Waituna Catchment: Technical Information and Physiographic Application

Clint Rissmann, Lisa Pearson, Jessie Lindsay, Tapuwa Marapara, and Alexandra Badenhop

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Prepared by
Rissmann, C.[1], Pearson, L.[1], Lindsay, J.[1], Marapara, M.[1], and Badenhop, A.[2]

[1] Land and Water Science Ltd.
www.landwaterscience.co.nz
61c Leet Street
Invercargill, 9810
New Zealand

[2] e3 Scientific Ltd.
www.e3scientific.co.nz
11 Arrow Lane
Arrowtown, 9302
New Zealand

Corresponding Author
Dr Clint Rissmann
Contact number: 03 214 3003
Email: clint@landwatersci.net

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Story Map
The information contained in this report has been summarised in a web-based application. All maps have been provided over a base map of Southland, with main roads and land parcel boundaries to allow the user to easily locate areas of interest. Maps have an interactive component allowing the user to view maps at farm or catchment scale.

Access to the Story Map is through the following URL:
https://e3s.maps.arcgis.com/apps/MapJournal/index.html?appid=0c0fc1fa5afa423eb63d85bd9a1ec980

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Abstract

Water quality outcomes can vary spatially across the landscape, even when there are similar land use pressures. These differences are often the result of natural spatial variation in the landscape, which alters the composition of the water through coupled physical, chemical and biological processes. Living Water (a Fonterra and Department of Conservation partnership) commissioned a high-resolution physiographic assessment of the Waituna Catchment, Southland, to support water quality and biodiversity investment decisions for the catchment.

The physiographic approach is an integrated or 'systems view', predicated upon the spatial coupling between landscape attributes and the key processes governing water quality outcomes in surface and shallow groundwater. For example, the relationship between soil drainage class (*attribute*) and redox (*process*) can be used to predict soil denitrification potential. Unlike other mapping and classification approaches, the physiographic approach incorporates water quality, hydrochemical and/or hydrological response signals into a spatial format to identify, select, combine and classify those landscape gradients that drive variation in water quality outcomes. The key process-attribute layers identified for the Waituna Catchment are hydrology and redox.

Areas characterised by similar process-attribute classes for both hydrology and redox are defined as Physiographic Units (PGU). Each PGU responds in a similar fashion at the process level to broadly equivalent land use pressures. Through classification of the catchment into PGUs we demonstrate that: (i) physiographic mapping can be used to estimate the steady-state water composition of surface water and shallow unconfined groundwater with a high degree of confidence, and; (ii) process-attribute gradients and resultant PGUs are a powerful tool for informing and optimising efforts to improve water quality – matching efforts to the process level controls over water quality at the land parcel scale.

The key outputs of this report include:

- Exploration of the relationship between landscape attributes and processes.
- Development of a strong understanding of the processes controlling variation in water quality outcomes across the catchment.
- Production of four key process-attribute layers (A-PAL, H-PAL, R-PAL and W-PAL).
- Testing of the validity and performance of the redox (R-PAL) and hydrological (H-PAL) layers to estimate the main water quality contaminants across the catchment.
- Incorporation of a simple land use pressure layer to further refine estimations of spatial variation in water quality.
- Generation of numerical models for estimating each contaminant for receiving environments without water quality data.
- Combination of the R-PAL and H-PAL to produce a Physiographic Map of the inherent landscape controls over water quality.

1 Background

1.1 Project Overview

Living Water is a partnership between the Department of Conservation (DOC) and Fonterra to improve the natural habitats of five key catchments in significant dairying regions around New Zealand. The Waituna Catchment, Southland covers an area of 19,280 ha, in Southland, is one of the focus areas of the Living Water partnership due to increasing pressures on water quality and biodiversity. Living Water recognise the main environmental issues for the catchment are: (i) a significant loss of wetland, ecosystem integrity, and habitat for biodiversity; (ii) poor water quality caused by high levels of suspended sediment (S), nutrients (nitrogen (N) and phosphorus (P)), and microbial (M) contamination; and (iii) modified waterways, wetland and lagoon hydrology (Living Water, 2016).

One of the main goals of the Living Water partnership is to "*showcase examples of best practice sustainable dairy farming alongside thriving natural/native biodiversity*" (Living Water, 2016). The key focus areas are to "*work with farmers to protect, restore and reconnect fragmented wetlands; improve instream habitat and water quality; and support the uptake of best farm management practices*".

The Waituna Catchment forms part of the Awarua-Waituna wetland complex and has been recognised under the Ramsar Convention as a wetland of international importance since 1976 [1]. The Awarua-Waituna Wetlands is one of the largest (3,556 ha) remaining wetland complexes in New Zealand. It is important for its biodiversity and cultural values[2]. The Waituna catchment drains into the Waituna Lagoon, a brackish intermittently closed and open lagoon or lake (ICOLL), within the Waituna Wetland Scientific Reserve (Figure 1.1). The Waituna Lagoon is fed by Waituna, Moffat, and Carran Creeks. A tributary of Carran Creek, Craws Creek, is predominantly natural state and provides a good reference catchment for comparison with agriculturally land developed within a wetland setting.

Water quality monitoring within the catchment shows spatial variation in water quality outcomes in areas with broadly similar land use pressures (Rissmann et al., 2012; Rissmann and Hodson, 2013; Rekker and Wilson, 2016, Environment Southland, unpublished data). The effectiveness of farm management approaches to improve water quality will vary depending on the environmental properties of the land, such as hydrological pathways, soil type, and aquifer geology (Rissmann et al., 2016; Hughes et al., 2016; Monaghan, 2016). Therefore, a uniform approach to farm management across the catchment may not provide the desired benefits to water quality for the catchment. Understanding the environmental factors (*attributes*) that drive variation in water quality outcomes/risk is vital for sustainable primary production and enables the cost/benefits of water quality improvement controls to be targeted to areas where they have a higher chance of success (Larned et al. 2016; McDowell et al., 2017).

[1] http://www.wetlandtrust.org.nz/Site/Ramsar_Convention/Awarua_Waituna_Lagoon.ashx

[2] http://www.wetlandtrust.org.nz/Cache/Pictures/2280561/Waituna_Lagoon_Factsheet.pdf

Figure 1.1: Location of Waituna Catchment in Southland, New Zealand. Shading shows areas of subcatchments including the area of direct contribution to Waituna Lagoon.

1.2 Purpose of Work

The aim of this body of work is to identify and map the natural landscape gradients that determine variability in surface water and groundwater quality outcomes across the Waituna Catchment, and to provide a spatially and temporally refined platform for strategic investment in water quality. At the heart of this approach is an integrated or 'systems view' that requires an understanding of: (i) why the character of each of the main catchment compartments that water interacts with (i.e. surficial, soil and shallow aquifer) varies across the catchment; (ii) how the variable character of each compartment influences water quality via a range of fundamental processes; and (iii) the exchange and evolution of water composition as it moves between compartments (Rissmann et al., 2018).

The need for an integrated approach to inform catchment management for water quality reflects the fact that water is highly mobile, moving through and between compartments with each compartment leaving its own imprint upon the water. This approach recognises that the net result of any water quality outcome (e.g. the health of Waituna Lagoon), is seldom the result of a single process. Rather it acknowledges that the drivers of variability in water quality are complex and cannot be well understood via an isolationist perspective (e.g. consideration of a single compartment (e.g. soil) and/or a single process (e.g. hydrological processes)). For these reasons and out of respect for the community of the Waituna Catchment, this work seeks to provide an integrated platform that can be used to guide and inform investment in farm management, ecological diversity and mitigations in a manner that recognises and honours the inherent complexity of the Waituna Catchment.

Finally, the authors of this work recognise that no map, no measurement nor any conceptual knowledge is ever perfect nor will this understanding always align exactly with the fine-grained knowledge of individual property owners across the catchment. However, despite the imperfect nature of this work, it is hoped that this work will go some way to providing a meaningful representation of the natural variability that supports the aspirations of the community of the Waituna Catchment.

1.3 Report Structure

The Waituna Catchment is one of the most intensively studied catchments in New Zealand (including but not limited to Muirhead and Rutherford, 2007; Wilson, 2011; Burbury, 2012; Hamilton et al., 2012; Rissmann et al., 2012; McDowell et al., 2012; 2013; Muirhead, 2013; Waituna Lagoon Technical Group, 2013; McDowell and Monaghan, 2014; Tanner et al., 2013; 2014; Taylor et al., 2015; Rekker and Wilson, 2016; Guerin and Wourms, 2016). Rather than replicating the studies completed, this work builds upon and seeks to integrate the many layers of information currently available to obtain new insights over the nature of the controls governing water quality outcomes.

This report details background and technical information used for the application of the Physiographic methodology to the Waituna Catchment. The report is structured into 8 sections, as illustrated in Figure 1.2. Section 2 outlines the controls over water quality and is intended to provide the reader with background information to support the following chapters. The next three sections are about the landscape components and combine past research to provide an evolved understanding of the catchment: Section 3 details the inherent properties, such as geology and soils and how they influence water quality. Section 4 provides catchment scale resolution of the hydrological pathways, which is further refined in Section 5 which focuses on the farm scale hydrological pathways. This knowledge is integrated using the physiographic approach to produce process-attribute layers, which explain the key processes controlling water composition and quality (Section 6). Once combined they identify Physiographic or Fundamental Landscape Units – areas of similar water quality controls which can be used to provide targeted advice for farm planning (Section 8). A numerical model and validation are developed in Section 7. The model enables the estimation of each agricultural contaminant for receiving environments without water quality data.

