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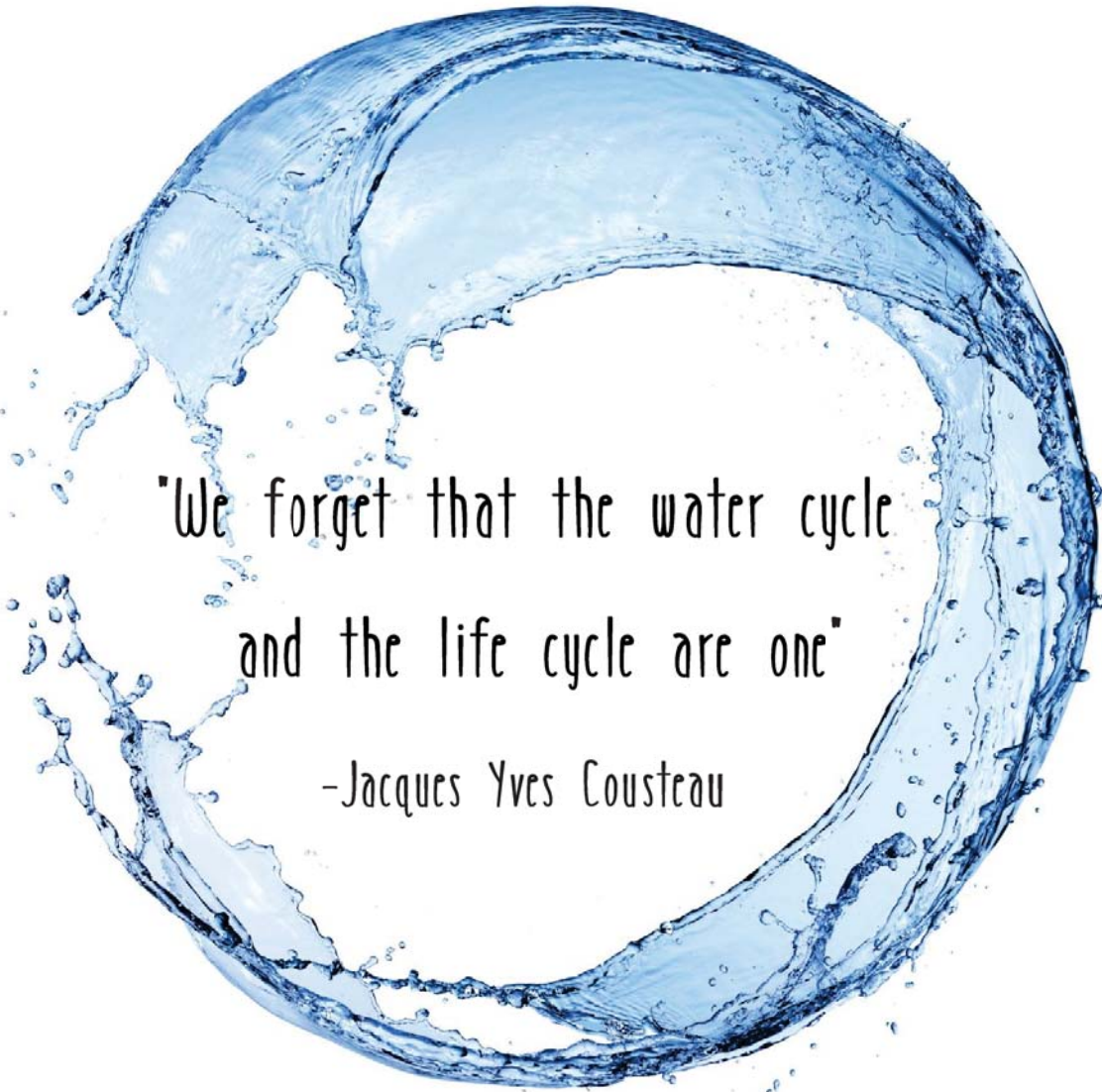
Assessing water footprint and associated water scarcity indicators at
different spatial scales: A case study of concrete manufacture in New
Zealand

A thesis presented
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A circular splash of water, captured in a high-speed photograph, forms a ring around the text. The water is a vibrant blue, and the splash is dynamic, with many small droplets and ripples visible. The background is plain white.

"We forget that the water cycle
and the life cycle are one"

-Jacques Yves Cousteau

Abstract

Water scarcity is a growing issue of concern across the globe. In recent times a complex suite of water footprint impact assessment tools and concepts have supplemented traditional management approaches. There are several methods proposed in the literature to both quantify water use and assess its environmental impacts at defined spatial scales. In New Zealand, case studies in the water footprinting space are sparse, and are for the majority focused on the agricultural industry.

This thesis focused on evaluation of different water footprint methods and their associated water scarcity indicators to assess water use impacts for the building and construction sector of New Zealand. The water footprints of 1 m³ ready mix concrete manufactured at 27 concrete batching plants throughout New Zealand were calculated at three distinct spatial scales: the freshwater management zone scale, catchment scale, and regional scale. Four water footprint characterisation factors (blue water scarcity (WS_{blue}) (Hoekstra et al., 2011), water stress index (WSI) (Pfister et al., 2009), water depletion index (WDI) (Berger et al., 2014), and available water minus demand (AMD) (Boulay et al., 2016)) were used to assess the environmental impact of water use for 1 m³ ready mix concrete at the three spatial scales.

The average volumetric blue water footprint of the 27 ready mix batching plants was quantified at 0.18 m³ (180 litres) of water per m³ of concrete, and ranged from 0.15 (150 litres) to 0.29 m³ (290 litres) of freshwater per m³ of concrete. For three of the four water footprint methods used (WDI , WSI and WS_{blue}), and across the three spatial, the Ashburton boundary ranked highest in terms of the environmental impacts of a specified quantity of water use. In contrast, the AMD method ranked the Palmerston North boundary highest across the three spatial scales. At the freshwater management zone and catchment scales, the WDI , WSI and WS_{blue} methods ranked the Wanganui area lowest, and the AMD method ranked the Greymouth area lowest. At the regional scale, all the four water footprint methods ranked the West Coast region lowest in terms of the environmental impact of water use, due mainly to the fact that the West

Coast has more available water and a lower allocation demand than other regions studied. The analysis indicated that volumetric water use varied by a factor of two across the different plants (per m³ concrete). For three of the four WF methods (*WDI*, *WSI* and *WS_{blue}*), the WF results were similar in their rankings of the different plants at all the geographical scales; however, the *AMD* method resulted in different rankings at all the geographical scales. Overall, the *WDI* and *WSI* water scarcity indices calculated by Berger et al. (2014) and Pfister et al. (2009) were less readily adaptable to the finer resolution in New Zealand. The *WS_{blue}* and *AMD* calculated by Hoekstra et al. (2011) and Boulay et al. (2016) however, were found to be more readily adaptable. It is recommended that these methods be explored further with respect to their potential use at the finer resolution in New Zealand.

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List of Abbreviations

Acronym	Definition
EFR	Environmental Water Requirement
LCA	Life Cycle Assessment
WFN	Water Footprint Network
ISO	International Standardisation Organisation
GIS	Geographic Information System
WSI	Water Stress Index
LCIA	Life Cycle Impact Assessment
WAVE	Water Accounting and Vulnerability Evaluation
WTA	Withdrawal-to-Availability Ratio
LCI	Life Cycle Inventory
BIER	Basin Internal Evaporation Recycling Ratio
WDI	Water Depletion Index
CTA	Consumption-to-Availability Ratio
SWS	Surface Water Stocks
WULCA	Water Use Life Cycle Assessment
AWaRe	Available Water Remaining
FMZ	Freshwater Management Unit
SI	Supplementary Information

Glossary of Terms

Water Stress	The ability, or lack thereof, of water to meet human <i>and</i> ecological demands
Water Scarcity	The lack of available freshwater required to meet water usage demands for a particular region
Surface Water	Water that is on the Earth's surface, such as in a stream, river, lake, or reservoir.
Groundwater	Groundwater is the water stored beneath Earth's surface in aquifers (layers of water-bearing rock or sand). Groundwater exists in the zone of saturation, and may be fresh or saline
Blue Water	Refers to the consumption of surface and groundwater resources
Green Water	Refers to the consumption of precipitation on land- insofar as it does not become runoff
Grey Water¹	Refers to pollution, and can be described as the volume of freshwater required to assimilate a pollutant load given natural background conditions and existing ambient water quality standards

¹ Defined according to water footprint terminology

Chapter 1

Introduction

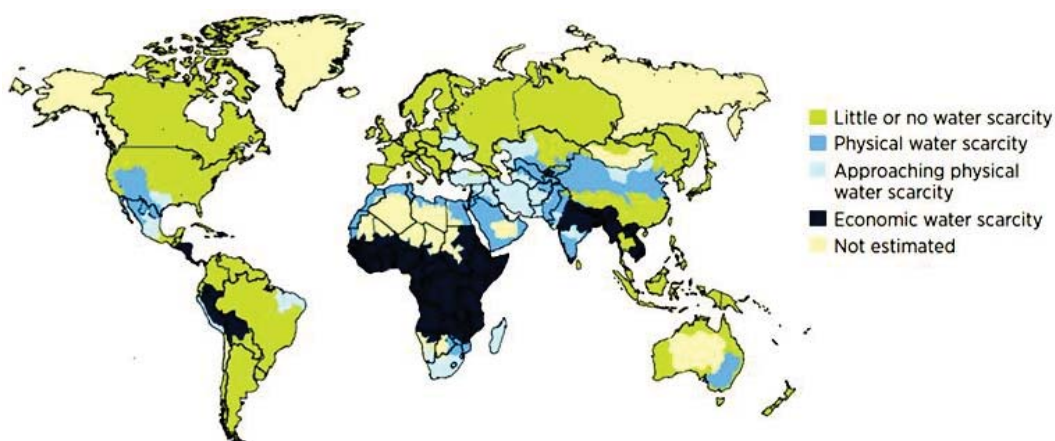
1.1 The Global Freshwater Situation

Freshwater constitutes a mere 2.53% of the total volume of water on Earth, of which 1.74% is unobtainable because it is held within glaciers and the ice sheets of the Polar Regions (Shiklomanov & Rodda, 2003). This leaves a meagre 0.79% of the total volume of water on Earth in the form of readily available freshwater (approximately 11 million km³/year) in aquifers, surface water bodies, soil pores and the atmosphere (Shiklomanov & Rodda, 2003). With its relatively low availability for human consumption (compared with salt water), it represents a significant challenge then, that freshwater is the most extracted natural resource on the planet by mass (Rockstrom et al., 2009). Approximately 4,000 km³ of surface water and 700 km³ of groundwater are withdrawn annually at the global level (Rockstrom et al., 2009).

Moreover, the distribution of freshwater across the globe is extremely variable and corresponds poorly to the distribution of the world population. For example, Asia contains approximately 61% of the world's population but has access to only 36% of global runoff which, when compared with South America (which has 5.5% of the world's population and access to 25.6% of global runoff), demonstrates the disparity between water demand and availability in different regions (Postel, Daily, & Ehrlich, 1996). In many countries, especially the larger ones, there can be both water-scarce and water-abundant areas. These areas are often geographically separated with no chance for inter-basin water transfers (United Nations Water [UN Water], 2007). At present, there are around 663 million people in developing nations that lack access to safe drinking water, and a further 2.4 billion people lacking adequate water for sanitation (World Health Organization [WHO], United Nations Childrens Fund [UNICEF], 2015). It has been estimated that more than 842,000 people die annually from diseases associated with unsafe drinking water, and lack of water for hygiene (Prüss-Ustun et al., 2014). All the while, water usage continues to increase to meet the

demands of a growing population, economic prosperity, urbanisation and industrialisation.

Not surprisingly then, water scarcity, with its associated impacts, is becoming an issue of increasing concern across the globe. The term water scarcity refers to a lack of sufficient freshwater to meet the water usage demands of a region. This can occur in the form of physical water scarcity (a region does not have enough freshwater available to meet requirements), or economic water scarcity (where there is a lack of infrastructure and/or mismanagement of the freshwater resource). Alternatively, there are instances in which this may occur due to extraordinary events such as extreme drought or war (Pfister, Koehler, & Hellweg, 2009). Approximately 1.2 billion people (a fifth of the world population) live in areas of physical water scarcity, whilst 1.6 billion people (a quarter of the world population) face economic water scarcity (Figure 1.1) (UN Water, 2007).



Definitions and Indicators

- *Little or no water scarcity:* Less than 25% of water from rivers withdrawn for human purposes.
- *Physical water scarcity:* More than 75% of river flows are withdrawn for agriculture, industry, and domestic purposes (accounting for recycling of returned flows).
- *Approaching physical water scarcity:* More than 60% of river flows are withdrawn. These basins will experience physical water scarcity in the near future.
- *Economic water scarcity:* Water resources are abundant relative to water use, with less than 25% of water from rivers withdrawn for human purposes, but malnutrition exists.

Figure 1.1 A global map of physical and economic water scarcity (Comprehensive Assessment of Water Management in Agriculture, 2007)

The increasing consumption of freshwater, coupled with water quality degradation, are the major contributing factors to the increase in water scarcity being experienced by an ever expanding proportion of the global populace. Ten of the worst affected river basins based on population size and months of water scarcity are shown in Figure 1.2. Countries with the largest populations (India, China, and the USA) suffer from water scarcity for most of the year (Hoekstra & Mekonnen, 2012b; World Wide Fund for Nature [WWF], 2014). With the world population forecast to reach 9.6 billion in 2050 (United Nations [UN], 2013), it has been estimated that by that time, half a billion people will experience some form of 'water-stress' first-hand (Rockstrom et al., 2009).

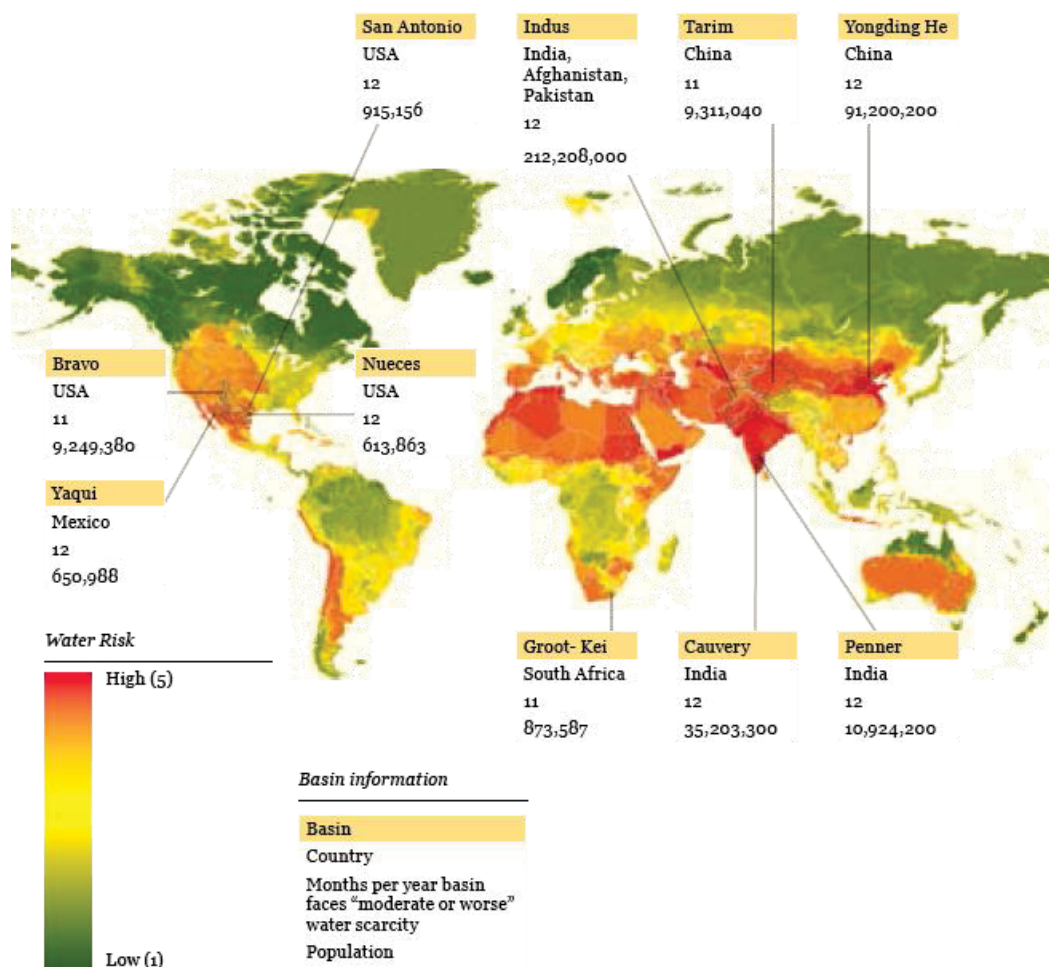


Figure 1.2 The ten worst affected river basins by population and months of water scarcity, 1996-2005 (Hoekstra & Mekonnen, 2012b; WWF, 2014).

It is important here to differentiate between the terms 'water scarcity' and 'water stress'. These terms are often used concurrently in the literature, but have different definitions. Water scarcity refers to a lack of available freshwater required to meet water usage demands for a particular region. The term water stress is more inclusive, and refers to water scarcity in conjunction with water quality, environmental flows and freshwater accessibility (Figure 1.3). Water stress is therefore defined as the ability, or lack thereof, of water to meet human *and* ecological demands (Schulte, 2014). Both water scarcity and water stress can result in water risk, defined as the probability and severity of an entity experiencing a negative water related event.

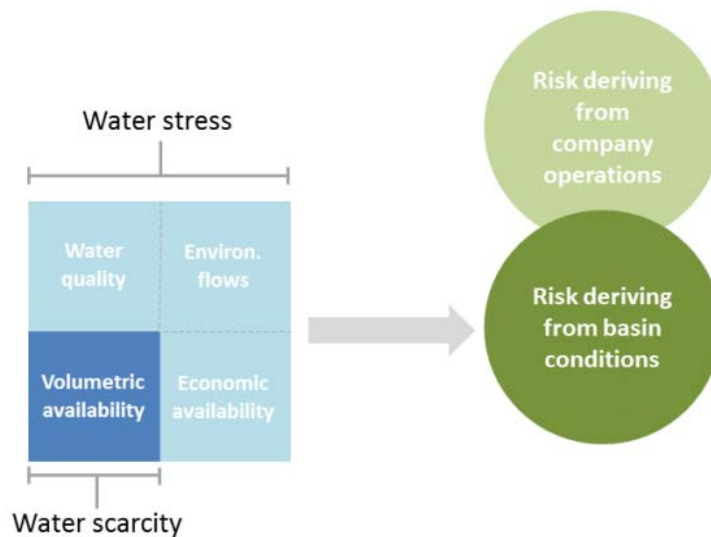


Figure 1.3 A schematic diagram of the relationship between water scarcity, stress and risk (Schulte, 2014).

Water scarcity and water stress both affect terrestrial and aquatic ecosystem functioning, as well as water, food and health security for humans. The deterioration of freshwater source can have detrimental effects on human livelihoods. Changes in freshwater availability, for instance, can threaten water supply and cause collapse of both the terrestrial and aquatic ecosystems, and a loss of biodiversity. Biodiversity provides the link between the multitude of essential ecosystem services which are indispensable to human wellbeing (Figure 1.4). Examples include, food, fibre and

water provision, disease and climate regulation, primary production, soil formation and nutrient cycling, to name a few (Millennium Ecosystem Assessment, 2005).

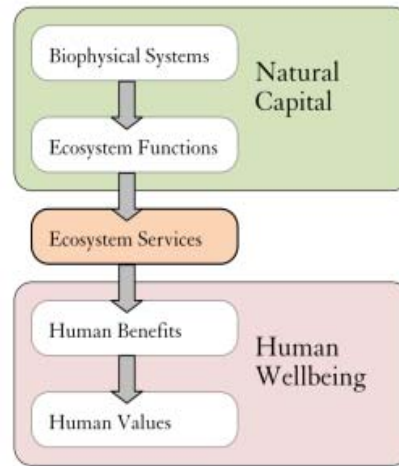


Figure 1.4 Linkages between natural capital, ecosystem services and human wellbeing (Baker et al., 2011).

A recent study conducted by the WWF indicates that populations of freshwater species are most at risk, having declined more rapidly than both their terrestrial and marine counterparts². Freshwater populations declined by 76% from 1970-2010, compared with 39% for both terrestrial and marine populations (WWF, 2014).

The depletion and degradation of global freshwater resource is therefore an expanding issue of concern, as the effects of water stress and water scarcity become more prevalent, and concerns heighten for the future sustainability of this crucial resource. As the human population continues to increase, the demand for water intensive production of food and energy is predicted to rise by 70% and 60%, respectively by the year 2050 (Bruinsma, 2009; UN, 2013). This increase in water for food and energy demand is expected to create resource competition on the global scale. Therefore, the need for effective management of the freshwater resource is likely to increase into the future in order to sustainably manage increasing water scarcity and/or water stress in many parts of the world.

² This is based on trends in 3,066 populations of 757 mammals, bird, reptile, amphibian and fish species (WWF, 2014).

1.2 The Water Footprinting Concept

Given these challenges, traditional approaches to water management are being supplemented with a more complex suite of concepts and tools to assess freshwater use and its associated environmental, economic, social, and cultural impacts. Included in these recent advances is the concept of water footprint, and related analytical water footprint methods and tools.

1.2.1 Origin and the Contemporary Definition

The water footprint concept has its origins in the 1990s when the term 'virtual water' was coined by Professor John Anthony Allan, after he identified that the consumptive water use of citrus fruit production in water scarce countries of the Middle East, was much greater than the water content of the fruit itself (Allan, 1998). The term virtual water is used in water footprinting to describe water 'embodied' in a product. It refers to the volume of water consumed and/or polluted, as measured throughout the entire production chain of a product (Allan, 1998).

In 2002, Professor Arjen Hoekstra and colleagues expanded this concept by distinguishing different types of water consumed and/or polluted (Hoekstra, Chapagain, & Mekonnen, 2011). Traditionally, water use assessments have been concerned with defining the amount of surface and groundwater withdrawn and/or directly consumed during a process or in the production of a product (Hoekstra et al., 2011). The water footprint concept is more complex and comprehensive than typical water withdrawal measurements as it quantifies the volume of freshwater used both directly *and* indirectly in the production lifecycle (Figure 1.5). Water footprint methods are therefore able to account for virtual water flows, in other words, the 'embodied' water being transferred between two geographically delineated areas as a result of product trade (Hoekstra et al., 2011). Furthermore, they allow for differentiation between different 'colours' of water use (based on source and function), and omit the non-consumptive part of water withdrawal (return flow), because it is returned to the environment (Hoekstra et al., 2011).

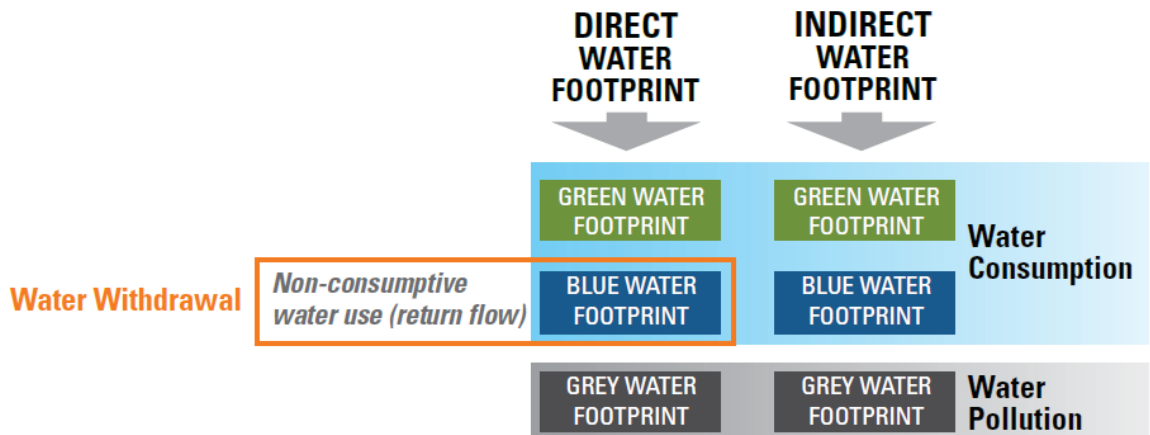


Figure 1.5 A schematic diagram of the components of a water footprint, according to Hoekstra et al. (2011).

1.2.2 Defining Water 'Colour' Criteria

As can be seen in Table 1.1, three 'colours' of water are commonly differentiated in water footprinting methods. These are: surface and groundwater (blue water), soil moisture (green water)³, and freshwater required to assimilate pollution (grey water). The green and blue water colours account for water quantity, whilst the grey water colour accounts for water quality aspects of freshwater use and management.

Table 1.1 Water colour definitions as provided by Hoekstra et al. (2011).

Water Colour	Definition
Blue	Refers to the consumption of surface and groundwater resources
Green	Refers to the consumption of precipitation on land- insofar as it does not become runoff
Grey	Refers to pollution, and can be described as the volume of freshwater required to assimilate a pollutant load given natural background conditions and existing ambient water quality standards

³ Soil moisture is a measure of the rainwater stored in the soil profile which is ultimately consumed as evapotranspiration by plants in primary production (Hoekstra et al., 2011).

The distinction between blue and green water is an important development, as the hydrological, environmental, social and economic opportunity costs of blue and green water can differ significantly (Hoekstra & Chapagain, 2008). Green water is particularly important in primary production, as it ensures water-dependent food security by sustaining rain-fed crop production. Green water allocation and use is intrinsically linked to the land use type and primary production, while blue water could be transported and allocated to a range of water users including domestic, industrial and agricultural water use. Globally, green water represents the largest share of virtual water in the international trade of agricultural commodities (Aldaya, Allan, & Hoekstra, 2010). The grey water footprint accounts for water quality impacts associated with a production or consumption process. The grey water concept was first introduced by Falkenmark and Lindh (1974), expressed as the volume of water required to 'dilute' a volume of polluted water to bring it within accepted standards. The concept has been updated and debated in the literature by numerous academics (Chapagain, Hoekstra, Savenije, & Gautam, 2006; Hoekstra & Chapagain, 2008; Postel et al., 1996). A brief history of the concept and its development are discussed in Appendix A.

1.2.3 Volumetric vs. Impact Orientated Water Footprints

The methodology for water footprinting is still an emerging subject area and different methods are being proposed, debated, and revised to define and standardise water footprint assessments (Bayart et al., 2010; Berger, van der Ent, Eisner, Bach, & Finkbeiner, 2014; Boulay et al., 2016; Hoekstra, Chapagain, Aldaya, & Mekonnen, 2009; Hoekstra et al., 2011; Milà i Canals et al., 2009; Pfister et al., 2009; Ridoutt & Pfister, 2010). There are two main schools of thought with regards to water footprinting method. The first is concerned with measuring the *volume of water used* (Hoekstra et al., 2011)⁴, and the second is concerned with directly assessing the *impacts associated with water use* (Bayart et al., 2010; Berger et al., 2014; Boulay et al., 2016; Boulay, Bouchard, Bulle, Deschenes, & Margni, 2011a; Milà i Canals et al., 2009; Pfister et al., 2009; Ridoutt & Pfister, 2010). Those working on impact-orientated

⁴ Note: a sustainability assessment was later incorporated in the method of Hoekstra et al. (2011) allowing for an impact assessment of water use in-line with the aforementioned LCA methods.

assessments take a Life Cycle Assessment (LCA) approach (refer section 2.2) to assess and quantify resource and environmental impacts associated with water use.

Water footprint methods that take an impact assessment approach develop water scarcity indicators. These indicators are then used to characterise the impacts associated with water withdrawal or water consumption. Pertinent impact indices discussed include: The Blue, Green and Grey Water Footprint Impact Indices (Hoekstra et al., 2011), Water Stress Index (Pfister et al., 2009), Water Depletion Index (Berger et al., 2014) and the Water Deprivation Potential (Boulay et al., 2016).

Volumetric water footprint assessments may select between the different colours of water or include all water types, depending on the product or process under assessment (Hoekstra et al., 2011). In contrast, methods which focus on impact assessment differentiate water flows with regards to their environmental impact (Bayart et al., 2010; Berger et al., 2014; Milà i Canals et al., 2009; Pfister et al., 2009; Ridoutt & Pfister, 2010). Generally, they disregard the green water use arguing that this could be better associated with land use management than water use management (Pfister et al., 2009; Ridoutt & Pfister, 2010).

1.2.4 Scale and Scope of Water Footprint Assessments

Importantly, water footprint assessments can be undertaken at a range of spatial and temporal scales. For example, water footprints exist at the global, national, and regional scales, as well as for localised products, processes, and organisations. The temporal scale of the assessments is also an important consideration, as water availability is likely to change throughout the year due to seasonal changes and extreme weather events - such as flooding and droughts (Hoekstra et al., 2011; Pfister & Bayer, 2014).

At present, the development of water footprint methodologies have been focused on the global and national scales. Notably less research has been conducted at the catchment or finer resolution, and there is little in the way of assessing the effects of the different water footprint methods and comparisons between different spatial scales, particularly for products, processes and organisations in New Zealand.

1.3 Problem Statement

There are few water footprinting studies using New Zealand data. Current literature regarding the water footprint of New Zealand products, processes and organisations is sparse, with some global level studies using data that may be outdated and/or averaged at the national level. Yet Hoekstra and Mekonnen (2012b) stress the importance of scale in water footprinting and make the point that national level data can often have significantly different results than data taken at the catchment level. Thus, the results of water footprinting studies can often yield dramatically different results depending on both the spatial and temporal scale of the assessment.

This is demonstrated for New Zealand with a simple comparison of reports by Hoekstra et al. (2010) and the New Zealand Business Council for Sustainable Development (NZBCSD, 2008). Hoekstra et al. (2010) provide a global map of water stress that depicts New Zealand as a singular eco-region, said to have low or no water stress (Hoekstra et al., 2010). In contrast, the NZBCSD (2008) state that water scarcity is increasing in some regions of New Zealand, with a number of catchments having water allocations that exceed total available water. The scale of water footprint assessment is clearly a very important consideration. Hoekstra and Mekonnen (2012b) explained that assessments of water use and its impacts are more robust at a finer level of resolution (i.e. river-basin/ catchment scale). This is particularly true in New Zealand, where there is high spatial and temporal variability in availability and consumption of freshwater.

In addition to this, a number of different water footprint methods have been proposed in the literature (Berger, van der Ent, Eisner, Bach, & Finkbeiner, 2014; Boulay et al., 2016; Boulay et al., 2011a; Boulay, Bouchard, Bulle, Deschenes, & Margni, 2011b; Frischknecht, Steiner, & Jungbluth, 2009; Hoekstra et al., 2011; Milà i Canals et al., 2009; Pfister et al., 2009). Each of these proposed water footprint methods uses a different approach to assess the environmental impacts associated with water consumption and/or pollution. Consequently, a secondary source of variation arises from methodological choice. It is possible to get results that are quite different

depending on which impact assessment method and subsequent characterisation factors are employed to assess water footprint of a process or product.

Finally, New Zealand water footprint studies to date have focused mainly on agricultural products and processes (Herath, Green, Horne, Singh, & Clothier, 2014a, 2014b; Herath et al., 2013a; Herath et al., 2013b; Zonderland-Thomassen & Ledgard, 2012; Zonderland-Thomassen, Lieffering, & Ledgard, 2014). However, concern about industrial water consumption and the potential for adverse environmental effects have increased more recently (Pfister, Saner, & Koehler, 2011).

1.4 Research Aims

This research aims to investigate the feasibility of adapting different water footprint methods, and water scarcity indicators in particular, for the assessment of water use in the building and construction sector of New Zealand. The building and construction sector is the fifth largest economic sector in New Zealand, with both domestic and export markets (PricewaterhouseCoopers [PwC], 2011). More specifically, this research focuses on a case study of the water footprint of 1 m³ ready mix concrete produced at 27 concrete batching plants throughout the North and South Islands of New Zealand. The case study is used to investigate the effects of using different water footprint methods, at different spatial scales, on the overall water footprint results for 1 m³ ready mix concrete produced at different locations in New Zealand.

1.5 Research Objectives

The specific research objectives are as follows:

- I. Review and identify water footprint methods that can potentially be used for New Zealand-based water footprint studies.
- II. Calculate New Zealand specific characterisation factors for selected water footprint methods at three different scales (the freshwater management zone scale, catchment scale and regional scale).
- III. Conduct a New Zealand specific water footprint case study of concrete manufacture to investigate the influence of the different water footprint

methods and spatial scales on quantification of water footprint of 1 m³ ready mix concrete produced at different locations in New Zealand.

- IV. Evaluate the case study results and their implications for the concrete industry and for future water footprint assessments in New Zealand.
- V. Make recommendations on areas that could benefit from further research and development.

1.6 Relevance of the Proposed Research

The evaluation of different water footprint methods using local data sets aims to provide some insight into the feasibility and adaptability of using internationally developed water footprint methods for New Zealand studies. Moreover, by conducting water footprint analysis at a finer spatial resolution in New Zealand, using local data, the results of the water footprint assessment should be more accurate and meaningful than those using currently averaged national (or global) datasets at larger spatial scales. This will facilitate a more accurate calculation of the water footprint associated with products and processes in New Zealand.

Using a case study from the building and construction sector⁵, as opposed to the agricultural sector (which has notably received more attention in this arena), this thesis seeks to broaden the scope of application for water footprint and associated impact assessments in New Zealand.

This information could then be used to inform management decisions in order to better sustain our natural resources and reduce any adverse environmental impacts from production and manufacture in New Zealand. Furthermore, given the increasing concern over water scarcity worldwide, there is the potential to provide marketing opportunities for New Zealand export products by demonstrating the water footprint credentials of these products.

⁵ Specifically from the manufacturing classification C203300 (Statistics New Zealand & Australian Bureau of Statistics, 2006)

1.7 Thesis Outline

Chapter Two: Background

A New Zealand context to issues of water quantity, quality and scarcity are provided, alongside a discussion of how these issues are perceived both internationally and locally, and how they are affecting particular regions. The chapter then goes on to broaden the concept of Life Cycle Assessment, and provide some context to the New Zealand building and construction sector. Recent sustainability initiatives in the industry, and the growing importance of water footprinting and LCA as environmental assessment tools within New Zealand are discussed.

Chapter Three: Literature Review

This chapter comprehensively and critically reviews existing water footprint methods with a focus on the chosen methods of Hoekstra et al. (2011); Berger et al. (2014); Pfister et al. (2009) and Boulay et al. (2016). A particular focus is given to the way in which these methods characterise water scarcity at different spatial scales. Studies conducted in New Zealand using these water footprint methods (amongst others) are also reviewed and summarized to inform the method and data collection used in this study.

Chapter Four: Methods and Materials

A detailed account of how different water footprint and impact indices are calculated using the four selected methods, and at the three spatial scales. A detailed explanation of data requirements, data quality, local data sources and data limitations encountered during the process are provided.

Chapter Five: Results

This chapter presents and describes the results of the water footprint and impact assessment results obtained using the four selected water footprint methods, applied at the three spatial scales. A simple statistical analysis is conducted to assess the relationships between different variables considered in each of the water footprint methods studied, and their impact on the calculated variability.

Chapter Six: Discussion

A discussion and explanation of the results of the case study, alongside a general discussion surrounding the feasibility and applicability of adapting each of the four methods to the finer resolution in New Zealand. Key areas of discussion include the effect of method and scale on the characterisation factors themselves as well as their effect on the post-characterised water footprint of concrete. A discussion pertaining to the limitations of applying each method is provided.

Chapter Seven: Conclusions, Limitations and Recommendations

Finally, this chapter evaluates the overall feasibility of adapting each method to the three distinct spatial scales, highlights limitations in terms of data availability and quality, and provides both a spatial and a methodological recommendation for future water footprint and impact assessment in New Zealand.

Chapter 2

Background

2.1 Freshwater in New Zealand

Freshwater has been described as New Zealand's greatest natural asset (Ministry for the Environment [MfE], 2014a), and is regarded as essential to New Zealand's social, cultural and economic wellbeing (MfE, 2007). Freshwater is not only essential to many of the primary industries which earn New Zealand its wealth, but is also valued for recreation by locals and tourists alike. Furthermore, the health of freshwater bodies is culturally significant to Māori, as they continue to be used as mahinga kai (a source of food and resources). Freshwater is, in essence, fundamental to the livelihoods of New Zealanders and to the 'clean green' image touted by the national tourism industry. Moreover, freshwater sustains the ecosystems that are home to many indigenous species, 52% of which are endemic (MfE & Statistics New Zealand, 2015). It is this ecological diversity which has led to New Zealand being identified as one of the world's 35 biodiversity hotspots (Mittermeier et al., 2004).

2.1.1 Freshwater Availability and Demand in New Zealand

The availability and demand for freshwater are both spatially and temporally variable across different regions of New Zealand. A series of mountain chains span the length of New Zealand, providing a barrier to prevailing westerly winds and essentially dividing the country into different climatic regions. Nationally, climate ranges from warm subtropical in the far north, to cool temperate in the far south, with severe alpine conditions in the mountainous regions (National Institute of Water and Atmospheric Research [NIWA], 2001). Thus the abundance of freshwater in New Zealand can differ drastically depending on topography and regional characteristics. The West Coast of the South Island, for example, experiences the most rainfall (can be greater than 7,000 mm/year in some areas) and is the wettest region in New Zealand. In contrast, a mere 100km east of the Southern Alps mountain range, the Canterbury and Otago Regions are the driest (can be less than 1,000 mm/year in some areas) in New Zealand (refer to Figure 2.1).

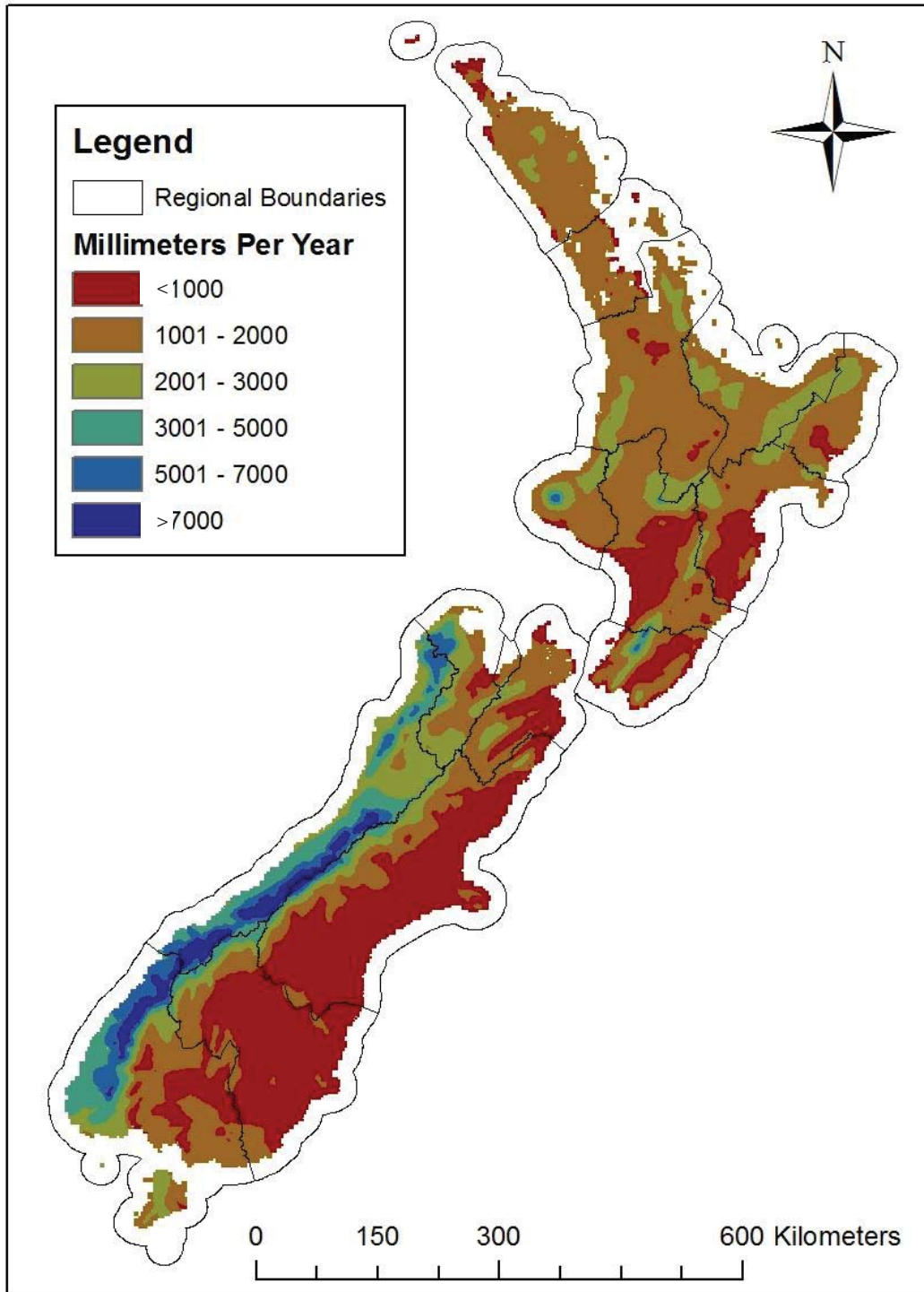


Figure 2.1 New Zealand median annual rainfall for the period 1981-2010 (data provided by NIWA)

Freshwater availability in New Zealand is, by international standards, abundant due to plentiful rainfall experienced across the country, and comparatively low population density (MfE, 2007). In fact, the total freshwater available per person in New Zealand is reported to be higher than 180 other countries across the world (MfE, 2007). Per capita, freshwater availability in New Zealand is ranked fourth amongst members of the Organisation for Economic Co-operation and Development (OECD), behind Canada, Iceland and Norway (New Zealand Institute of Economic Research, 2014). In a global scale study conducted by the World Resources Institute, baseline water stress⁶ (amongst other variables) was ranked. New Zealand ranked 100th (out of a total of 176) for overall baseline water stress. According to the method, areas with a baseline water stress of greater than 20% are thought to experience heightened risk associated with environmental water stress. All sectors⁷ in New Zealand were reported to have a low to medium (10-20%) water stress (Gassert, Reig, Luo, & Maddocks, 2013).

However, at the same time, New Zealanders use two to three times more freshwater per person than most other OECD countries (MfE, 2007). Figure 2.2 shows how freshwater is allocated within New Zealand (both by source and user). Surface and stored water are the most allocated sources of freshwater in the country, making up 80% of all allocated water collectively. Nationally, groundwater constitutes 20% of consumptive allocation in New Zealand. Freshwater is used predominantly for irrigation, followed by hydropower, industry, domestic water use and stock water requirements. It is important to note that hydropower generation takes in New Zealand are generally non-consumptive; after hydropower generation, water is returned to source and becomes available once more to be allocated to other water users. There is one major exception in New Zealand: a singular hydropower plant in the Southland region. In this instance, the consented weekly allocation is discharged to sea, depriving other water users. The hydropower component in Figure 2.2 depicts this single hydropower consent *only* (MfE, 2010).

⁶ Calculated as the ratio of total annual water withdrawal to total available annual renewable supply (accounting for upstream consumptive use)

⁷ Agricultural, industrial and domestic

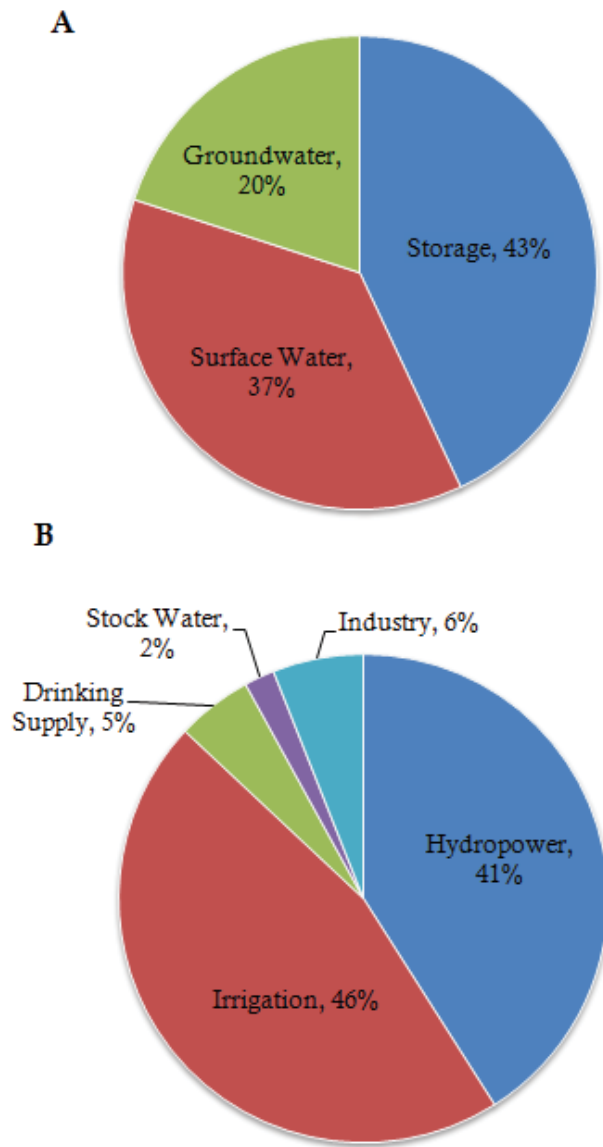


Figure 2.2 The distribution of weekly consumptive allocation in New Zealand by source (A) and use (B) (MfE, 2010).

Distribution and allocation of freshwater (by source) in New Zealand, differs greatly between regions. The proportion of allocated freshwater by source, for each of New Zealand's 16 regional boundaries can be seen in Figure 2.3

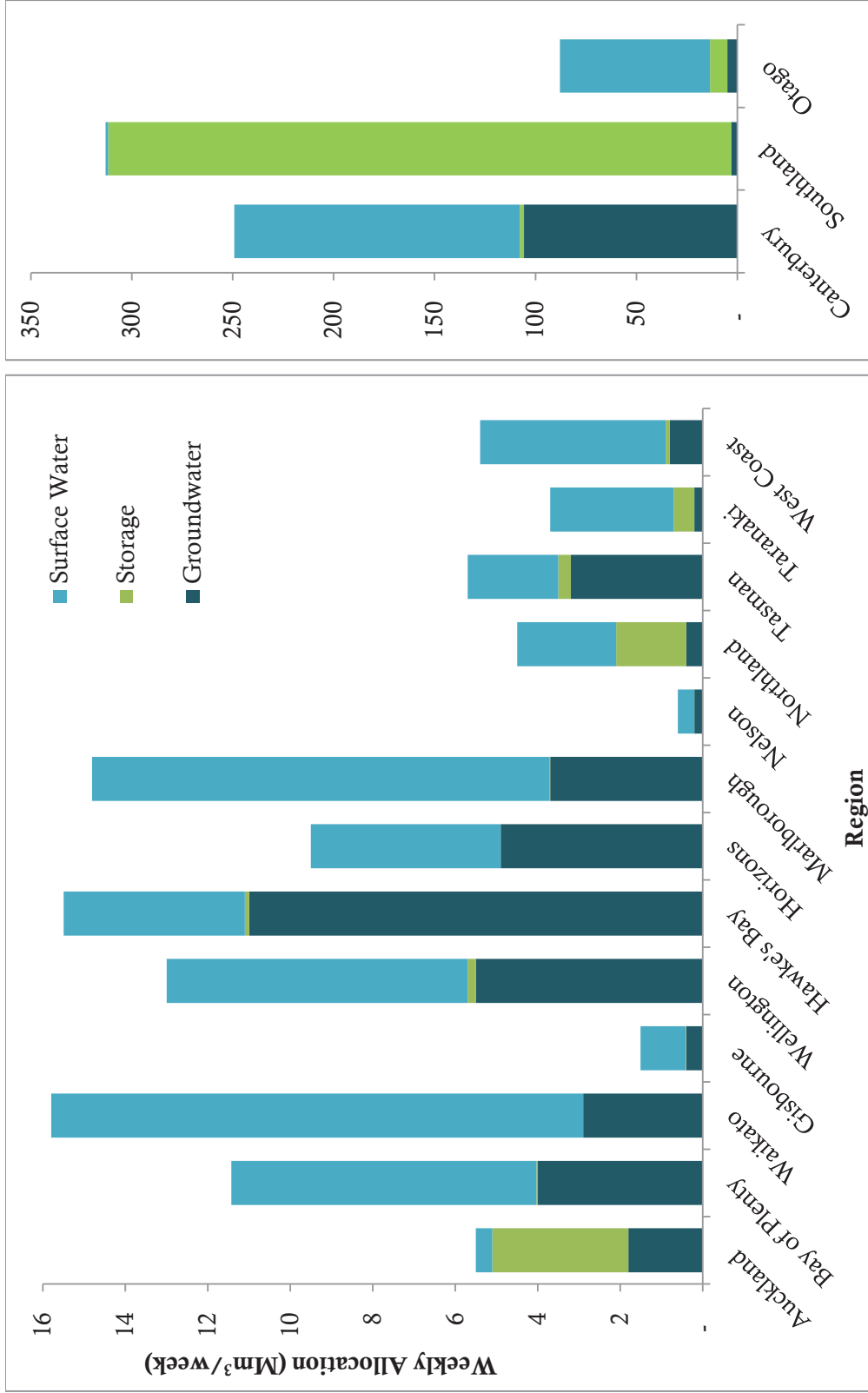


Figure 2.3 Weekly consumptive water allocation, by region and source in New Zealand (MFE, 2010)

As mentioned prior, groundwater allocation is about 20% at the national scale (Figure 2.2). However, in some regions (Hawkes Bay for example) the majority of freshwater is allocated from groundwater aquifers. Aquifers lie beneath 26.3% of New Zealand's total land area, and are an important source of freshwater (MfE & Statistics New Zealand, 2015). There are around 200 mapped aquifers throughout the country (refer to Appendix C) holding, on average, an estimated 711 billion m³ of freshwater. Aquifers of the Canterbury region hold approximately 73% of this freshwater (averaging around 519 billion m³), followed by the Waikato, Bay of Plenty and Taranaki regions (MfE & Statistics New Zealand, 2015).

2.1.2 Environmental Concerns

In recent times, the concern over the country's freshwater resource has begun to grow as some regions reach environmental limits for part or all of the year. Generally, the demand for freshwater peaks during the summer months (December to February), particularly for agricultural uses (irrigation). It is during these months, however, that the availability of freshwater is at its lowest. This has resulted in most regions having at least one river and/or aquifer that is either fully or over allocated, or is likely to be so in the next five years (NZBCSD, 2008).

In addition, the availability and demand for freshwater differ notably between regions. Figure 2.4 shows the maximum allocated surface water as a percent of the calculated minimum flow expected during dry periods (MfE, 2010). Eastern regions such as Otago, Canterbury and Marlborough have a larger proportion of catchments at risk from potential allocation pressures during dry periods. Around 30% of the Otago region and 20% of the Canterbury and Marlborough regions have high potential water allocation pressures. These regions tend to have higher freshwater allocations from catchments considered potentially pressured. For example, nearly three-quarters of allocated water in the Otago region is from highly pressured catchments (MfE, 2010).

Comparatively, the West Coast and Manawatu-Wanganui regions have less than one percent of catchments considered at risk. Freshwater abstractions from regions with a higher proportion of at risk catchments may be considered to have reduced water

security, when compared with regions with lower potential pressure, as restrictions are more likely to be imposed to conserve water (Mfe, 2010).

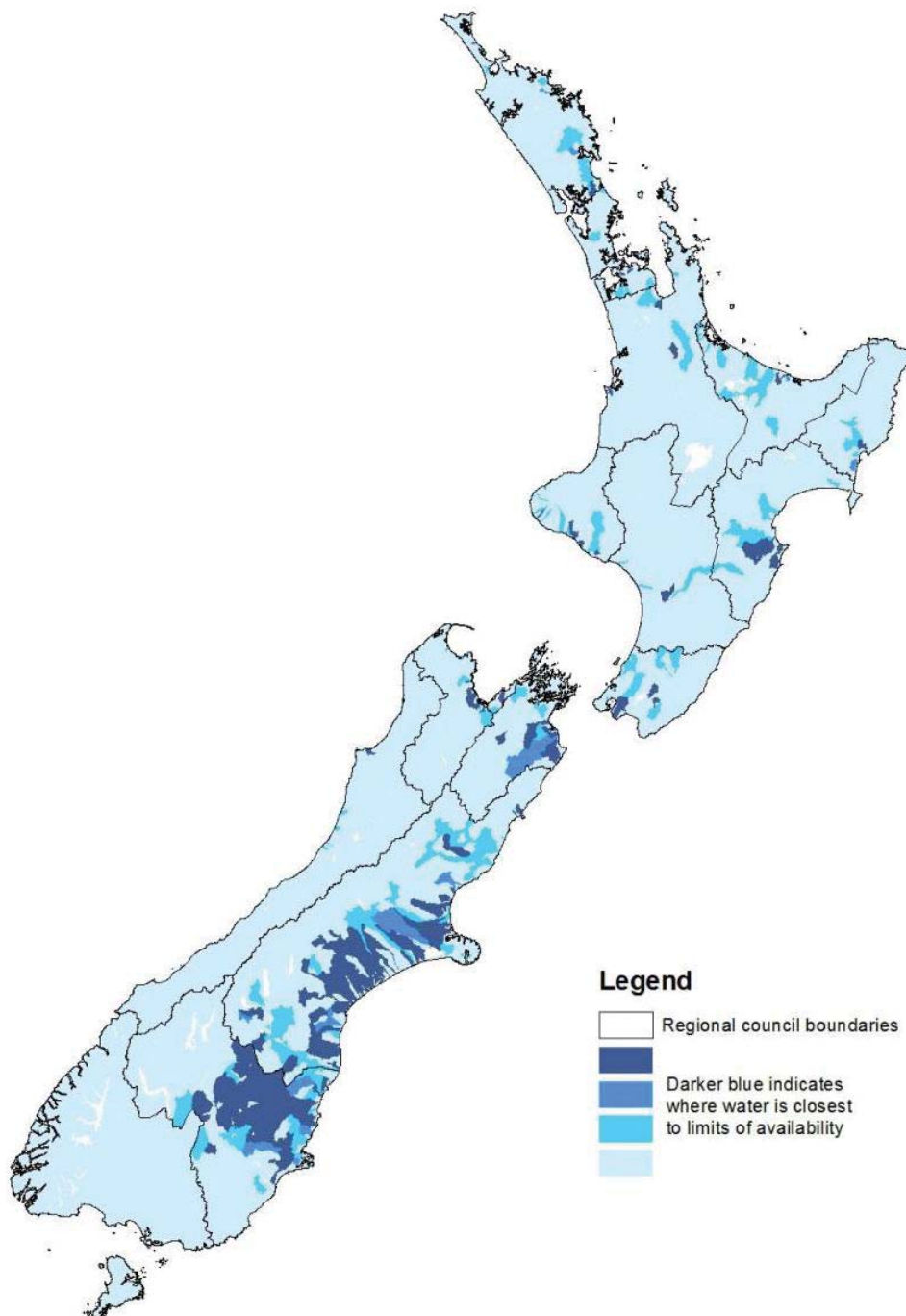


Figure 2.4 Potential surface water allocation pressures across regions of New Zealand (based on 2010 data) (MfE, 2010).

2.1.3 Freshwater Quality in New Zealand

The quality of freshwater in New Zealand is good, by international standards. In fact, a national level Water Quality Index (WQI) produced by the UNEP GEMS Water Programme found New Zealand to have the best water quality of 92 other nations assessed (Carr & Rickwood, 2008). New Zealand scored a WQI of 100 based on five indicative water quality parameters⁸. For all parameters tested, New Zealand did not exceed threshold values as defined in the report (Carr & Rickwood, 2008). Similarly, Yale University's annual Environmental Performance Index (EPI) which ranks countries based on the quality of their resources (e.g. air, water quality, land use and resource management), ranked New Zealand's access to safe drinking water amongst the top 10% of countries in the world (Yale University, 2015).

However, the overall picture emerging from New Zealand's National Water Quality Network (NWQN) is that water quality, while generally good by international standards, is under pressure and showing a declining trend at some locations (NIWA, 2010). At the regional level, water quality in New Zealand is variable and heavily influenced by land-use (MfE, 2007; MfE & Statistics New Zealand, 2015). Aspects of water quality have deteriorated in catchments that are dominated by intensive land uses (such as agricultural or industrial development), whereas catchments that remain unmodified have better water quality (MfE, 2007). Research has indicated a gradient in water quality from excellent in native forest, to good in plantation forestry, and poor in pastoral and urban streams. Streams in areas with dairying activities are amongst the most polluted (NIWA, 2010).

The NWQN results alongside State of the Environment Monitoring (administered by regional councils) indicate diffuse pollution from land-use is the overwhelming cause of water quality degradation in New Zealand (NIWA, 2010). Nitrogen (in the form of dissolved nitrate), phosphorus, faecal microbes and sediment are identified as the key contaminants from diffuse sources. During the period 1989-2007 nitrogen concentrations increased 1.4% per annum over the country (NIWA, 2010). Whilst no

⁸ Water quality parameters used were: Dissolved oxygen, pH, conductivity and nutrients (phosphorus and nitrogen) (Carr & Rickwood, 2008).

catchment saw a decrease in nitrogen, pristine rivers such as the Haast River remained stable. Similarly, phosphorus showed an upwards trend per annum across the country (refer to Appendix D and Appendix E). A comparison with data from the period 1989-2003 shows an increase in temperature, nitrogen and phosphorus across the country (NIWA, 2010). Pastoral farming makes up 40% of New Zealand land use and is unquestionably the main source of diffuse pollutants to rivers and streams (NIWA, 2010). The Waikato, Southland and Canterbury regions are amongst the worst affected in New Zealand (NIWA, 2010).

The way in which the freshwater resource is used can also be seen to have an impact on its quality. For example, non-consumptive processes (such as cooling) have the potential to change the quality of freshwater discharged back into the system. Whilst changes in water quality due to consumptive and degradative water use are beyond the scope of this research, it is important to note that water quality is an important consideration as this generally governs the utility of the freshwater resource for other users (Boulay, Bouchard, Bulle, Deschenes & Margni, 2011b).

2.1.4 Water Scarcity in New Zealand

As previously discussed, rainfall patterns and freshwater availability are not uniformly distributed throughout New Zealand, due to the localised topography and weather patterns. When assessing the water scarcity of a country, it is therefore essential to investigate at the local/regional scale in order to capture the variability influencing freshwater availability, and to understand the local impacts of water use (Berger et al., 2014; Herath, Clothier, Horne, & Singh, 2011a; Hoekstra et al., 2011; Pfister et al., 2009).

In a New Zealand case study conducted by Herath et al. (2011a), regional water scarcity was assessed at the annual scale using three key indicators of water scarcity:

- The Falkenmark Water Stress Index (Falkenmark, 1986),
- The Water Scarcity Index (Alcamo et al., 2003; Hoekstra & Hung, 2002) and,
- The Water Stress Indicator (Smakhtin, Revenga, & Döll, 2004b).

Data on water availability were taken from the physical stock account of water (1995-2005) (Statistics New Zealand, 2010) alongside water allocation data (2002-2005) (MfE, 2006). As actual water use data were unavailable for the most part, it was assumed that 70% of allocated water was actually used.

The Falkenmark Water Stress Index

The Falkenmark Index is an indication of water resource availability per capita per year in the region considered. It can be used to assess both water stress and scarcity based on the following guidelines.

Table 2.1 Water stress categories and thresholds, according to Falkenmark (1986).

Index (m³ per capita)	Definition
>1,700	No Stress
1,000- 1,700	Stress
500- 1,000	Scarcity
<500	Absolute Scarcity

As can be seen in Table 2.1, the threshold values of 1,700 m³ and 1,000 m³ per capita/per year serves as the standard indicator of water stress and water scarcity, respectively (Falkenmark, 1986). Regions which cannot sustain these values are said to be water stressed or water scarce accordingly.

Herath et al. (2011a) found all regions in New Zealand were above the threshold value for water stress, with the lowest value (5,160 m³ per capita/ per year in the Auckland region) measured to be more than three times greater than the threshold value for water stress (Figure 2.5).

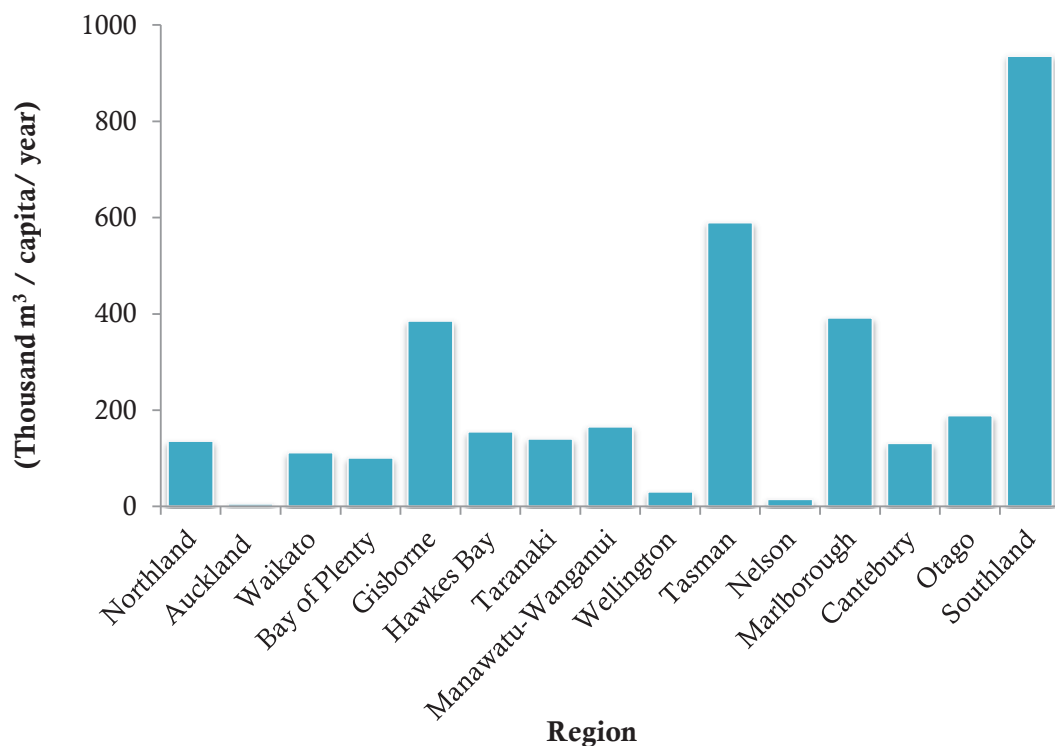


Figure 2.5 The Falkenmark Index (thousands m³ / capita / year) for all regions in New Zealand (Herath et al. 2011).

The Falkenmark method does not, however, account for actual demand or allocation of freshwater within the regions of New Zealand. Nor does it reflect any regional changes in demand due to different consumption patterns or economic factors (Herath et al., 2011a).

The Water Scarcity Index

The Water Scarcity Index as specified by Hoekstra and Hung (2002) differs from the Falkenmark Index, by assessing water stress as the ratio of water use (withdrawal) to total water availability (Alcamo et al., 2003), as opposed to just water availability. The severity of water scarcity according to the Water Scarcity index can be seen in Table 2.2.

Table 2.2 Water scarcity categories and thresholds, according to Hoekstra and Hung (2002)

WSI Rank	Definition
<0.1	No water stress
0.1 < WSI < 0.2	Low water stress
0.2 < WSI < 0.4	Moderate water stress
> 0.4	High water stress
> 0.8	Very high water stress

According to Herath et al. (2011a), none of New Zealand's 16 regions displayed water stress according to the Water Scarcity assessment method. The lowest value was found in the Southland at 0.001, and the highest values (0.041) were found in the Canterbury and Wellington regions, respectively (Figure 2.6). Regardless, all regions fell into the category 'no water stress'.

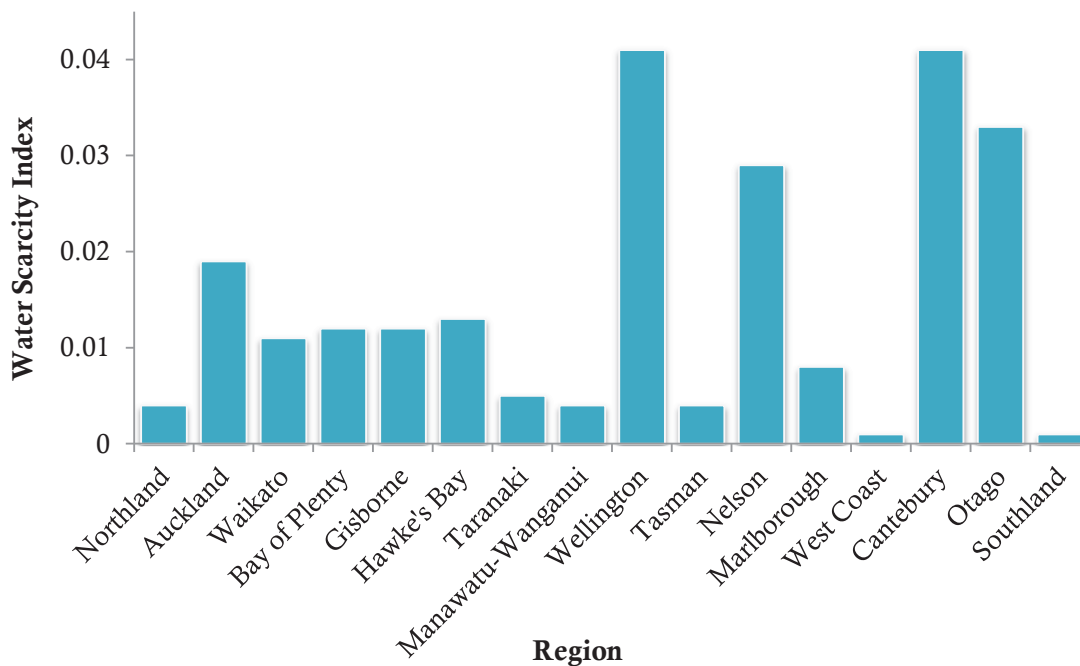


Figure 2.6 The Water Scarcity Index for all regions of New Zealand (Herath et al. 2011)

Water Stress Indicator

Finally, the Water Stress Indicator is the ratio of water withdrawal to the actual water resource available for human use (Smakhtin et al., 2004b). Unlike the Water Scarcity Index or the Falkenmark Water Stress Index, this method accounts for the environmental flow requirement (*EFR*) necessary for ecosystem functioning, and subtracts this value from the available water quantity. This provides a more balanced indication of the true water resource availability, as it accounts for the water necessary to sustain aquatic ecosystem functioning, as well as for human use. This is similar to the Freshwater Ecosystem Impact Indicator proposed by Milà i Canals et al. (2009), which indicates the impacts of blue water consumption on reducing the availability of water for ecosystem function and health. It has been noted that the quantification of the *EFR* is both location and time specific, and the *EFR* has been proposed and revised by several authors (Pastor, Ludwig, Biemans, Hoff, & Kabat, 2013; Richter, Davis, Apse, & Konrad, 2012; Smakhtin, Revenga, & Döll, 2004a).

According to Smakhtin et al. (2004b), the *EFR* required to maintain healthy aquatic ecosystems varies globally from 20-50% of mean annual river flow in a basin. Herath et al. (2011a) assigned an *EFR* of 30% for all regions of New Zealand. Categories of water stress according to the Water Stress Indicator method can be seen in Table 2.3.

Table 2.3 Water stress categories and thresholds, according to Smakhtin et al. (2004b).

WSI Rank	Definition
< 0.3	Slightly exploited
0.3 < WSI < 0.6	Moderately exploited
0.6 < WSI < 1.0	Heavily exploited
> 1	Over exploited

The estimated Water Stress Indicator values ranged from 0.002 in the Southland to 0.059 in Wellington (Figure 2.7). As such, all regions of New Zealand fall into the 'slightly exploited' category (Herath et al. 2011).

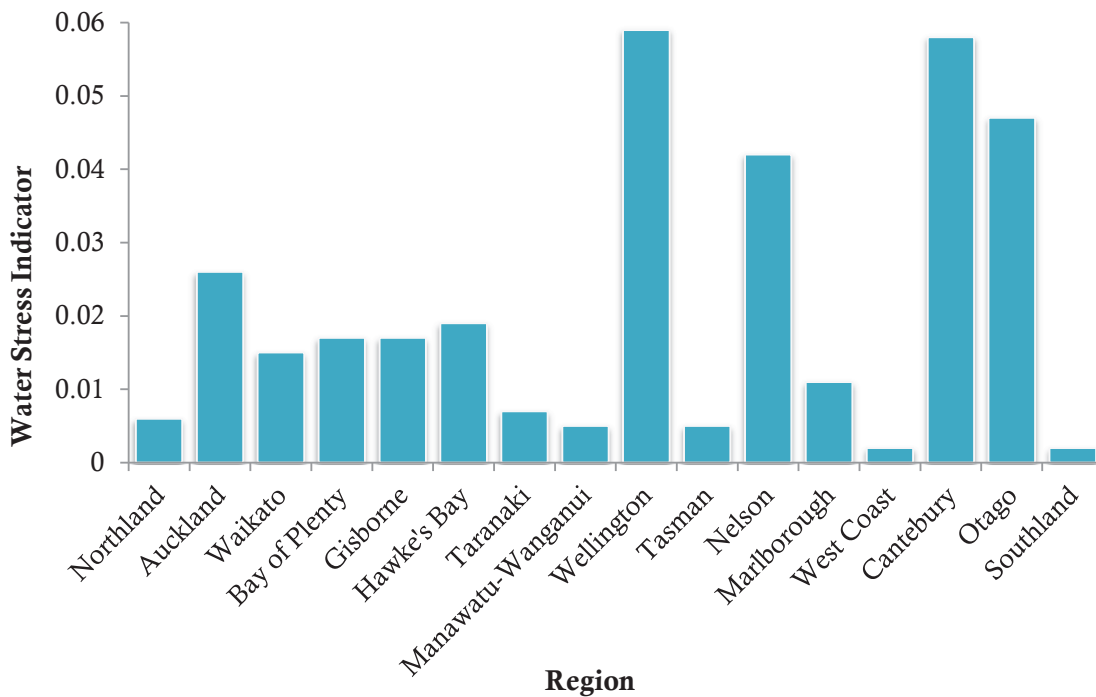


Figure 2.7 Water Stress Indicator for all regions of New Zealand (Herath et al. 2011).

According to the three methods assessed, no region of New Zealand was found to be over exploited or to experience high water stress, although some regions fell into the 'slightly exploited' category according to the Water Stress Indicator proposed by Smakhtin et al. (2004b). However, as the Water Stress Indicator has no category lower than 'slightly exploited' (Table 2.3) as with the Water Scarcity Index (Table 2.2), the Water Stress Indicator will always indicate at minimum this level of exploitation. During this particular study, the variation between water scarcity and water stress indicators results as a function of the chosen method was apparent. It can be seen that, for all regions of New Zealand, water stress is a more prominent issue than water scarcity, with water stress estimates permitting a more comprehensive indication of freshwater appropriation than water scarcity. It should be noted here that water stress is a precursor to water scarcity. In this case, the results from the Water Stress Indicator were considered more intuitively meaningful as this method captured the intricacies of water availability, hydrological and environmental considerations.

A limitation of the Herath et al. (2011a) study was the scale of the assessment. Whilst a regional level assessment provides a good overview of the current state of the nation's water scarcity/stress, a catchment level assessment would provide a more detailed and accurate account of specific areas of interest, and provide a more robust analysis. A further limitation as stipulated by Herath et al. (2011a) arises due to lack of data (for example, the volumes of water consumed by different users). Estimates of water scarcity for all three methods are based on annual water availability data; the results therefore do not account for seasonal stresses due to the temporal variations in water availability throughout the year. It is also important to note that none of the above methods account for water quality in their assessments of water scarcity/stress, and as previously mentioned this is often an important factor governing the utility of freshwater (Boulay et al., 2011b).

2.2 Life Cycle Assessment and Water Use

Life Cycle Assessment is an analytical environmental management tool, used to systematically evaluate the environmental performance and impacts of a product or a service throughout its entire life cycle. This extends from raw material acquisition, through to processing, use and disposal. LCA takes a "cradle to grave" approach (Nebel, 2006).

In 2006, the International Standardisation Organization (ISO) published *ISO 14040: Environmental Management- Life Cycle Assessment, Principles and Framework*, and later, *ISO 14044: Environmental Management- Life Cycle Assessment, Requirements and Guidelines*, to provide consistency and transparency amongst LCA practitioners worldwide (ISO, 2006). *ISO 14040* describes LCA as being capable of assisting with:

- Identification of opportunities to improve environmental performance throughout product lifecycles;
- Informing key stakeholders and decision makers (e.g. for purposes of strategic planning, priority setting, product or process design or redesign);
- The selection of relevant environmental performance indicators, and their measurement techniques; and

- Marketing (e.g. introduction of an eco-labelling scheme, environmental claims, or environmental product declarations).

