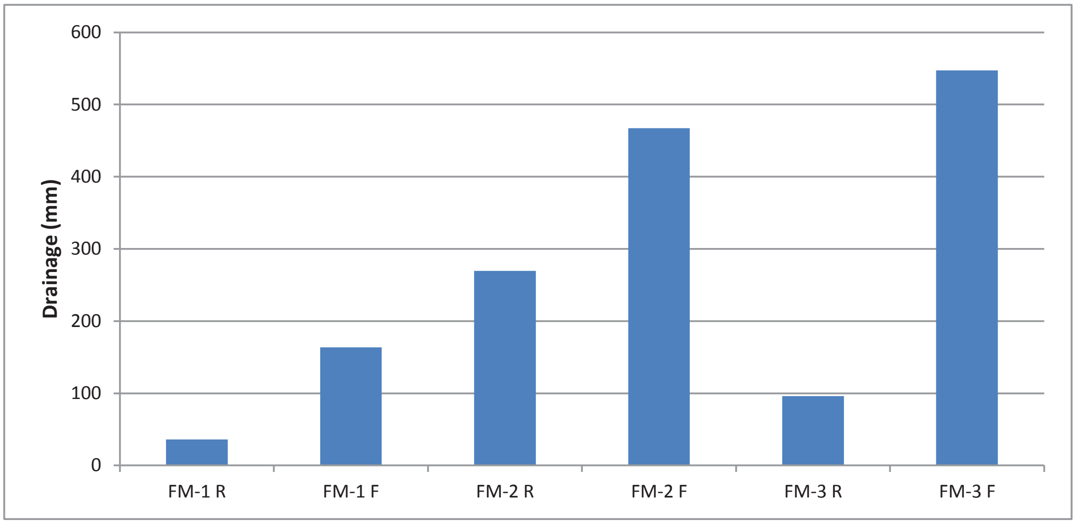

Figure 6.3 Cumulative drainage measured from six fluxmeters during the 2011/2012 potato-growing season (03 October 2011 to 15 April 2012). The 'R' and 'F' indicate fluxmeters located in the ridge and furrow, respectively.

At the beginning of the season, there were high intensity and large rainfall events (Table 6.1 and Fig 6.4). Immediately after the installation of fluxmeters, there was an intense and large rainfall event. Therefore, the first drainage event was exceptionally high with a large variability in drainage among the fluxmeters (Fig. 6.4). Immediately after planting, because of ridging and the insertion of the fluxmeters, the top soil could possibly have loosened to a lower bulk density, which might have eventually caused high infiltration through macrospores. In addition, the localised infiltration was obviously affected by local runoff, and local run-on induced by microtopography and local soil-water repellency characteristics. Furthermore, at the time of planting, just when measurements commenced, the soil was very wet from previous rainfall events. This would have contributed to the large drainage events at the onset of this potato cropping season.

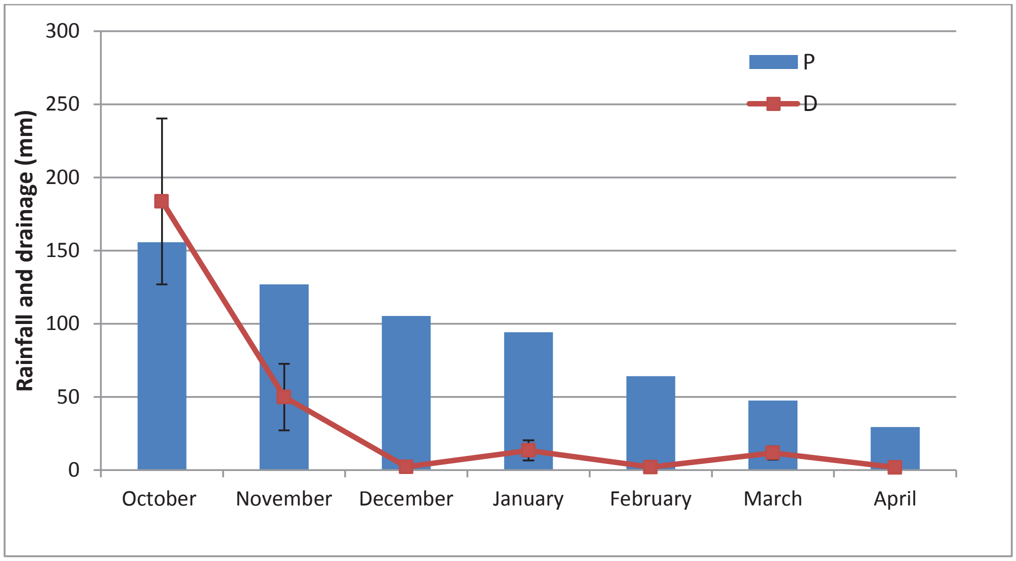

Figure 6.4 Monthly rainfall (P) and mean drainage (D) measured from six fluxmeters during the 2011/2012 potato-growing season (3 October 2011 to 15 April 2012). The bar graph indicates precipitation (rainfall) and the line graph shows the drainage. The errors bars of the line graph indicate standard error of mean drainage.

The total rainfall during the potato-growing season (October- April) was 600 mm, and the mean cumulative drainage from six fluxmeters was 263.8 mm (SE±84). This drainage contributes to groundwater recharge. Since irrigation was not used, the net use of groundwater (blue water) was negative. This means that rain-fed potato cultivation did not have any deleterious impact on the water quantity in the region.

The potato yield from the experimental site for the 2011/2012 season was 45 tonnes/ha. Thus, the blue WF was calculated as -58.6 L/kg. This indicates that the blue-water resources were being recharged under potato cultivation, at the rate of 58.6 L per kilogram of potato harvested in this area.

6.4.1.2 The net use of soil water (green-WF)

As expected, the soil moisture content over 0-90 cm fluctuated during the cropping season, and it was different in both value and pattern with depth down the profile (Fig. 6.5). The soil moisture content showed less variability with increasing soil depth. There was a drop of $0.071 m^3/m^3$ in the total soil water content in the top 0.9 m, by the end of the growing season, compared to that at planting (Eq. 6.3). The calculated green-WF (Eq. 6.4) for this change in soil water was, therefore, 15.8 L/kg. Nevertheless, the impact of this soil-water deficit is minor, since it can only be used by the current land use and therefore, the opportunity cost of this green water use is small. Furthermore, there is a probability of replenishment of the soil moisture reserve with winter rainfall following the cropping season, before subsequent planting, or changed land-use by the following October. This winter-return to field capacity of soil has been observed in many other studies (Deurer et al., 2011; Herath et al., 2013). In life cycle assessment based methods of water footprinting, the impact of green water consumption is considered negligible because, regardless of the system, green water is used by both productive and natural ecosystems (Canals et al., 2009; Ridoutt and Pfister, 2009).

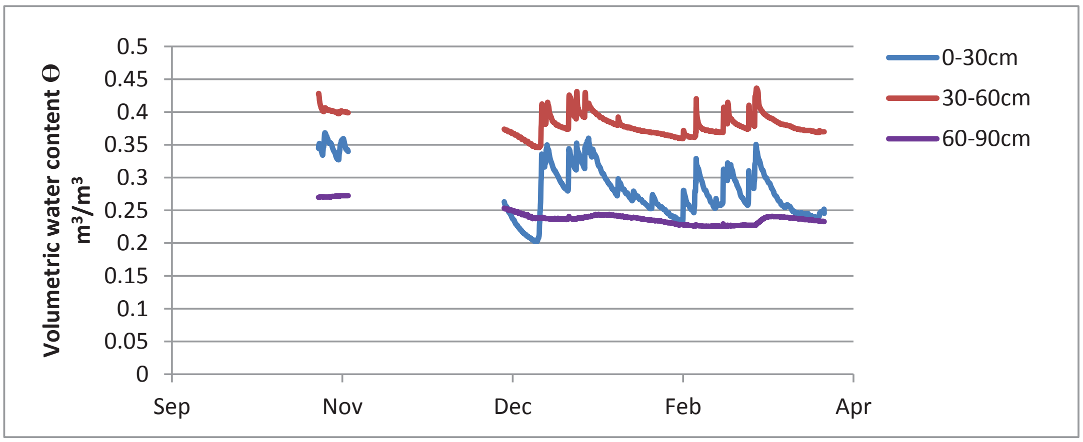

Figure 6.5 Seasonal dynamics of soil water content measured using Time Domain Reflectometer (TDR) probes at different soil depths over growing season of potatoes through to harvest. Potatoes were planted on 3 October 2011 and harvested on 15 April 2012. The break in the data was due to battery failure in data loggers.

6.4.2 The impact on water quality

6.4.2.1 Approach 1: The grey-water footprint

Following the definition of Hoekstra et al. (2011) , the grey-WF can be calculated as the water required to 'dilute' the NO_3-N in the leachate to an acceptable water-quality standard (Eq 6.5). The standard we used here was the New Zealand drinking water standard of 11.3 mg NO_3-N/L (MoH, 2008). The mean of the cumulative NO_3-N leached during the potato-growing season was measured at 67.7 kg-N/ha (SE±56.6). This value is less than most N-leaching estimates under potato production systems, both in New Zealand (Crush et al., 1997; Francis et al., 2003; Sinton et al., 2009) and internationally (Jiang et al., 2011; Prunty and Greenland, 1997).

In most of the current protocols for grey-water footprinting, the calculations have been based on the simple assumption that, on average, 10% of the nitrogen applied as fertiliser is lost through leaching (Chapagain et al., 2006; Dabrowski et al., 2009). In this potato cropping system, nitrogen was applied as an N P K mixture, with 12% of N (5.2% nitrate and 6.8% ammonium), and the rate of application was 1 tonne/ha. Therefore, the total N sourced from fertiliser was 120 kg/ha, applied at the time of planting. Our results show that, by the time of harvest, the leaching loss was equivalent to 56% of the applied fertiliser. Some of this lost N could also have been sourced from the mineralisation of soil organic matter in the root zone.

In regards to this potato system, the grey-WF calculated, using Eq. 6.1, was 133.1 L/kg. This is an indication of the purported impact on the receiving local groundwater body quantified as the amount of water needed to dilute the nitrate in the drainage, in order to meet the drinking water standard. This method could be used to compare the impact of different products, but the absolute value is less meaningful in understanding local impacts on water resources. Therefore, its use in regulatory policy formulation is limited, as a value, when embarking on reduction options.

6.4.2.2 Approach 2: The Impact fraction

In this second approach, the impact on water quality was calculated, by accounting for the fraction of assimilation capacity of nitrate nitrogen in the groundwater that is being used by the system (Eq 6.6). The estimated groundwater volume of the Manawatu region was 3.77×10^{9} $m^{3,}$ for the period 1995-2010, according to data from the Institute of Geological and Nuclear Sciences (GNS) supplied to Statistics New Zealand (Statistics New Zealand, 2012). The concentration difference between the drinking water standard and the extant background concentration is 8.27 mg-NO_3-N/L. Using these values, the assimilation capacity for NO_3-N in the local groundwater system was estimated to be 31.19×10^{6} kg-NO_3-N. Therefore, the fraction of assimilation capacity used in producing 1 kg of potato was 4.82×10^{-11}. This is a small number, but it would need to be compared cumulatively with other land-uses across the landscapes that are above these groundwater systems. The results can also be expressed based on area: 1 ha of potatoes, grown under the conditions of this study, uses just 0.0002% of NO_3-N assimilation capacity of the groundwaters of the region. Such an accumulation method would allow comparison across the various land uses that comprise our productive landscapes.

Although we have considered a fixed and static volume of groundwater, the aquifer resources are being recharged by lateral flow, and therefore they are quite dynamic. Here, it was assumed that the effect of the water quality from the lateral recharge is represented by the prevailing NO_3-N concentration of the whole groundwater system. This calculation for potatoes was made by considering the groundwater quantity and quality of the Manawatu region of New Zealand as a single hydrological system. More accurately, this assessment can then be performed by considering the water quantity and quality of individual aquifers. At the catchment, or watershed level, this method could be used to understand the non-point source contribution, from different land uses, on the pollution of local water resources. It could be considered separately for each pollutant, if need be. Therefore, Approach 2 can provide guidance to set sustainable limits to leaching losses from different land uses across various soil types, to ensure maintenance of the ecosystem services demanded of ourgroundwaters.

6.5 Improvement options to reduce the water footprints

Irrigation was not applied and there was little direct blue-water use for pesticide application. Furthermore, there is little opportunity to alter the green-WF, since transpiration is essentially linked to plant productivity. Therefore, we mainly focused on improvement options, in order to reduce the grey-WF and thus reduce the impact on water quality due to nitrate leaching. The processes controlling nitrate leaching vary widely and depend on soil properties and meteorological conditions, in addition to crop management practices (Jiang et al., 2011).