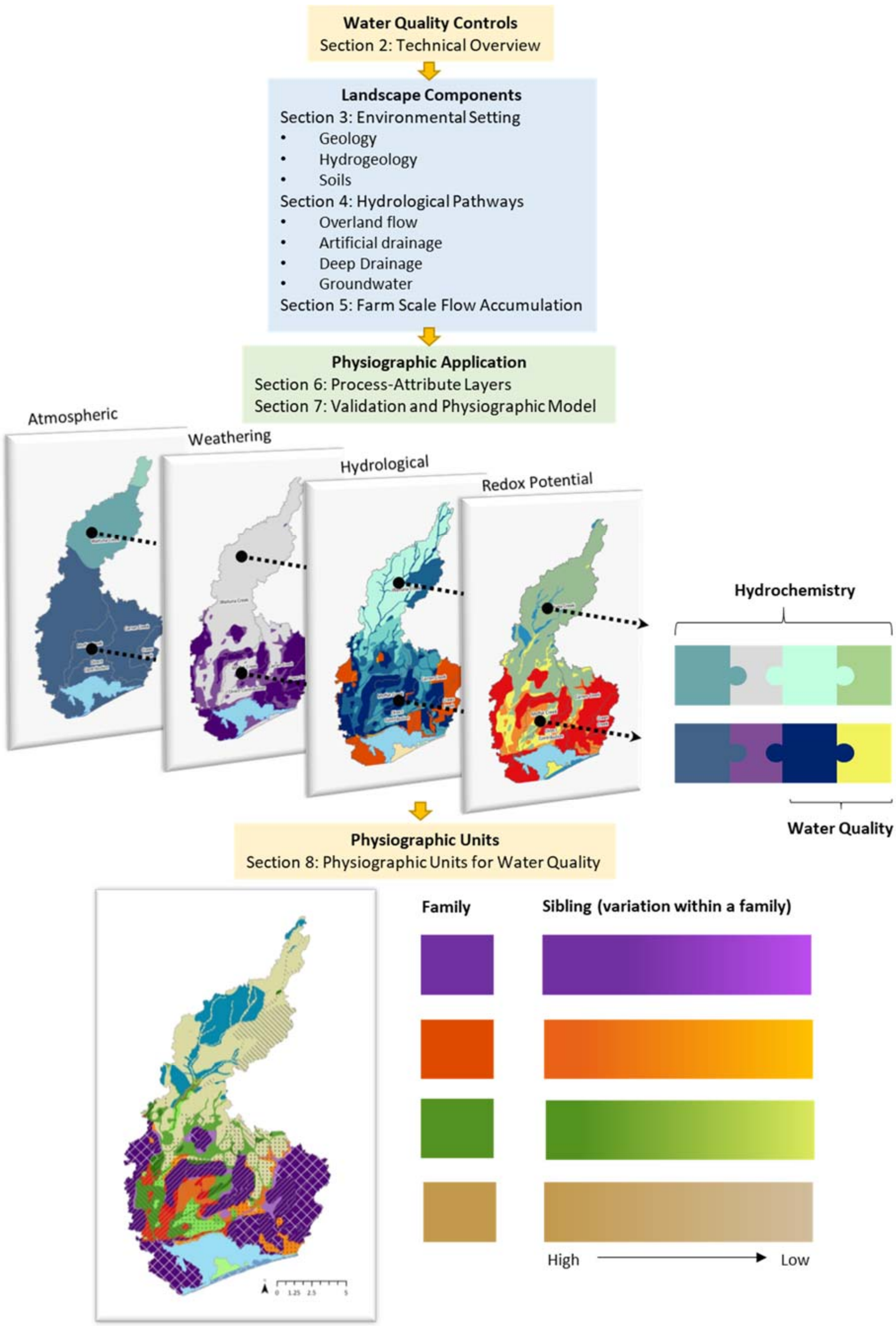

Figure 1.2: Overview of report structure.

2 Technical Overview of Water Quality Controls

2.1 Introduction

The key controls over variability in water quality outcomes across the Waituna Catchment are associated with both natural and anthropogenic features. The inherent natural properties of a landscape are important as they are often responsible for a significant degree of variation in water composition and quality, both in space and in time. Inherent properties are defined as natural topography, geology, hydrology and soil composition and associated relationships with water and land use activities. Importantly, the character of these inherent properties of a catchment also determines the degree to which they require modification for land use.

Modification of the inherent properties for land use is often restricted to the shallow surface of the earth, mainly vegetative clearance and modification of the drainage characteristics of the soil zone as well as the sinuosity, length and depth of river channels and streams.

Other modifications are associated with agricultural practices that work the soil and add animal manures and chemicals to the land surface in order to improve vegetative yields and productivity. Below the soil zone, the physical modification of geological materials is typically minimal. However, modification of the surficial hydrological response modifies the hydrological response of shallow aquifer systems by increasing the lateral routing of water and or suppressing shallow groundwater tables.

Modification of the inherent properties of the natural landscape has been an essential component of economic and social wellbeing. Without these modifications, there would not have been economic growth and prosperity. However, there is a need to limit contaminant losses, particularly nitrogen (N), phosphorus (P), sediment (S) and microbes (such as *E. coli*, M) to waterways across the Waituna Catchment (McDowell et al., 2012; Rekker and Wilson, 2016). This section explores the relationship between land use losses and water quality within the Waituna Catchment. A deeper understanding of the controls over water quality outcomes is an important platform to further support a wide range of community and industry activities looking to reduce land use losses across the catchment.

2.2 Source versus Transport Limited Settings

Landscapes or areas with minimal excess contaminants are said to be source limited - meaning that there is little excess nutrient or contaminant available for transport by water moving across or through the different compartments of the landscape. These areas are commonly associated with conservation or natural state land that has not been modified for agricultural or urban land use. In such settings, nutrients are often tightly cycled and of low abundance (Rissmann, 2011; Rissmann, 2012 and references therein; Zaimes et al., 2008), and although naturalised microbes are present, the export of *E. coli* from natural state settings is typically lower than that associated with areas of intensive livestock farming (Moriarty et al., 2011). Under such settings, the variation in contaminant transport and contaminant flux with stream flow is expected to be minor.

Conversely, if a system has a large store of land use derived contaminants it is referred to as being transport limited - meaning there is a significant potential for excess contaminants to be transported by water to a connected water body. Areas associated with a high land use loading are commonly associated with intensive agriculture, industrial and urban activities. If a system is transport limited, the variability of contaminant flux with stream flow is expected to be large, reflecting the concentration of contaminants within the shallow portion of the soil zone (Rissmann et al. in prep; Rissmann et al. 2016; Martin et al. 2017).

Often, land use derived contaminants accumulate over the drier months and are variably exported during the wetter months in response to soil drainage and surficial runoff (overland flow) (Ledgard et al., 2006; Cameron et al., 2014; Monaghan, 2014). The seasonal cycle of contaminant accumulation over the warmer months and subsequent export over the cooler months is a common feature of most intensive land use in cool temperate humid climates, such as Southland (Monaghan et al., 2010; Goldsmith and Ryder, 2013; Rekker and Wilson, 2016). Exceptions to this general observation are associated with high-intensity loading of land use contaminants to the land surface during the wetter months of the year (e.g. winter grazing practices).

2.3 Hydrological Modification

In the Waituna Catchment, Southland, and much of New Zealand, hydrological modification of the landscape has often included clearance of vegetation, drainage of swamps, and installation of extensive artificial drainage. Artificial drainage includes surficial drains, mainly as open farm drains or ditches, or subsurface drainage as mole-pipe or tile, over the last c. 150 years (Poole, 1990; Gay Munro 2014; Pearson 2015a). Streams have also been straightened to improve drainage and stream flow, restricting flooding of agricultural and urban land.

Region-wide modification of natural drainage properties has resulted in a marked change in the hydrological response of river and stream networks across Southland (Rissmann et al., 2012; Ledgard, 2013; Moran et al., 2017). Furthermore, increased stocking rates and a recent shift to heavier animals associated with dairying is known to reduce soil permeability via compaction, pugging, and has increased the risk of surficial runoff (Greene et al., 1994; Drewry and Paton, 2000; McDowell, 2006; Ballinger, 2011, Pearson 2015b). Importantly, drainage modification has not been static over time, with phases of drainage modification associated with technological advances, increasing land values, and land use change (Poole, 1990; Ledgard, 2014; Pearson, 2016a). Accordingly, drainage modification has been cumulative resulting in a shift in the hydrological response of catchments characterised by a long history of changing land use.

A net result of the widespread modification is that water now moves much more rapidly from the land to waterways and stream flow is more peaked (flashy) than in the past. More peaked stream response drives an increase in stream power which is a measure of the rate of energy dissipation against the bed and banks of a river or stream per unit downstream length (Fetter, 2001). Increased stream power is implicated with increased stream bank and stream bed erosion and higher sediment loads to lowland rivers, lakes, lagoons, and estuaries (Fahey and Row, 1992; Hicks et al., 2000; Henshaw, 2009; Davies-Colley, 2013; Clapcott, 2014).

The typical response of streams to vegetation clearance using sediment flux as an example is shown in Figure 2.1 (Bierman and Montgomery, 2013). A change in the hydrological response due to vegetation clearance and artificial drainage drives a change in the boundary conditions that govern the hydrological response and associated fluvial sediment flux. This occurs as water leaves the modified landscape more rapidly, cultivation exposes bare earth of which a portion is entrained and transported, entraining soil and land use derived contaminants and transporting them to waterways. Wintering paddocks are another example of bare ground that will shed water more rapidly than an equivalent area of forest or wetland.