For LCA practitioners, *ISO 14044* details the requirements necessary to conduct an LCA (ISO, 2006). LCA methods are typically undertaken in four distinct phases, as can be seen in Figure 2.8.

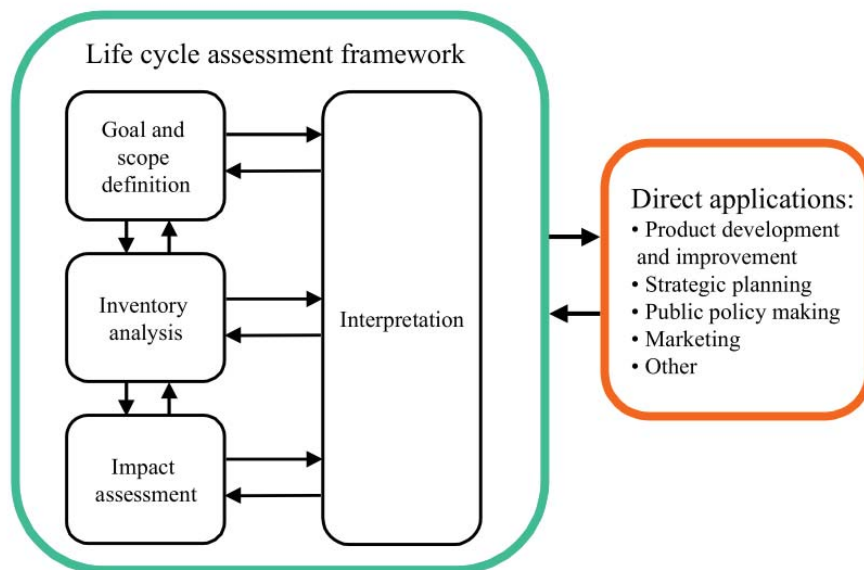


Figure 2.8 The LCA framework derived from *ISO 14040* (Nebel, 2006).

2.2.1 Goal and Scope

The goal and scope of LCA must be clearly stated, for example, a hotspot analysis or comparative analysis of two or more alternatives. During this phase, a functional (or declared) unit and system boundary need to be defined.

A functional unit refers to a quantified level of functionality or service that is provided. For example, a comparative assessment of different external wall constructions might use a functional unit of 1 m² of external wall over a 50 year time frame (Nebel, 2006). The system boundaries define what will be included in the analysis. Using the external wall example, the system of interest would typically include manufacture of materials needed in 1 m² of the wall, their transport and installation in the building, the

contribution the wall makes to thermal performance (through heat loss and heat gain), maintenance, repair and replacement during the building life, and demolition or deconstruction. Considering the potential for reuse, recycling or recovery at the building end of life.

2.2.2 Life Cycle Inventory

The Life Cycle Inventory (LCI) phase involves collecting relevant input and output data for processes within the defined system boundary. Data to be collected can include any emissions to land, air and water, as well as energy and mass flows (Ortiz, Castells, & Sonnemann, 2009). Ideally, the type, source and geographic location of the water is documented (Pfister et al., 2009). Further, some methods suggest in-stream⁹ and off-stream¹⁰ uses are to be distinguished (Owens, 2001; Pfister et al., 2009). It has been well documented that the use of localised data is more meaningful than national or globally averaged data (Hoekstra et al., 2011; Nebel, 2006, n.d; Pfister et al., 2009). This is particularly important if the results of the LCA are to be used for policy formation and decision support at the local level. The scale and quality of the data collected during this phase should be clearly defined. The LCI results in a list of quantified environmental inputs and outputs that comprise the defined functional unit.

2.2.3 Life Cycle Impact Assessment

The Life Cycle Impact Assessment (LCIA) phase aims to present environmental impacts calculated based on the LCI data, to facilitate understanding by key stakeholders, whilst meeting the requirements of the study. LCIA may result in calculation and reporting of different impacts separately e.g. climate change, resource depletion, eutrophication. It may also include normalisation and weighting, which result in aggregation of individual calculated impacts into one number, ranking or score. Impacts may be calculated at the “midpoint” or “endpoint” level.

⁹ In-stream use refers to water that is used in the natural context i.e. not removed from the water body. For example hydropower generation

¹⁰ Off-stream use refers to water withdrawals

Midpoint vs. Endpoint Impact Assessments

There are two major pathways for impact assessment: mid-point or end-point impact assessments (Figure 2.9 provides an overview of impact assessment pathways used in different impact assessment methods). Midpoints can be considered links in the cause-effect chain (environmental mechanism) of an impact category, prior to the endpoints. A midpoint assessment translates inputs and outputs calculated at inventory analysis into environmental impact categories; typical categories include climate change, stratospheric ozone depletion, eutrophication, and acidification. An endpoint assessment derives characterisation factors or indicators to reflect the impacts at the end of the cause-effect chain. Typical end-point impact categories are human health, ecosystems and resources. Typically an end-point assessment recognises impacts on a particular aspect (x) (depending on the study's goal and scope) (Bare, Hofstetter, Pennington, & Udo de Haes, 2000). Ortiz et al. (2009) refers to midpoint assessment as 'problem orientated' and endpoint assessment as 'damage orientated'.

During an industry meeting with experts in 2000, it was concluded that a common framework in which both midpoint and endpoint indicators were used was desired (Bare et al., 2000). The consensus gave rise to the creation of the ReCiPe method in 2008. The basic cause-effect chain of the ReCiPe method can be seen in Figure 2.9. The method accounts for 18 midpoint categories and just three endpoint categories (human health, ecosystem quality and resources) (Goedkoop et al., 2013). End-point impact assessments are generally regarded as having a higher level of uncertainty compared to midpoint results, but are easier to communicate and use for decision support.

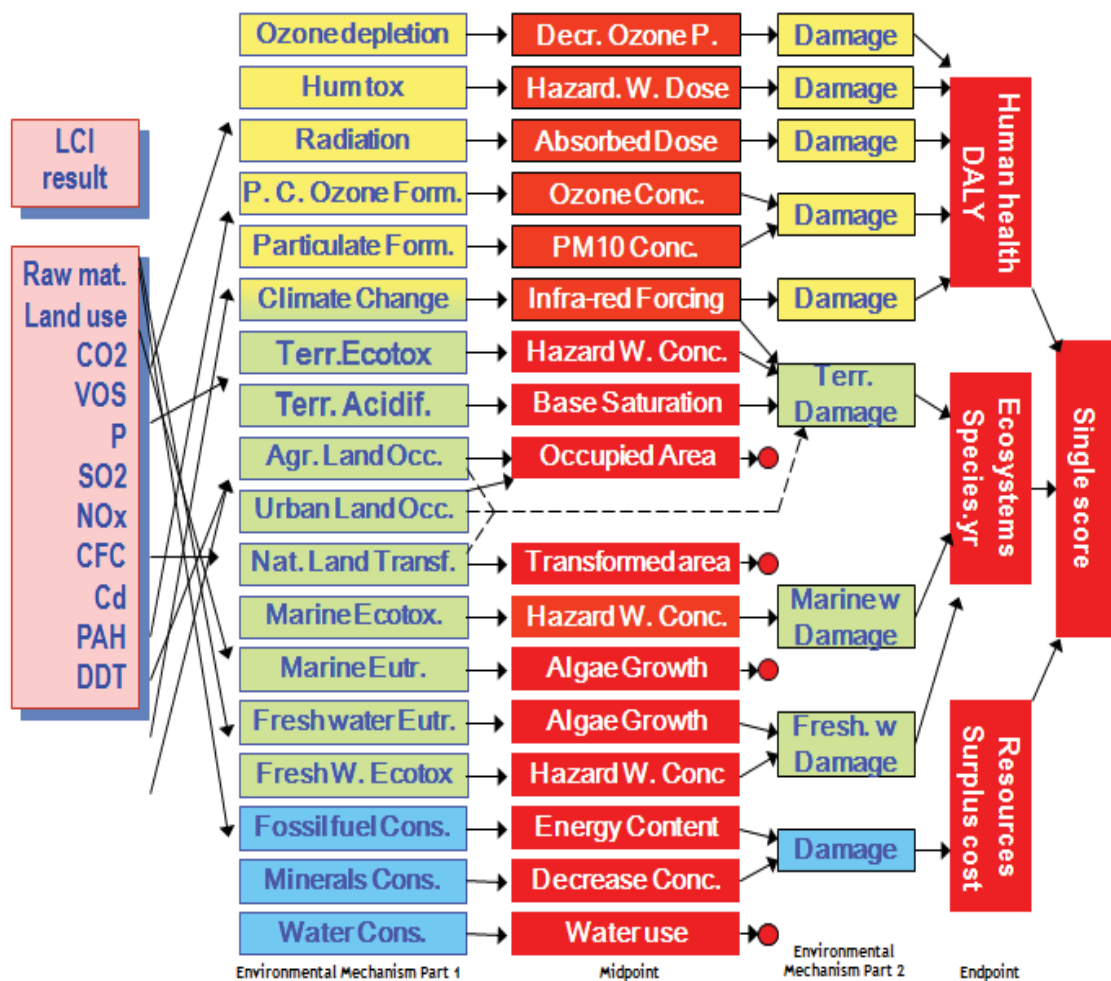


Figure 2.9 The relationship between LCI parameters (left), midpoint indicators (centre) and endpoint indicators (right) in the ReCiPe method (Goedkoop et al., 2013).

Whether the user chooses to conduct a midpoint or endpoint impact assessment will be a function of the goal and scope, and the method employed (Ortiz et al., 2009). Regardless of the pathway chosen, LCIA consists of three mandatory components (ISO, 2014):

1. Selection of impact categories
2. Assignment of LCI results (classifications)¹¹

¹¹ Classification of the LCI results involves assigning the emissions, wastes, and resources used, to the chosen impact categories

3. Modelling category indicators (characterisations)

At the user's discretion, the impact assessment may also include normalisation and weighting steps. Normalisation allows comparability of results with a known figure (e.g. global warming). Weighting, on the other hand, ranks the impact categories against each other in order to determine their importance (ISO, 2000).

2.2.4 Interpretation and Communication

The final stages of an LCA assessment are the interpretation and communication. This generally involves discussion of the results, and any opportunities to reduce the environmental impacts throughout the life cycle of the functional unit. If the results of an LCA study are to be published in any form, the assessment must comply with the *ISO 14040 & ISO 14044* standards, and gain third party review (Nebel, 2006). Additionally, this phase may include sensitivity analysis, scenario testing and consideration of the findings in the context of the goal of the study. Moreover, discussion surrounding any limitations and assumptions that have been made is disseminated during this phase.

2.2.5 Water and LCA

In the past, LCAs did not analyse the environmental impacts associated with water use. If included, water use was accounted for in terms of an inventory of the total water withdrawn (rather than consumed) throughout a product life cycle. It was therefore neither locally specific, nor contained any form of impact assessment (CEO Water Mandate, 2011).

In recent times, however, given the growing concern for water scarcity, LCA method has been developed and applied to account for water withdrawals *and* consumption. In 2014, the ISO published the first version of the *ISO 14046: Environmental Management- Water Footprint- Principles, Requirements and Guidelines* in order to support standardisation of LCA method for future use.

A water footprint assessment conducted according to *ISO 14046* is described as having the following benefits (ISO, 2014):

- It allows for identification and preparation of future water risks to water users;
- Identifies ways in which environmental impacts can be reduced;
- Improves water use efficiency at the process, product and organisation level;
- Shares knowledge and best practice with industry and government;
- Assists with meeting customer expectations of increased environmental responsibility, and
- Provides transparency, consistency, reproducibility and credibility for assessing and reporting water footprints.

Whilst water footprinting is still relatively new, the use of LCA case studies has contributed knowledge about the water consumption and impacts of various products, businesses and nations.

Table 2.4 gives a summary of water footprint studies conducted in New Zealand, along with the methods used. For the most part these have focused on agricultural products and processes. Examples include kiwi fruit production (Deurer, Green, Clothier, & Mowat, 2011; Landcare Research, 2011), the grey water footprint of potatoes (Herath et al., 2014a, 2014b), and the water footprint of dairy farming in the Canterbury and Waikato regions (Zonderland-Thomassen & Ledgard, 2012). Furthermore, Herath, Deurer, Horne, Singh, and Clothier (2011b) calculated the blue water footprint of hydroelectricity in New Zealand.

Table 2.4 Adoption of water footprint and impact assessment methods in New Zealand.

Product / Process	Method: Volumetric WF	Method: Impact Assessment
NZ kiwifruit water footprint (Landcare Research, 2011)	Hoekstra et al. (2011)	Hoekstra et al. (2011); Milà i Canals et al. (2009); Pfister et al. (2009)
NZ kiwifruit water footprint (Deurer et al., 2011)	Hoekstra et al. (2011)	Hydrological approach (Deurer et al., 2011)
NZ Hydroelectricity water footprint (Herath et al., 2011b)	Hoekstra et al. (2011)	Hydrological approach (Deurer et al., 2011; Herath et al., 2011b)
NZ wine water footprint (Herath et al., 2013a; Herath et al., 2013b)	Hoekstra et al. (2011) and Ridoutt and Pfister (2010)	Hydrological approach (Deurer et al., 2011; Herath et al., 2011b; Ridoutt & Pfister, 2010)
NZ dairy farming water footprint (Waikato and Canterbury) (Zonderland-Thomassen & Ledgard, 2012)	Hoekstra et al. (2011), Ridoutt and Pfister (2010) and Ridoutt et al. (2012)	Pfister et al. (2009)
NZ apples water footprint (Plant and Food Research, unpublished manuscript)	Hoekstra et al. (2011)	Hydrological approach (Deurer et al., 2011; Herath et al., 2011b)
NZ Potatoes grey water footprint (Herath et al., 2014a, 2014b)	Hoekstra et al. (2011)	Hydrological approach (Deurer et al., 2011; Herath et al., 2011b)

Although the focus of water footprint studies to date in New Zealand has primarily been the agricultural sector (responsible for approximately 70% of global freshwater consumption) (UN Water, 2007), in other parts of the world the consumption of water by industrial sectors has, in recent times, been receiving more attention (Pfister et al., 2011).

2.3 New Zealand's Building and Construction Industry

2.3.1 Background

The construction industry is the fifth largest economic sector in New Zealand, and plays an important role in the nation's economy. In 2011, the industry contributed 4% of overall gross domestic product (GDP), and had an annual export earning valued at NZD \$3 billion (PwC, 2011).

Figure 2.10 compares imported and exported construction materials during this period, where it can be seen that exported products far surpassed those imported, with Japan, Australia, China, and the USA amongst the largest markets for New Zealand's export produce (Dowdell, 2013).

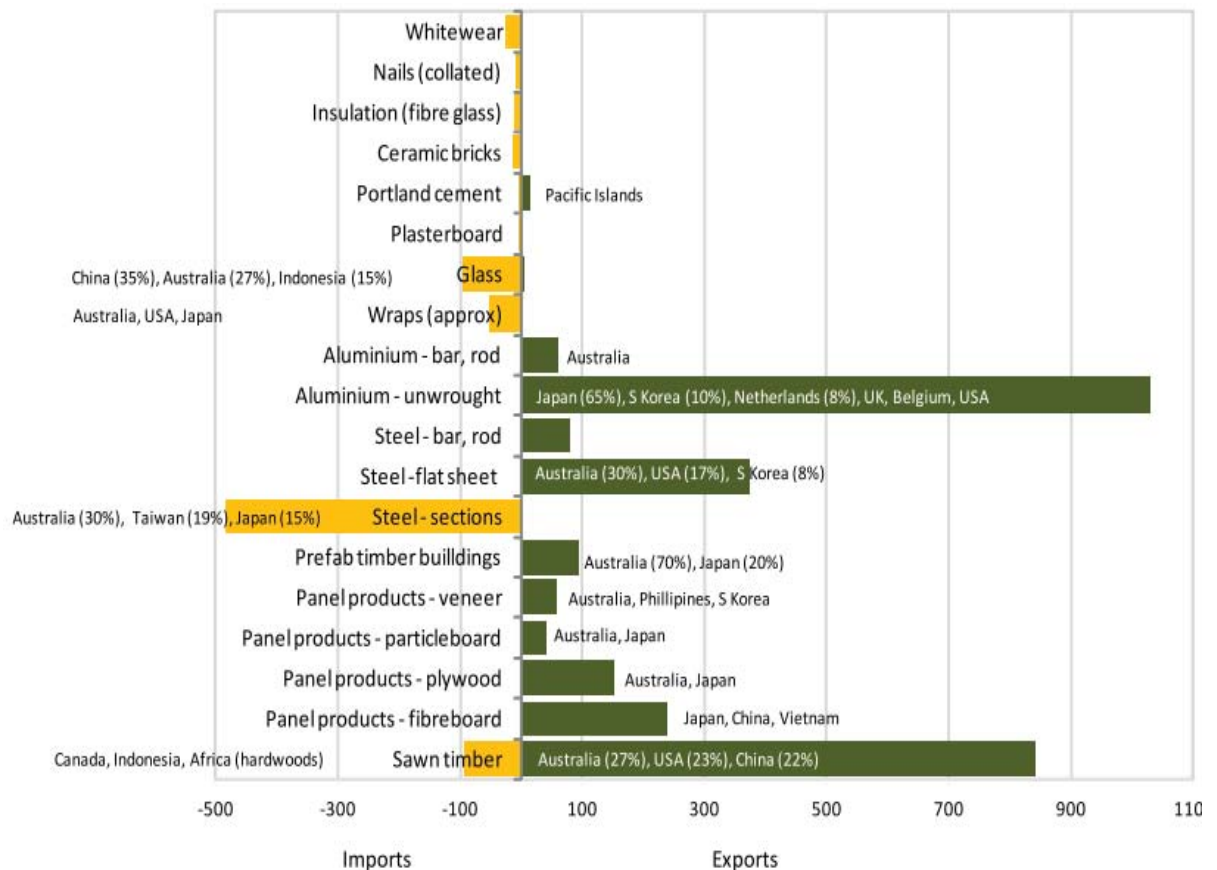


Figure 2.10 Export and import of selected building products in New Zealand for the 2011 period (Dowdell, 2013).

Investment in the construction industry is high; it is currently 45% of the national investment (PwC, 2011). The industry is less reliant on imports than other sectors producing significant shares of capital (PwC, 2011). Further, the industry is a major source of employment in New Zealand, accounting for one in 12 jobs. Over the last decade 14% of all new employment in New Zealand was within the construction sector, and as such the industry is an integral driver of economic growth (PwC, 2011).

2.3.2 Sustainability and LCA

Sustainability, within the building and construction sector, refers to the act of constructing and maintaining buildings in an economical way, so as to preserve environmental integrity and resources for present and future generations (Ortiz et al., 2009). Of the early sustainability initiatives within the sector, most focused on the embodied energy of building and construction products, as a result of the 1970s energy crisis in New Zealand (Jaques, McLaren, & Nebel, 2011). However, in more recent times, the focus has shifted to the 'whole-of-life' (or LCA) impacts of buildings and building products (Nebel, 2006). Many LCA tools have been developed for environmental assessments. These can be broadly separated into three levels with specific regard to the building and construction industry (Ortiz et al., 2009):

- Level 1: Whole building assessment framework or systems,
- Level 2: Whole building design decision or decision support tools, and
- Level 3: Product comparison tools.

By considering the wider environmental impacts of building and construction products, assessments become more environmentally representative and robust, allowing for more informed decision making (Jaques et al., 2011).

LCA studies have been conducted in New Zealand since the late 1990s, with the number of studies steadily increasing in recent times (Nebel, 2006). However, in comparison to the agricultural industry, the use of LCA within the building and construction sector is fairly under represented (Nebel, 2006) and this is particularly true for the use of water footprinting LCA methods within the sector. Nevertheless, the importance of good water stewardship within the building and construction sector has

been a focus of a number of large organisations in recent times. At the global scale, a United Nations Environment Programme (2012) survey of 72 companies, found water scarcity to be among the top three risks identified. In New Zealand, the Ministry of Business, Innovation and Employment (MBIE) stated in their 2007-2010 statement of intent, that the development of sustainability indicators (including water) for building and housing was amongst key projected outcomes.

In 2010, the Building Research Association of New Zealand (BRANZ) began to develop a 'Whole of Building Whole of Life Framework' (hereby referred to as 'the framework'). Informed by international standards (*ISO 21929-1* & *EN-15978*¹²) covering the application of LCA to the building and construction sector. The framework is to be used to assess the environmental performance of office buildings in New Zealand.¹³ Currently, such an approach is already available in Australia, Canada, Europe, Japan and the USA. Importantly, the framework aims to increase the availability of New Zealand-specific environmental performance data (Dowdell, 2013).

As well as environmental impacts, *EN 15804* also requires 'net freshwater use' to be reported for construction products (similar to both the *ISO 21929-1* and *EN-15978* which measure only net water consumption in terms of cubic meters). As an indicator, 'net freshwater use' neglects any temporal and spatial variations in extraction and discharge.

As part of the New Zealand Whole Building Whole of Life Framework research, a workshop was held with construction industry and other stakeholders in 2013 to discuss indicators relevant to New Zealand. During this workshop, water scarcity was identified for future inclusion. More specifically, how to measure water scarcity at the New Zealand catchment level using LCA methods was identified an area of further investigation. Current data regarding water scarcity in New Zealand is only available at the regional scale (refer section 2.1.4).

¹² Note: the framework will ultimately comply with these standards.

¹³ The method is said to be applicable with other building types e.g. schools and industrial buildings with some modification (Dowdell, 2013).

Chapter 3

Literature Review

Water footprinting is still an emerging subject area and different methods are being proposed, debated, and revised to define and standardise water footprint assessments. There are numerous proposed methods in the literature to calculate the water footprint and associated impacts of products and processes worldwide. Each considering different hydrological parameters and modelling choices. This chapter comprehensively and critically reviews these proposed methods, with a particular focus on the methods of Hoekstra et al. (2011), Pfister et al. (2009), Berger et al. (2014) and Boulay et al. (2016).

3.1 Water Footprint Methods

Water footprint methods have been the subject of much scientific debate in recent years. Water footprints can be calculated based on the *volume* of freshwater consumed and/or degraded or the subsequent *environmental impacts*. An overview of existing methods, distinguishing between chosen cause-effect pathways, can be seen in Figure 3.1.

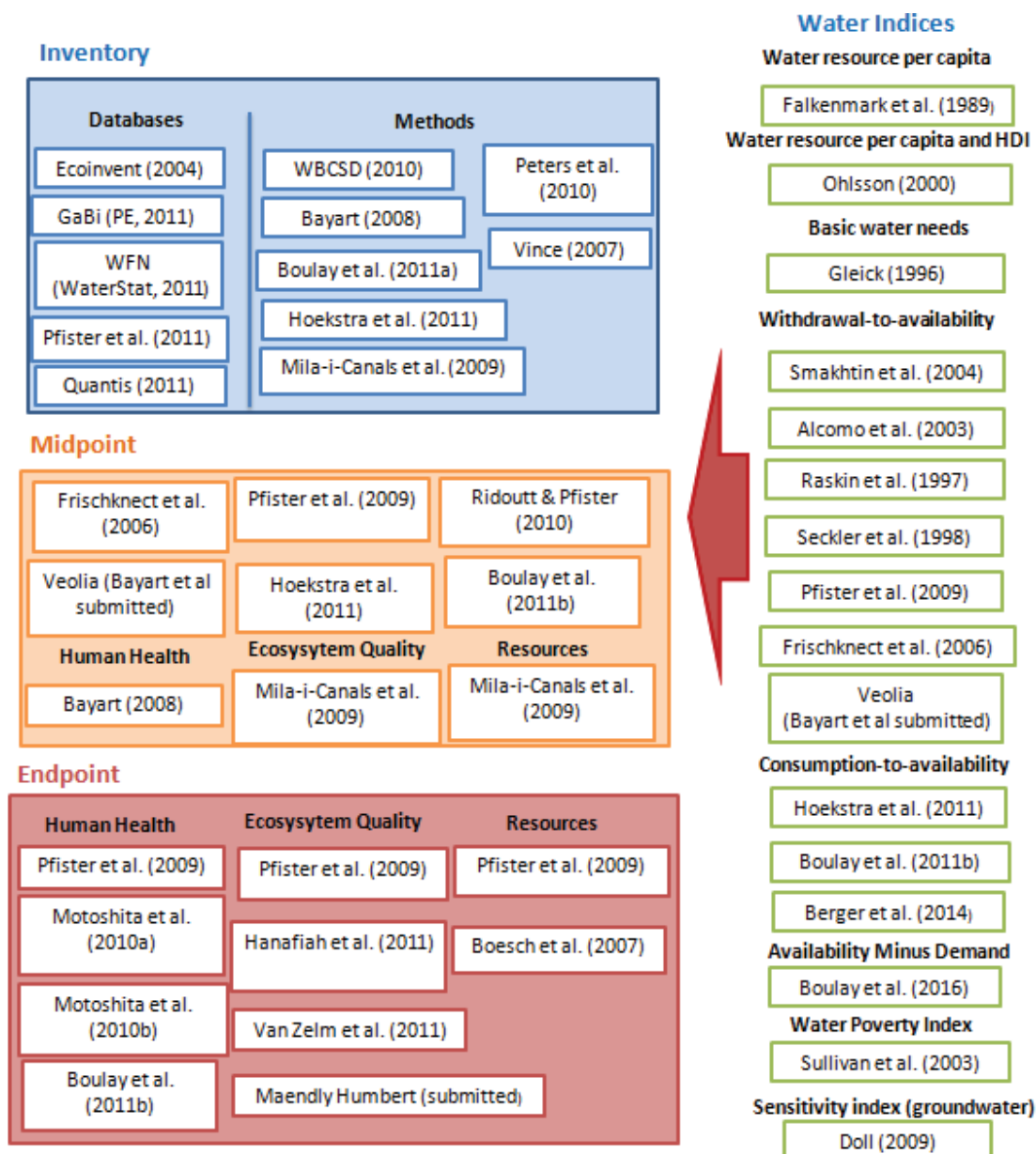


Figure 3.1 An overview of existing water footprint methods, adapted from Kounina et al. (2013).

3.2 Volumetric Water Footprinting Methods

Water footprints based on volumetric calculations determine the appropriation of freshwater for processes, products, and organisations. These methods can be described as taking a "resource efficiency" perspective by focusing on the total quantities of water

available, consumed and/or polluted during the production of a product (Berger & Finkbeiner, 2012).

3.2.1 The Water Footprint Network Method (Hoekstra et al., 2011)

The volumetric water footprint method (Hoekstra et al. 2011) produced by the Water Footprint Network (WFN) was the first comprehensive water footprint method to be developed. The WFN was initiated by a number of institutions in order to coordinate efforts to further develop and disseminate knowledge of water footprint concepts, methods, and tools, following the expansion of the virtual water concept by Allan (1998). In 2011, the WFN expanded upon their initial 2009 method to produce the *Water Footprint Assessment Manual: Setting the Global Standard* (Hoekstra et al. 2011). The WFN method consists of four key phases (Figure 3.2).

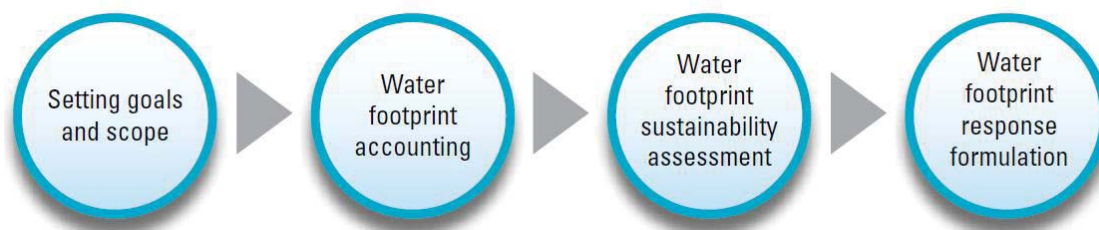


Figure 3.2 The four distinct phases of a water footprint assessment according to the WFN (Hoekstra et al., 2011).

Phase 1: Setting Goals and Scope

Water footprinting can be undertaken at a multitude of scales (global, national, regional etc.), and for a range of entities (products, processes, organisations etc.). Depending on the chosen scale, the scope of the study and any assumptions and considerations made will vary. It is therefore necessary to be explicit regarding the scope and inventory boundaries (what is to be included and excluded from the study). This ensures results are transparent to the wider water footprinting community (Hoekstra et al., 2011). Decisions to be made include:

1. The colours of water to be included (refer to Table 1.1);
2. Where to truncate the analysis along a supply chain;

3. Which period of data, and
4. The level of spatiotemporal explication (Table 3.1).

Table 3.1 Levels of spatiotemporal explication in water footprint accounting, according to Hoekstra et al. (2011).

Level	Spatial Explication	Temporal Explication	Source of Required Water Use Data	Typical use
Level A	Global average	Average annual or annual	Literature and global databases on typical water consumption and pollution by product or process	<ul style="list-style-type: none"> • Awareness-raising • Broad level identification of contributing factors to the overall water footprint • Development of global projections of water consumption
Level B	National, regional or catchment-specific	Annual or monthly	As for level A but with national, regional or catchment specific data	<ul style="list-style-type: none"> • Broad level identification of spatial variation • Hotspot identification • Water allocation decisions
Level C	Small catchment or field-specific	Monthly or daily	Empirical data or best estimates on water consumption and pollution, temporally and spatially specified	<ul style="list-style-type: none"> • Knowledge base for water footprint sustainability assessment • Strategy formulation

Phase 2: Water Footprint Accounting

The accounting phase of the WFN's assessment refers to the compilation of relevant data used to answer the question "What is the actual appropriation of flow within a defined period?" (Hoekstra et al., 2011). It is a multidimensional indicator showing volumes of consumed water by source and volumes of polluted water by type, whilst accounting for direct and indirect water use (Hoekstra et al., 2011).

As the volume of water withdrawn is not the same as the volume of water consumed, consumptive water use as defined by the WFN refers to any one of the following cases:

1. Water is evaporated¹⁴;
2. Water is incorporated into a product;
3. Water is not returned to the same catchment as it was withdrawn, or is returned to the sea; or
4. Water does not return in the same period, for example, it is withdrawn in a water scarce period and returned in a wet period.

In order to calculate the water footprint of a product, the water footprints of the predominant processes (defined during the initial phase) are calculated as follows:

Equation 1

$$WF_{proc} = \text{Water Evaporation} + \text{Water Incorporation} + \text{Lost Return Flow [volume/time]}$$

The results of each individual process step are aggregated to provide a product's volumetric consumptive water footprint (Hoekstra et al., 2011). Although the WFN method distinguishes between the different colours of water consumed during each process step, when calculating the final water footprint, colours could be aggregated to give a single value (Hoekstra et al., 2011). This has garnered some criticism amongst the water footprinting community, with Berger and Finkbeiner (2012) describing the aggregated volumetric water footprint as "lacking scientific rationale". In addition, Ridoutt and Pfister (2010) state that the volumetric water footprint method is incomplete and misleading (refer section 3.4.1). It is important to note though, that the water footprints of final products¹⁵ calculated using this method *are* comparable.

Despite the criticisms surrounding the use of an aggregated volumetric water footprint value, industry support for the WFN method is high. In 2010, the WFN's partners reached 130 spanning a multitude of countries and sectors worldwide (including large corporations and institutions). Case studies using the WFN method are numerous, and include: animal products, crops, energy, and entire nations, refer to the WFN's Water

¹⁴ Includes production related evaporation e.g. evapotranspiration for primary production, evaporation from storage ponds, or during transport, processing and disposal.

¹⁵ Also applies to the final water footprint values for communities, businesses, nations, and catchments etc.

Footprint Statistics (Water Footprint Network, 2016). Specific examples of where the WFN method has been adopted in New Zealand can be seen in Table 2.4.

Phase 3: Water Footprint Sustainability Assessment

In essence, the sustainability phase of the WFN assessment, is a comparison of the calculated volumetric water footprint to the available freshwater resource (Hoekstra et al., 2011). Consideration is given to the three sustainability dimensions (environmental, social, and economic), level of impact (primary or secondary impacts), and water colours (blue, green and grey). The sustainability assessment can be conducted by one of two pathways depending on the desired outcome and scope of the study;

- I. Hotspot identification
- II. Aggregated water footprint impact indices

I. Hotspot Identification

A hotspot refers to a catchment (and a period in time), where the water footprint of a product or process, for example, is not sustainable and must be reduced. An unsustainable water footprint occurs if environmental, social or economic wellbeing are compromised. Hotspots are typified by water scarcity, pollution or conflict (Hoekstra et al., 2011). Whilst hotspot identification can identify where the largest impacts are, it is useful in some instances, such as LCA, to summarise this information into one index or a few indices to facilitate comparison and evaluation among different products or processes.

II. Aggregated Water Footprint Impact Indices

Hoekstra et al. (2011), note that the blue, green and grey water footprints of a product calculated using the WFN approach are directly transferable as indicators in LCA (as they measure the volume of freshwater appropriated). Nevertheless, the WFN method has been further developed to assess the local *environmental impacts* associated with appropriation of the freshwater resource. The local environmental impact depends on local water scarcity (for blue and green water) and/or pollution level (grey water) in the catchment within which the water footprint is calculated (Hoekstra et al., 2011).

The WFN have incorporated an impact assessment in the form of three water footprint impact indices:

1. **Blue** water footprint impact index,
2. **Green** water footprint impact index, and
3. **Grey** water footprint impact index.

To determine the contribution of a product to the total water footprint in a specified boundary, the localised impacts of that specific product depend on two matrices. Table 3.2 indicates the relevant matrices depending on the colour(s) of water under assessment (Hoekstra et al., 2011).

Table 3.2 Matrices used to determine the localised impact of a water footprint depending on water colour

	Blue	Green	Grey
How large the specific water footprint is specified by boundary (x) and month (t)	✓	✓	✓
What the <u>water scarcity</u> is, specified by boundary (x) and month (t)	✓	✓	✗
What the <u>water pollution level</u> is, specified by boundary (x) and month (t)	✗	✗	✓

In the case of blue water, the blue water scarcity of a catchment (or specified boundary) is defined as the ratio of the total blue water footprints in the catchment (WF_{blue}) to the blue water availability (WA_{blue}) in that catchment. Where the blue water footprint of a catchment is defined as the sum of the blue water consumed by *all* water using processes in the area:

Equation 2

$$WF_{blue} = \sum_q WF_{proc,blue} [q] \text{ [volume/time]}$$

Where, $WF_{proc,blue}[q]$ refers to the blue water footprint of a process q in the catchment¹⁶. The blue water availability (WA_{blue}) in each catchment (x) is defined as the natural runoff in the catchment (R_{nat}) minus an environmental flow requirement (EFR) over time (t):

Equation 3

$$WA_{blue}[x,t] = R_{nat}[x,t] - EFR[x,t] \quad [\text{volume/time}]$$

Subsequently the blue water scarcity (WS_{blue}) in a catchment is calculated, as follows:

Equation 4

$$WS_{blue}[x,t] = \frac{\sum WF_{blue}[x,t]}{WA_{blue}[x,t]} \quad [-]$$

A water scarcity of 100% indicates that the available water has been fully consumed. Both the blue and green water scarcity indicators, and the grey water pollution level have been designed to indicate unsustainable conditions when they exceed 100%. In terms of seasonal variability, Hoekstra et al. (2011) recommend calculating water scarcity at the monthly scale for a more accurate representation of temporal changes in water scarcity. Assessment at the annual scale should be conducted with caution, as this does not allow for differentiation between the drier months, when freshwater resources are least replenished naturally, and water scarcity will have its greatest environmental impacts.

The local water footprint impact indices ($WFII$ s) are obtained by multiplying elements of the two matrices (water footprint and water scarcity in the case of the blue and green water footprint indices, and water footprint and water pollution level in the case of the grey water footprint impact index) (refer Table 3.2). The calculated $WFII$ s can be interpreted as the aggregated and weighted measure of the environmental impacts of the water footprints, in the places and periods in which the various water footprint components occur. The blue water footprint impact index ($WFII_{blue}$), for example, is calculated as follows:

¹⁶ In the case of inter-basin transfers the volume of water is accounted for in the region from which the water is exported (Hoekstra et al., 2011).

Equation 5

$$WFII_{blue} = \sum x \sum t (WF_{blue}[x, t] \times WS_{blue}[x, t]) [-]$$

Separately, the three water footprint impact assessments (blue, green and grey) refer to different things and are not directly comparable. However, the three are able to be aggregated to give a single water footprint impact index (Hoekstra et al., 2011). By aggregating the water footprint impact indices, a broad impression of the environmental impact of a water footprint can be gained. This is useful if the overall aim of the study is to compare water footprint impacts from one area with those of another area (or one product from another product). It is less useful, however, if the aim of the study is to identify specific processes that have higher water consumption and apply response strategies (as it disregards all underlying temporal and spatial information important for smaller scale analysis). If this is the desired outcome a hotspot analysis would be more favourable (Hoekstra et al., 2011).

Unlike hotspot identification analysis, the aggregated water footprint impact index is able to be incorporated into LCA studies (Hoekstra et al., 2011). It is important to note that the blue, green and grey water footprint impact indices as described by Hoekstra et al. (2011), are concerned with the local environmental impacts only (no consideration is given for the social and economic impacts as with some LCA methods).

Phase 4: Water Footprint Response Formulation

Finally, as water footprint results are made available, response options, strategies and policies are developed (Hoekstra et al., 2011). This thesis will not focus on response options and strategies; however for reference the *Water Footprint Assessment Manual: Setting the Global Standard* (Hoekstra et al. 2011) provides an extensive inventory of options for various stakeholders. Thus, reflecting the need for water response strategies to be locally orientated, as opposed to a 'one size fits all' approach.

3.3 Impact Orientated Water Footprint Methods

Water footprint methods that are concerned with the environmental, social and economical impacts associated with water consumption, generally take an LCA approach (refer to section 2.2). They reflect the fact that the environmental impacts of

a litre of water used in one region may have very different implications than in another region, depending on local water scarcity levels (Ridoutt, Sanguansri, Nolan, & Marks, 2012). LCA methods involve the characterisation of individual water flows prior to aggregation into a water footprint indicator(s) (Berger & Finkbeiner, 2012).

In some impact-orientated water footprint methods, assessments may be based on the functionality of the affected water to different users. One such LCA method that accounts for impacts on human health related to water consumption and degradation is that proposed by Boulay et al. (2011a). The method assesses the extent to which a user's withdrawn and released water changes in terms of its functionality, based on key characteristics such as: water quality, quantity, type of resource (ground or surface water), and geographical location. Water is considered functional if it can meet user needs without generating adverse effects or a change in practice (Boulay et al., 2011a). A loss of water functionality is caused by consumption (i.e. water becomes unavailable for use in the same watershed, is integrated within a product, or is evaporated) or degradation (i.e. water is too contaminated to be used for a specific function) (Bayart et al., 2010). This is important as malnutrition and disease may occur if water availability and quality is reduced.

There are a range of different methods in the literature concerned with calculating the impacts of water withdrawals and consumption (refer to Figure 3.1). The LCA methods of Pfister et al. (2009), Berger et al. (2014) and Boulay et al. (2016) have been chosen for the following analysis.

3.3.1 The Water Stress Index Method (Pfister et al., 2009)

The Water Stress Index (*WSI*) builds upon the *Water Resource Vulnerability Index* concept, commonly referred to as the water-to-availability (*WTA*) ratio (Raskin, Gleick, Kirshen, Pontius, & Strzepek, 1997). The *WTA* concept can be defined as the ratio of total annual withdrawal ($\sum WU_{ij}$) per user category (*j*) and per basin(*i*), to water availability (WA_i) within a basin, as follows:

Equation 6

$$WTA_i = \frac{\sum WU_{ij}}{WA_i} [-]$$

According to the *WTA* method, a country is considered water scarce if the annual water withdrawal is between 20% and 40% of annual water availability. Severe water scarcity occurs at over 40% of water withdrawals (Raskin et al., 1997). This method and the proposed 40% threshold value are commonly used and have been termed the "criticality ratio" by some, referring to the ratio of water withdrawals for human use to total renewable water resources (Alcamo, Henrichs, & Rösch, 2000).

The use of the *WTA* (Pfister et al. 2009) ratio has been criticised in the literature. According to Berger & Finkbeiner (2012), the two principle components, freshwater withdrawal and freshwater availability, can lead to deceptive effects by overestimating physical water scarcity. As the *WTA* comprises freshwater 'borrowing' (e.g. water for cooling) and degradative water use. Furthermore, the *WTA* does not consider the sensitivity of a watershed to water scarcity, resulting from additional water use/consumption. For example, catchments with small renewable water supplies are more sensitive to additional water use than basins with large renewable resources (Berger & Finkbeiner, 2012). When the goal of an LCA is to assess the effects to ecosystem and human health, it is the opinion of Berger and Finkbeiner (2013) that surface water stocks need be included in the analysis, as they can compensate for over-use of the renewable supply.

If water storage is insufficient, it is stressed by Pfister et al. (2009) that any variability in monthly or annual precipitation data, may lead to increased water stress during specific periods of the year. Similarly, if evaporation is high, (and thus the effects of increased water stress cannot be fully compensated by periods of low water stress) there will be increased water stress. To this end, Pfister et al. (2009) adapted the *WTA* ratio by introducing a variation factor (*VF*) to correct for increased water stress (calculating a modified *WTA**). Differentiation is also made for catchments with strongly regulated flows (*SRF's*) as identified by Nilsson, Reidy, Dynesius, and Revenga (2005). Three classes of *SRF's* were constructed using fragmentation and flow

regulation characteristics (Nilsson et al., 2005). Table 3.3 indicates the New Zealand rivers identified as having *SRF*'s according to Nilsson et al. (2005).

Table 3.3 The degree of flow modification in New Zealand rivers, according to Nilsson et al. (2005).

Degree of Flow Modification	River	Region(s)
Strongly Affected Flows	Waikato	Waikato
	Waitaki	Canterbury and Otago
	Clutha	Otago
Moderately Affected Flows	Waiiau	Canterbury
Not Affected	Grey	West Coast
	Buller	West Coast

The *VF*'s for rivers with *SRF*'s are defined as the aggregated measure of dispersion of the multiplicative standard deviation of monthly (s_{*month}) and annual (s_{*year}) precipitation (assuming a log normal distribution):

Equation 7

$$VF = e^{\sqrt{\ln(s_{*month})^2 + \ln(s_{*year})^2}} \quad [-]$$

The application of the *VF* and calculation of the modified *WTA** is dependent on the status of each respective river system (refer to Table 3.3). If the river in focus has a strongly regulated flow the modified *WTA* (*WTA* *) is calculated as the square root of the *VF* multiplied by the *WTA*. In the case of non-*SRF*'s the *VF* is simply multiplied by the *WTA*.

Equation 8

$$WTA^* = \sqrt{VF} \times WTA \text{ (for } SRF's)$$

$$WTA^* = VF \times WTA \text{ (for non } SRF's)$$

The traditional *WTA* ratio is further progressed by Pfister et al. (2009) through the development of the *WSI*. The *WSI* indicates the proportion of water that deprives other users of freshwater. A *WSI* of above 0.5 indicates the threshold between moderate and severe water scarcity.

Equation 9

$$WSI = \frac{1}{1 + e^{-6.4 \times WTA^* \left(\frac{1}{0.01} - 1\right)}} [-]$$

The *WSI* can be used as a characterisation factor for the mid-point category "water deprivation" in LCIA. It is also able to be incorporated into either a mid-point *or* an end-point LCIA (refer section 2.2.3). Damages to three key areas of protection are considered: human health, ecosystem quality and resources. Damage assessment is performed in accordance with the Eco-Indicator-99 method (Goedkoop & Spriensma, 2001). For both pathways, data from the 'virtual water database'¹⁷ produced by Chapagain and Hoekstra (2004) is used in order to produce a regionalised inventory for water consumption (Pfister et al. 2009).

However, unlike the volumetric water footprint method (Hoekstra, Chapagain, et al., 2009; Hoekstra et al., 2011), the framework proposed by Pfister et al. (2009) only accounts for blue water consumption¹⁸, and omits green water consumption and its effects on blue water recharges¹⁹. Grey water may or may not be included (see later additions below). Whilst the virtual water database is convenient for estimating the amount of virtual water used in agricultural products, it is criticised by Pfister et al. (2009) due to a lack of spatial resolution and reliability. It is noted that the database considers only a few industrial products and that water use within the supply chain is neglected (Pfister et al., 2009).

As per the definition of the *WTA* ratio, thresholds above 20-40% are considered moderately to severely water stressed (figures may reach as high 60%). The *WSI* is

¹⁷ The database contains the virtual water content of numerous crop products in most countries.

¹⁸ Consumptive water use as described by Pfister et al. (2009) is consistent with the definition provided by Hoekstra et al. (2011).

¹⁹ Although this is remarked as an potential area of further research.

logarithmic, and is tuned to result in a *WSI* of 0.5 for a *WTA* of 40% (the threshold between moderate and severe water stress). The minimum water stress that can be calculated using the *WSI* is 0.01 (maximum is 1), as it is regarded that all consumption will have a marginal local impact.

Although the *WSI* only accounts for water consumption (not degradation), additional assessment of degradative water use is suggested because water quality loss can be seen as essentially a loss in water quantity due to the functionality of the freshwater resource decreasing (Pfister et al., 2009). As previously mentioned, an approach for assessing water degradation has been developed (Boulay et al., 2011a, 2011b).

The *WSI* has been calculated for approximately 10,000 watersheds (Figure 3.3) using the WaterGAP2 global model, with modifications²⁰. The model incorporates both socio-economic²¹ and hydrological²² data in order to quantify the *WSI* values for different watersheds around the world (Pfister et al., 2009). Due to the global nature of the dataset, modelled results have a level of uncertainty associated with them. Although *WSI* values for the whole world have been calculated, it is also theoretically possible to calculate compatible site-specific values using local hydrological data (higher spatial resolution) and the same formula described in Pfister et al. (2009).

²⁰ Note: the traditional *WTA* ratio is amended for the *WSI* calculation to account for the monthly and annual variability of precipitation, and to correct for watersheds with strongly regulated flows (*SRF*'s)

²¹ E.g. water withdrawal for human activities, degree of flow regulations

²² E.g. water availability in different watersheds, monthly and annual variability of precipitation

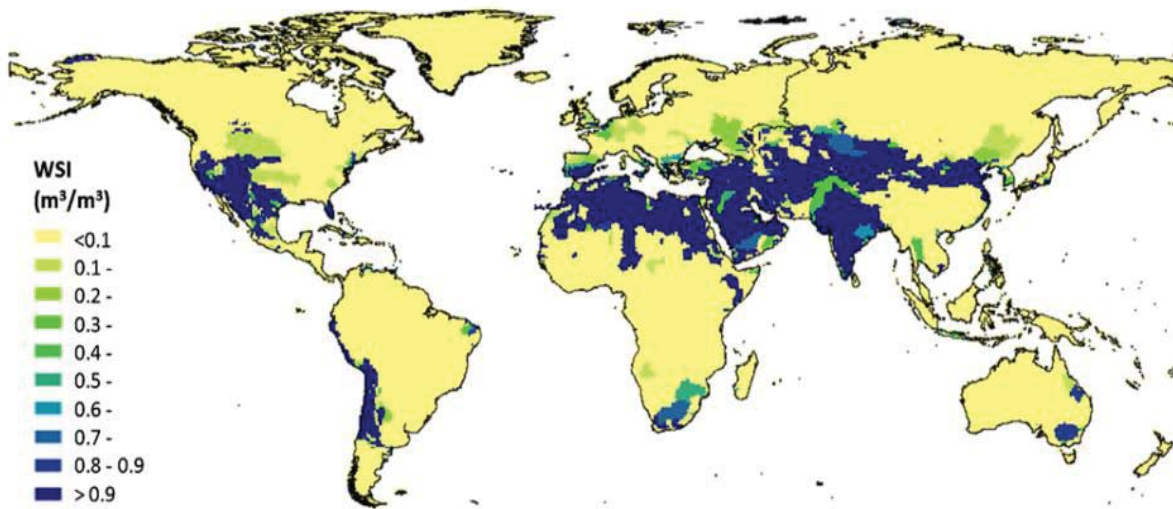


Figure 3.3 A global representation of the Water Stress Index, as developed by Pfister et al. (2009).

It is recommended that the *WSI* be calculated at the country or major watershed level, although it is asserted that the watershed level is the most appropriate, as hydrological processes are connected by watersheds (Pfister et al., 2009). Unfortunately, in many cases, LCI data are derived from global averages and estimations, further highlighting the need for improved regional data when conducting LCA.

As with the volumetric water footprint method of Hoekstra et al. (2011), the *WSI* method has been well used throughout the literature. Case studies include condiments, confectionary and meat products (Ridoutt & Pfister, 2010; Ridoutt et al., 2012). Since its publication, the method of Pfister et al. (2009) has been further updated as follows:

- Ridoutt and Pfister (2010) adapted the work of Pfister et al. (2009) to calculate a stress-weighted water footprint. The method seeks to understand freshwater consumption (blue water only) in relation to the degree of water scarcity at source. The stress-weighted water footprint is achieved by multiplying the blue water footprint by the regional *WSI*.
- Ridoutt and Pfister (2013) assessed the consumptive *and* degradative use of blue water taking into consideration the local *WSI* value relative to a global average

WSI value of 0.602 (Ridoutt, Sanguansri, Nolan & Marks, 2012). The characterisation factor used for consumptive water use, for each spatially differentiated instance, was the locally relevant *WSI* value divided by this global average *WSI* value. The ReCiPe endpoint method was used. The method presented here is useful only for comparisons as it references a water footprint to the global average *WSI*, therefore results are not appropriate at the local level.

3.3.2 Water Accounting and Vulnerability Evaluation Model (Berger et al., 2014)

The Water Accounting and Vulnerability Evaluation (WAVE) model was proposed in response to a number of criticisms in the existing water footprinting literature (Berger et al., 2014). As with all LCAs, the method comprises an LCI and a LCIA phase in the form of a water accounting model and impact assessment, respectively. However, the WAVE method is a departure from traditional impact assessments in LCA as it does not predict the impact of water consumption on the three endpoints (human health, ecosystem quality and resources). Instead, the method evaluates the vulnerability of basins to freshwater depletion based on a new midpoint indicator, the water depletion index (*WDI*).

It is the authors' opinion that existing volumetric (Hoekstra et al., 2011) and impact-orientated (Boulay et al., 2016; Pfister et al., 2009) methods have shortcomings. Included in these, is the definition of freshwater consumption. In alignment with Hoekstra et al. (2011) freshwater consumption is defined as the fraction of total water use (withdrawal) that does not return to the originating drainage basin. This can be due to integration in products, discharge into other basins (or the sea), or loss via evapotranspiration. This definition of freshwater consumption used by both the volumetric and impact-orientated water footprint methods of Hoekstra et al. (2011) and Pfister et al. (2009) is criticised by Berger and Finkbeiner (2012). It is the view of the authors that the inclusion of evaporation as a consumptive water use is incorrect. Berger et al. (2014) argue that evaporated water is often returned to the same drainage basin in a relatively short time frame as precipitation, and therefore this is more accurately described as water *use* not consumption. Similarly, water vapour that is

created synthetically can return to its originating basin and contribute to water availability.

Berger and Finkbeiner (2012) state the assumption that evapo(transpi)rated water is lost is based on the fact that 70% of all precipitation occurs over the ocean (New, Todd, Hulme, & Jones, 2001). They argue that when the fate of evaporation is analysed that, on average, 57% of terrestrial evaporation returns as precipitation over land (Van der Ent, Savenije, Schaefli, & Steele-Dunne, 2010). Thus its inclusion as a consumptive water use is incorrect as water consumption in one river basin may lead to precipitation in the same or another basin. Thus the definition of freshwater 'consumption' as used in the WAVE method differs from the aforementioned definition adopted by Hoekstra et al. (2011) and Pfister et al. (2009).

Berger et al. (2014) also criticised the use of the *WTA* ratio (as used in Pfister et al., 2009), arguing that not all freshwater withdrawals are consumptive. For example, industrial cooling processes often return freshwater to the original basin almost immediately. Therefore, accounting for this withdrawal can lead to overestimation of the water scarcity (particularly in industrial regions). Furthermore, the term "availability" in the *WTA* context refers only to the run-off fraction (water flowing in rivers and streams), and thus neglects groundwater and surface water stocks. Lakes and aquifers in particular, are an important consideration when assessing impacts as they can act to buffer temporal water scarcity. Finally, the *WTA* ratio is tuned to become zero when water withdrawal is zero, as it assesses relative water stress only (neglecting absolute water shortage). As a result, arid and semi-arid regions appear uncritical when water withdrawal is close to zero (as a result of low population or industry). Consequently, it is argued that true water scarcity at the local level is not reflected in current indicators (Berger et al., 2014). The WAVE model aims to address the aforementioned criticisms through the creation of the *WDI*.

Phase One: The Water Accounting Model

At the water accounting level, a new LCI method is introduced. In addition to freshwater withdrawals and wastewater discharges considered in other LCA methods,

evaporation recycling is considered for the first time in the WAVE model. As opposed to just evaporative losses considered by both Hoekstra et al. (2011) and Pfister et al. (2009).

Atmospheric evaporation recycling has been found to be an important consideration for water accounting studies. As an example, moisture evaporating from the Eurasian continent has been determined to be responsible for 80% of China's freshwater resource (Van der Ent et al., 2010). Inclusion of this parameter is thought to allow a more accurate calculation of water consumption figures. It is reported that evaporation recycling may account for 32% of calculated water consumption volumes within drainage basins (Van der Ent & Savenije, 2011; Van der Ent et al., 2010). Moreover, consideration is given to synthetically created water vapour²³. Figure 3.4 shows the systematic overview of water inventory flows required by the WAVE method.

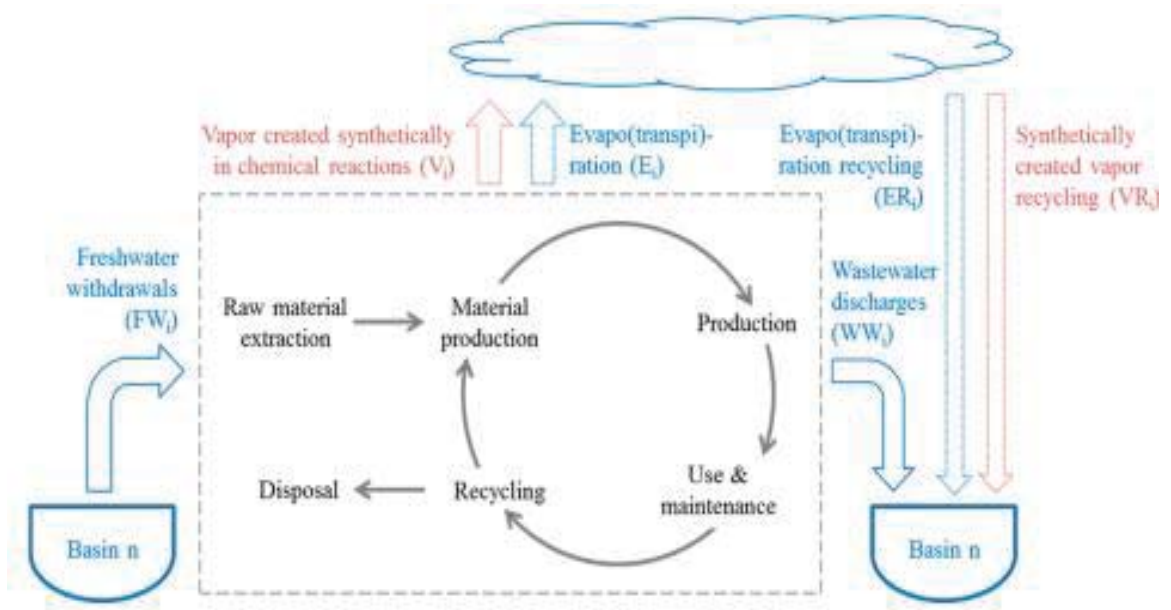


Figure 3.4 Water inventory flows considered along the lifecycle of a product according to the WAVE model (Berger et al., 2014).

²³ Defined as vapour created synthetically chemical reactions e.g. burning fossil fuels (Berger et al., 2014)

In order to calculate the effect of atmospheric evaporation and synthetic vapour recycling, an effective water consumption (WC_{eff}) component is introduced, in order to calculate the risk of freshwater depletion (RFD)²⁴, as follows:

Equation 10

$$RDF = \sum WC_{eff,n} * WDI_n$$

The WC_{eff} is defined as the sum of the total effective water consumption in each basin(n). In order to calculate $WC_{eff,n}$ the total wastewater discharges (WW_i), evapo(transpi)ration recycling (ER_i), and synthetically created vapour recycling (VR_i) are subtracted from the freshwater withdrawals (FW_i) occurring within the basin (n) as follows:

Equation 11

$$WC_{eff,n} = \sum (FW_i - WW_i - ER_i - VR_i) \quad [-]$$

Note: The WAVE method accounts for consumptive blue water only.

Atmospheric evaporation recycling is dependent on the degree to which terrestrial evaporation supports the occurrence of precipitation within a region (e.g. the regional moisture recycling). This is a function of the scale and shape-dependence of the regional moisture recycling ratios. The Basin Internal Evaporation Recycling ($BIER$) ratio was developed through the work of Van der Ent et al. (2010) and Van der Ent and Savenije (2011) to quantify this relationship. $BIER$ is estimated based on the length of the basin in the direction of the main moisture flux (x) and the average local length scale of the evaporation recycling process (λ).

Equation 12

$$BIER = \frac{-x - \lambda \cdot \exp\left(\frac{-x}{-\lambda}\right) + \lambda}{-x} \quad [-]$$

²⁴ Note: the RFD is a separate calculation to the WDI characterisation factor, which is the focus of this thesis.

The *BIER* ratio has been calculated for over 11,000 basins on the global scale (Figure 3.5). Evaporation recycling increases with distance, consequently larger drainage basins have higher *BIER* values than smaller basins (Van der Ent et al., 2010). Globally, *BIER* varies from 0% in the Sahel zone to 38% in the Congo basin.

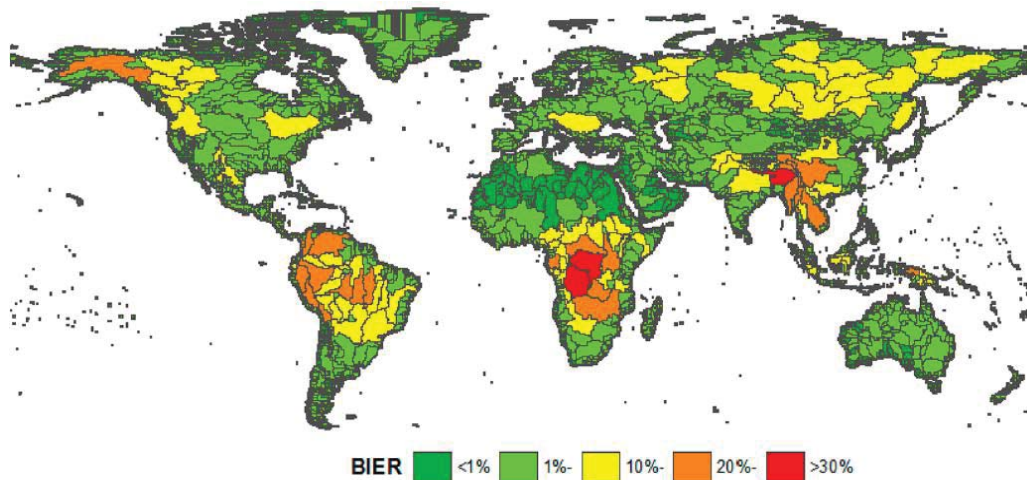


Figure 3.5 Global map of *BIER* ratios denoting the fractions of evaporated water returning to the originating basins via precipitation (Berger et al., 2014)

A limitation of *BIER* is the lack of validity due to the nature of the calculation required, hence the only method for evaluating the accuracy of *BIER* is a sensitivity analysis. It is important to consider that sensitivity analysis determines the sensitivity of the parameters in the equation *only* and not the sensitivity of the parameters in nature. If the model is incorrect, or if it poorly represents a parameter in reality then analysing the sensitivity of the parameter is a meaningless pursuit (Pilkey & Pilkey-Jarvis, 2007).

In New Zealand *BIER* values vary from 0% in parts of the lower North Island and upper South Island to 0.03% in the Otago region (Figure 3.6).

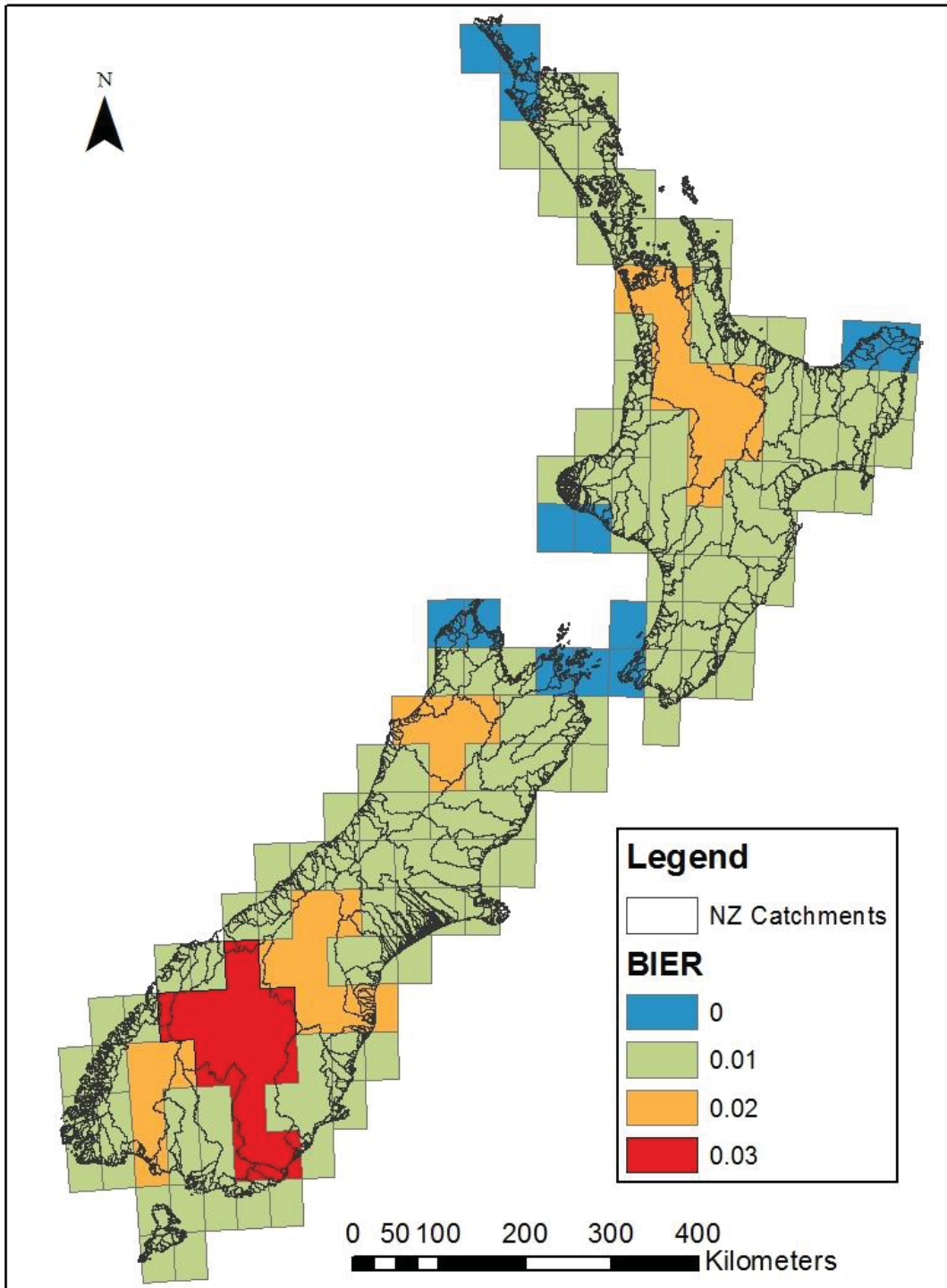


Figure 3.6 The *BIER* ratios for New Zealand catchments (as calculated by Berger et al. 2014).

In order to determine the volume of the evapotranspiration that is recycled within the catchment, the volumes of evapotranspiration (E_i) within each basin are multiplied by $BIER$ and the runoff fraction (a).

Equation 13

$$ER_i = E_i \times BIER_n \times a \quad [-]$$

Similarly, the volume of synthetic vapour (V_i) recycling is calculated as follows:

Equation 14

$$VR_i = V_i \times BIER_n \times a \quad [-]$$

In recognition that not all water that returns to land as precipitation is going to be available as ground and/or surface water stocks, the method includes a runoff fraction (a) (based on data from the WaterGAP2 global model). The runoff fraction is calculated as the long term average runoff (R), (groundwater recharge and surface runoff) divided by the total precipitation (P) within a drainage basin, as follows:

Equation 15

$$a = \frac{R}{P} \quad [-]$$

When $BIER$ is multiplied with the runoff fraction, a hydrologically effective $BIER$ ($BIER_{hydro\ eff}$) is produced. Figure 3.7 provides $BIER_{hydro\ eff}$ values for major water basins. When compared with Figure 3.5, it can be seen that large $BIER$ ratios are reduced when consideration for the local runoff is included. Most of the world's basins have a $BIER_{hydro\ eff}$ of below 5%. By including the runoff fraction the blue water consumption of the Himalayas and Alaska for example, is reduced from >80% when using $BIER$ (Figure 3.5) to between 10-33% when using $BIER_{hydro\ eff}$ (Figure 3.7). Giving a more accurate representation of the proportion of recycled precipitation.

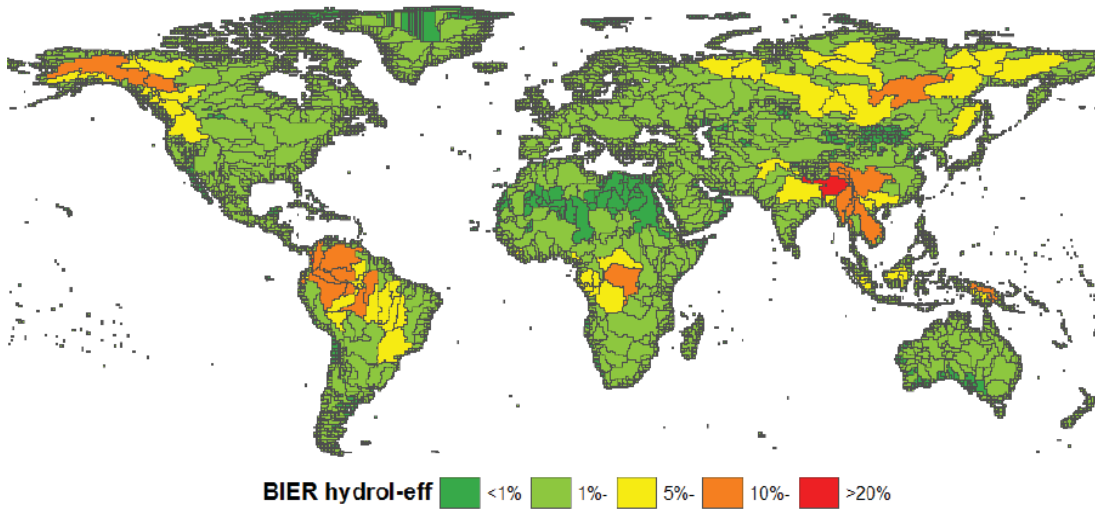


Figure 3.7 Hydrologically effective basin internal evaporation recycling ($BIER_{hydro\ eff}$) ratios denoting the fractions of evaporated water returning to the originating basin as blue water (Berger et al., 2014).

Phase Two: Vulnerability Evaluation Modelling

The WAVE model differs from impact assessments which describe damages to human health, ecosystem quality and resources. It is the opinion of the author that these impact assessment methods rely on various assumptions, and the regression models used are often of low statistical significance (Berger et al., 2014). As an alternative, the WAVE model proposes to assess impacts in terms of vulnerability to the freshwater resource through the *RFD* impact category (refer to Equation 10 for calculation of the *RFD*). More specifically, the method considers the regional vulnerability of drainage basins to blue water depletion.

The *WDI* is defined as the vulnerability of the basin to freshwater depletion based solely on physical blue water scarcity and based on a consumption-to-availability *CTA* ratio (as opposed a *WTA* ratio):

Equation 16

$$WDI_n = \frac{1}{1 + e^{-40 \times CTA} \left(\frac{1}{0.01} - 1 \right)} \quad [-]$$

WDI_n turns 1 above a CTA^* of 0.25, which is regarded as the threshold of extreme water stress. The calculation of CTA accounts for annual water consumption (C), annual water availability (A), and annually useable surface water stocks (SWS) such as lakes, wetlands and dams. The freshwater availability (A) of a drainage basin is defined as the volume of annually renewable freshwater within the basin, which is quantified by means of runoff²⁵. Therefore, in order to consider the usable surface water stocks of lakes, wetlands and dams the SWS are added to the freshwater availability (A). Additionally, for the first time, consideration is given to the volume of groundwater stocks. As groundwater data are not available at the global level, an adjustment factor (AF_{gws}) has been introduced to reduce the final scarcity ratio depending on the availability of groundwater²⁶.

Equation 17

$$CTA^* = \frac{C}{A + SWS} \times AF_{gws} \quad [-]$$

For SWS an annual useable fraction was determined by multiplying the area of each wetland and/or lake within a specified boundary by an effective depth. Effective depths of 0.2m and 0.5m were applied for wetlands and lakes, respectively. Data for the volume of freshwater in dams is assumed to be readily available. A useable fraction of 1% of the total volume of lakes, wetlands and dams per selected boundary is used for the volume of SWS , as follows in Equation 18. The rationale being that surface and groundwater stocks can be utilised for a minimum of 100 years, even if no renewability occurs.

Equation 18

$$SWS = \frac{\sum_i (V_{dam,i} + (A_{lake/wetland,i} \times d_{eff}))}{100 \text{ years}} \quad [-]$$

The WDI is therefore aimed at assessing a basin's vulnerability to freshwater depletion based on a modified CTA (CTA^*), which accounts for both surface and groundwater

²⁵ Plus upstream inflows if the catchment is divided into sub-catchments.

²⁶ Based on geological structure and annual recharge of the drainage basin.

stock availability. It can be interpreted as an equivalent volume of depleted water resulting from a volume of water consumption. Similar to the *WSI* of Pfister et al. (2009), the *WDI* follows a logistic function and turns 1 above the *CTA* ratio of 0.25. This is considered the threshold for extreme water stress.

In arid and semi-arid basins²⁷ *WDI* must be set to maximum (1), as in its current form *WDI* does not account for absolute freshwater shortage (only relative freshwater scarcity). This is an important step when assessing the vulnerability of basins in these conditions, as freshwater resources are often highly vulnerable to depletion regardless of the relative water scarcity (Berger et al., 2014). In other methods water scarcity indicators are calculated as zero (or near to zero) in these environments due to lack of consumption (Frischknecht et al., 2009). This does not, however, accurately reflect local scarcity conditions. The *WDI* for the world's major river basins has been calculated (Figure 3.8). It can be seen that *WDI* is highest in arid and semi-arid environments such as central Asia.

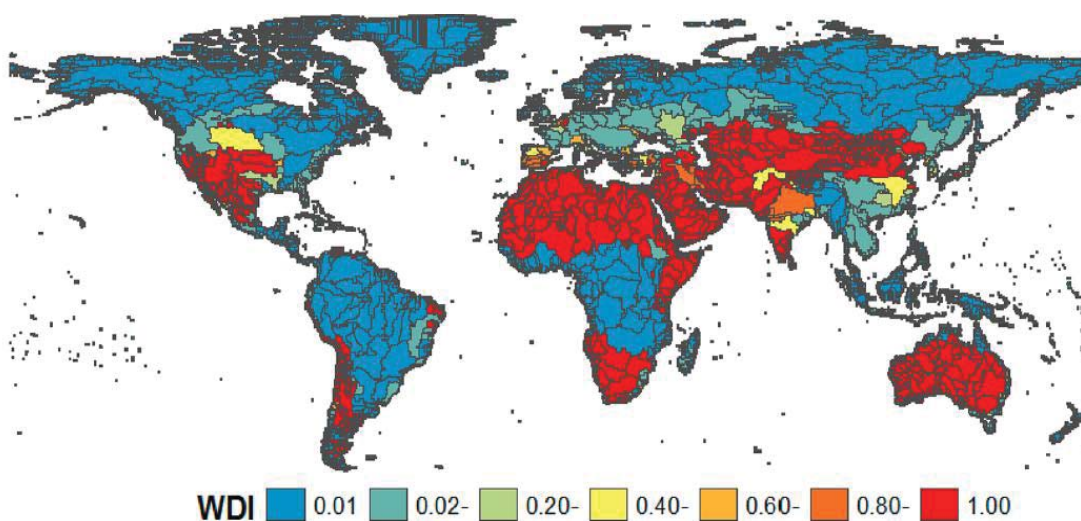


Figure 3.8 A global representation of the *WDI* expressing the vulnerability of basins to freshwater resource depletion (Berger et al., 2014).