We found that 71% of leached NO_3-N occurred during the first 30 days after planting, and 90% occurred during the first 60 days. This is mainly due to low plant nitrogen demand during this period, since seed potatoes germinate and they do not have any (or limited) leaf area to drive nutrient uptake at this initial stage. According to our observations, this uptake took 3-4 weeks, from planting to emergence. However, fertiliser was applied at the time of planting. During this time, the weather was very wet, with high rainfall. This resulted in high drainage volumes, which caused leaching of most of the equivalent amount of the applied N fertiliser (Fig. 6.6).

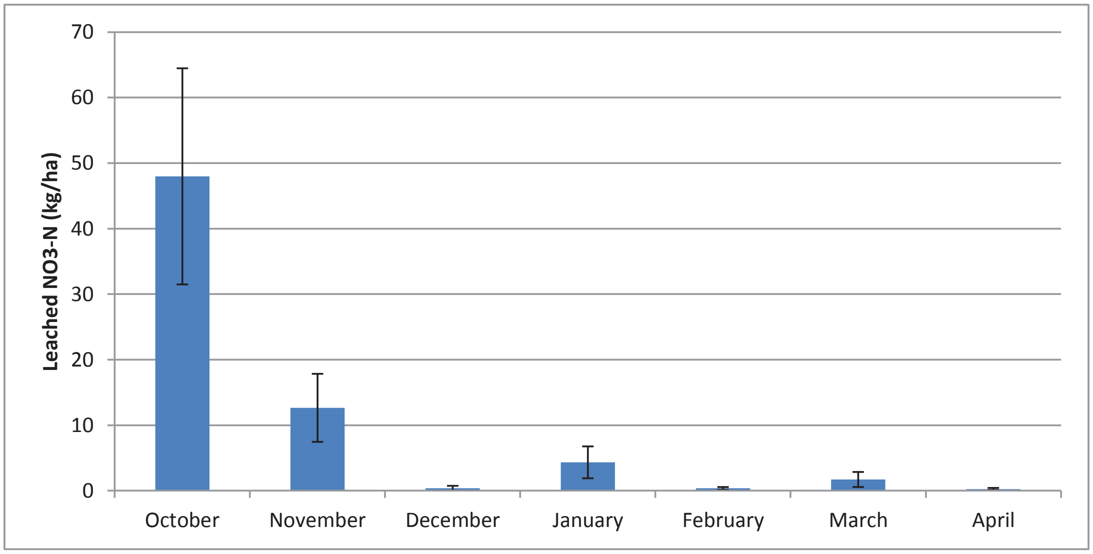

Figure 6.6 Monthly pattern of nitrate load leached during potato cropping 2011/2012. Bars indicate standard errors of the mean.

The dry matter accumulation and nitrogen uptake, by different parts of the potato plant, were also measured over the season, in order to develop an understanding of the crop's N demands. This helped us with our aim to match nitrogen supply with plant demand, in order to minimise leaching losses. The dry-matter partitioning and nitrogen uptake pattern will be used to validate the model for prediction of the nitrogen dynamics in the potato production system.

Crop N uptake was low during the first five weeks after planting. Thereafter, N demand increased as the crop developed, until approximately 80-90 days after planting (Fig. 6.7). Other studies of temporal patterns in N uptake have also shown that up to 80% of the total N uptake by potatoes occurs between 20 and 60 days after emergence (Munoz et al., 2005). Therefore, having large amounts of labile N available early on, means that the nitrogen is prone to leaching. This risk can be realised should wet conditions prevail early in the season, as happened in this study.

There have been many studies carried out on the split application of N and its effect on N leaching and potato yield. Rosen and Bierman (2008), in their experiment with the split application of the same amount of N fertiliser in different proportions at planting, emergence and hilling, found that the lowest N leaching and the highest marketable yield of potato were achieved by splitting the application equally between emergence and hilling, and with none at planting. In another experiment, increasing the amount of N applied at planting did not affect the total marketable yield (Errebhi et al., 1998). Therefore, application of nitrate fertiliser as a side-dressing, when the root system is well developed and plant nitrogen demand is high, would help to reduce leaching losses. In the next chapter, the magnitude of this saving will be predicted, by using a mechanistic model. This side-dressing could be done during the next field operation after planting, which is the ridging operation to re-form the mounds. This involves soil disturbance, since the mounds are formed around the potato ridges, and it would be possible to devise a fertiliser injection system to provide nitrogen exactly where and when it is needed.

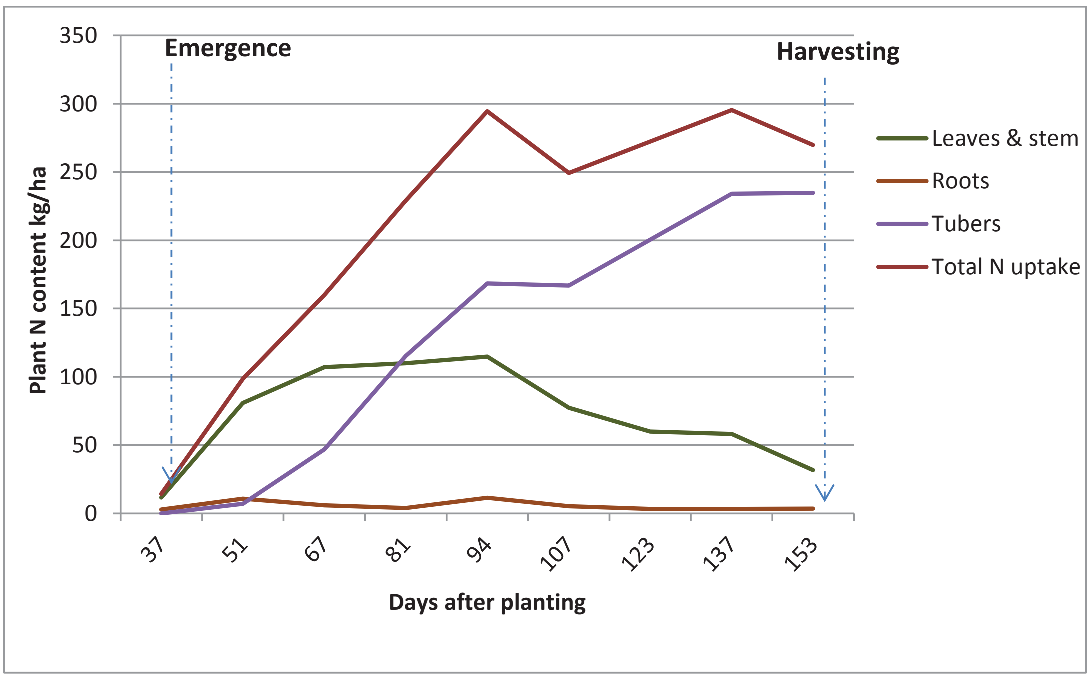

Figure 6.7 Total nitrogen (N) content of different plant parts during potato crop development.

Plant available N is derived either from fertiliser, or indigenously from the soil's mineralisation of organic material. Therefore, further understanding of the nitrogen mineralisation potential of soil and accordingly managed fertiliser application is the key to reducing leaching losses, without compromising the yield. It is probable that the surge of soil mineralisation, due to soil disturbance at the time of both planting and ridging, would provide a flush of nutrients that would sustain the emerging plant, at least until the supply from the fertiliser could take over.

6.6 Practical limitations and focus for future work

Potatoes are typically grown in this region in rotation with a cover crop, or other crops such as maize. This study only considered the potato cultivation phase, and it was only conducted during one growing season. In order to obtain a better estimation of the WF, it is important to consider the complete crop rotation. Furthermore, in order to capture the year-to-year variability in weather, modelling of water use is important, through consideration of periods of long-term weather data, in order to provide a stochastic assessment of the impacts that will account for the natural variability.

From a life cycle assessment perspective, it is important to assess the impacts across the whole life cycle, including inputs from the background system. For example, seed potatoes are sourced externally, having gone through a series of multiplication cycles. Therefore, the impacts from those inputs might also make a significant contribution to the potato life cycle. Our future research endeavours will address these aspects.

In this study, measurements were collected from six fluxmeters paired in the ridge and furrow at three local sites. As we have explained above, there was a large variability in drainage among the six fluxmeters, for reasons such as local micro-topographical differences and variability in soil hydrophilic and hydrophobic characteristics. The use of more fluxmeters would be one possible way to capture this field-scale variability, in order to obtain a better representation of the variability in drainage, and a better estimate of the mean value. Based on the standard errors in our measurements, we have estimated the change in the error of measurement that would occur, if more fluxmeters were used (Fig. 6.8). This estimation shows that having 10 pairs (seven more pairs than used here) would reduce the standard error of the mean to 12.5%. This is approximately half of the current error ratio. There would not be a significant improvement in the standard error by increasing the number of fluxmeter pairs from 15 (10% error ratio) to 50 (6%).

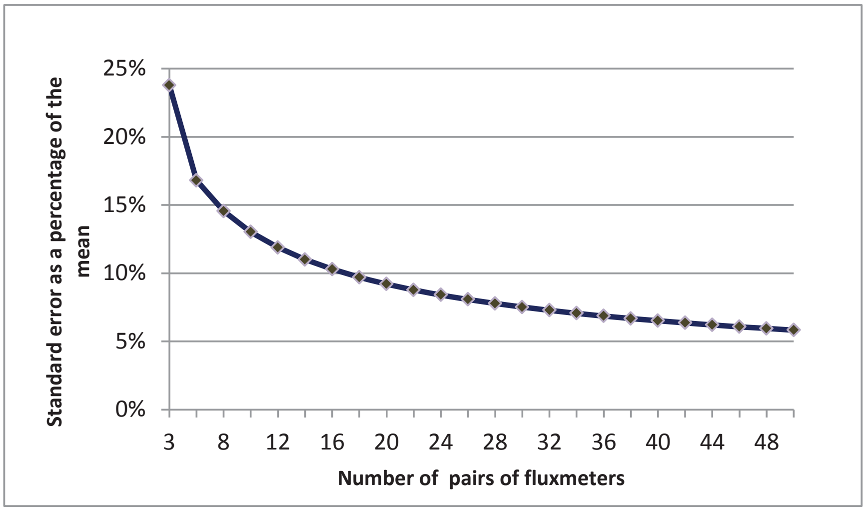

Figure 6.8 Predicted change in error of drainage measurements when using increasing number of flux meters.

The use of a larger number of flux meters would obviously reduce the errors of observation. However, this would be expensive in terms of equipment, and time consuming in order to obtain the drainage volumes. This would be impractical. Furthermore, the 2011-2012 year was wetter than average (Table 6.1). In order for a product's WF to be assessed accurately, there is a need to consider a wider temporal scale of measurements, which again would involve practical difficulties. Therefore, modelling, supported with sufficient measurements, is the better option for simulating long-term weather, when representing WF components.

6.7 Conclusions

From our measurements of the quantitative impacts of potato production, the blue-WF, which is the net use of groundwater, was negative, at -58.6 L/kg. This means that rain-fed potato cultivation does not have any deleterious impact on the water quantity of the Manawatu Region in New Zealand. The green-WF of the change in soil-water storage, over the potato growing season from October 2011 to April, 2012, was 15.8 L/kg. This change was due to a drop in soil moisture content at the end of the season, compared to that at the time of planting. However, this deficit would certainly be replenished by winter rainfall, before the beginning of the next season, as we have found in our previous studies on other perennial and annual crops.

Our assessment of the impact of potato production on water quality indicates that current practices could pose certain risks to groundwater quality, local assimilation capacity notwithstanding. According to Approach 1, the water needed to 'dilute' nitrate nitrogen to drinking water standard was 133.1 L/kg of potato. According to Approach 2, the used-fraction of nitrate assimilation capacity of groundwater was 4.82×10^{-11}/kg of potato.

The assessment of nitrate leaching over the growing season indicates that most leaching losses can potentially be minimised through changed fertiliser management practices. We suggest avoiding fertiliser application at the time of planting since, at this time, crop demand for nitrogen is very low, and there would be other sources of nitrogen to enable emergence and early growth. Alternatively, splitting the application of N fertiliser to match crop demand would significantly reduce nitrate leaching at the early stage of the season, with a second more appropriate application of fertiliser application at mounding, in order that the trafficking cost of the later application can be minimised. These options will be explored in more detail through the modelling in Chapter 7.

Despite the intensity of the field measurements, which revealed critical insights, the field data were bedevilled by a lack of spatial and temporal representativeness. Modelling will, therefore, always be the critical means by which WFs will be quantified. Nonetheless, model validation will always be necessary.