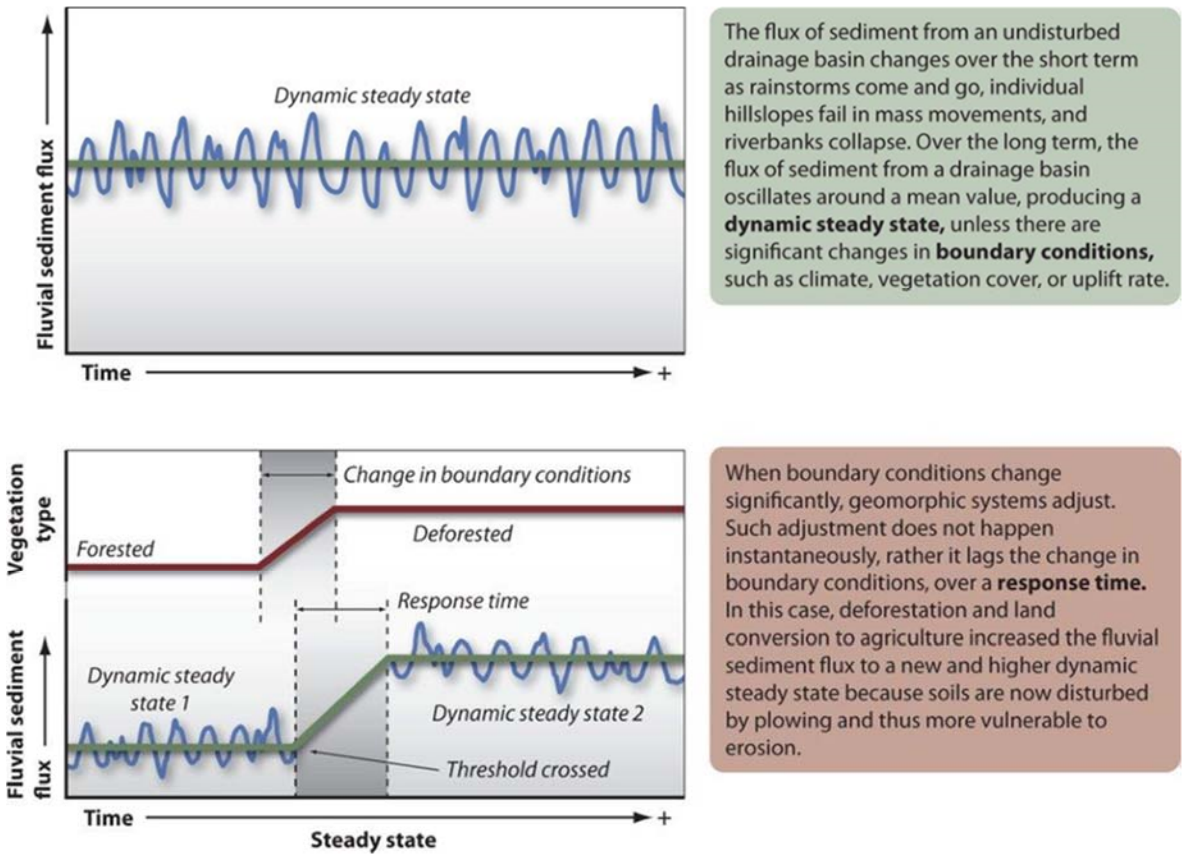

Figure 2.1: Conceptual control over enhanced sediment loss from transport limited settings. Where the bottom text box refers to ploughing, we redefine this as any activity that has significantly modified the hydrological response of the land surface (from Bierman and Montgomery, 2013).

Lateral soil zone flow (also known as interflow) commonly occurs at the interface between two soil horizons (layers) characterised by contrasting permeability (e.g., rapid or moderately permeable top soils overlying a slowly permeable subsoil or pan). For example, for poorly drained soils the majority of soil zone drainage may move laterally, either via subsurface artificial drainage or along slowly permeable horizons within the soil zone, with a minor component percolating to depth (McLaren and Cameron, 1996; Monaghan, 2014). Artificial drainage also increases the rate at which water is flushed through the soil zone, with the most rapid flow occurring via mole drains and/or macropores that intersect the tile drain network (Monaghan et al., 2016). Tile drains are widely recognised as one of the key pathways for soil zone solutes and contaminants to reach waterways (Cameron et al., 2014; Monaghan, 2014; Ledgard, 2014). Importantly, it is possible to estimate the likelihood and density of artificial drainage according to the unmodified hydrological characteristics of a soil (Pearson, 2015a). However, land use type and intensity likely play an important role over the density of drainage occurring, with heavier stock necessitating drier ground in order to limit treading damage as much as possible.

Studies in Pukemutu soil series by AgResearch indicate that up to 95% of drainage exits via mole-pipe drainage with <5% percolating to the underlying aquifer (Monaghan et al., 2016). Similar observations of dominance by lateral flow have been made at the Wallacetown Demo Farm (Cameron et al., 2014, Monaghan, 2014). Naturally, the propensity for lateral movement of soil water is enhanced across areas with subsurface artificial drainage and as such, has likely reduced the fraction of water draining to underlying aquifers. By comparison, for well-drained soils, especially those that are rapidly permeable, the majority of soil drainage percolates to depth and the shallow aquifer system before discharging as baseflow to streams (Monaghan, 2014; 2016). In reality, a hydrological continuum exists between exclusively deep vertical percolation and shallow lateral soil zone flux and accordingly, the pathway a contaminant takes will vary in according to variation in soil and geological materials and in response to modification (Figure 2.2) (Beyer et al. 2016; Rissmann et al. 2016). Therefore, it is important to consider the hydrological modification of a catchment when assessing the controls over water quality. This is especially true of the large developed wetland component of the Waituna Catchment, which has by far the highest density of artificial drains (both subsurface and open ditch).

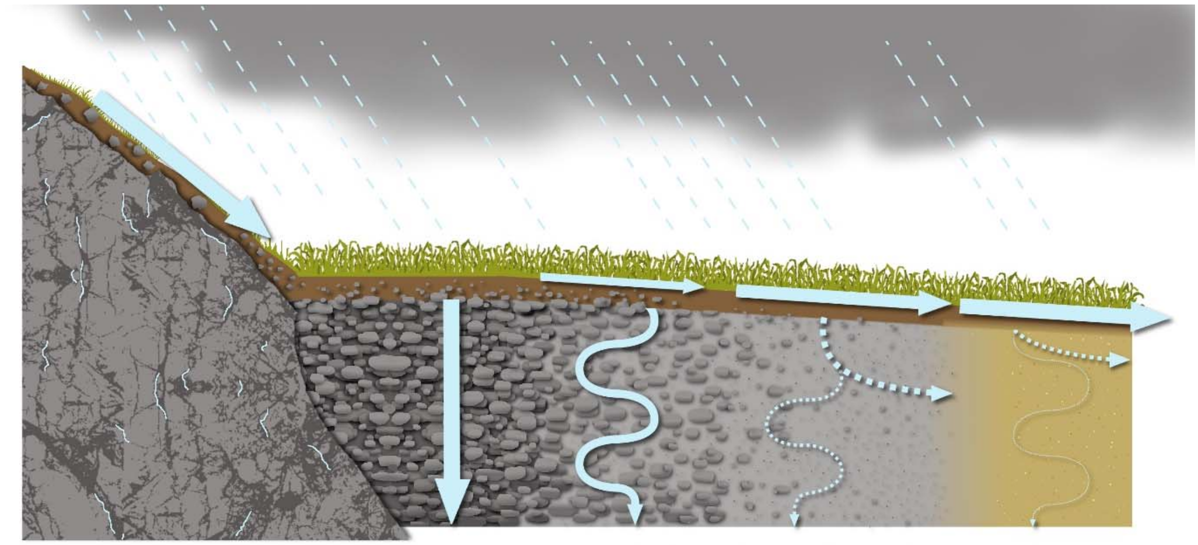

Figure 2.2: Conceptual diagram of gradients in soil properties which control the hydrological response ie. deep drainage, lateral flow and overland flow.

2.4 Chemical Modification

In order to sustain agricultural production, lime and fertilisers are added to soils to amend acidity and increase the availability of plant available nutrients and trace elements for livestock, respectively (Rogers et al., 2017). Direct deposition and irrigation of animal wastes onto soils also modifies the nutrient and contaminant content of a soil. Addition of lime raises the pH of the soil and is an important tool for managing the acidity and adding base cations, especially across developed wetland areas. Potassium (K), nitrogen (N), phosphorus (P) and a range of other amendments modify but do not necessarily change the inherent properties of a soil. Subsurface drainage also plays a critical role in modifying the biogeochemical environment within the soil zone by increasing the supply of oxygen to the subsoil (Brady and Weil, 2007; Schoonover and Crim, 2015).

As with artificial drainage, the degree to which the chemical composition of a soil is modified depends upon the inherent properties of the soil. For example, large amendments of lime are required in areas of naturally acidic soils, higher P inputs are required for soils with very high or very low P-retention and trace element additions are also dependent on soil mineralogy and weathering status (McLaren and Cameron, 1996). Modification of the chemical environment of the soil in conjunction with the introduction of different plant and animal species has also changed the biological composition of the soil environment (Doran, 1979). However, as with artificial drainage, the type and intensity of land use is likely an additional driver over the chemical and biological modification of the soil. Figure 2.3 shows the land use and land cover change from a natural state (*c.*

1000 years ago) to 2015 in the Waituna Catchment (adapted from Pearson and Couldrey, 2016). Along with land use change, the number of stock within the catchment and region has also intensified (Ledgard, 2013; Pearson and Couldrey, 2016).

In the 1960s, the national Land Use Capability (LUC) classification system identified the south of the Waituna Catchment as Class 4-8 land, identifying the severe limitations to arable and pastoral farming (DSIR, 1968). This land, considered marginal under the LUC due to the severe wetness, rooting zone limitations, and climatic limitations, was further developed through the support of government subsidies in the late 1970s which aided the development of the catchment through additional drainage and chemical modification[3]. If this assessment was to be undertaken today, the classes identified in Waituna Catchment would be significantly different, with much of the limitations reduced in severity by hydrological and chemical modification.

[3] These included the Livestock Incentive scheme (1977) which encouraged farmers to carry more stock, the Land Development Encouragement Loans (1978) made cheap loans available to develop unproductive land, and the Supplementary Minimum Price scheme (1978) guaranteed farmers price stability for their products, despite declining international prices.