²⁷ Refer to method supplementary materials (Figure S4) for a global map of basins classified as dry sub-humid, semi-arid and arid according to UNEP.

As it stands, the WAVE method does not account for green water. However, the authors stipulate that both *BIER* and *BIER_{hydro,eff}* have the capacity to account for the basin internal recycling of plant evaporation (green water). Similarly, no consideration is given to water quality degradation (grey water) in the method's original form. Although the authors do provide a means of incorporating water quality aspects into the *RFD*, following the Water Impact Index tool (developed by Veolia Water). This involves relating water quality standards, in this case a target concentration of a pollutant, to the actual concentration in the water flow (see methods SI). This approach is similar to that provided by Hoekstra et al. (2011) for grey water assessment.

The WAVE method is compatible with both volumetric and LCA (impact-orientated) water footprint methods. For the former, it offers interpretation of volumetric water footprint methods on a qualitative level, and for the latter it provides a compatible characterisation factor (the *WDI*) for use in LCA studies. At present case studies conducted using the *WDI* method are significantly less than the aforementioned methods of Hoekstra et al. (2011) and Pfister et al. (2009), and are limited to the water footprint of bio-fuels (Berger et al. 2014).

3.3.3 The Available Water REMaining (AWaRe) method (Boulay et al. (2016))

The Water Use in Life Cycle Assessment (WULCA) working group developed the Available Water Remaining (AWaRE) method, in accordance with the *ISO 14046* standard (ISO, 2014), after members of the group identified a lack of consistency when applying the principals of LCA to water footprinting assessment (Boulay et al., 2016). The group's aim was to provide a consensual and consistent framework to assess, compare and disclose the environmental performance of product lifecycles with regard to freshwater use. Further, the method can be used to calculate water scarcity footprints in accordance with *ISO 14046* (Boulay et al., 2016).

Boulay et al. (2016) has proposed the water scarcity mid-point indicator method, AWaRE. AWaRE represents the water deprivation potential of water consumption, quantifying the relative Available WATER REMaining, per unit of surface area in a given watershed, relative to the world average, after human and ecosystem demand

has been met. Answering the question "What is the potential to deprive another user (human or ecosystem) when consuming water in this area?" (Boulay et al., 2016). The method builds upon the assumption that the potential to deprive another user of freshwater is directly proportional to the amount of freshwater consumed (accounted for in the LCIA phase), and inversely proportional to the available freshwater remaining (specified spatially and temporally).

Expanding upon the currently used *WTA* (Pfister et al., 2009), and *CTA* (Berger et al., 2014) ratios in LCA, Boulay et al. (2016) identified a need for a shift in focus to a demand-to-availability (*DTA*) ratio. Since both ecosystem and human water consumption can be considered in "demand". However, during the consultation period, it was identified that the *DTA* ratio, as with other *WTA* and *CTA* ratios (with the exception of Berger et al. 2014), failed to account for absolute water availability (or scarcity) per area. As the metric used is relative to use, a loss of information occurs. Boulay et al. (2016), thus agreed on a characterisation factor based on the availability-minus-demand (*AMD*) per basin (*i*), where consideration is given to both human water consumption (*HWC*) and ecosystem water requirements (same principal, and hereby referred to as environmental flow requirement [*EFR*]) relative to the area (m^3 per m^2 per month), as follow:

Equation 19

$$AMD_i = \frac{Availability - HWC - EFR}{Area} \quad [\text{m}^3/\text{m}^2 \text{ month}^{-1}]$$

Volumes of actual water availability were derived from the WaterGAP 2.2 global database. Human water consumptions (*HWCs*) are derived from the WaterGAP global database, and are based on statistical data for consumption of freshwater withdrawals (Flörke et al., 2013) alongside an underlying global hydrology model created by Müller Schmied et al. (2014). The volumes of *HWC* were derived from WaterGAP and are calculated at 0.5 by 0.5 degree scale across earth (with the exception of Antarctica). *HWCs* were calculated at the annual time step for all user groups with the exception of agriculture, which are calculated at the monthly time step.

As the *AMD* factor is expressed relative to the area ($\text{m}^3/\text{m}^2 \text{ month}^{-1}$) results across regions are comparable. For example, consumption of freshwater in regions with the same regionally remaining water per m^2 per month (after the *HWC* and *EFR* have been met) are equal as no other regional specification is considered in the *AMD* calculation.

The *EFR* as used by Boulay et al. (2016) differs slightly from the *EFR* used by Hoekstra et al. (2011). Unlike Hoekstra et al. (2011), which accounts for the *EFR* as 80% of natural flow at the annual scale (following Richter et al. (2012)), Boulay et al. (2016), distinguishes *EFR* by month, following the method of Pastor et al. (2013). The monthly model was validated using local case studies of five major habitat types encompassing the main freshwater eco-regions. It is the first global scale method to account for *EFR* at the monthly scale (Boulay et al., 2016).

The *EFR* model as presented by Boulay et al. (2016) evaluates the minimum water required as a fraction of the available flow required to maintain freshwater ecosystems in "fair" condition (with respect to pristine flows i.e. no human influence), where "fair" refers to an ecological state between poor and good (Pastor et al., 2013). The *EFR* for fair-conditioned freshwater ecosystems was found to range from 30-60% of pristine flow as a function of seasonal flow patterns. In terms of the global scale analysis, *EFR* was calculated in conjunction with monthly natural flow data from the WaterGAP database²⁸.

Although Boulay et al. (2016) recommend calculating *AMD* at the sub-watershed level and the monthly time-step, values may also be aggregated to country level and/or annual time-step using a new aggregation concept. The aggregation method selected is dependent on the water user category concerned (agricultural, domestic or industrial). Characterisation factors for agricultural and non-agricultural water use are incorporated in the method for aggregation where monthly values and/or the location of water consumption data are unavailable. This provides an additional layer of information, in order to bridge the gap between spatial and temporal resolutions of the LCIA phase, with the relevant resolutions for water use impact assessments, as these

²⁸ Note: terrestrial and ground-water dependant ecosystems were not accounted for.

are often conducted in union. Boulay et al. (2016) suggest that for New Zealand, the spatial resolution should take priority over the temporal resolution²⁹. The annual consumption-based average for unknown users, non-agricultural and agricultural users in New Zealand are 9.53, 3.21 and 12.16, respectively (Boulay et al., 2016). The relative AWaRE in Figure 3.9 has been calculated at the global sub-watershed level, using monthly data (aggregated to the annual scale).

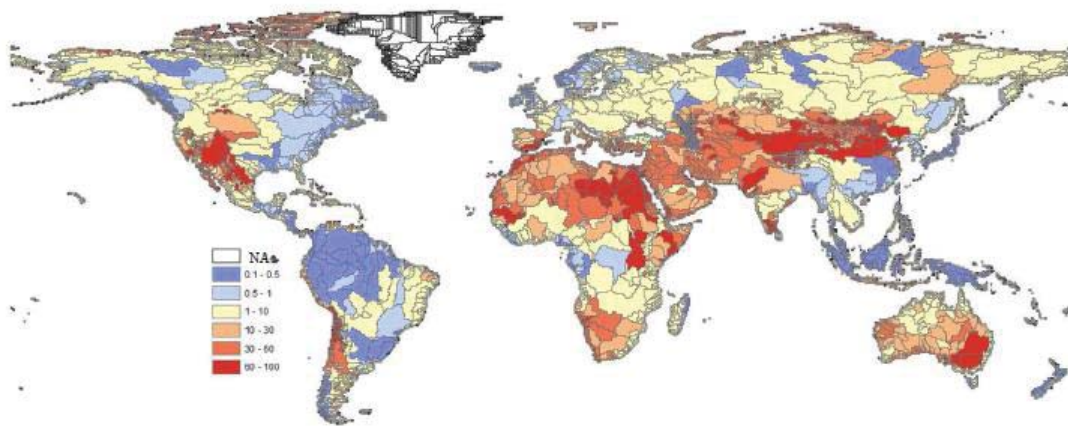


Figure 3.9 The AWaRE characterisation factor for water scarcity footprint (in m³ world eq./m³ consumed in region). Represented at the annual level for non-agricultural users (Boulay et al. (2016)).

The Surface- Time- Equivalent

The inverse of the *AMD*, is the *STe*. Depending on the desired output, the surface-time-equivalent may be calculated per basin. The surface-time-equivalent (*STe*) per basin (*i*) is the inverse of the difference between availability and demand (*AMD_i*), as follows;

Equation 20

$$STe_i = \frac{1}{AMD_i} \text{ [m}^2 \text{ month, m}^3\text{]}$$

²⁹ Due to the high standard deviation

It can be interpreted as the surface-time equivalent necessary to generate one cubic metre of unused water in a region (assuming a given level of water demand), to the stage where water availability can equal the water demand (Boulay et al., 2016). When the water demand is equal to, or exceeds, the available water in a particular region (which would result in a negative AMD), the STe_i is set to maximal (100), as the equation would no longer be continuous nor hold the same meaning (refer to cut-off criteria in Equation 21).

Both the AMD_i and its inverse, the STe_i range from 0.1 to 100³⁰, with a factor value of 1 corresponding to the world average³¹. Values of less than 1 indicate regions with lower water scarcity than the world average. A value of 10 for example, indicates that the region has 10 times less available water remaining per unit area than the world average. In these regions it takes 10 times more surface-time to generate an amount of unused water, compared with the world average (assuming a given level of water demand). The upper limit of 100 affects regions in which the demand exceeds the available water. This represents 33% of world consumption at the monthly level (Boulay et al., 2016).

In a step similar to the Global Warming Potential (Intergovernmental Panel on Climate Change, 2013) and the Water Stress Index (Ridoutt & Pfister, 2013) the "virtually occupied" area is normalised by a consumption-weighted world average and inverted, representing the value in comparison with the average m^3 consumed worldwide (for demand < availability). In this instance the global average AMD has been calculated using the WaterGAP global database as $0.0136 m^3 m^{-2} month^{-1}$ (Boulay et al., 2016). Characterisation factors (CF's) are calculated according to Equation 21 and/or Equation 22 as follows;

³⁰ Following the cut-off criteria referred to in the methods SI.

³¹ It should be noted that the factor value of 1 is not equivalent to the factor for the average water consumption in the world, i.e. the world average factor to use when the location is unknown. This value is calculated as the consumption-weighted average of the factors based on the $1/AMD$ calculation (as opposed to the AMD calculation). Thus the world consumption-based average for unknown users, non-agricultural and agricultural users are 43, 20 and 46 respectively.

Equation 21

$$CF = \frac{0.0136}{AMD_i} [-]$$

In order to deal with future changes in consumption and availability to the global *AMD* (and the local *AMD*'s as calculated by Boulay et al. (2016), the *CF*'s presented by Boulay et al. (2016) are to be updated as, and when, the WaterGAP global database is updated. The global *AMD* of 0.0136 m³ m⁻² month⁻¹ was calculated based on consumption estimates for the 2010 period (personal communication, A-M Boulay, April 26th, 2016).

Alternatively the *STe_i* can be applied, depending on the desired interpretation, which is either the area of virtually occupied land (using *AMD*) or the surface time equivalent to generate 1 m³ of unused water in a region (using *STe*). Note: calculated outcomes are identical.

Equation 22

$$CF = \frac{STe_i}{STe_{world\ ave}} [-]$$

Cut-offs are applied to the *CF* (either *AMD* or *STe_i*), as follows:

- $CF = MAX = 100$, for demand \geq Availability or $AMD_i > 100 AMD_{world\ ave}$
- $CF = MIN = 0.1$ for $AMD_i < 0.1 * AMD_{world\ ave}$

The AWaRe method and the calculation of *AMD* differs from the aforementioned *WSI* (Pfister et al., 2009) and *WDI* (Berger et al., 2014) with regards to the use of scaling (refer to Table 3.4). Both the *WSI* and the *WDI* use a scaling function to fit the obtained values within a certain range (e.g. a logistic curve between 0.01 and 1, as used in the *WSI* and *WDI* methods). The use of scaling is criticised by Boulay et al. (2016), as it implies normative choices for curve tuning parameters, and there is little information to support this choice. It is for this reason, and unlike the other LCA methods used in this case study, that the *AMD* uses the aforementioned cut-off criteria with no additional modelling choices.

The AWaRe method is the most recently developed LCA method for water footprint assessment. At present there are few case studies in which the AWaRe method is applied. Case studies to date include; comparative water footprint profiles of vegetable oil for bio-fuel production (Caldiera et al., n.d.), water scarcity footprint of a Volkswagen car (Emara & Berger, 2016a), the water scarcity footprint of a flow regulator (NEOPERL) (Emara & Berger, 2016b), and the application of AWaRe to bottled beverages (Bayart & Ekambi, 2016).

3.4 Contentions Between Water Footprint Methods

There is much debate in the literature as to the applicability of the two approaches to water footprinting (volumetric vs. impact assessment). A number of communications have been exchanged between supporters of the two approaches (Hoekstra, Gerbens-Leenes, & Van der Meer, 2009; Hoekstra & Mekonnen, 2012a; Pfister & Hellweg, 2009; Ridoutt & Huang, 2012), and some effort has been made to co-ordinate the two trains of thought, though unsuccessful at present (Boulay, Hoekstra, & Vionnet, 2013).

The disagreement between the LCA community and the volumetric water footprint method stem from the use of quantitative indicators at different stages in the analysis (WFN method in the inventory phase, and LCA methods in the impact assessment phase) (Boulay et al., 2013). The most important difference in the final water footprints of the two pathways, is that a volumetric water footprint has a water resource management focus, whilst an LCA water footprint has an environmental impact focus (Boulay et al., 2013).

3.4.1 Criticisms of the Volumetric Water Footprint

The volumetric water footprint method proposed first by Hoekstra et al. (2009), and updated by Hoekstra et al. (2011) has seen some opposition by LCA practitioners over the years. A critique provided by both Ridoutt and Pfister (2010) and Berger and Finkbeiner (2012) states the inclusion and importance of green water consumption as questionable. Berger and Finkbeiner (2012) concluded that the importance of green water consumption is doubtful, as soil moisture is only available locally for plants, and cannot be utilised by the surrounding ecosystems or for human withdrawal. Similarly,

Ridoutt and Pfister (2010) argued that green water consumption does not contribute to blue water scarcity, and that as green water is inseparable to land, green water is best considered in the context of land-use.

Further, it is stated that the volumetric water footprint identifies the need to address inefficient water use, without considering that any investments to improve water use efficiencies are best implemented if priority is given to water stressed regions (Ridoutt & Huang, 2012). Similarly, Ridoutt and Pfister (2010) argue that because the volumetric water footprint does not allow for an assessment of the consequences of freshwater consumption, that they are misleading and incorrect. Ridoutt and Huang (2012) concluded that water footprints that do not take into account environmental relevance have the potential to misinform and motivate behaviours that adversely affect the overall goal of reducing pressure on freshwater systems. At the core of the LCA community's argument is that the need to reduce humanity's water footprint arises from the current pattern of water use, which is skewed towards highly stressed watersheds, as opposed to an absolute shortage of freshwater globally. Therefore, it is argued by LCA practitioners that the volumetric water footprint lacks meaningful interpretation (Ridoutt & Huang, 2012).

Moreover, Berger and Finkbeiner (2013) criticised the aggregation of the blue, green and grey water footprints into a single index, as this implies the three components have equal weighting, and that compensation between categories is possible. It is argued, that in most cases, this is not possible. Further, aggregation of the blue and green water consumption hides the advantages of rain-fed compared to irrigated agriculture.

3.4.2 Criticisms of the Impact-Orientated Water Footprint

In response to the aforementioned criticism, and the release of the *ISO 14046* standard (which supports taking an LCA approach to water footprinting), Hoekstra (2016) critiques the weighting of the volumetric water footprint with the water scarcity in the local environment (in particular the *WSI* method proposed by Pfister et al., 2009). Hoekstra et al. (2011) noted that for the purposes of a sustainability analysis the volumetric accounts provided by the water footprint are more useful than the aggregated impact indices proposed by LCA practitioners. The author regards impact-

orientated methods as "lacking physical interpretation" and using questionable weighting choices (Hoekstra et al., 2009).

Further, Hoekstra et al. (2016) made the point that the LCA community emphasises the issue of water use as a category of environmental impact, although it is the authors' opinion that insufficient information has been presented on what the environmental issue actually is. Instead of a water-scarcity based characterisation factor, Hoekstra (2016) proposes that volumetric water footprints should be characterised by water availability. It is the authors' point of view that allocation is about allocating litres, not about allocating scarcity-weighted litres. Moreover, it is suggested that water footprint assessments according to LCA method are inconsistent with how other environmental footprints are defined.

Moreover, as water scarcity increases with water consumption in a catchment, the multiplication of consumptive water use (of a specific product/process for example) with water scarcity is thought to imply that the resultant weighted water footprint is affected by the water footprint of other processes or activities within the area. Hoekstra (2016) argues that this cannot be the purpose of an environmental performance indicator.

In terms of the inclusion of green water in impact-orientated water footprint methods, Hoekstra (2016) make the justification that green water shortage in agriculture, for example, is in fact the reason behind blue water demand, and thus blue water scarcity. However, this LCA view ignores the need to assess and improve green water use efficiency to reduce impacts of human water consumption on global water resources.

Finally, Berger and Finkbeiner (2012) make the argument that water footprint focused LCA methods have the potential to double count water quality degradation, as water quality is covered by existing LCA pathways (e.g. eutrophication, eco-toxicity). Thus there is the potential for overestimations if mutually exclusive impact pathways are considered.

3.5 Methodological Comparison

There are numerous water footprint assessment studies based on the calculation of both volumetric water footprints, and the impact assessment of water consumption. As discussed, the variables considered in each method differ depending on the goal and scope of the assessments. The methods of Hoekstra et al. (2011), Pfister et al. (2009), Berger et al. (2014) and Boulay et al. (2016) are used in the case study. Table 3.4 provides a summary of the relevant variables and parameters considered in these methods.

Table 3.4 A comparison of the four selected water footprint methods used in this research.

	Hoekstra et al. (2011)	Pfister et al. (2009)	Berger et al. (2014)	Boulay et al. (2016)
Subject of Study	Product, process & organization	Product, process & organization	Product, process & organization	Product, process & organization
Focus is on Consumption and/or Degradation?	Consumption & degradation	Consumption	Consumption	Consumption
Indirect/ Direct Water Use	Indirect & Direct	Indirect & Direct	Indirect & Direct	Indirect & Direct
Water Colour(s)	Blue, Green & Grey	Blue water only	Blue water only	Blue water only
WTA/CTA/AMD	CTA	WTA	CTA	AMD
Characterisation Factor	Blue and Green Water Scarcities and/or the Grey Water Pollution Index	Water Stress Index	Water Depletion Index	Available Water Remaining
Midpoint/ Endpoint	N/A	Midpoint & Endpoint	Midpoint & Endpoint	Midpoint
Order of Magnitude (CF)	7	2	2	3

Chapter 4

Methods and Materials

4.1 Scope of the Research

The aim of this research is to investigate the feasibility of adapting existing water footprint methods, and associated water scarcity indicators, presented by Hoekstra et al. (2011), Pfister et al. (2009), Berger et al. (2014) and Boulay et al. (2016), to assess water use in products and processes by the building and construction sector of New Zealand. The focus is on assessing the impacts of any spatial variation in the various water footprint characterisation factors used, and the influence of this spatial scale on the results of a water footprint assessment of New Zealand products. This is investigated using a case study of ready mix concrete production in New Zealand.

The analysis is separated into three distinct spatial scales of analysis:

- Scale 1: Assessment of the water footprint at freshwater management zone level using locally provided data at the management zone scale;
- Scale 2: Assessment of the water footprint at catchment level using locally provided data at the catchment scale;
- Scale 3: Assessment of the water footprint at regional level using locally provided data at the regional scale.

4.2 Case Study: The Water Footprint of Ready Mix Concrete

Ready mix concrete in New Zealand is categorised as 'normal' or 'special', and produced according to NZS3104 (Standards New Zealand, 2013). Normal ready mix concrete has a 28-day compressive strength between 17.5 MPa and 50MPa, and is the focus of this case study. Special concrete is not considered in the scope of this research. Subsequently, from this point onwards, the term ready mix refers to normal ready mix concrete only.

Ready mix concrete is a multi-purpose product that is used in infrastructure and building construction applications. It primarily consists of a mix of aggregates (coarse and fine), cement, water and admixtures, the proportions of which are varied in order to achieve desired characteristics.

Ready mix concrete may be made at the construction site, either using a dedicated batch process (for larger projects) or using portable mixers (for small projects). Alternatively, it may be ordered from dedicated batching plants located across New Zealand, and delivered ready for use in truck mixers. There are around 170 ready mix concrete plants throughout New Zealand (Jaques, 2001). Approximately 90% of these ready mix plants are represented by three companies with links to the major cement companies. The remaining 10% are independently owned companies (Jaques, 2001). In 2006, over three million m³ of ready mix concrete was produced throughout the country; the industry employed 24,315 staff, and contributed 1.8% of New Zealand's GDP in the same year (New Zealand Institute of Economic Research, 2008).

This case study assesses water use in 27 concrete batching plants located throughout New Zealand for which data were provided by Allied Concrete (Allied Concrete, 2014).

4.2.1 Goal and Scope

This case study is intended as an exploratory investigation, providing a comparison of both the applicability and feasibility of adapting the four water footprint methods selected to three distinct spatial scales. The intended audience is the building and construction sector of New Zealand, alongside LCA specialists and those interested in water resource management.

It is important to note that the water footprint of concrete calculated using the selected methods considers only freshwater resources, and associated local environmental impacts. Other resources used and impacts associated with the ready mix concrete production are not included in this case study, although LCA can facilitate their inclusion. The impact assessment methods as described by Pfister et al (2009), Berger et al (2014) and Boulay et al. (2016) are wholly aligned with LCA, and were developed

to be integrated into LCA studies and/or used as stand-alone indicators. The WFN (2011) method was developed independently of LCA method and can be considered a stand-alone method but can also be used alongside LCA results (Hoekstra et al., 2011).

4.2.2 Functional Unit

Data for eight compressive strengths of ready mix concrete, ranging from 17.5 MPa to 50MPa were provided by Allied Concrete. The difference in the volume of water required to produce 1 m³ of concrete during the manufacturing phase was found to be negligible between compressive strengths (refer to Appendix I). Therefore, data for the eight compressive strengths were aggregated in order to calculate the water footprint of 1 m³ of concrete (for each of the 27 concrete batching plants). The functional unit was therefore defined as 1m³ of ready mix concrete.

4.2.3 System Boundary

Water footprint accounting requires information pertaining to the water consumption during different phases of a product's life cycle. Product water footprints can include all processes in the life cycle. Using concrete as an example, this could include raw material acquisition, through to manufacture, use and disposal ("cradle to grave" approach according to LCA).

Alternatively, product water footprint assessments can be truncated along the lifecycle(Hoekstra et al., 2011)(Hoekstra et al., 2011)(Hoekstra et al., 2011)(Hoekstra et al., 2011). The system boundary of this particular case study is the manufacturing phase of the concrete lifecycle (or a 'gate to gate' approach according to LCA). This includes all water consumed during the batching processes, through to concrete loading and truck washout. A 'gate to gate' analysis was chosen, as the data required to calculate the water footprint of concrete during the manufacture and production processes were available (or able to be calculated/estimated), for the desired spatial scales. The phases of the concrete lifecycle concerning raw material extraction, transportation, use and disposal of the final product were omitted from this assessment. It was deemed beyond the scope of the current research to include these phases of the lifecycle. Figure 4.1 shows the activities included in the analysis for the water footprint assessment of 1 m³ of ready mix concrete.

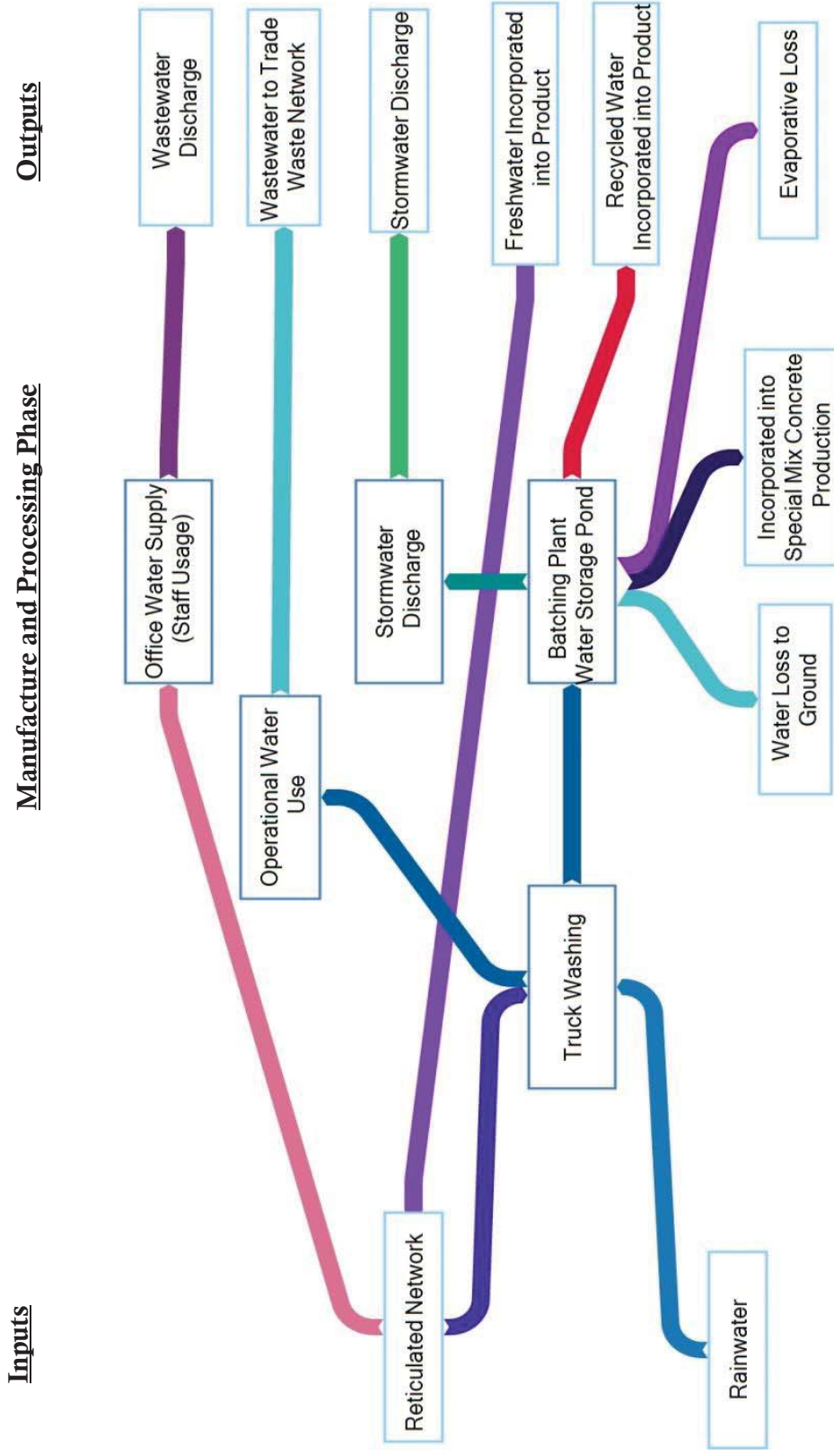


Figure 4.1 Activities included in the system boundary for concrete production within selected Allied Concrete batching plants across New Zealand.

4.2.4 Geographic Resolution

The local environmental impact assessments conducted according to the methods of Hoekstra et al. (2011), Pfister et al. (2009), Berger et al. (2014) and Boulay et al. (2016) are conducted at three distinct scales, i.e. regional, catchment or freshwater management zone, defined as follows:

Regional Scale

There are 16 regional councils throughout New Zealand. As can be seen in Figure 4.2, the 27 concrete batching plants are located within seven of these regional councils. Hereby referred to as *selected regions*.

Catchment Scale

The 27 case study concrete plants are located within 20 catchments throughout the North and South Islands of New Zealand (Figure 4.3). Hereby referred to as *selected catchments*.

Freshwater Management Zone Scale

In New Zealand, the National Policy Statement for Freshwater Management (NPS-FM) (2014), issued under the Resource Management Act³² (1991), has introduced the term 'Freshwater Management Units' (hereby referred to as freshwater management zones [FMZs]) for management of freshwater resources at local scale. As part of the Regional Councils' obligations under the NPS-FM, FMZs are defined and assigned by the relevant regional authorities in their regions. FMZs are described as "the water body, multiple water bodies, or any part of a water body determined by the regional council as the appropriate spatial scale for setting freshwater objectives and limits for freshwater accounting and management purposes" (Ministry for the Environment, 2014b, p. 7). As such, FMZ boundaries were selected for inclusion in this case study as they can differ significantly from the catchment boundaries, and are widely used throughout New Zealand for management and decision-making purposes.

³² The Resource Management Act (1991) is the primary piece of environmental legislation in New Zealand.

The FMZ boundaries presented in this thesis are consistent with those of the Land and Water Aotearoa (LAWA) database (LAWA, 2016). The 27 selected concrete batching plants are located within 21 FMZs, as can be seen in Figure 4.4. These will hereby be referred to as *selected freshwater management zones (FMZs)*.

4.2.5 Temporal Resolution

The data used to quantify water consumption in each concrete batching plant was provided for the year 2013. All other hydrological data used for the environmental impact assessment of concrete's water consumption is based on the average annual datasets (as opposed to averaged monthly datasets) due to the availability and scale of data collected by the relevant organisations. The collated data used for impact assessment are described in the following sections. It should be noted, however, that there is some variation in the temporal resolution of the data used in the assessment. This variation was unavoidable due to data availability from the relevant organisations. Furthermore, it was assumed that long-term average data is preferred for a better understanding of the local hydrological conditions, as this reduces effects from seasonal biases (such as drought conditions or flood events).

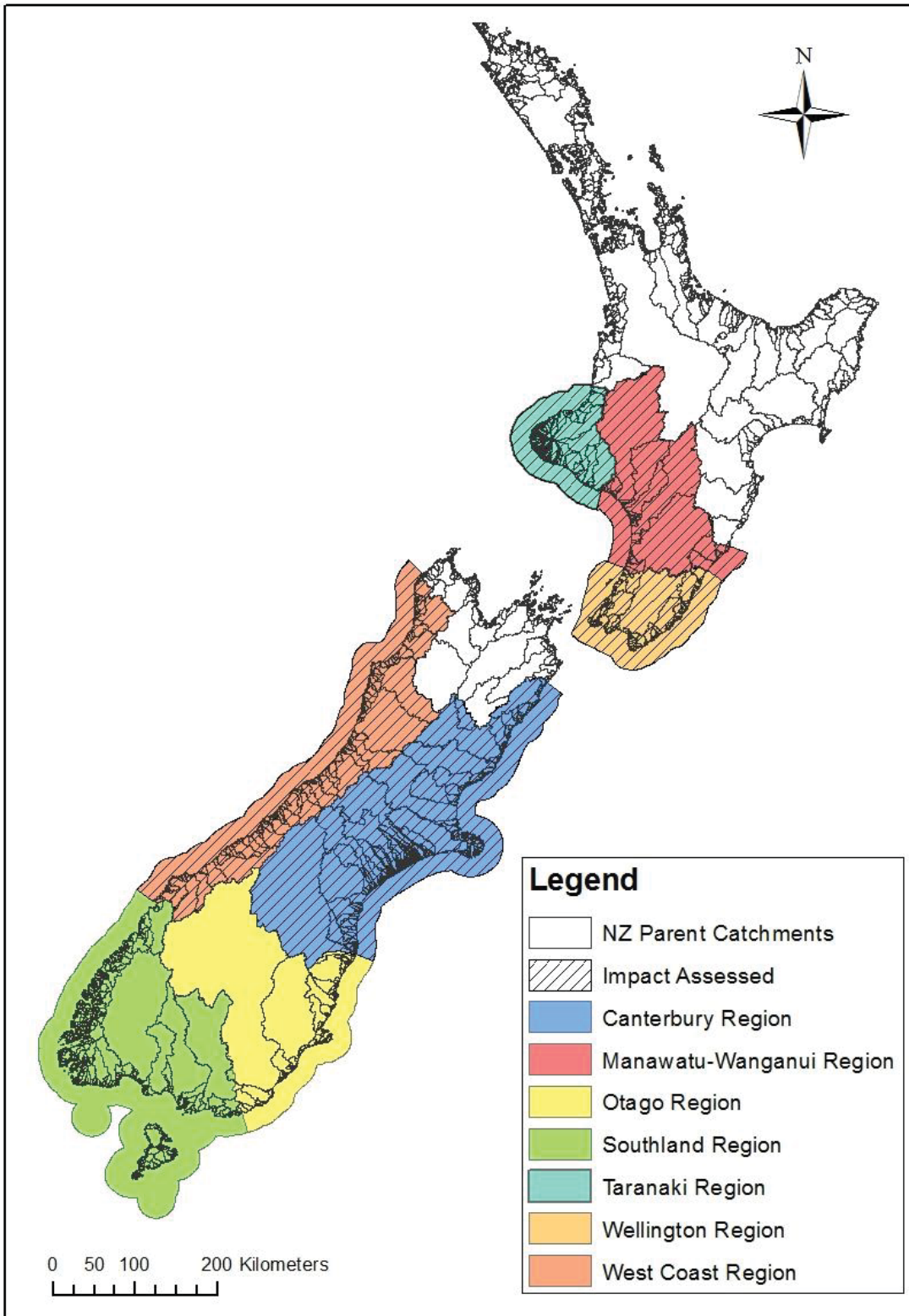


Figure 4.2 Map showing the regions where selected concrete batching plants are located throughout New Zealand.

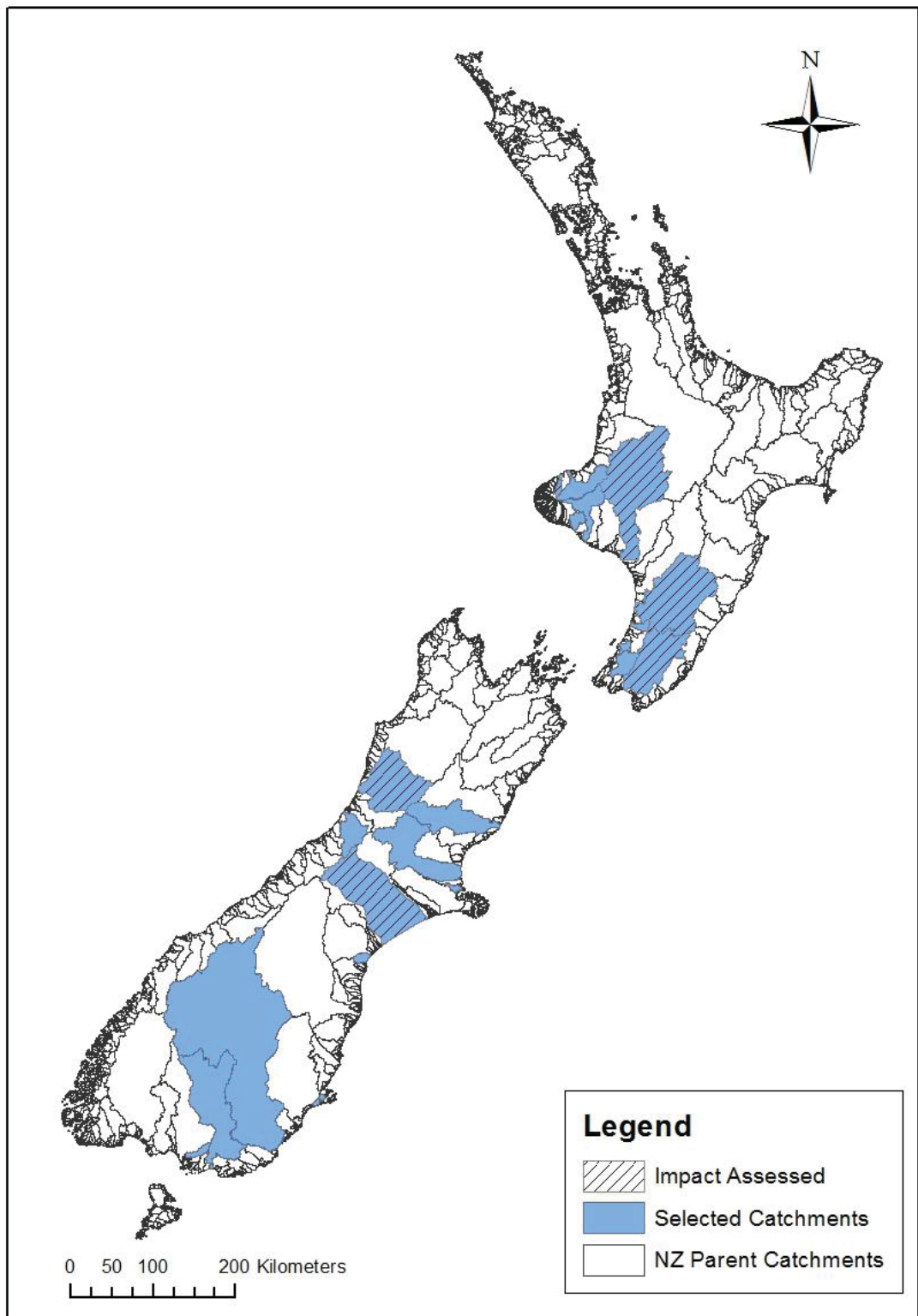


Figure 4.3 Map showing the catchments where selected concrete batching plants are located throughout New Zealand.

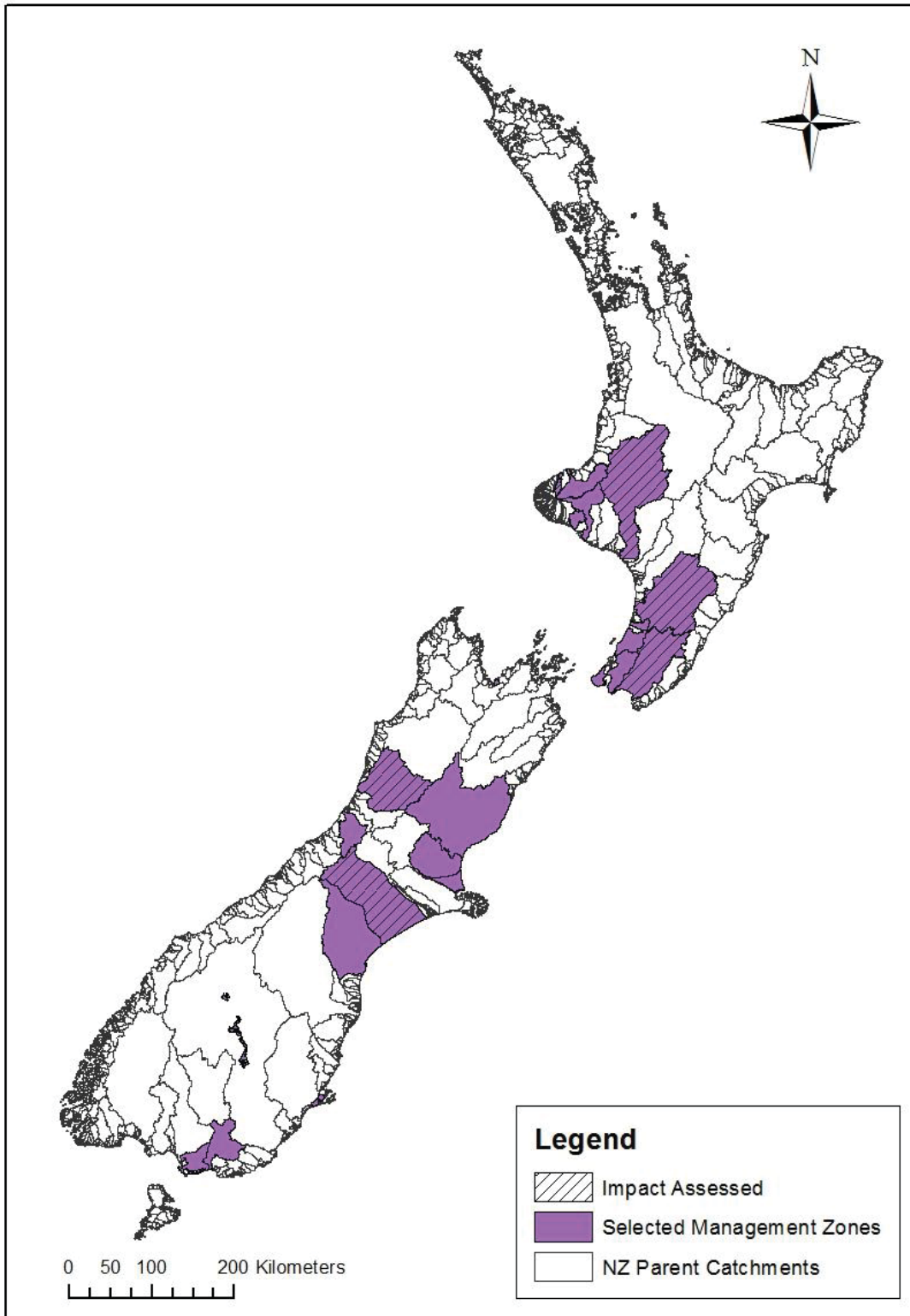


Figure 4.4 Map showing the freshwater management zones where selected concrete batching plants are located throughout New Zealand.

4.3 Data and Quality Requirements (LCI phase)

4.3.1 Data Used for Water Footprint Accounting

The production and water use data used in quantification of the water footprint of ready mix concrete across the 27 case study concrete batching plants were provided by BRANZ with the permission of Allied Concrete. This data was originally compiled for use in an Environmental Product Declaration of the environmental impacts of concrete production (which excluded water scarcity) (Allied Concrete, 2014). The datasets included both primary and secondary data as follows:

Primary data (based predominantly on measurements):

- Total volume of concrete produced by each batching plant for the year 2013.
- Volumes of freshwater entering the batching plants, either from the reticulated network or captured rainwater during the year 2013. Of all the batching plants, seven were metered for freshwater entering via the reticulated network, and ten had meters for recycled freshwater entering the system from storage tanks on-site. There were six plants in which the freshwater input from the reticulated network were un-metered, and three in which the recycled freshwater was un-metered. For the method used to estimate freshwater use by un-metered plants, refer to the secondary data described below.
- Volumes of freshwater and recycled water in concrete, which were based on the automated batching system records for each batching plant. From this, the total freshwater used in the concrete for each plant, and the average water content in the concrete was derived.
- Volumes of water used for washing out trucks, which drains to on-site storage ponds (and may be subsequently used in concrete production).
- Wastewater disposal (either the reticulated wastewater system or a septic tank) were provided for each plant along with the number of staff at each plant.
- Volumes of wastewater discharges were provided by the plant managers. These discharges may be to a wastewater system, a consented discharge to a watercourse or disposed by truck to landfill or other Allied Concrete site (when water levels in storage ponds were high).

- Volumes of lost return flow were derived as storage water that had to be disposed of off-site (and out of the selected boundary). This information was provided by the plant managers for the two batching plants in which this occurred for the 2013 period.

Secondary data (based predominantly on estimates):

- For the six un-metered plants that use freshwater in the concrete mix, a typical water use figure was calculated based on an average derived from the other 18 metered plants. Average water use was multiplied by the concrete production volumes per plant to provide an estimate of freshwater in the product. The same exercise was repeated for the three un-metered plants that use recycled water in the concrete.
- The total freshwater used at the batching plants for other purposes was derived by subtracting the total freshwater used in the ready mix (and special mix³³) in the year, from the total freshwater entering each batching plant. This may be for drinking water/toilets or truck washes. Allocation of this freshwater was based on compressive strengths of the ready mix concrete (17 MPa - 50 MPa), and according to the volume of concrete produced.
- The volume of staff wastewater per plant was estimated based on a singular plant with measured wastewater discharges. A “wastewater discharge/person” figure of 20 L/ per person/ per day was derived. This was based on the assumption that water use per person for kitchen/toilet applications is similar for all sites, and absolute volumes would therefore be dictated by the numbers of employees. By applying the number of days in which each plant is open per week, and the number of weeks per year (which varied between plant), an annual figure of wastewater discharge was derived for each plant.
- Total rainfall volumes collected over each plant site were estimated by taking regional rainfall data, and estimates of the batching plant areas (including the rooftop areas that drain to storage ponds) from the plant managers.

³³ Note: Freshwater used in special mix concrete was accounted for in the collection of this data but is not a focus of the case study

- Total evaporation from storage ponds was calculated based on the published evaporation rates for the North and South Islands of New Zealand (provided by Moot, Mills, Lucas, and Warwick (2009)). Storage pond areas were estimated by the plant managers and provided.
- The volumes of “water loss to ground” were calculated based on the difference between inputs and outputs for each concrete plant. It was identified in the Environmental Product Declaration that water losses could result for a variety of reasons. However, it was assumed that all water losses would eventually make it to the ground. This assumption neglects any hydrological interaction between soil moisture and evaporative losses to the atmosphere.

4.3.2 Data Used for Water Footprint Impact Assessment

Data used for the environmental impact assessment of concrete production were gathered from a multitude of sources, depending on the method choices. There were a few datasets, however, that were used in all four water footprint impact assessment methods. These are presented in Table 4.1, and described along with additional datasets specific to each method in the following subsections.

Table 4.1 Summary of local datasets used for the environmental impact assessment of 1 m³ of ready mix concrete in New Zealand.

Data	Source	Period	Description	Primary/Secondary Data
Water Availability (i.e. Natural Runoff)				
<i>FMZ Scale:</i>	Woods et al. 2006	1996-2006	Modelled mean flow of NZ rivers	Secondary
<i>Catchment Scale:</i>	Woods et al. 2006	1996-2006	Modelled mean flow of NZ rivers	Secondary
<i>Regional Scale:</i>	Woods et al. 2006	1996-2006	Modelled mean flow of NZ rivers	Secondary
Water abstractions (based on consented water per user group)				
<i>FMZ Scale:</i>	LAWA	Can vary depending on the council in focus	Estimated abstractions based on regional scale calculations and FMZ scale consented surface and groundwater volumes	Secondary
<i>Catchment Scale:</i>	Regional Council's & LAWA	Current	Estimated abstractions based on regional scale calculations and catchment scale consented surface and groundwater volumes	Secondary
<i>Regional Scale:</i>	Ministry for the Environment	2009-10	Estimated abstractions based on regional scale calculations and surface and groundwater abstractions	Secondary

4.3.2.1 Water Availability

The environmental impact assessment of concrete production requires information on local water availability at appropriate scale of analysis. Woods et al. (2006) modelled the mean flow of all New Zealand rivers using four experimental models, over the period 1996-2006. The data presented in this thesis originates from the model four output, referred as 'model 4' hereafter (refer to Appendix B). This model 4 estimated the actual evapotranspiration according to the ratios of potential evapotranspiration with annual precipitation, using a single water balance parameter estimated by an independent calibration (Zhang et al., 2004). A regional bias correction is applied to the model results. The model 4 estimates were deemed the most accurate when compared with measured and synthesised runoff for each catchment (Woods et al. 2006). The modelled runoff volumes can be interpreted as the natural runoff to New Zealand rivers. It should be noted that during the model calibration phase sites were selected to have little effect with regards to water use or diversion (personal communication, Roddy Henderson, March 31st, 2016). The model 4 estimated runoff was clipped and converted (into m³/year) in ArcGIS (v.10.2) to derive the average runoff per selected boundary.

4.3.2.2 Accounting for Consented vs. Consumptive Freshwater Use

The volume of surface and groundwater consented to be abstracted were provided by various organisations (see Table 4.1). However, not all consented water abstractions are used for actual water withdrawals by water users. In terms of the actual appropriation of consented freshwater, estimates of actual water withdrawals at the regional scale can be seen in Figure 4.5. Estimates of actual freshwater abstractions [%] have been calculated for the 2009-10 period for all regions of New Zealand (MfE, 2010).

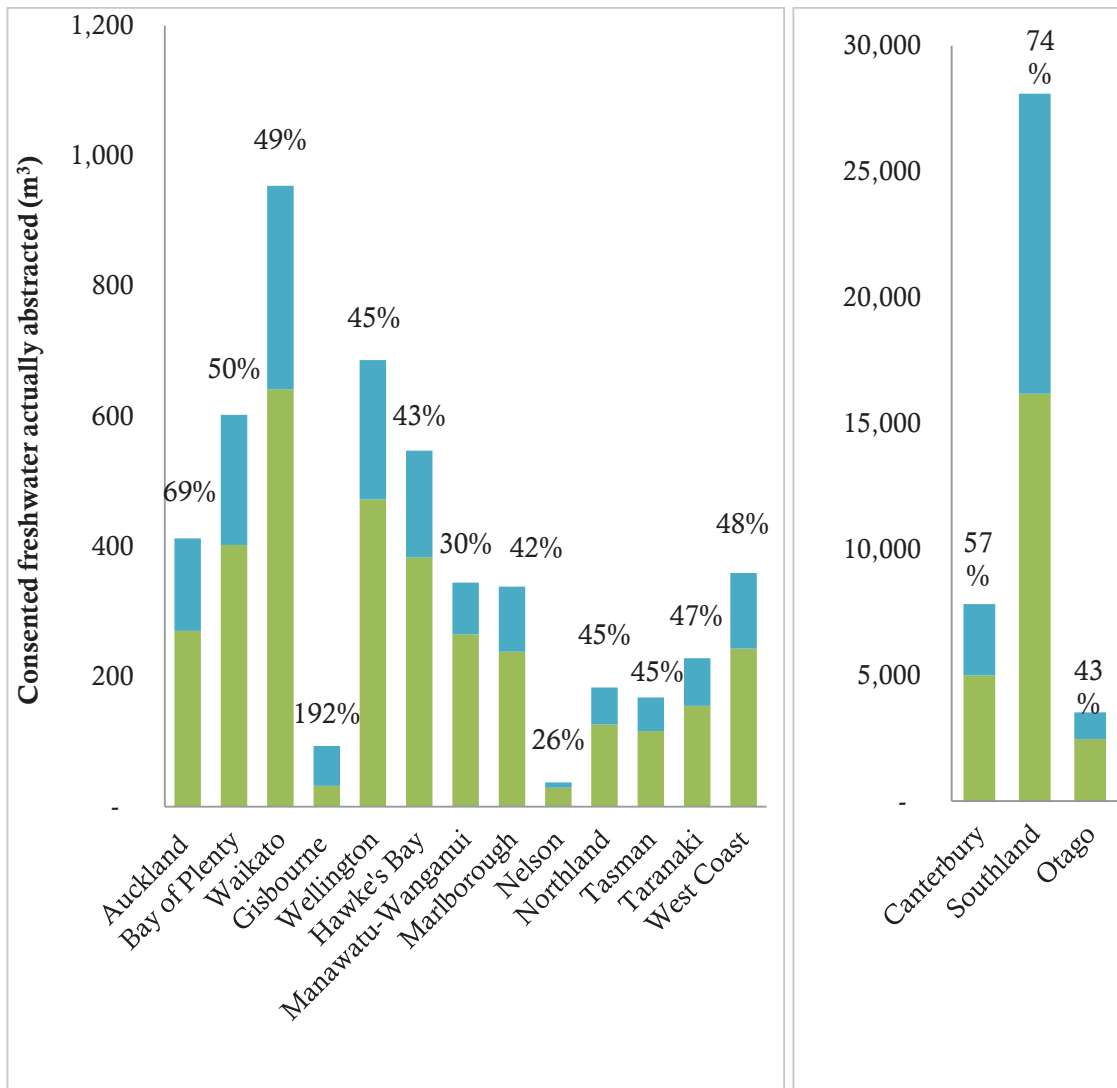


Figure 4.5 Estimates of actual water use per region of New Zealand (based on MfE, 2010)

Expanding on Figure 4.5, the Ministry for the Environment (2010) provide estimates of the actual abstraction rates per user group, and per region. Table 4.2 shows the percentage of consented freshwater per user group that is actually abstracted (for selected regions only). These estimates were used to convert consented freshwater abstractions to actual freshwater abstractions at all spatial scales. It is important to note that irrigation and stock water abstractions have been combined and averaged to provide the agricultural user group.

Table 4.2 Estimates of actual water use as a percent of consented freshwater abstractions per user group, per region in New Zealand (MfE, 2010).

Region	Domestic Users	Industrial Users	Agricultural Users
Canterbury	32 %	40%	60%
Southland	45%	36%	47%
Wellington	53%	67%	41%
Horizons	34%	19%	44%
Marlborough	41%	52%	41%
Nelson	26 %	14 %	58%
Otago	24%	21%	63%
West Coast	98%	40%	67%
Taranaki	56%	37%	64%

4.3.2.3 Accounting for Water Use Efficiency

In terms of freshwater use, it is also recognised that not all freshwater abstracted is actually consumed, i.e. returned to another catchment (or the sea), evapotranspired, or incorporated into a product (following the definition of consumption provided by Hoekstra et al. 2011). The volume of freshwater consumed is effected by the efficiency of the water use system. Therefore, the efficiency of water use per user group: agricultural, domestic, industrial and 'other' are considered in order to translate actual water abstractions to actual water consumption.

It should be noted that the definition of water 'consumption' used by authors cited below varies slightly from the definition provided in ISO 14046, by including leaks to groundwater, for example. Where necessary (and provided³⁴) definitions will be stated for the different user groups.

Irrigation Efficiency as a Proxy for Agricultural Water Use Consumption

In reference to Figure 2.2b it can be seen that agricultural water users (irrigators and stock water sources) are by far the largest user of freshwater, as compared with domestic or industrial water users in New Zealand. Agriculture accounts for 48% of

³⁴ The definition of consumption using the method of Shiklomanov (2000) is not explicitly defined

weekly water abstractions throughout New Zealand, whilst domestic (drinking water supply) and industrial water abstractions account for 5% and 6%, respectively³⁵.

Not all agricultural water use (e.g. irrigation) is actually consumed as a fraction returns back as runoff to surface waters and/or percolation to groundwater in the catchment. This agricultural water use consumption could be represented by the efficiency of irrigation systems (Shiklomanov, 2000). The efficiency of irrigation system is defined as actual water consumed (as evapotranspiration) as a fraction of the total water applied for irrigation (Equation 23). The efficiency of irrigation systems was chosen as a metric to represent all agricultural water use, as the majority of freshwater withdrawals in the sector are for irrigation. As can be seen in Figure 2.2b, of the total weekly water withdrawals in New Zealand 46% is for irrigation and a mere 2% for stock water requirements³⁶. As such, the consumption of stock water was assumed to be insignificant when compared to irrigation.

Based on New Zealand irrigation efficiency research (refer Appendix H) for various irrigation systems, it was found that the efficiency of spray irrigators (centre-pivot systems only) vary from 78% to 96% and flood irrigation schemes (border strip systems only) vary from 13% to 56%³⁷. It would therefore be feasible to assume 80% and 50% efficiency rate, for spray and flood irrigators respectively, based on the best practice standards for irrigation (McIndoe, 2002). Spray irrigation systems account for the majority of irrigation systems used in New Zealand at 74%, followed by flood irrigation at 18% (McIndoe, 2002). Therefore, for the purposes of this research a figure of 80% irrigation efficiency is used to account for the volume of freshwater consumed in agricultural water use³⁸. Moreover, from an administrative perspective an 80% irrigation efficiency allows farmers to change to a more efficient system without the need to alter the calculated consumption volumes. As McIndoe (2002) noted, all

³⁵ Note: The hydroelectricity component refers to a singular scheme that discharges freshwater to the sea, all other hydropower schemes in New Zealand are non-consumptive, thus the hydropower component of Figure 7b is not included here.

³⁶ Note: stock water is usually a permitted activity that does not require a resource consent. As such this figure will be under representative of the total freshwater consumption due to stock water requirements.

³⁷ Micro-irrigation systems (7% of national systems), and unspecified irrigators (6% of systems) were not accounted for due to lack of data.

³⁸ It is important to note that there is no distinction between crop types, as this was found to a minimal effect on achievable application efficiency

irrigators (with the exception of travelling guns³⁹) should be able to achieve 80% efficiency. Irrigation system efficiency is defined in accordance with McIndoe (2002) as follows:

Equation 23

$$\text{Irrigation System Efficiency} = \frac{\text{Water applied that is stored in root zone}}{\text{Total amount of water delivered to the farm}}$$

Shiklomanov (2000) conducted a global scale assessment of water consumption across the three water user categories: agriculture, domestic, and industry. A comparison of total agricultural withdrawal and consumption⁴⁰ for Australia and Oceania can be seen in Figure 4.6

Figure 4.6. Note, years 2000, 2010 and 2025 are forecasted.

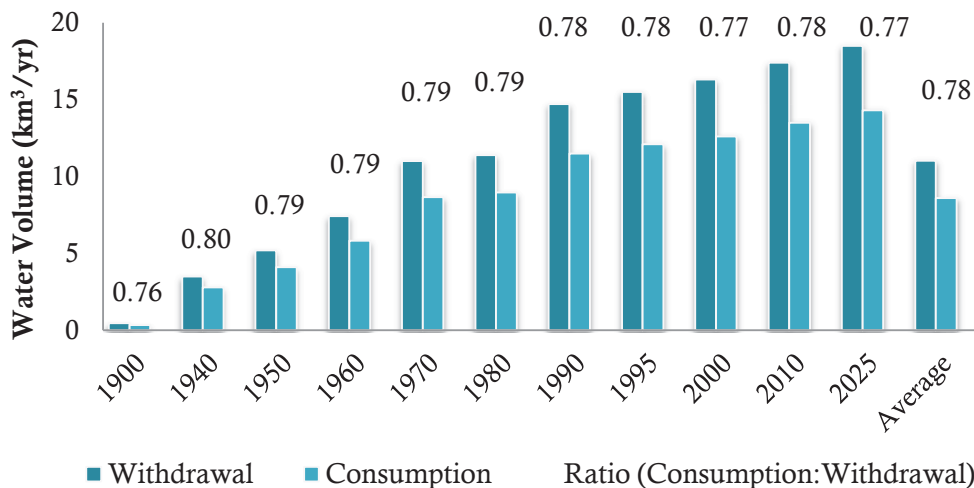


Figure 4.6 Freshwater withdrawal and consumption for agricultural activities in Australia and Oceania (km³/yr) (Shiklomanov, 2000).

It can be seen that the ratio of freshwater consumed to freshwater withdrawn varies from 0.76 to 0.80 from the period 1900 to the forecasted year 2025. The average consumption to withdrawal ratio is 0.78 (Shiklomanov, 2000). This closely compares to the used ratio of 0.80 for agricultural water consumption: withdrawal in New

³⁹ Based on international data.

⁴⁰ No definition of agricultural consumption is provided, however Shiklomanov (2000) acknowledge that the efficiency of irrigation systems is affected by physiographic conditions, serviceable condition of irrigators, watering techniques and crop composition.

Zealand. As such, data produced by Shiklomanov (2000) has been used to estimate domestic and industrial water use efficiencies as described below.

Domestic Water Use Efficiency as a Proxy for Domestic Water Consumption

New Zealand specific data pertaining to the efficiency of domestic water users are at present unavailable. Therefore, an estimate of domestic water use efficiency was derived from Shiklomanov (2000).

The greater part of water withdrawals from the urban network are returned (treated or not) as wastewater. The principal part of domestic water consumption consists of water losses as leakages from the water supply and drainage systems, and evaporation of water used for plants, streets, recreation zones and house lawns (Shiklomanov (2000)). It is important to note that the *ISO 14046* (2014) definition of freshwater consumption does *not* include freshwater that is returned to the same catchment. Therefore, the aforementioned inclusion of freshwater leakages is not in line with the standard. However, due to the lack of research in this area, especially in New Zealand, the aforementioned definition is used. It is recognised, however, that this may cause slight over-estimations of freshwater consumption for domestic water use. It is reported that for domestic purposes water consumption is between 5-10% of the total water intake in modern cities, and can reach as high as 40% to 60% in less developed cities (likely caused by an increase in leakages due to infrastructure). Climatic conditions play a large role in the volume of domestic water that is consumed in this way.

Figure 4.7 shows the ratio of consumption to withdrawal for domestic users from 1990 to 2025 (note: 2000, 2010 and 2025 are predicted) in Australia and Oceania. The consumption to withdrawal ratio ranged from 0.10 to 0.21, with an average of 0.13. For the purposes of this study, a ratio of 0.10 is utilised, based on the aforementioned research and level of development in infrastructure in New Zealand, to estimate domestic freshwater consumption from actual domestic water abstractions.

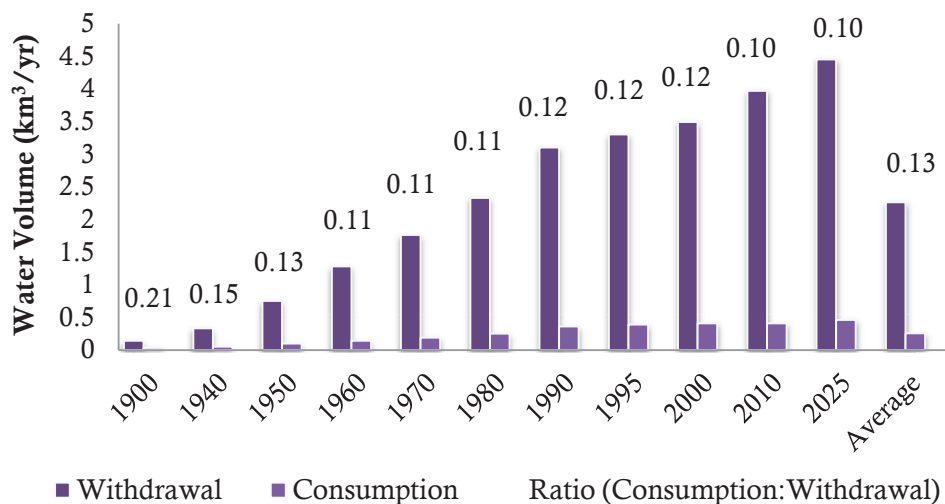


Figure 4.7 Freshwater withdrawal and consumption for domestic activities in Australia and Oceania (km³/yr) (Shiklomanov, 2000).

Industrial Water Use Efficiency as a Proxy for Industrial Water Consumption

New Zealand specific data pertaining to the efficiency of industrial water users was also unable to be located or calculated. Therefore, an estimate of industrial water use efficiency was derived from Shiklomanov (2000).

According to Shiklomanov (2000), the principal water use in industry is power generation (thermal and atomic power generation, specifically). It is reported that typical industrial water consumption varied between 5% and 20%, but in some industries could reach as high as 30% to 40% (Shiklomanov, 2000). Figure 4.8 shows estimates of water consumption figures for industry for the Australia and Oceania regions. The ratio of consumption to withdrawal ranged from 0.06 to 0.15, with an average of 0.11. As New Zealand's primary power producer is hydro-electricity, where freshwater is typically returned to the source soon after withdrawal (therefore not consumed), the ratio of 0.10 was deemed an appropriate estimation of industrial water consumption as fraction of actual industrial water abstractions. An assessment of the water footprint of hydro-electric power generation throughout New Zealand, was conducted by Herath et al. (2011b). It was found that the average water footprint of New Zealand hydro-electric power (GJ) ranged from $1.55e^{-9}$ km³/GJ to $6.05e^{-9}$ km³/GJ depending on the method employed. Thus the volume of consumptive

freshwater used for power generation in New Zealand is lower than that calculated by Shiklomanov (2000) which might result in a slight overestimation of the ratio of consumption to withdrawal.

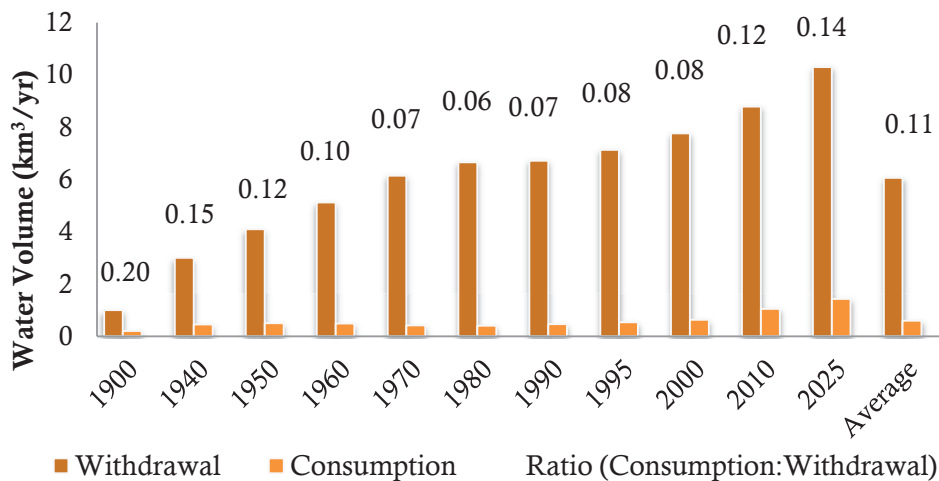


Figure 4.8 Freshwater withdrawal and consumption for industrial activities in Australia and Oceania (km³/yr) (Shiklomanov, 2000).

'Other' Water Use Efficiency as a Proxy for 'Other' Water Consumption

There is a lack of information about the 'other' consumptive freshwater takes across New Zealand and in the literature (Shiklomanov, 2000). In order to account for efficiency of the 'other' water category, a withdrawal to consumption ratio of 0.10 was used, in line with domestic and industrial water uses.

4.4 Quantification of Blue Water Footprint of 1 m³ of Concrete

In accordance with the method of Hoekstra et al. (2011), the volumetric blue water footprint of concrete production was quantified and normalised, using 1 m³ of concrete as the functional unit. The green and grey water footprints were omitted from the analysis. The green water footprint was assessed to be negligible in this case of concrete batching plants. According to the WFN method (Hoekstra et al., 2011), stored rainwater at each concrete plant is considered blue water as opposed to green water. The rationale is that, had it not been stored, the water would have become run-

off to a surface or ground water body thus meeting the criteria for blue water⁴¹ (Hoekstra, 2011). In the case of grey water, discharges from batching plants are typically alkaline. If significant, these are required to be neutralised before entering water bodies. Of the 27 concrete batching plants in the study, only one plant was required to do so before discharge into the industrial wastewater system, during the 2013 study period. This process is not included in the system boundary of the case study and as such is omitted from the environmental impact assessment.

In accordance with Equation 1, the consumptive blue water used by each concrete batching plant was accounted for using data pertaining to the volume of evaporation from storage ponds, the volume of freshwater incorporated into the product itself, and the volume of lost return flow (i.e. volume of 'wastewater' transported to other batching plants). Following the WFN method, each respective plant's process water footprint was then divided by the plant's total concrete production for the year 2013. This enabled the blue water footprint of 1 m³ of ready mix concrete to be quantified and displayed in 'm³ water per m³ concrete', following Equation 23.

Equation 24

$$WF_{blue,concrete} = \frac{WF_{blue}}{Concrete\ Production\ Volume} \text{ [m}^3\text{water/m}^3\text{product]}$$

4.5 Assessing Environmental Impact of 1 m³ Concrete

Four water footprint methods (Table 3.4) were used to assess the environmental impacts of water used by concrete plants at three spatial scales. It is important to note that only the resource related *environmental* impacts associated with blue water consumption are considered in this case study. No consideration is given to any water quality degradative effects from water consumption in the studied concrete production plants. Further, no consideration is given to any social and/or economic impacts of water use and consumption in concrete production. In the case of concrete and cement

⁴¹ Hoekstra et al. (2011) note, however, that if this harvested rainwater is being stored for agricultural purposes then it is to be categorised as 'green water' as some of the water may be stored in the soil profile.

production, however, an economic impact assessment has been conducted by the New Zealand Institute of Economic Research (2008).

Table 4.1 presents a summary of local water use and hydrological data used for the environmental impact of 1 m³ of concrete at the three different spatial scales. All the data used to calculate the blue water footprint and associated environmental impacts of 1 m³ concrete production were based on average annual rather than monthly quantities. This is mainly due to the unavailability of many of the required datasets at the monthly scale. It is acknowledged, however, that data at the monthly scale would provide more robust results (Hoekstra et al., 2011; Boulay et al., 2016).

4.5.1 Water Footprint Network's Sustainability Assessment

According to the method of the WFN (Hoekstra et al, 2011), the blue water footprint impact index was calculated at the three spatial scales. The data required for these calculations are described below.

Local Blue Water Availability

In accordance with Equation 3 (refer to section 3.2.1), the blue water availability (WA_{blue}) for each spatial scale was calculated as the natural runoff (R_{nat}) minus the environmental flow requirement (EFR). With regard to EFR , in this instance, a value of 80% R_{nat} was used based on the precautionary default value stipulated in Appendix V of *The Water Footprint Assessment Manual: Setting the Global Standard* (Hoekstra et al., 2011).

Local Blue Water Footprint

A blue water footprint will have an adverse environmental impact for a specific catchment/boundary, and a specific period, if the blue water footprint exceeds the local blue water availability, causing EFR to be violated in that catchment/boundary (Hoekstra et al., (2011). In order to calculate the total blue water footprint in a specified boundary ($\sum WF_{blue,boundary}$), the WFN suggest aggregating the blue water footprints for all process steps within that boundary (Equation 2). However, given that the case study spans a multitude of boundaries (and at different spatial scales), the conventional calculation proposed by Hoekstra et al. (2011) was deemed too data and

resource demanding, and thus beyond the scope of this assessment. This level of detail was unfortunately not feasible for this study, and proved to be a significant limitation when applying the WFN method to finer resolutions.

Alternatively, the total blue water footprint of a selected boundary was calculated based on the consented surface and groundwater abstractions for different water user groups (agriculture, domestic, industry and 'other'), as follows:

Equation 25

$$\sum WF_{blue,boundary}[x,t] = eff * fc * CWA \text{ [volume/time]}$$

Where the *CWA* is the total volumes of consented surface water and groundwater abstractions for different main water user groups (agriculture, domestic, industry and 'other') per selected boundary, *fc* is the fraction of consented water estimated to be actually abstracted for water use by different water user groups, and *eff* is the fraction of actual abstracted water estimated to be consumed (i.e. abstraction – return flow) by different water groups. These required datasets are described in section 4.3.2 above.

Characterising the Volumetric Blue Water Footprint of 1 m³ Concrete

The volumetric water footprints of 1 m³ concrete (*WF_{blue,concrete}*) for each batching plant were then characterised by the blue water scarcity (*WS_{blue}*) accounting for local water use and hydrological conditions. The blue water scarcity (*WS_{blue}*) for a specified boundary was calculated by dividing the estimated total blue water footprint of all processes ($\sum WF_{blue,boundary}$) (Equation 25) with the blue water availability (*WA_{blue}*) in the specified boundary. The respective blue water footprints of 1 m³ concrete (*WF_{blue,concrete}*) for each batching plant were then multiplied with the blue water scarcities (*WS_{blue}*) of the specified boundary (refer to Equation 5 for calculation of *WFII_{blue}* of specified boundaries) as follows:

Equation 26

$$Ch. WF_{blue,concrete}(WFII_{blue}) = WS_{blue} * WF_{blue,concrete} \text{ [-]}$$

4.5.2 Water Stress Index (WSI) of Pfister et al. 2009

A stress weighted water footprint was quantified to assess the local environmental impacts associated with the production of 1 m³ of ready mix concrete production at three spatial scales (refer to section 3.3.1).

Calculating the Withdrawal to Availability Ratio

The Water Stress Index (*WSI*) developed by Pfister et al. (2009) advances the *WTA* ratio created by Raskin et al. (1997). In accordance with Equation 6, the *WTA* ratio was calculated as the ratio of water use (*WU*) to water availability (*WA*). In this instance, total water use ($\sum WU_{ij}$) was calculated as the total volume of actual surface and groundwater takes⁴² (refer to Table 4.1), accounting for the typical percentage of consented water that is actually abstracted for use (for abstraction assumptions refer to section 4.3.2). In this case, total *WA* is considered in the form of natural runoff available for water use (refer to Table 4.1).