Since a potato crop is grown in a rotation, it is important to consider a complete growing sequence, and for a longer time period, in order to capture the effects of long-term cultivation, including potato-cover crop rotation and annual climate variability. We will also model this rotational sequence, and devise an allocation scheme to assign WFs to the potato phase.

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CHAPTER 7

The water footprint of potato production: A hydrological assessment using measurements and modelling

The water footprint of potato production was assessed in Chapter 6, by using measured soil water dynamics and measured drainage and leachate for just one potato growing season. It is important to recognise that, when quantifying the water footprint, long-term weather patterns need to be accounted for, in order to ensure that a representative footprint value is established. This chapter presents water footprint results from extended mechanistic modelling, for the cropping sequence of a potato crop and green cover-crop rotation. This assessment covers the water footprint of the full lifecycle, and the system boundary extends from field cultivation through to the packhouse gate. Both foreground and background systems are considered. It also considers the water footprint associated with the production of the seed potatoes, as they go through six generations of multiplication. The modelling was carried out using a long-term weather record of 40 years. Furthermore, modelling was used to establish fertiliser management practices that would reduce the grey-water footprint, due to the leakage of nutrients from the rootzone of the potatoes. A number of methodological issues will also be discussed.

The content of this Chapter has been submitted for publication in the international journal of *Agricultural Water Management*

7.1 Abstract

Water footprinting (WF) is being mooted as a metric to quantify the impacts of production on water resources. The impact of water use in the potato-production chain on water resources was assessed. This was specifically modelled for the Manawatu region of New Zealand, using mechanistic modelling supported by field measurements of soil water content, drainage and leaching under rain-fed potatoes over the 2011/2012 growing season. The hydrological water-balance method was used to assess the impact of potato production on local water resources in the region. This method accounts for all inflows and outflows to and from water resources, in order to quantify the net use of groundwater as blue-WF, and of the soil-water as green WF. The functional unit was a kilogram of potatoes at the packhouse gate. The green-WF was found to be negligible. The blue-WF was negative at -72 L/kg, thus indicating that groundwater is recharged by 72 L for a kilogram of potatoes harvested in the region. This suggests that the modelled rain-fed potato production system has no deleterious impacts on the blue water quantity in the region.

The grey-WF was 61 L/kg, of which 56 L/kg is from the cropping stage. The impact of the packhouse phase and the background system, including seed-potato production, was found to be small on the region's water resources. The use of the absolute value of the grey-WF, to understand the impact on water quality, is not straightforward. However, average concentrations and loading of nitrate from the cultivation phase do indicate that current practices are having some impact on water quality. The average concentration of nitrate leaching below the root zone was at 11.3 mg/L, which is just at the drinking water standard. The average loading rate of NO_3-N was at 27.8 kg/ha/y. Modelling of different fertiliser application scenarios of two splits, three splits and a late application at 55 days after planting, reduced nitrate concentrations and loads from the production system. The nitrate concentrations were simulated 10.5, 10.3 and 9.5 mg/L, and the loading rates were 25.6, 25.2 and 24.3 kg NO_3-N/ha/y, respectively. Together with the added nitrogen, there would be sufficient nitrogen mineralisation in the soil and therefore, yield would not be compromised in this potato-cover crop rotation, and the grey WF would be reduced to 50.6, 50.1 and 48.9 L/kg respectively.

Keywords: nitrate leaching; environmental impact; soil water; ground water; agriculture

7.2 Introduction

Potatoes provide more calories, vitamins and nutrients per area of land sown than other staple crops (Nunn and Qian, 2011). In New Zealand, potato is the largest vegetable crop in terms of area under cultivation, accounting for 10,500 ha of production (Potatoes New Zealand, 2012). Potatoes are grown in all parts of the country and all year round under different management practices including irrigation and rain-fed, at both commercial and home gardening scales. Crop development programmes in recent years have led to highly productive potato cultivars that require intensive management and high levels of fertilisers and other agrichemicals, plus irrigation. Drainage from such highly productive agricultural lands is increasingly being perceived as a major contributor to off-site environmental impacts (Crush et al., 1997; Francis et al., 2003; Skaggs et al., 1994).

Water footprinting is being considered as a metric that can be used to understand the environmental impacts related to water use and water-borne emissions. Among the different methods that have been proposed for water footprinting, the hydrological water-balance method has shown to provide a better understanding of the local hydrological impact of agricultural production systems (Deurer et al., 2011; Herath et al., 2011; Herath et al., 2013). Due to the difficulty of measuring the green and blue water fluxes and the challenge of quantifying the leaching and runoff of agrichemicals to surface and/or groundwater bodies, the WF calculations have been based on many assumptions. For instance, most grey-water footprint calculations have been simply based on the crude assumption that a fixed fraction of the applied fertiliser is lost through leaching (Chapagain et al., 2006; Dabrowski et al., 2009). This is a rough approximation that excludes critical factors, such as different soil types, various agricultural practices, variable local soil hydrological conditions and interactions among the different chemicals within the soil.

As presented in Chapter 6, the water footprint of potato production was assessed, by using measured soil water content dynamics, drainage and leachate over the 2011/2012 potato growing season in the Manawatu Region of New Zealand. However, the water and nutrient dynamics varies from year to year, depending on weather conditions and hence, a significant temporal variability in the measured WFs is expected. Field measurements used to capture this variability are time consuming, expensive and labour intensive. Combining field measurements with modelling provides a useful tool to model and estimate the long-term average WFs of a production system. In this study, the analysis from the measurements presented in Chapter 6 was extended by modelling the cropping sequence of a potato crop and green cover-crop rotation using the SPASMO model (Green and Clothier, 1999; Green et al., 2008).

The main objective of this study was to assess the impact of the life cycle of a fresh potato production system on water resources in the Manawatu Region in New Zealand. This analysis was undertaken using a mechanistic modelling approach that was robustly supported by field measurements, in order to verifiably quantify the hydrological components of the production system. We also aimed to assess the effect of nitrogen fertiliser management on both the grey-water footprint and the nitrate loadings from the production system. We modelled improvement options that would reduce the impact on water quality.

7.3 Methodology

A life-cycle based approach was used to assess freshwater use and its impacts along the potato production chain. A system boundary was established from raw-material acquisition, through cultivation and into the packhouse. The functional unit was a kilogram of fresh potatoes ready for dispatch at the packhouse gate. The life cycle of this system primarily consists of two phases: the crop production stage in the field, and the packhouse stage, when the harvested potatoes are cleaned and packed ready for market distribution. In regards to the potato cropping system, the seed potatoes are sourced externally. These two systems were separately studied in detail.

7.3.1 The crop production phase

The study was conducted on a commercial-scale potato-production system in the Manawatu Region of New Zealand. The soil type was Manawatu fine sandy loam (Dystric Fluventic Eutrochrept) and the average annual rainfall was 940 mm over a 40-year period (1972-2012). Irrigation was not applied, since there was generally sufficient rainfall to meet the crop's water demand.

7.3.1.1 Quantifying water and nutrient dynamics

In the previous chapter, it was found that, for a reasonable assessment by measurements alone, there would need to be many more measuring devices than would be practicable, in order to overcome the spatial variability, and that longer term measurements would be required to capture the temporal variability. Here, a mechanistic modelling approach was used and it was robustly supported by field measurements, in order to quantify the water and nutrient dynamics and fluxes of the production system. Field measurements were conducted for soil moisture, drainage and leaching over a full year (from October 2011 to October 2012) of the potato-cover crop sequence. The soil moisture content was measured using eight, three-wire Time Domain Reflectometer (TDR) probes. Six tension fluxmeters (Deurer et al., 2008 ; Gee et al., 2009) were used to measure the drainage and leaching under the root zone of the potatoes, as explained in Chapter 6. The field measurements were combined with the Soil Plant Atmosphere System Model (SPASMO) (Green and Clothier, 1999; Green and Clothier, 1995), in order to simulate the soil-water dynamics and solute transport, by considering a 40-year period (1972 to 2012) of actual weather data for the site (NIWA, 2012). Here, the analysis considered a typical practice of potato planting in early October, harvesting in late March and the planting of a green cover-crop that would be ploughed in at the end of the season. The model was validated by comparing the model predictions of the daily water and nitrate fluxes with the field measurements of drainage and nitrate leaching at the field sites. Subsequently, the long-term water and nitrate predictions by SPASMO were used to assess the average blue, green and grey WFs, and to explore various management options by splitting the fertiliser applications to reduce the grey WF.

7.3.1.2 Quantifying blue, green and grey water footprints

The blue water footprint

The blue-water footprint was quantified by calculating the net use of blue-water resource, which we take as the groundwater (Herath et al., 2013). The model-predicted hydrological components were used to calculate the blue WF using following equation.

$$WF_{Blue} = \frac{10\{I - (D+R)\}}{Y} \qquad \textbf{(Eq. 7.1)}$$

Here, WF_{Blue} is the net use of blue water [L/kg of potatoes] from groundwater and *I* is the amount of irrigation water used [mm/y]. Since no irrigation is used in this system, uptake from groundwater is zero. The *D* in Eq. 7.1 is the drainage from the root zone [mm/y], and *R* is the surface runoff [mm/y]. Since the landscape is flat to undulating, we have assumed that any surface runoff eventually recharges groundwater resources. Here, *Y* is the potato yield [tonne/ha/y]. The factor 10 is for the conversion of units. The measured blue-WF was taken as the average drainage (*D*) quantified, by using the eight fluxmeters over the 2011/12 growing season.

The green water footprint

The green-WF was quantified as the net use of soil-water storage, and this was modelled as the difference between outflows and inflows of the soil-water system, as given in the Eq.7.2 (Herath et al., 2013).

$$WF_{Green} = \frac{10\{(T+E+D+R) - (P-P_i)\}}{Y} \qquad \textbf{(Eq. 7.2)}$$

where, WF_{Green} is the net green-water consumption [L/kg of potatoes] from soil-moisture store; *T* is transpiration from the vegetation [mm/y]; *E* is the evaporation from soil [mm/y]; *P* is the precipitation [mm/y]; and P_i is precipitation intercepted [mm/y] by the canopy. The factor 10 is for conversion of units.

The hydrological components were simulated, by using the SPASMO model to calculate the green-WF. The measured green-WF is the difference of the average TDR soil water content at the beginning of the season, and at the end of the season.

The grey water footprint

Following the definition of Hoekstra et al. (2011), the grey-WF was quantified as the volume of freshwater that is required to 'dilute' a pollutant, so that its concentration meets the prevailing water quality standard:

$$WF_{Grey} = [L/(C_m - C_n)]/Y \qquad \textbf{(Eq. 7.3)}$$

Here, WF_{Grey} is the freshwater required [L/kg of potatoes] to dilute the runoff and leachate down to an accepted water quality standard; *L* is the net-load of pollutants from the system [mg/ha]; C_n is the natural concentration [mg/L] of the pollutant in the receiving water body if there were no human intervention (Hoekstra et al., 2011); and C_m is the maximum acceptable concentration [mg/L] for the pollutant given by the prevailing water quality standard.

At the cropping stage, NO_3-N was considered as the dominant pollutant. We used the same principle to assess the impact of the packhouse phase, and for the input sourced from the background system. For packaging materials, such as the plastic and wooden boxes used in the packhouse, we used reported WF values from the literature (Herath et al., 2013; Li and Nwokoli, 2011).

7.3.2 Seed potato production

The seed potatoes are sourced externally, having already gone through a series of multiplication cycles (Fig. 7.1). For this system, the seed potato production process was undertaken in the Canterbury region. Therefore, all data related to crop water use and the management practices were collected from a major seed potato producing company. The process starts with tissue cultured plantlets, through glasshouse multiplication, and then another four generations of multiplication in the field. The crop water use and soil water dynamics of the field multiplication process was modelled, by using SPASMO and by considering local Canterbury weather and local soil information. The WF of G_2, the glasshouse phase, was quantified using data from direct and indirect water use through inputs, such as electricity that were sourced from the producers.

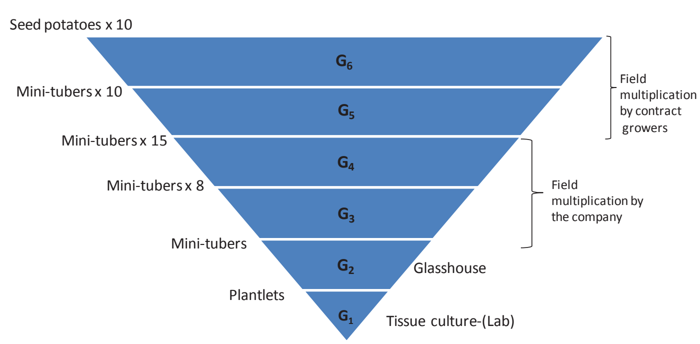

Figure 7.1 Seed potato production process starting from tissued-cultured plantlets through to seed potatoes used in cultivation of ware potatoes. Seed potatoes undergo five generations (G_{1-5}) of multiplication at different rates in each generation (left).