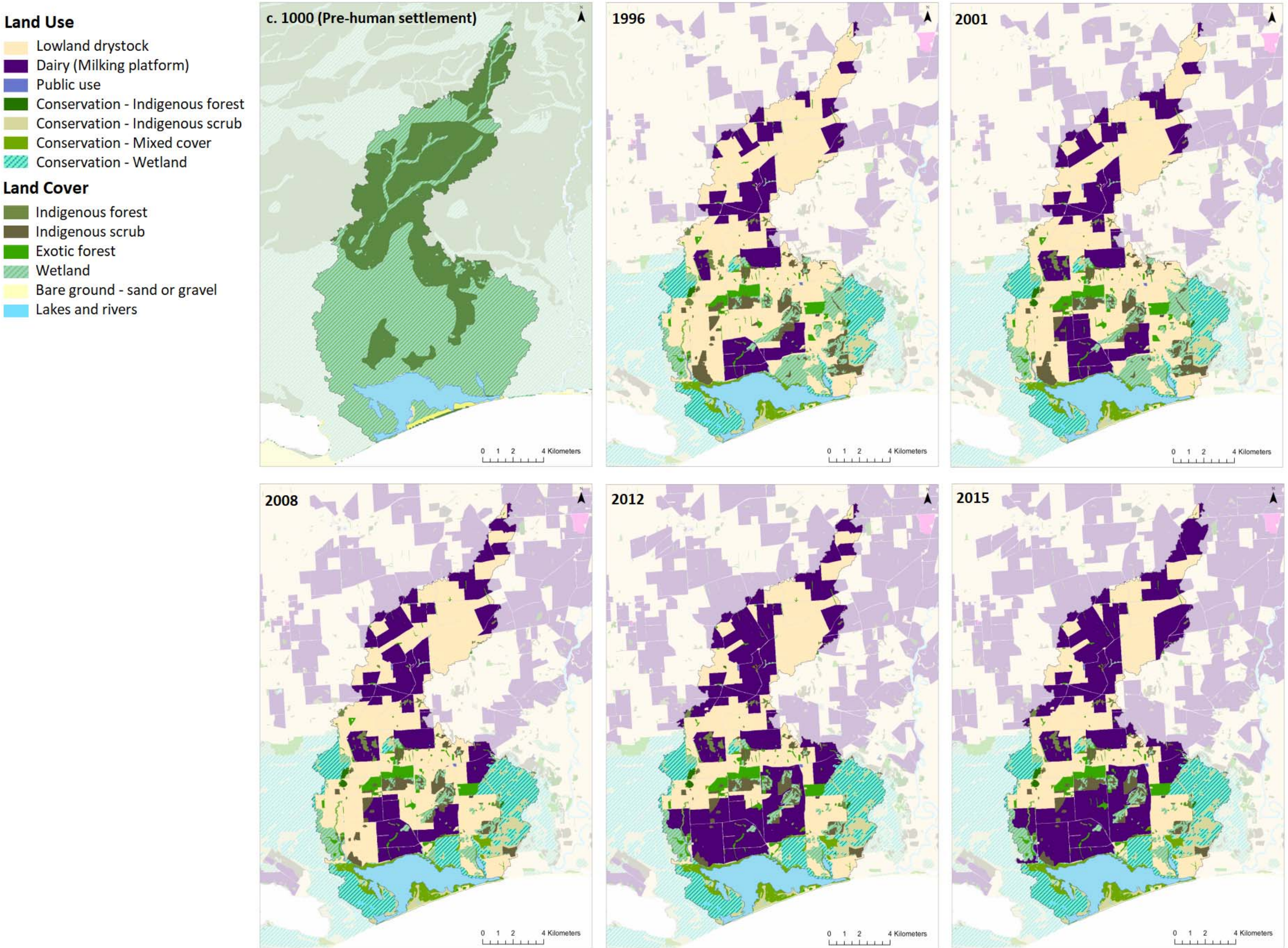

Figure 2.3: Land use and land cover change in the Waituna catchment from a natural state (c. 1000) to 2015. Figure is adapted from data produced by Pearson and Couldrey (2016) and references within).

2.5 Distribution and Transport of Land Use Contaminants with Depth

Most commonly, land use derived nutrients and contaminants are concentrated in the upper layers of the soil (<600 – 1,000 mm) and decline in concentration with depth (McLaren and Cameron, 1996; Rissmann et al., 2016; Martin et al. 2017), with only the most mobile contaminants transported to deeper levels or exported via subsurface drainage. The mobility and persistence of a nutrient or contaminant varies according to the inherent properties of the soil and/or aquifer, the degree of modification of the hydrological setting and the intensity and type of land use (Fraterrigo and Downing, 2008; King et al., 2005). Accordingly, the behaviour of land use derived contaminants varies between areas comprised of different assemblages of soil and geological materials (Rissmann et al. 2016).

Under specific conditions, accumulated contaminants may migrate below the upper layers of the soil (root zone), percolating through the subsoil. Once below the root zone a contaminant may then be transported laterally or percolate vertically to depth until it reaches the shallow aquifer. The fraction of a contaminant that moves laterally versus the component that percolates to depth is strongly dependent upon soil and unsaturated zone hydrological properties and can vary significantly (Nimmo et al., 2002; Wilson and Shokri, 2015). For example, for poorly drained soils a significant proportion of drainage moves laterally either via subsurface artificial drainage or along slowly permeable horizons within the soil zone.

Horizon permeable or lateral flow is also known as interflow. With well-drained soils, especially those that are rapidly permeable, a greater proportion of soil drainage will percolate to depth and the shallow aquifer system. In reality, a hydrological continuum exists between exclusively deep vertical percolation and shallow lateral soil zone flux. Accordingly, the pathway a contaminant takes will vary in response to natural variation in soil and geological materials and in response to anthropogenic modification of the inherent properties of soil, geology and topography (Beyer et al. 2016; Rissmann et al. 2016).

Not surprisingly, artificial modification of the drainage properties of soils is largely dictated by the initial hydrological properties of the soil (Pearson, 2015a). Specifically, very poorly drained and slowly permeable soils are commonly associated with the highest density of artificial drainage. Whereas well drained and rapidly permeable soils (i.e. recent gravel soils) have little if any artificial drainage.

In terms of contaminant transport, P-mobility is highly variable and strongly dependent on the properties of the soil and aquifer. For example, P is not typically mobile in mineral soils that contain abundant oxides and oxyhydroxides of iron (Fe) and/or aluminium (Al) (referred to as oxides hereafter). Such soils are said to have high anion exchange capacity or P-retention. In the absence of iron or aluminium oxides, P can be highly mobile. Simmonds (2016) reported very high levels of P leaching under winter grazing of virgin peat soils within the Waituna catchment (within the zone of direct contribution) and noted the dissemination of P throughout the peat soil profile.

In summary, most land use contaminants are concentrated at the or near the surface of the soil and decline in concentration with depth, reflecting the important and highly effective role of soil and unsaturated zone materials in storing and variably attenuating contaminants. However, the mobility and persistence of a nutrient or contaminant varies according to the inherent properties of the soil and/or aquifer and the degree of modification of the hydrological setting for a given land use pressure. Therefore, it is important to recognise the different behaviour of land use derived contaminants between areas comprised of different assemblages of soil and geological materials.

2.6 Attenuation Mechanisms and Transport

A significant proportion of land use derived contaminants that are not exported via the farm production cycle are naturally attenuated by processes occurring with soil and/or aquifer materials. Attenuation mechanisms include, but are not limited to, physical filtering out of contaminants, electrostatic attraction, chemical sorption, precipitation and redox reactions. The degree to which attenuation occurs is dependent upon both the inherent and modified properties of the soil and aquifer. In highly complex geological settings, such as the Waituna Catchment, the type and degree of attenuation varies sharply across the catchment.

Despite the spatial variation, the degree to which land use contaminants are attenuated typically increases with residence time within the soil or shallow aquifer system. The deeper water percolates to depth, the greater the degree of removal of land use derived contaminants. This is one of the key reasons the majority of land use contaminants are concentrated in the upper layers of the soil zone. Large molecules, particulate matter, and particle-reactive contaminants are often retained within the upper 300 – 600 mm of the soil (Martin et al. 2017). As water percolates down through the soil the concentration of land use derived contaminants generally declines in response to a number of key physical, chemical, and biological processes. As a general rule, thick soil and unsaturated zones have a greater capacity to store and variably attenuate/assimilate land use contaminants.

Under a particular range of conditions, several key land use contaminants can be mobilised beyond the soil zone and discharge to surface and ground waters. As noted above, P is one of these species, which can be highly mobile in soils and aquifer materials with low abundances of oxides of Fe and/or Al, such as organic soils and peat aquifers. Organic soils and peat aquifers, comprised of organic matter, have a low capacity to retain P (Rissmann et al. 2012; McDowell and Monaghan, 2015; McDowell et al. 2015). If P leaches to a strongly reducing (low oxygen) peat aquifer it commonly forms colloids that are negatively charged, small (1 – 100 nm), and highly mobile (Vande Voort et al., 2013; Wolthoorn et al., 2004; Gangloff et al., 2016). The high mobility of P-colloids means that they are readily transported through soils and aquifers to streams or discharge directly to connected surface water bodies.

A key property of colloids is that they remain in suspension so that gravity forces do not cause settling or precipitation. Aggregation or colloid neutralisation and precipitation occurs when colloids transported by freshwater mix with brackish water high in salts[4]. Seawater is a key example where the high concentration of positively charged salts causes colloids to aggregate and subsequently precipitate out of solution. Importantly, viruses, bacteria, algae and organic matter are primarily in colloidal form or at the very least behave as colloids. Recognising the role of colloidal formation and transport in natural systems is therefore critically important to understanding variation in contaminant transport. As observed elsewhere, colloid neutralisation is likely a key process driving the precipitation of P colloids within the brackish waters of the Waituna Lagoon (Gunnars and Blomqvist, 1997; Simmonds, 2016).