Accounting for Variation in Annual and Monthly Precipitation

Further, in accordance with Equation 8, the water stress capacity of rivers and streams is accounted for by calculating New Zealand specific variation in annual and monthly precipitation (Equation 7). In this instance, the variation factor (*VF*) was calculated for the whole of New Zealand at 500m by 500m grid cells. This was based on the rainfall data provided at monthly and annual intervals for each grid across New Zealand, over the period 1972 to 2013 (R.S. Westerhoff, personal communication, December 8th, 2015). The original data were acquired from NIWA's Virtual Climate Station (Tait, Henderson, Turner, & Zheng, 2006). The *VFs* per grid cell were aggregated to the specified boundaries, and weighted by the mean annual precipitation in each grid cell within the specified boundary (using Arc GIS v10.2). The average *VF* for the whole of New Zealand was found to be 1.29 compared with the median variation factor of 1.80 calculated according to Pfister et al. (2009). Figure 4.9 and Figure 4.10 show the *VF* as calculated for the North and South Islands of New Zealand, respectively. The final *VF*'s were applied to *WTA* ratios to calculate *WTA** (Equation 8) for each selected boundary depending on whether the river in the selected boundary was deemed

⁴² User groups identified in the case study are; industry, agriculture, domestic and 'other'

strongly modified, or not (as per the definition provided by Nilsson et al. (2015), refer to Table 3.3).

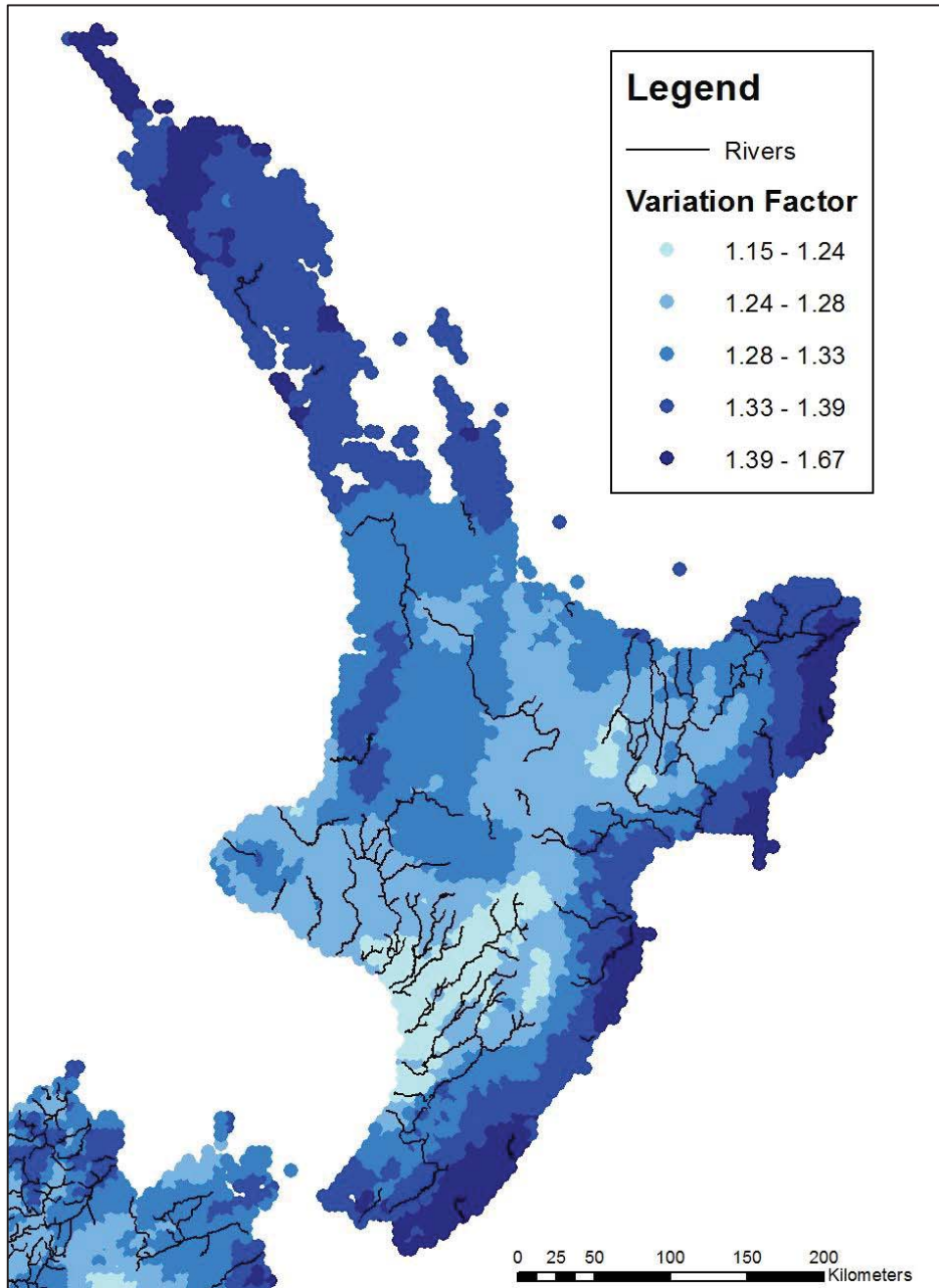


Figure 4.9 The calculated variation factor per 500m grid cell across the North Island of New Zealand, according to Pfister et al. (2009).

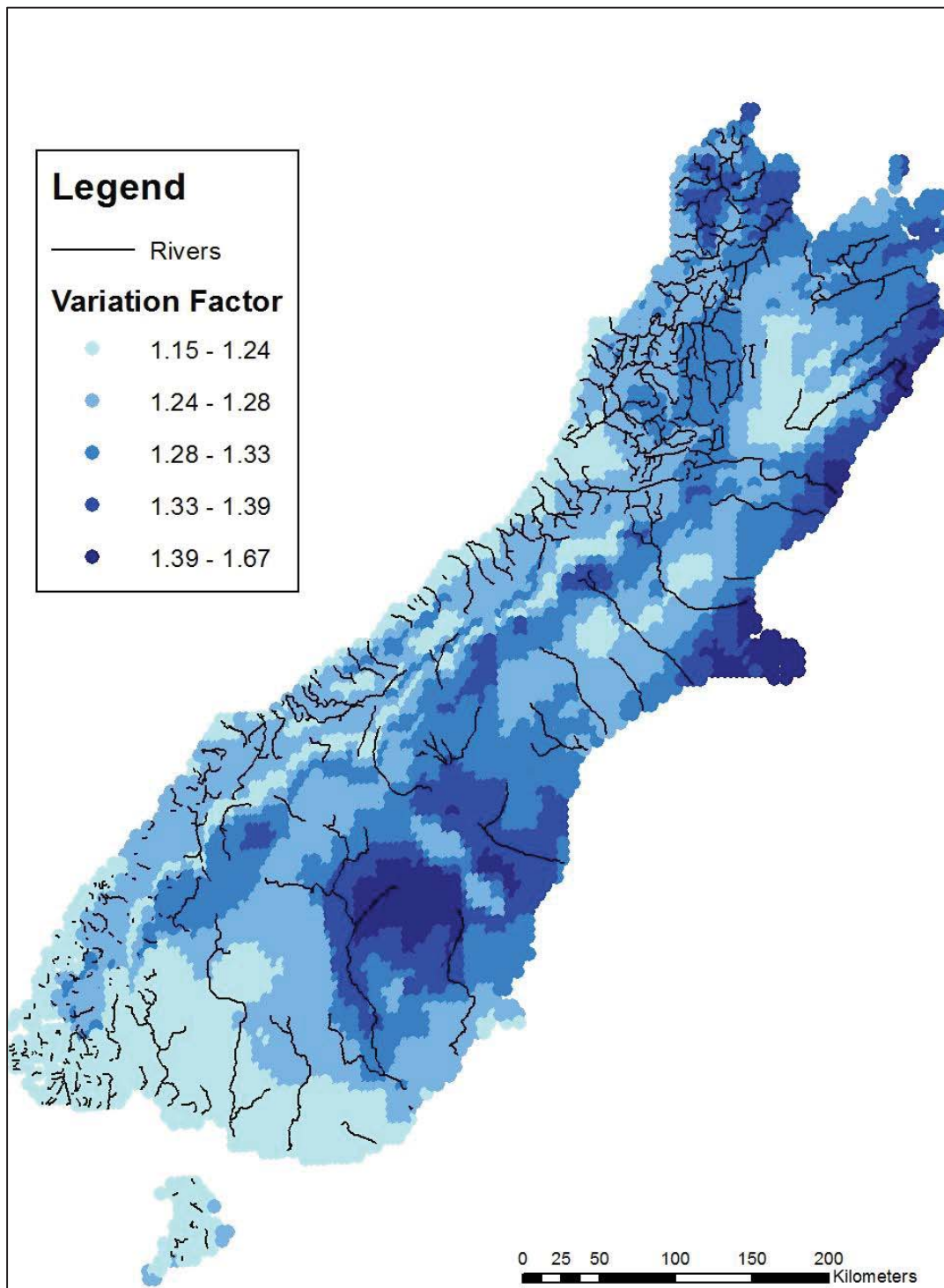


Figure 4.10 The calculated variation factor per 500m grid cell across the South Island of New Zealand, according to Pfister et al. (2009).

It is important to note, that the above study by Nilsson et al. (2005) was conducted at the global scale, and is concerned with larger river systems only. Consequently some of the smaller river systems in New Zealand that have potentially modified flows were not accounted for, or included in the analysis of Pfister et al. (2009). Table 4.3 presents some of these smaller river systems that have hydroelectric power schemes, and potentially modified flows. Only those schemes within selected boundaries have been presented, for a more complete list of affected river in New Zealand refer to Appendix F.

Table 4.3 River systems affected by flow modification in New Zealand (Young, Smart, & Harding, 2004)

River	Scheme	Region
Taieri	Paerau & Patearoa power scheme (hydro)	Otago
Raikai	High bank power scheme (hydro)	Canterbury
Mangahoe	Mangahoe power scheme (hydro)	Manawatu-Wanganui
Patea	Patea power scheme (hydro)	Taranaki
Takaka & Cobb	Cobb power scheme (hydro)	Nelson
Branch	Branch river power scheme (hydro)	Marlborough
Clyde	Clyde dam	Canterbury
Manapouri	Manapouri dam	Southland
Waipori	Waipori power scheme (hydro)	Otago

Characterising the Volumetric Blue Water Footprint of 1 m³ Concrete

The volumetric water footprint of 1 m³ concrete ($WF_{blue,concrete}$) was characterised using the local water stress index (WSI) per specified boundary (i), to quantify the stress-weighted water footprint (Pfister et al., 2009) as follows:

Equation 27

$$Ch. WF_{blue,concrete}(WSI) = WSI_i \times WF_{blue,concrete} \quad [-]$$

The final $Ch.WF_{blue,concrete}$ (WSI) can be interpreted as the water footprint of 1 m³ concrete weighted according to local water stress.

4.5.3 Water Depletion Index of Berger et al. 2014

The Water Accounting and Vulnerability Evaluation (WAVE) model was employed to assess the local environmental impacts associated with the production of 1 m³ of ready mix concrete production at three spatial scales (refer to section 3.3.2).

Calculating a Modified CTA* and WDI

The traditional consumption-to-availability (CTA) ratio (which relates annual water consumption to availability) is modified to account for the volume of annual useable surface and groundwater stocks in the study area (CTA^*). This modification accounts for effective volumes of dams, lakes, wetlands and groundwater sources available for freshwater use (Berger et al., 2014).

In terms of the volume of surface water stocks, the area of all lakes (which encompasses lakes created by dams) and wetlands were derived from the Freshwater Ecosystems of New Zealand database (D. West, personal communication, August 28th, 2015). Lake and wetland areas were aggregated per selected boundary and multiplied with an effective depth of 5 m or 2 m, respectively, following the recommendation of Berger et al. (2014) (Note: this is also the protocol for the WaterGAP2 global model).

As groundwater stocks are not available at the global scale, adjustment factors for groundwater stocks (AF_{gws}) have been provided (Table 4.4) based on the WHYMAP method (Richts, Struckmeier, & Zaepke, 2011). Groundwater recharge values specific to each selected boundary were not included in the analysis due to the difference in scale. Groundwater management areas were found, in some instances, to vary significantly from surface water management zones, making it difficult to obtain useful data for each specified boundary. At the regional scale, estimations of the volume of groundwater are available for New Zealand. However, no such compilation of groundwater volumes at the FMZ and/or catchment scale were found to be available during this case study.

Whilst recharge could be assumed to be lower for catchments in drier areas of New Zealand (e.g. Canterbury region), a midpoint AF_{gws} of 0.950 was used for all selected boundaries and scales, as groundwater recharge could not be obtained for all the selected boundaries in a consistent manner. This is an area that could benefit from further research.

Table 4.4 Adjustment factors for available groundwater stocks (AF_{gws})

Geological Structure	Annual Recharge (mm)	Scarcity Reduction	AF_{gws}
Major Groundwater Basin	>300	10.00%	0.900
	100-300	7.50%	0.925
Complex Hydrogeological Structure	>300	5.00%	0.950
	100-300	2.50%	0.975
Others		0.00%	1.000

The modified CTA^* (Equation 16) was then used to quantify the water depletion index (WDI) for each specified boundary.

Characterising the Volumetric Blue Water Footprint of 1 m³ Concrete

The volumetric blue water footprint of 1 m³ concrete was characterised using the calculation of WDI per specified boundary(i), to quantify the characterised water footprint $Ch.WF_{blue,concrete}(WDI)$ as follows:

Equation 28

$$Ch.WF_{blue,concrete}(WDI) = WDI \times WF_{blue,concrete} \quad [m^3 \text{water}/m^3 \text{product}]$$

The final $Ch.WF_{blue,concrete}(WDI)$ can be defined as a water footprint weighted according to risk of freshwater depletion.

4.5.4 The AWaRe method: Available WATER REMaining (Boulay et al. (2016))

The recently developed AWaRe method was employed to assess the local environmental impacts associated with the production of 1 m³ of ready mix concrete production at three spatial scales (refer to section 3.3.3).

[Note: At the time of publishing, this method was still in a preliminary version. The calculations as discussed below are correct at time of publication]

Calculating Available Water Minus Demand

The AWaRE method is based on the availability minus the demand (*AMD*) of freshwater (refer Equation 18). In this instance, water availability was represented by natural runoff (refer section 4.3.2.1). Consideration is given to both human water consumption and the environmental flow requirement (*EFR*). The freshwater consumed in each specified boundary were calculated and consistent with the method based on consented surface water and groundwater for main water users (agriculture, industry and domestic) in the specified boundary. This method is described in detail in section 4.3.2. The *EFR*s, in this instance, were calculated using the method of Pastor et al. (2013) for each respective basin, and provided (A-M Boulay, personal communication, May 1st, 2016). Accordingly, the *EFR* for selected basins in New Zealand ranged from 33%, in the Canterbury and West Coast regions, to 37% in the Wellington region.

Characterising the Volumetric Blue Water Footprint of 1 m³ Concrete

The availability minus demand (*AMD*) (and/or *ST_e*) per basin (*i*) is introduced as the characterisation factor in the AWaRE method. The estimates of *AMD_i* for each specified boundary (as per Equation 19) were first normalised by the world average *AMD_{world ave}* of 0.0136 (Equation 21). Finally, the volumetric blue water footprint of 1 m³ concrete was characterised using the estimates of *CF* per specified boundary(*i*), to quantify the characterised water footprint *Ch.WF_{blue,concrete}(AMD)* as follows:

Equation 29

$$Ch.WF_{blue,concrete}(AMD) = CF * WF_{blue,concrete}$$

The characterised blue water footprint *Ch.WF_{blue,concrete}(AMD)*, as calculated above (Equation 28), can be interpreted as a water footprint weighted by the remaining volume of freshwater left for use once demand has been met.

4.6 Statistical Analysis

A statistical analysis was conducted with the overall aim of determining which of the water use and hydrological variables (considered in each water footprint methods) were responsible for the majority of variation between the water footprints. As there are some datasets that are specific to New Zealand and others that use global estimations, this analysis was deemed useful to determine the source of the variation and explore potential issues surrounding data source and accuracy.

4.6.1 Co-efficient of Variation

In order to understand the relative variation in estimates of water footprint by different methods, and at different spatial scales, the co-efficient of variation (CV) was calculated as the ratio of the standard deviation (SD) to the mean (M) of the dataset, as follows:

Equation 30

$$CV = \frac{SD}{M} * 100 \text{ [%]}$$

4.6.2 Variance Based Sensitivity Analysis

The coefficient of determination was calculated and compared across all water footprint methods and spatial scales, to assess which freshwater variables have the greatest effect on the variation in the resultant characterisation factors and characterised water footprints. This analysis was conducted for the six concrete batching plants selected for the environmental impact assessment. A limitation of this analysis arises from the interconnectedness of the freshwater variables used. Subsequently, the variables cannot be considered independent, and this may have a bias effect on the coefficients determined. However, this is considered an indicative analysis to identify the freshwater variables that have an influence on estimates of water footprints at different spatial scales.

It is important to note that the volume of freshwater consumed per batching plant (Equation 1) is used to calculate the variance in the volumetric water footprints of the concrete batching plants, as opposed to the volumetric water footprints themselves

(Figure 5.4). It was determined that due to the nature of the calculation for volumetric water footprints (Equation 23) that dividing freshwater consumption by concrete production would cause the underlying relationships to be masked (as x is a function of y). In order to evaluate why the volumetric water footprint varied between plants, it was deemed necessary to consider the underlying relationships between relevant variables used to calculate the volumetric water footprints.

Chapter 5

Results

5.1 The Volumetric Blue Water Footprint of 1 m³ of Ready Mix Concrete

The volumetric blue water footprints of 1 m³ concrete were calculated for the 27 concrete batching plants using the method of Hoekstra et al. (2011) (discussed in section 4.4).

In order to calculate the volumetric blue water footprint of 1 m³ of concrete (m³ water per m³ concrete), it was first necessary to quantify the volume of freshwater *consumed* by each respective plant (m³/year). Figure 5.1 shows the water consumption of *each plant*, as a percentage of the total water consumption across all batching plants. Where consumption is calculated according to Equation 1.

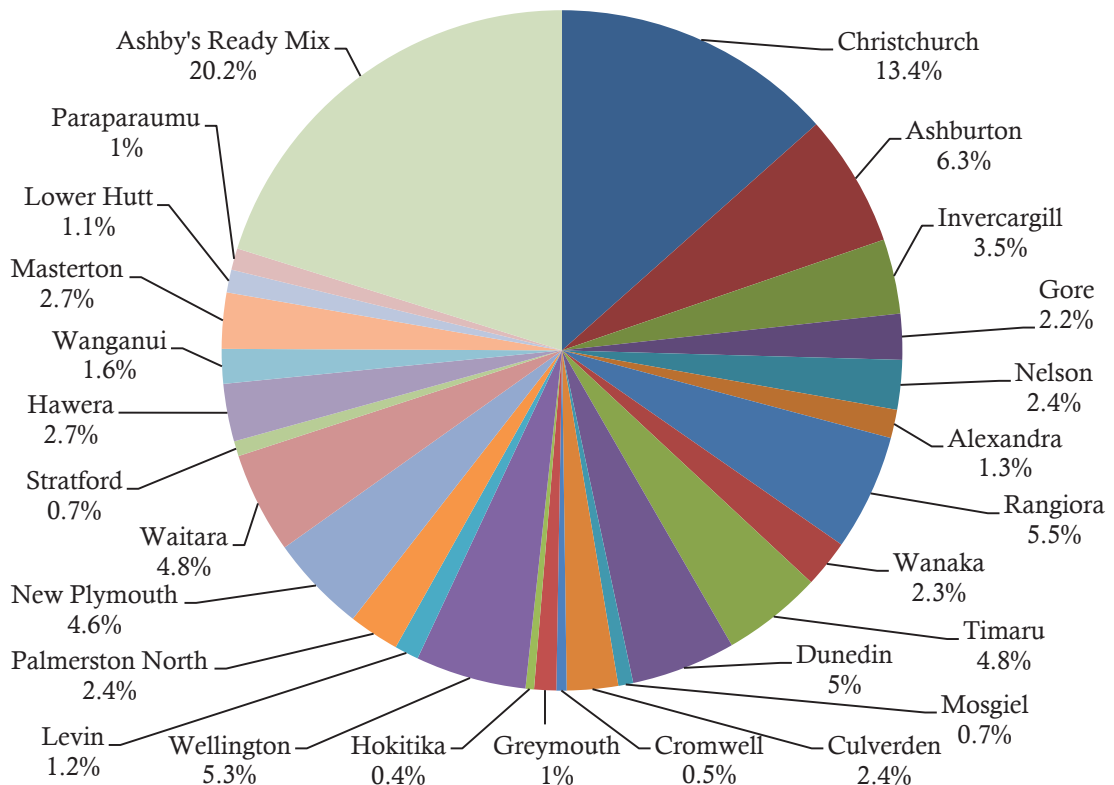


Figure 5.1 Contribution of Allied Concrete batching plants to the total volume of freshwater consumed (m³/year), for the 2013 period.

Table 5.1 summarises a regression analysis comparing the volume of freshwater consumed (refer to Figure 5.1) to variables accounted for in the calculation of the blue water footprint for each concrete batching plant (refer to Equation 1). The purpose is to understand which factors are accounting for the variability in freshwater consumption between concrete batching plants.

Table 5.1 A summary of the regression analysis between the volume of freshwater consumed at each of the 27 concrete batching plants and the variables accounted for to calculate consumption (% of variation).

	Freshwater Consumed (m ³ /year)
Blue Water Evaporation (m ³ /year)	0.64%
Blue Water Incorporated in Product (m ³ /year)	99.95%
Lost Return Flow (m ³ /year)	2.14%

As can be seen in Table 5.1, the volume of blue water incorporated into the product accounts for the largest proportion of variation in consumption across batching plants. The total volume of freshwater incorporated into the product, comprises both freshwater from the reticulated network (40%) and recycled water (60%).

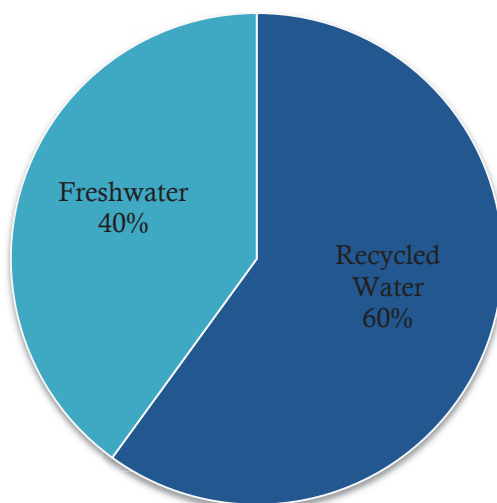


Figure 5.2 The percentage of freshwater and recycled water withdrawn, averaged across the 27 concrete batching plants, for the 2013 period.

Using the guidelines stipulated in *The Water Footprint Assessment Manual: Setting the Global Standard (2011)*, the volume of freshwater consumption shown in Figure 5.1 can be adapted to account for the volumes of concrete production per plant (refer Equation 23). Figure 5.3 represents the proportion of total concrete produced at each plant.

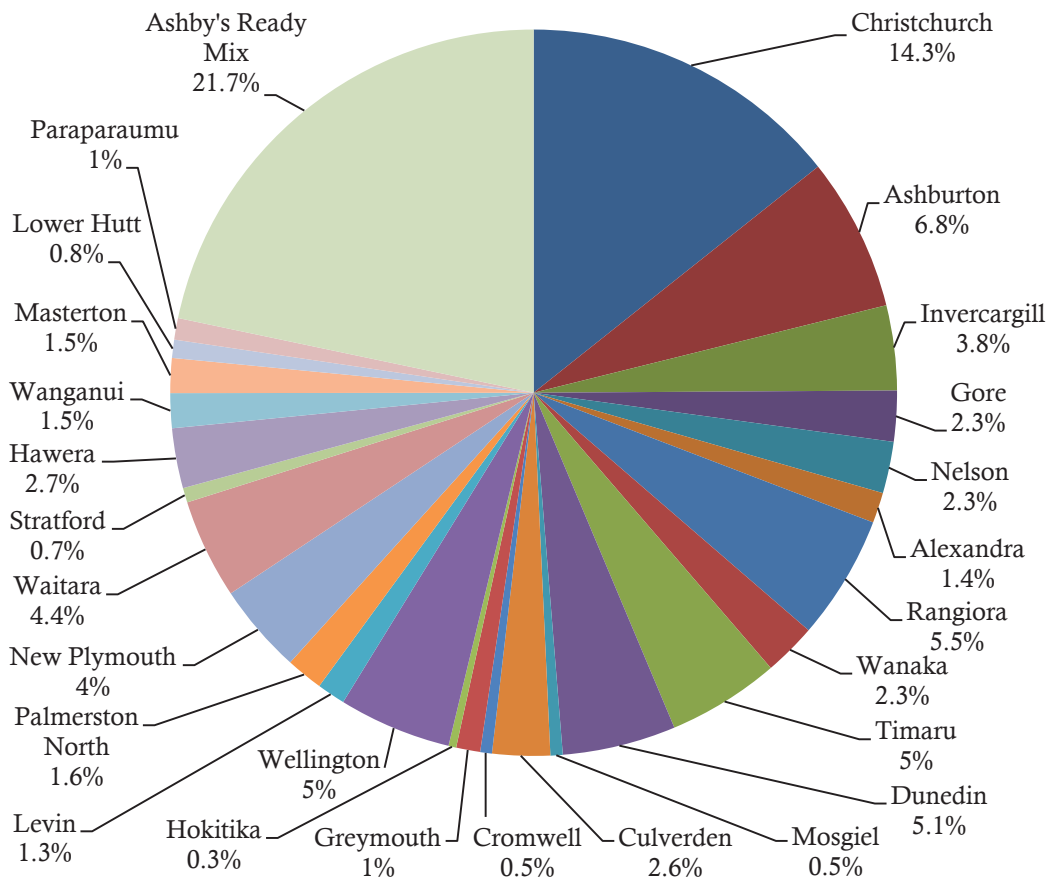


Figure 5.3 Contribution of Allied Concrete batching plants to total ready mix concrete production (m^3 ready mix concrete/year) during the 2013 period.

Accounting for the volume of freshwater consumption (Figure 5.1) divided by the volume of concrete production (Figure 5.3), in accordance with Equation 24, the volumetric water footprint of each concrete batching plant (m^3 water per m^3 concrete) can be seen in Figure 5.4. The water footprint of concrete varies between batching plants. Eight batching plants have water footprints greater than the average, of 0.18 m^3 water per m^3 concrete. Three of plants in particular: Mosgiel, Palmerston North and

Masterton, can be seen to be greater than all other batching plants at 0.22, 0.24 and 0.29 m³ water per m³ concrete, respectively. The lowest volumetric blue water footprint of 0.15 m³ water per m³ concrete was calculated at the Ashburton, Levin, Cromwell and Ashby's Ready Mix batching plants.

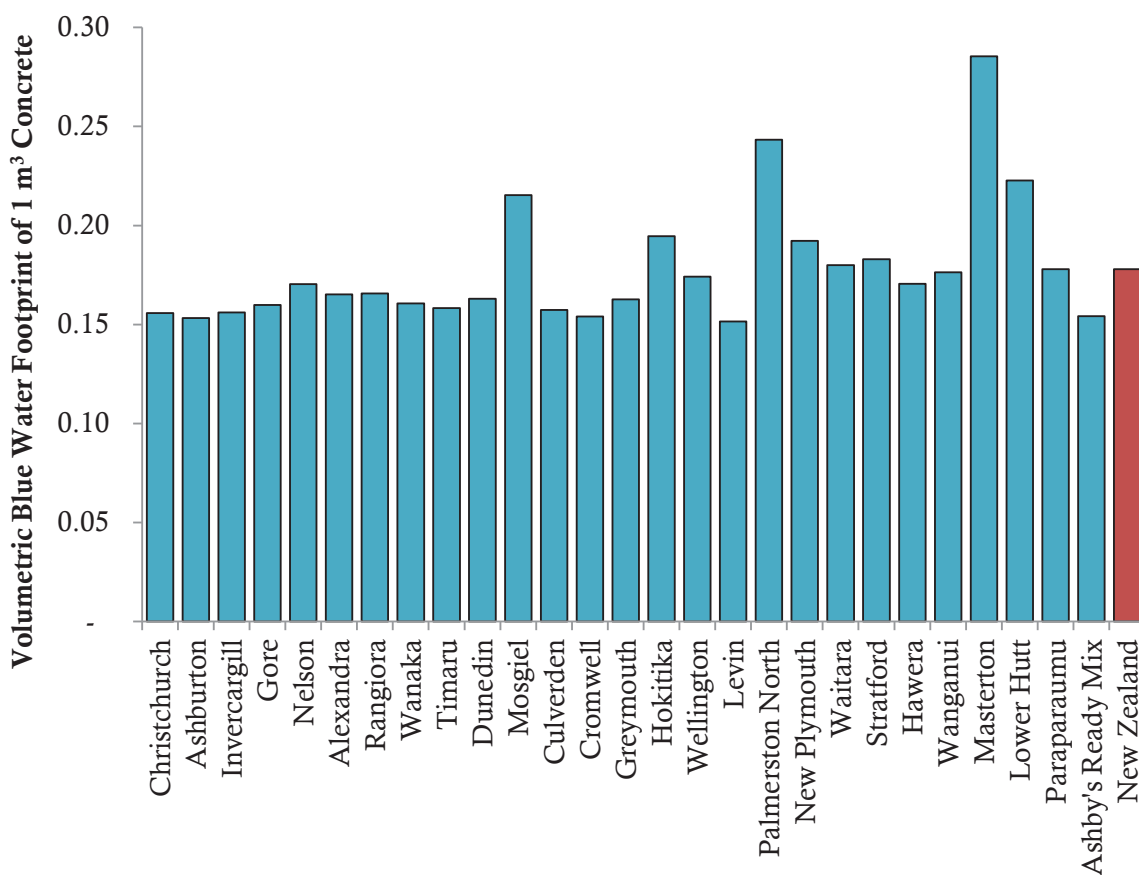


Figure 5.4 The volumetric blue water footprint of 1 m³ of concrete throughout 27 New Zealand batching plants (2013 period), according to the WFN method.

For comparative purposes, the volumetric water footprints shown in Figure 5.4 have been normalised with the average. Figure 5.5 shows the normalised water footprints for each of the 27 concrete batching plants throughout New Zealand alongside the volume of concrete production and freshwater consumption (shown as a % of the total across the 27 batching plants). Where concrete produced exceeds the volume of freshwater consumed, the water footprint of 1 m³ concrete across the 27 concrete

batching plants can be seen to be below the average of one. Where freshwater consumed exceeds the volume of concrete produced the water footprint can be seen to be closer (or above) the average, compared to plants of a similar size with regards to concrete production and freshwater consumption volumes.

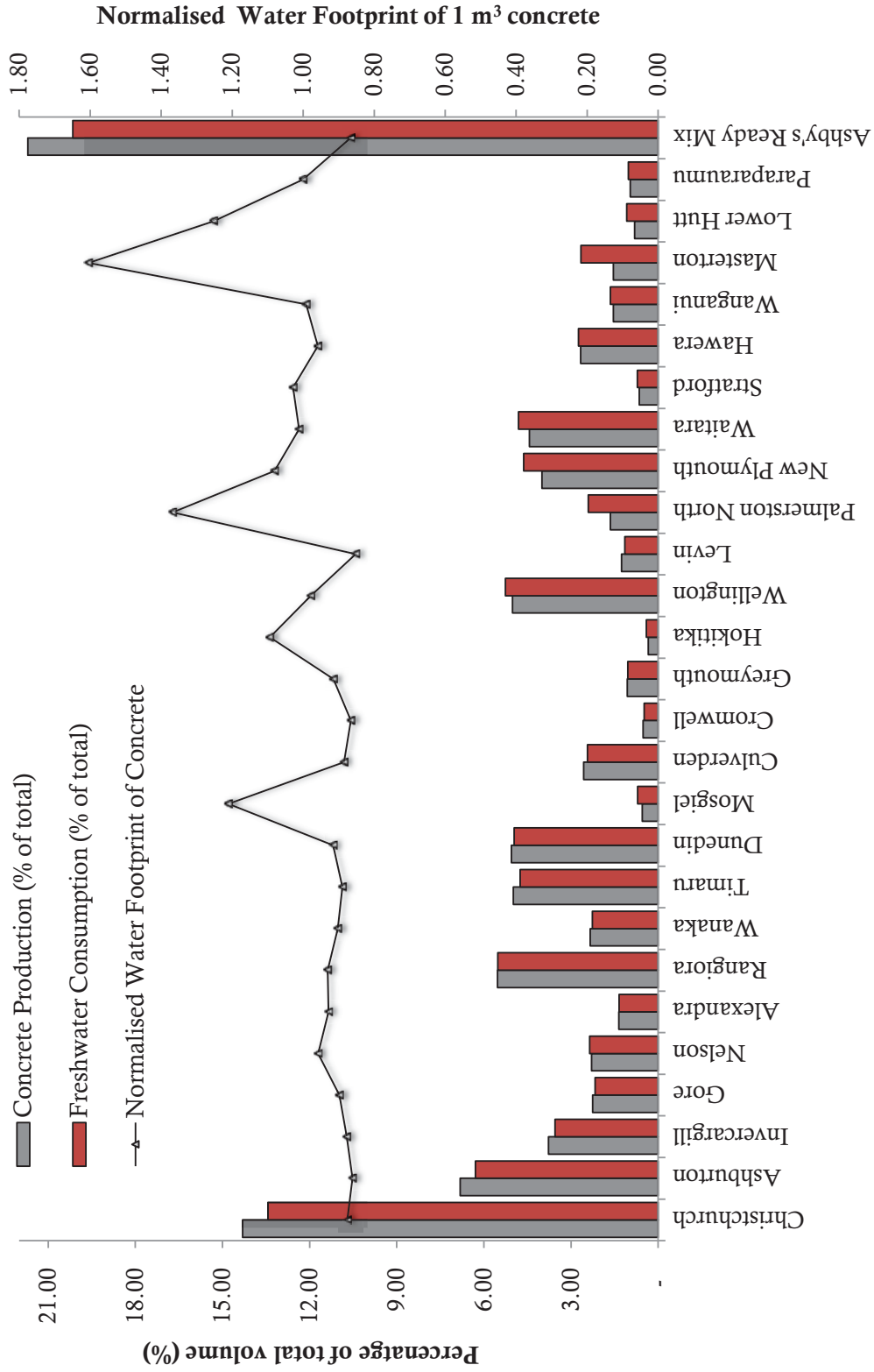


Figure 5.5 A comparison of concrete production and freshwater consumption volumes (as a % of the total) with the normalised water footprint of 1 m³ concrete at concrete batching plants throughout New Zealand, for the 2013 period.

5.2 Environmental Impact Assessment of the Concrete Water Footprint

5.2.1 Selection of Concrete Batching Plants

For the purposes of the environmental impact assessment, a sub-set of six concrete batching plants located throughout New Zealand were selected. Table 5.2 provides a reference of these concrete batching plants and the associated labelling at the FMZ, catchment or regional scale.

Table 5.2 A reference for the labelling of batching plants at different scales.

Batching Plant	Freshwater Management Zone Scale	Catchment Scale	Regional Scale
Ashburton	Ashburton	Ashburton	Canterbury
Greymouth	Grey	Greymouth	West Coast
Palmerston North	Manawatu	Manawatu	Horizons
New Plymouth	Waiwhakaiho	Waiwhakaiho	Taranaki
Wanganui	Wanganui	Wanganui	Horizons
Masterton	Ruamahanga	Ruamahanga	Wellington

These six plants were selected to represent hydrologically diverse regions of New Zealand, in order to capture any variability in the water scarcity as a result of spatial scale. They also span the range of volumetric water footprints (refer Figure 5.4). The criteria used to select the six batching plants were (a) a range of volumetric blue water footprints, (b) data quality of the underlying datasets, (c) spatial diversity of the plants.

5.2.2 Schematic Flow Diagrams of Withdrawals and Consumption

Schematic diagrams showing the proportion of freshwater withdrawn and consumed at the 27 concrete batching can be seen in Appendix K. The schematic flow diagrams for the selected six plants chosen for environmental impact assessment can be seen below (Figures 5.6 through to 5.11). Freshwater withdrawals are either made from the reticulated network and/or from rainwater storage tanks on-site⁴³ (refer to Figure 5.3). It can be seen below that the majority of plants, with the exception of Wanganui,

⁴³ Line thickness is indicative of volume.

withdraw the majority of freshwater from the reticulated network. Freshwater consumption occurs in varying volumes, across the 6 batching plants, as water consumed through the product incorporation, evaporation, loss to ground or disposal of wastewater in a few cases.

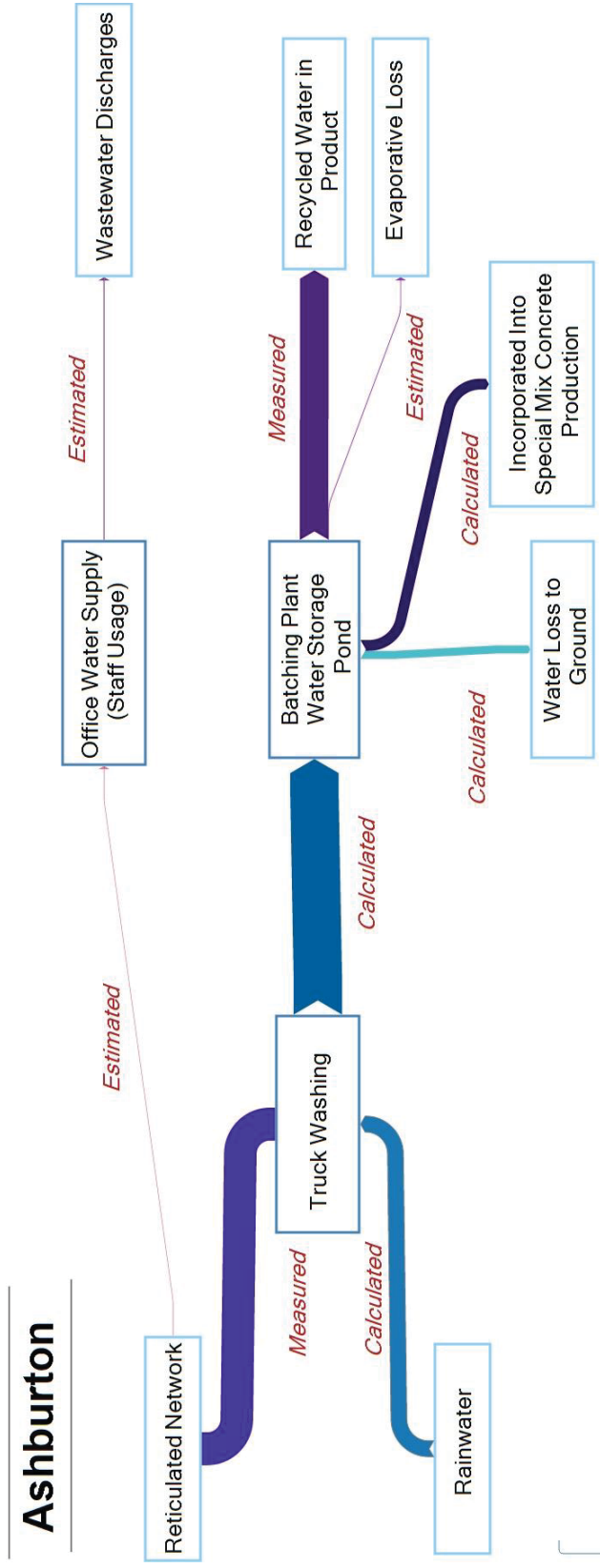


Figure 5.6 Schematic diagrams of freshwater withdrawal and consumption at the Ashburton batching plant, for the 2013 period.

Note: line thickness is indicative of volume (I.e. the thicker the line the greater the volume).

Greymouth

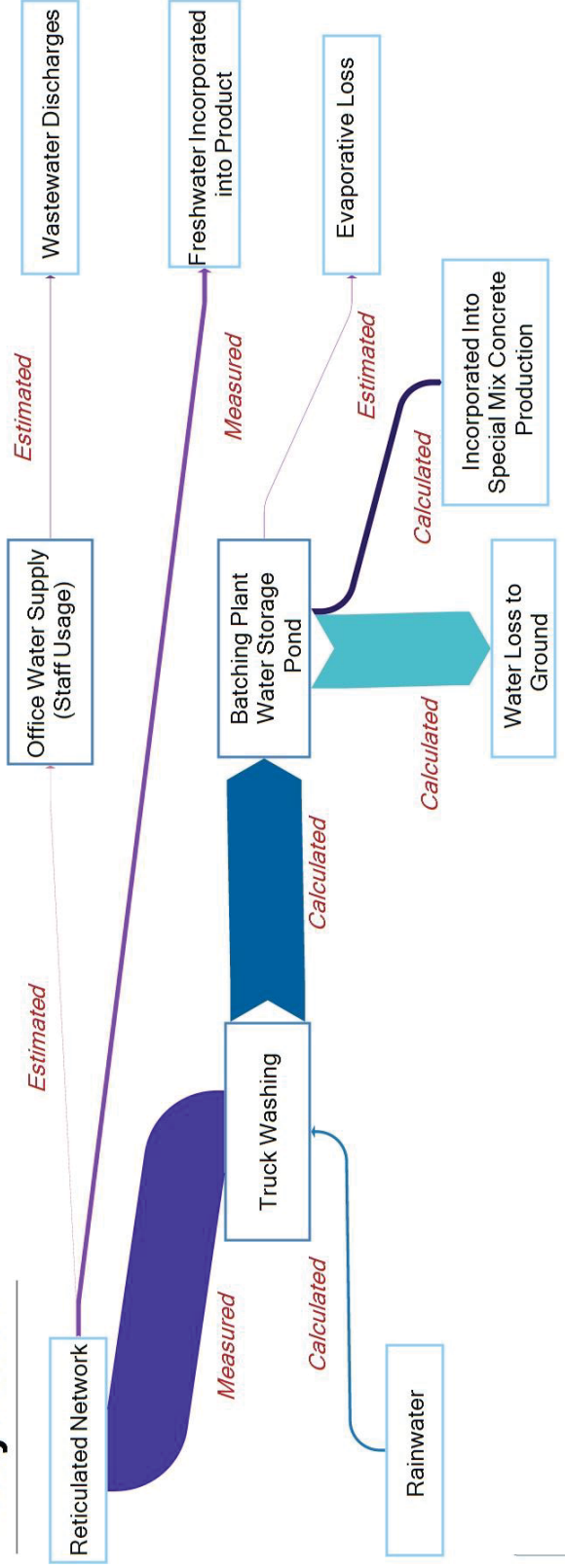


Figure 5.7 Schematic diagrams of freshwater withdrawal and consumption at the Greymouth batching plant, for the 2013 period.

Note: line thickness is indicative of volume (I.e. the thicker the line the greater the volume).

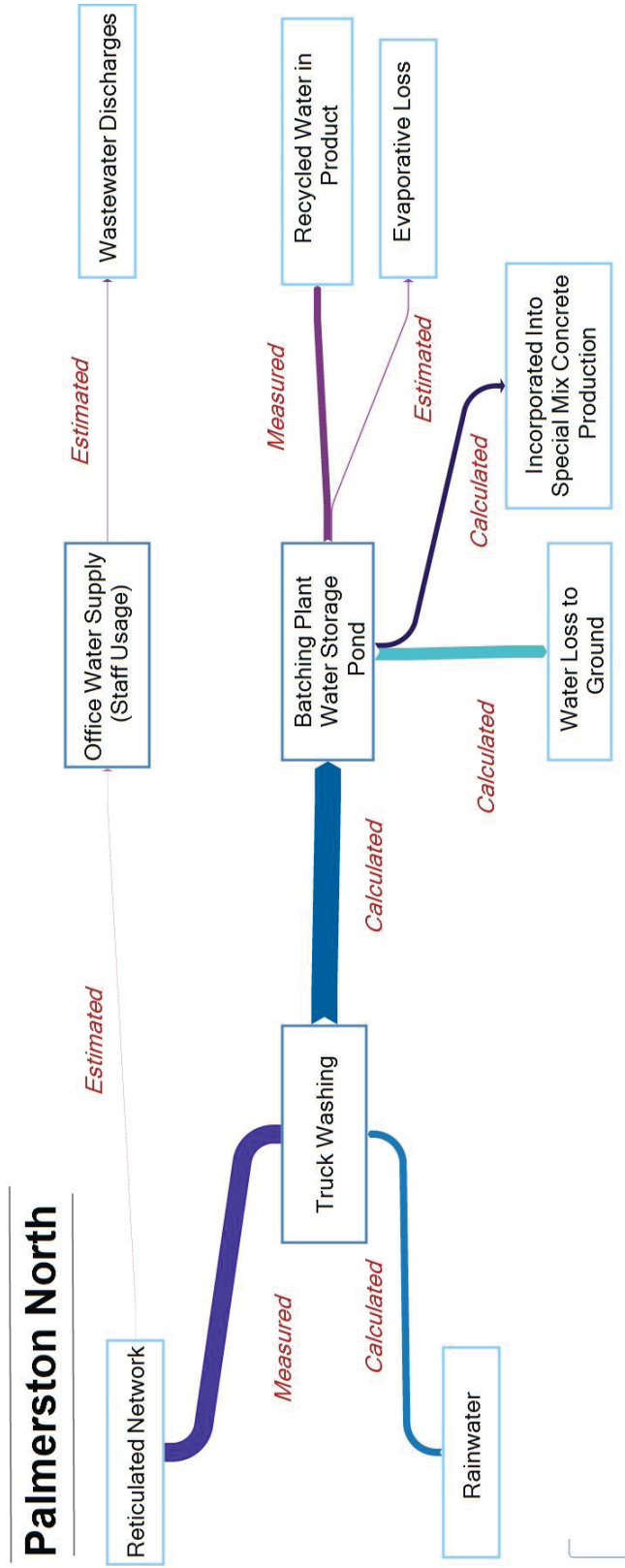


Figure 5.8 Schematic diagrams of freshwater withdrawal and consumption at the Palmerston North batching plant, for the 2013 period.

Note: line thickness is indicative of volume (I.e. the thicker the line the greater the volume).

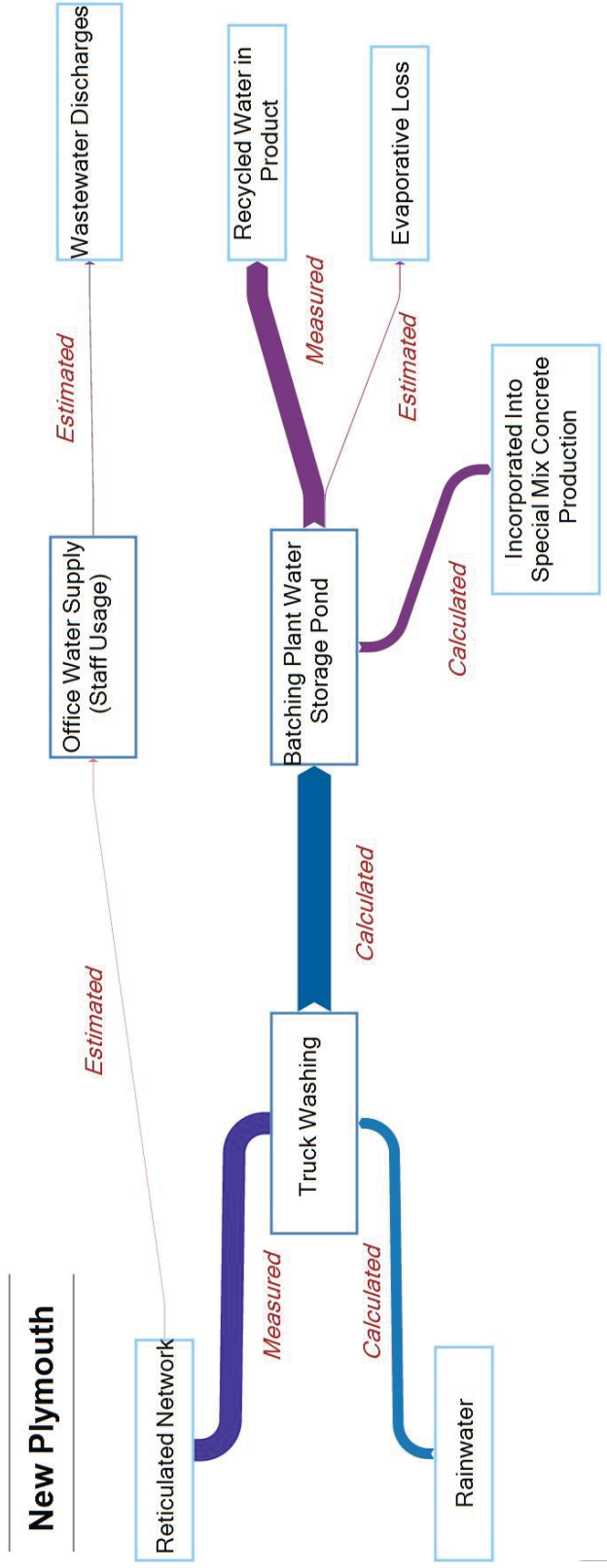


Figure 5.9 Schematic diagrams of freshwater withdrawal and consumption at the New Plymouth batching plant, for the 2013 period.

Note: Line thickness is indicative of volume (I.e. the thicker the line the greater the volume).

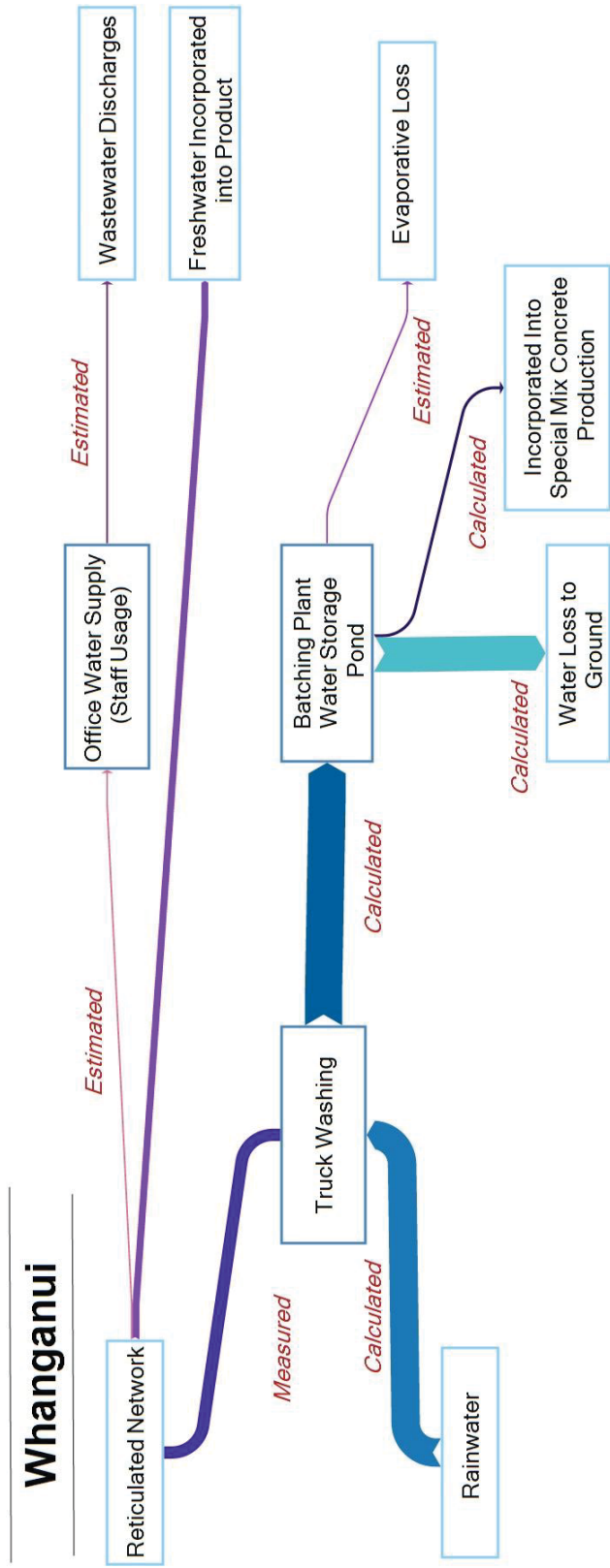


Figure 5.10 Schematic diagrams of freshwater withdrawal and consumption at the Wanganui batching plant, for the 2013 period.

Note: line thickness is indicative of volume (I.e. the thicker the line the greater the volume).

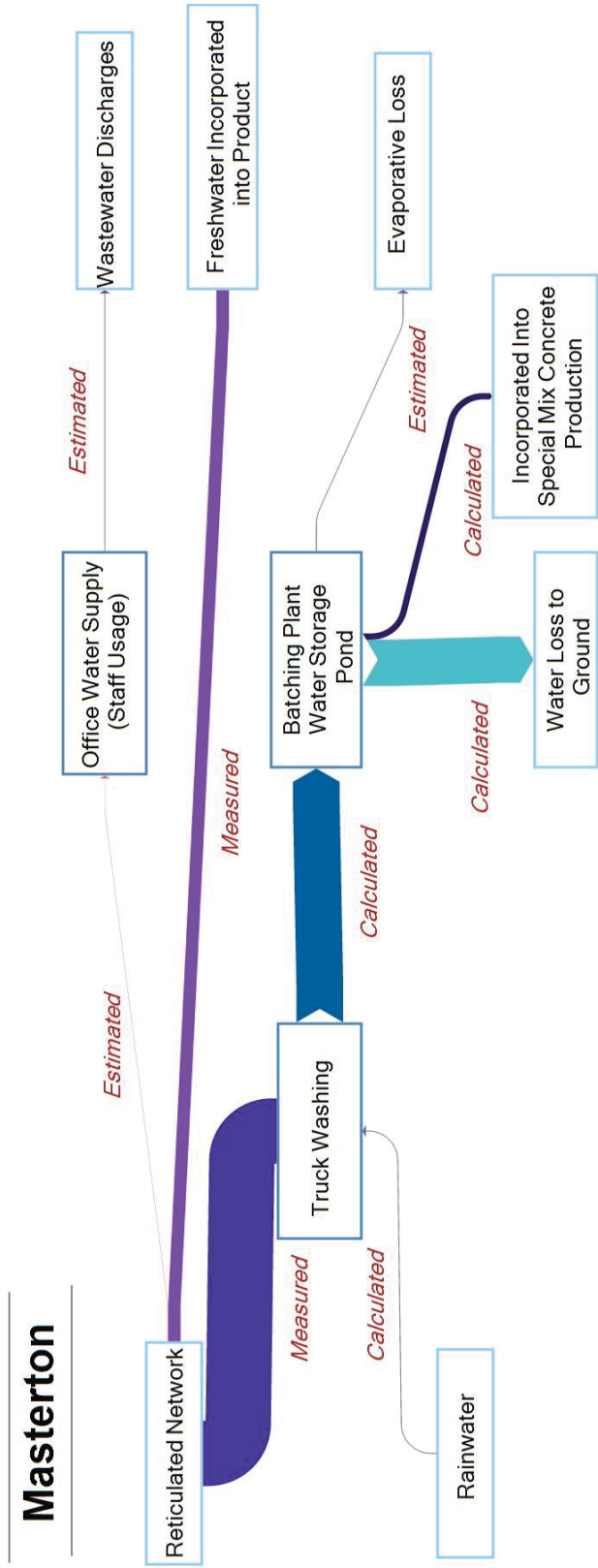


Figure 5.11 Schematic diagrams of freshwater withdrawal and consumption at the Masterton batching plant, for the 2013 period.

Note: line thickness is indicative of volume (I.e. the thicker the line the greater the volume).

5.2.3 Freshwater Management Zone Scale

In order to characterise the volumetric water footprint of 1 m³ concrete (refer Figure 5.14) at the selected concrete batching plants throughout New Zealand, the total volumes of withdrawn, consumed and available freshwater were required for each FMZ where the selected batching plants are located. Figure 5.12 shows the proportion of freshwater available (R_{nat}), withdrawn and consumed in each selected freshwater management zone. It can be seen that the Grey FMZ has the greatest available freshwater, more than ten billion m³ per year. Of this, only a very small volume is withdrawn and/or consumed (0.7%). Of the six selected batching plants, the Ashburton FMZ has the greatest proportion of available freshwater withdrawn and/or consumed on an annual basis (36%). Moreover, the Ashburton FMZ withdraws and consumes almost ten times more than Ruamahanga FMZ, which has the second highest withdrawal and consumption.

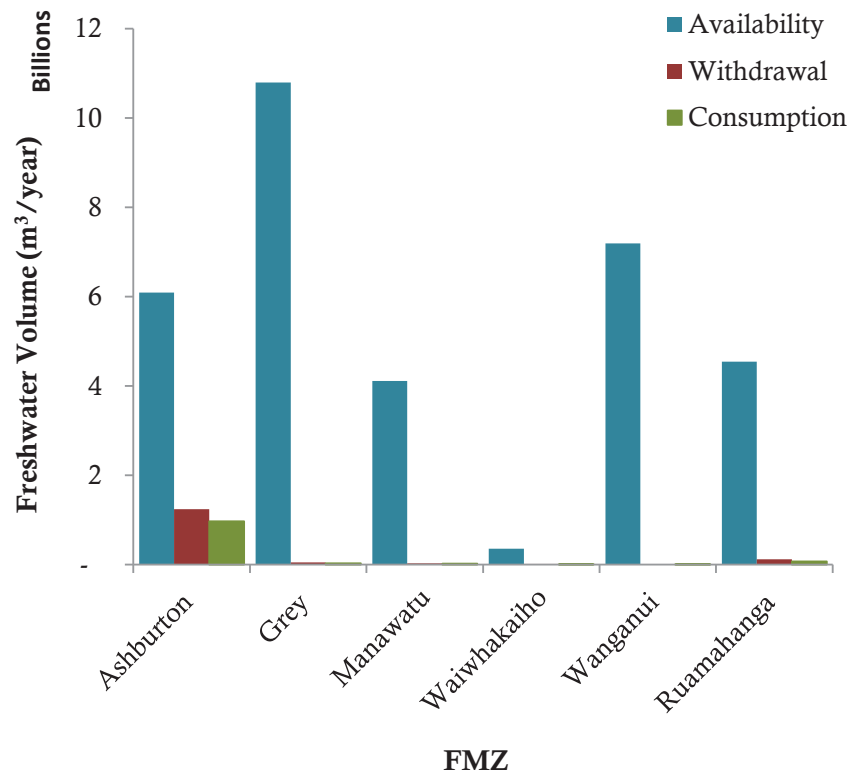


Figure 5.12 Volumes of available, withdrawn and consumed freshwater in selected freshwater management zones (billions m³/ year).

5.2.3.1 Blue Water Footprint Impact Index (Hoekstra et al., 2011)

The blue water scarcity (WS_{blue}) for the six freshwater management zones (FMZs) where the selected plants are located, are shown in Figure 5.13. It can be seen that the Ashburton FMZ has the highest WS_{blue} at 0.79 and is almost ten times greater than the second highest WS_{blue} calculated at the Ruamahanga FMZ. The Wanganui FMZ was found to have the lowest calculated WS_{blue} at just 0.001, followed by Grey, Manawatu and Waiwhakaiho FMZ's at 0.01, 0.02 and 0.03, respectively.

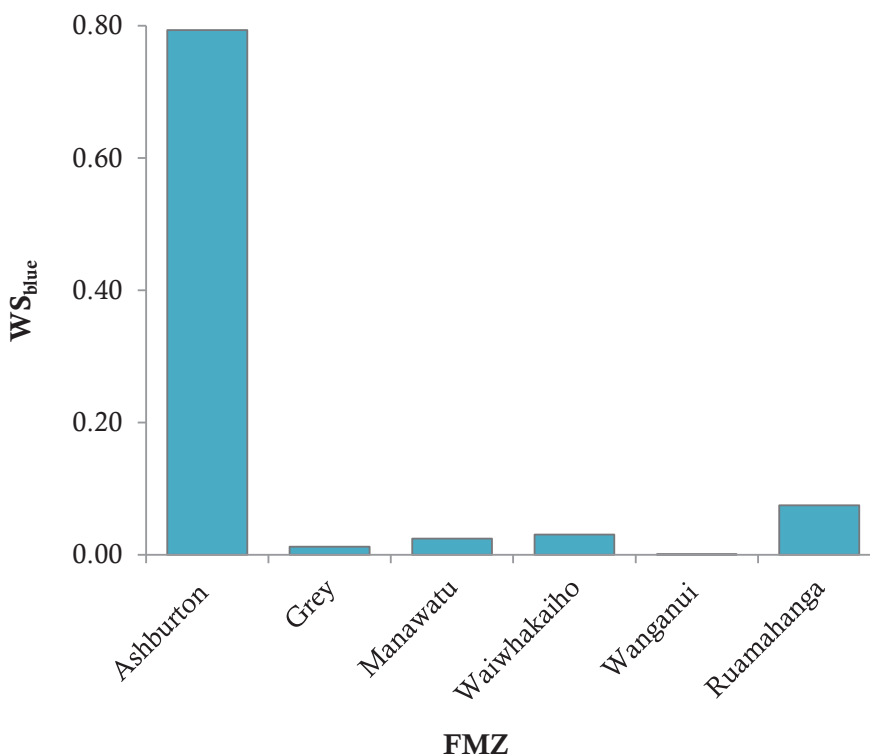


Figure 5.13 The calculated blue water scarcity (WS_{blue}) for selected freshwater management zones.

The characterised blue water footprints ($Ch.WF_{blue,concrete}(WFII)$) of each selected batching plant were calculated (according to Equation 27) and can be seen in Figure 5.14 alongside the volumetric blue water footprint results ($WF_{blue,concrete}$). It can be seen that when the volumetric blue water footprint of concrete is characterised by the

local blue water scarcity index (WS_{blue}) in the freshwater management zones, the results alter markedly between the plants. According to the volumetric water footprints, the Masterton batching plant and the Palmerston North batching plant have the highest water footprints at 0.29 and 0.24 m^3 water/ m^3 concrete, respectively. When characterised by the local WS_{blue} , the Ashburton batching plant (Ashburton FMZ) has the highest characterised water footprint, at 0.12. This is over five times greater than the second highest characterised water footprint calculated for the Masterton batching plant at 0.02. The lowest characterised water footprint was calculated for the Wanganui batching plant at 0.0002.

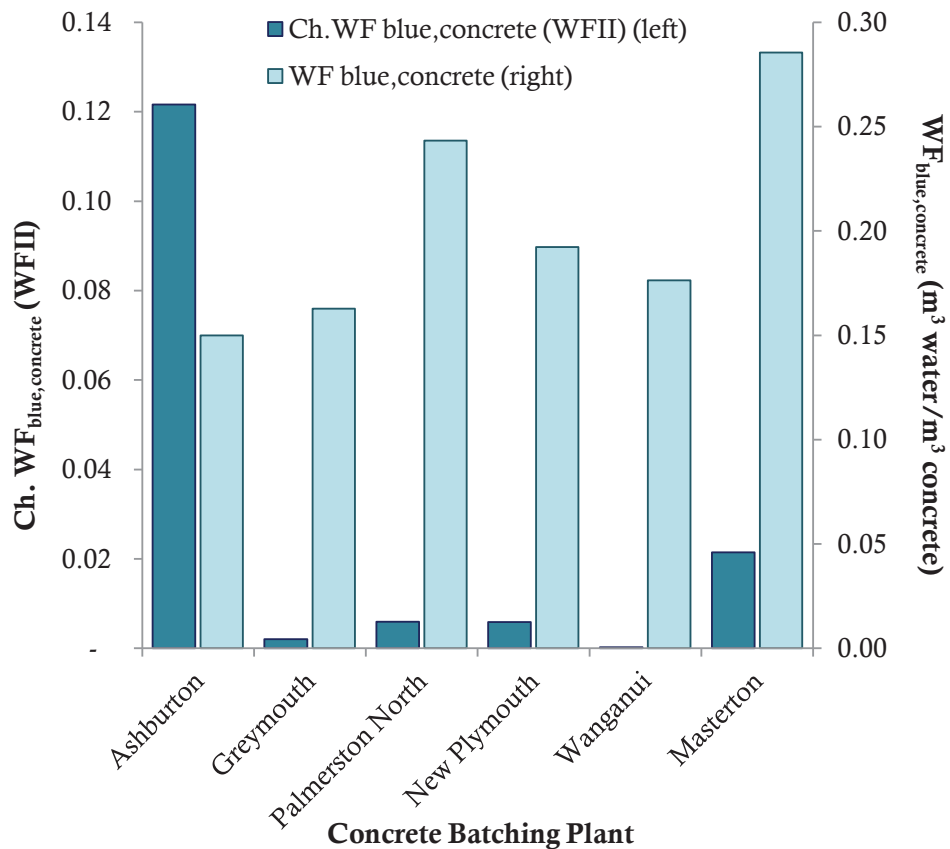


Figure 5.14 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch.WF_{blue,concrete}(WFII)$) at selected freshwater management zones, for the 2013 period.

5.2.3.2 Water Stress Index (Pfister et al. (2009))

The *WSI* calculated for selected FMZ's can be seen in Figure 5.15. Based on the withdrawal to availability ratio (*WTA*), the highest *WSI* highlights the FMZ's with the greatest water stress. It can be seen that the Ashburton FMZ has the highest *WSI*, but that all FMZ's rank quite closely in this instance, ranging from 0.005 in the Grey, Manawatu and Wanganui FMZ's through to 0.008 in the Ashburton FMZ.

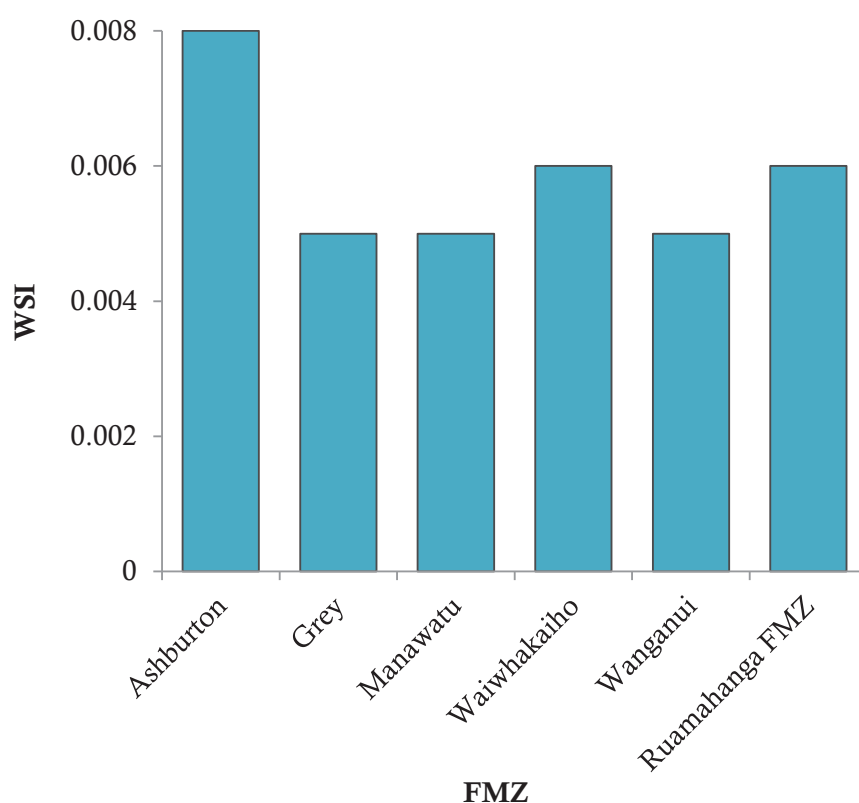


Figure 5.15 The calculated water scarcity index (*WSI*) for selected freshwater management zones.

When the volumetric water footprints of each selected batching plants are characterised using the *WSI* values, the Masterton plant can be seen to have the largest water footprint at 0.0017 (Figure 5.16). The Greymouth plant was found to have the lowest water footprint at 0.0008, after characterisation with the local water stress index.

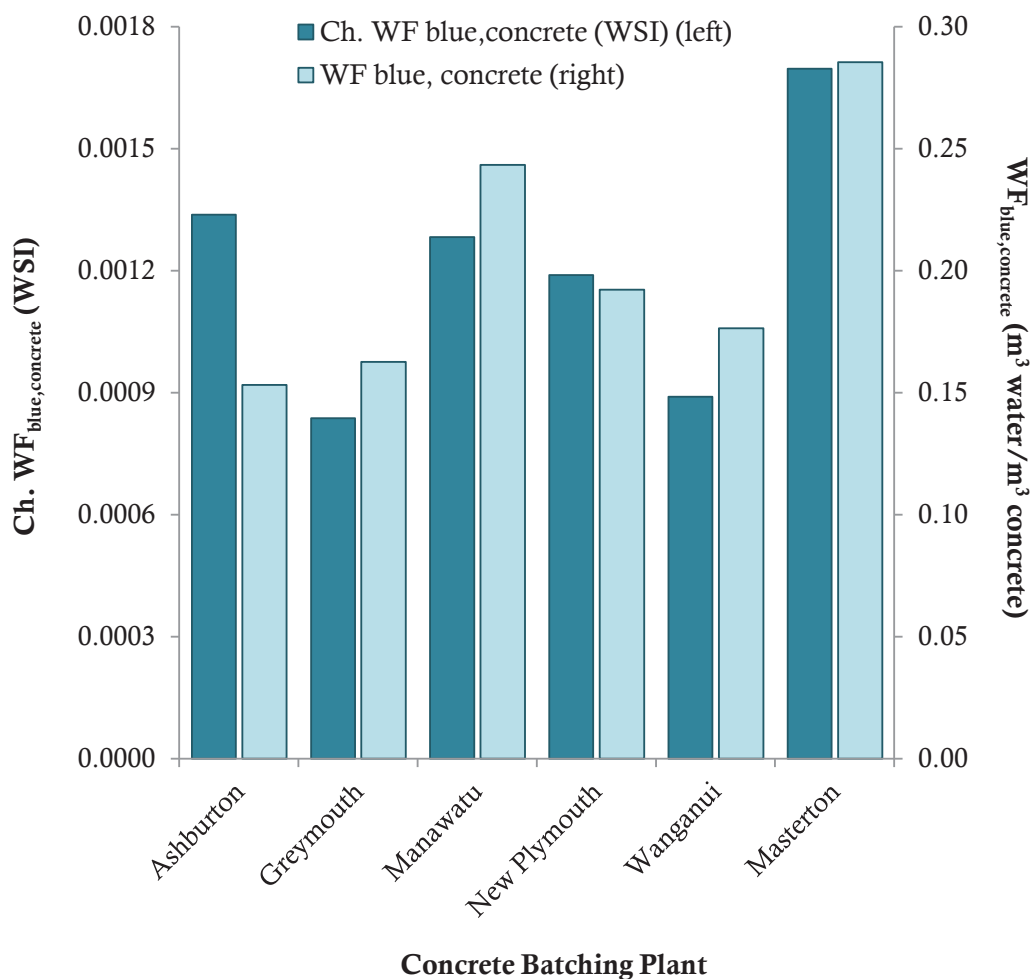


Figure 5.16 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}(WSI)$) at the selected freshwater management zones, for the 2013 period.

5.2.3.4 Water Depletion Index (Berger et al., 2014)

The Water Depletion Index (WDI) for the respective FMZ's can be seen in Figure 5.17. Overall, the WDI ranged from 0.0010 in the Ashburton FMZ to 0.005 in the Manawatu, Grey and Wanganui FMZ's.

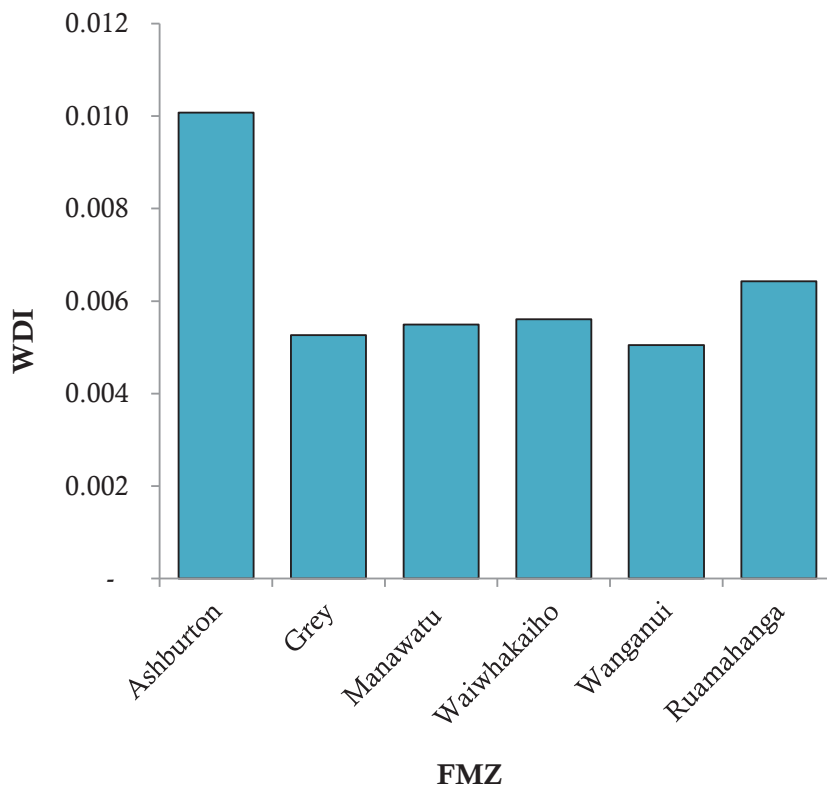


Figure 5.17 The water depletion index (*WDI*) calculated for selected freshwater management zones.