The WF of each generation of seed potato production was quantified by considering the WF from the previous generation divided by the multiplication rate and then adding the WF of that generation. The WF assessment for the seed potatoes was started from G_2, the glasshouse phase. The WF of the G_1 of the tissue culture phase was ignored, since the water use was considered to be minimal.

7.3.3 Packhouse phase

Direct water use and waste-water disposal information was collected from a large-scale packhouse in the Manawatu region. The direct water use for washing and cleaning was obtained from a survey, direct interviews and observations. The indirect water consumption through electricity and fuel was calculated, by following the method used by Herath et al.,(2013). Except for the electricity, water use impacts from the background were based on consumptive use. In regards to the packaging materials, the blue, green and grey WFs of wood and paper were assessed, following Herath et al., (2013), and for the WF of plastic we used the results from Li and Nwokoli (2011).

7.3.4 Improvement options to reduce the water footprint

Since this production system does not use blue water for irrigation, the main focus was to explore the improvement options for the grey-WF, in order to reduce the impact of nitrate leaching on the groundwater resources. It is evident from the measurements that the early application of N fertiliser, at planting, tends to lead to leaching of nitrogen at the beginning of the season, when it is often wet, and when plant demand is very low (Chapter 6). The SPASMO model was used to simulate nitrate leaching both for the 2011/12 growing season and for an average year, with four potential fertiliser application scenarios, including split fertiliser applications (Section 7.3.5). The grey WF was predicted, together with the NO_3-N concentration of the drainage, and the annual NO_3-N load leaving the root zone.

7.4 Results and Discussion

7.4.1 The crop production phase: water use and impacts

For the cultivation phase of ware potatoes, water use and its impacts on blue and green water resources are discussed, by considering both the measurements and mechanistic modelling.

7.4.1.1 Green water footprint: the net use of soil-stored water

The measurements of soil water content showed the annual change in soil water content was negligible since, in winter, the profile soil moisture content returned to field capacity. The soil water contents, simulated by the SPASMO mode,l showed a good agreement with the field measurements using TDR (Fig 7.2). The model simulations also showed that the net change in soil water content was indeed negligible. Although there were fluctuations with the intermittent rainfall, the soil-water store has been replenished. Therefore, the impact of green water consumption, by the crop, on the green water resource is insignificant. In other words, the green-WF is zero.

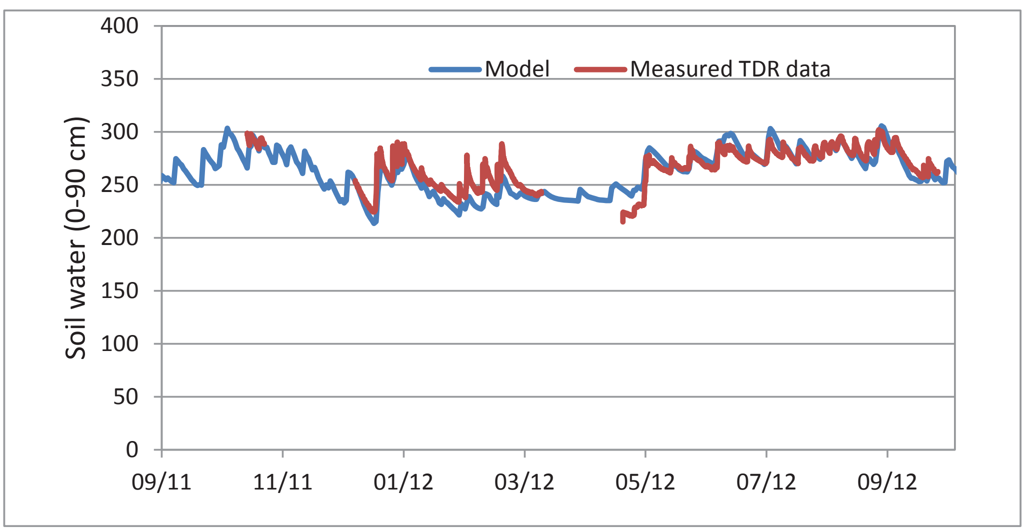

Figure. 7.2 Dynamics of soil water stored in the top 90 cm of the soil profile in Manawatu sandy loam soil (Dystric Fluventic Eutrochrept) under ware-potato cultivation (2011/2012 season), as measured using the mean of eight Time Domain Reflectometry (TDR) probes (red), together with the prediction using SPASMO model (blue). Irrigation was not used.

7.4.1.2 The blue water footprint: the net use of groundwater resource

The measurements of drainage using fluxmeters were exceptionally high during the 2011/12 season (Fig 7.3). In addition, the number of tension fluxmeters used was not sufficient to capture the high variability and complexity of water fluxes in the field, as discussed in Chapter 7. The local infiltration rate has been significantly affected by various factors, such as patchiness in the local hydrophilic and hydrophobic nature of soil, especially after cultivation and mounding and the local microtopography, plus the larger-scale relief of the ridges and furrows.

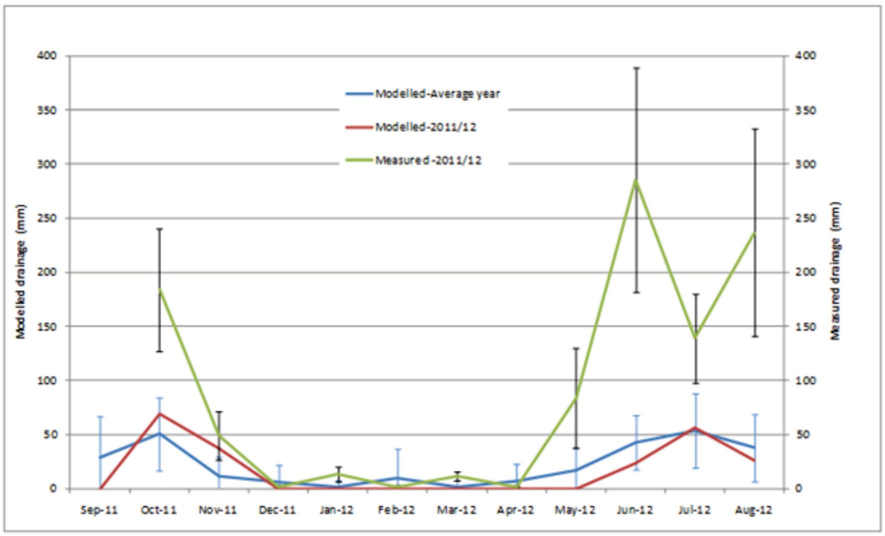

Figure 7.3 Monthly drainage over a year modelled for an average year (blue) and the year 2011/2012 (red), compared to the measured drainage (green). Error bars of measurements are standard errors of mean of measurements from six fluxmeters, while bars of the average year (blue line) are the standard deviation of monthly averages over 40 years (1972-2012).

The disturbance of the soil during the fluxmeters installation might have influenced the infiltration rate during the first couple of drainage events. Furthermore, wide cracks were observed in the soil surface (Fig 7.4), especially during tuber bulking, which would cause preferential flow into these connected macropores. Furthermore, the canopy would act as a 'reverse umbrella', thus rapidly directing the stem-flow directly into the surface vented orifices of the macrospores created by the cracks (Fig 7.4). This could well be a major contribution factor to the high drainage rates observed in the measurements. In addition, the redistribution and convergence of this local runoff into surface depressions, which we observed over the fluxmeters, would have contributed to the higher drainage volumes recorded by these devices. Nonetheless, the model predictions show a similar trend, compared to the measurements for both the average year and the year of study.

The first three months of October, November and December of the 2011/12 season were wetter (rainfall total 388 mm) than the sum of the average rainfall (250 mm) during these months. This led to a higher measured and modelled drainage for that season. Therefore, given the uncertainty in the measured values of 2011/12, the modelled average drainage of 245mm/y, was used together with the predicted yield of 44 tonne/ha, in order to calculate the blue-WF using the Eq. 1. This was negative for the crop-production phase, because no groundwater was extracted for the cultivation. The seasonal drainage contributed to the recharge of the groundwater resource at the rate of 72 L/kg of potatoes. Therefore, the blue-WF is -72 L/kg, which indicates the potato cultivation phase has no deleterious impact on the quantity of the groundwater resource in the region.

Figure 7.4 Images showing the surface-vented cracks, connected to the plant stems, in the soil just above one of the fluxmeters under the ridge at 12 weeks after planting.

The blue-WF from the background system, as a result of the use of inputs such as pesticides, fuel, and fertiliser, was also considered. This was estimated to be just 2.9 L/kg, and it was dominated by a water footprint arising, due to fuel consumption.

7.4.1.3 The grey water footprint: the nitrate concentrations and loads

By using Eq. 7.3, the grey WF was calculated as 55.9 L/kg, by considering NO_3-N as the pollutant. For the C_m, which is the maximum acceptable concentration, we used the

New Zealand drinking water standard of 11.3mg NO_3-N/L (MoH, 2008). The natural concentration, C_n , is the NO_3-N concentration in the receiving water body, if there had been no human intervention (Hoekstra et al., 2011). This was assumed to be zero, considering the NO_3-N concentration found to be in the groundwater of this region (Daughney and Randall, 2009). The use of the absolute value of the grey WF, to understand the impact on water quality, is not straightforward (Herath et al., 2013), while it might be useful for inter-product comparisons. Given this lack of grey-WF utility, the nitrate concentrations in the drainage and the nitrate loading to groundwater were simply used to assess the impact on water quality. Figure 7.5 shows predicted nitrate leaching and the timing of the fertiliser application, together with the field measurements. The measured concentrations are somewhat variable; however, the modelled and measured average concentrations show reasonable agreement. The measured average NO_3-N concentration in the drainage, at 60cm below the surface, was 12.6 mg/L, while the model predicted concentration for an average year was 11.3 mg/L. This concentration is at the New Zealand drinking water quality standard (MoH, 2008). This implies that the current practice here is just at the limit of an impact on the quality of underlying water resources.

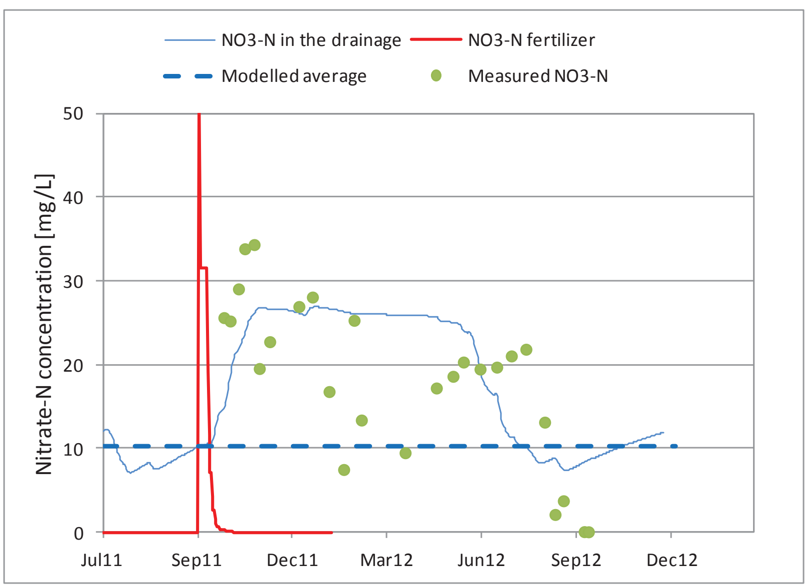

Figure 7.5 Modelled NO_3-N concentration in the drainage at 60cm for the long-term average (dashed line), and predicted concentration over the season 2011/12 (blue line), together with the fertiliser application spike (red line). Measured concentrations in the drainage from 6 fluxmeters measured over the 2011/12 growing season are shown as green dots.