Another key contaminant of waterways is nitrogen. Nitrogen exists in various forms including dissolved inorganic, dissolved organic, and undissolved (particulate) forms. The dissolved inorganic forms include ammonia and ammonium nitrogen (NH_3 and NH_4^+, respectively), nitrite (NO_2^-) and nitrate (NO_3^-). Of these nitrogen species, NO_2^- and NO_3^- are mobile under most natural settings which is why they are often problematic to manage. Of these two species NO_2^- commonly occurs in very low concentrations relative to NO_3^-. The high mobility of NO_3^- reflects its negative charge and large hydrated radius which precludes it from being retained by positively charged sites on clays and/or

[4] Especially in di- and tri-valent cations such as Ca^{2+}, Al^{3+}, Mg^{2+}

organic matter[5]. Further, a significant proportion of soil and aquifer materials are also negatively charged, resulting in weak electrostatic repulsive forces that further enhance the mobility of NO_3^-.

Despite being highly mobile, NO_3^- is susceptible to denitrification under low oxygen conditions within soil or aquifer systems (Rissmann, 2011). Denitrification is mediated by naturally occurring bacteria in soil and or aquifer materials and is the only truly assimilative process that functions via the conversion of nitrate to gaseous species (i.e., NOx and N_2) which do not readily participate in nutrient cycling within water or subsoil environments. High rates of denitrification can, therefore, remove most if not all leached nitrate.

Optimal conditions for nitrate removal are associated with soils that are poorly drained and/or aquifers that contain a high content of organic carbon (Rissmann, 2011; Rissmann et al., 2012; Clough et al., 1998; Tanner et al., 2015; Beyer et al., 2016). For most imperfect or poorly drained soils, denitrification rates are lowest in the topsoil and typically increase with depth (Rissmann et al. 2016b, and c; Luo et al., 1998; Ernstsen, 2006). However, subsurface drainage can reduce the capacity of soil and bacteria to denitrify by increasing oxygenation and/or nitrate is exported at a rate that exceeds the metabolic capacity of the microbial biomass to denitrify (Brady and Weil, 2007; Tanner et al., 2014; Schoonover and Crim, 2015). High nitrate transfer rates are associated with periods of peak drainage when hydraulic loading rates at the land surface are very high (Cameron et al., 2014; Monaghan, 2014; Wells et al., 2016). Well-drained soils are often poor denitrifiers and are associated with high nitrate leaching losses in areas of intensive land use (Ledgard et al., 2006; Monaghan, 2014; Wells et al., 2016).

Ammonia (NH_3) and ammonium (NH_4^+) are important chemical species in natural waters, and especially across areas of intensive land use including urban and rural. Ammonia is both highly soluble in water and highly toxic to aquatic species. Large quantities of ammonia are produced by animal wastes and during the hydrolysis of fertilisers such as urea. However, ammonia is rapidly converted to the ammonium ion (NH_4^+) under the pH range typical of most soils and natural waters (Figure 2.4).

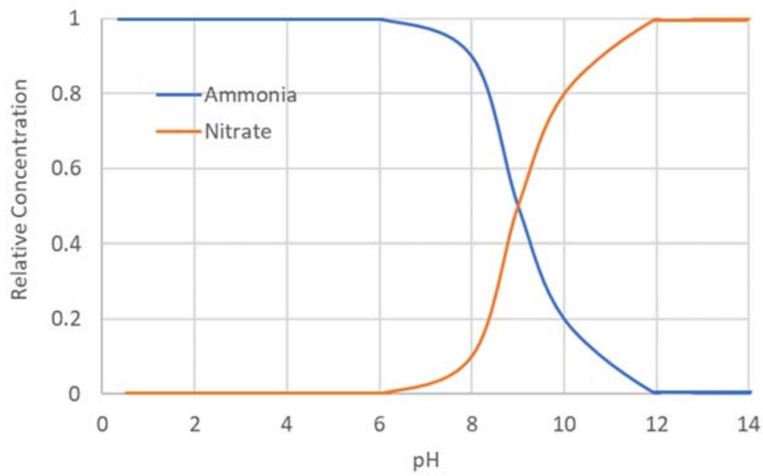

Figure 2.4: Nitrogen form under pH range.

Ammonium is less toxic and far less mobile due to a strong attraction to negatively charged particles by this positively charged ion and is eventually converted to mobile nitrate under oxidising

[5] charge is pH dependent.

conditions. However, due to its affinity for sediment, ammonium can be mobilised in significant quantities during surficial runoff.

Ammonium is also produced naturally in reducing, carbon-rich aquifers with concentrations as high as 3.5 mg/L NH_4-N measured for a deep lignite aquifer within the Waituna Catchment (unpublished Environment Southland Data). Reduced peat aquifers also typically contain elevated NH_4-N e.g., 0.2 – 1.0 mg/L) due to natural sources. However, Total Nitrogen (TN) concentrations for reduced aquifers systems are five times lower than across the northern portion of the catchment, where oxidised forms of nitrogen dominate (Rissmann et al., 2012).

Organic forms of nitrogen are often associated with animal urine and manures (urea), urea fertiliser and organic matter bound N, both dissolved and particulate. As with ammonia and ammonium, the majority of organic nitrogen forms are eventually oxidised to nitrate and either taken up by algae or plant biomass or removed via denitrification. Particulate nitrogen is often highest under surficial runoff conditions, due to the mobilisation of nitrogen-rich animal wastes and partially decomposed plant matter with small C:N ratios.

Of relevance to the Waituna catchment is the different behaviour of N and P species according to soil and geological materials. Specifically, aquifers and soils containing a high organic carbon content have a large capacity to remove nitrate via denitrification but a low capacity to retain and store P. Conversely, soils and aquifer materials formed in alluvial and aeolian sediments have a high mineral content and have a high capacity to retain and store P, and with the exception of poorly drained soils, are poor denitrifiers. If nitrate is lost from the root zone in a well-drained soil it can migrate rapidly to depth and may accumulate in the shallow aquifer before draining to surface water.

With regards to microbial attenuation, harmful microbes are removed via physical exclusion by fine textured materials in soil and aquifers (i.e. fine sand, silt, clay) and via sorption or electrostatic attraction onto soil or aquifer materials (i.e., poorly ordered and structural clays). It is uncommon to find *E. coli* in groundwater due to the high efficacy of the soil and unsaturated zone in microbial removal. However, soils that have high void ratios and/or macropores provide a pathway for microbial organisms to bypass the soil matrix and discharge straight to aquifers (Rissmann and Lovett, 2016; Hughes et al., 2016 Central Plains Physiographic zone) or surface water bodies via tile drainage (Hughes et al., 2016; Central Plains). There is some thought that fibrous peat may be less effective at excluding microbes due to a high void ratio, and yet there is currently little information to directly support this hypothesis.

Sediment is attenuated by physical exclusion, although it is also known to infiltrate and clog/fill voids in highly permeable soil and rock. However, as with any contaminant it can still be transported via macropores and subsurface artificial drainage with sediment discharge from tiles recorded at Tussock Creek (Monaghan, 2014). However, by far the most common process transporting sediment from land to waterways is overland flow (OLF) (e.g., surficial runoff). Overland flow is most often associated with flood events driven by high intensity or long periods of precipitation that results in saturation of the soil zone. Under these conditions, infiltration of precipitation is limited resulting in runoff across the land surface accumulating in swales or running off in sheets in areas of sloping land. As the water flows across the land it can entrain a significant proportion of land use contaminants and transport them directly to waterways. Ammonium, bacteria, viruses and both organic and inorganic forms of P have a high affinity for sediment and are commonly carried to stream during the entrainment of sediment during surficial runoff events.

The process of OLF is why streams and rivers in flood often look dirty or brown in colour. Streams from both pristine natural state, urban and intensively farmed rural areas often show the same discolouration at high flows. However, as natural state environments are commonly source limited there are few if any land use derived contaminants to be entrained and mobilised by OLF and as such the water quality impacts are typically negligible.

Of the various pathways water takes, surficial runoff associated with the surface and upper 150 – 300 mm of the soil zone is often the source of the majority of contaminants to surface water bodies (Rissmann et al., in prep). Surficial runoff is favoured in areas with steep or rolling topography that also host soils that are slowly permeable and/or poorly drained. Areas with naturally elevated water tables are also at risk of OLF due to limited subsurface volume for water storage. Where this type of land is associated with intensive land use, especially wintering of livestock, the losses to waterways can be large (Simard et al., 2000; Monaghan et al., 2016). Importantly, one or even a handful of surficial runoff events may transmit the majority of land use derived contaminants to surface water bodies and for this reason, it is important to evaluate how OLF susceptibility varies across a catchment when looking to reduce land use losses. By comparison, the risk of surficial runoff is lower across areas where soils are rapidly permeable (≥72 mm/hr) and free draining, with flat (<2º) topography. However, one or even a handful of surficial runoff events may transmit the majority (~90%) of land use derived contaminants to surface water bodies (Smith and Monaghan, 2003; Curran Cournane, 2011; Orchiston et al., 2013) and for this reason it is important to evaluate the susceptibility of an area to surficial runoff (Pearson, 2015b).

Due to the nature of surficial runoff, mobilisation can include scouring of the soil surface and subsequent entrainment of soil, animal wastes and fertiliser. Consequently, all four critical contaminants are transported to surface water waters during a runoff event. By comparison, the discharge of land use derived contaminants via subsurface drainage typically contains a lower load of contaminants due to the filtering and storage capacity of the overlying soil. The same is true of baseflow, which is the groundwater component directly discharged from an aquifer to a stream. By the time water has percolated to an aquifer the majority of land use derived contaminants, especially microbes and sediment, have been removed, resulting in relatively pure water. However, under specific settings both N and P can be highly mobile and readily transported through the topsoil, subsoil and/or underlying aquifers. Once in an aquifer, nitrogen and phosphorus may be exported to a connected surface water body as baseflow, thereby transmitting land use losses to the surface water network.