When the volumetric water footprints are characterised with the locally derived *WDI*, it can be seen that the Masterton batching plant has the highest characterised water footprint ($Ch.WF_{blue,concrete}(WDI)$) at 0.0018. This is followed closely by the Ashburton plant at 0.0015 (refer to Figure 5.18). Conversely, the Greymouth and Wanganui batching plants have the lowest water footprint when characterised with the *WDI* at 0.0009, respectively.

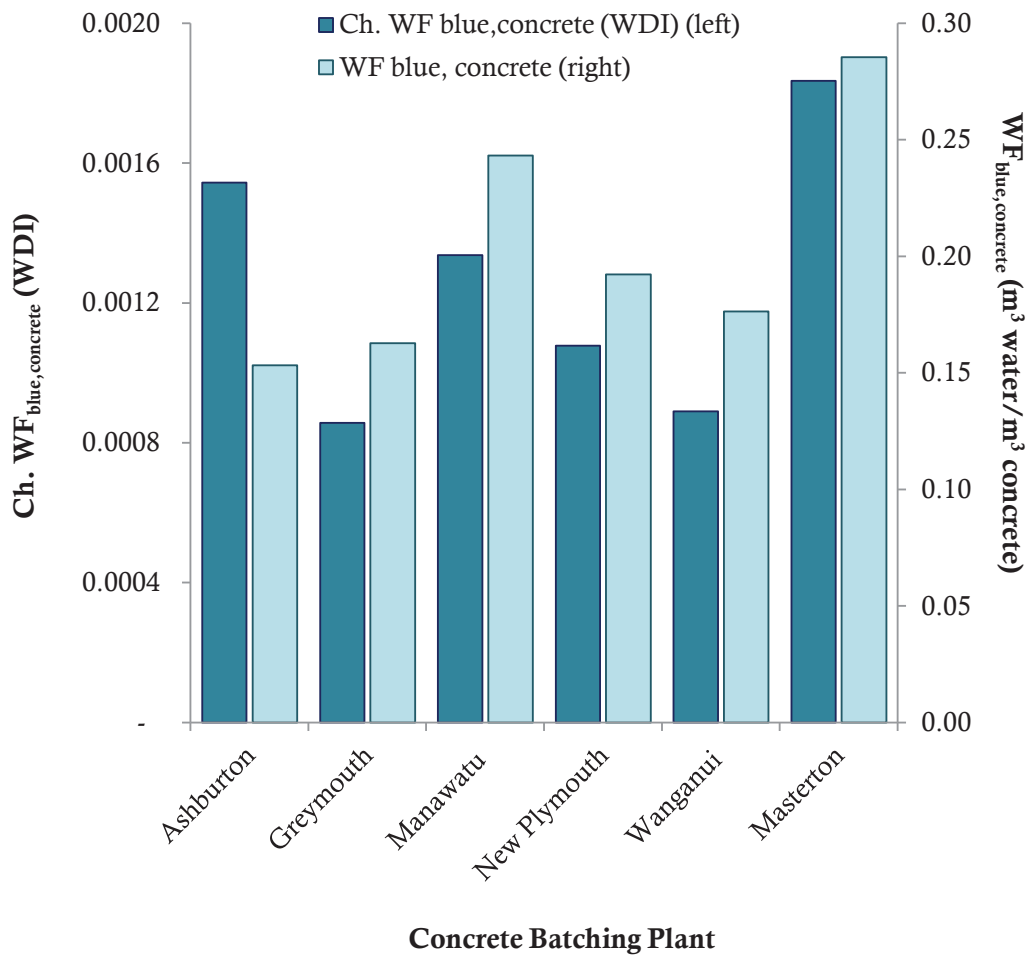


Figure 5.18 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch.WF_{blue,concrete}(WDI)$) at selected management zones, for the 2013 period.

5.2.3.5 Available Freshwater Remaining (Boulay et al., 2016)

The AMD characterisation factor for each selected management zone can be seen in Figure 5.19. The Manawatu FMZ has the highest AMD at 0.43, followed closely by the Ashburton FMZ at 0.42. Of the least concern in terms of environmental impacts are the Grey and Waiwhakaiho FMZ's, respectively at 0.11.

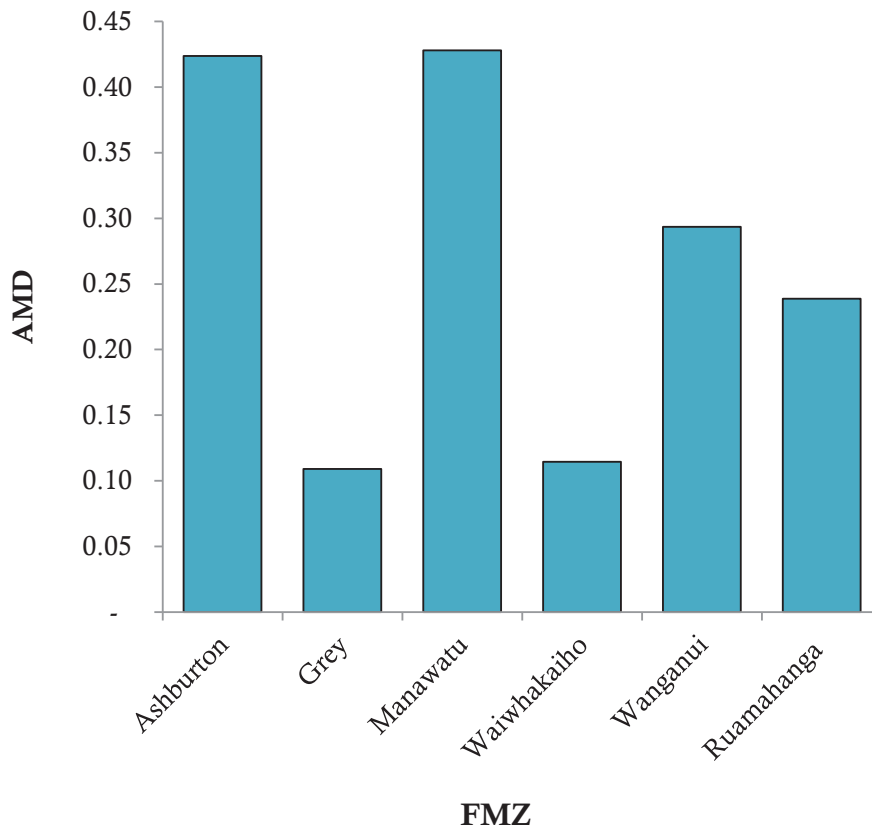


Figure 5.19 The availability minus demand (*AMD*) calculated for selected freshwater management zones.

The water footprint of the concrete batching plants ($WF_{blue,concrete}$) after characterisation by the local *AMD* values, ($Ch.WF_{blue,concrete}(AMD)$) can be seen in Figure 5.20. It shows that the characterised water footprint of the Palmerston North plant is the highest at 0.009. The lowest characterised water footprint was calculated at the Greymouth batching plant at 0.001.

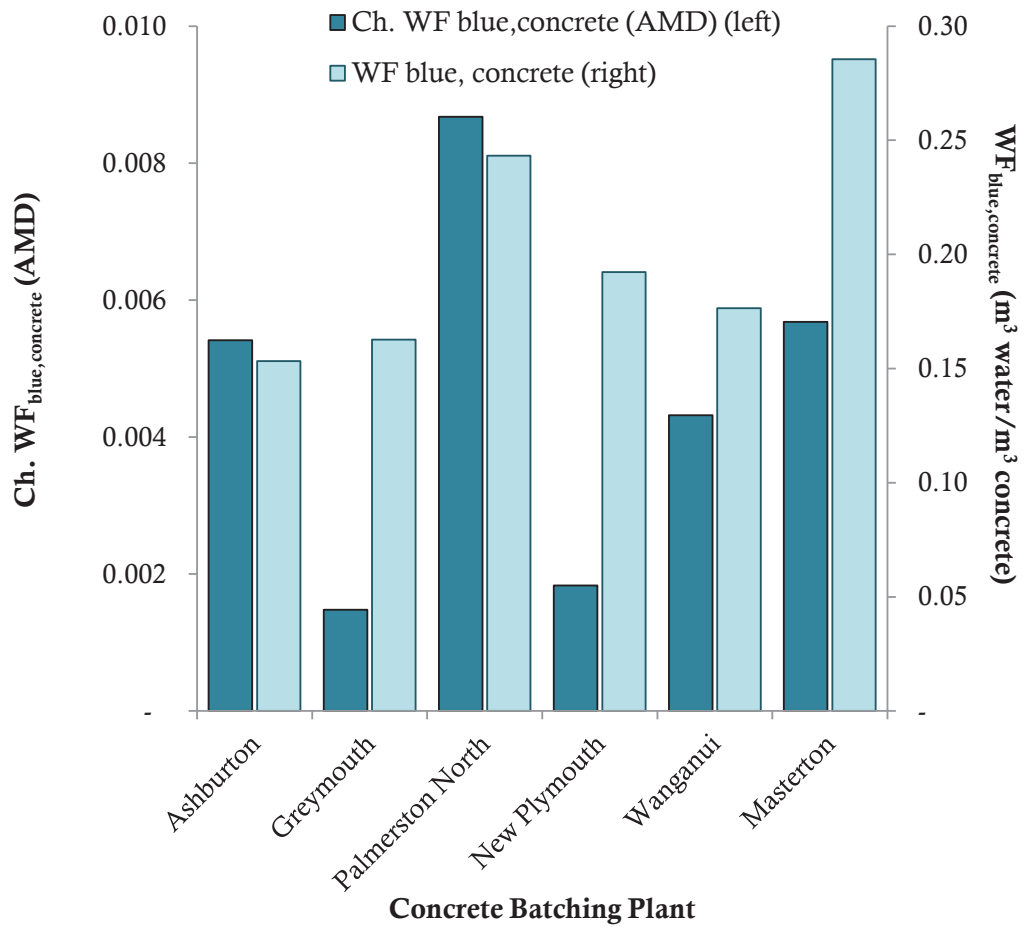


Figure 5.20 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}(AMD)$) at the selected freshwater management zones, for the 2013 period.

5.2.3.6 Comparison of Water Footprint Methods at the Freshwater Management Zone Scale

In order to compare the relationship between characterisation factors and the final characterised water footprints calculated using the methods of Hoekstra et al. (2011); Pfister et al. (2009); Berger et al. (2014) and Boulay et al. (2016), a ranking system was applied to both the characterisation factors themselves (Figure 5.21) and the volumetric and characterised water footprints of concrete (Figure 5.22). Both the characterisation factors and the characterised water footprints of concrete derived from

each method (and for each selected batching plant), were ranked in ascending order from lowest (1) to largest (6).

At the freshwater management zone (FMZ) scale, Figure 5.21 indicates that the *WDI* and *WS_{blue}* characterisation factors ranked each FMZ identically. For three of the four methods, the Ashburton FMZ was ranked highest with the exception of the *AMD* method. The *AMD* method ranked the Ashburton FMZ 5th, behind the Manawatu FMZ, ranked 6th. The *WSI* method, however, ranked Grey, Manawatu and Wanganui FMZs equal to 1st, (lowest water stress), Waiwhakaiho and Ruamahanga FMZs equal to 4th, and Ashburton FMZ equal to 6th (highest water stress).

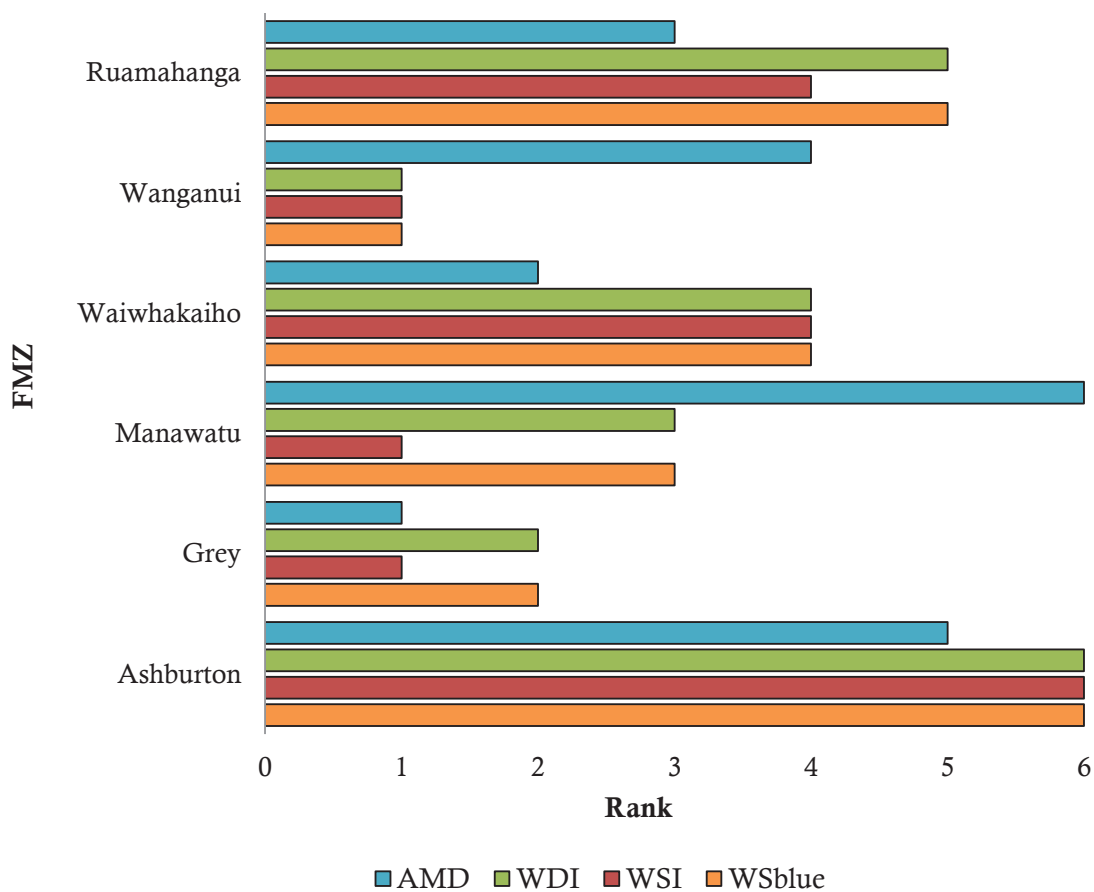


Figure 5.21 Ranking of the selected characterisation factors (in ascending order) at the freshwater management zone scale.

Figure 5.22 shows some variation in the post-characterised water footprint of each concrete batching plant depending on which method is employed. The results for Greymouth, Palmerston North and New Plymouth batching plants appear to have the greatest agreement between different water footprint methods. Interestingly, post-characterisation, the *WSI* and *WDI* methods consistently give the same ranking for each batching plant. In terms of the *WS_{blue}* and the *AMD*, results are only ever 'one out' with the exception of the Palmerston North batching plant which is 'two out' indicating a higher environmental impact.

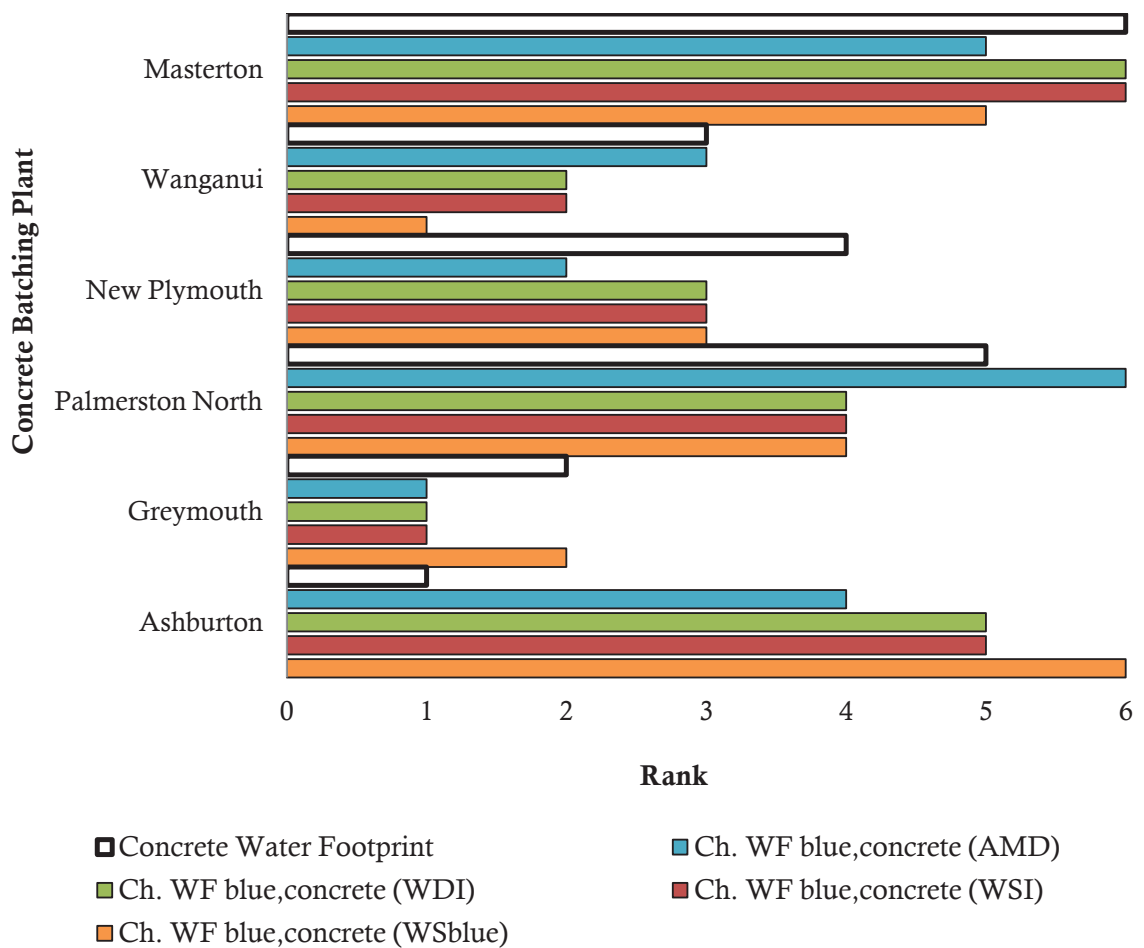


Figure 5.22 Ranking of the selected characterised water footprint results (in ascending order) at the freshwater management zone scale, for the 2013 period.

5.2.4 Catchment Scale

Figure 5.12 shows the proportion of freshwater available, withdrawn and consumed in each selected catchment scale. It can be seen in Figure 5.23 that the volume of available freshwater for the Greymouth catchment is higher than all other selected catchments. In contrast, both the freshwater withdrawals and consumption in the Greymouth catchment could be considered negligible in comparison (at 0.7% of the total availability). The Ashburton catchment has the highest proportion of freshwater withdrawals and consumptions (10.5% of the total availability). The Ruamahanga catchment has the second highest percentage of withdrawal/consumption in comparison to other selected catchments, at 4% of the freshwater available.

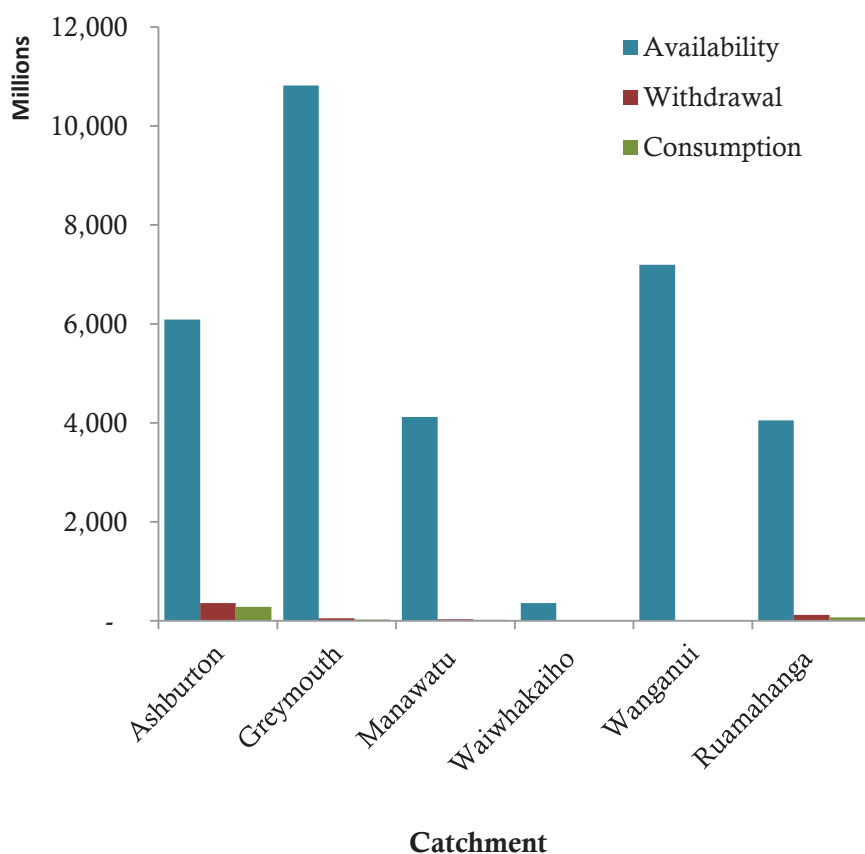


Figure 5.23 Volumes of available, withdrawn and consumed freshwater in the selected catchments (millions m³/ year).

5.2.4.1 Blue Water Scarcity Impact Index (Hoekstra et al., 2011)

The WS_{blue} for selected catchments can be seen in Figure 5.24. It can be seen that the WS_{blue} characterisation factor is much greater for the Ashburton catchment than all other selected catchments, at 0.23. The Ashburton catchment's WS_{blue} for example, is almost three times greater than that of the second highest catchment, Ruamahanga (at 0.08). The lowest WS_{blue} is calculated for the Wanganui catchment (at 0.001), followed closely by the Greymouth catchment (at 0.01).

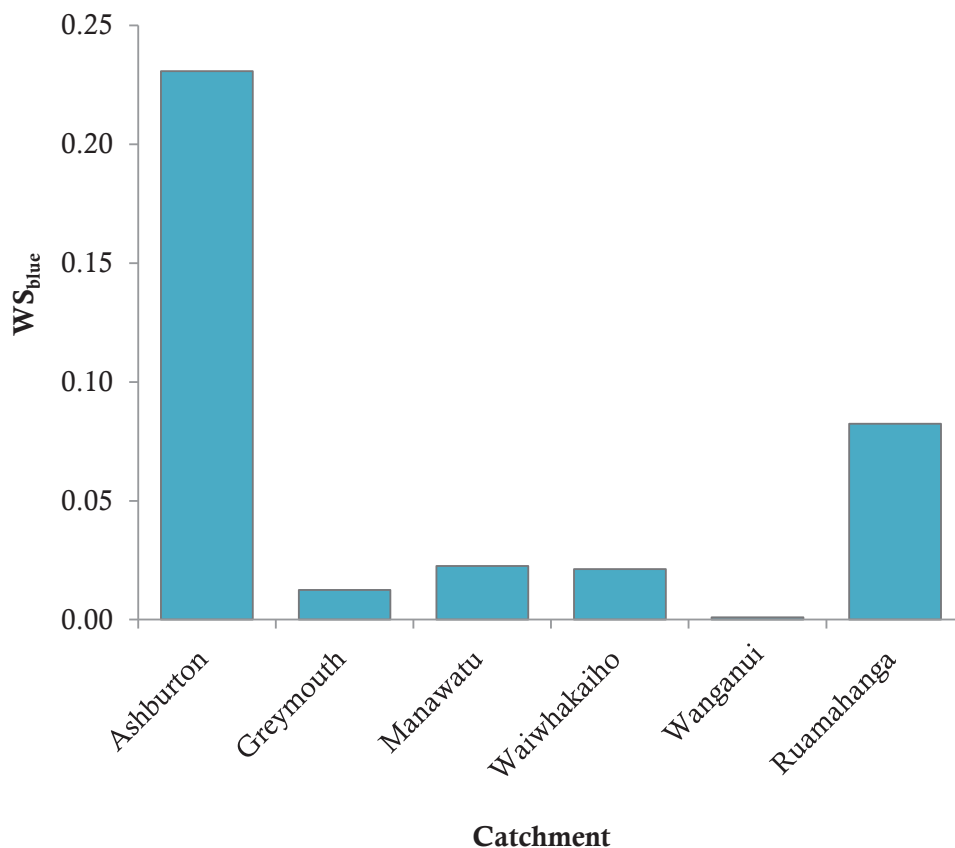


Figure 5.24 The calculated blue water scarcity (WS_{blue}) for selected catchments.

In accordance with the method, Figure 5.25 shows the water footprint of concrete post-characterisation with the WS_{blue} . It can be seen that when the volumetric water footprint is characterised by the local water scarcity at the catchment scale, that the

Ashburton batching plant has the highest water footprint at 0.035. The batching plant with the lowest characterised water footprint is the Wanganui batching plant at 0.001.

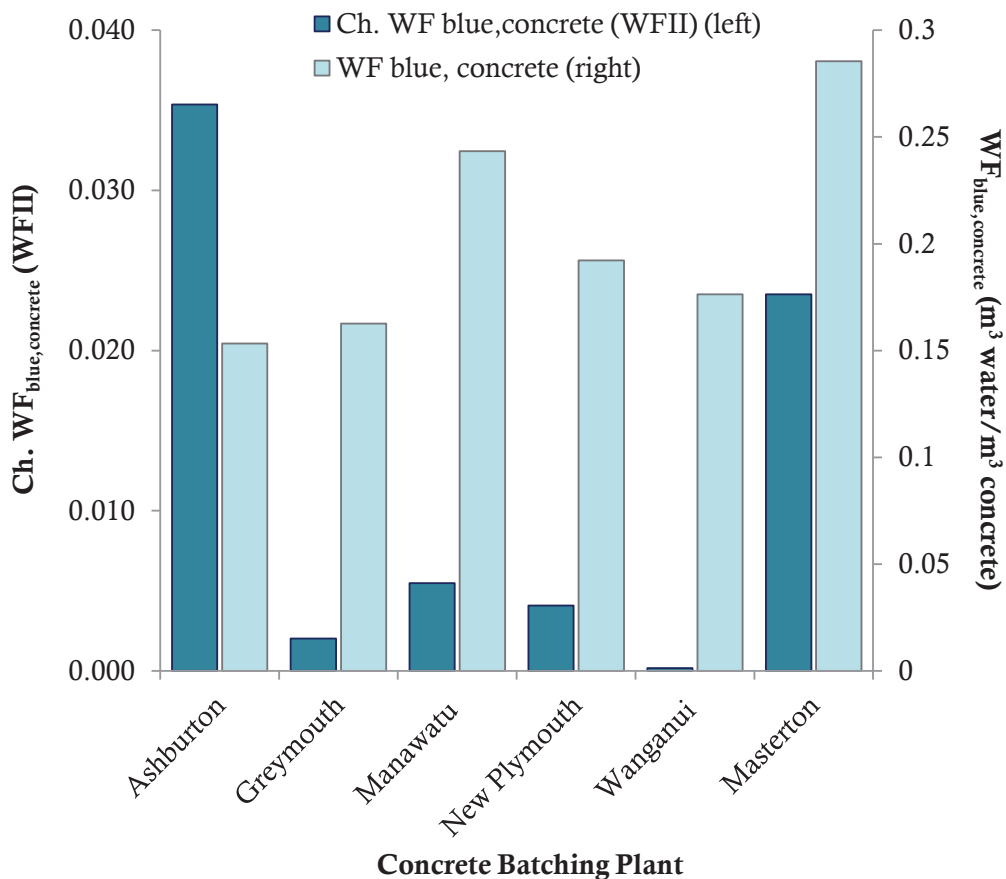


Figure 5.25 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}$ (WFII)) at the catchment scale, for the 2013 period.

5.2.4.2 Water Stress Index

Catchment level *WSI* characterisation factors for six selected batching plants can be seen in Figure 5.26. The calculated *WSI* ranges from 0.010 in the Greymouth and Wanganui, catchments, through to 0.016 in the Ashburton catchment.

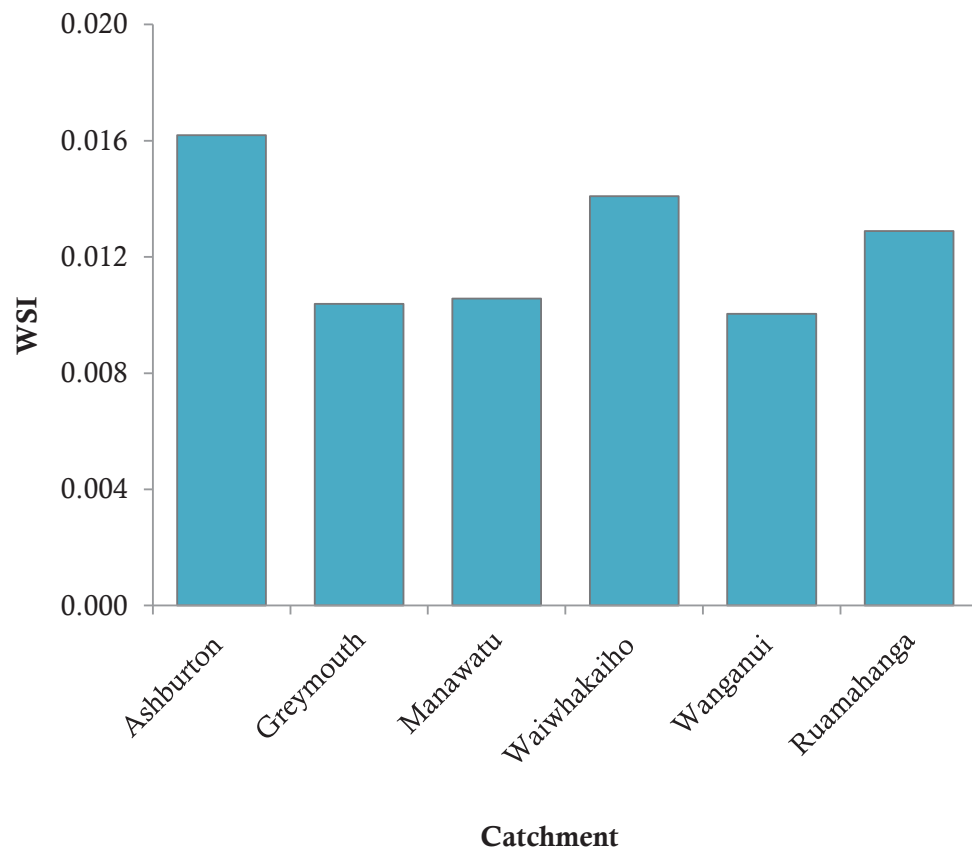


Figure 5.26 The calculated water stress index (*WSI*) for selected catchments.

As can be seen in Figure 5.27, when the volumetric water footprint of each concrete plant is characterised by the local *WSI*, the Masterton batching plant can be seen to have the largest environmental impact (0.0037) in terms of freshwater scarcity. In contrast, the Wanganui and Greymouth batching plants can be seen to have the lowest environmental impact when characterised by local water stress indices (0.0018 and 0.0017, respectively).

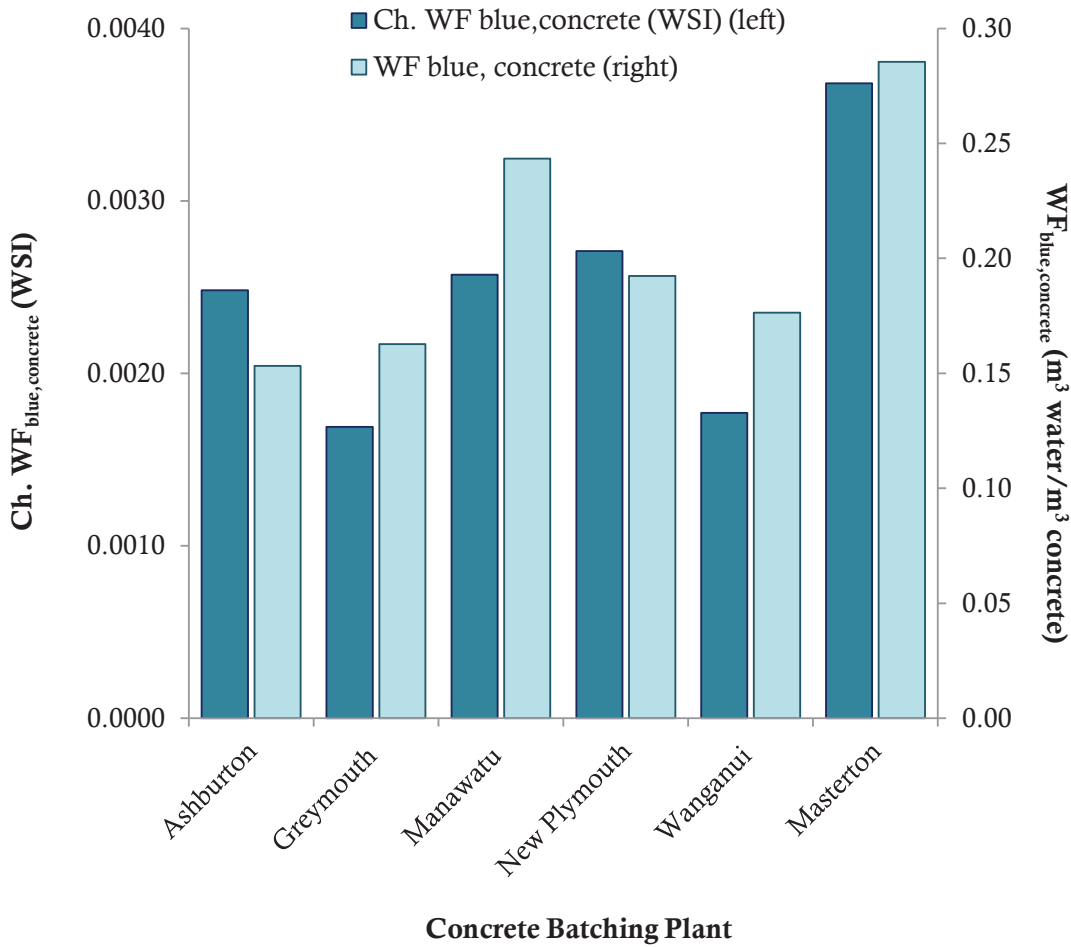


Figure 5.27 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprint ($Ch. WF_{blue,concrete}$ (WSI)) at the catchment scale, for the 2013 period.

5.2.4.3 Freshwater Depletion Index (Berger et al., 2014)

As can be seen in Figure 5.28, when the *WDI* characterisation factor is calculated at the catchment scale it ranges from 0.01 in the Greymouth, Manawatu, Waiwhakaiho and Wanganui catchments, through to 0.06 in the Ashburton catchment. The Ashburton catchment has a *WDI* three times greater than the second highest calculated *WDI* in Ruamahanga (0.02).

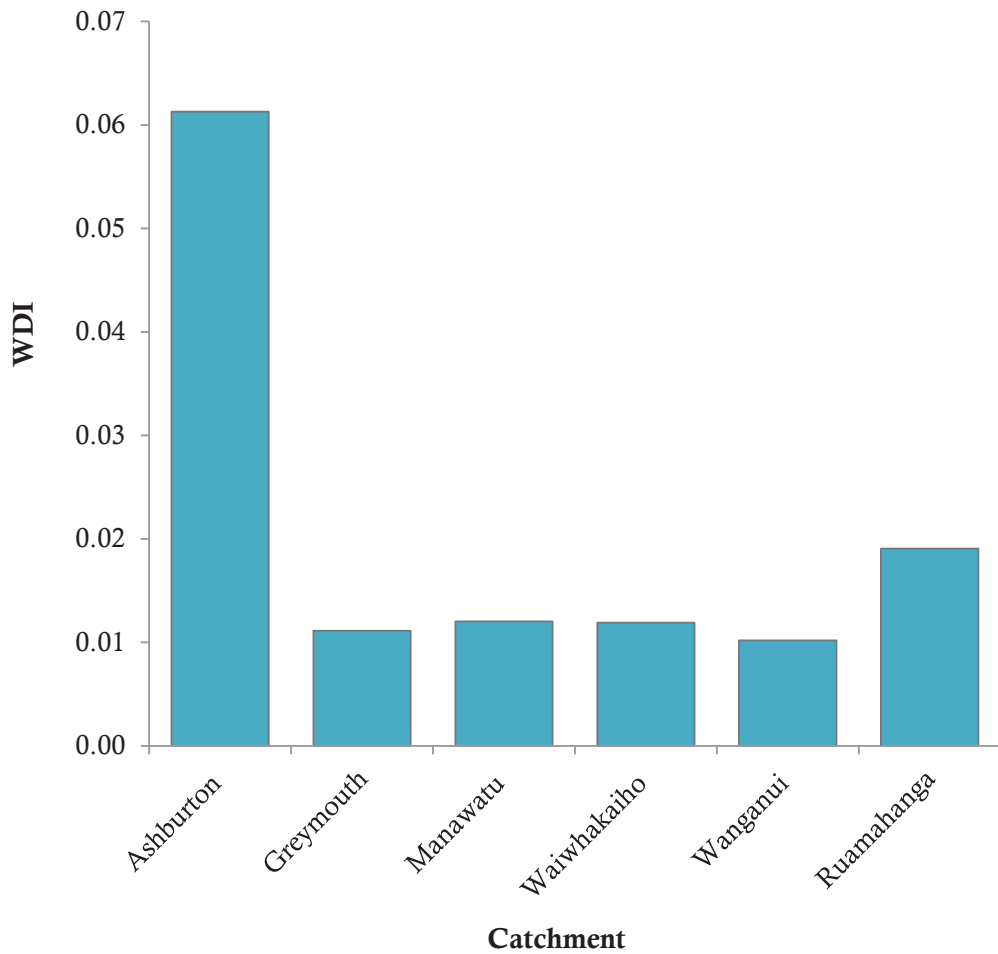


Figure 5.28 The calculated water scarcity index (*WDI*) for selected freshwater management zones.

The volumetric water footprint of 1 m³ concrete has been characterised at the catchment scale by the *WDI* (Figure 5.29). According to Berger et al. (2014) the risk of freshwater depletion is by far the highest for the Ashburton batching plant at 0.009. All other batching plants having comparatively low water footprints post characterisation ($Ch.WF_{blue,concrete}(WDI)$). The lowest water footprints were calculated for the Wanganui and Greymouth batching plants at 0.002.

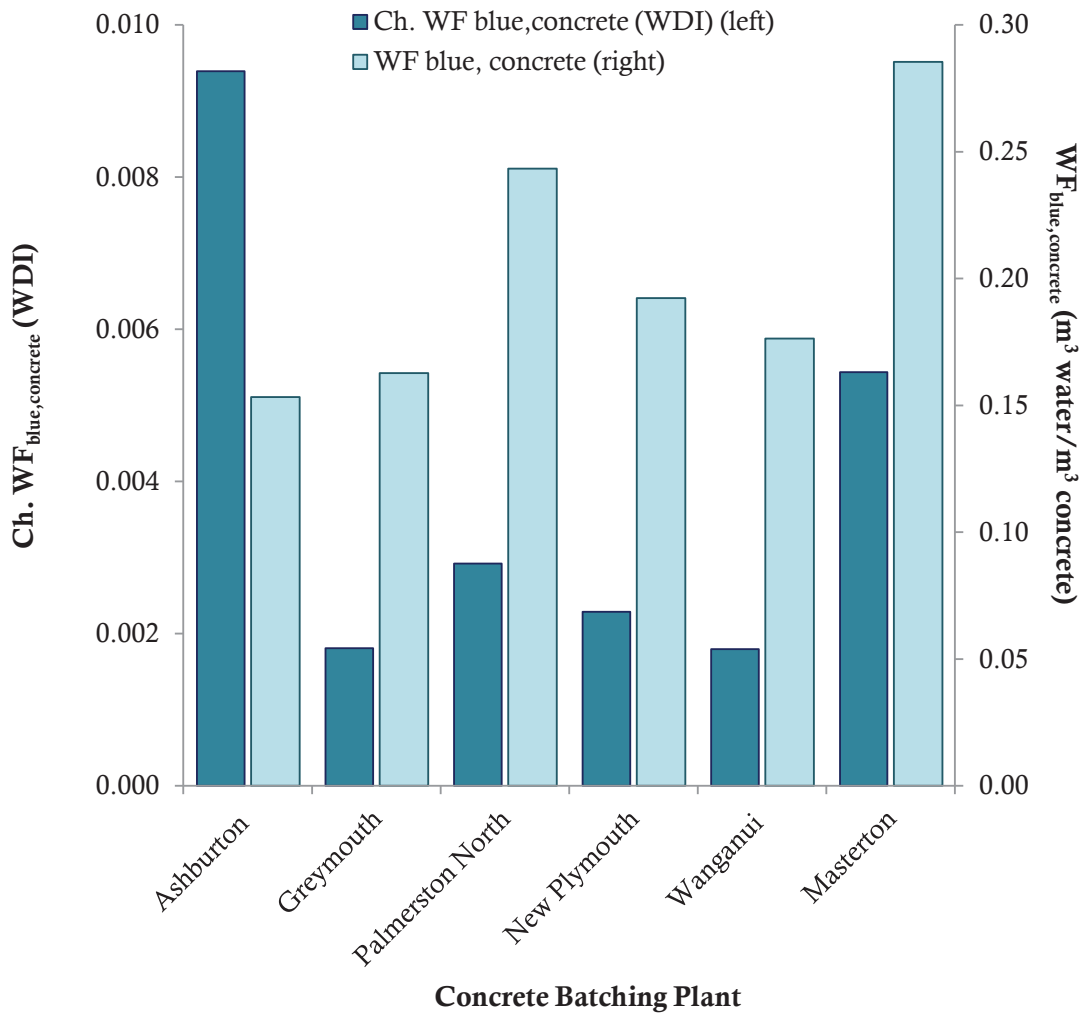


Figure 5.29 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch.WF_{blue,concrete}(WDI)$) at the catchment scale, for the 2013 period.

5.2.4.4 Available Water Remaining (Boulay et al., 2016)

The AMD characterisation factor has been calculated at the catchment scale (Figure 5.30). The AMD ranged from 0.009 in the Greymouth catchment, to 0.037 in the Manawatu catchment. The Wanganui and Ashburton catchments follow closely, at 0.025 and 0.027, respectively.

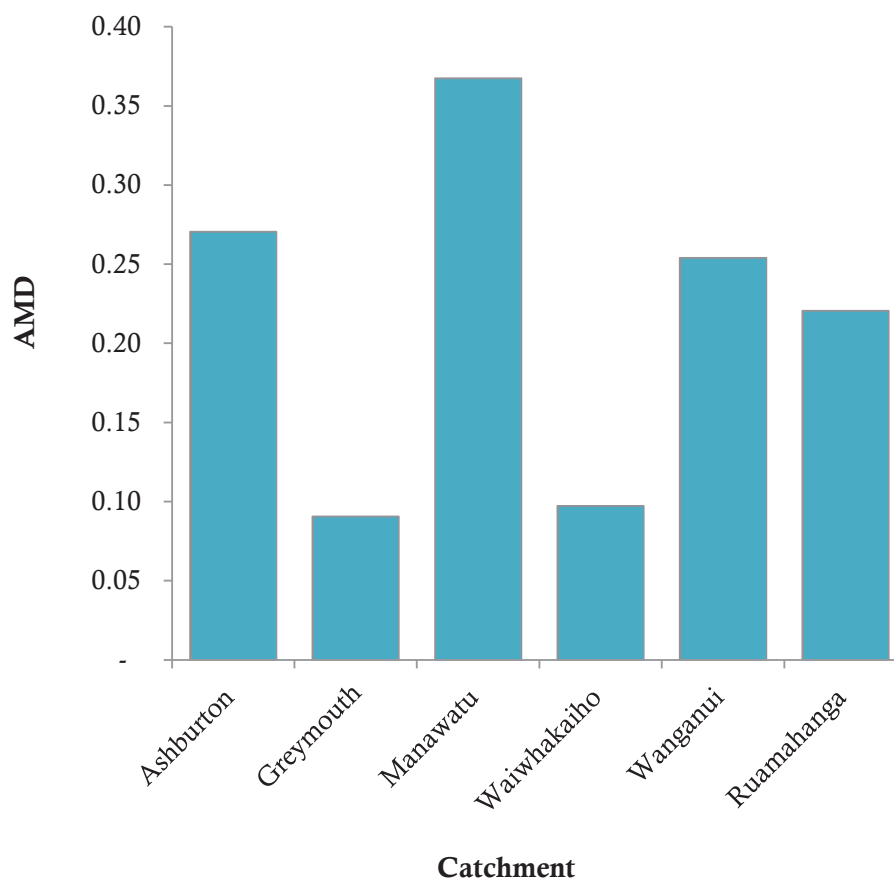


Figure 5.30 The calculated water scarcity index (*AMD*) for selected freshwater management zones.

As can be seen in Figure 5.31, the volumetric water footprint characterised by the local *AMD* can be seen to be the greatest for the Palmerston North and Masterton batching plants (at 0.009 and 0.006, respectively). The lowest recorded water footprint was calculated for the Greymouth batching plant (at 0.01), followed closely by the New Plymouth plant (at 0.02).

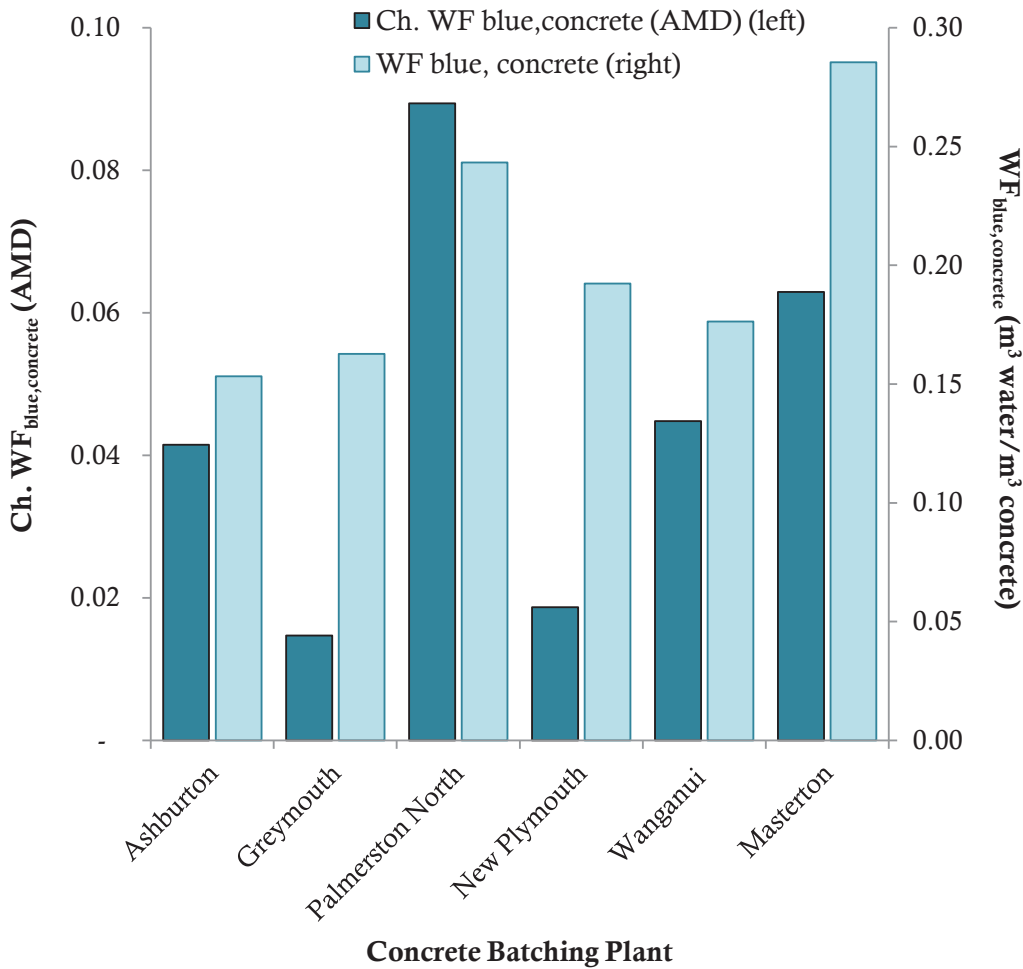


Figure 5.31 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch.WF_{blue,concrete}(AMD)$) at the catchment scale.

5.2.4.5 Comparison of Water Footprint Methods at the Catchment Scale

In order to compare the relationship between characterisation factors and the final characterised water footprints calculated using the methods of Hoekstra et al. (2011); Pfister et al. (2009); Berger et al. (2014) and Boulay et al. (2016), a ranking system was applied to both the characterisation factors themselves (Figure 5.32) and the volumetric water footprint of concrete post-characterisation (Figure 5.33). Both the characterisation factors and the post-characterised water footprints are ranked in ascending order from 1 (lowest) to 6 (largest).

Figure 5.32 shows some variation in the characterisation factors of the selected catchments. The Wanganui catchment can be seen to rank lowest in three of the four methods (WDI , WSI and WS_{blue}), followed by the Greymouth catchment which ranked second using the same three methods. Similarly, the Ashburton catchment ranks greatest in the three of the four methods (WDI , WSI and WS_{blue}),. In each instance, it is the AMD characterisation factor that is causing the variation in the ranks of these catchments. The WDI , WSI and WS_{blue} characterisation factors can be seen to rank the Ashburton, Greymouth and Wanganui catchments in the same order, and for all other catchments the rank is only ever 'one out' amongst methods. In contrast, the AMD does not follow the same ranking pattern and can be seen to be greatest for the Manawatu catchment and lowest for the Greymouth catchment.

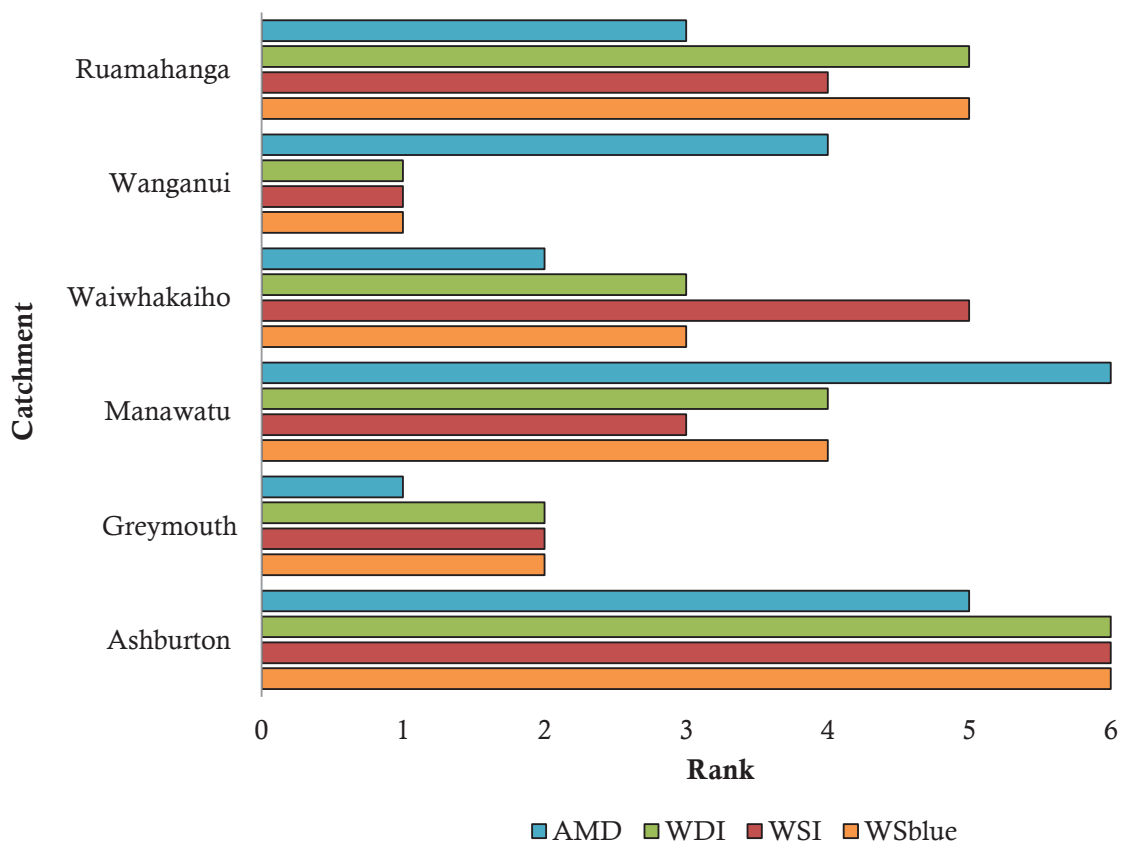


Figure 5.32 Ranking of the selected characterisation factors (in ascending order) at the catchment scale.

Of all methods the Ashburton batching plant is ranked the greatest in terms of environmental impacts by two of the four methods evaluated (*WDI* and *WFII*⁴⁴). The Masterton batching plant is ranked fifth highest by three of the four methods (*AMD*, *WDI* and *WFII*). Similarly, the Palmerston North plant is ranked fourth highest by three of the four methods (*WDI*, *WSI* and *WFII*). Depending on which method is employed the Ashburton, Masterton and Palmerston North batching plants all rank highest for environmental impacts at least once. In contrast, the batching plant ranked the lowest in terms of environmental impacts associated with freshwater is the Greymouth plant, which ranked either first or second depending on the chosen method. Interestingly, at the catchment scale the *WDI* and *WFII* methods characterise all six of the selected batching plants in the same order.

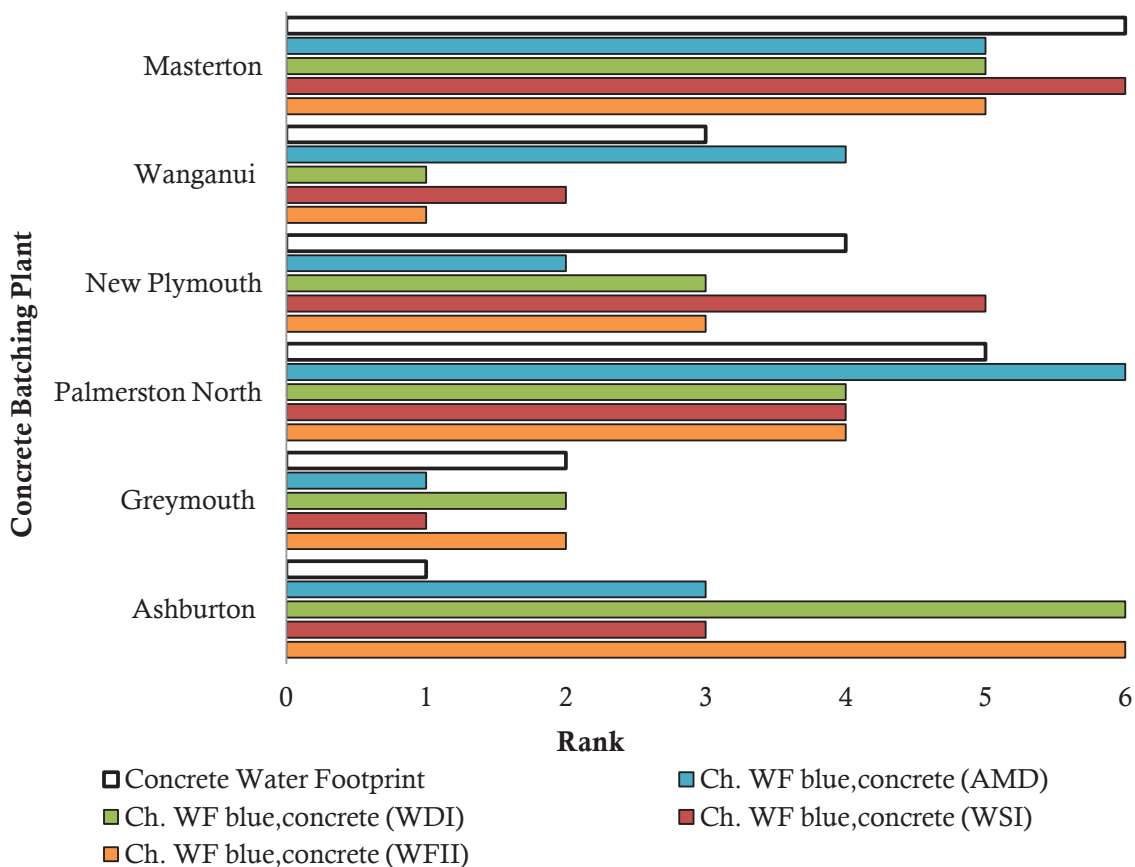


Figure 5.33 Ranking of the characterised water footprint results at the catchment scale (in ascending order), for the 2013 period.

⁴⁴ Calculated using the WS_{blue} characterisation factor

5.2.5 Regional Scale

At the regional scale, it can be seen that the West Coast region of New Zealand has, by far, the most available freshwater of the six selected regions. The West Coast, for example, has over three times the volume of the second highest region, Canterbury. The region with the lowest available freshwater can be seen to be the Taranaki region of New Zealand.

In terms of withdrawal and consumption of available freshwater it can be seen that the Canterbury region withdraws and consumes the highest volume compared to the other five selected regions. Canterbury's withdrawal and consumption, is 9.1% of the available freshwater in that region, compared with 1.72% in the Wellington region. The region with the lowest withdrawal and consumption is the West Coast region at 0.08% of the available freshwater.

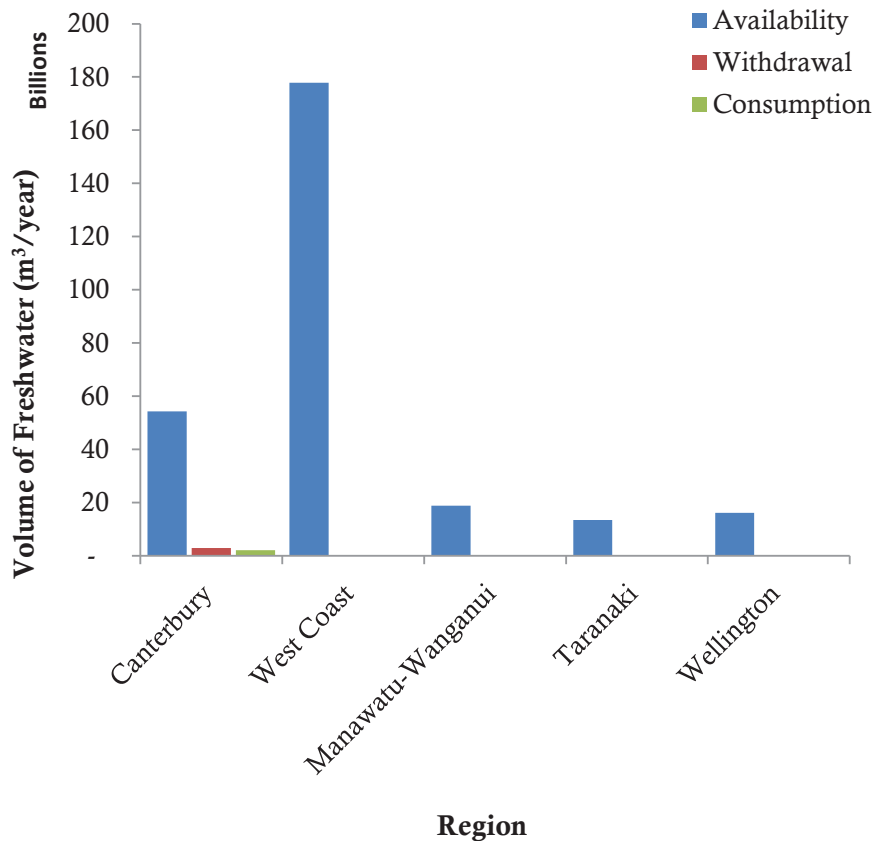


Figure 5.34 Volumes of available, withdrawn and consumed freshwater in the selected regions (billion m³/ year).

5.2.5.1 Blue Water Footprint Impact Index (Hoekstra et al. 2011)

Regional scale WS_{blue} characterisation factors for the six selected batching plants can be seen in Figure 5.35. It can be seen that WS_{blue} varies substantially depending on regional characteristics. The Canterbury region displaying the highest WS_{blue} (at 0.19) Contrastingly, the West Coast has the lowest calculated WS_{blue} at 0.001.

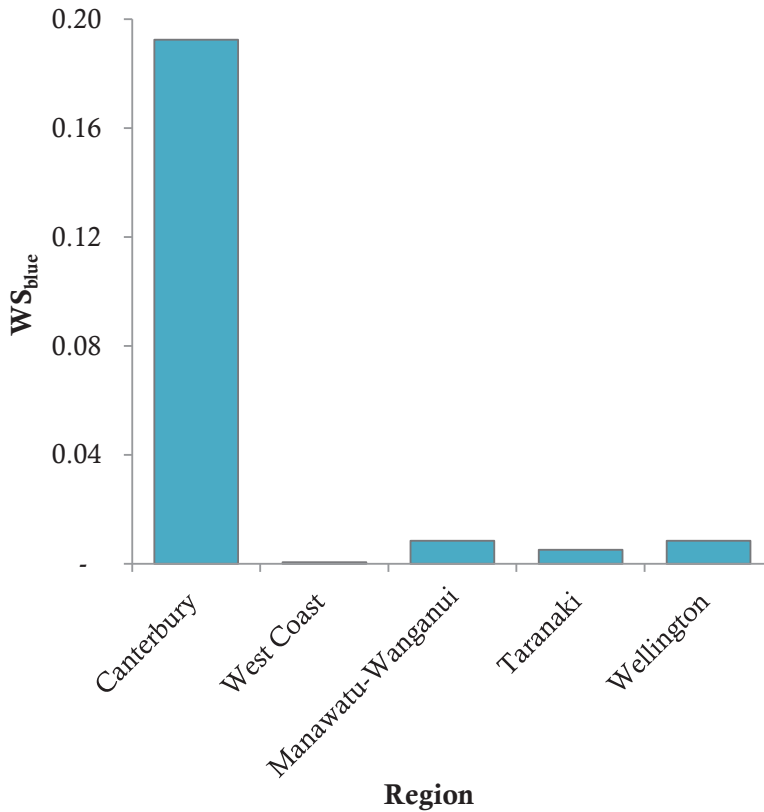


Figure 5.35 The calculated water scarcity index (WS_{blue}) for selected regions.

According to the characterised water footprint calculated at the regional scale (Figure 5.36), the Ashburton concrete batching plant (Canterbury region) has the highest water footprint at 0.029. Of the least concern is the Greymouth batching plant (West Coast region) at 0.001. As far as the volumetric water footprints of the selected concrete batching plants, the Masterton plant (Wellington region) consumes the most water, however, it can be seen that the environmental impact of this (when characterised by the WS_{blue}) are proportionally low. In contrast, Ashburton batching plant has a lower

volumetric water footprint, but when characterised by the local water scarcity it can be seen to have the greatest impact of all batching plants.

When compared with the $WFII$ characterisation factors in Figure 5.35, it can be seen that the order in which plants are ranked post-characterisation with the WS_{blue} (refer Figure 5.36) does not alter. However, when compared with the volumetric water footprints of the selected concrete batching plants, the order can be seen to notably alter in comparison.

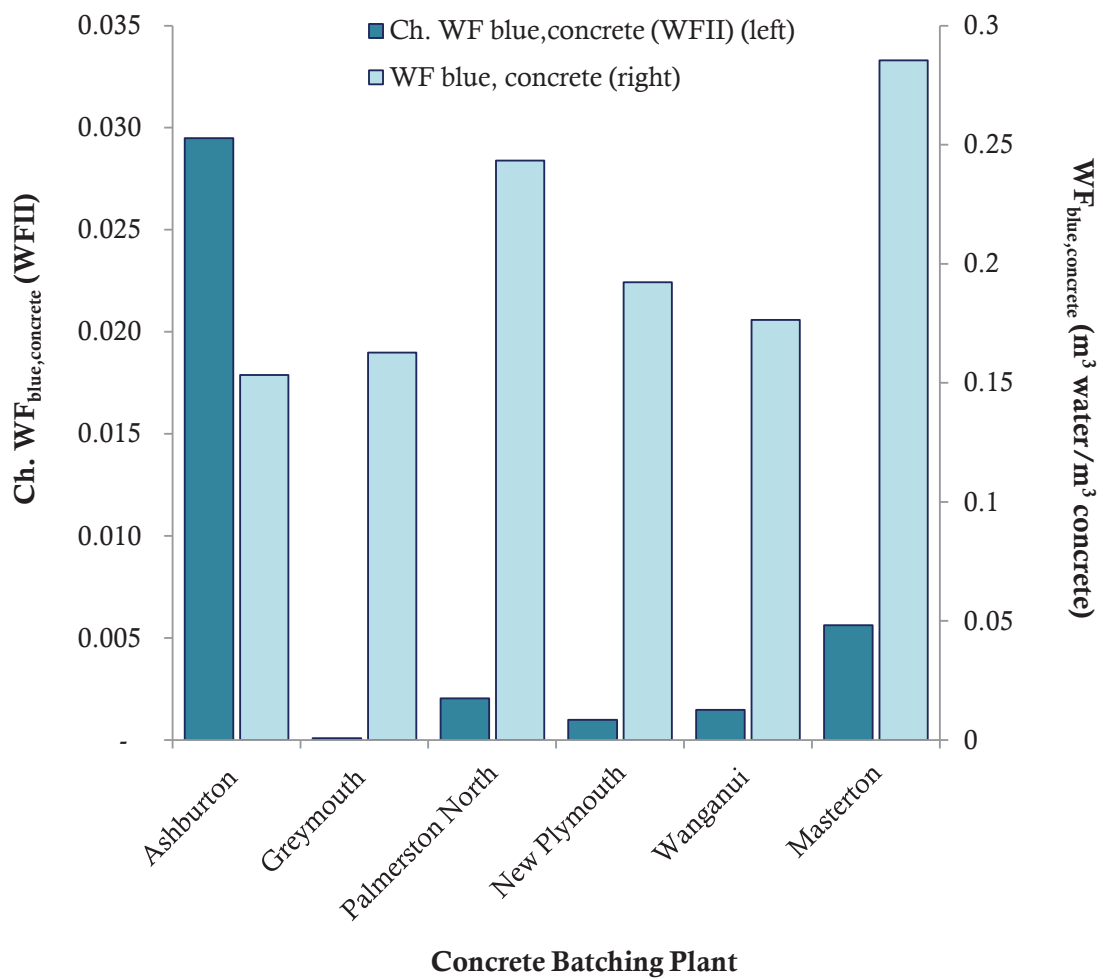


Figure 5.36 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}$ (WFII)) at the regional scale, for the 2013 period.

5.2.5.2 Water Stress Index (Pfister et al., 2009)

The *WSI* calculated at the regional scale can be seen in Figure 5.37. The *WSI* ranges from 0.010 in the Taranaki, West Coast and Manawatu-Wanganui regions to 0.015 in the Canterbury region. The *WSI* can be seen to have less variation in the calculated characterisation factor across the selected batching plants than other methods employed (e.g. the *WFII* in Figure 5.35).

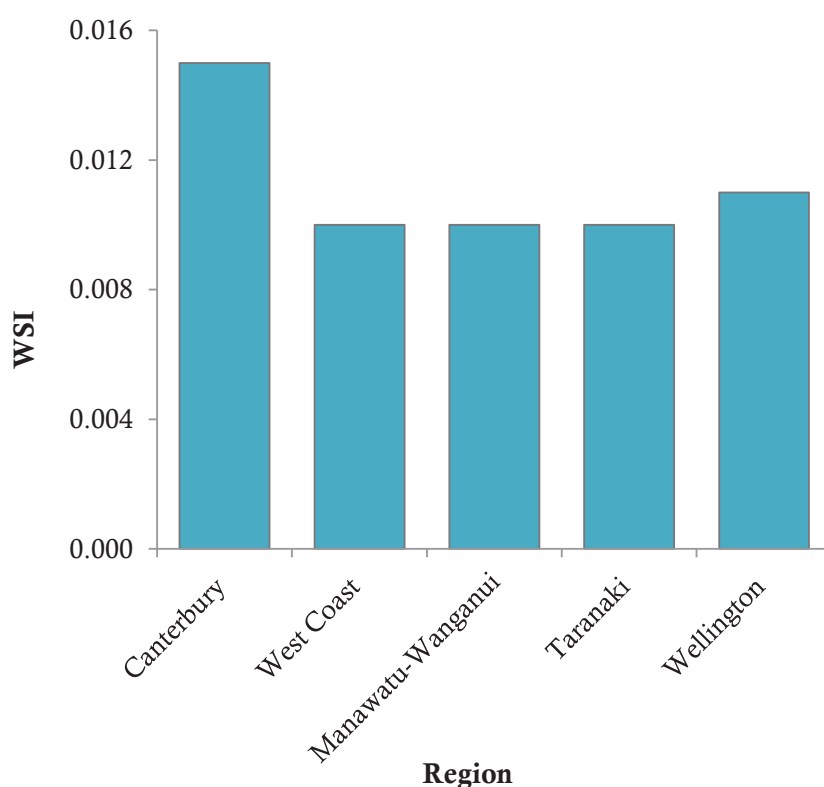


Figure 5.37 The calculated water scarcity index (*WSI*) for selected regions.

The volumetric water footprints of the six concrete batching plants are characterised by the regional *WSI* in Figure 5.38. It can be seen that the environmental impact of the Masterton plant is the greatest at 0.0032. This is followed by the Palmerston North and Ashburton plants, at 0.0025 and 0.0024, respectively. The batching plant with the lowest characterised water footprint is Greymouth, at 0.0016. Figure 5.38 shows a

close relationship between all batching plants characterised water footprints (in comparison with Figure 5.36 for example).

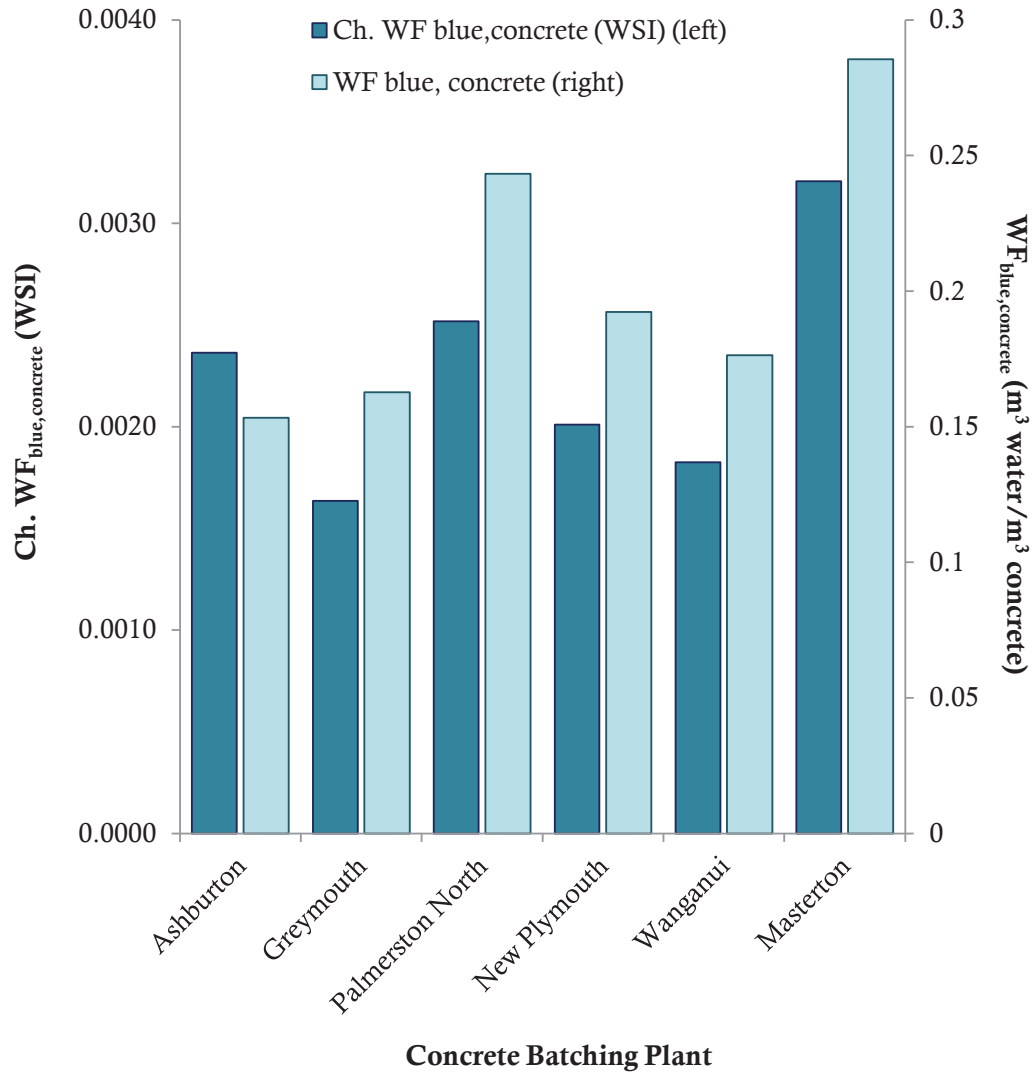


Figure 5.38 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}(WSI)$) at the regional scale, for the 2013 period.