The predicted annual average NO_3-N load was 27.8 kg/ha, with a high monthly variability. Due to the exceptionally high rainfall during the first three months of the 2011/12 season, the modelled nitrate loading during this period is predicted to be higher than the average year (Fig. 7.6)

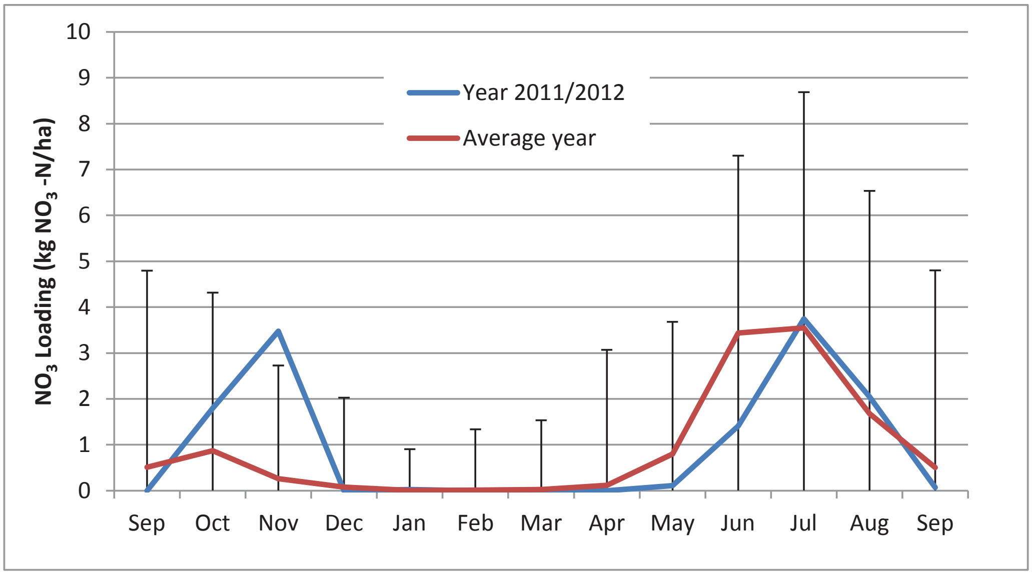

Figure 7.6 Modelled monthly NO_3-N load for study year 2011/12, and for average year computed from the record of 40 years. Bars indicate monthly standard error of the mean.

7.4.2 Seed potato production

The major water impact of the glasshouse phase of seed-potato production is from electricity consumption, followed by direct water use for the system. The mini-tubers from the glasshouse phase were calculated to have a blue-WF of 247.8 L/kg (Table 7.1).The grey-WF of the glasshouse phase was not considered, since the discharge water quality information was not available. Furthermore, any discharge would need a resource consent that met local water quality standards. The blue-WF of the field multiplication process was calculated to be 7.2 L/kg, and the grey-WF was 0.37 L/kg. The seed potatoes in this study were sown at the rate of 2.7 tonne/ha. After consideration of multiplication across generations of seed potato production, the contribution to the blue-WF from seed potatoes was estimated to be just 0.6 L/kg of potatoes, and the grey-WF was only 0.02L/kg. The green-WF of this process was, as before, considered to be negligible.

Table 7.1 Blue and grey water footprints of different generations of seed potato production process. The symbol G_2 is for the glasshouse, while G_3- G_6 is field multiplications.

Generation	Blue WF(L/kg of seed potato)	Grey WF (L/kg of seed potatoes)
G_2	247.8	-
G_3	38.23	0.37
G_4	9.78	0.39
G_5	8.21	0.41
G_6	8.05	0.41

7.4.3 Packhouse phase

In the packhouse, harvested potatoes are washed or cleaned, and packed for different destinations. The packaging materials include large wooden pallets for the export market, together with plastic bins and crates, plus paper bags for the local market. The blue and grey WFs of the packaging materials were 0.6 and 5.3 L/kg, respectively. The direct water use in the packhouse is mainly for washing the potatoes. In the assessment of water quality, it was found that turbidity was the main concern. However, the system is strongly regulated by local government authorities, and the wastewater is discharged only after going through two settling ponds. The discharge of the clear waste-water is to a small stream which leads to the main river. Evaporative loss from the settling ponds was considered as blue water use. Chemical analyses of the discharged water showed no noxious chemicals leaving the system and therefore, the grey WF from direct water use in the packhouse is considered negligible.

Table 7.2 Contribution from different inputs to total blue water footprint of packhouse phase

Input	Blue water footprint(L/kg)
Direct water use in the pack house	0.25
Packaging materials	0.60
Electricity use	0.11
Other fuel use	0.01
Total	**0.97**

7.5 Summary results of the analysis

The crop production stage is the major contributor to the impact of the fresh ware-potato production chain on water resources, both in terms of quantity and quality (Table 7.3). Due to the groundwater recharge during the crop production phase, the overall blue water footprint of the potato production system is negative, which is a positive indication for the state of the groundwater resource. However, from a broader perspective of environmental impacts, this needs to be assessed. Is this recharge rate sufficient to maintain environmental flow requirements, and also ensure the continued flow of ecosystem services from these water resources?

Table 7.3 Impacts of water use in potato production on quantity (blue-WF) and quality (grey-WF) of water resources in the Manawatu region of New Zealand (Units: L/kg of potato at the packhouse gate)

System		Impact on water quantity (Blue-WF) L/kg	Impact on water quality (Grey-WF) L/kg
Crop production	Foreground	-72	55.9
	Background	3.5	0.2
Packhouse	Foreground	1.27	-
	Background	0.72	5.3
Total		**-66.5**	**61.4**

The major impact of the potato supply chain on water quality is also from the crop production phase, as indicated by the grey WF (Table 7.3). The analysis of nitrate loading and concentration, within current management practices, shows that there is some impact on water quality with the leachate being, on average, at the drinking water standard. Therefore, it is worthwhile investigating improvement options, in order to reduce the impact of the current system on the quality of water resources.

7.6 Improvement options to reduce the impact on water quality

Three potential fertiliser-application scenarios were considered, together with the current practice. In regards to all the scenarios, the amount of fertiliser used was 120 kg N/ha, of which 68 kg/ha was in the form of NH_4^+, and 52 kg/ha was NO_3^-.

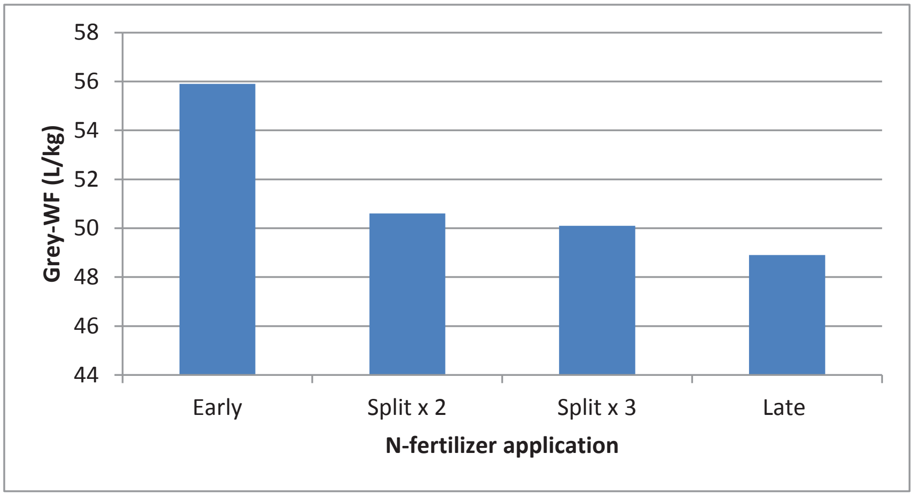

Figure 7.7 Effect of different fertiliser scenarios on grey-WF. For all scenarios, the amount of fertiliser used was 120kg-N/ha. Early application is at the time of planting: half Splitx2 is applied at planting and the remainder applied at ridging, 28 days after planting (DAP). Splitx3 is each 1/3 of fertiliser applied at planting, 28 days, and at mounding at 55 DAP. Late application is once, at 55 DAP.

The modelling predicts that the split applications and late application of N fertiliser has decreased the grey WF (Fig. 7.6). The NO_3^--N concentrations and loads are significantly lower compared to the current practice of applying all the fertiliser at the time of planting (Fig. 7.7). However, there was no significant difference between Split x2 and Splitx3. With the split application, and the late application of N fertiliser, the predicted average NO_3-N concentrations were less than the New Zealand drinking water standard of 11.3 mg NO_3-N /L. The average loading rate of NO_3-N was 27.8 kg/ha/y under the current practice. It was reduced to 25.6, 25.2 and 24.3 kg NO_3-N/ha/y with the Early, Splitx2, Splitx3 and Late applications, respectively. However, from the growers' point of view, the effect of fertiliser application on yield is also important. The model predictions do not show any significant difference in yields, which were 44, 44.7, 44.5 and 43.9 tonne/ha for the Early, Splitx2, Splitx3 and Late applications, respectively. Therefore, managing fertiliser with split applications is a good and viable option, in order to reduce the impact on the water quality of groundwater resources.

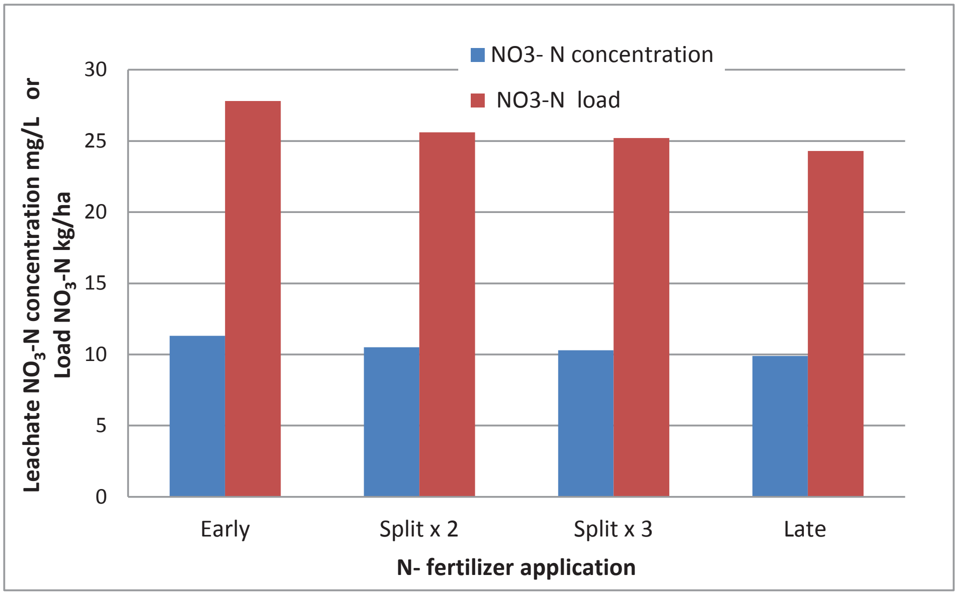

Figure 7.8 Effect of different fertiliser application practices on loads and concentrations of nitrate leaving the root zone of ware potatoes. For all scenarios, the amount of fertiliser used was 120kg-N/ha. Early application is at the time of planting: half Splitx2 is applied at planting and remainder at ridging, 28 days after planting (DAP). Splitx3 is each 1/3 of fertiliser applied at planting, 28 DAP, and mounding at 55 DAP. Late application is once, at 55 DAP.

7.7 Conclusions

The impact of the fresh ware-potato production chain on the quality and quantity of water resources in the Manawatu region was assessed, by using the hydrological water balance method of water footprinting. The measurements and modelling results indicate that this potato production system has no deleterious impact on water resources, in terms of quantity. This was indicated by the negative blue-WF (-67 L/kg). However, current practices do suggest some impacts on groundwater quality, the local assimilation capacity of groundwater system notwithstanding. The average concentration of nitrate in the drainage below the root zone was predicted at 11.3 mg-NO_3/L, which is just at the drinking water standard. The modelling of different fertiliser application scenarios indicated that splitting N rate into two or three applications, or having a late application at 55 days after planting, would reduce the leaching of nitrate, shrink the grey-WF and lower the nitrate loading, without compromising the yield. These predictions are for a potato-cover crop rotation, as is common in New Zealand. Therefore, there are options to reduce the grey-WF, and thereby lower the impact on groundwater water quality in the region.

The green-WF, the impact of potato production on soil water store was negligible, according to our measurement, in addition to our modelling results. This result is because, every winter, the soil-water store returns to field capacity. The impact of the packhouse phase and the background system, including seed potato production, was small and insignificant, being just 2 L/kg for the blue WF, and 5.3 L/kg for the grey WF.

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CHAPTER 8

Overall Summary and Future Research Imperatives

8.1 Summary

Freshwater is a fundamental natural resource that is necessary for human life, as well as for the maintenance of environmental integrity. Agriculture has significantly affected the chemical and biological properties of receiving water bodies and modified natural flows. Growing awareness of threats to the world's freshwater resources has driven the development of methods to measure and report the impacts arising from water use. The concept of water footprinting has gained momentum, as a metric that quantifies the environmental impacts related to water use. Water footprinting is also considered to have the potential to underpin the development of eco-labels or environmental product declarations, which can communicate, to stakeholders, information on the environmental performance of production systems. The multifarious issues surrounding the sustainability of water use in New Zealand are, therefore, of growing importance to primary industries, not just from a strictly environmental point of view, but also for product marketing (Sinha and Akoorie, 2010) under the 'clean green' label. The research presented in this thesis attempts to quantify the water footprint (WF) of two economically important agricultural products in New Zealand. These are grape-wine, the top horticultural export earner; and potatoes, the largest vegetable crop in terms of area under cultivation in New Zealand.