Over the last 150 years, artificial drainage and stream channel straightening have in places dramatically increased the power of water to entrain and transport contaminants across and through the landscape as well as via the hydrological network. Increased drainage rates are also responsible for an increase in stream power and higher rates of stream bank erosion. Although channel migration is a natural process, many studies demonstrate that levels of stream bank erosion have dramatically increased in response to both clearance of indigenous vegetation for agricultural and urban land use and artificial drainage (Fahey and Row, 1992; Hicks et al., 2000; Henshaw, 2009; Davies-Colley, 2013; Rissmann et al., 2017). Speeding up the rate at which water leaves the landscape also increases to varying degrees the flushing of the soil zone and associated transport of land use contaminants.

The net result of landscape modification for intensive land use has been an increase in the amount of sediment transported downgradient to lakes, lagoons and estuaries in areas of intensive land use since European colonisation of New Zealand (Thoms, 1981; Ledgard, 2013; Trompetter et al., 2014; Swales et al., 2012, 2015; Gibbs et al., 2015). Significantly, over the same time period as vegetation clearance and drainage, soil nutrient levels have also increased along with stock rates so that eroded sediment now contains higher concentrations of N and P but also microbes that are subsequently transported to surface waterways (McDowell et al., 2009; Swales et al., 2015). Recognition of the pathways and mechanisms that govern variation in land use losses is fundamental to understanding how and why water quality outcomes vary in both space and time.

2.7 Spatial Variation in Water Quality

For catchments with steep spatial gradients in soil and geological properties, the variation in water quality outcomes for the same or similar land use can be large. Various authors report that variation in landscape attributes account for more than two times the variation in water quality outcomes than land use alone (Johnson et al.,1997; Dow et al., 2005; Shiels, 2010; Becker et al., 2014). This reflects the observation that spatial variation in landscape attributes (e.g. soil texture, drainage, geology) play a critical role in determining the type(s) of pathway water takes and the potential attenuation along such a given pathway of agricultural contaminants. The complexity of the geological, soil and hydrological environment within the Waituna Catchment is well recognised (McDowell et al., 2012; Rissmann et al., 2012; Rekker and Wilson, 2016).

Application of Q-mode factor analysis[6] to water compositional data for 16 monitoring sites within the Waituna Catchment revealed greater variation in water composition between sites than within, reflecting a high degree of heterogeneity in catchment attributes. At the individual site level, flow is the dominant control over variation in water composition (Rissmann et al. in prep.). Mapping of the spatial controls over variation in water quality outcomes is a key focus of this report and forms the bulk of Sections 3-6.

2.8 Temporal Variation in Water Quality

It is well recognised that water composition varies with flow and that stream waters can at any one time be a mix of shallow groundwater discharge, soil water and/or surficial runoff. Here we define the surficial zone as the land surface and the upper 150 – 300 mm of the soil ('O' or 'A' horizons); whereas the soil zone is defined as the depth of pedogenically differentiated material occurring below the root zone (~300 mm).

Depending on landform age, pedogenic differentiation has been observed to extend to depths ≥3 m across large areas of the Southland Plains (Topoclimate South, 2001; Crops for Southland, 2002). If the water table is deeper than 3 m below ground level (m BGL), pedogenically undifferentiated sediment or rock may occur between the soil zone and the groundwater table. This zone of sediment or rock is commonly referred to as the 'C' horizon. However, across the Waituna catchment, the shallow nature of the water table (median = 2.4 m BGL) means that soils are often in direct contact with the shallow aquifer system[7].

Critically, of the three main compartments that contribute to flow all three are seldom active at once. Rather, drainage from each compartment occurs in response to seasonal climatic cycles and lower frequency high-intensity precipitation events (Figure 2.3). For example, soil drainage varies according to soil moisture status, increasing or decreasing in response to evapotranspiration and the magnitude of precipitation events. When soils are wet, tile drains are often flowing, contributing water to the stream network. When soils dry up in response to warmer weather and higher rates of evapotranspiration the flow of soil water decreases and/or stops.

Surficial runoff transports water, solutes, and particulates from the land surface and upper 150 – 300 mm of the soil zone to a surface water body (Winter et al., 1998; Inamar, 2011). Contaminants occurring at shallow levels generally have had insufficient time to migrate deeper into the soil zone where a range of beneficial process aid in the retention and variable attenuation of contaminants.

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[6] On log transformed and z-scored hydrochemical and water quality data.

[7] In areas with shallow water tables, the groundwater water table may rise up in response to recharge and subsequently discharge through the subsoil to discharge via artificial drainage. Rising groundwater is particularly relevant for the wetland area of the catchment and likely to be less prevalent in the north of the catchment.

For this reason, surficial runoff commonly delivers the largest load of land use derived contaminants directly to stream (Monaghan et al., 2010; Preston et al., 2011; Khatri and Tyagi, 2015).

As with soil water drainage, surficial runoff switches on and off in response to evapotranspiration and the magnitude of precipitation. Surficial runoff events are of two general types, infiltration excess and saturation excess. Infiltration excess occurs when the intensity of precipitation exceeds the infiltration rate of the soil causing runoff prior to infiltrating the soil (Horton, 1941). Infiltration excess derived surficial runoff is exacerbated when soils become compacted by heavy stock or machinery or when the soil surface becomes hydrophobic in response to extended dry periods (Drewry and Paton, 2000; Goldsmith and Ryder, 2013; Monaghan et al., 2016). In urban areas, surficial runoff is common due to the sealing of the land surface by asphalt and concrete. In the summer months when soils are drier and plants are transpiring, precipitation intensity has to be high to drive surficial runoff. However, during the cooler months when evapotranspiration rates are low and soils approach or are at saturation, surficial runoff can be caused by even low-intensity precipitation (Srinivasan et al., 2002). Surficial runoff due to soil saturation is referred to as saturation excess overland flow and is more common than infiltration excess across the Waituna Catchment (Section 4.2).

The flux of water, solutes, and particulates from an aquifer to a stream are termed the baseflow component of stream flow. Surface water baseflow is often dominant during the warmer months when soils are dry and reflects the gradual outflow of water stored in aquifers that underlie the soil zone (Fetter, 2001). The baseflow index for a stream is a hydrological measure of the total volume of water contributed to a stream from an aquifer(s). Importantly, the relative contribution of groundwater to a stream varies strongly according to the hydrogeological setting, with relatively low groundwater inputs associated with areas of poorly drained soils and low aquifer storage (Environment Southland, unpublished data). Various publications have noted that the volume of groundwater contributing to stream increases as the area of well-drained soils and alluvial aquifers increase (Inamdar, 2011 and references therein; Inamdar et al., 2013). In some wetland settings, groundwater contributions to streamflow over the hydrological year can be negligible, reflecting the poor drainage status of soils and limited aquifer storage (Inamdar, 2011 and references therein). In these systems, soil water drainage, especially where the density of artificial subsurface drainage is high, may dominate the volumetric flux of water (Inamdar, 2011 and references therein).

As soils wet up and dry, the volume of water and the flux of solutes and particulate matter varies within the stream reflecting the variable contributions from the three key compartments (Rissmann et al, in prep). Of these compartments, surficial runoff occurs at the lowest frequency and yet generates the greatest volumetric flux of water and contaminants. In transport limited settings, surficial runoff may be responsible for the bulk (c. 90%) of the nutrient, contaminant sediment and microbial load to surface waters (Smith and Monaghan, 2003; Goldsmith and Ryder, 2013; Orchiston et al., 2013; Curran Cournane et al., 2011; McKergow et al., 2007).

In Southland, soil drainage typically occurs over an extended period during the cooler months when evapotranspiration rates are lowest (Smith and Monaghan, 2003; McDowell et al. 2005; Monaghan et al, 2016). During periods of soil drainage, tiles flow and aquifers are recharged by percolating soil water. As soils dry up over the warmer months of the year in response to increased evapotranspiration, soil drainage declines and may stop entirely. Under these conditions, baseflow from aquifers underlying the soil zone is the main source of water, solutes and particulates to stream.

Importantly, the composition of each compartment strongly influences the type and magnitude of solute and particulate flux. For example, a stream receiving drainage from a peat aquifer will show a different compositional signature to a stream receiving drainage from an aquifer formed in gravel, sand, silt and clay even for a similar land use intensity (Rissmann et al. 2016).