5.2.5.3 Water Depletion Index (Berger et al., 2014)

The *WDI* indicates the Canterbury region to have the greatest risk of environmental impact at 0.043. Of least concern are the West Coast and Taranaki regions of New Zealand at 0.10, respectively. There can be seen to be a lot of variation in the

calculated *WDI* amongst regions of New Zealand. The Canterbury region, for example, is over three times as high as the second highest region - Wellington.

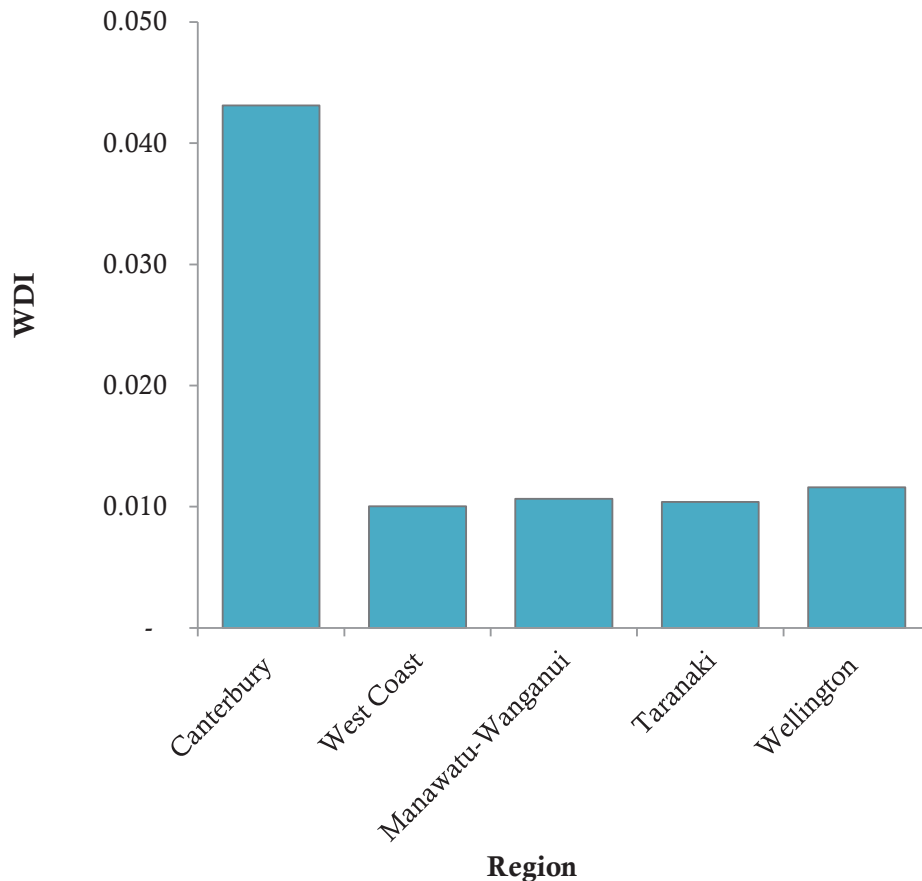


Figure 5.39 The calculated water depletion index (*WDI*) for selected regions.

The *WDI* clearly shows that the Ashburton batching plant has the greatest environmental impact of all concrete batching plants assessed at 0.006. Interestingly, the Ashburton batching plant does not have the highest volumetric water footprint, with the Masterton plant exceeding the amount of freshwater required to produce 1 m³ of concrete (0.15 and 0.29 m³ water/ m³ concrete, respectively). In terms of the characterised water footprint however, the Masterton batching plant has only the third highest water footprint post-characterisation (at 0.003). The Greymouth, Palmerston North and Wanganui batching plants, can be seen to have the lowest characterised water footprint at 0.002, respectively.

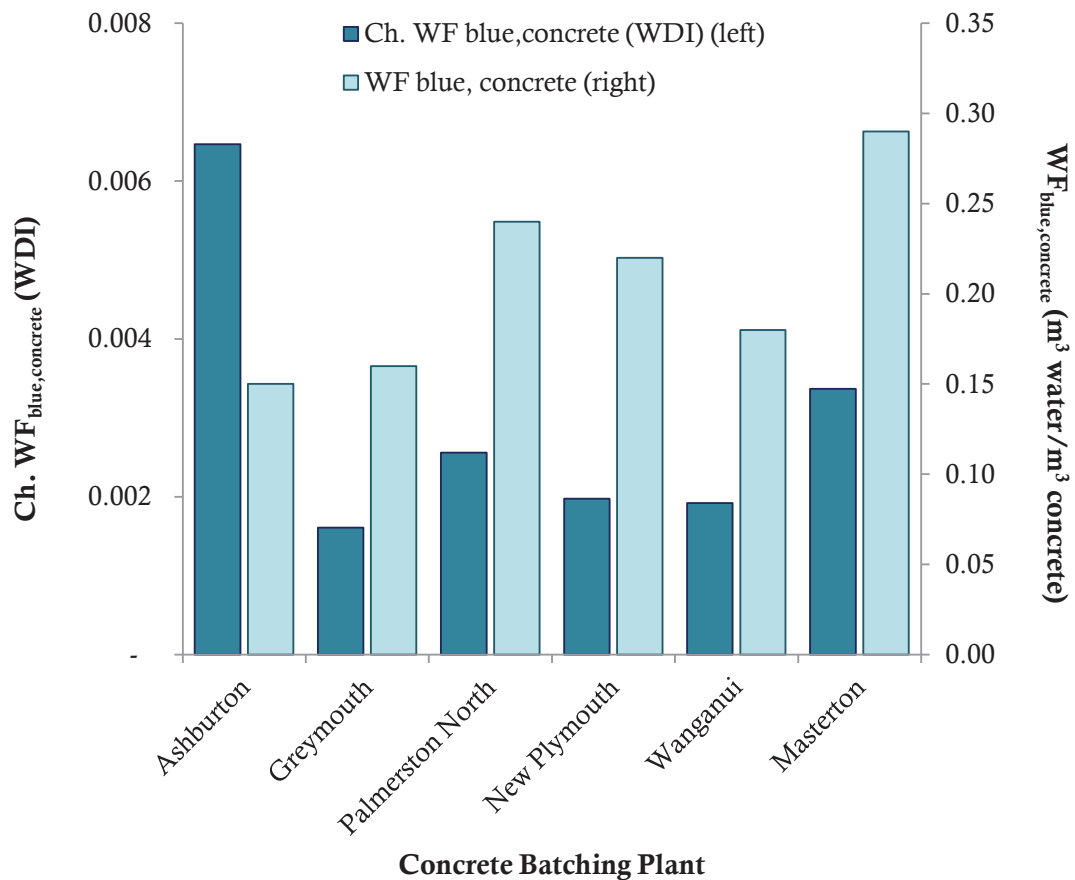


Figure 5.40 The volumetric water footprint of selected concrete batching plants ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}(WDI)$) at the regional scale, for the 2013 period.

5.2.5.4 Available Water Remaining (Boulay et al., 2016)

According to Boulay et al. (2016), the *AMD* in selected New Zealand regions can be seen in Figure 5.41. The Manawatu-Wanganui region has the highest calculated *AMD* at 0.35, this is followed by the Wellington and Canterbury regions at 0.26 and 0.27, respectively. The region with the lowest calculated *AMD* is the West Coast region, at 0.05.

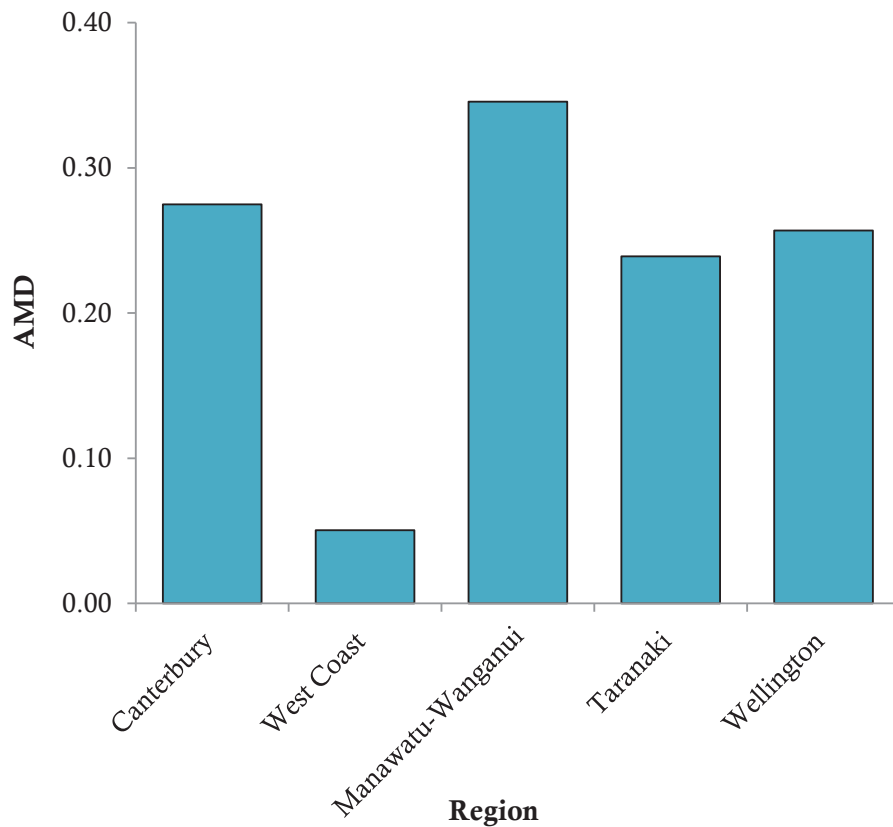


Figure 5.41 The calculated availability minus demand (*AMD*) for selected regions.

In terms of the environmental impact of the six concrete batching plants, as calculated using the AWaRE method (Figure 5.42), the Palmerston North plant appears to have the highest impact at 0.084. This is followed closely by the Masterton plant (at 0.073). In contrast, Greymouth is the identified as having the lowest water footprint post-characterisation at 0.008.

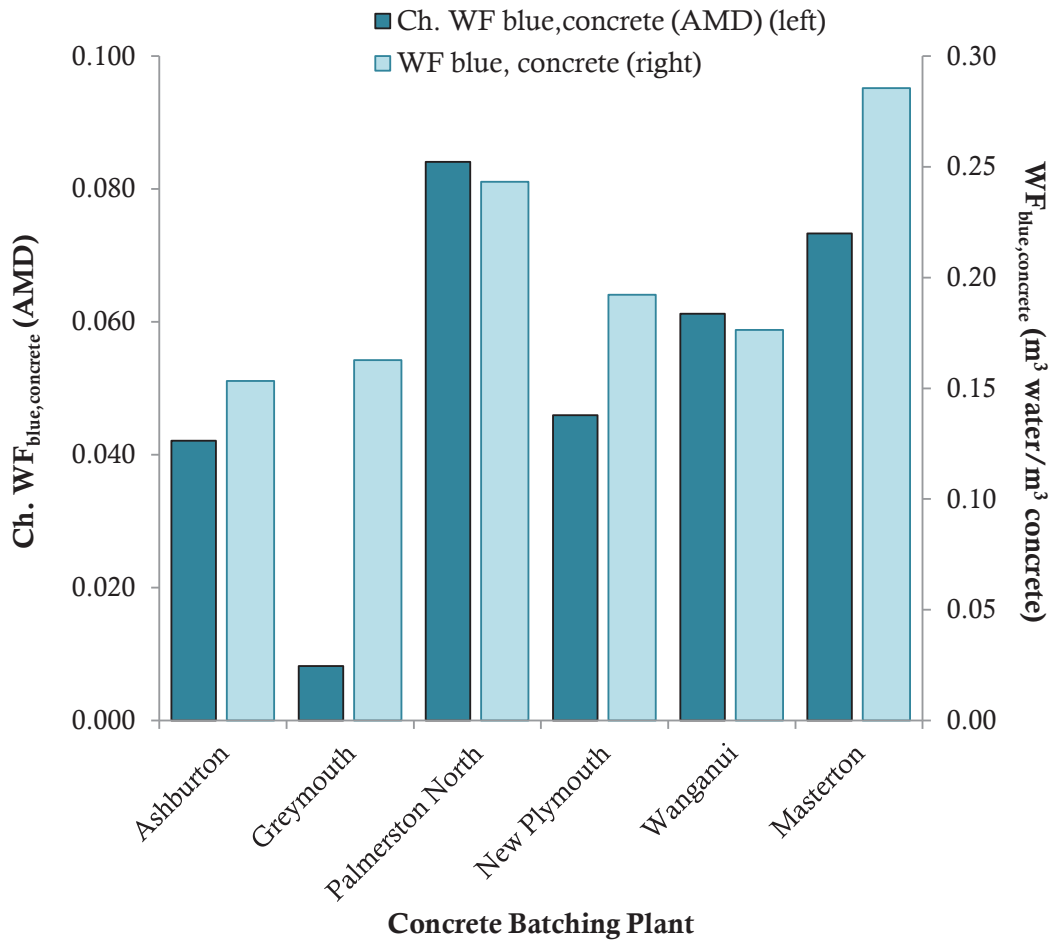


Figure 5.42 The volumetric water footprint of concrete ($WF_{blue,concrete}$) alongside the characterised water footprints ($Ch. WF_{blue,concrete}(AMD)$) at the regional scale, for the 2013 period.

5.2.5.5 Comparison of Water Footprint Methods at the Regional Scale

In order to compare the difference in the ranks of each region and each characterised water footprint, the characterisation factors and post-characterised water footprints as calculated using the methods of Hoekstra et al. (2011); Pfister et al. (2009); Berger et al. (2014) and Boulay et al. (2016), are ranked, and can be seen in Figure 5.43 and Figure 5.44, respectively. Both the characterisation factors and the post-characterised water footprints are ranked in ascending order from 1 (lowest) to 5 (largest).

According to Figure 5.43, the Canterbury region of New Zealand ranks highest for three of the four methods (the exception being the *AMD*). In contrast, the West Coast region of New Zealand ranks lowest for all four of the characterisation factors calculated. The Manawatu-Wanganui region displays varying ranks from greatest when using the *AMD* to lowest when using the *WSI*. Similarly, the Wellington region ranges from fourth highest (using the *WDI* and *WSI* methods) to second (using the *AMD* and *WS_{blue}* methods). The *AMD* can be seen to rank the Wellington, Taranaki and Canterbury regions equal second. Interestingly, across all regions, the *WDI* and *WSI* characterisation factors are identical, with the exception of the Manawatu-Wanganui region, where the *WSI* ranks two greater.

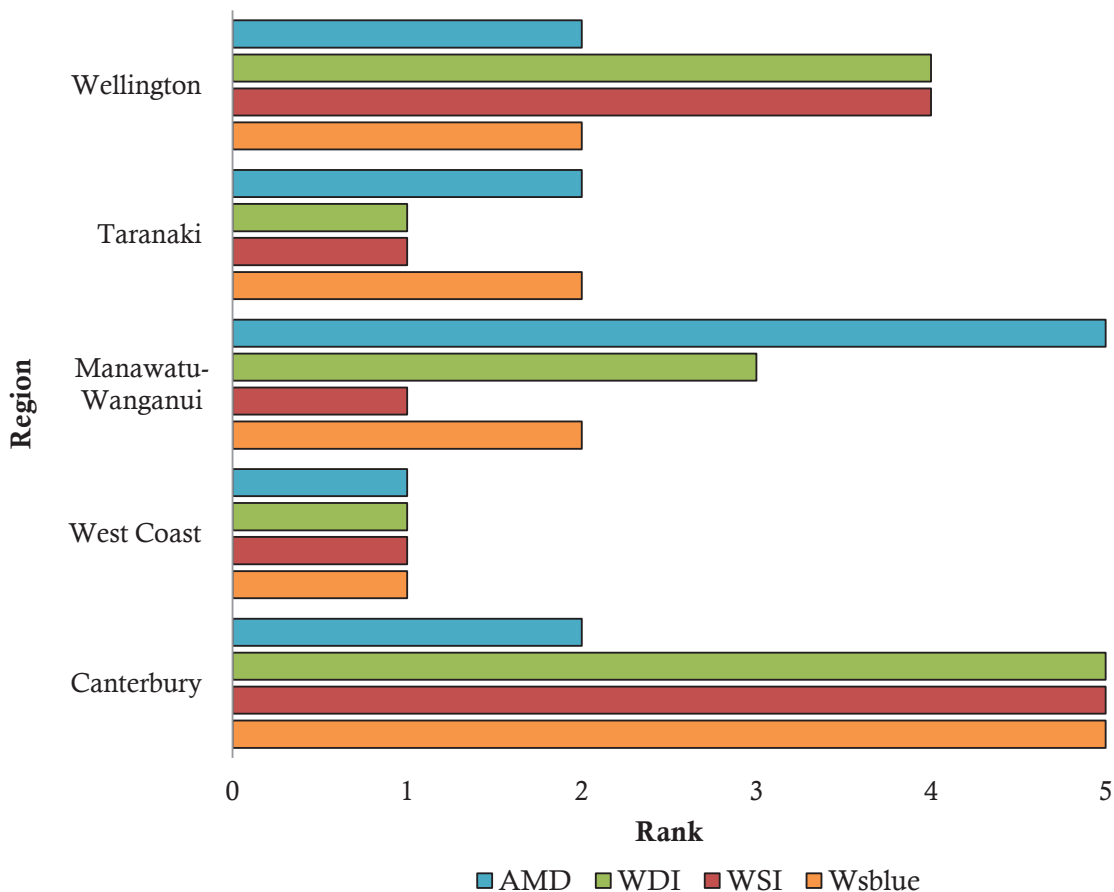


Figure 5.43 Ranking of the selected characterisation factors (in ascending order) at the regional scale.

As can be seen in Figure 5.44, the Greymouth plant can be seen to rank equally across methods as having the lowest environmental impact. For all other batching plants there appears to be more variation across methods. The Ashburton plant ranks highest when calculated using the *WDI* and *WFII* CFs. Masterton also ranks amongst the highest in terms of environmental impact (at fifth and sixth depending on the method). When comparing the results of the initial volumetric water footprint to the water footprints post-characterisation, it can be seen that the Ashburton plant ranks lowest in terms of volumetric water footprint, but amongst the highest post-characterisation (with the exception of the *AMD*). In contrast, Masterton has the highest volumetric water footprint rank (fifth), and ranks between fourth and fifth post-characterisation.

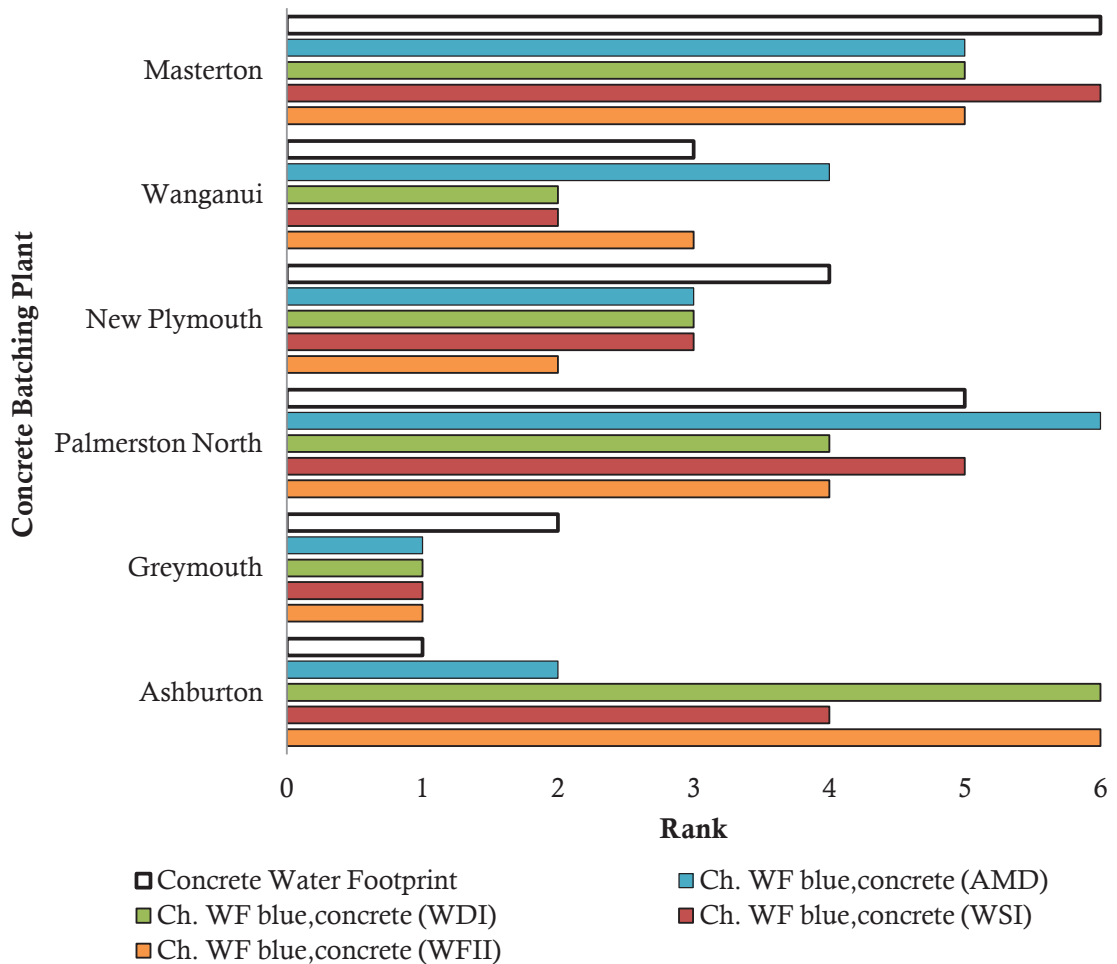


Figure 5.44 Ranking of the selected characterised water footprint results (in ascending order) at regional scale, for the 2013 period.

5.3 The Effect of Spatial Scale on Characterisation Factors

The Blue Water Scarcity Index (Hoekstra et al. 2011)

Figure 5.45 shows that the WS_{blue} can be seen to vary as a function of the spatial scale. Freshwater management zone (FMZ) scale analysis appears to yield a much higher WS_{blue} than all other scales with regard to the Ashburton plant at 0.79. However, across all scales, the Ashburton batching plant consistently has a higher WS_{blue} than all other batching plants. In contrast, the Wanganui plant has the lowest overall WS_{blue} , at both the FMZ and catchment scale. At the regional scale, however, the Greymouth batching plant can be seen to have the lowest calculated WS_{blue} , at 0.0006.

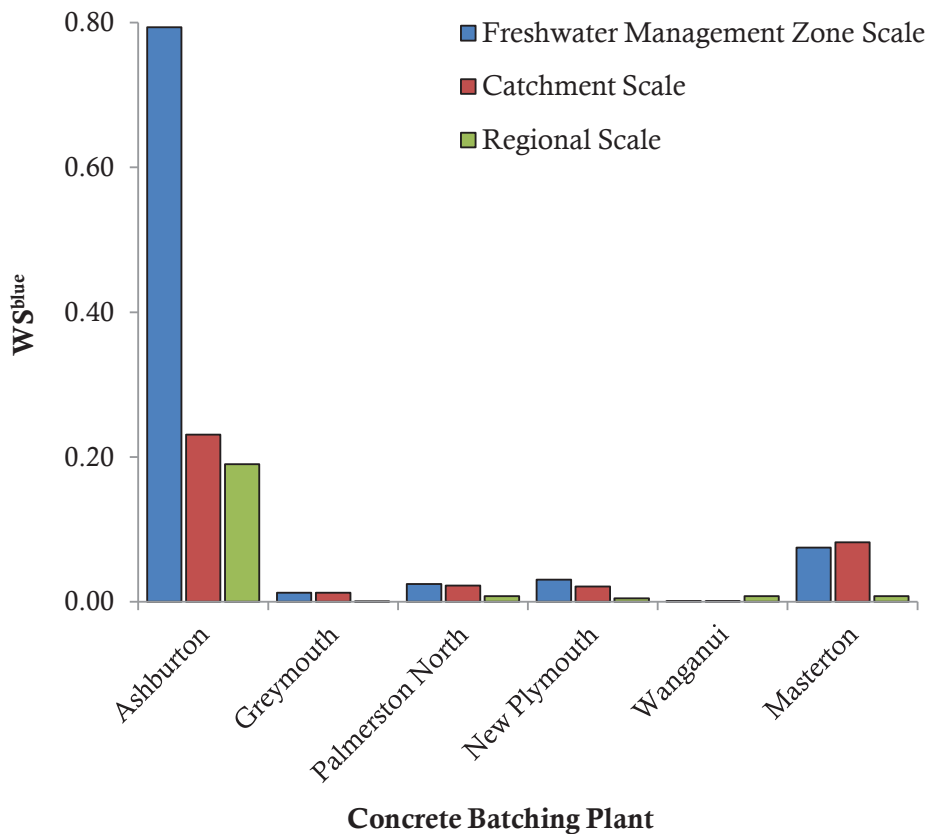


Figure 5.45 A comparison of blue water scarcity (WS_{blue}) across all spatial scales at selected concrete batching plants throughout New Zealand.

The Water Stress Index (Pfister et al. 2009)

The water stress index (*WSI*) in Figure 5.46 can be seen to vary as a function of the spatial scale. However, a clear trend can be seen across concrete batching plants with the catchment scale analysis yielding the highest *WSI*, followed by the regional scale, and lowest by the FMZ scale. The *WSI* calculated at the catchment and regional scales appear more closely related than that of the FMZ scale, which can be seen to be much lower. For example, at the Palmerston North batching plant, the regional scale *WSI* is almost the same as the catchment scale *WSI* (0.011 compared with 0.010, respectively). At the Wanganui batching plant this is, in fact, the case with the catchment and regional scale analysis yielding identical *WSI* (at 0.010).

In contrast with Figure 5.45, the *WSI* calculated at the FMZ scale yield the smallest values for all batching plants. The *WSI* can be seen to range from 0.005 in the Manawatu, Waiwhakaiho and Grey FMZs, to 0.016 in the Ashburton FMZ.

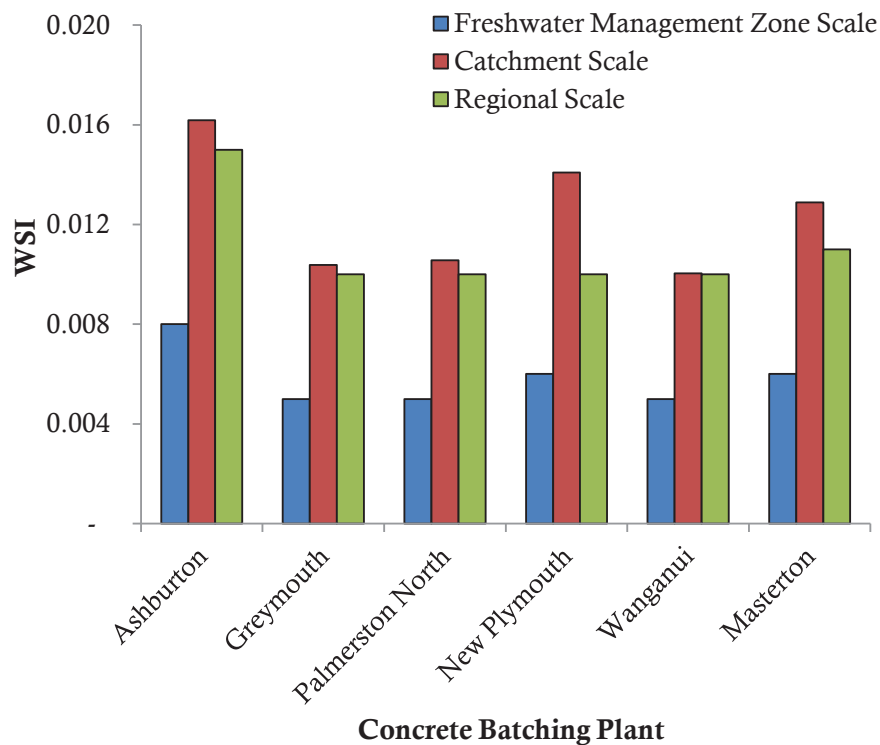


Figure 5.46 A comparison of the water stress index (*WSI*) across all spatial scales at selected concrete batching plants throughout New Zealand.

The Water Depletion Index (Berger et al. 2014)

Similar to the water stress index (*WSI*) results in Figure 5.46, the water depletion index (*WDI*) in Figure 5.47 is highest at the catchment scale (with the exception of the Wanganui batching plant, in which calculated *WDI* is highest at the regional scale). The highest calculated *WDI* of 0.06 was calculated at the catchment scale for the Ashburton batching plant. Similar to Figure 5.46, the catchment and regional scale *WDI* are closely related, especially when compared with the FMZ scale results, which are notably lower for all selected batching plants. When calculated at the FMZ scale, *WDI* ranges from 0.005 in the Grey FMZ, Manawatu FMZ and Wanganui FMZ through to 0.01 in the Ashburton FMZ.

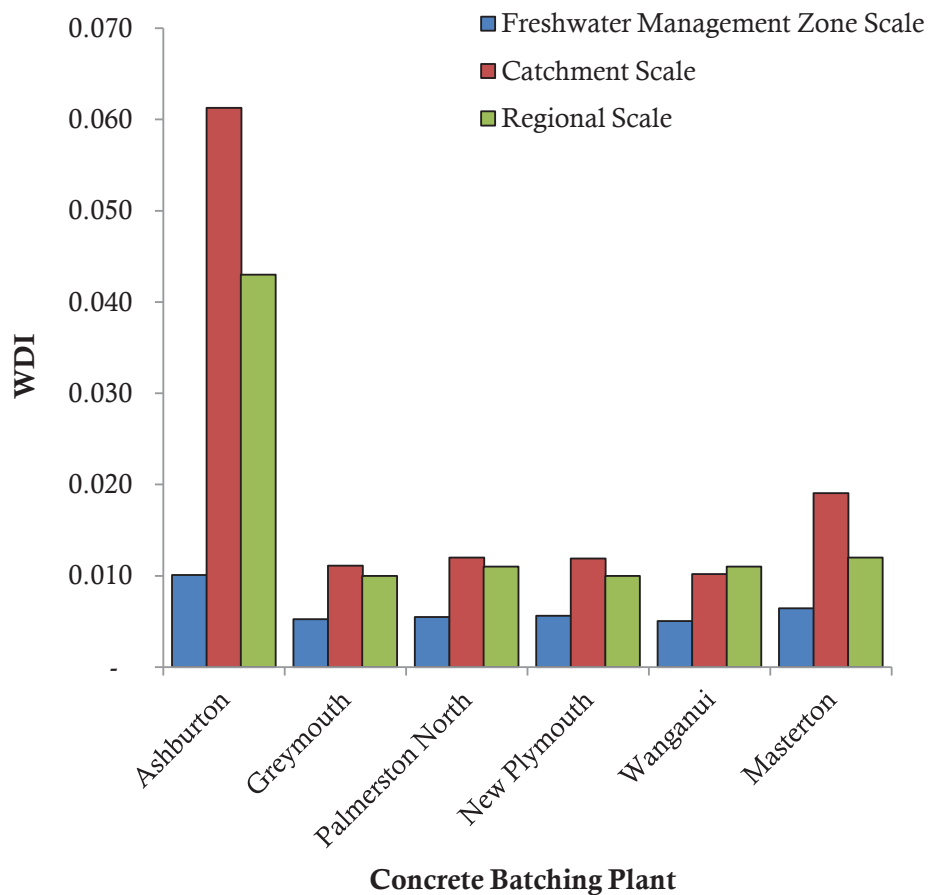


Figure 5.47 A comparison of the water depletion index (*WDI*) across all spatial scales at selected concrete batching plants throughout New Zealand.

The Availability Minus Demand (WULCA, 2012)

The *AMD* varies depending on scale *and* batching plant concerned (Figure 5.48). For example, the New Plymouth, Wanganui and Masterton batching plants have a higher *AMD* at the regional scale. Interestingly, the FMZ and catchment scale *AMD*s are identical for all selected batching plants (with the exception of Ashburton and Masterton). The *AMD* can be seen to range from 0.004 in West Coast region to 0.031 in the Manawatu FMZ and catchment.

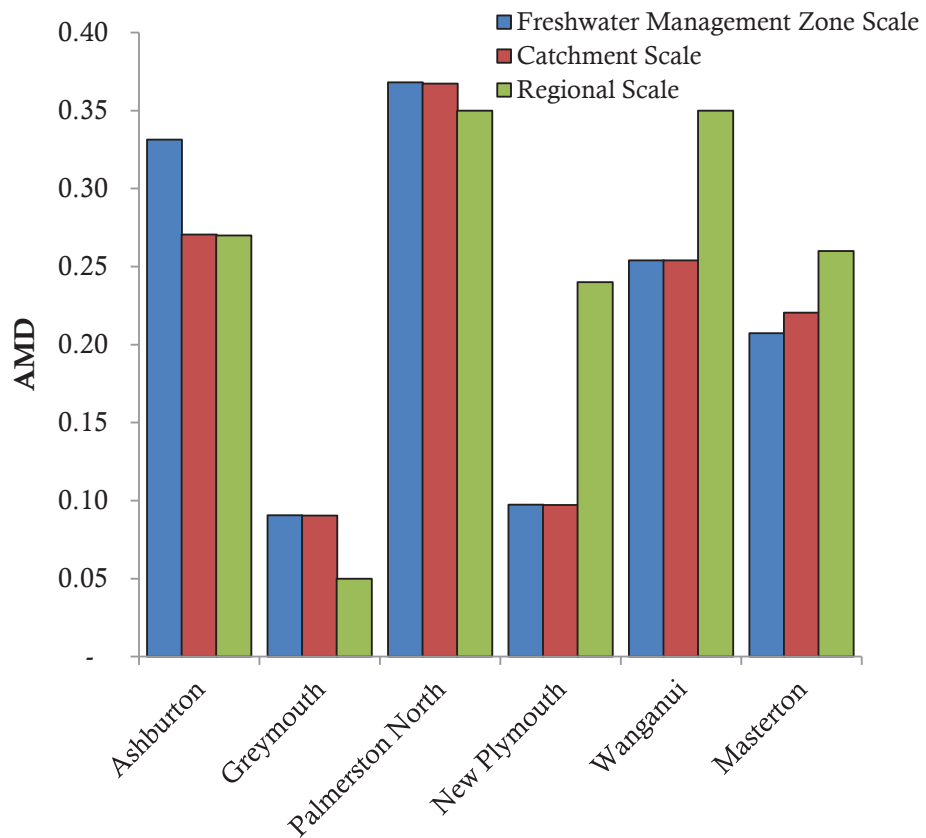


Figure 5.48 A comparison of the availability minus demand (*AMD*) across all spatial scales at selected concrete batching plants throughout New Zealand.

5.4 Co-efficient of Variation

Referring to Figure 5.49, it can be seen that the co-efficient of variation (CV) for the estimates of WS_{blue} (Hoekstra et al., 2011), WSI (Pfister et al., 2009) and *AMD* (Boulay et al., 2016) characterisation factors across all scales are greatest at the freshwater

management zone (FMZ) scale. The *WDI* (Berger et al., 2014), however, has a greater CV at the catchment scale.

In terms of methodological comparison, it can be seen that the *WS_{blue}* has a notably higher CV at all scales, compared with all other methods employed, ranging from 142% to 201%. In contrast, the *WSI* has the lowest of all calculated CV's across all spatial scales, ranging from 18% to 22%.

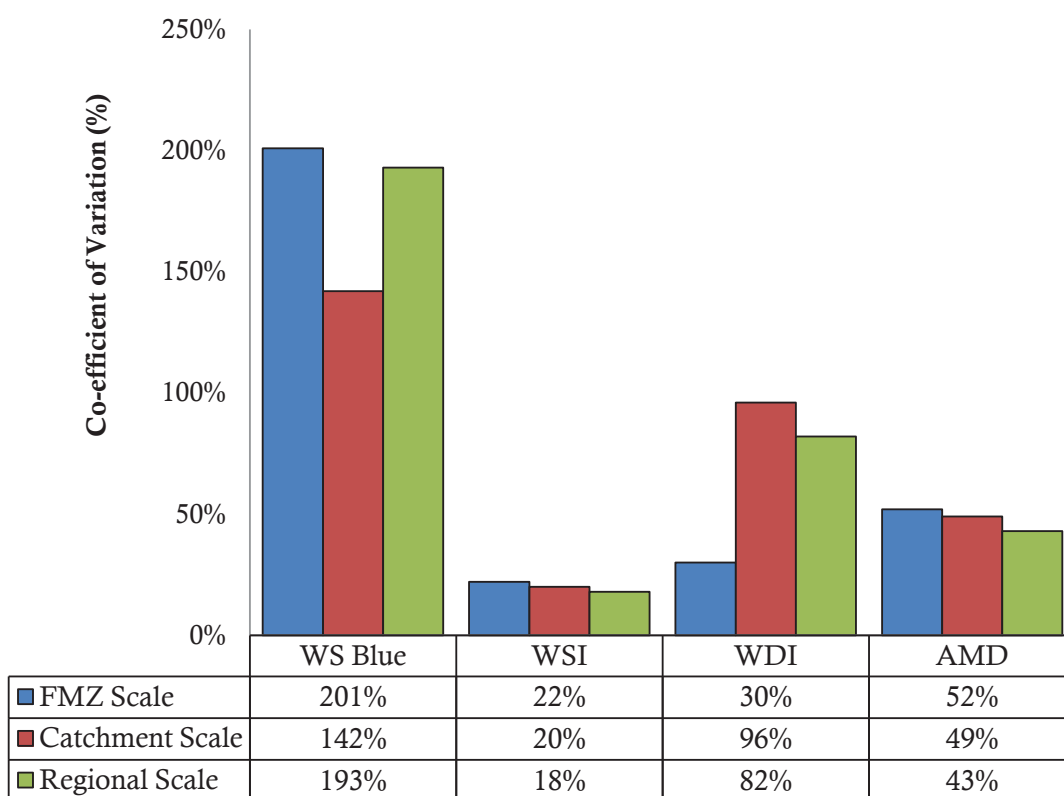


Figure 5.49 Co-efficient of variation for all methods and spatial scales considered during impact assessment

5.5 Variance Based Sensitivity Analysis

5.5.1 The *WS_{blue}* (Hoekstra et al., 2011)

The variation in the key variables used to calculate the *WS_{blue}* characterisation factor across all spatial scales can be seen in Table 5.3. In accordance with Equation 4, the

volume of freshwater consumption in each selected boundary, and the volume of available water (defined as natural runoff minus the environmental flow requirements) are assessed.

At all spatial scales it can be seen that variation in the volume of freshwater consumed in the selected boundary explains the majority of variation in the calculated WS_{blue} . The volume of available freshwater accounted for little variation between scales, ranging from 0.03% to 0.41% of variation.

Table 5.3 Coefficient of determination (%) for variables that contribute to the calculation of the WS_{blue} characterisation factor.

	Consumption	Availability
FMZ Scale	98.73	0.41
Catchment Scale	97.53	0.04
Regional Scale	99.58	0.03

Table 5.4 shows the co-efficient of determination between key variables considered in the calculation of the WS_{blue} , against the characterised water footprint of concrete at the three distinct spatial scales. When the volumetric water footprints of each batching plants (refer Figure 5.4) are characterised by the local WS_{blue} , the volume of freshwater consumed within each respective boundary can be seen to contribute the most to the variation in the calculated water footprints. The volumetric water footprint, however, accounts for between 0.78% and 16.03% dependant on the scale of the analysis.

Table 5.4 Coefficient of determination (%) for variables that contribute to the water footprint of concrete post-characterisation with the WS_{blue} .

	Consumption	Availability	Volumetric WF
FMZ Scale	96.69	0.44	12.73
Catchment Scale	82.23	1.09	0.78
Regional Scale	96.99	0.6	16.03

5.5.2 The *WSI* (Pfister et al., 2009)

The variation in the key variables used to calculate the *WSI* characterisation factor across all spatial scales can be seen in Table 5.5. In accordance with Equation 6, the volume of freshwater abstracted in each selected boundary, the volume of available freshwater and the variation factor (*VF*) are assessed.

When calculated at different spatial scales, the volume of abstracted freshwater accounts for the majority of the variation in the calculated *WSI*. Depending on the spatial scale of the assessment, the volume of available freshwater and the *VF* parameters have varying significance to the overall *WSI*. At the catchment scale, both the available freshwater and *VF* are more important to the *WSI* than at the freshwater management zone or regional scale.

Table 5.5 Coefficient of determination (%) for variables that contribute to the calculation of the *WSI* characterisation factor.

	Availability	Abstraction	<i>VF</i>
FWM Scale	0.14	91.7	1.95
Catchment Scale	19.94	61.22	1.58
Regional Scale	0.54	97.53	9.06

Table 5.5 shows the co-efficient of determination between key variables considered in the calculation of the *WSI*, against the post-characterisation water footprint of concrete at the three distinct spatial scales.

Table 5.6 Coefficient of determination (%) for variables that contribute to the water footprint of concrete post-characterisation with the *WSI*.

	Availability	Abstracted	<i>VF</i>	Volumetric WF
FMZ Scale	26.15	5.38	56.93	54.3
Catchment Scale	43.85	4.45	71.5	64.93
Regional Scale	27.87	1.55	77.94	69.85

When the volumetric water footprints of concrete (Figure 5.14) are multiplied by the aforementioned *WSI* characterisation factor, it can be seen in Table 5.6 that the variation in the characterised water footprint is more distributed, when compared with the variation in the characterisation factor itself (refer to Table 5.5). At all scales, the *VF* can be seen to account for the most variation in the characterised water footprints, followed by the volumetric water footprint. At all spatial scales, the volume of freshwater abstracted appears to account the least for the variation in the characterised water footprints of concrete. In contrast, freshwater abstractions can be seen to account for the most variation in the *WSI* characterisation factor (Table 5.5).

5.5.3 The WDI (Berger et al., 2014)

The variation in the key variables used to calculate the *WDI* characterisation factor across all spatial scales can be seen in Table 5.7. In accordance with Equation 17 the volume of annual freshwater consumption, annual water availability, and the annual useable surface water stocks (*SWS*) are considered. Note: the volume of groundwater stocks accounted for in Equation 16 were assumed to be 0.95 for all selected boundaries, and as such groundwater stocks do not account for any of the variation in the *WDI*.

Table 5.7 Coefficient of determination (%) for variables that contribute to the calculation of the *WDI* characterisation factor.

	Consumption	Availability	<i>SWS</i>
FMZ Scale	98.11	0.06	0.03
Catchment Scale	98.96	0.18	0.11
Regional Scale	99.96	0.0001	71.88

Table 5.8 shows the co-efficient of determination between key variables considered in the calculation of the *WDI*, against the post-characterised water footprint of concrete at the three distinct spatial scales. When characterised by the aforementioned *WDI*, variation in the characterised water footprint of concrete plants stems from the volume of freshwater consumption in the selected boundary (Table 5.8). The volume of

available freshwater accounts for only 0.58% to 11.83% of the total variation in the characterised water footprints at different spatial scales. At the regional scale, the volume of surface water stocks can be seen to have a much larger impact on the total variation in the characterised water footprints, compared with freshwater management zone and catchment scale analysis.

Table 5.8 Coefficient of determination (%) for variables that contribute to the water footprint of concrete post-characterisation with the *WDI*.

	Consumption	Availability	SWS	Volumetric WF
FMZ Scale	27.26	11.83	1.94	41.46
Catchment Scale	91.74	0.58	0.31	0.13
Regional Scale	89.42	2.39	53.57	1.87

5.5.4 The *AMD* (Boulay et al., 2016)

The variation in the key variables used to calculate the *AMD* characterisation factor across all spatial scales can be seen in Table 5.9. In accordance with Equation 19, the volume of annual freshwater demand (where demand is the available freshwater minus human and ecosystem demand) and annual water availability are considered.

It can be seen that at the regional scale, the volume of both freshwater demand and availability could account for higher variation in the calculated *AMD*. In contrast, at the FMZ scale and catchment scale, the volume of human and ecosystem demand, and the available freshwater appear to account for little of the variation, at below 2%.

Table 5.9 Coefficient of determination (%) for variables that contribute to the calculation of the *AMD* characterisation factor

	Demand	Availability
FMZ Scale	1.55	0.38
Catchment Scale	1.02	0.86
Regional Scale	80.0	79.8

Table 5.10 shows the co-efficient of determination between key variables considered in the calculation of the *AMD*, volumetric water footprint and the characterised water footprint of concrete at the three distinct spatial scales. As with the characterisation factor itself (Table 5.9) at the regional scale, both the volume of freshwater availability and demand can be seen to account for the majority of the variation in the final characterised water footprint. At the freshwater management zone and catchment scales these variables account for less of the total variation amongst the selected batching plants. It can be seen that the volumetric water footprints account for significant variation in the characterised water footprints at each spatial scale.

Table 5.10 Coefficient of determination (%) for variables that contribute to the water footprint of concrete post-characterisation with the *AMD*.

	Demand	Availability	Volumetric WF
FMZ Scale	1.67	4.12	34.47
Catchment Scale	3.85	5.49	46.99
Regional Scale	73.93	73.39	54.45

Chapter 6

Discussion

This chapter will firstly explore the freshwater consumption and volumetric water footprints of concrete, and how these vary between batching plants (section 6.1 and 6.2). This is followed by a discussion about the influence of methodological choices on the final water footprint results. The variation in the calculated characterisation factors as a function of the methods is evaluated and discussed (section 6.3.1), as well as how the spatial scale of the analysis alters the results across batching plants (section 6.3.2).

6.1 Variation in Freshwater Consumption Between Concrete Batching Plants

A visual comparison of the proportion of freshwater consumed by each concrete batching plant (Figure 5.1) and the volume of concrete produced by the respective plants (Figure 5.3) confirms that a relationship exists between production volume and freshwater consumption. For example, the Ashby's Ready Mix and Christchurch batching plants have the highest share of water consumption at 20.2% and 13.4%, respectively (Figure 5.1). In terms of their total production, these plants produce the greatest proportion (21.7% and 14.3%, respectively) of the total volume of concrete manufactured in 2013 (Figure 5.3).

The relationship between freshwater consumption at each batching plant, and a range of relevant variables accounted for in calculation of the volumetric water footprint (Equation 1, and visualised in Figure 5.1), can be seen in Table 5.1.

Referring back to the freshwater consumption calculation (Equation 1), it can be seen that three factors are accounted for:

1. Freshwater evaporation,
2. Freshwater incorporated into the product, and
3. Lost return flow.

Referring to Table 5.1, it can be seen that evaporation from storage ponds appears to have very little significance to the variation in plant consumption, explaining only 0.64% of total variation. Similarly, the lost return flow, is not considered to have a significant impact on the variability across batching plants as it occurred very little during the 2013 period (on only two occasions across all batching plants). In contrast, the volume of blue water incorporated into the product can be seen to account for 99.95% of the total variation across the 27 concrete batching plants. Figure 5.2 further investigated the volume of freshwater incorporated into the product and provides a comparison of the average volume of freshwater withdrawn from the reticulated network with the volume of recycled water. It can be seen on average 60% of the freshwater withdrawn by the 27 concrete plants is recycled, and 40% is withdrawn from the reticulated network.

It is important to note that whilst Hoekstra et al. (2011) acknowledge the importance of water recycling and re-use, stating that these can be instrumental in reducing the blue water footprint, there is no differentiation in Equation 1 between freshwater derived from the reticulated network, and that recycled from rainwater storage ponds on-site. Although the volume of recycled water from the rainwater storage ponds on-site is not explicitly considered, plants in which freshwater is recycled and re-used (as opposed to derived from the reticulated network) are not contributing to freshwater allocation problems. This is a particularly important consideration in areas that have allocation problems (refer to Figure 2.4) e.g. the Canterbury region (in which the Ashby's Ready Mix, Ashburton and Christchurch batching plants are located).

Referring briefly to the volumetric water footprint results in Figure 5.4, it can be seen that the volumetric blue water footprint of 1 m³ concrete at the Ashburton and Wanganui batching plants are 0.15 and 0.18 m³ water per m³ concrete, respectively. This indicates that the Wanganui batching plant consumed more freshwater to produce the same volume of concrete, and thus has the greater environmental impact of the two plants. However, it can be seen in Figure 5.6 that the Ashburton plant withdraws the majority of freshwater from the reticulated network as opposed to

rainwater storage ponds on-site⁴⁵. In comparison, the Wanganui batching plant derives the majority of freshwater from rainwater storage ponds onsite (Figure 5.10). Thus, the Ashburton batching plants has a larger impact on the freshwater resource than plants such as Wanganui, for example, which derive the majority of freshwater from rainwater storage ponds onsite.

It is therefore important to distinguish the 'type' of freshwater used by each concrete batching plant, alongside their location and the local freshwater allocation status, in order to gain a more comprehensive picture of freshwater consumption between plants (Pfister et al., 2009). To this end, the schematic diagrams for each plant (refer to Appendix K) can be used alongside the results of Figure 5.4, in order to distinguish between the freshwater source and provide additional information for response formulation.

6.2 Variation in the Volumetric Water Footprint of 1 m³ Concrete

When the volume of freshwater consumed by each concrete batching plant is divided by the concrete production volume in that period (in accordance with Equation 24) variability in the volumetric water footprint between plants is apparent. For example, according to Figure 5.4, the volumetric blue water footprint of 1 m³ of ready mix concrete at the Ashby's Ready Mix and Christchurch batching plants are relatively low (at 0.15 and 0.16 m³ water/ m³ concrete, respectively). In contrast, the Masterton plant has the highest volumetric blue water footprint at 0.29 m³ water/ m³ concrete. This variation in the water footprints amongst concrete plants is interesting because, in theory, the volume of water required to produce 1 m³ of concrete should be similar (if not identical) for each concrete plant since the manufacturing procedure is the same throughout the 27 concrete batching plants studied, and there is no notable difference in water consumption between compressive strengths (refer to Appendix I).

6.2.1 Concrete Production and Freshwater Consumption

An analysis between the relative percentages of concrete production and freshwater consumption in relation to the volumetric water footprint, across all 27 concrete

⁴⁵ Similarly the Ashby's Ready Mix and Christchurch batching plants also withdraw more freshwater from the reticulated network than is recycled from rainwater storage ponds onsite (refer to Appendix K).

batching plants, can be seen in Figure 5.5. The variation in the volumetric water footprints between batching plants is almost solely explained by the variation in the volume of concrete produced or freshwater consumed (as x is a function of y). It can be seen that for batching plants where the total volume of concrete production exceeds the volume of freshwater consumption, that the normalised water footprint of 1 m^3 is below the average (of one). This is demonstrated clearly for the Christchurch and Ashby's Ready Mix plants, where the relative percentages of concrete production and freshwater consumption are the highest of all 27 concrete batching plants studied. In contrast, they have amongst the lowest normalised blue water footprints at 0.88 and 0.87 m^3 concrete per m^3 freshwater, respectively (refer to Figure 5.5). In both instances, the volume of concrete produced exceeded the volume of freshwater consumed (by 1.55% at Ashby's ready mix plant and 0.88% at the Christchurch batching plant).

Contrastingly, the Masterton, Palmerston North and Mosgiel batching plants have the highest normalised blue water footprints at 1.60, 1.37 and 1.21 m^3 concrete per m^3 freshwater, respectively. As can be seen in Figure 5.5, these plants have comparatively lower concrete production and freshwater consumption volumes. However, for each of these plants, the volume of freshwater consumed is greater than the volume of concrete produced. For the Masterton, Palmerston North and Mosgiel plants freshwater consumption is 1.2%, 0.8% and 1.3% greater than the concrete production volume for the 2013 period.

Similarly, freshwater consumption exceeds concrete production at the Hokitika and Nelson batching plants, but to a lesser extent (0.06%, respectively). It can be seen in Figure 5.5 that even for these two plants, in which freshwater consumption is only marginally greater than concrete production, that the normalised water footprints are slightly higher, and closer to the average than other batching plants with similar production volumes. This is demonstrated clearly with a comparison of the Gore and Nelson batching plants. The Gore batching plant has a normalised blue water footprint of 0.90 m^3 concrete per m^3 freshwater, and the Nelson plant 0.96 m^3 concrete per m^3 freshwater. Concrete production volumes at the two plants are similar (2.26% and 2.29% of the total concrete production, respectively). In terms of the volume of

freshwater consumed, the two plants are again similar (at 2.17% and 2.35% of the total volume, respectively). However, the Nelson plant has a normalised water footprint greater than the Gore plant because the proportion of consumed water is greater than the volume of concrete production (by 0.06%). In contrast, the Gore plant had 0.08% more concrete production relative to freshwater consumption, and can be seen to have a smaller normalised blue water footprint (refer to Figure 5.5).

6.2.2 Batching Plant Efficiency

Whilst the regression analysis in Table 5.1 goes some way to explain the relationships between the relevant variables and the freshwater consumption of each plant, it does not consider the efficiency of batching plant processes. This is another potential source of variation in the water footprints amongst concrete batching plants that needs to be considered. As can be seen in Figure 5.1 the Masterton plant has much higher freshwater consumption than that of the Ashby's Ready Mix plant (2.7% to 20.2% of the total, respectively). However, the Masterton batching plant is amongst the smallest of Allied Concretes national plants, compared with Ashby's Ready Mix, which is the largest Allied plant in the country⁴⁶. To this end, incentives to improve the freshwater efficiency are targeted at larger plants such as Ashby's. Compounding this, the Ashby's Ready Mix plant is located in an area with strict wastewater discharge controls, hence the focus on recycling and re-using freshwater (personal communication, C. Munn, 13th Jan 2016).

6.3 Environmental Impact Assessment of 1 m³ Concrete Production

6.3.1 Variation in Characterisation Factors as a Function of Method

The chosen water footprint characterisation factors; WS_{blue} (Hoekstra et al., 2011), WSI (Pfister et al., 2009), WDI (Berger et al., 2014) and AMD (Boulay et al., 2016) were all chosen as they provide different approaches, using varying hydrological considerations, to characterise volumetric water footprints at the finer resolution. The results displayed in Chapter 5 show that there is clear variation in the calculated CFs as a function of both the spatial scale and the method employed. A ranking system was

⁴⁶ Based on concrete production volumes during the 2013 period (refer Figure 5.3).

applied to the CFs for all methods at the FMZ scale (Figure 5.21), catchment scale (Figure 5.32) and regional scale (Figure 5.43). At all scales, results of the *WDI*, *WSI* and *WS_{blue}* CFs ranked the Ashburton FMZ and catchment/ Canterbury region the highest. In contrast, the *AMD* CF ranked the Manawatu FMZ and catchment/ Manawatu-Wanganui region highest in terms of water stress, at all the spatial scales analysed. Similarly, the FMZ/catchment/region with the lowest calculated CF can be seen to be to vary. At all spatial scales the Greymouth and Wanganui FMZs and catchments/ West Coast and Manawatu-Wanganui regions can be seen to have the lowest CF values, depending on the chosen method. For example, at the FMZ scale, the *WSI*, *WDI* and *WS_{blue}* ranked the Wanganui FMZ as having the lowest water stress. However, the *AMD* ranked the Greymouth FMZ the lowest.

It appears that in some instances the *WSI*, *WDI* and *WS_{blue}* characterisation factors are more closely related (and thus rank more closely) than the *AMD*. For example, as can be seen at the FMZ scale (Figure 5.21) and the catchment scale (Figure 5.32) the *WSI* and *WS_{blue}* ranked FMZs identically. Similarly, at the regional scale the *WDI* and *WSI* ranked identically in four of the five instances.

Whilst there can be seen to be some trends in the relative ranking of the selected boundaries, there is also clear variation in the characterisation factors between boundaries and methods. A notable example of the variation in the rank of the CFs themselves can be seen in Figure 5.43, where it can be seen that the Manawatu region has the lowest possible rank of one when calculated according to the *WSI*. In contrast, when calculated according to the *AMD*, the Manawatu region can be seen to have the highest rank (five) (i.e. the highest water stress level).

A possible source of this apparent variation between the calculated CFs (and thus their overall ranks) are the background methods and modelling choices employed by each method. For the *WSI* and *WDI* methods, the CFs are each scaled to fit obtained values within a logistic curve between 0.01 and 1. In contrast, neither the *WS_{blue}* nor the *AMD* apply a scaling factor to the final CF. Whilst Boulay et al. (2016) do not scale the *AMD*, they do apply cut-off criteria (refer to Equation 22). Another point of difference

between the *AMD* and the calculated *WSI*, *WDI* and *WS_{blue}* at all spatial scales is the normalisation with the world average. Referring back to Equation 22, it can be seen that the *AMD* is normalised by the world average *AMD* (*AMD_{world ave}*). A further source of variation between the results as calculated using the *WS_{blue}* and *AMD* characterisation factors, is the calculation of *EFR*, which differs between the two methods. Hoekstra et al. (2011) using a value of 80% of natural flow and Boulay et al. (2016) using the guidelines of (Pastor et al., 2013).

In order to understand which variables in each method are accounting for the most variation in the calculated characterisation factors, a regression analysis (see section 5.5) was conducted. It can be seen in Table 5.3 for the *WS_{blue}* CF that the volume of consumed freshwater accounts for a much higher proportion of the observed variation than the freshwater availability (defined as natural flow minus the environmental flow requirement). Similarly, Table 5.7 shows that across all spatial scales, variation in the calculated *WDI* is caused by variation in consumptive volumes between batching plants. As the *WSI* method is based on a withdrawal-to-availability (*WTA*) ratio and not a consumption-to-availability (*CTA*) ratio (as is the case with the *WS_{blue}* and *WDI*), the volume of withdrawn freshwater was assessed in Table 5.5. It can be seen that, as with the volume of freshwater consumption for the *WS_{blue}* and *WSI*, the volume of withdrawal account for the majority of variation in the calculated *WSI* across all spatial scales. Of less importance to overall variation were the method specific *VF*s (refer to Figure 4.9 and Figure 4.10). Lastly, the key variables used to calculate *AMD* are assessed in Figure 5.10. In contrast with the regression results for all other calculated CFs, it can be seen that the volume of freshwater demand (i.e. consumption) is not responsible for the majority of variation across all scales, as with other methods. In this instance, there is no clearly dominant parameter at any spatial scale.

Thus it is clear that for some methods there is a dominant parameter driving the variation in the overall water footprint. For example, for the *WS_{blue}* (Table 5.3) and *WDI* (Table 5.7) methods that is the volume of freshwater consumed (which accounts for almost all variations throughout the spatial scales). Similarly, for the *WSI* method it

is the volume of withdrawal (Table 5.5). For others, such as the *AMD*, there appears to be no dominant driver of variation, but a range of contributing variables. It is possible that the interactions between these variables explain more variation than if the variables are considered alone. A limitation of the aforementioned regression analysis is that the variables are not truly independent, as one hydrological process will affect the next. Thus, results are considered indicative only.

6.3.2 Variation in Calculated Characterisation Factors as a Function of Spatial Scale

Scale is an important consideration when conducting water footprint analysis as the scale of the assessment can affect how appropriate for decision support a water footprint may be. For example, whilst useful for global comparisons, a national scale analysis will not provide the level of detail required for decision support at the regional (or finer) resolution.

The calculated CFs varied amongst methods as a function of the spatial scale of the analysis. As can be seen in Chapter 5, across all spatial scales and methods, the Ashburton batching plant has the highest calculated CF (9 out of the 12 times), and is ranked second highest for the remaining three occurrences. This is followed by the Palmerston North batching plant (across all spatial scales) which had the highest calculated CF (4 out of the 12 times).

The Ashburton catchment high rank across all methods and scales can be explained by the localised hydrological conditions alongside the freshwater demand. Ashburton is located within the Canterbury region, and more specifically in a region that is reported to exceed (or border on exceeding) the limits of sustainable water use (Land Air Water Aotearoa, 2016). This is illustrated in Figure 2.4.

A comparison of the Ashburton and Manawatu catchments shows that, on average, Ashburton has more available freshwater than Manawatu (refer to Figure 5.23), receiving more annual rainfall (9.30 and 7.90 billion m³/year, respectively). This suggests that the Ashburton batching plant has more natural runoff to utilise than the Manawatu catchment. However, in terms of the volume of consented freshwater, the

Ashburton catchment allows over ten times the volume of freshwater to be consented than the Manawatu catchment (refer to Figure 5.23). In accordance with MfE (2010), the percentage of abstracted water is accounted for in the environmental impact assessments, across all of the water footprint methods. It can be seen, on average, that the freshwater abstractions across all user groups in Ashburton are 57%, whilst in Masterton actual abstractions are 30% of consented freshwater (refer to Figure 4.5). Therefore, in terms of total freshwater abstractions, Ashburton abstracts, on average, more than nine hundred million m³/year than the Manawatu catchment. This accounts for the higher environmental impacts associated with freshwater consumption in Ashburton in comparison with all other batching plants.

The batching plant associated with the lowest characterisation factors, according to all methods and scales is Greymouth, which ranked lowest 8 out of the 12 times (being ranked second lowest for the remaining four occurrences). Furthermore, the Wanganui catchment ranked lowest in 6 out of the 12 occurrences (being ranked between second and fourth for two of the remaining occurrences). Greymouth is located in the West Coast of the South Island of New Zealand, often regarded as the wettest region of New Zealand (NIWA, 2001). In terms of total rainfall entering both catchments, the Greymouth catchment receives 11.68 billion m³/year, whilst the Wanganui receives more rainfall with 11.83 billion m³/year of rainfall on average (refer to Figure 2.2). In terms of the volume of consented freshwater, it can be seen in Figure 5.23 that the Greymouth catchment allows more freshwater to be consented and abstracted compared with the Wanganui catchment. The average actual abstraction rate for the Greymouth and Wanganui catchments are 48% and 30%, respectively (refer to Figure 4.5).

Thus in order to explain why Greymouth frequently has the lowest environmental impact from water scarcity according to all water footprint methods, and scales, it is necessary to look at the interactions between hydrological components within the catchments. For example, the natural runoff (R_{nat}) volumes to the Greymouth and Wanganui catchments are 10.81 and 7.19 billion m³/ year, respectively. It can be seen that there is a notable difference in the rainfall to R_{nat} ratio between the Greymouth

and Wanganui catchments. The difference between rainfall and R_{nat} is five times greater for the Wanganui catchment compared with the Greymouth catchment⁴⁷. A potential explanation for the apparent loss of freshwater in the Wanganui catchment, when compared to the Greymouth catchment, is the higher rate of evapotranspiration (refer to the national map of Penman potential evapotranspiration produced by Woods et al. (2006)). Furthermore, the Wanganui catchment has approximately triple the volume of R_{act} (runoff to sea) than the Greymouth catchment, meaning a greater volume of freshwater exists the catchment via the sea, than in the Greymouth catchment.

Co-efficient of Variation Comparison Across Scales

Comparing calculated CV's for the various methods, and all across spatial scales, it can be seen in Figure 5.49 that the WS_{blue} calculated according to the method of Hoekstra et al. (2011) clearly has the greatest variation across all spatial scales compared with all other methods. This indicates that the WS_{blue} has greater variability relative to the mean (refer to Figure A.9). Of all CFs calculated, the WSI (Pfister et al. 2009) shows the lowest variability relative to the mean, and can be seen not to vary greatly between spatial scales.

Potential causes of the variation seen in the calculated CV's could be dictated by either the variables themselves (e.g. natural runoff) as the New Zealand climate differs from region to region (refer to Appendix B). In contrast, the variation encountered could be a function of the methods themselves. As discussed in section 6.2.1, the regression analysis in section 5.5 goes some way to explain the dominant drivers of variation in the calculated CFs.

6.3.3 Variation in the Characterised Water Footprints of Concrete as a Function of Spatial Scale and Chosen Method

The Freshwater Management Zone Scale

Post-characterisation, the water footprint of 1 m³ concrete can be seen to be highest for the Ashburton batching plant (as a result of the WS_{blue} and WSI characterisation

⁴⁷ The rainfall to runoff ratio for the Greymouth catchment being 0.87 billion m³/year and 4.64 billion m³/year for the Wanganui catchment

factors) and Masterton (as a result of the *WDI* and *AMD* CFs). Interestingly, the Masterton plant had the highest volumetric blue water footprint ($0.29 \text{ m}^3 \text{ water/ m}^3 \text{ concrete}$), whilst the Ashburton plant had one of the lowest volumetric blue water footprints recorded at $0.15 \text{ m}^3 \text{ water/ m}^3 \text{ concrete}$ (refer to Figure 5.4). It's only when the existing volumetric water footprint is multiplied by the *WDI* and *WSI* characterisation factors (refer Figure 5.17 and Figure 5.15, respectively) that the Ashburton plants rank exceeds Masterton, for both the *WSI* and *WDI* methods at the freshwater management zone scale.

The WS_{blue} of a specified boundary is defined as the ratio of the total blue water footprints in the defined boundary to the blue water availability (refer to Equation 2). Thus, when calculating the WS_{blue} , for example, consideration is given to *all* processes in a specified boundary that are consuming freshwater. Therefore, Masterton is likely ranked highest due to its initially high volumetric water footprint (which is caused by a higher proportion of freshwater consumption compared with concrete production (refer to section 6.2.1)). The ranking of the Ashburton batching plant is reversed post-characterisation because the characterisation factor reflects the greater shortage of water in the Ashburton area, as it located in an area that is bordering over allocation (MfE, 2010) .

Contrastingly, the water footprints post-characterisation of the Greymouth and Wanganui batching plants were ranked lowest in terms of their potential environmental impacts. Greymouth was ranked the lowest for all methods, with the exception of the WS_{blue} , in which the Wanganui batching plant was reported to have the lowest environmental impact. The WS_{blue} ranking, in this instance, can be explained by the modified water footprint calculation (Equation 24) which has been adapted from the original (refer to Equation 2) to include the volume of consumed freshwater. Figure 4.5 shows the typical regional freshwater abstraction per selected region. It can be seen that on the West Coast (containing the Greymouth batching plant) the average freshwater abstraction for all users is 68%, compared with the Manawatu-Wanganui region at 29%. Thus when applying the modified water

footprint calculation, the Greymouth boundary is regarded as having a greater water footprint, and is thus ranked higher.

A regression analysis (see section 5.5) was conducted to explore the variability between characterised water footprints as a result of the varying hydrological parameters considered in each method. When the volumetric water footprints (Figure 5.4) are characterised, the *VF* becomes more significant (compared to the *WDI* in Table 5.5, to explaining the variation in final water footprints for the *WSI* method (Table 5.6). As the *VF* accounts for strongly regulated flows, it thus increases water stress which could account for the higher water footprint calculated using this method. In contrast, the parameter that accounts for the majority of variation in the water footprints post-characterisation with the *WDI* method is the volume of consumed freshwater (as is the case for the *WDI* CF in Table 5.7). Thus these variables are responsible for the majority of the variation that can be seen between methods at the FMZ scale.

The Catchment Scale

The relative ranking of characterised water footprints at the catchment scale shows a unanimous rank across all methods for the Greymouth batching plant⁴⁸ (Figure 5.33). Greymouth can be seen to have the lowest environmental impact associated with freshwater withdrawal/consumption. In terms of the batching plant with the highest rank (and therefore predicted environmental impact), there is noticeably more variation as a function of the chosen method. Amongst those ranked the highest are Palmerston North (*AMD*), Ashburton (*WS_{blue}* & *WDI*), and Masterton (*WSI*). As discussed for the FMZ scale analysis, the variation in the characterised water footprint could be due to the difference in the background criteria required to calculate the respective CFs. Post-characterisation, the Palmerston North plant is considered to have the greatest environmental impact, according to Boulay et al. (2016). Factors influencing this characterisation are the high volumetric blue water footprint of the Palmerton North batching plant (0.24 m³ water/ m³ concrete) and the normalisation with the global average water *AMD* (0.0136 m³/m²-month). The AWaRe method is the

⁴⁸ It should be noted that Wanganui was ranked joint lowest on two occasions.

only one, in this instance, that normalises the characterised water footprint with the global average.

The co-efficient of determination at the catchment scale (refer to section 4.6.2) is similar to the aforementioned variation in the FMZ scale results. Freshwater consumption can be seen to account for the majority of the calculated variation in the WS_{blue} , WDI and AMD methods, whilst the volume of freshwater withdrawal is the dominant factor accounting for variation using the WSI method (due to the use of the WTA not CTA ratio in this method).

The Regional Scale

At the regional scale, the Ashburton batching plant (Canterbury region) has the largest water footprint impact according to the WS_{blue} and WDI methods. In contrast, the WSI calculates the water footprint of the Masterton plant to exceed Ashburton post-characterisation. In terms of the AMD , the Palmerston North plant is calculated as having the largest water footprint impact.

The regression analysis (refer to section 4.6.2) clearly shows that at the regional scale the volume of freshwater consumed is almost entirely responsible for the variation in the final characterised water footprint using the WS_{blue} and WDI method. As the WSI does not consider the volume of freshwater consumption, merely abstracted the change in ranking, post-characterisation, is likely due to the difference in calculation, as all other plants account for consumptive volumes. In terms of the AMD , the Palmerston North plant is ranked highest, as can be seen in Figure 5.43. In contrast to all other methods, the dominant driver of variation when using the AMD at the regional scale is the natural runoff (R_{nat}) and the environmental flow requirement (EFR).

Similar to analysis at the catchment scale, Figure 5.44 shows how each method ranks the characterised water footprint of the selected batching plants. It can be seen that the Greymouth plant is unanimously ranked lowest (as at the catchment scale), and that there is more variation amongst all other rankings and methods. Interestingly, Ashburton is ranked second lowest using the AMD method (Figure 5.42). However, as

can be seen in Figure 5.44, this can be explained by the dominant parameters causing variation in the final water footprint, as these are different from all other methods.

6.4 International Comparison

To put the water footprint of the 27 concrete batching plants into perspective, the average water footprint of 1 m³ concrete throughout New Zealand is compared to an international case study of the water footprint of a construction project in Sweden (Wärmark, 2015), and the water footprint of concrete produced in Sweden (Netz & Sundin, 2015) and European-made steel (Unger, Zhang, & Mathews, 2013).

For comparative purposes, the water footprint of New Zealand concrete has been converted to units of m³/kg, in order to compare the water footprint with international examples. The average density of concrete (2363 kg/m³) was derived as an average across all compressive strengths, according to data in Allied Concrete's environmental product declaration (EPD) (Allied Concrete, 2014). Both the water footprint of Netz and Sundin (2015) and Unger et al. (2013) are calculated only for the manufacturing stage, and as such are comparable with the New Zealand case study.

Table 6.1 shows that the average water footprint of New Zealand concrete is an order of magnitude smaller than a similar case study conducted in Sweden⁴⁹. Perhaps more significant is the comparison of the two concrete water footprints with that of the water footprint of steel made in the Netherlands, where steel can be seen to have a much higher blue water footprint.

⁴⁹ The comparison of the two water footprints should be taken as relative not absolute

Table 6.1 An international comparison of the volumetric blue water footprints of building and construction products

Product	Water Footprint (m³/kg)	Source	Notes
Swedish Concrete	0.00078	Netz and Sundin (2015)	Manufacturing stage only
NZ Concrete	0.000076	This case study	Manufacturing stage only
European Steel	0.00421	Unger et al. (2013)	Product blue water footprint

In contrast to Table 6.1, the results of Wärmark (2015) are not comparable to the case study of New Zealand concrete as consideration is given to a larger proportion of the lifecycle, including raw material extraction and transportation. However, the results are presented in Table 6.2 to give an indication of the relative difference in the water footprints of different industrial products throughout a greater proportion of the lifecycle. The water footprints calculated by Wärmark (2015), in accordance with the methods of Hoekstra et al. (2011) and Pfister et al. (2009) are compared in Table 6.2, in accordance with the research of Wärmark (2015). In agreement with the results presented in Table 6.1, it can be seen that the water footprint of concrete is much lower than the water footprint of steel.