At a time when international standards for water footprinting are being developed, this thesis evaluates existing WF protocols and develops a hydrological water-balance method for water footprinting. This method accounts for all the inflows and outflows of both the liquid and vapour forms of water within a system boundary.

Electricity is a major input into the primary-production supply chain and, because hydropower is a major component of New Zealand's electricity mix, it was first decided to determine the WF of hydro-electricity.

The WF of New Zealand hydroelectricity has been assessed in Chapter 3, by using three different methods by considering all the major hydroelectric power plants which account for more than 95% of hydropower generated in the country. The first (WF_1) and second (WF_2) methods only consider the consumptive water use of the hydro-electrical generation system, while the third method (WF_3) introduced in this thesis considers the net hydrological water balance, including rainfall. Irrespective of the method, the WF of New Zealand's hydroelectricity was found to be much smaller than the commonly cited international value of 22 m^3/GJ. Depending on the method, the national WF for hydropower generation ranged from 1.55 m^3/GJ (WF_3) to 6.05 m^3/GJ (WF_1). Precipitation is the key driver for replenishing water resources. The water balance approach provides meaningful information that helps in understanding the differences in the WF for different locations, which are diverse in terms of water resource availability. This study also highlights the effects of local climatic differences and the structural specifics of various hydroelectricity schemes on the WF of hydropower. The large variation in the WF of hydropower across New Zealand illustrates the inappropriateness of using global average values. Local values, calculated by using a hydrologically rational method, must be used.

The locally assessed WF-value of hydropower in New Zealand has then been used in the assessments for quantifying the WFs of wine and potato production in New Zealand. The WF associated with the life cycle of wine-grape production was assessed in Chapter 4, for two regions in New Zealand: Marlborough and Gisborne. This assessment considered some 12,600 ha under grapes: and 36 wineries across both the regions. The vineyards were on 29 different soil types spread across 19 climatic regions. The functional unit (FU) was a 750-mL bottle of wine at the winery gate. The WF was assessed, by using the full water-balance, which takes into account all hydrological inflows and outflows in the system and identifies two main resources: soil water (the green water resource) and groundwater (the blue water resource). The net uses of these two resources were quantified as the green and blue water WFs. There was a large variability in the blue-WF of grape cultivation across the different vineyards, even within a region. At the grape-growing stage, the average blue-WFs were -81 L/FU and -415 L/FU for Marlborough and

Gisborne, respectively (Table 8.1). The negative WF value indicates that the groundwater resources are being recharged on an annual timescale in these regions. The green-WFs were found to be negligible because the soils are returned to field capacity by rainfall every year, at some stage during winter. The average grey-WFs, that is, the water required to dilute the nitrate nitrogen (NO_3-N) leached in the vineyard phase, were 40 and 188L/FU for Marlborough and Gisborne, respectively (Table 8.1). However, the average concentration of NO_3-N in the leachate was much smaller than the drinking water standard of 11.3 mg/L. On average, they were just 5.01 mg/L and 8.7 mg/L for Marlborough and Gisborne, respectively (Table 8.1). The WFs of the winery phase were found to be very small, compared to those of the vineyard stage. The spatial variability found here highlights the importance of considering WFs and related water issues at the local scale. Locale is the essence of 'terroir' for wine.

Table 8.1 Summary results of the water footprint analyses of wine and potato. The results are presented for the cultivation phase, in terms of kilogram of grapes and potatoes.

Product	Region	Impact on the quantity of sourced water resources		Impact on the water quality of receiving water resources		
		Green WF from the hyrological method of this thesis	**Blue WF** from the hyrological method of this thesis (L/kg)	**Grey WF** of Hoekstra et al., 2011 (L/kg)	Average NO_3-N **load** (kg NO_3-N/ha/y)	Average NO_3-N **concentration** (mg NO_3-N/L)
Wine-grape	Marlborough	Negligible (<0.1 L/kg)	- 81	40	4.9	5.0
	Gisborne	Negligible (<0.1 L/kg)	- 415	188	29.3	8.7
Potatoes	Manawatu	Negligible (<0.1 L/kg)	- 72	56	27.8	11.3

The hydrological water-balance method developed in Chapters 3 and 4 was then evaluated, together with three other existing methods, in order to assess the WFs of wine produced in the Marlborough and Gisborne regions (Chapter 5). The three other water-footprinting methods were: the consumptive water use method of the Water Footprint Network (WFN); and two life cycle assessment (LCA) based methods. The latter were the stress-weighted WF, and the method of Canals et al. (2009) that considers freshwater ecosystem impacts and freshwater depletion. The outcomes of these methods were evaluated for their ability to indicate the local impacts on water resources, and their usefulness to key stakeholders.

The WF method of the WFN quantifies blue and green-water consumption, but further information on water resource availability is required, in order to assess the impacts of consumption on local water resources. The grey-WF indicates the impact on water quality, and comparisons can be made between products, but the absolute values are less meaningful. The LCA-based methods of the stress-weighted WF, and the freshwater ecosystem impact and freshwater depletion, do certainly indicate impacts to some extent. However, their ability to show local impacts is limited, due to the constraints associated with their characterisation factors. The WFN method and the two LCA methods are based on consumptive water use. Hydrologists have, however, shown that the water consumed through evapotranspiration soon returns as precipitation, within a reasonably short local scale. Consequently, any water footprinting based on water consumption alone is limited, in terms of assessing the local impact of primary production on water resources. Furthermore, the applicability of the outcomes of these three methods, to growers, consumers and resource regulators, is not straightforward. In contrast, the hydrological water-balance method indicates the impact on water quantity, as a volumetric measure and therefore, it can be understood by the non-technical community. In the case of growers, it provides a sensible measure of the impact of their production, as well as useful information for setting measurable targets to reduce the WF. It can also help resource regulators to manage water resources, by matching water demands to water availability, or the rate of their replenishment.

The principles of the hydrological water balance method were then used in Chapter 6 to measure, under field conditions, the WF of a crop of fresh potatoes in the Manawatu region of New Zealand. The large spatial and temporal variability in the field measurements presented here proved to be very challenging. It would appear that the number of measurement devices required to obtain reliable values would not be practicable. Potatoes are typically grown in this region in rotation, with a cover crop or pasture. Therefore, it was thought to be more accurate to consider the whole crop sequence through mechanistic modelling. Furthermore, modelling is the most cost-effective means to capture the effects of the year-to-year variability in the weather, thereby quantifying a representative WF. Nonetheless, model validation by measurements will always be an imperative.

Finally, the WF of a kilogram of potatoes at the packhouse gate in the Manawatu region of New Zealand was quantified in Chapter 7, by using this mechanistic modelling, which was robustly supported by the field measurements. The green-WF was again found to be negligible. The blue-WF was -72 L/kg (Table 8.1), thus indicating that groundwater is recharged by 72 L for every kilogram of potatoes harvested in the region. This suggests that the modelled rain-fed potato production system has no deleterious impacts on the blue water quantity in the region.

The grey-WF was 61 L/kg, of which 56 L/kg is from the cropping stage. The impacts of the packhouse phase and the background system, including seed-potato production on the two regions' water resources, were found to be small. The use of the absolute value of the grey-WF, in order to understand the impact on water quality, is not straightforward. However, average concentrations and loading of NO_3-N, from the cultivation phase, indicate that the current practices are having some impacts on water quality. Under the current practice of applying all the nitrogen fertiliser at planting, the average concentration of nitrate leaching below the root zone was at 11.3 mg/L (Table 8.1). This is just at the drinking water standard. The average loading rate of NO_3-N was at 27.8 kg/ha/y.

The potential to reduce the grey WF of potato production was investigated, by modelling three different fertiliser-application scenarios related to split applications and timings. They were: two splits, three splits and a late application at 55 days after planting (see Chapter 7 for a fuller description). All three scenarios reduced the NO_3-N concentrations and loads from the cultivation phase. The NO_3-N concentrations were simulated to be 10.5, 10.3 and 9.5 mg-NO_3/L, and the loading rates were 25.6, 25.2 and 24.3kg /ha/y, respectively.

The findings of this thesis highlight the importance of considering the full hydrology in water footprinting and to understand the local impacts on water resources. Furthermore, it is important to derive improvement options that will reduce water-related impacts. The content of this thesis also emphasises the synergy found between measurements and modelling. It details how measurements are critically important to modelling schema. The variability in water use and the impacts found in this study indicate the importance of considering water related assessments at the local scale.

The analysis presented in this thesis reveals that, in terms of the impacts on water quantity, most of New Zealand's agricultural products are not net consumers of water. Indeed, because of New Zealand's humid climate, the groundwater under New Zealand's agricultural fields is, in net, recharged. However, contamination of water resources, through agrichemicals, needs consideration, in order to reduce the impact of agriculture on the water quality of receiving bodies. The hydrological water-balance method developed in this thesis is useful for water management at the orchard scale; and also for policy formulation concerning both the quantity and quality of receiving water resources.

8.2 Implications and suggestions for future work

Although water footprinting has already been used as a metric for quantifying water use and indicating impacts, there are still a number of methodological issues around current water footprinting protocols. These have been highlighted in Chapters 3 to 7. In particular, the challenges that need to be resolved are around the following:

- Lack of agreement between the different water footprinting protocols. A consistency of terminology is an urgent need.
- Regardless of the methodological differences, there are constraints, which are due to a lack of geographically specific inventories. This has been shown in Chapter 4 and therefore, the development of accurate inventories is vital.
- It is also important that the various WF methods capture spatial and temporal variability. This is especially important for agricultural production systems with their inherent spatial variability. This is a result of differences in soils across diverse landscapes, together with the temporal variability of weather. This variability could be better addressed through the combined strength of field measurements and mechanistic modelling, in order that the water use values and impact assessments are representative and meaningful.
- This research, together with most other reported studies, has only considered impacts directly related to water-use, and not the emissions that do not have an immediate impact on water resources. Impacts on water resources exist across different timescales. For example, the contribution of SO_2 emissions from any production system might have long-term impacts on water resources. Moreover, carbon emissions might affect water resources, both quantity and quality, through climate change. Further improvements in methodologies are needed to assess these impacts.
- All methods to date lack rational protocols for assessing the impact on the water quality of receiving water resources. This is an important aspect. It clearly needs further development, since the utility of available water quantity depends on its quality.
- This thesis has provided important analyses and signposts as to what is needed, in order that the world's water resources are best used to sustain food production and also to maintain environmental integrity.

APPENDIX – A

Assessing Freshwater Scarcity in New Zealand

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A.1 Introduction

"Freshwater problems" are among today's most acute and complex scientific and technical problems. They increasingly reach beyond regional and national borders and are becoming global in nature. Decreasing availability, declining quality and growing demand for water are creating significant challenges to businesses and investors who have taken clean reliable and inexpensive water for granted. Although water is apparently plentiful on some global and national averages, there are enormous disparities between regions. Due to the high spatial and temporal variability of freshwater availability and demand, regional assessment of water scarcity, that is demand in relation to availability, is of importance in understanding the impacts of water use. Therefore, this study aims to assess freshwater status of New Zealand on regional scale using three key indicators; Falkenmark Index (Falkenmark, 1986) Water Scarcity Index (Hoekstra and Hung, 2002) and Water Stress Indicator Smakhtin *et al.* (2004).

A.2 Data and Methodology

Freshwater scarcity was assessed using three indicators: Falkenmark Index (per capita water availability), Water Scarcity Index and Water Stress Indicator following their definitions. Water resource availability was determined using the physical stock account for water to the period of 1995-2005 as published by Statistics New Zealand (StatisticsNZ, 2010) and data on water use and allocations for 2002-2005 were extracted from a snapshot of water allocation in New Zealand by Ministry for the Environment

(MfE, 2006). In this assessment we found that actual water-use data were not available for most of the regions. Therefore, when water-use data were not available, it was assumed that 70% of the allocated water was being used.

A.3 Results and Discussion

A.3.1 Water Availability per Capita/ Falkenmark Index

This is an indication of water resource availability per capita per year in the country or the region considered. This is also known as the Falkenmark Index (Falkenmark, 1986). A threshold value of 1,700 m^3 per capita per yr has been identified and the countries of which renewable water supplies cannot sustain this figure are said to experience water stress. This concept is also known as water resource per capita –WRPC (Canals et al., 2009). This concept is used here to assess regional water availability and stress in New Zealand.