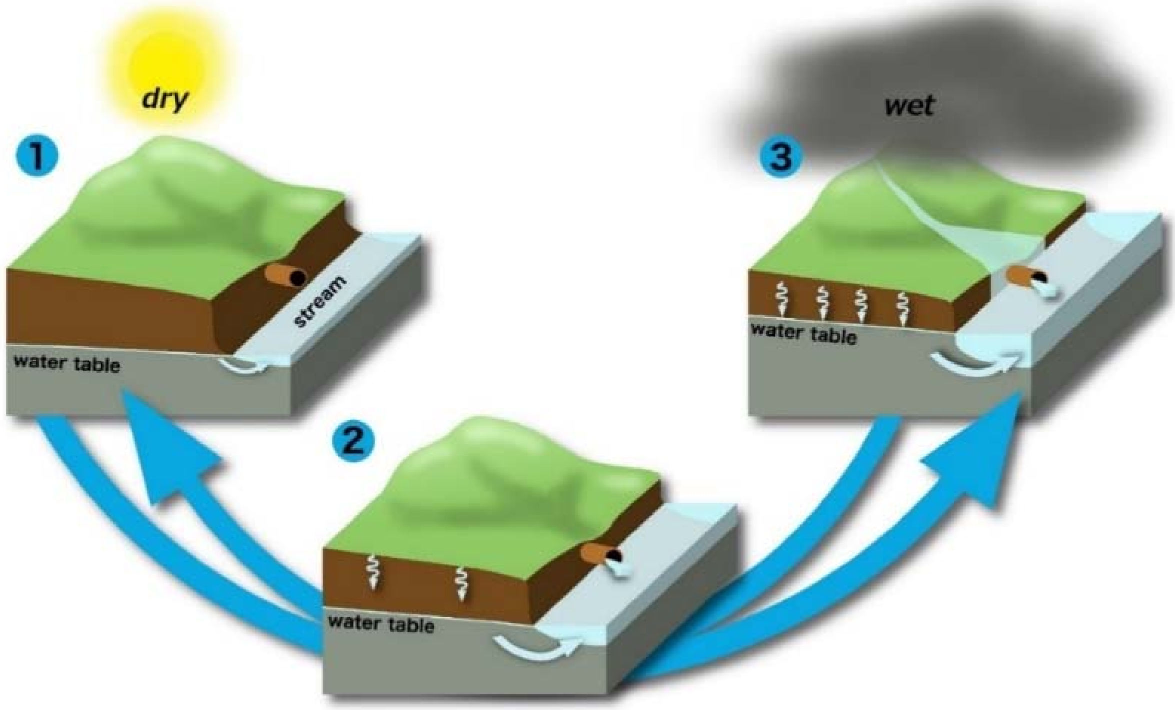


Figure 2.5: Conceptual diagram of temporal variation in hydrological pathways to streams.

3 Environmental Setting

3.1 Introduction

The inherent natural properties of a landscape, such as the topography, geology and soils constrain much of the hydrological and geochemical variation observed in the water chemistry. In this section, the environmental setting of the Waituna Catchment is described, including the relevance to water quality. The landscape properties that are described in this section are integrated for the development of Physiographic Units (see Section 6).

3.2 Topography

The topography of the Waituna Catchment is predominantly undulating (<10° slope) with an elevation change of 72 m from the north of the catchment to sea level in the south (Figure 3.1). Figure 3.1 was produced using LiDAR imagery at 1 m resolution. Areas of undulating topographic relief are predominantly located to the north of Waituna Creek and Carran sub-catchment, whilst Moffat Creek, Craws Creek sub-catchments and the area of direct contribution are predominantly flat. Topographic breaks occur in conjunction with paleoshorelines (discussed further in Section 3.2.1). The stream network follows a dendritic pattern with incised channels predominantly in the north of the catchment.

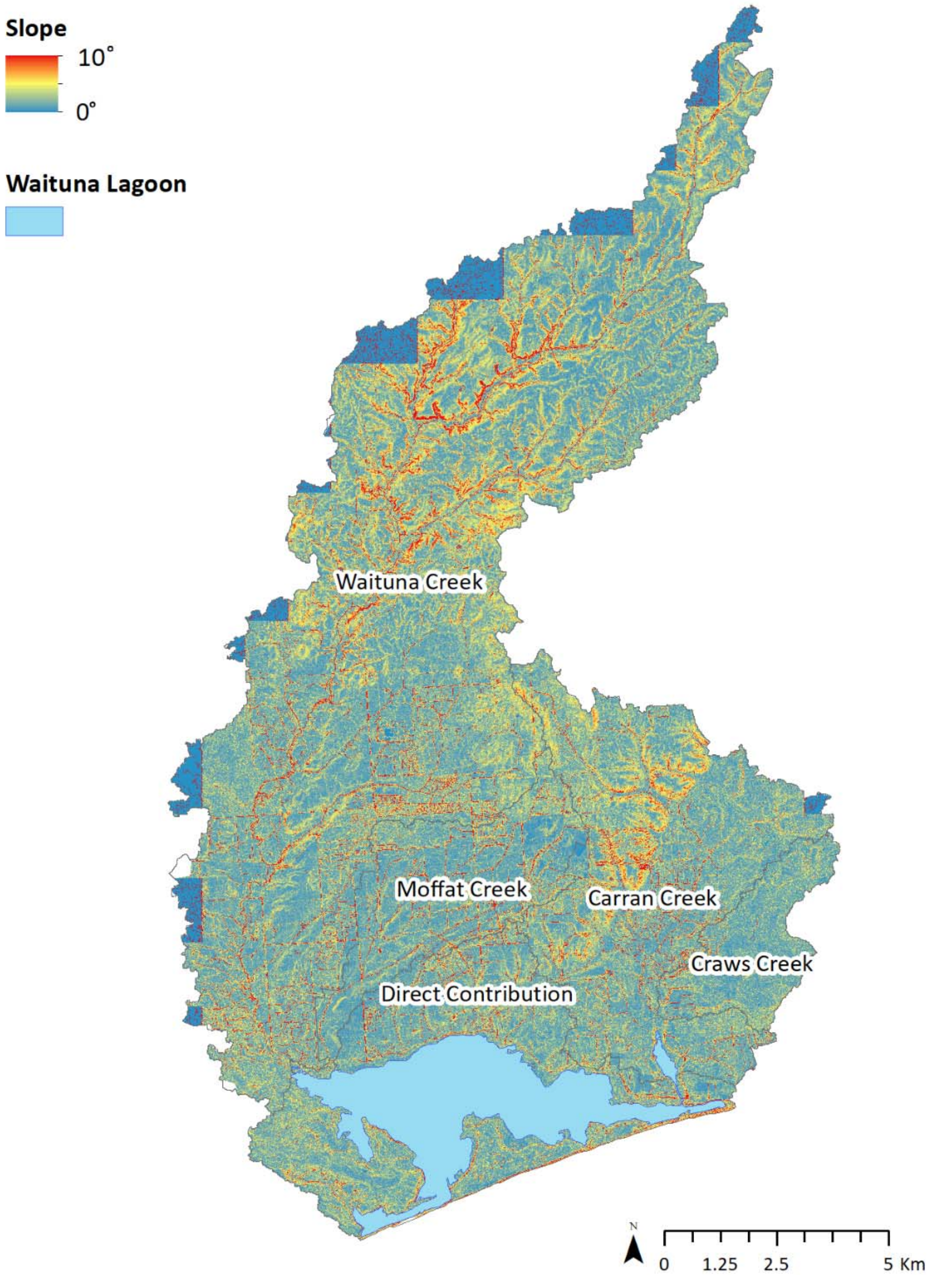


Figure 3.1: Topography of the Waituna catchment developed from 1 m resolution LIDAR data. Areas without LiDAR coverage around the catchment boundary have poor slope definition.

3.3 Geology

The geology of the Waituna Catchment may be subdivided into three general stratigraphic layers: (i) unconsolidated alluvial, marine terrace and peat wetland deposits of Quaternary age (0 – 1.8 million years) that constitute the modern day surface of the catchment; (ii) thick deposits (up to 250 m) of mixed terrestrial and marine sediments of the East Southland Group deposited during the Tertiary Period (>1.8 to 65 million years ago) and underlie the Quaternary deposits, and; (iii) much older basement rocks (170 – 260 million years) of the Brook Street and Murihiku Terranes that constitute the primary basement rock and underlie the Tertiary sediments of the East Southland Group (Figure 3.2; Wilson, 2011; Rissmann et al., 2012; Rekker and Wilson, 2016).

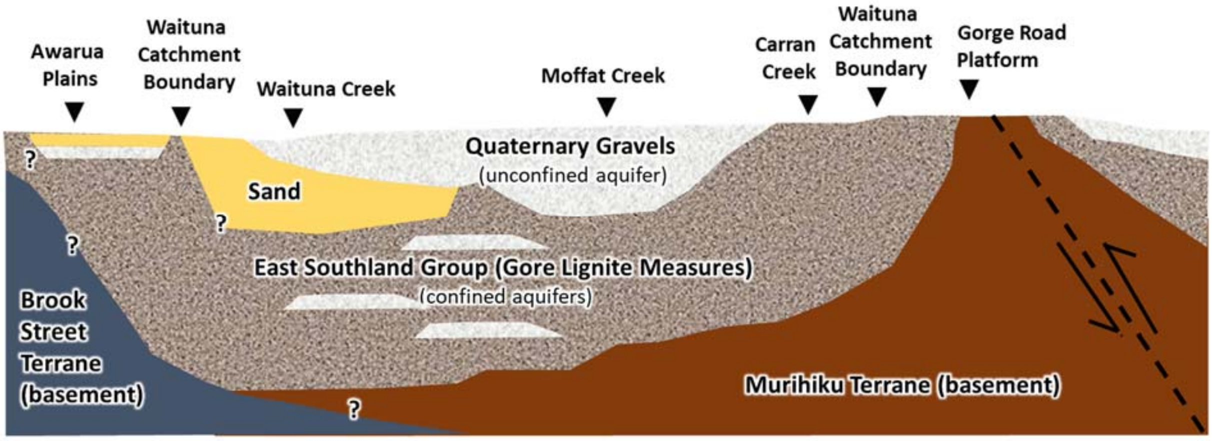


Figure 3.2: Schematic hydrogeological cross-section of the Waituna catchment from Awarua Bay towards Gorge Road (not to scale) (adapted from Wilson, 2011).

Quaternary deposits were laid down in response to a succession of glacial and interglacial cycles associated with the erosion and transport of sediments from the northern mountains to the lowland plains (Turnbull and Allibone, 2003). This included the deposition of the Q8 – Q10 and Q6 – Q8 aged alluvial terraces of the Kamahi Formation and reworked Waikiwi Terrace alluvium (Figure, 3.3). Both of these units are classified as being associated with moderately weathered clay-rich sand and greywacke (quartz) gravels (Turnbull and Allibone, 2003).