Table 6.2 Water footprint results of a construction project in Sweden (Wärmark, 2015)

Product	Hoekstra et al. (2011) (m³ freshwater per kg)	Pfister et al. (2009) (m³ freshwater per kg)
Concrete	0.002	0.192
Galvanized Steel	0.019	6.062
Granite	0.006	2.302

A comparison of the calculated water footprints according to both Hoekstra et al. (2011) and Pfister et al. (2009) shows calculated water footprints to be two orders of magnitude larger when calculated according to Pfister et al. (2009).

6.5 Implications for the Concrete Industry in New Zealand

6.5.1 Decision Support

The volumetric water footprint of 1 m³ concrete presented in Figure 5.4 can be interpreted as one component used to measure freshwater efficiency amongst concrete batching plants. When displayed alongside the volume of concrete produced and the volume of freshwater consumed, it can be seen clearly in Figure 5.5 that plants which have a higher freshwater consumption to concrete production ratio, have the greatest volumetric water footprints. Therefore, if the goal of the assessment is to target batching plants in which efficiency practices could be employed, then Figure 5.4 is sufficient.

However, if the goal of the assessment is to understand which batching plants are consuming the largest volume of freshwater, then the volumetric water footprint results should be used alongside the consumption and production statistics in Figure 5.1 and Figure 5.2. For example, by simply looking at the results in Figure 5.4, the Masterton batching plant appears to have the highest water footprint. When assessed alongside the production statistics, it can be seen that whilst Masterton requires more freshwater to produce 1 m³ concrete, overall the plant produces a mere 2% of total concrete. In contrast, the Ashby's Ready Mix site produces 20% of all concrete. Clearly, whilst Masterton requires a greater freshwater input to produce a unit of concrete, in terms of total resource consumption, the Ashby's Ready Mix plant consumes substantially more freshwater per year in absolute terms. Furthermore, as discussed previously, the schematic diagrams of freshwater flows at each batching plant (refer Appendix K) are useful alongside the information in Figure 5.4, as they aid in the visualisation of freshwater flows between batching plants. In particular, withdrawals from the reticulated network vs. recycled freshwater from rainwater storage ponds, as these can have a greater impact depending on the local hydrological conditions.

An environmental impact assessment (refer to section 6.2) can assist in putting the volumetric water footprints in the context of the local hydrological conditions, which can be beneficial in selecting batching plants that could benefit from freshwater management initiatives.

Chapter 7

Limitations, Recommendations and Conclusions

7.1 Water Footprint Accounting (Inventory)

Whilst calculating the water footprint and associated environmental impacts of the 27 concrete batching plants throughout New Zealand, a number of limitations and recommendations were identified, as follows.

7.1.1 Obtaining Primary Data

Assessing the environmental impact of the volumetric water footprint of 1 m³ concrete using the method of Hoekstra et al (2011), as described in *The Water Footprint Assessment Manual: Setting the Global Standard (2011)*, requires a multitude of site specific data for all process steps included in the system boundary (refer to Figure 4.1). In this study, a number of required parameters (discussed in more detail in section 4.3) were estimated, as measured data were unavailable. Ideally, primary data would be used in the LCI phase; however, process-specific water consumption data for the entire life cycle are often unavailable. This is an area that could greatly benefit from the compilation of a local database similar in nature to the WaterGAP inventories used in larger scale studies. This would be beneficial for future LCA studies of New Zealand products and processes.

7.1.2 Type and Location of Water Use

It is acknowledged in the literature that the type and source of freshwater should be identified during water footprint studies (Pfister et al. 2009). This information was provided for each batching plant (refer Appendix K); however, with regards to the water footprint equation (Equation 1), there is no differentiation between recycled water and that sourced from the reticulated network. As previously discussed in the context of the Ashburton and Masterton batching plants (refer to section 6.1.2), the volumetric blue water footprint results, which are used in all environmental impact assessments, do not account for the source of blue water. In this case study, the blue water could be sourced from the reticulated water supply or on-site rainwater

harvesting. However, the opportunity costs of these two different sources of blue water could be different. The batching plants that operate solely on recycled rainwater (in this case, stored rainfall⁵⁰) do not contribute to over-allocation problems. Thus, the differentiation between water types is an important distinction that is neglected in the current methods.

7.2 Water Footprint Impact Assessment

In terms of calculating the local environmental impacts associated with the production of 1 m³ concrete across the 27 batching plants, limitations and areas of improvement were identified in section 4.3.2. There are a few fundamental issues, however, that were identified when adapting the methods to the finer resolution in New Zealand. These are discussed below.

7.2.1 Data Availability and Quality

At both the FMZ and the regional scale, various water quantity data were derived from the Land and Water Aotearoa (LAWA) domain. Whilst the LAWA database was a useful source for water quantity datasets, the standardisation of data provided by the numerous organisations is a potential limitation, and an area for improvement. For example, actual run-off values are calculated by respective regional councils and collated; these may therefore be calculated using different methods depending on the organisation. This being said, it became apparent when collating data at the catchment scale in New Zealand that there is no singular database which provides catchment level hydrological information. A lack of standardisation on the scale of reporting amongst regional councils proved limiting when attempting to collect data at the catchment scale; data were difficult to obtain and varied in quality.

Data derived from the LAWA database are not intended to be used for hydrological studies at this level. The primary function of this database is to make water quantity and quality data more readily available. In terms of future data collection, a database platform, similar in nature to LAWA, but containing water quantity data required for

⁵⁰ As per the definition provided by Hoekstra et al. (2011) this type of rainwater storage can be considered blue water as opposed to green water, and therefore should be considered in the blue water accounting phase.

water footprint and LCA studies would be a hugely beneficial tool for conducting future water footprinting studies at a finer resolution in New Zealand.

7.2.2 Temporal Resolution

In terms of the environmental impacts of water withdrawal/consumption, seasonal variations in temperatures and precipitation play a vital role in determining water stress. Use of annual scale data can result in the magnitude of seasonal changes to water availability being overlooked. Ideally, data used to calculate the environmental impact would be at the monthly scale. Unfortunately, this level of detail was unavailable for many of the critical datasets and so analysis was conducted at the annual scale. It is recommended for future water footprinting studies that data at the monthly scale be utilised, in order to show the full magnitude of seasonal variability in freshwater throughout the regions of New Zealand.

7.3 Method Specific Limitations

Methodological limitations (and proposed alternatives) that arose when applying the four water footprint methods of Hoekstra et al. (2011), Pfister et al. (2009), Berger et al. (2014), and Boulay et al. (2016) to the case study of New Zealand concrete, were discussed in Chapter 4. Some key limitations of the chosen methods are discussed below.

7.3.1 Estimation of the Environmental Flow Requirement

Environmental flow requirements (*EFRs*) are considered only by Hoekstra et al. (2011) and Boulay et al. (2016). Hoekstra et al. (2011) recommend a blanket value of 80% of natural flow, whilst Boulay et al. (2016) recommend an *EFR* in the range of 30% to 60% for pristine flows. In New Zealand, *EFRs* as calculated by Boulay et al. (2016) were found to range from 33% in the Canterbury and West Coast regions to 37% in the Wellington region (Boulay et al. 2016). A New Zealand case study conducted by Herath et al. (2013) utilised an *EFR* of 30% when calculating the water footprint of hydro-electricity.

Limitations arise from using an estimated value for *EFR* at the annual scale, as freshwater availability alters seasonally. Using the percentage of natural flow averaged

over long time periods neglects this seasonal variation. Boulay et al. (2016) identified that the currently applied proportions of natural flow do not account for the local diversity of the aquatic environment. It is suggested that substitution of these values with the minimum low flow requirements would provide a more robust recommendation in terms of the *EFR*, and would be based on the aquatic environment at ground level, as described in Boulay et al. (2016).

7.3.2 Temporal Resolution

With regards to the method proposed by Berger et al. (2014), a limitation arose from the temporal resolution of the data. Evaporation recycling and synthetically created vapour recycling are calculated at the annual scale. This does not accurately reflect climatic changes influencing water recycling and availabilities. Monthly data would give a better representation. The WAVE model does not provide monthly or weighted annual *BIER* and *WDI* factors but acknowledges this as an area of further research.

Similarly, the method of Boulay et al. (2016) is intended to be performed using data at the monthly scale. However, for the water footprint of concrete calculated in this thesis, data at the monthly scale were unavailable and thus annual scale data were used throughout. It is recognised that monthly data would be much more robust and allow for a more accurate representation of water scarcity throughout the year. Future research that provides hydrological data at the monthly timescale would be hugely beneficial to application of both the Berger et al. (2014) and Boulay et al. (2016) methods at the finer resolution in New Zealand.

7.4 Decision Support

7.4.1 Spatial Recommendation

As can be seen in Figure 5.49, the coefficient of variation (CV) calculated for each method and scale shows some variation in the calculated CFs as a function of scale. On average, across all four methods, the catchment scale has the lowest CV, followed by the FMZ and regional scale. In terms of communicability, it is recommended that the catchment scale be utilised over the FMZ scale, especially if the goal of the assessment is international comparison, as the catchment scale is more readily

comparable with international case studies in which watershed CFs and impact assessments are often calculated. Further, the FMZ scale may, in some instances, include only small areas of a larger catchment. Whilst this can assist in targeting specific management practices to portions of the catchment deemed valuable for recreation, for example, when this scale is used for impact assessment it is not representative of the hydrological processes occurring in the wider catchment.

The regional scale is best suited if the purposes of the case study are to compare the wider regions of New Zealand and should, in this instance be utilised over the catchment scale analysis.

7.4.2 Methodological Recommendation

The methods of Hoekstra et al. (2011), Pfister et al. (2009), Berger et al. (2014) and Boulay et al. (2016) were all evaluated and calculated at three distinct spatial scales to determine how suited these methods were to adaptation at the finer resolution in New Zealand. This section summarises the findings of the water footprint and impact assessment case study of concrete batching plants throughout New Zealand, and makes recommendations for future water footprint and impact assessment studies in New Zealand.

Local Availability of the Datasets Required

For each aspect of the CFs, local data were sourced where possible. During the data collection process it became apparent that for some of the methods, the data required were simply not available either at an appropriate scale or quality for the method to be adapted at the finer resolution in New Zealand. This was the case for the WS_{blue} characterisation factor, as calculated by Hoekstra et al. (2011). A fundamental calculation required to calculate the WS_{blue} was the calculation of the blue water footprint of all processes within a distinct boundary (Equation 2). Due to a distinct lack of data availability concerning blue water consumption across New Zealand, this was unfeasible and an alternative calculation was utilised (refer to Equation 25).

Global Data Sets and Normalisation

Due to the global nature of the WaterGAP database used to determine both available freshwater and human consumption, local results normalised using the global *AMD* may be inappropriate. At present, analytical uncertainty propagation is not possible due to the discrete steps in the function. Boulay et al. (2016), note however, that stochastic uncertainty assessment is to be conducted in the future.

A proposed alternative to normalisation with the global *AMD*, would be to normalise the local *AMD* (catchment for example) by a nationally calculated *AMD*, calculated using locally measured or modelled datasets. This would allow catchment specific *AMD*, for example, to be normalised with New Zealand specific datasets. This could be used to better represent the diverse hydrological conditions within New Zealand, without considering countries with starkly different hydrological conditions.

Environmental Flow Requirement

A weakness in both the *WDI* and also the *WSI* characterisation factors is the lack of consideration of the environmental flow requirement. The *EFR* is not considered in either method but is an important freshwater management consideration in New Zealand, as protection of the country's natural resources (including freshwater and ecosystem health) is a paramount component of the Resource Management Act (1991). Thus the adaptability of the *WDI* or *WSI* characterisation factor to the finer resolution in New Zealand would neglect an important aspect of current freshwater management in New Zealand.

The *AMD* CF is similar to that of the WS_{blue} in that it accounts for a proportion of flow necessary to maintain environmental stability. The environmental flow requirement (*EFR*) according to Hoekstra et al. (2011) is calculated at 80% of natural flow (R_{nat}). The method according to the Boulay et al. (2016) working group is perhaps more scientifically robust as it has calculated at 0.5 by 0.5 grid cells, using monthly natural flow data from the WaterGAP global database, thus accounting for both spatial and temporal variability. Moreover, a case study of five major habitat types was conducted by Boulay et al. (2016), in order to validate the monthly model created by Pastor et al. (2013). The *EFR* for the six selected boundaries, varied from

33% in the Canterbury and West Coast regions to 37% in the Wellington region. This equates to a much lower proportion of the natural flow required to sustain appropriate environmental flows than the blanket 80% proposed by Hoekstra et al. (2011). A potential option for improving the *EFR* as proposed in Hoekstra et al. (2011) would be to substitute using 80% of the natural flow, for *EFR* values calculated according to Boulay et al. (2016) in line with the method proposed by (Pastor et al., 2013).

However, neither of these *EFRs* account for local ecosystem conditions, which might cause variation in the *EFR* requirements both spatially and temporally. This is due to the global nature of the WaterGAP dataset in which natural flow data are acquired in these global scale assessments. A more accurate representation of *EFR* might stem from using local metrics such as permissible low flows. In New Zealand, these are set by the relevant territorial authorities under obligation from the Resource Management Act (1991), with the aim to ensure the economic use of water is maintained whilst also maintaining aquatic ecosystems and their natural character (Wellington Regional Council, 2014).

ISO 14046 Compliance

A valuable consideration when selecting a method to calculate the water footprint and associated impacts of products and processes is whether the method is compliant with the recent *ISO 14046* standard (refer to section 2.2.5). A water footprint assessment according to this standard is based on an LCA approach, and the term 'water footprint' may only be applied when referring to results after impact assessment (ISO, 2014). A non-impact assessment-based analysis can be calculated in accordance with the new *ISO 14046* standard but must be regarded as a water footprint inventory study and must not be reported as a water footprint study (ISO, 2014). Thus, in its original form, without the inclusion of the *WFII*, an analysis conducted using this method of Hoekstra et al. (2011), cannot be considered a water footprint assessment.

A water footprint must also be comprehensive (i.e. all significant environmental impact parameters relating to water are addressed). If this is not demonstrated, the results of a non-comprehensive water footprint analysis must include a qualifier, and be reported as such (e.g. water scarcity footprint, water availability footprint). In the

case of the Hoekstra et al. (2011), only if the grey water footprint is considered in the impact assessment can it be seen as comprehensive according to *ISO 14046*. This is because the grey water component assesses water quality aspects (it should be noted, however, that this is not as comprehensive as methods presented by the LCA community e.g. the method proposed by Boulay et al. (2011a)). If the grey water component is not included, then a water footprint impact assessment calculated in accordance with the *WFII* will never be able to achieve a 'comprehensive' status in accordance with *ISO 14046*.

The *WSI* (Pfister et al., 2009), *WDI* (Berger et al., 2014) and *AMD* (Boulay et al., 2016) methods take an LCA approach using impact assessment. Thus results can be considered as either a water footprint or as a water footprint with a qualifier (depending on comprehensiveness) (ISO, 2014).

In terms of compliance, the method of Boulay et al. (2016) was developed with the *ISO 14046* in mind, and is fully compliant with the requirements of the new standard.

7.4.3 Summary

Table 7.1 provides a brief overview of important considerations to account for when adapting an impact assessment method to the New Zealand context and whether the four methods evaluated meet these criteria.

Table 7.1 A summary of components considered in each of the evaluated water footprint methods.

	<i>ISO 14046</i> Compliant	Comprehensive (<i>ISO 14046</i>)	Adaptable for NZ?	EFR Considered?
Hoekstra et al. (2011)	✓*	✓**	✓***	✓****
Pfister et al. (2009)	✓	✓	✓	✗
Berger et al. (2014)	✓	✓	✓	✗
Boulay et al. (2016)	✓	✓	✓*****	✓

* Compliant as long as an impact assessment using the *WFII* is calculated

**Comprehensive as long as it includes a grey water impact assessment (otherwise there is no other metric for water quality impacts to human health, as with LCA methods).

*** Calculations for the water footprint of a specified boundary had to be adapted due to data unavailability.

**** Using a blanket value of 80% of natural flow (but could be adapted to use more site specific *EFR*)

***** Difficulty in adaptation may arise from the normalisation with a global average *AMD*

It is based on the aforementioned strengths and weakness of the four methods assessed that the methods of Boulay et al. (2016) and Hoekstra et al. (2011) are recommended for adaptation at the finer resolution in New Zealand. The impact assessment proposed by Boulay et al. (2016) is *ISO 14046* compliant and comprehensively considers the environmental flow requirement (*EFR*) (as opposed to the methods of both Pfister et al. (2009) and Berger et al. (2014) who do not consider *EFR*, and Hoekstra et al. (2011) who consider *EFR* as 80% of natural flow regardless of seasonal changes in water availability). Furthermore, the *AMD* characterisation factor is based on an availability-minus-demand ratio, which is considered more comprehensive with respect to the actual appropriation of the freshwater resource, when compared to the with the withdrawal-to-availability ratio of Pfister et al. (2009), for example.

As can be seen in Table 7.1, of all the methods considered both Boulay et al. (2016) and Hoekstra et al. (2011) are compliant with the *ISO 14046* and can achieve comprehensiveness in accordance with this standard. In terms of the feasibility of adapting the two methods at the finer resolution in New Zealand, the method proposed by Hoekstra et al. (2011) had a few shortcomings. Firstly, a key component of the impact assessment 'the blue water footprint of the specified boundary' (refer to

Equation 2), could not be calculated due to the sheer amount of data required, compounding by the lack of data availability in this area in New Zealand. Thus an alternative calculation was used (refer to Equation 25). Furthermore, the *EFR* according to Hoekstra et al. (2011) is 80% of natural flow, regardless of seasonality. This does not represent the sensitivity of the freshwater ecosystem throughout the year. It's feasible, however, to adapt the method of Hoekstra et al. (2011) to include site-specific *EFR* values. Future research could calculate site specific *EFR* for catchments in New Zealand, which would be beneficial for the methods of both Boulay et al. (2016) and Hoekstra et al. (2011)

The method proposed by Boulay et al. (2016) was found to be more readily adaptable to the finer resolution in New Zealand than for the other methods. Further, more comprehensive consideration is given to the *EFR* within selected boundaries, and this can be calculated seasonally. The main limitation when applying the method of Boulay et al. (2016) to the finer resolution is the proposed normalisation with the global average. As discussed prior, the potential to normalise the locally calculated *AMD* with a national average *AMD* could be explored in order to further adapt this method at the finer resolution in New Zealand, and elsewhere.

7.4.4 Conclusions

In summary, the water footprint and associated environmental impact of 27 concrete batching plants throughout New Zealand were calculated at three distinct spatial scales using the methods of Hoekstra et al. (2011); Pfister et al. (2009); Berger et al. (2014) and Boulay et al. (2016). The volumetric water footprint of each concrete batching plant was found to vary depending on the ratio of freshwater consumption to concrete production. Batching plants in which the volume of freshwater consumption exceeded concrete production (per m³) were found to have water footprints closer to or greater than the average across all plants, when compared with plants of a similar size that produced more concrete per m³ of freshwater.

Across New Zealand, the Ashburton batching plant was deemed to have the greatest environmental impact (ranked highest 9 out of 12 occurrences and second highest for the remaining three occurrences) for the methods of Hoekstra et al. (2011), Pfister et al.

(2009) and Berger et al. (2014). In contrast, Boulay et al. (2016) calculated the Palmerston North batching plant as having the greatest environmental impact across all spatial scales (4 out of 12 occurrences). Batching plants with the lowest environmental impacts were found to be Greymouth (ranked highest 8 out of 12 occurrences and second highest for the remaining four occurrences) and Wanganui (6 out of 12 occurrences).

Based on the aforementioned case study, the methods of Hoekstra et al. (2011) and Boulay et al. (2016) are recommended for further exploration in New Zealand case studies. Both methods were found to be adaptable to the finer resolution in New Zealand.

Works Cited

- Alcamo, J., DÖLL, P., Henrichs, T., Kaspar, F., Lehner, B., RÖSch, T., & Siebert, S. (2003). Development and testing of the WaterGAP 2 global model of water use and availability. *Hydrological Sciences Journal*, 48(3), 317-337.
doi:10.1623/hysj.48.3.317.45290
- Alcamo, J., Henrichs, T., & Rösch, T. (2000). *World water in 2025: Global modeling and scenario analysis for the world commission on water for the 21st century*: Center for Environmental Systems Research, University of Kassel.
- Aldaya, M., Allan, J., & Hoekstra, A. (2010). Strategic importance of green water in international crop trade. *Ecological Economics*, 69(4), 887-894.
- Allan, J. A. (1998). Virtual water: a strategic resource: global solutions to regional deficits. *Ground Water (USA)*, 36(4), 545-546.
- Allied Concrete. (2014). *Environmental product declaration: Ready mixed concrete using Holcim manufactured cement*. Retrieved from Invercargill, New Zealand:
- Baker, N., Blackett, P., van der belt, M., Carbines, M., Clothier, B., Bradford-Grieve, J., . . . Young, R. (2011). *Ecosystem services: Emerging issues*. Retrieved from
- Bare, C. J., Hofstetter, P., Pennington, W. D., & Udo de Haes, A. H. (2000). Mid-points versus End-points: The sacrifices and benefits. *International Journal for Life Cycle Assessment*, 5(6), 319-326.
- Bayart, J.-B., Bulle, C., Deschênes, L., Margni, M., Pfister, S., Vince, F., & Koehler, A. (2010). A framework for assessing off-stream freshwater use in LCA. *The International Journal of Life Cycle Assessment*, 15(5), 439-453.
- Bayart, J.-B., & Ekambi, E. (2016). *WULCA- AWARE Case study: Application of AWARE to bottled water and beverages*. Quantis.
- Berger, M., & Finkbeiner, M. (2012). Methodological Challenges in Volumetric and Impact-Oriented Water Footprints. *Journal of Industrial Ecology*, 17(1), 79-89.
- Berger, M., van der Ent, R., Eisner, S., Bach, V., & Finkbeiner, M. (2014). Water accounting and vulnerability evaluation (WAVE): Considering atmospheric evaporation recycling and the risk of freshwater depletion in water footprinting. *Environmental science & technology*, 48(8), 4521-4528.

- Boulay, A.-M., Bare, J., Benini, L., Berger, M., Lathuilliere, M., Manzardo, A., . . . Pfister, S. (2016). The WULCA consensus characterisation model for water scarcity footprints: Assessing impacts of water consumption based on Available Water REmaining (AWARE). *Environmental Science and Technology*.
- Boulay, A.-M., Bouchard, C., Bulle, C., Deschenes, L., & Margni, M. (2011a). Categorizing water for LCA inventory. *The International Journal of Life Cycle Assessment*(7), 639-651. doi:10.1007/s11367-011-0300-z) contains supplementary material, which is available to authorized users.
- Boulay, A.-M., Bouchard, C., Bulle, C., Deschenes, L., & Margni, M. (2011b). Regional characterization of freshwater use in LCA: modeling direct impacts on human health. *Environmental science & technology*, 45(20), 8948-8957.
- Boulay, A.-M., Hoekstra, A. Y., & Vionnet, S. (2013). Complementarities of water-focused life cycle assessment and water footprint assessment. *Environmental science & technology*, 47(21), 11926-11927.
- Bruinsma, J. (2009). *The resource outlook to 2050: by how much do land, water, and crop yields need to increase by 2050? FAO Expert Meeting on 'How to feed the world in 2050'*. Retrieved from Rome:
- Caldiera, C., Quinteiro, P., Castanheira, E., Dias, C. A., Arroja, L., & Freire, F. (n.d.). *Comparative water footprint profile of vegetable oils for biofuel production*. University of Coimbra; University of Aveiro.
- Carr, M. J., & Rickwood, C. J. (2008). *Water quality: Development of an index to assess country performance*. Retrieved from Gatineau, Canada:
- CEO Water Mandate. (2011). *Water footprint and corporate water accounting for resource efficiency*. Retrieved from New York, USA:
- Chapagain, A. K., & Hoekstra, A. Y. (2004). *Water footprints of nations: Volume 1: Main report*. Retrieved from Delft, Netherlands:
- Chapagain, A. K., Hoekstra, A. Y., Savenije, H. H. G., & Gautam, R. (2006). The water footprint of cotton consumption: An assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. *Ecological Economics*, 60, 186-203. doi:10.1016/j.ecolecon.2005.11.027

- Comprehensive Assessment of Water Management in Agriculture. (2007). *Water for food, water for life: A comprehensive assessment of water management in agriculture*. London; Colombo: Earthscan; International Water Management Institute.
- Deurer, M., Green, S., Clothier, B., & Mowat, A. (2011). Can product water footprints indicate the hydrological impact of primary production?—A case study of New Zealand kiwifruit. *Journal of hydrology*, 408(3), 246-256.
- Dowdell, D. (2013). *Application of environmental profiling to Whole Building Whole of Life Assessment – A plan for New Zealand; BRANZ Study Report (SR 276), Judgeford*. Retrieved from
- Emara, Y., & Berger, M. (2016a). *Water scarcity footprint of a Volkswagen car: Case study for the WULCA consensus midpoint impact assessment model "Available Water Remaining" AWARE*. Technische universität berlin Berlin, Germany.
- Emara, Y., & Berger, M. (2016b). *Water scarcity footprint of a water flow regulator (NEOPERL): Case study for the WULCA consensus midpoint impact assessment model "Available Water Remaining" AWARE*. Technische universität berlin Berlin, Germany.
- Evans, C. (1999). Effects of irrigation on groundwater recharge between the Rakaia and Selwyn Rivers, Canterbury, New Zealand. *Unpublished MSc Thesis, University of Canterbury*.
- Falkenmark, M. (1986). Fresh water: Time for a modified approach. *Ambio*, 15, 192-200.
- Falkenmark, M., & Lindh, G. (1974). How Can We Cope with the Water Resources Situation by the Year 2015? *Ambio*, 3(3/4), 114-122. doi:10.2307/4312063
- Flörke, M., Kynast, E., Bärlund, I., Eisner, S., Wimmer, F., & Alcamo, J. (2013). Domestic and industrial water uses of the past 60 years as a mirror of socio-economic development: A global simulation study. *Global Environmental Change*, 23(1), 144-156.
- Frischknecht, R., Steiner, R., & Jungbluth, N. (2009). The Ecological Scarcity Method—Eco-Factors 2006—A method for impact assessment in LCA (Methode der ökologischen Knappheit—Ökofaktoren 2006—Methode für die Wirkungsabschätzung in Ökobilanzen).

- Gassert, F., Reig, P., Luo, T., & Maddocks, A. (2013). *Aqueduct country and river basin rankings: A weighted aggregation of spatially distinct hydrological indicators. Working Paper*. Retrieved from Washington, DC:
- Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A. D., Struijs, J., & Zelm, R. v. (2013). *ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level- Report I: Characterisation*. Retrieved from
- Goedkoop, M., & Spriensma, R. (2001). The eco-indicator99: A damage oriented method for life cycle impact assessment: Methodology report.
- Heiler, T. (2015). 'Irrigation and drainage', Te Ara - the Encyclopedia of New Zealand. Retrieved from <http://www.TeAra.govt.nz/en/irrigation-and-drainage>
- Herath, I., Clothier, B., Horne, D., & Singh, R. (2011a). Assessing freshwater scarcity in New Zealand. *New Zealand Life Cycle Management Centre Working paper Series 02/11*, 21-26.
- Herath, I., Deurer, M., Horne, D., Singh, R., & Clothier, B. (2011b). The water footprint of hydroelectricity: a methodological comparison from a case study in New Zealand. *Journal of Cleaner Production*, 19(14), 1582-1589. doi:<http://dx.doi.org/10.1016/j.jclepro.2011.05.007>
- Herath, I., Green, S., Horne, D., Singh, R., & Clothier, B. (2014a). Quantifying and reducing the water footprint of rain-fed potato production, part I: measuring the net use of blue and green water. *Journal of Cleaner Production*, 81(0), 111-119. doi:<http://dx.doi.org/10.1016/j.jclepro.2014.06.026>
- Herath, I., Green, S., Horne, D., Singh, R., & Clothier, B. (2014b). Quantifying and reducing the water footprint of rain-fed potato production part II: a hydrological assessment using modelling supported by measurements. *Journal of Cleaner Production*, 81(0), 103-110. doi:[10.1016/j.jclepro.2014.06.025](http://dx.doi.org/10.1016/j.jclepro.2014.06.025)
- Herath, I., Green, S., Horne, D., Singh, R., McLaren, S., & Clothier, B. (2013a). Water footprinting of agricultural products: evaluation of different protocols using a case study of New Zealand wine. *Journal of Cleaner Production*, 44(0), 159-167. doi:<http://dx.doi.org/10.1016/j.jclepro.2013.01.008>
- Herath, I., Green, S., Singh, R., Horne, D., van der Zijpp, S., & Clothier, B. (2013b). Water footprinting of agricultural products: a hydrological assessment for the water footprint of New Zealand's wines. *Journal of Cleaner Production*, 41(0), 232-243. doi:<http://dx.doi.org/10.1016/j.jclepro.2012.10.024>

- Hoekstra, A. Y. (2016). A critique on the water-scarcity weighted water footprint in LCA. *Ecological Indicators*, 66, 564-573.
doi:<http://dx.doi.org/10.1016/j.ecolind.2016.02.026>
- Hoekstra, A. Y., & Chapagain, K. A. (2008). *Globalisation of water: Sharing the planet's freshwater resource*. Oxford, UK: Blackwell Publishing.
- Hoekstra, A. Y., Chapagain, K. A., Aldaya, M. M., & Mekonnen, M. M. (2009). *Water Footprint Manual: State of the Art 2009*. Enschede, Netherlands: Water Footprint Network.
- Hoekstra, A. Y., Chapagain, K. A., & Mekonnen, M. M. (2011). *The water footprint assessment manual: Setting the global standard*. London, UK: Earthscan.
- Hoekstra, A. Y., Gerbens-Leenes, W., & Van der Meer, T. H. (2009). Reply to Pfister and Hellweg: Water footprint accounting, impact assessment and life-cycle assessment. *Proceedings of the National Academy of Sciences*, 106(40), 114.
- Hoekstra, A. Y., & Hung, P. (2002). Virtual water trade: A quantification of virtual water flows between nations in relation to international crop trade. *Value of Water Research Report Series*, 11, 27-29.
- Hoekstra, A. Y., & Mekonnen, M. M. (2012a). Reply to Ridoutt and Huang: From water footprint assessment to policy. *Proc Natl Acad Sci U S A*, 109(22), E1425-E1425. doi:10.1073/pnas.1205186109
- Hoekstra, A. Y., & Mekonnen, M. M. (2012b). The water footprint of humanity. *Proc Natl Acad Sci U S A*, 109(9), 3232-3237. doi:10.1073/pnas.1109936109
- Hoekstra, A. Y., Moinar, J. L., Jennings, M., Revenga, C., Spalding, M. D., Boucher, T. M., . . . Ellison, K. (2010). *The Atlas of Global Conservation; Changes, Challenges, and Opportunities to Make a Difference*. Berkeley: University of California Press.
- International Standardization Organization. (2000). Environmental management -- Life cycle assessment -- Life cycle impact assessment. ISO 14042.
- International Standardization Organization. (2006). Environmental Management- Life Cycle Assessment- Principles and Framework. ISO 14040. Geneva, Switzerland: ISO.
- International Standardization Organization. (2014). Environmental management -- Water footprint -- Principles, requirements and guidelines. ISO 14046.

- Jaques, R. (2001). *Environmental impacts associated with New Zealand concrete manufacture - Preliminary study*. Retrieved from Wellington, New Zealand:
- Jaques, R., McLaren, J., & Nebel, B. (2011). *Environmental profiling of New Zealand building material products: Where to for the New Zealand building sector?- Discussion Paper*. Retrieved from
- Kounina, A., Margni, M., Bayart, J.-B., Boulay, A.-M., Berger, M., Bulle, C., . . . Humbert, S. (2013). Review of methods addressing freshwater use in life cycle inventory and impact assessment. *The International Journal of Life Cycle Assessment*(3), 707. doi:10.1007/s11367-012-0519-3) contains supplementary material, which is available to authorized users.
- Land Air Water Aotearoa. (2016). Canterbury Region. Retrieved from <http://www.lawa.org.nz/explore-data/canterbury-region/water-quantity/surface-water-zones/ashburton-surface-water/>
- Landcare Research. (2011). *Assessment of the water footprint of fresh kiwi fruit - methods and scoping: Final report*. Retrieved from Wellington, New Zealand:
- McIndoe, I. (2002). *Efficient and reasonable use of water for irrigation*. Retrieved from
- McIndoe, I., & Carran, P. (1998). Testing of best management guidelines. *Lincoln Environmental MAF Technical Paper 00/06*.
- Milà i Canals, L., Chenoweth, J., Chapagain, A., Orr, S., Antón, A., & Clift, R. (2009). Assessing freshwater use impacts in LCA: Part I—inventory modelling and characterisation factors for the main impact pathways. *The International Journal of Life Cycle Assessment*, 14(1), 28-42. doi:10.1007/s11367-008-0030-z
- Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being* (Vol. 5): Island press Washington, DC.
- Ministry for the Environment. (2006). *Snapshot of Water Allocation in New Zealand*. Retrieved from Wellington: New Zealand:
- Ministry for the Environment. (2007). *Environment New Zealand 2007*. Retrieved from Wellington, New Zealand:
- Ministry for the Environment. (2010). *Update of water allocation data and estimate of actual water use of consented takes 2009-10*. Retrieved from Wellington, New Zealand:

- Ministry for the Environment. (2014a). Fresh Water. Retrieved from <http://mfe.govt.nz/issues/water/>
- Ministry for the Environment. (2014b). *National Policy Statement for Freshwater Management*. Retrieved from Wellington, New Zealand:
- Ministry for the Environment, & Statistics New Zealand. (2015). *New Zealand's Environmental Reporting Series: Environment Aotearoa 2015*. Retrieved from www.mfe.govt.nz
- Mittermeier, R. A., Gil, P. R., Hoffman, M., Pilgrim, J., Brooks, T., & Mittermeier, C. G. (2004). *Hotspots revisited: Earth's biologically richest and most endangered terrestrial ecoregions*. Mexico City, Mexico: Conservation International.
- Moot, D., Mills, A., Lucas, D., & Warwick, S. (2009). *Country Pasture/Forage Resource Profiles: New Zealand*. Retrieved from <http://www.fao.org/ag/agp/AGPC/doc/Counprof/newzealand/newzealand2.htm>
- Müller Schmied, H., Eisner, S., Franz, D., Wattenbach, M., Portmann, F. T., Flörke, M., & Döll, P. (2014). Sensitivity of simulated global-scale freshwater fluxes and storages to input data, hydrological model structure, human water use and calibration. *Hydrol. Earth Syst. Sci.*, 18(9), 3511-3538. doi:10.5194/hess-18-3511-2014
- National Institute of Water and Atmospheric Research. (2001). Overview of New Zealand climate. *NIWA*. <https://www.niwa.co.nz> Retrieved from <https://www.niwa.co.nz>
- National Institute of Water and Atmospheric Research. (2010). *How clean are our rivers?* Retrieved from <https://www.niwa.co.nz/publications/wa/water-atmosphere-1-july-2010/how-clean-are-our-rivers>
- Nebel, B. (2006). *White Paper - Life cycle assessment and the building and construction industry*. Retrieved from
- Nebel, B. (n.d). *Is it valid to use international data in New Zealand LCA studies?* Retrieved from Rotorua, New Zealand:
- Netz, J., & Sundin, J. (2015). *Water Footprint of Concrete*. (Environmental Strategies, Second Cycle), Royal Institute of Technology.

- New, M., Todd, M., Hulme, M., & Jones, P. (2001). Precipitation measurements and trends in the twentieth century. *International Journal of Climatology*, 21(15), 1889-1922. doi:10.1002/joc.680
- New Zealand Business Council for Sustainable Development. (2008). *A best use solution for New Zealand's water problem*. Retrieved from Auckland:
- New Zealand Institute of Economic Research. (2008). *Cement and Concrete Product: Economic Impact Assessment*. Retrieved from
- New Zealand Institute of Economic Research. (2014). *Water management in New Zealand: A road map for understanding water value*. Retrieved from Wellington, New Zealand:
- Nilsson, C., Reidy, C. A., Dynesius, M., & Revenga, C. (2005). Fragmentation and flow regulation of the world's large river systems. *Science*, 308(5720), 405-408.
- Ortiz, O., Castells, F., & Sonnemann, G. (2009). Sustainability in the construction industry: A review of recent developments based on LCA. *Construction and Building Materials*, 23(1), 28-39.
- Owens, J. W. (2001). Water Resources in Life-Cycle Impact Assessment: Considerations in Choosing Category Indicators. *Journal of Industrial Ecology*, 5(2), 37-54. doi:10.1162/10881980152830123
- Parliamentary Commissioner for the Environment. (2015). *Update Report- Water quality in New Zealand: Land use and nutrient pollution*. Retrieved from Wellington: New Zealand:
- Pastor, A., Ludwig, F., Biemans, H., Hoff, H., & Kabat, P. (2013). Accounting for environmental flow requirements in global water assessments. *Hydrology and Earth System Sciences Discussions*, 10, 14987-15032.
- Pfister, S., & Bayer, P. (2014). Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. *Journal of Cleaner Production*, 73, 52-62. doi:http://dx.doi.org/10.1016/j.jclepro.2013.11.031
- Pfister, S., & Hellweg, S. (2009). The water “shoesize” vs. footprint of bioenergy. *Proceedings of the National Academy of Sciences*, 106(35), E93-E94. doi:10.1073/pnas.0908069106

- Pfister, S., Koehler, A., & Hellweg, S. (2009). Assessing the environmental impacts of freshwater consumption in LCA. *Environmental Science and Technology*, 43(11), 4098-4104.
- Pfister, S., Saner, D., & Koehler, A. (2011). The environmental relevance of freshwater consumption in global power production. *The International Journal of Life Cycle Assessment*, 16(6), 580-591.
- Pilkey, O. H., & Pilkey-Jarvis, L. (2007). *Useless arithmetic: why environmental scientists can't predict the future*: Columbia University Press.
- Postel, S. L., Daily, G. C., & Ehrlich, P. R. (1996). Human appropriation of renewable fresh water. *Science-AAAS-Weekly Paper Edition*, 271(5250), 785-787.
- PricewaterhouseCoopers. (2011). *Valuing the role of construction in the New Zealand economy: A report to the Construction Strategy Group*. Retrieved from Porirua, Auckland:
- Prüss-Ustun, A., Bartram, A., Clasen, T., Colford Jr, J. M., Cumming, O., Curties, V., . . . Wolf, J. (2014). Burden of disease from inadequate water, sanitation and hygiene in low- and middle-income settings: a retrospective analysis of data from 145 countries. *Tropical Medicine and International Health*, 19(8), 894-905. Retrieved from <http://www.ncbi.nlm.nih.gov/pmc/articles/PMC4255749/pdf/tmi0019-0894.pdf>
- Raskin, P., Gleick, P., Kirshen, P., Pontius, G., & Strzepek, K. (1997). *Water futures: assessment of long-range patterns and problems. Comprehensive assessment of the freshwater resources of the world*: SEI.
- Richter, B. D., Davis, M., Apse, C., & Konrad, C. (2012). A presumptive standard for environmental flow protection. *River Research and Applications*, 28(8), 1312-1321.
- Richts, A., Struckmeier, W. F., & Zaepke, M. (2011). WHYMAP and the groundwater resources map of the world 1: 25,000,000 *Sustaining Groundwater Resources* (pp. 159-173): Springer.
- Ridoutt, B. G., & Huang, J. (2012). Environmental relevance—the key to understanding water footprints. *Proceedings of the National Academy of Sciences*, 109(22), E1424-E1424.

- Ridoutt, B. G., & Pfister, S. (2010). A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. *Global Environmental Change*, 20(1), 113-120. doi:<http://dx.doi.org/10.1016/j.gloenvcha.2009.08.003>
- Ridoutt, B. G., & Pfister, S. (2013). A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. *The International Journal of Life Cycle Assessment*, 18(1), 204-207.
- Ridoutt, B. G., Sanguansri, P., Nolan, M., & Marks, N. (2012). Meat consumption and water scarcity: beware of generalizations. *Journal of Cleaner Production*, 28(0), 127-133. doi:<http://dx.doi.org/10.1016/j.jclepro.2011.10.027>
- Rockpoint. (2002). *Irrigation in New Zealand*. Retrieved from
- Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, S. F. I., Lambin, E., . . . Falkenmark, M. (2009). Planetary Boundaries: Exploring the safe operating space for humanity. *Ecology and Society*, 14(2).
- Schulte, P. (2014). Defining Water Scarcity, Water Stress, and Water Risk: It's Not Just Semantics. Retrieved from <http://pacinst.org/water-definitions/>
- Shiklomanov, I. A. (2000). Appraisal and Assessment of World Water Resources. *Water International*, 25(1), 11-32. doi:10.1080/02508060008686794
- Shiklomanov, I. A., & Rodda, J. C. (2003). *World water resources at the beginning of the 21st century*. Cambridge, UK: UNESCO and Cambridge University Press.
- Smakhtin, V., Revenga, C., & Döll, P. (2004a). A pilot global assessment of environmental water requirements and scarcity. *Water International*, 29(3), 307-317.
- Smakhtin, V., Revenga, C., & Döll, P. (2004b). *Taking into account environmental water requirements in the global-scale water resource assessment*. Retrieved from Colombo, Sri Lanka:
- Standards New Zealand. (2013). NZS3104: Specification for Concrete Production (pp. 43): Standards New Zealand.
- Statistics New Zealand. (2010). Water physical stock account 1995-2005. Retrieved from http://www.stats.govt.nz/browse_for_stats/environment/environmental-economic-accounts/water-physical-stock-account-1995-2005.aspx

- Statistics New Zealand, & Australian Bureau of Statistics. (2006). Australian and New Zealand Standard Industrial Classification (ANZSIC).
- Stronge, J. (2001). *Field efficiency of border strip irrigation in Canterbury, New Zealand*. Lincoln University.
- Tait, A., Henderson, R., Turner, R., & Zheng, X. (2006). Thin plate smoothing spline interpolation of daily rainfall for new zealand using a climatological rainfall surface. *International Journal of Climatology*, 26(14), 2097-2115.
- Unger, K., Zhang, G., & Mathews, R. (2013). *Water Footprint Assessment - Tata Chemicals, Tata Motors, Tata Power, Tata Steel: Results and Learning*. Retrieved from waterfootprint.org
- United Nations. (2013). *World populations prospects: The 2012 revision, Volume 1: Comprehensive Tables*. Retrieved from New York, USA:
- United Nations Environment Programme. (2012). *The business case for the green economy- Sustainable return on investment*. Retrieved from
- United Nations Water. (2007). *Coping with water scarcity: challenge of the twenty first century*. Retrieved from
- United States Environmental Protection Agency. (2013). Water: Total Maximum Daily Loads (303d). Retrieved from <http://water.epa.gov/lawsregs/lawsguidance/cwa/tmdl/overviewoftmdl.cfm>
- Van der Ent, R. J., & Savenije, H. H. G. (2011). Length and time scales of atmospheric moisture recycling. *Atmospheric Chemistry and Physics*, 11(5), 1853-1863.
- Van der Ent, R. J., Savenije, H. H. G., Schaeffli, B., & Steele-Dunne, S. C. (2010). Origin and fate of atmospheric moisture over continents. *Water Resources Research*, 46(9).
- Wärmark, K. (2015). *Assessment of water footprint for civil construction projects*. Swedish University of Agricultural Sciences, Lennart Hjelm, Uppsala.
- Water Footprint Network. (2016). Water Footprint Statistics (WaterStat). Retrieved from <http://waterfootprint.org/en/resources/water-footprint-statistics/>
- Wellington Regional Council. (2014). *Minimum flow recommendations for the Wellington region - Technical report to support the draft Natural Resources Plan*. Retrieved from Wellington, New Zealand:

- World Health Organization, & United Nations Childrens Fund. (2015). *Progress on sanitation and drinking water- 2015 update and MDG assessment*. Retrieved from
- World Wide Fund for Nature. (2014). *Living Planet Report 2014- Species and spaces, people and places*. Retrieved from Gland, Switzerland:
- Yale University. (2015). Environmental Performance Index. Retrieved from <http://epi.yale.edu/epi>
- Young, R., Smart, G., & Harding, J. (2004). Impacts of hydro-dams, irrigation schemes and river control works. In Harding JS, Mosely MP, Pearson C, Sorrell B. (eds.). *Freshwaters of New Zealand, New Zealand Hydrological Society, and New Zealand Limnological Society*, 37.31-37.16.
- Zhang, L., Hickel, K., Dawes, W., Chiew, F. H., Western, A., & Briggs, P. (2004). A rational function approach for estimating mean annual evapotranspiration. *Water Resources Research*, 40(2).
- Zonderland-Thomassen, M. A., & Ledgard, S. F. (2012). Water footprinting – A comparison of methods using New Zealand dairy farming as a case study. *Agricultural Systems*, 110(0), 30-40.
doi:<http://dx.doi.org/10.1016/j.agry.2012.03.006>
- Zonderland-Thomassen, M. A., Lieffering, M., & Ledgard, S. F. (2014). Water footprint of beef cattle and sheep produced in New Zealand: water scarcity and eutrophication impacts. *Journal of Cleaner Production*, 73(0), 253-262.
doi:<http://dx.doi.org/10.1016/j.jclepro.2013.12.025>

Appendices

Appendix A - The History and Developments of Grey Water

The idea of expressing water pollution in terms of the amount of water required to 'dilute' waste was first proposed by Falkenmark and Lindh (1974). As a rule of thumb, a dilution factor of 10-50 times the wastewater flow was deemed appropriate (Falkenmark & Lindh, 1974). Later, Postel et al. (1996) applied a dilution factor for waste absorption of 28 L per second/ per 1000 of the population. It was only in 2006 that pollution was examined by type and consideration was given to any wastewater treatment. This resulted in the use of ambient water quality standards as the criterion for the dilution factor required (Chapagain et al., 2006).

The term 'Grey Water Footprint' was first introduced by Hoekstra and Chapagain in 2008. Since its inception, the term has received some criticism in the literature, and has resulted in the formation of a grey water footprint working group. The currently accepted definition as proposed by the WFN is concurrent with Table 1.1. Depending on whether pollution is entering surface water or groundwater, the local ambient water quality standard for the pollutant, or the drinking water standards are applied respectively. It is therefore plausible in some situations to calculate the grey water footprint for surface and groundwater bodies independently^{51,52}. The grey water footprint is analogous to the 'critical-load approach' which is reference to the load of pollutants, which if exceeded, consumes the assimilation capacity of the receiving water body (Hoekstra et al., 2011). Similar again, is the US EPA's total maximum daily load (TMDL), which calculates the maximum pollutant load that a water body can receive, whilst continuing to meet water quality standards (United States Environmental Protection Agency, 2013). What these methods all have in common is

⁵¹ Separate values can be applied where precise loads to groundwater and surface water systems are known.

⁵² It is important to note when doing so, that groundwater often ends up as surface water. It is proposed as an alternative for pollutant loads to groundwater that the difference between the water quality standard and the natural background concentration for the most critical water body (being either the groundwater or surface water) be used. When the pollutant loads is to surface water the relevant data for that system is used.

that they recognise that the capacity for a receiving water body to assimilate pollutants is limited by the difference between maximum and natural concentrations.

These approaches however have received criticism in the literature, for example Ridoutt and Pfister (2013) stipulate that the critical-load approach is problematic when assessing compounds with no standards, and does not take advantage of LCA developments such as fate, exposure and effects modelling. Further, the grey water footprint as outlined by the WFN, has potential limitations as no consideration appears to be given to natural attenuation processes that may improve water quality downstream of the initial pollutant. Additionally, no consideration is given to the combined effect of pollutants which may be much greater than when the impacts of chemicals are treated individually (Hoekstra et al., 2011).

Appendix B - Mean Natural Runoff (mm/year) to New Zealand Rivers (1996-2006)

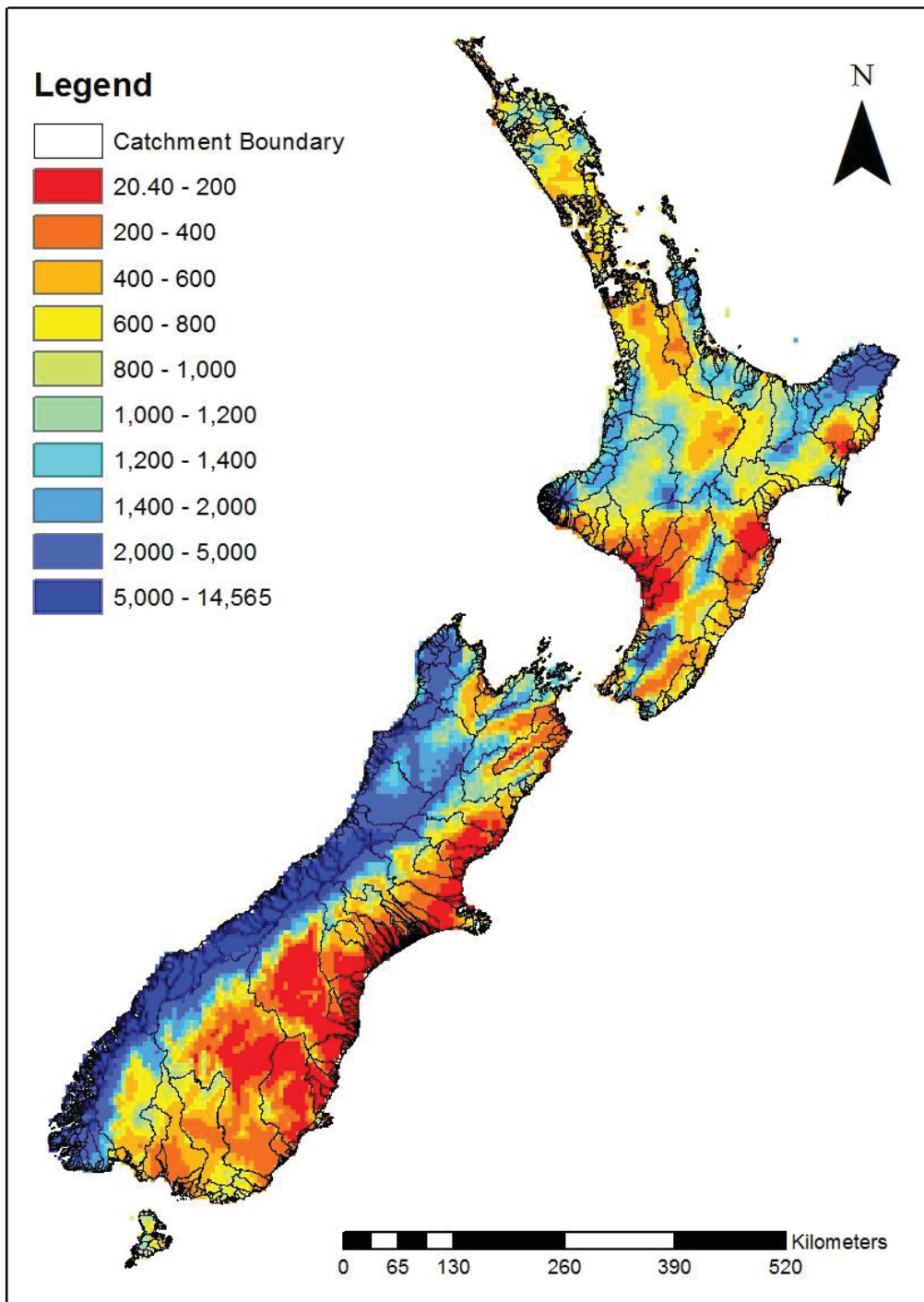


Figure A.1 Mean natural runoff (mm/year) to New Zealand rivers (1996-2006) according to Woods et al. (2006).

Appendix C Aquifer Locations and Extents Throughout New Zealand



Figure A.2 Aquifer Locations and Extents Throughout New Zealand (Ministry for the Environment & Statistics New Zealand, 2015).

Appendix D National Trends in Nitrogen Concentrations

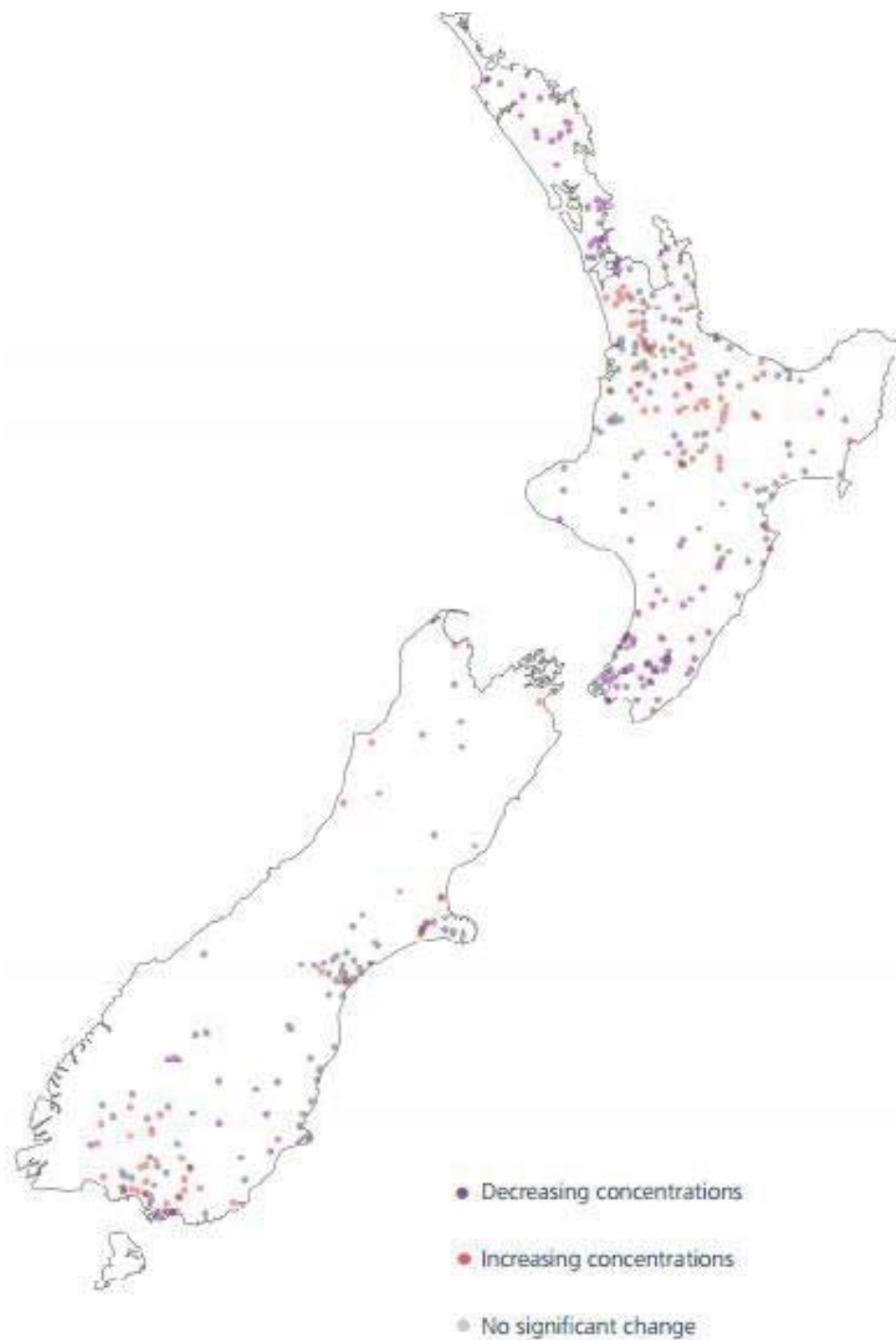


Figure A.3 National Trends in Nitrogen Concentrations (PCE, 2015).

Appendix E National Trends in Phosphorous Concentrations

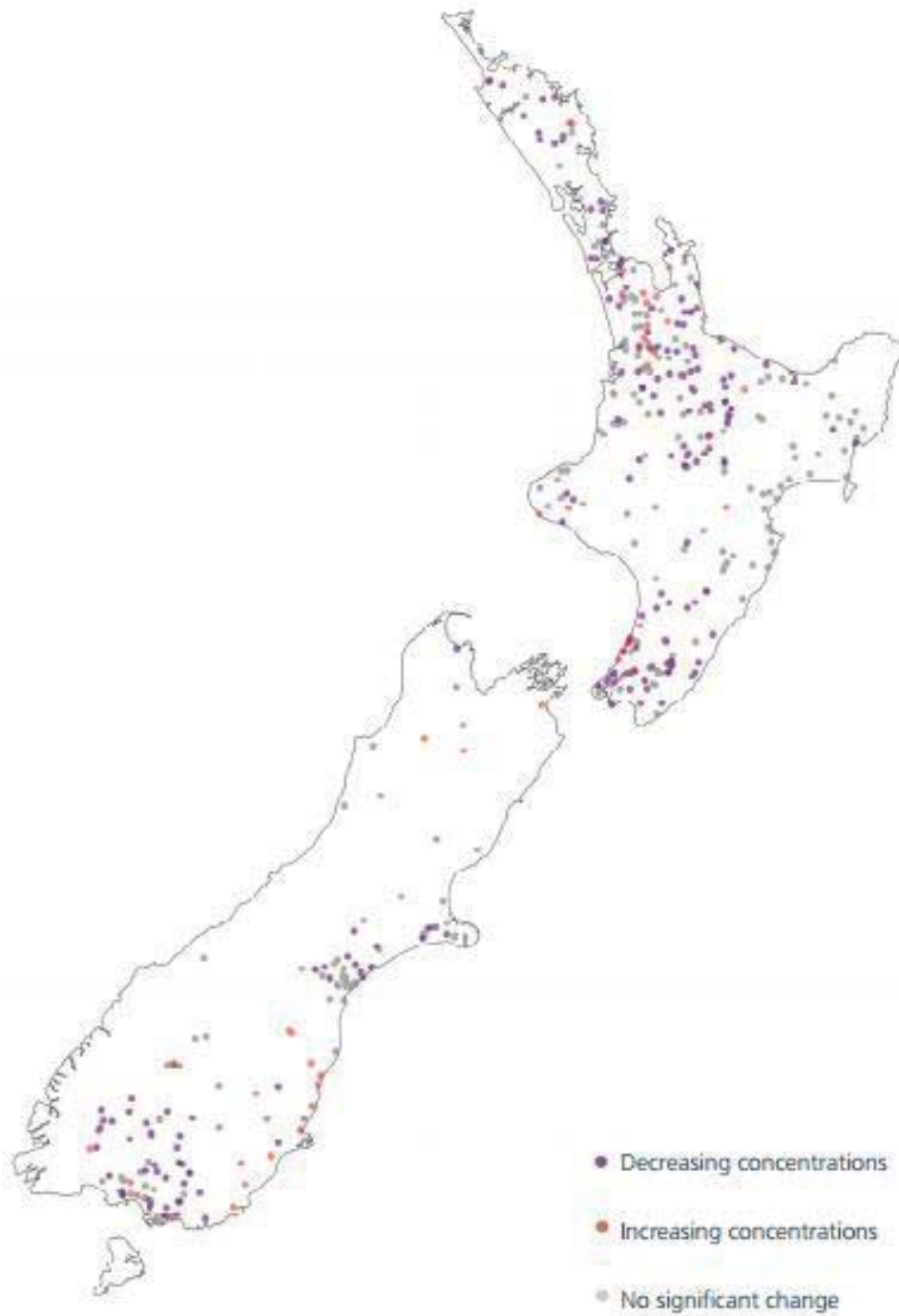


Figure A.4 National Trends in Phosphorous Concentrations (PCE, 2015).

Appendix F Rivers Affected by Hydro-Electricity Generation in New Zealand

Table A.1 Rivers Affected by Hydro-Electricity Generation in New Zealand

Hydropower Scheme	Affected River	Installed Capacity (MW)
Lake Karapiro	Waikato River	90
Arapuni	Waikato River	197
Waipapa Dam	Waikato River	51
Maraetai Dam	Waikato River	360
Whakamaru Dam	Waikato River	100
Atiamuri	Waikato River	84
Ohakuri Dam	Waikato River	112
Aratiatia Dam	Waikato River	84
Tokanuu	Tongariro River	240
Rangipo Dam	Tongariro River	120
Mangaio power station	Tongariro River	1.8
Moawhango Dam	Moawhango River/ Mangaio stream	
Kaitawa	Waikaretaheke River	37
Tuai	Waikaretaheke River	52
Piripaua	Waikaretaheke River	44
Managahao	Mangahao River	19
Matahina Dam	Rangitikei River	72
Patea	Patea River	31
Llyoyd Mandeno	Wairoa River	16
Lower Mangapapa	Wairoa River	5.6
Kaimai 5	Wairoa River	0.3
Ruahihi	Wairoa River	20
Wheao	Rangitaiki River	26
Cobb	Takaka / Cobb River	32
Branch	Branch River	11
Lake Coleridge	Rakaia River	45
Highbank	Rakaia River	25
Tekapo A Dam	Waitaki River	25
Tekapo B Dam	Waitaki River	160
Ohau A	Waitaki River	264
Ohau B	Waitaki River	212
Ohau C	Waitaki River	212
Benmore Dam	Waitaki River	540
Aviemore Dam	Waitaki River	220
Waitaki	Waitaki River	105
Paerau	Taieri River	10
Clyde Dam	Clutha River	432
Roxburgh Dam	Clutha River	320
Lake Manapouri	Waiau River	680
Waipori	Waipori River	54

Appendix G Installed hydroelectricity generation in New Zealand

Note: Each dot represents a power scheme larger than 5 MW

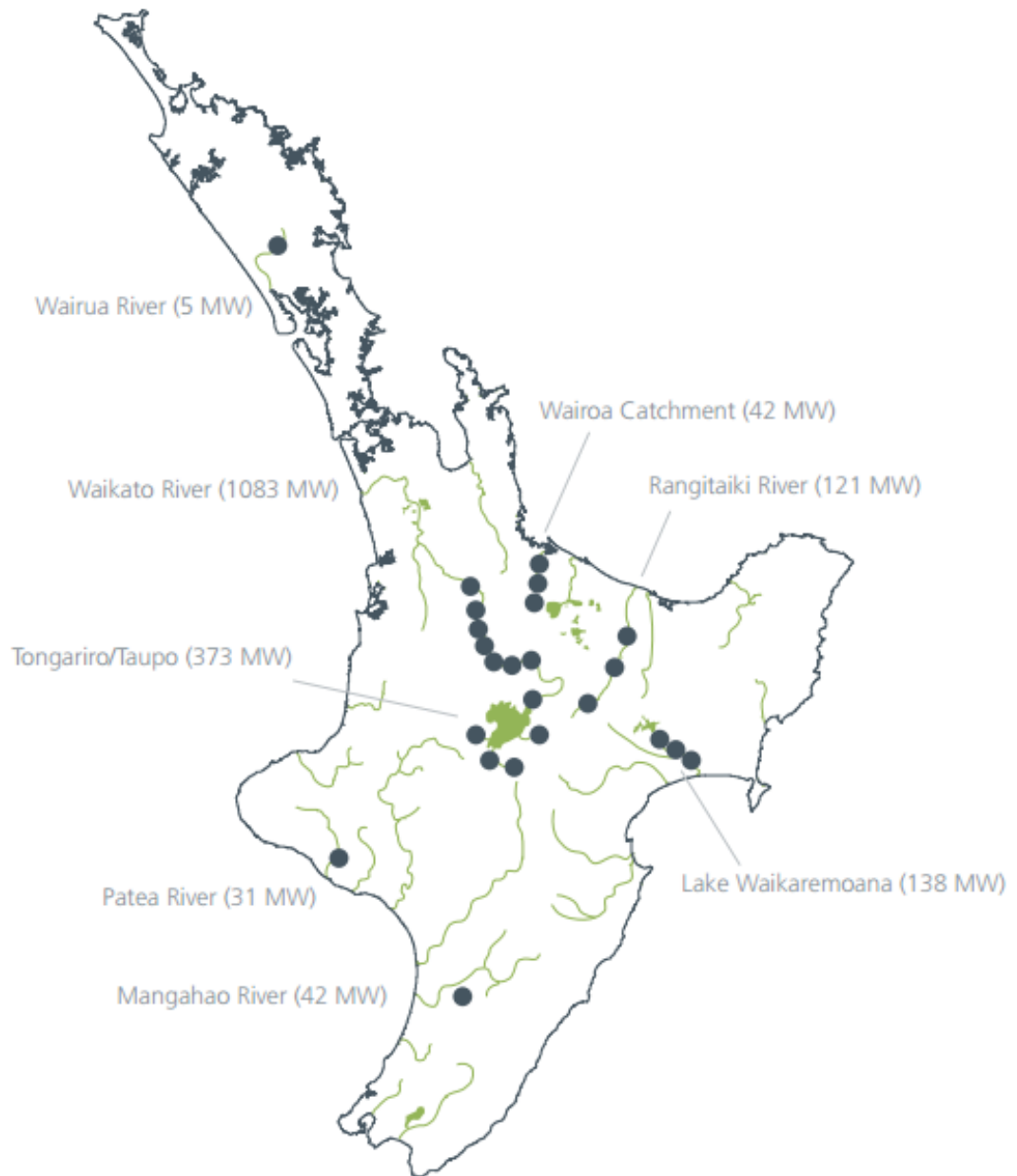


Figure A.5 Installed hydroelectricity generation in the North Island of New Zealand

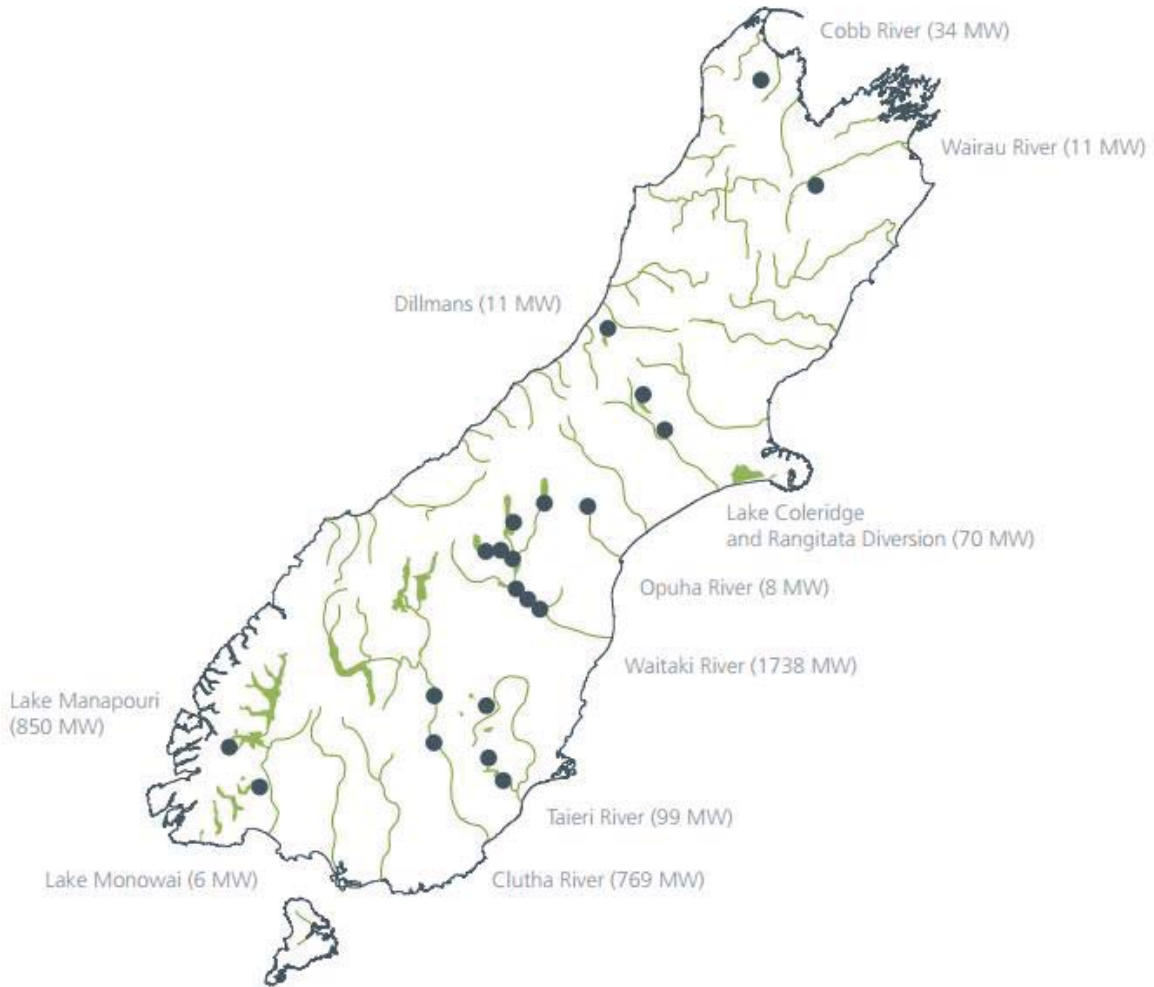


Figure A.6 Installed hydroelectricity generation in the South Island of New Zealand

Appendix H - Irrigation System Efficiency in New Zealand

There are several types of irrigation systems used throughout New Zealand. Figure A.7 shows the proportion of spray, flood and micro-irrigation systems.

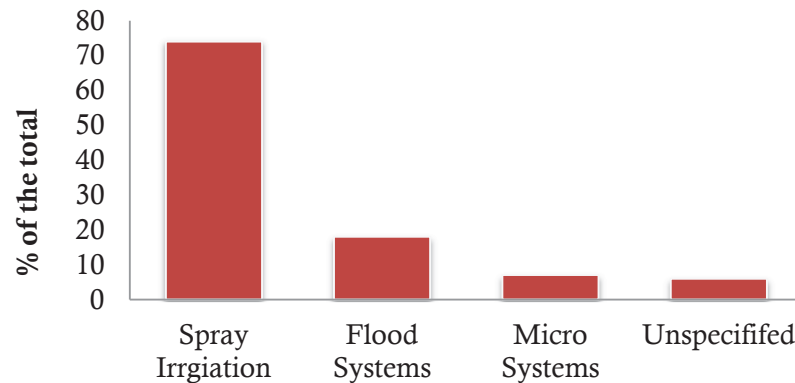


Figure A.7 Irrigation systems of New Zealand (Rockpoint, 2002)

Spray irrigation systems account for the majority of the systems used in New Zealand at 74%, followed by flood systems at 18%. Within those classes, centre-pivot systems are the most popular form of spray irrigator, and border strips are a widely used form of flood irrigation (Heiler, 2015). In New Zealand, there appears to be less difference in water use efficiency between the various spray irrigator types, than between border-strip (flood systems) and spray irrigation (McIndoe, 2002).

Spray Irrigation System Efficiency

Some research has been conducted in New Zealand to look at the efficiencies of the various irrigation systems. McIndoe and Carran (1998) reported efficiencies of 78% in 1977 and 96% in 1988. Whereas, others⁵³ found the efficiency of a centre-pivot system in New Zealand to be 88% (McIndoe, 2002).

Flood Irrigation Efficiency

Similarly, some variation occurs with border strip schemes. McIndoe (2002) report efficiencies of 45% and 34% for 1997 and 1988 respectively. Alternatively, Evans (1999) proposes an efficiency of 13%, although the accuracy of this value has been

⁵³ A Daveron, Hydro Services Ltd, pers comm. as cited in McIndoe (2002)

questioned as there is some concern that best-practice irrigation was not performed (McIndoe, 2002). Finally, Stronge (2001) calculated a seasonal water efficiency of 56% for border strip systems.