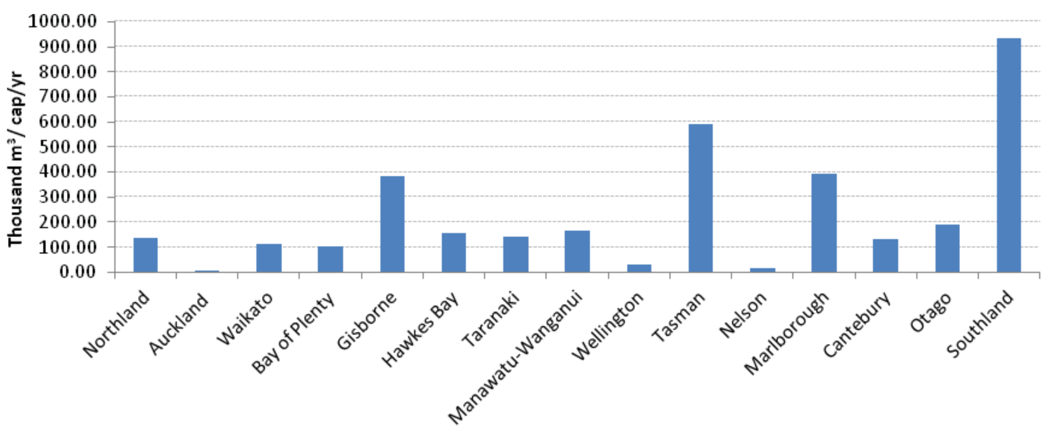

Figure A.1 The Falkenmark Index (thousand m3/capita/year) for the regions in new Zealand

According to this indicator, all the regions in New Zealand are well above the threshold value. The Auckland region shows the lowest value of 5160 m^3 per capita per year, but well over the threshold value of 1700 m^3 per capita per year. This indicator has been adopted as the standard indicator of water scarcity. However, it does not take into account water demand, or actual use. A simple threshold does not reflect variations in the demand for water in different countries or regions due to lifestyle and economic factors. Even though New Zealand is well endowed with fresh water, there are evidences of its high levels of water use, for instance; New Zealanders use 653 litres of water a day per capita for domestic purposes. This rates as “excess use”(Lawrence et al., 2002).

A.3.2 Water Scarcity Index

This index has widely been used to assess water stress as the ratio of water use (withdrawal) to total water available (Alcamo et al., 2003). According to this index, the severity of water scarcity has been ranked as follows: WSI<0.1 - no water stress; 0.1 ≤ WSI<0.2 - low water stress; 0.2 ≤ WSI<0.4 - moderate water stress; WSI ≥ 0.4 - high water stress; and WSI>0.8 - very high water stress (Smakhtin et al., 2004). A similar concept is also known as water use per resource -WUPR (Canals et al., 2009). Table 1 shows the estimated Water Scarcity Index for regions in New Zealand.

Table A.1 The Water Scarcity Index and Water Stress Indicator for the regions of New Zealand

Region	Precipitation (Mm^3/y)*	Inflow from other regions (Mm^3/y)*	Total Inflow (Mm^3/y)*	Water use (Mm^3/y)	**Water Scarcity Index**	Available water for human use (Mm^3/y)	**Water Stress Indicator**
Northland	20227	0	20227	79.8	**0.004**	14158.9	**0.006**
Auckland	6901	0	6901	127.93	**0.019**	4830.7	**0.026**
Waikato	42054	1088	43143	467.60	**0.011**	30200.1	**0.015**
Bay of Plenty	24293	2074	26366	307.09	**0.012**	18456.2	**0.017**
Gisborne	15966	1273	17239	207.48	**0.012**	12067.3	**0.017**
Hawke’s Bay	21548	1692	23240	310.17	**0.013**	16268	**0.019**
Taranaki	14774	0	14774	73.92	**0.005**	10341.8	**0.007**
Manawatu-Wanganui	37495	21	37516	138.60	**0.004**	26261.2	**0.005**
Wellington	13962	40	14003	581.07	**0.041**	9802.1	**0.059**
Tasman	26194	1294	27488	104.23	**0.004**	19241.6	**0.005**
Nelson	693	0	693	20.44	**0.029**	485.1	**0.042**
Marlborough	15631	1112	16743	130.34	**0.008**	11720.1	**0.011**
West Coast	127531	7160	134691	191.03	**0.001**	94283.7	**0.002**
Cantebury	67011	2082	69093	2811.06	**0.041**	48365.1	**0.058**
Otago	37000	85	37086	1224.93	**0.033**	25960.2	**0.047**
Southland	86830	160	86991	116.55	**0.001**	60893.7	**0.002**

*data were extracted from statistics of New Zealand : Mm^3/y-million cubic meters per year

According to the Water Scarcity Index (Table A.1), none of the regions in New Zealand are freshwater stressed. This indicator is more meaningful than the Falkenmark index as it actually accounts for water use. The Southland region shows the lowest value of 0.001, while Canterbury and Wellington having the highest values of 0.041. All the regions fall into the category of 'no water stress'.

A.3.3 Water Stress Indicator

An alternative indicator for environmental water stress has been explored by Smakhtin *et al.* (2004). It is the ratio of water withdrawal (WU) to the actual water resources available for human use. This is derived by subtracting environmental water requirement (EWR) from the total available water resources (WR): $WSI^2 = WU/(WR-EWR)$. This is considered as a more accurate indication of the water resources available for further human use by 'reserving' the necessary resources for ecosystem functioning (Canals et al., 2009).This concept was used initially for river basins by Smakhtin et al. (2004), and recently it has used to assess the impact of the external water footprint of the UK on the water resources of other countries (Chapagain and Orr, 2008). In that study, 30% of the total available water resources was considered as the EWR. Nonetheless, the quantification of EWR is location and time specific and is often highly debated. According to Smakhtin *et al.* (2004), the EWR to maintain a fair condition of freshwater ecosystems ranges globally from 20-50% of the mean annual river flow in a basin. We have used 30% of total inflow to the region as the EWR, and then estimated the Water Stress Indicator for different regions in New Zealand (Table A. 1). According to the Water Stress Indicator, environmental water scarcity is categorised as follows: $WSI^2 > 1$ – over exploited; $0.6 \leq WSI^2 < 1$ - heavily exploited; $0.3 \leq WSI^2 < 0.6$ - moderately exploited; and $WSI^2 < 0.3$ - slightly exploited. All the regions of New Zealand show a WSI of less than 0.3 (Table A.1), and therefore they fall into the slightly exploited category. The regional values range from 0.002 (Southland) to 0.059 (Wellington). This indicator captures vital hydrological aspects in terms of ecosystem health, and therefore it is more meaningful in understanding the impacts.

However, none of the indicators considered have taken into account the quality of water, which frequently governs its utility. This is an important aspect to be included in freshwater scarcity assessments.

A.4 Conclusions

New Zealand is well endowed with freshwater according to the estimated water scarcity indicators, although it does have very high levels of usage. Among the three indicators considered, Water Stress Indicator gives a better estimation of freshwater scarcity. The quality of the available water is however an important aspect yet to be included in scarcity indicators. If New Zealand wisely allocates its water resources and parsimoniously uses its plentiful water without compromising its quality, then New Zealand has enormous opportunities to achieve competitive marketing advantage through ecoverification of the water footprint of its products.

A.5 References

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APPENDIX- B

Publications by the candidate during the PhD programme

Refereed Research Articles in Scientific Journals

1. **Herath, I.,** Deurer, M., Horne, D., Singh, R., and Clothier, B., 2011. The water footprint of hydroelectricity: A methodological comparison from a case study in New Zealand. *Journal of Cleaner Production* 19: 1582-1589.

2. **Herath, I.,** Green, S., Singh, R., Horne, D., van der Zijpp, S., and Clothier, B. 2013. Water footprinting of agricultural products: a hydrological assessment for the water footprint of New Zealand's wines. *Journal of Cleaner Production*, 41: 232-243.

3. **Herath, I.,** Green, S., Horne, D., Singh, R., McLaren, S. and Clothier, B. 2013. Water footprinting of agricultural products: Evaluation of different protocols using a case study of New Zealand wine. *Journal of Cleaner Production*, 44: 159-167.

4. **Herath, I.,** Green, S., Horne, D., Singh, R. and Clothier, B. 2013. Measuring water and nutrient dynamics to quantify and reduce the water footprint of potato cultivation. *Agricultural Water management*, under review.

5. **Herath, I.,** Green, S., Horne, D., Singh, R. and Clothier, B. 2013. The water footprint of potato production: a hydrological assessment using measurement and modelling. *Agricultural Water management*, under review.

Papers in Conferences/Forums/Workshops

6. **Herath, I**., Clothier,B., Green, S., Horne, D. and Singh, R. & van den Dijssel, C. 2013. Is the grey-water footprint helpful for understanding the impact of primary production on water quality? In: ***Accurate and efficient use of nutrients on farms***. (Eds L.D. Currie and C L. Christensen). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 26. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand.

7. **Herath, I**., Clothier,B., Green, S., Horne, D. and Singh, R. 2012. Methodological challenges in agricultural product water footprinting: A case study of New Zealand wine. Proceedings of the Joint Australian and New Zealand Soil Science conference, Hobart, 2 - 7 December 2012.

8. **Herath, I.,** Clothier, B., Green, S., Deurer, M., Horne, D., Singh, R. and van der Zijpp, S. 2012. Variability in agricultural product water footprints: A case study of

New Zealand wine. Proceedings of the 2nd New Zealand Life Cycle Assessment Centre Conference 28th–29th Mar 2012. Auckland, New Zealand.

9. **Herath, I.,** Clothier, B., Green, S., Horne, D., Singh, R., Marsh, A., Buchanan, A. and Burgess, R. 2012. Measuring the Grey Water Footprint of Potatoes. In: Advanced Nutrient Management Gains from the Past - Goals for the Future. (Eds L.D. Currie and C.L. Christensen). Occasional Report No. 25. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand.

10. **Herath, I.,** Deurer, M., Horne, D., Singh, R. and Clothier, B. 2011.The water footprint of hydropower: a vital input for the water footprinting of New Zealand's agricultural products. Proceedings of the 2nd Annual Doctoral Students' Symposium. December 5, 2011. Massey University Palmerston North, New Zealand.
11. **Herath,I.,** Deurer, M. Clothier, B., Horne, D. and Singh, R. 2011. Water footprints of agricultural products: indicators of water sustainability. Proceedings of the 10th International Conference of East and Southeast Asia Federation of Soil Science Societies. October 10-13, 2011, Colombo, Sri Lanka.

12. **Herath,I.,** Clothier,B., Horne,D. and Singh, R. 2010. Assessing Freshwater scarcity in New Zealand. Life Cycle Management Centre Working paper Series 02/11(pp 21-26). ISSN (online) 2230-4282. Massey University Palmerston North, New Zealand.

13. **Herath, I. K.,** Clothier B., Horne D, 2010. Indices of the status of freshwater resources for impact analyses. Proceedings of 19th World Congress of Soil Science, Brisbane, Australia.
14. **Herath, I.K.,** Clothier B., Horne D, 2010. Referencing water footprints to indices of water resource status. In: Farming's future: Minimising footprints and maximizing margins. (Eds L.D. Currie and C.L. Christensen). Occasional Report No. 23. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand.

15. **Herath, I**., Clothier, B., Green, S., Horne, D. and Singh, R. & van den Dijssel, C. 2012. The water footprint of New Zealand wine: Evaluation of different water footprinting protocols. 1st New Zealand student symposium on Life cycle management 16th October 2012 Massey University. Palmerston North, New Zealand.

Articles in other Periodicals/Magazines/Reports

16. **Herath, I**., Green, S., Singh, R., Horne, D., van der Zijpp, S., and Clothier, B. 2013. Assessment of different water footprinting protocols using New Zealand's wine production. WISPAS- Water in the soil-plant atmosphere system. Issue 114. ISSN 1176-2292. The New Zealand Institute for Plant and Food Research.

17. **Herath, I.,** Green, S., Singh, R., Horne, D., van der Zijpp, S. and Clothier, B., A hydrological assessment for the water footprint of New Zealand's wines. 2012. WISPAS- Water in the soil-plant atmosphere system. Issue 113. ISSN 1176-2292. The New Zealand Institute for Plant and Food research.

18. **Herath, I.,** Clothier, B., Green, S., Horne, D., Singh, R., Marsh, A., Buchanan, A. and Burgess, R. Quantifying the grey-water footprint with potatoes. 2012. WISPAS- Water in the soil-plant atmosphere system. Issue 111. ISSN 1176-2292. The New Zealand Institute for Plant and Food research.

19. **Herath, I.,** Deurer, M., Clothier, B., Horne, D. and Singh, R. 2011. The water footprint of New Zealand's hydroelectricity. WISPAS- Water in the soil-plant atmosphere system. Issue 108. ISSN 1176-2292. The New Zealand Institute for Plant and Food research.
20. **Herath, I.,** Deurer, M., Clothier, B., Horne, D. and Singh, R. 2011. The water footprint of New Zealand's goods and services. Water New Zealand. Issue 169 (28-31).

21. **Herath, I.,** Clothier, B. Deurer, M. Horne, D. and Singh, R. 2011. A look at the water footprint of New Zealand's electricity. Irrigation New Zealand. Autumn 2011 (12-13). Irrigation New Zealand, Christchurch, New Zealand.

Co-authored publications

22. Clothier, B., **Herath**, **I**. Green, S., 2013. Reducing the Water Footprint of Irrigated Agriculture. In *sharing Irrigation Knowledge for Better Outcomes*. Irrigation Australia 2013 Regional Conference-Griffith. 28-30 May 2013, Australia.

23. Clothier, B., **Herath**, **I**. Green, S., 2013. Water footprint of Primary Production-treading lightly? Land Treatment Conference. 11 April 2013. Blenheim, New Zealand.

24. Clothier, B., Green,S., **Herath**, **I**. and Deurer, M. 2012. Measuring and Modelling the Water Footprint of Agricultural Products. 2012. The second International Conference on Hydropedology, 22-27 July 2012, Leipzig, Germany.

25. Clothier, B., Green,S., **Herath, I**., Périé E,. van den Dijssel C., Deurer, M. Mason, K., Dryden, G and Butcher, M. 2012 The Carbon and Water Footprints of Fruit Products. Acta Horticulturae.

26. Clothier, B., Green,S., **Herath**, **I**. Périé E. van den Dijssel C., Deurer, M. Mason, K., Dryden, G and Butcher, M. 2012. The Carbon and Water Footprints of Fruit Products. 7th International Symposium on Irrigation of Horticultural Crops. July, 16-20, 2012, Geisenheim, Germany.

27. Clothier, B., Deurer,M., Green,S., **Herath I**. and Mowat, A. 2012. The water footprinting of New Zealand's fruit: A hydrological methodology. In: Advanced Nutrient Management Gains from the Past - Goals for the Future. (Eds L.D. Currie and C.L. Christensen). Occasional Report No. 25. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand.

28. Clothier, B., Deurer,M., Green,S., **Herath I**. and Mowat, A. 2012. A hydrological approach to the water footprinting of New Zealand's fruit products. Proceedings of the 2nd New Zealand Life Cycle Assessment Centre Conference 28th–29th Mar 2012. Auckland, New Zealand.

29. Clothier,B., Green, S., Deurer, M., Mueller,K., **Herath, I.** Mackay, A. & Roygard, J. 2012. Linking local soil transport processes to catchment hydrology and policy options. Japan Geocience Union Meeting. Chiba City. Japan. 20-25 May 2012.

30. Clothier, B., Green, S., **Herath**,I., van den Dijssel,C., Hall, A., Dryden, G. and Butcher, M. 2012. Measuring & Modelling the Water Footprint of Fruit Production: Eco-verification & Adapting to Climate Change. 11th Symposium of plant-water relationship. Sevilla, Spain.

31. Clothier,B., Green, S.,Hall, A., **Herath, I**., van den Dijssel, C. and Périé, E. 2012. Soils, Water and Carbon: Sustaining Returns on Investment into Natural Capital. The XII International Conference for Agricultural Biotechnology (ABIC), 3rd September 2012 Rotorua, New Zealand.

32. Clothier B, Green S, **Herath I**, van den Dijssel C, Mason K, Dryden G. 2012. The water footprint of export apples from New Zealand. A report prepared for: Pipfruit New Zealand, Ref: PF11P12.10. Plant & Food Research Client Report No. 45431. Plant & Food Research Contract No. 27625. SPTS No. 7074

APPENDIX- C

Research related activities the candidate attended during the doctoral programme

	Description of event, date and location	Description of the tasks involved	Remarks/outcome (publications are referenced in Appendix B)
1.	A **workshop was organised** to discuss the research with people working in the wine industry in Marlborough region. This workshop was collaboratively organised by Massey University, Plant and Food Research and Marlborough District Council. 85 Budge Street, Blenheim, 1 February 2010.	Undertaking a presentation on '**Quantifying and Reducing the Water Footprint of Wine Production'**, which described the proposed research plan. Engaged in the discussion that followed the presentation and responded to the concerns and suggestions of industry personnel.	There was interest shown and suggestions offered by stakeholders for a comparative assessment of the water footprint of wine in Marlborough and Gisborne. This suggestion has been considered within the study.
2.	Annual **workshop** of the Fertilizer and Lime Research Centre, Massey University on '**Farming's future: Minimising footprints and maximizing margins'.** Massey University, 11-12 February, 2010.	Presented a paper on 'Referencing water footprints to indices of water resource status.'	Engaged in discussions related the water footprint and networking. **Publication no. 14**
3.	**19th World Congress of Soil Science** 1-6 August 2010. Brisbane, Australia.	Presented a paper on 'Indices of the status of freshwater resources for impact analyses'.	Engaged in discussions and networking. **Publication no. 13**

4.	**Confirmation of registration of PhD seminar** 27 August 2010 University premises.	Candidate was expected to make a presentation upon the completion of the first year of the PhD programme, on the progress and structural framework of the remainder of the study. This was evaluated by a panel nominated by the university that included an external examiner.	PhD registration was confirmed. Received outstanding comments on progress made and capability of candidate.
5.	Attended the **Central District Field days, as a resource person.** March 2011.	The role was to explain the concept of water footprint to a non-specialist audience. Featured the theme poster for the year at the Massey University stand. Answered public queries and responded to media interviews.	Obtained feedback on the research and generally about the water footprinting concept from the general public, industry related personnel and journalists. Article on the university web site, which can be accessed from: http://www.massey.ac.nz/massey/about massey/news/article.cfm?mnarticle=responsible-water-use-for-better-returns-17-03-2011
6.	Contested for the **Three Minute Thesis Competition** of doctoral candidates held at Massey University.	Contestants had to present their PhD research to a non-specialist audience, during a period of three minutes with a single PowerPoint slide and no animation.	Candidate was judged to be a finalist to represent the Manawatu Campus in the Massey University finals.
7.	Attended, as a resource person, the **Technical workshop on water footprinting,** which was organised by Life Cycle Management Centre, New Zealand. 18 August 2010, Wellington, New Zealand.	Presented a paper on 'Assessing water scarcity in New Zealand'.	Contributed to discussions with LCA experts, hydrologists and water resource managers. **Publication no. 12**
8.	**NZLCM Centre Operations Group Meeting** held at LCR Scion offices, Wellington, New Zealand on 10 November, 2010.	**Presented a paper** on Quantifying and reducing the water footprint of agricultural products in New Zealand.	Actively involved in discussions related to research and was able to receive feedback from the LCA experts.

9.	6th **World Water Forum.** 29-30 September 2011, Marseille, France.	Attended, as an **invited speaker,** for the session on Harmonize Energy and Water: **Presented a paper** on 'Quantitative Aspects of Water Use in Energy Production: The water footprint of hydropower'.	Actively participated in the discussions on methodological issues on water footprinting and related problems with water-energy nexus.
10.	The **10th International Conference of East and Southeast Asia Federation of Soil Science Societies.** 10-13 October, 2011, Colombo, Sri Lanka.	**Presented a paper** on 'Water footprints of agricultural products: indicators of water sustainability'.	Successfully answered questions following the presentation. Received outstanding comments on the method proposed in the study. **Publication no. 11**
11.	The 2nd **Annual Doctoral Students' Symposium**. **December 5, 2011**. Massey University, Palmerston North, New Zealand.	**Presented a paper** on 'The water footprint of hydropower: a vital input for the water footprinting of New Zealand's agricultural products'.	Paper was selected among the best six papers for oral presentation Won the best presenter award at the session on 'Agricultural and Natural Sciences' **Publication no. 10**
12.	Annual **workshop** of Fertilizer and Lime Research Centre, Massey University on 'Advanced Nutrient Management Gains from the Past - Goals for the Future' 28–29 Feb 2012. Massey University, New Zealand.	**Presented a paper** on 'Measuring the grey water footprint of potatoes'.	Actively involved in discussions and received constructive comments on the research presented. **Publication no. 9**
13.	Attended, as a resource person, the **Workshop on Water Footprinting-Principles, Methods & Guidelines** at the 2nd New Zealand Life Cycle Assessment Centre Conference; 27 March 2012. Auckland, New Zealand.	**Presented a paper** on 'Assessing the grey water footprint.'	Actively participated in discussions on methodological issues on water footprinting.

14.	**2nd New Zealand Life Cycle Assessment Centre Conference** 28–29 March 2012. Auckland, New Zealand	**Presented a paper** on 'Variability in agricultural product water footprints: A case study of New Zealand wine'.	Successfully fielded questions and queries related to water footprinting. The importance of methodological comparison was emphasised. **Publication no. 8**
15.	**1st New Zealand student symposium on Life cycle management** 16 October 2012 Massey University, Palmerston North, New Zealand.	**Presented a paper on '**The water footprint of New Zealand wine: Evaluation of different water footprinting protocols'.	Actively participated in discussions on methodological issues on water footprinting. **Publication no. 15**
16.	**The Joint Australian and New Zealand Soil Science conference** 2 - 7 December 2012, Hobart, Australia.	**Presented a paper** on 'Methodological challenges in agricultural product water footprinting: A case study of New Zealand wine'.	Successfully engaged in discussions related water footprinting. **Publication no. 7**
17.	Annual **workshop** of Fertilizer and Lime Research Centre, Massey University on 'Accurate and efficient use of nutrients on farms' 12–14 Feb 2013. Massey University, New Zealand	**Presented a paper** on 'Is the grey water footprint helpful for understanding the impact of primary production on water quality?'	Engaged in discussions related to the water footprint and networking. **Publication no. 6**

APPENDIX- D
Statements of contribution for publications

DRC 16

MASSEY UNIVERSITY
GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Indika Herath

Name/Title of Principal Supervisor: Associate Professor David Horne

Name of Published Research Output and full reference:

A scientific paper in an international journal

Herath, I., Deurer, M., Horne, D., Singh, R., and Clothier, B. (2011). The water footprint of hydroelectricity: A methodological comparison from a case study in New Zealand. Journal of Cleaner Production 19 pp.1582-1589

In which Chapter is the Published Work: Chapter 3

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate: 95%

  and / or

- Describe the contribution that the candidate has made to the Published Work:

indika.herath@plantandfood.co.nz

Candidate's Signature

18.12.2012

Date

David Horne

Principal Supervisor's signature

18.12.2012

Date

GRS Version 3– 16 September 2011

DRC 16

MASSEY UNIVERSITY
GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Indika Herath

Name/Title of Principal Supervisor: Associate Professor David Horne

Name of Published Research Output and full reference:

A scientific paper in an international journal

Herath, I., Green, S., Singh, R., Horne, D., van der Zijpp, S., and Clothier, B. (2013). Water footprinting of agricultural products: a hydrological assessment for the water footprint of New Zealand's wines. Journal of Cleaner Production 41, pp 232-243.

In which Chapter is the Published Work: Chapter 4

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate: 90%

  and / or

- Describe the contribution that the candidate has made to the Published Work:

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Candidate's Signature

18.12.2012

Date

David Horne Digitally signed by David Horne DN: cn=David Horne, o=Massey University, ou=Institute of Natural Resources, email=D.J.Horne@massey.ac.nz, c=NZ Date: 2012.12.18 17:29:09 +13'00'

Principal Supervisor's signature

18.12.2012

Date

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DRC 16

MASSEY UNIVERSITY
GRADUATE RESEARCH SCHOOL

STATEMENT OF CONTRIBUTION
TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Indika Herath

Name/Title of Principal Supervisor: Associate Professor David Horne

Name of Published Research Output and full reference:

A scientific paper in an international journal

Herath, I., Green, S., Horne, D., Singh, R. McLaren, S. and Clothier, B. (2013). Water footprinting of agricultural products: Evaluation of different protocols using a case study of New Zealand wine. Journal of Cleaner Production, Accepted for publication.

In which Chapter is the Published Work: Chapter 5

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate: 95%

  and / or

- Describe the contribution that the candidate has made to the Published Work:

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GRS Version 3– 16 September 2011