Appendix I - A Comparison of Freshwater Use Between Compressive Strengths of Concrete

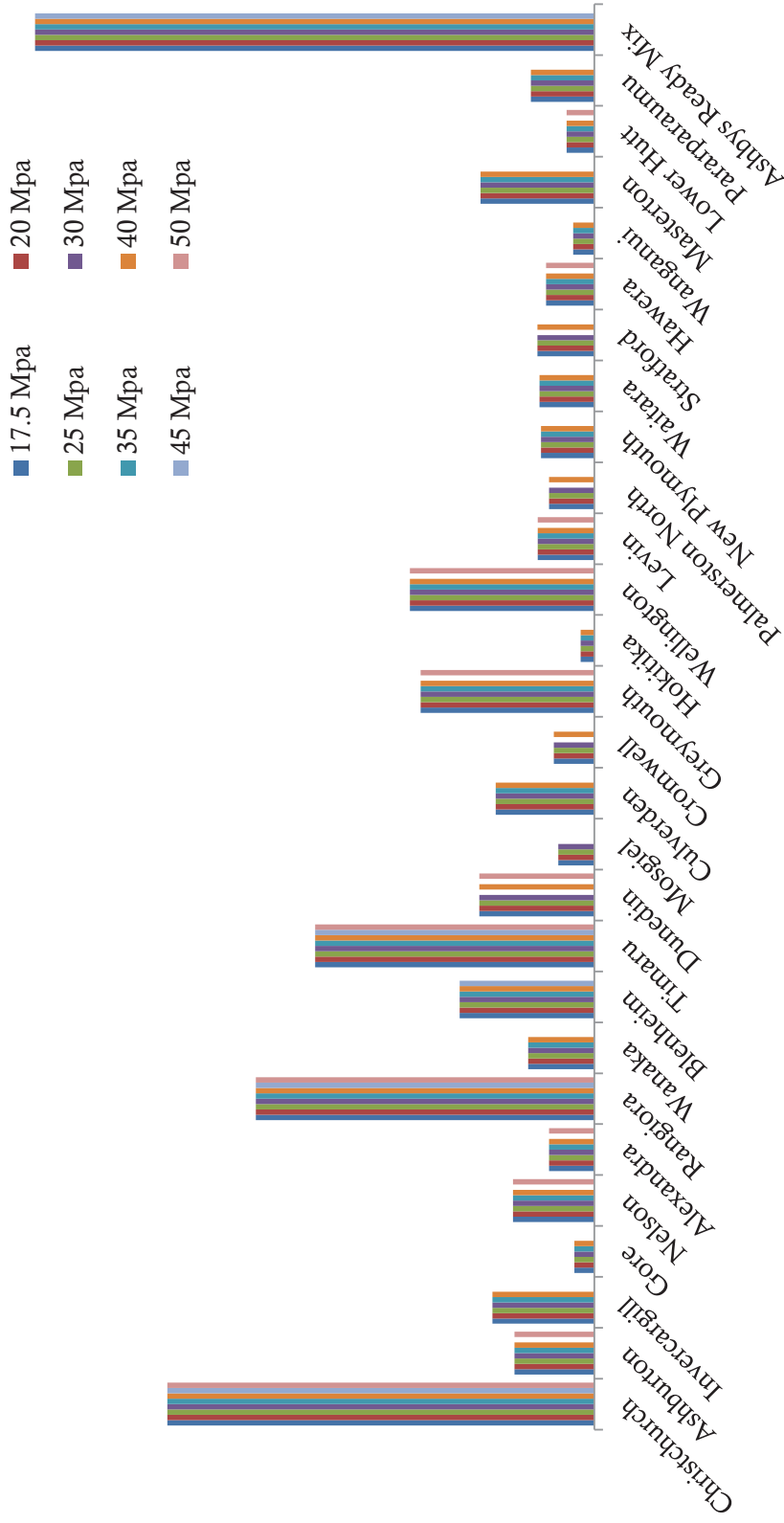


Figure A.8 A Comparison of Freshwater Use Between Compressive Strengths of Concrete

Note: Data presented in m³ freshwater per m³ concrete produced at each plant. For confidentiality reasons, y axis has been removed

Appendix J- Descriptive Statistics Across all Methods and Scales Considered During Impact Assessment

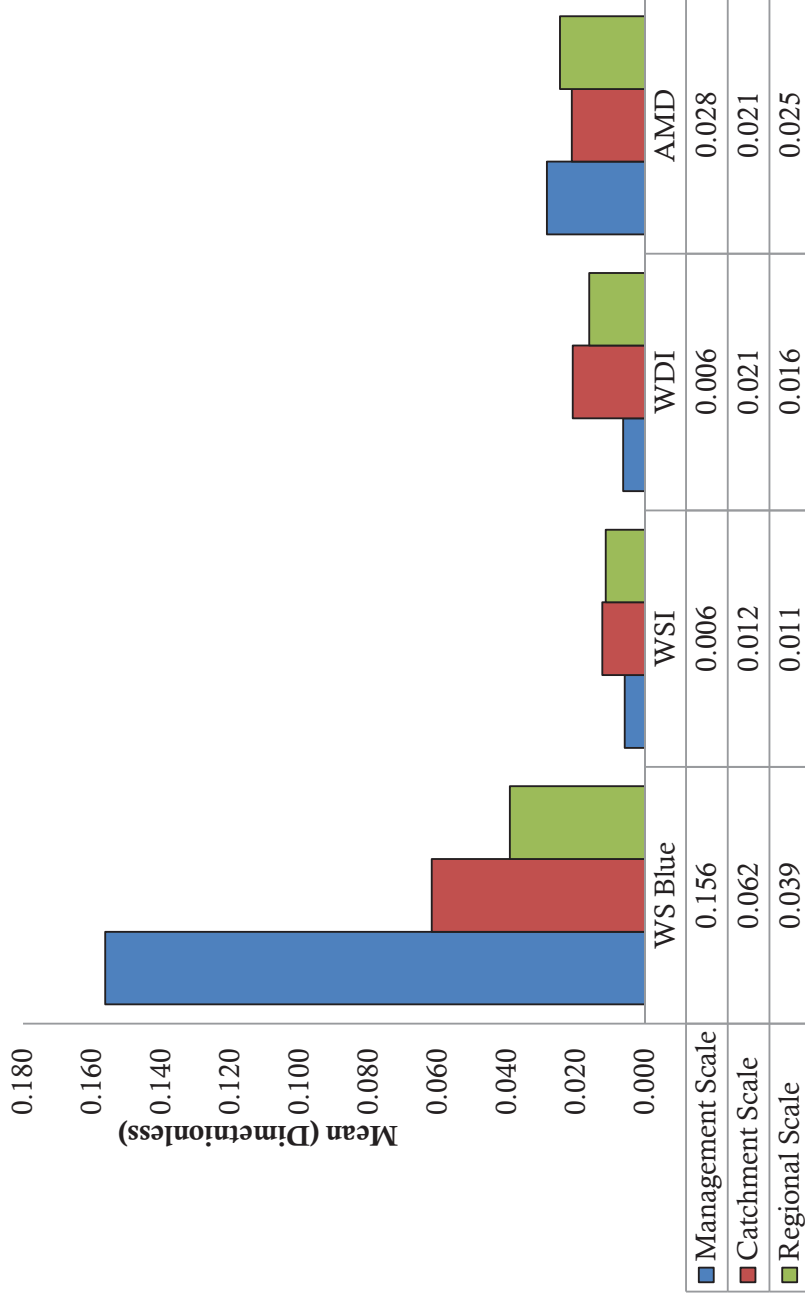


Figure A.9 Calculated mean across all methods and scales considered during impact assessment

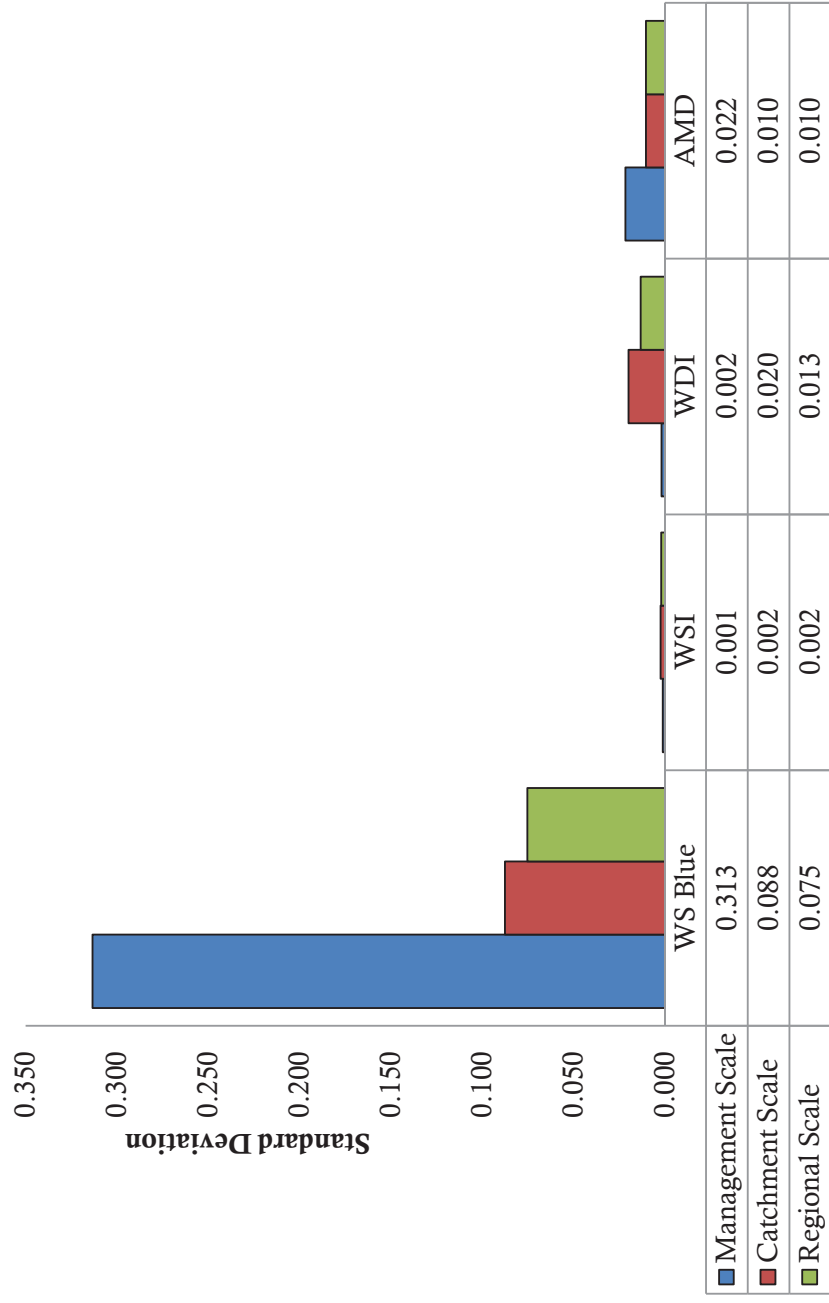


Figure A.10 Calculated standard deviations across all methods and scales considered during impact assessment

Appendix K - Schematic Diagrams of Concrete Batching Plants Freshwater Usage

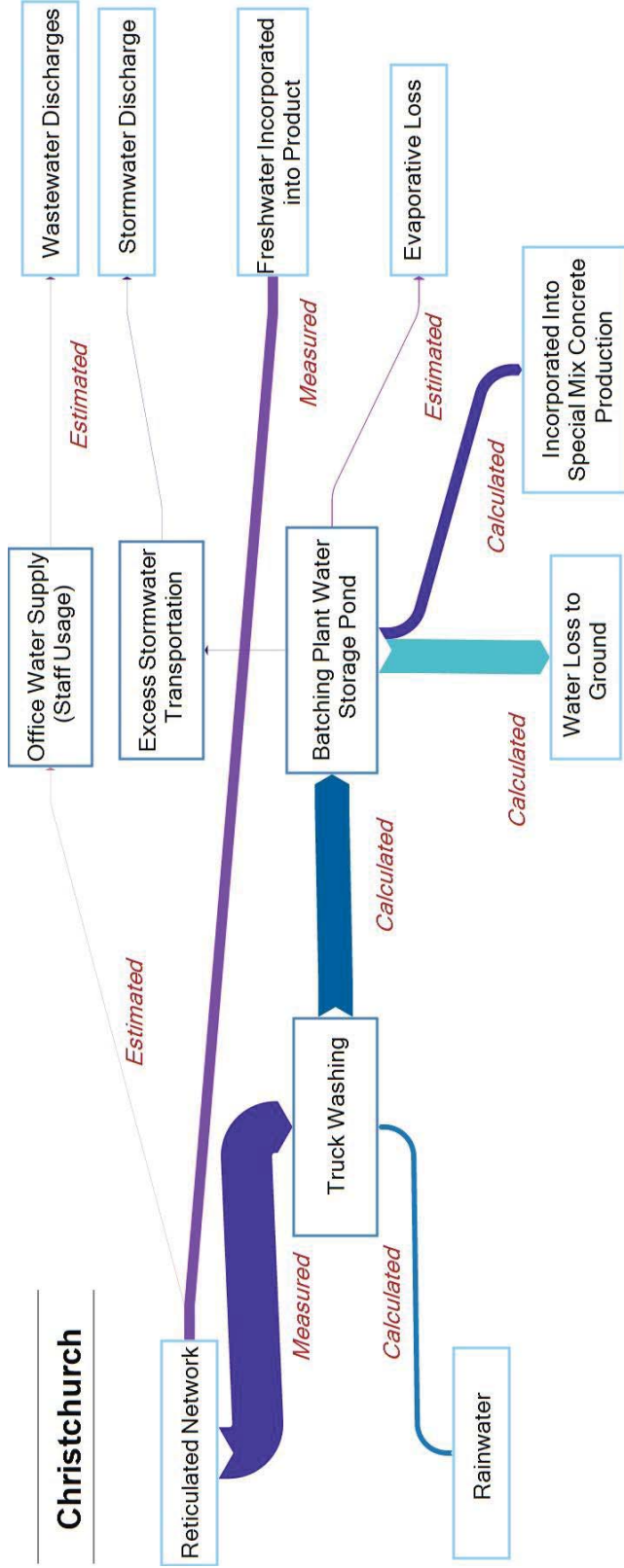


Figure A.11 Schematic diagram of freshwater withdrawals and consumption volumes at the Christchurch batching plant, for the 2013 period.

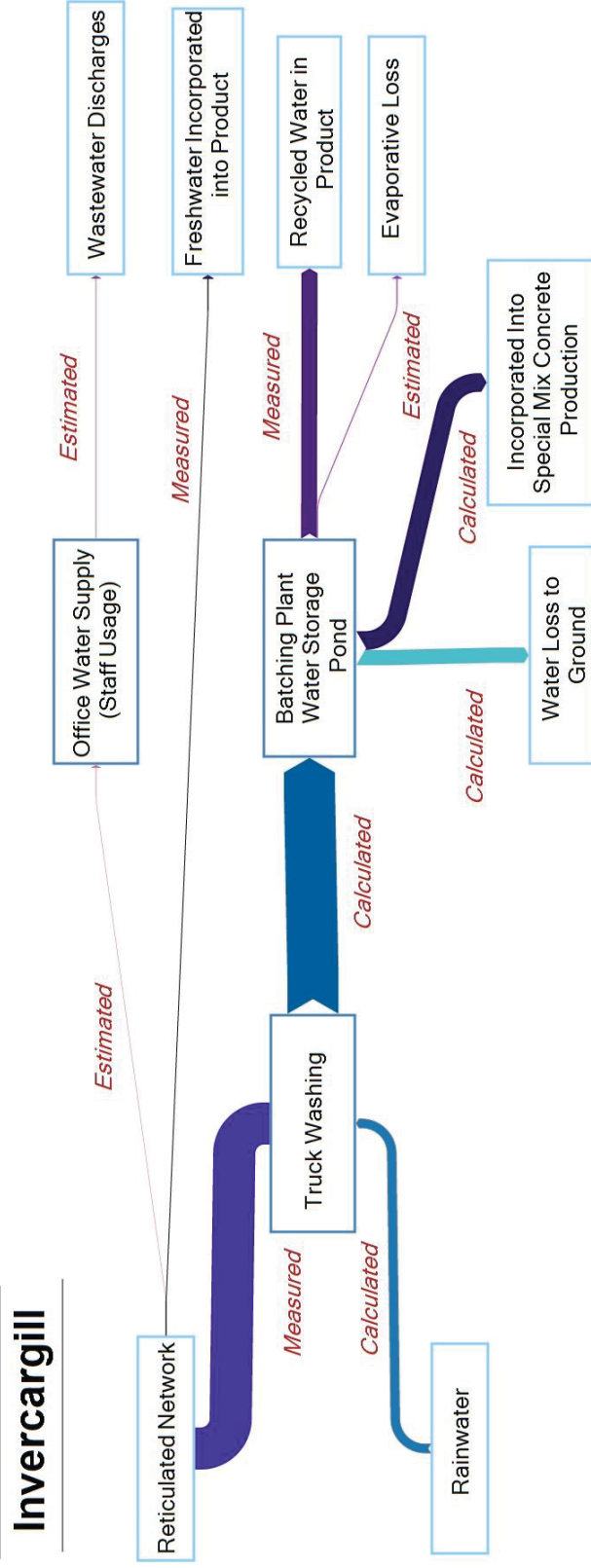


Figure A.12 Schematic diagram of freshwater withdrawals and consumption volumes at the Invercargill batching plant, for the 2013 period.

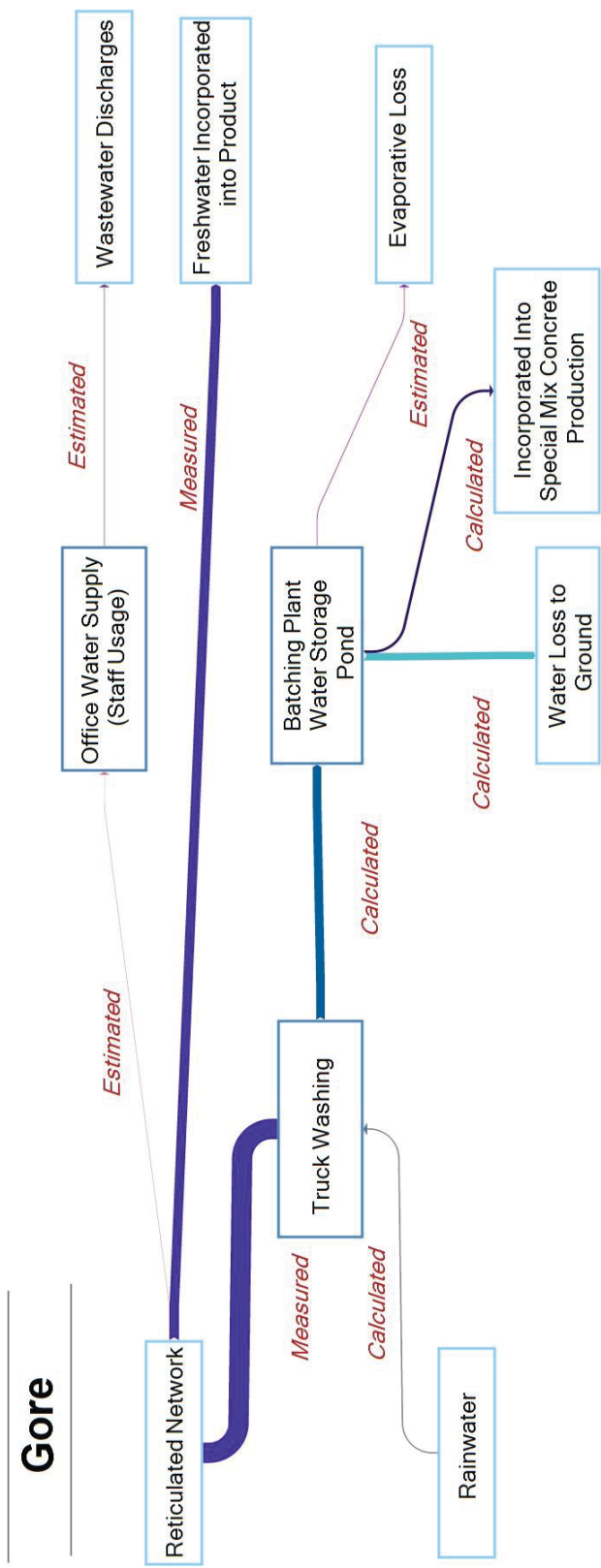


Figure A.13 Schematic diagram of freshwater withdrawals and consumption volumes at the Gore batching plant, for the 2013 period.

Nelson

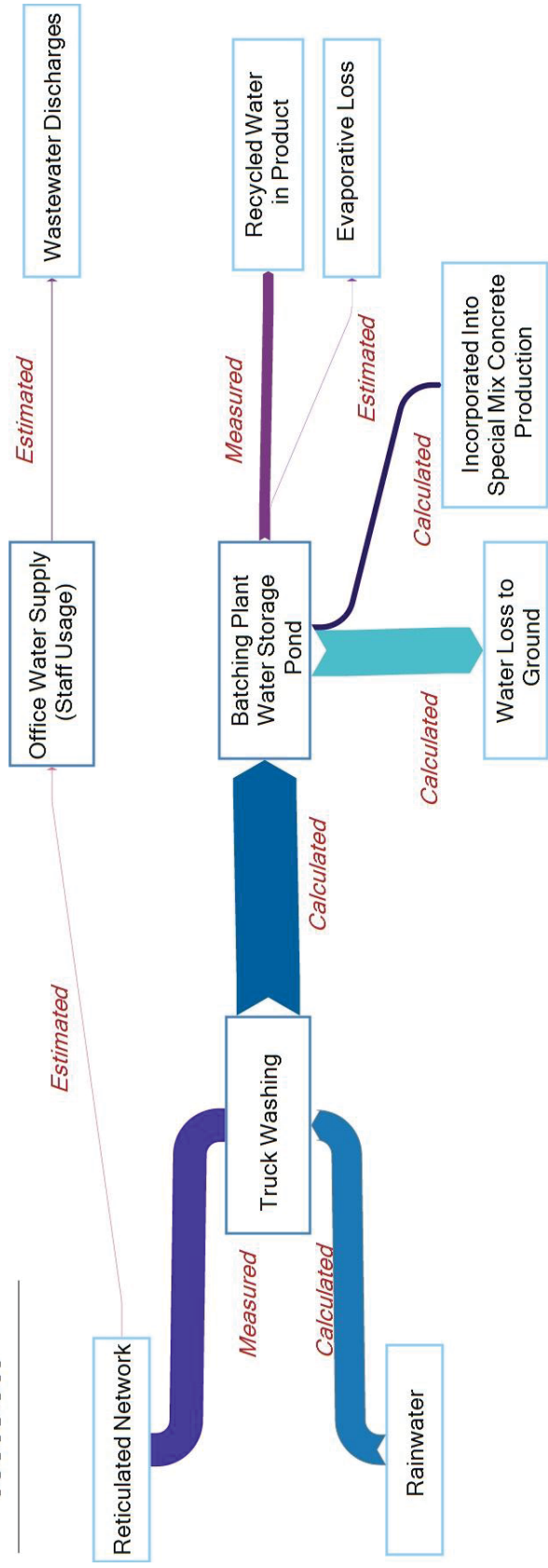


Figure A.14 Schematic diagram of freshwater withdrawals and consumption volumes at the Nelson batching plant, for the 2013 period.

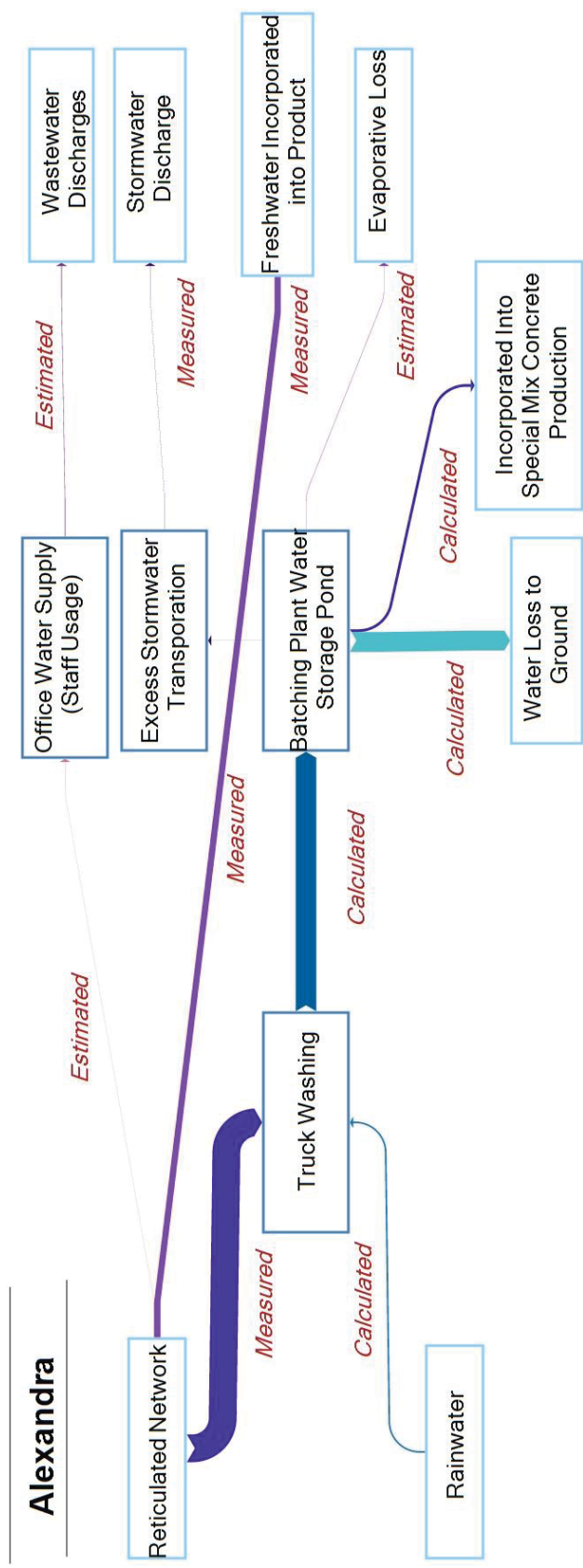


Figure A.15 Schematic diagram of freshwater withdrawals and consumption volumes at the Alexandria batching plant, for the 2013 period.

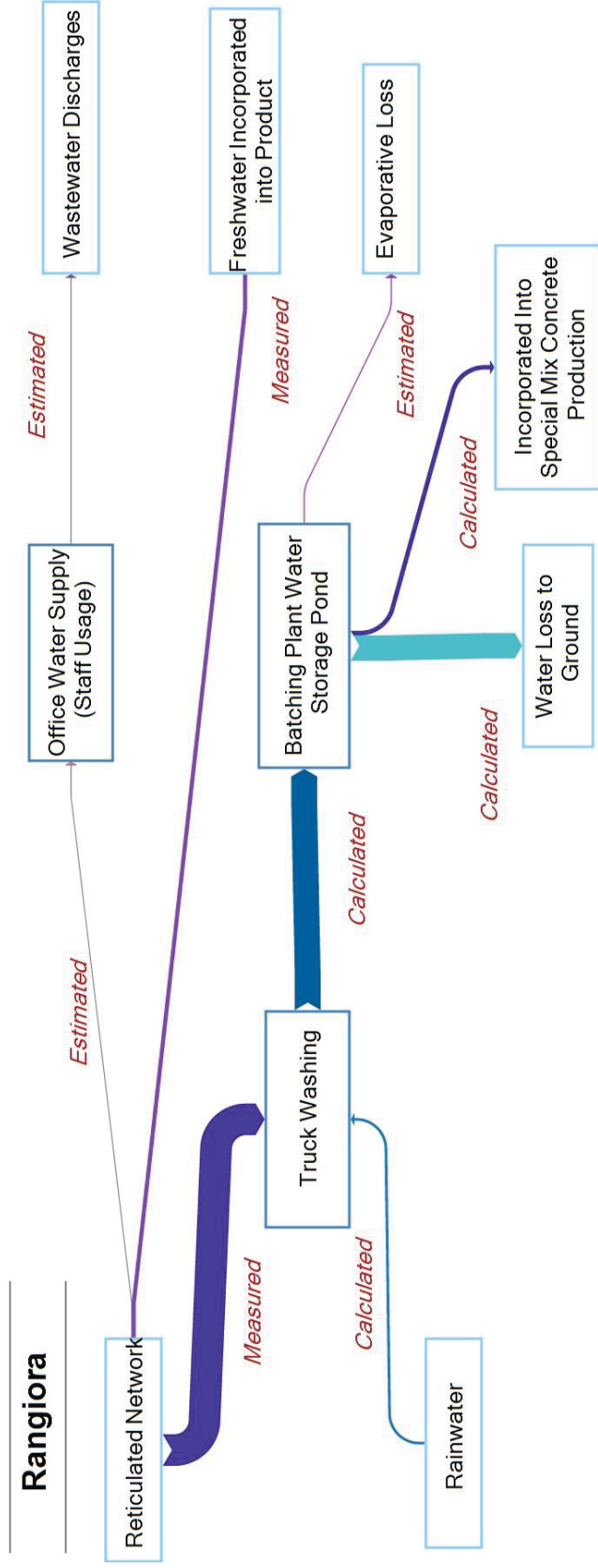


Figure A.16 Schematic diagram of freshwater withdrawals and consumption volumes at the Rangiora batching plant, for the 2013 period.

Wanaka

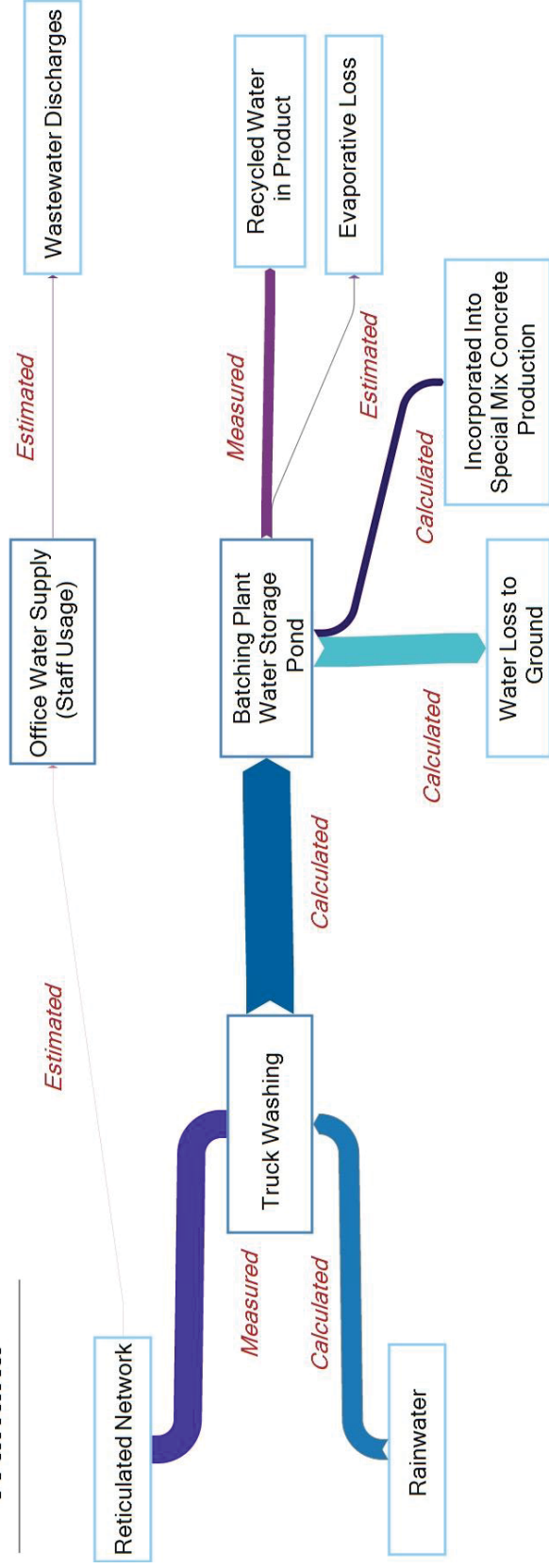


Figure A.17 Schematic diagram of freshwater withdrawals and consumption volumes at the Wanaka batching plant, for the 2013 period.

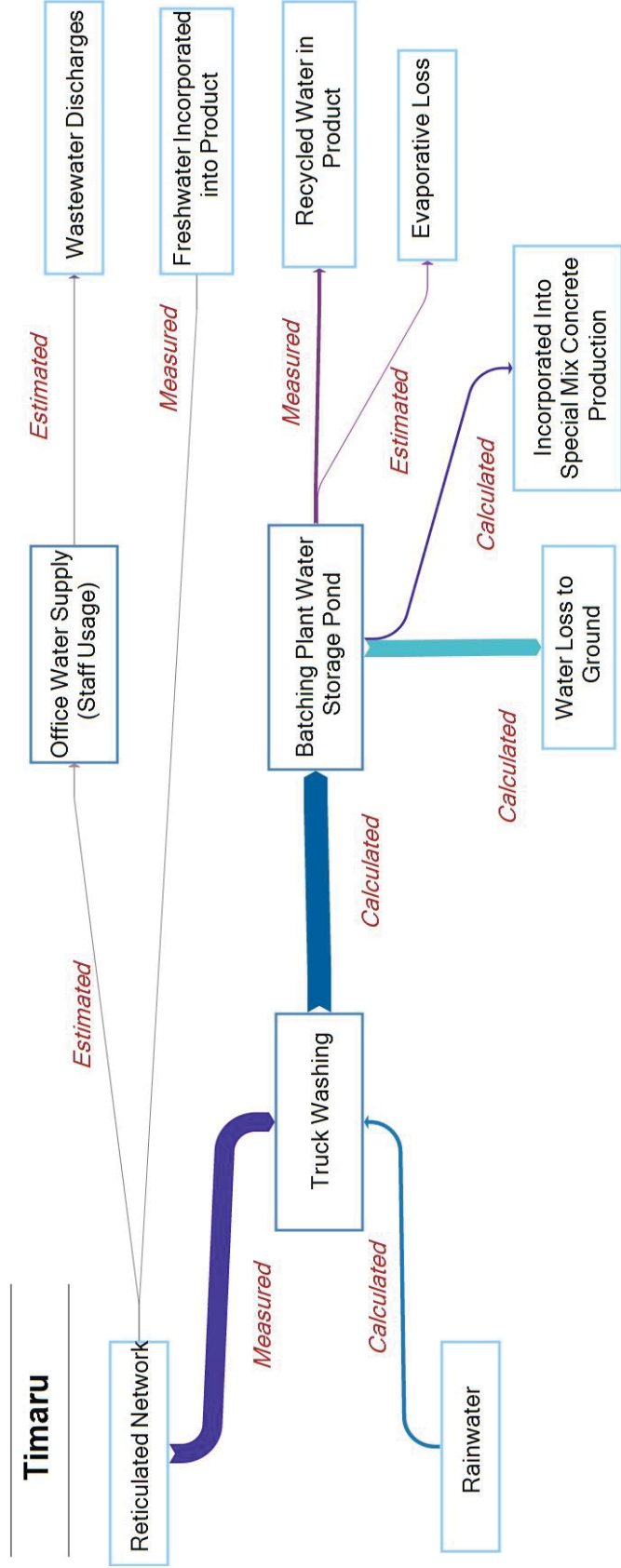


Figure A.18 Schematic diagram of freshwater withdrawals and consumption volumes at the Timaru batching plant, for the 2013 period.

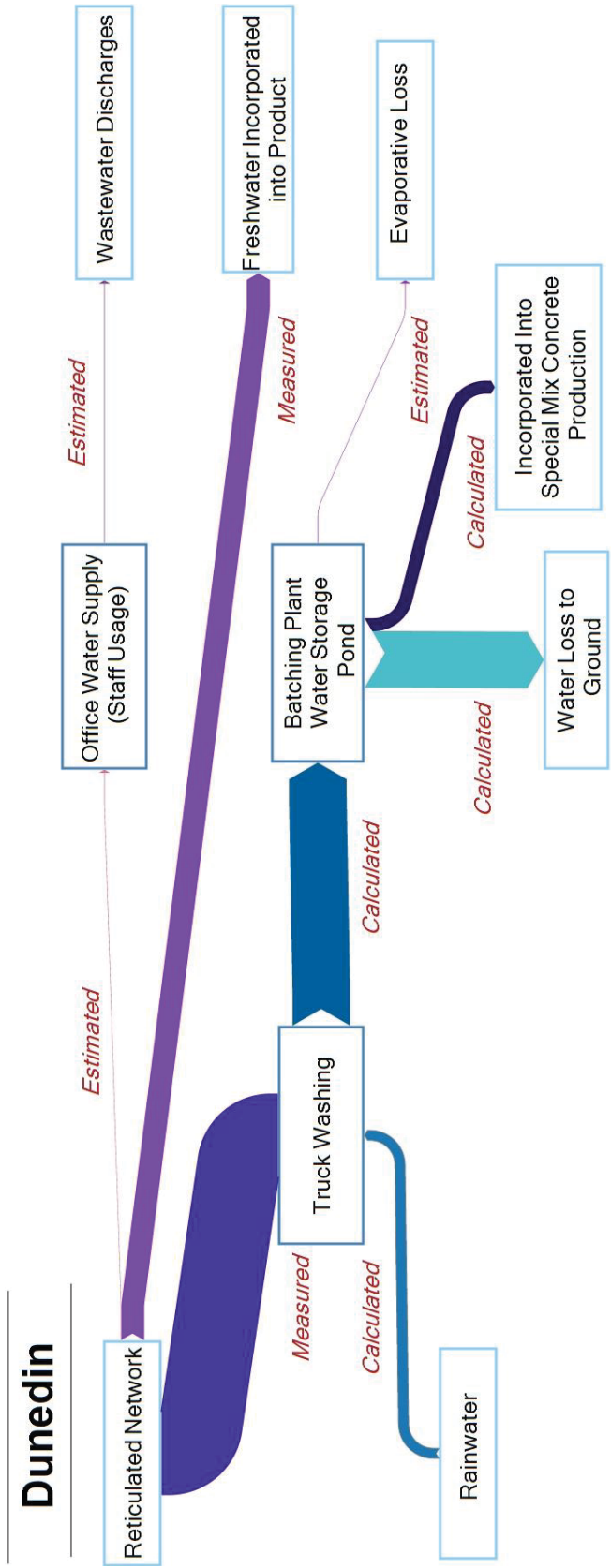


Figure A.19 Schematic diagram of freshwater withdrawals and consumption volumes at the Dunedin batching plant, for the 2013 period.

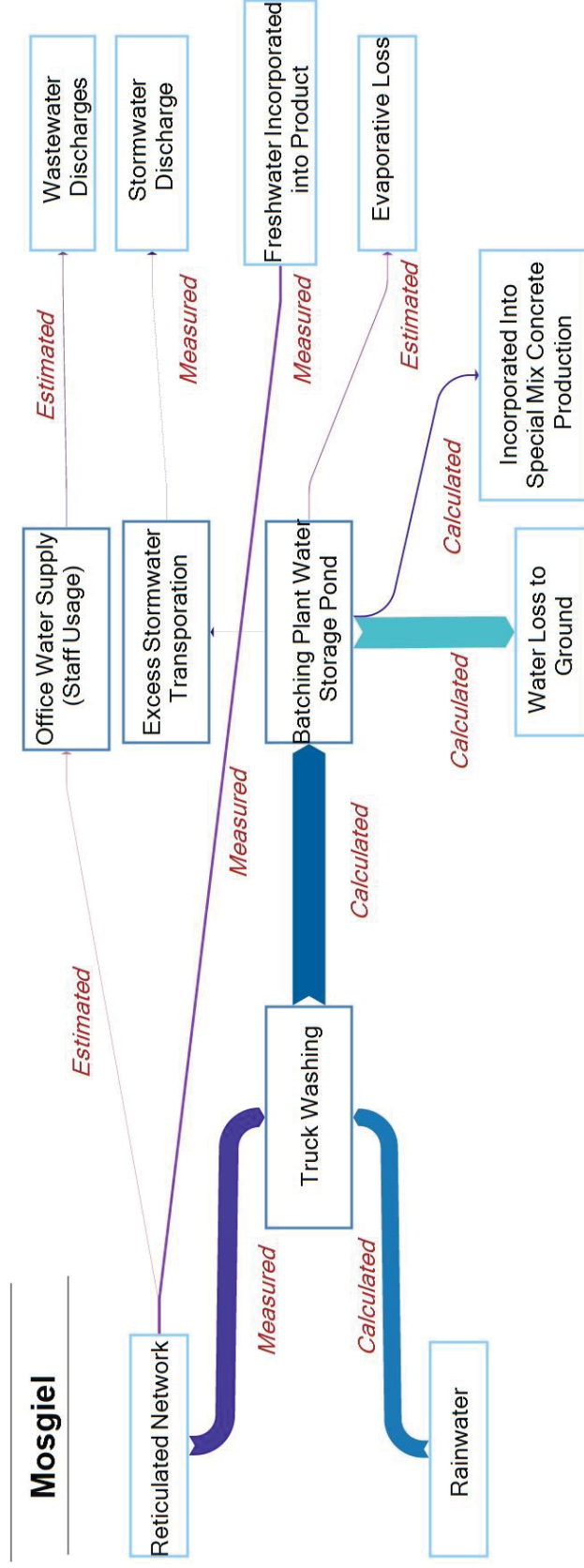


Figure A.20 Schematic diagram of freshwater withdrawals and consumption volumes at the Mosgiel batching plant, for the 2013 period.

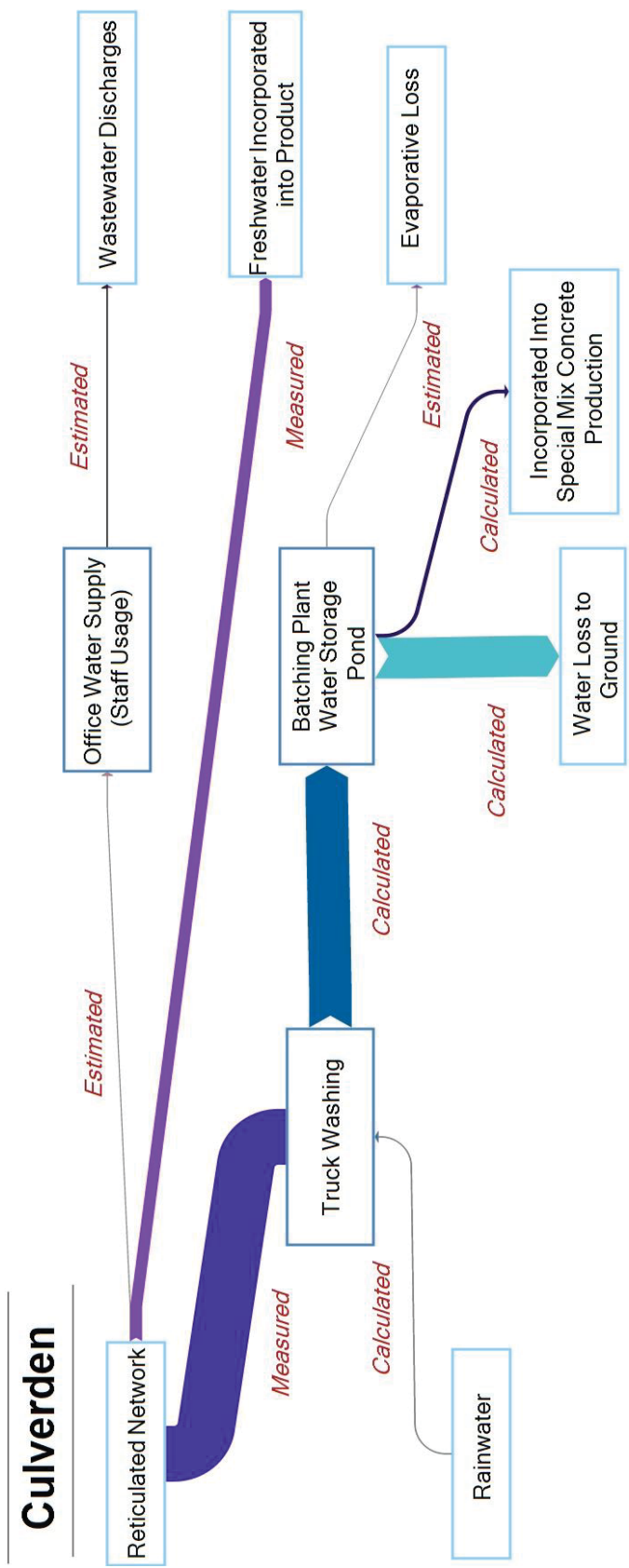


Figure A.21 Schematic diagram of freshwater withdrawals and consumption volumes at the Culverden batching plant, for the 2013 period.

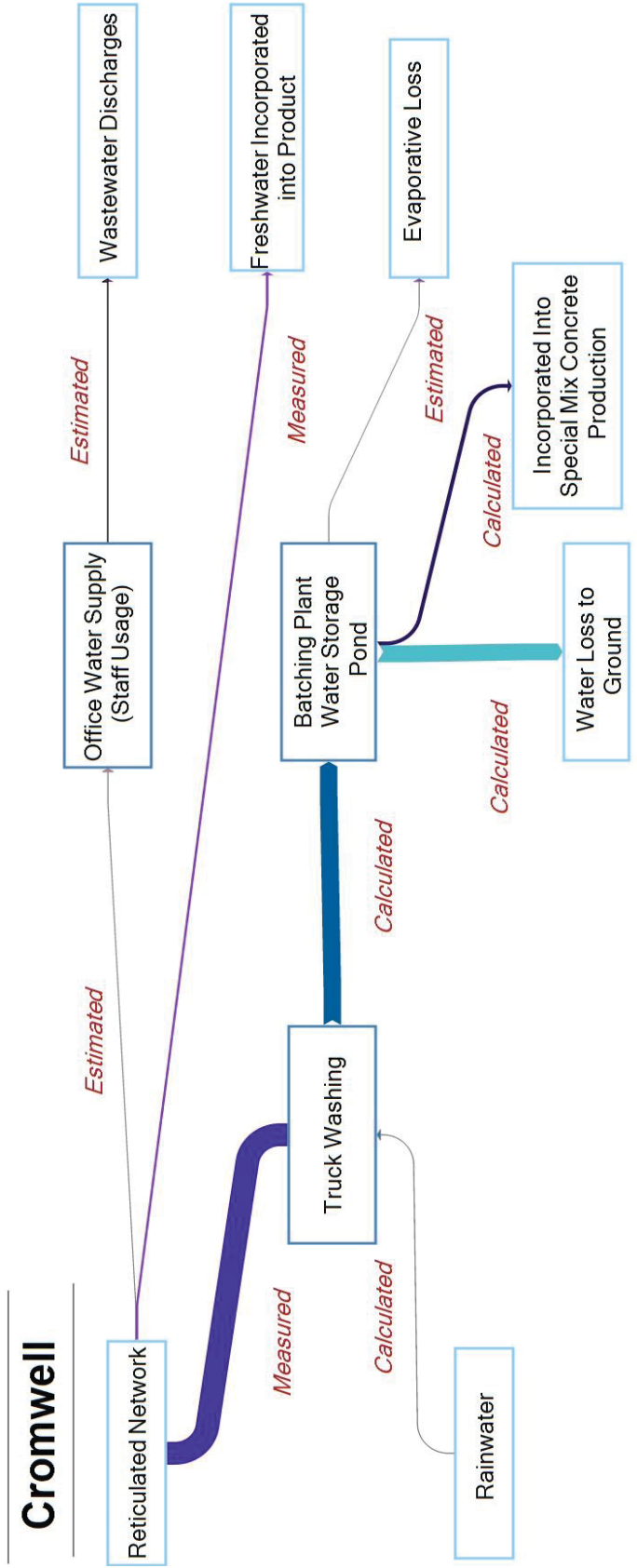


Figure A.22 Schematic diagram of freshwater withdrawals and consumption volumes at the Cromwell batching plant, for the 2013 period.

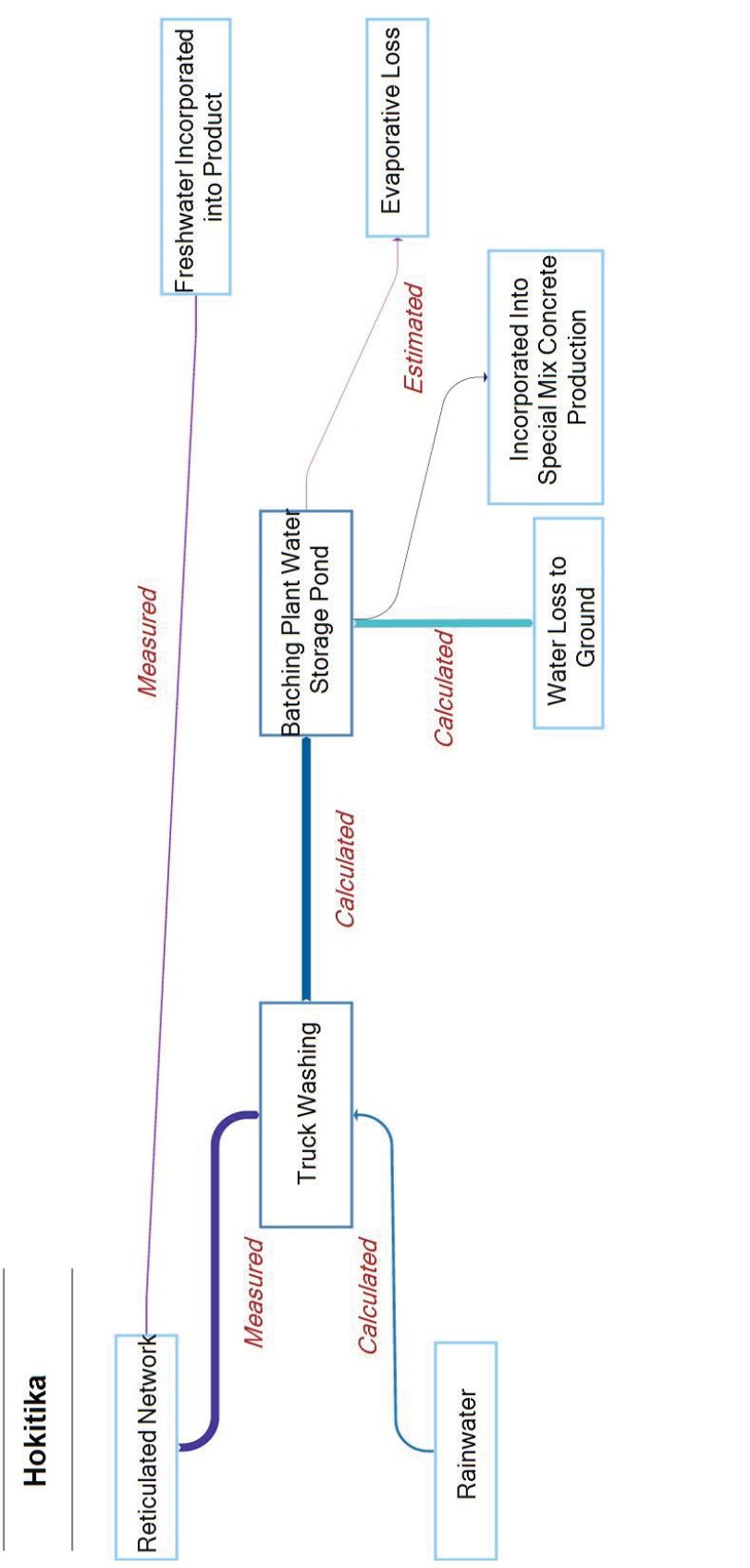


Figure A.23 Schematic diagram of freshwater withdrawals and consumption volumes at the Hokitika batching plant, for the 2013 period.

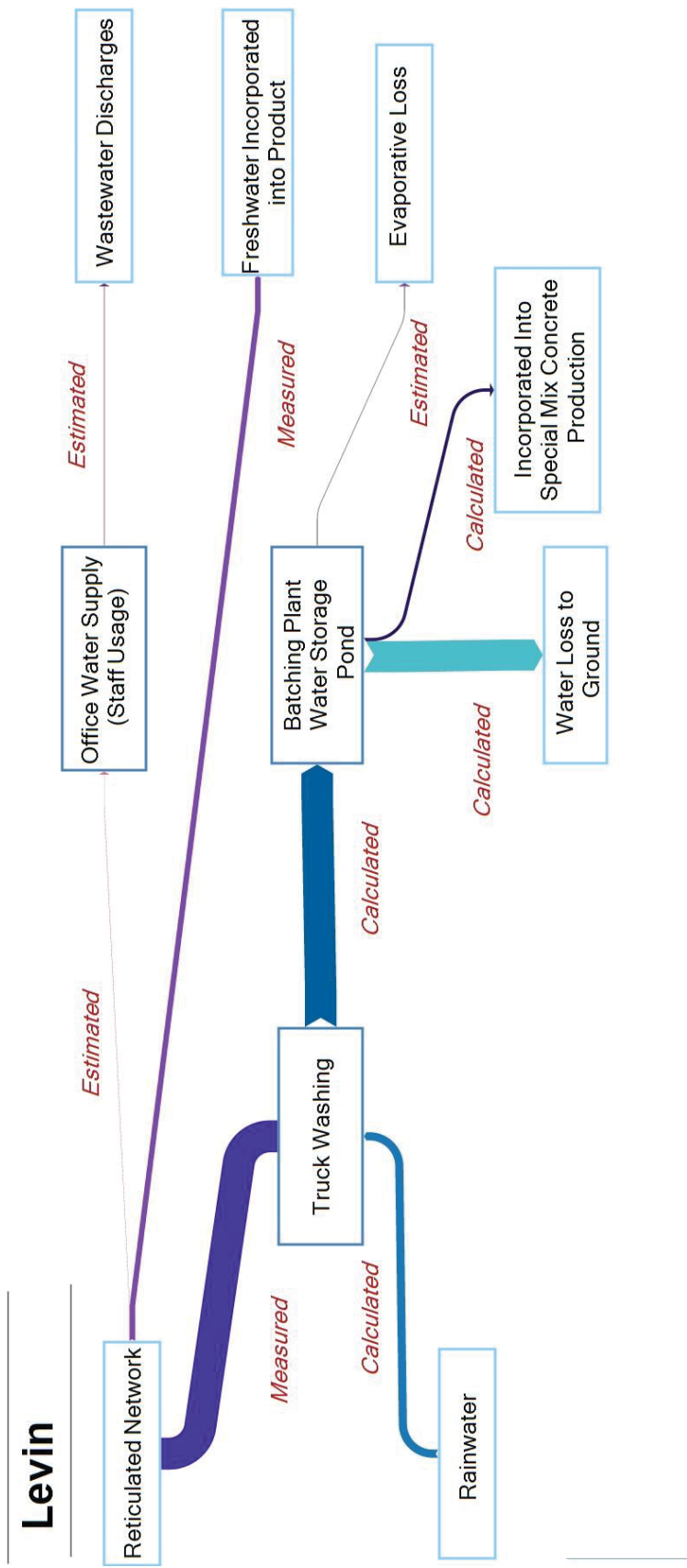


Figure A.24 Schematic diagram of freshwater withdrawals and consumption volumes at the Levin batching plant, for the 2013 period.

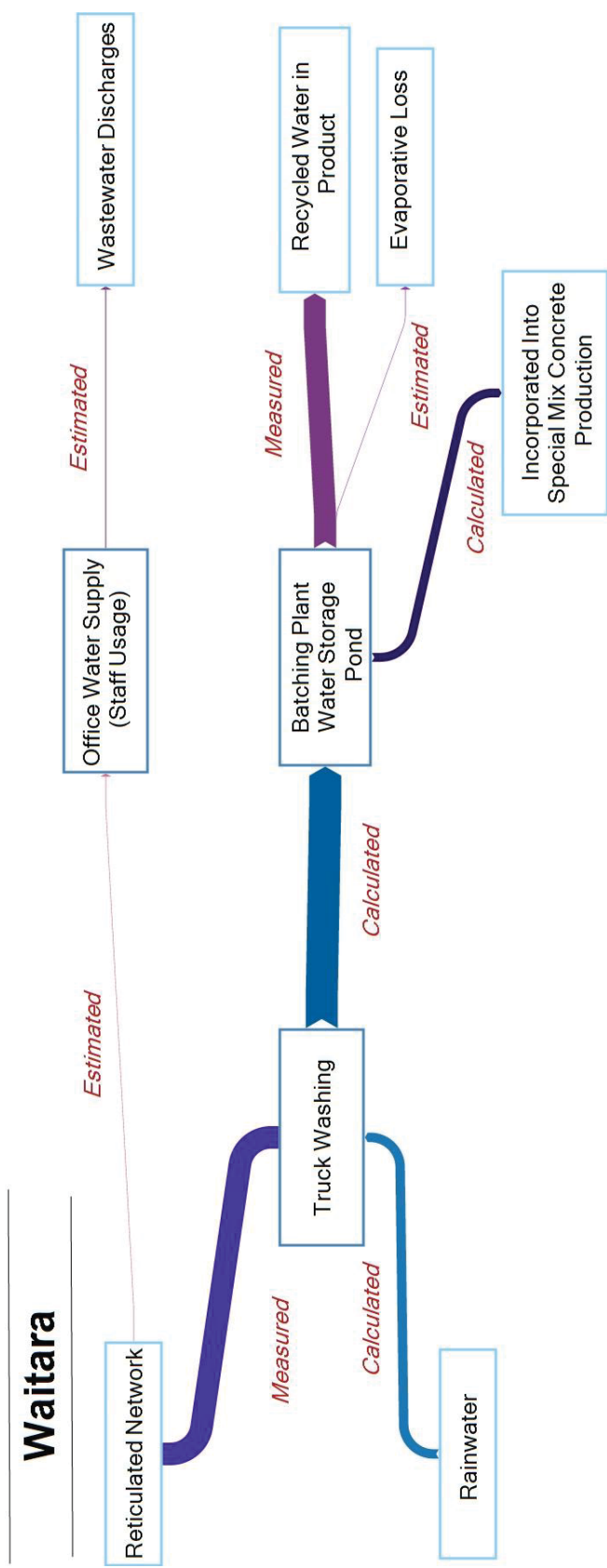


Figure A.25 Schematic diagram of freshwater withdrawals and consumption volumes at the Waitara batching plant, for the 2013 period.

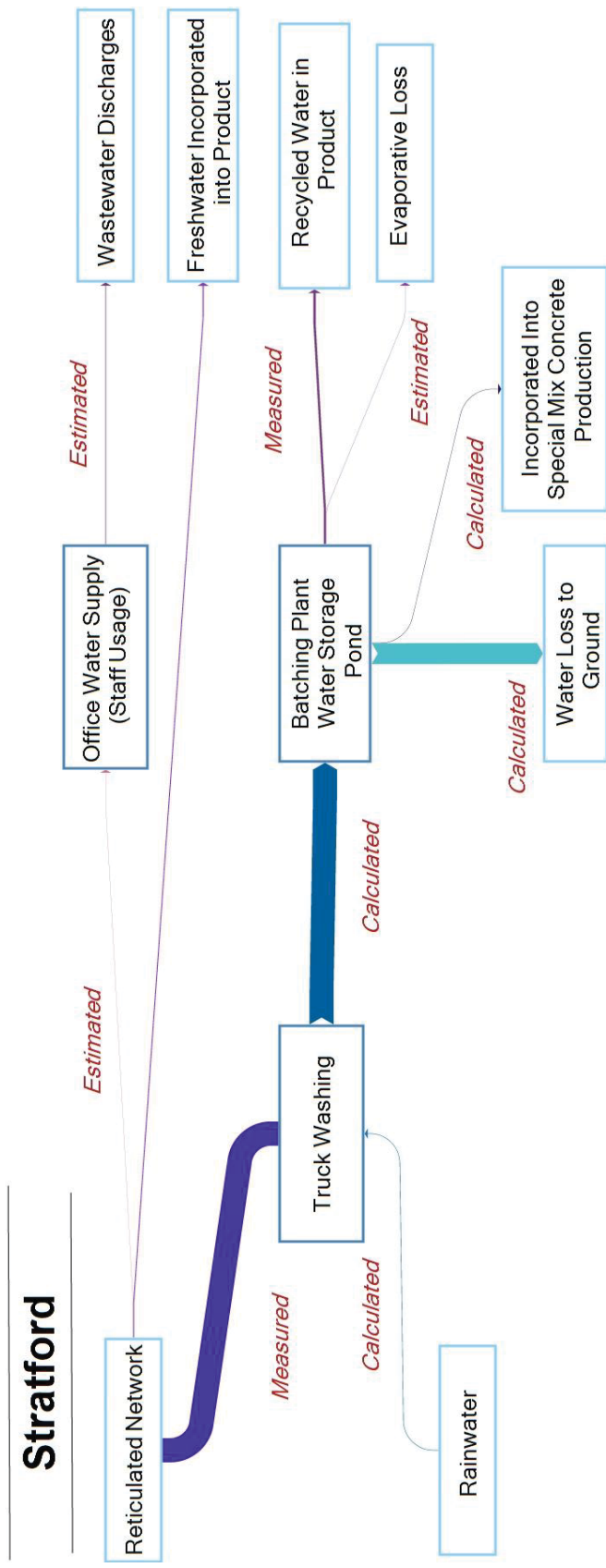


Figure A.26 Schematic diagram of freshwater withdrawals and consumption volumes at the Stratford batching plant, for the 2013 period.

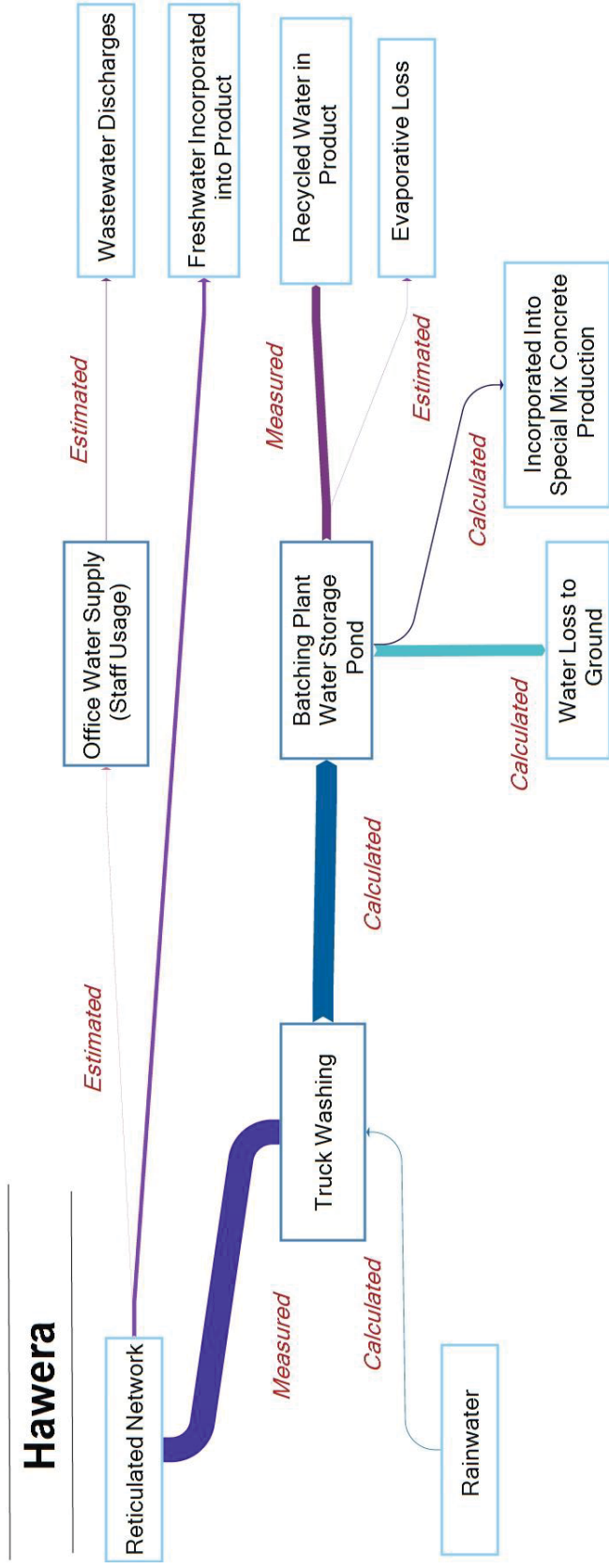


Figure A.27 Schematic diagram of freshwater withdrawals and consumption volumes at the Hawera batching plant, for the 2013 period.

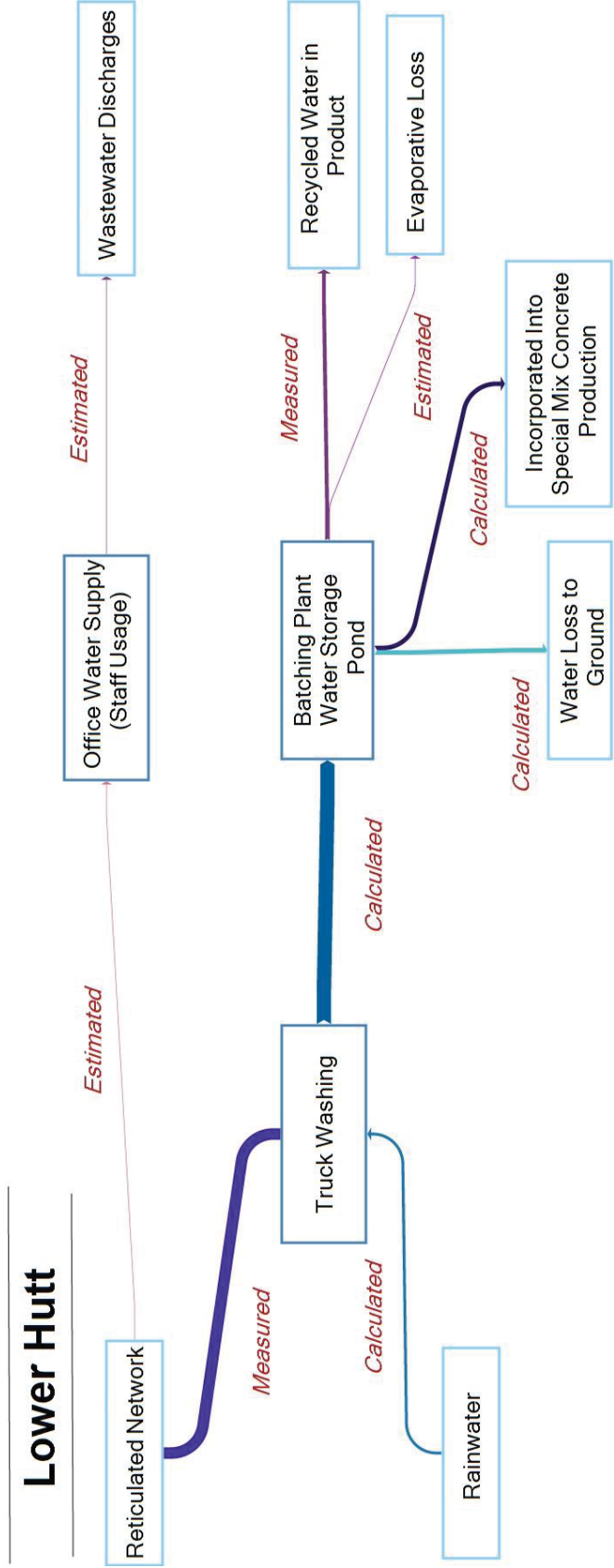


Figure A.28 Schematic diagram of freshwater withdrawals and consumption volumes at the Lower Hutt batching plant, for the 2013 period.

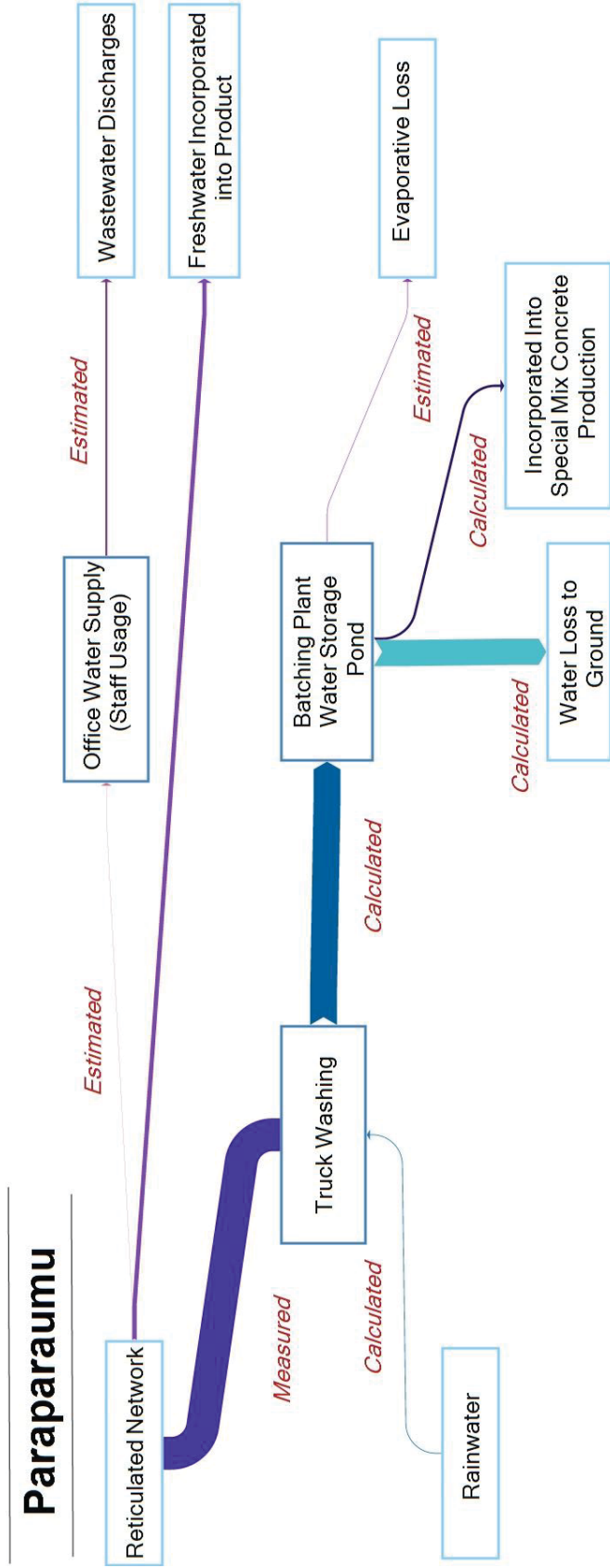


Figure A.29 Schematic diagram of freshwater withdrawals and consumption volumes at the Paraparaumu batching plant, for the 2013 period.

Ashby's Ready Mix

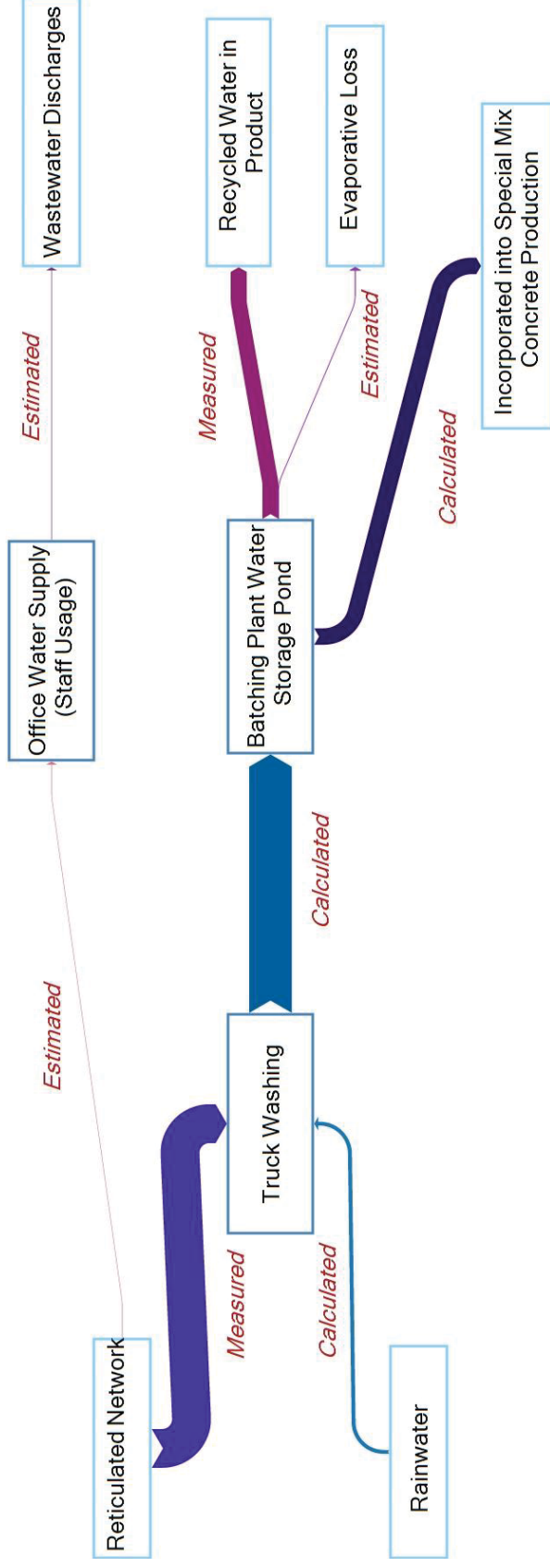


Figure A.30 Schematic diagram of freshwater withdrawals and consumption volumes at the Ashby's Ready Mix batching plant, for the 2013 period.