Later in the Quaternary, fluctuating sea level resulted in the shoreline migrating as far north as Caesar Road and as far south as Stewart Island (Turnbull and Allibone, 2003). The Q5 aged paleoshoreline separates the catchment into northern and southern Waituna (Figure 3.3). The trace of the paleoshoreline coincides with the southernmost extent of the Kamahi Formation alluvium, whereas the reworked Waikiwi Terrace alluvium extends above and below the Q5 paleoshoreline. The Waikiwi Terrace constitutes a significant fraction of the northernmost extent of the Carran Creek Catchment, and occurs as a remnant within the Moffat Creek Catchment, extending south of the paleoshoreline, slightly beyond Cook Road within the Waituna Creek Catchment.

As the shoreline retreated to its current position, it left behind a series of beach deposits comprised of quartz-rich, pebbly to bouldery gravel and sand with minor peat (Turnbull and Allibone, 2003). Subsequent fluvial avulsion resulted in the erosion of beach deposits leaving prominent remnants across the southern portion of the Moffat Creek Catchment, and as a minor unit within the Carran Creek Catchment. Ongoing stream channel avulsion drove the erosion and deposition of recent (Q1) alluvium across the lower third of the Waituna Catchment. Around the same time, a wetland complex developed in response to naturally high-water tables associated with the low elevation setting and poorly permeable basement rock (Turnbull and Allibone 2003; Wilson, 2011; Rissmann et al., 2012; Rekker and Wilson, 2016).

The organic deposits are mostly fibrous to woody peat, although amorphous peat is common at deeper levels (Rissmann et al. 2012). Where unmodified, peat bogs are well developed and actively growing with some deposits >15 m thick (Topoclimate South, 2001). Natural state peat bogs occur in the eastern part of Carran Creek (Craws Creek) and in places around the fringe of the lagoon associated with the area of direct contribution. Part of the Awarua Wetland borders the western boundary of the Waituna Creek catchment. The Morton Mains Fault has been observed to traverse the western edge of the Waituna Creek catchment; where Waituna Creek follows a similar trace to the fault there is no peat bog development in this area potentially due to greater permeability associated with fault-related fractures and permeable gravels.

In the east of Carran Creek, the GLM underlie the thin wetland deposits. Here the low permeability of the upper GLM (carbonaceous mudstone/sandstone alternating with gravels and sands) has led to saturated cover soils and the development of peat wetlands. Deeper within the 250 metre-thick GLM, drilling has encountered gravel/sand aquifers in between siltstone/sandstone aquitards which are often confined or slightly leaky (Rissmann et al. 2012; Rekker and Wilson, 2016).

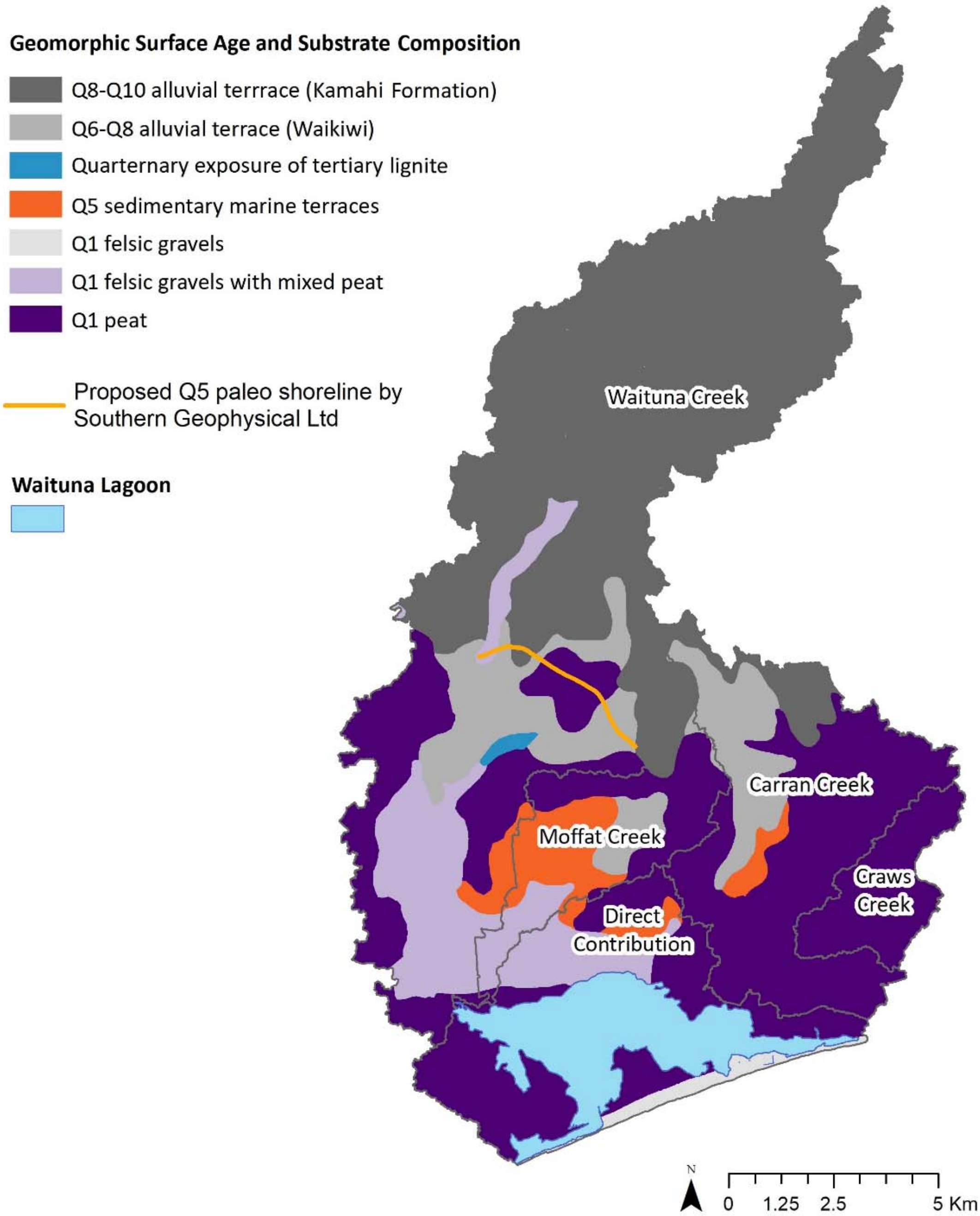


Figure 3.3: Geology of the Waituna catchment (adapted from QMap, Turnbull and Allibone, 2003). The Q5 paleoshoreline is identified as a yellow line, although the extent of the Kamahi Formation may be used as an approximation of the former shoreline.

3.3.1 Paleochannels

Paleochannels and the paleoshoreline may greatly influence the water response within a catchment, creating a preferential high conductivity pathway for recharge waters. The presence of paleochannels as conduits for groundwater recharge was raised in light of: (i) the rapid hydrological response of shallow aquifers to recharge across the northern portion of the catchment (Wilson, 2011; Environment Southland, unpublished; Burbery, 2012); (ii) the presence of poorly hydrochemically evolved groundwaters, and estimates of groundwater mean residence times less than one year and possibly as short as two months (see Section 3.3)(Burbery, 2012; Rissmann et al., 2012; Environment Southland, unpublished data, groundwater age estimates).

Subsequent geophysical investigations (Southern Geophysics Ltd, 2014), identified paleochannels of up to 200 m wide, where fluvial depositional features cut through older gravels and intersected the shallow aquifer. These channels are inferred to have been incised in the mid-Quaternary (Southern Geophysics Ltd, 2014), incising into the Kamahi Formation, as former stream channels. Subsequent alluvial and aeolian inputs resulting in these channels being buried beneath a thin (c. 0.3 to >1 m) accumulations of silts and sands.

Raw Ground Penetrating Radar (GPR) data was obtained from Southern Geophysics Ltd to determine whether the locations of the paleochannels could be further refined and presented in greater detail than points. GPR data was opened in Reflexw (version 7.5.9, Sandmeier Geophysical Research), a GPR data and seismic processing software. Lines from the GPR trace files that resembled channels were traced and exported as shapefiles. The shapefiles were loaded into ArcMap as a series of points. The points were overlaid on the Waituna DEM and elevation profiles across the GPR trace lines were examined to see if there were any land features that supported evidence of paleochannels. However, there was no firm correlation between surface topographic features and underlying paleochannels, due to masking by alluvial processes and aeolian deposition of silts and sands from the mid-Quaternary. The GPR lines exported from Reflexw could not depict the location of the paleochannels to a higher resolution than what has previously mapped by Southern Geophysical Ltd. The likely location of the paleoshoreline is shown in Figure 3.3.

3.4 Hydrogeology

The soils, surface water network, Waituna Lagoon and the limited aquifer resource of the Waituna Catchment are hosted primarily by unconsolidated sands and clay bound gravels of Quaternary age (Quaternary cover sequence). South of the paleo-shoreline, peat deposits and wetland soil types are also ubiquitous along with areas of marine deposited gravels of low permeability. The Tertiary sediments that underlie the Quaternary cover sequence are of low permeability and host a deeper confined aquifer system that shows some minor connectivity to the surface water network (Rissmann et al., 2012). Accordingly, variability in soil types, hydrological response, water composition and water quality outcomes are primarily determined by the composition of the Quaternary cover sequence.

3.4.1 Quaternary cover sequence: peat, sand, silt and clay bound gravel thickness and extent

Across the Waituna Catchment, the Quaternary cover sequence varies in thickness in response to topographic relief on the Tertiary and older basement and incision by stream channels. Although there have been a number of interpretations of the thickness of the Quaternary cover sequence preference has been given to that of Wilson (2011), since it accords with our local knowledge of catchment hydrogeology and local water table data. Specifically, Wilson (2011) observed that:

- The limited gravel thickness (generally less than 5 metres) of the northern third of the Waituna Catchment reflects the influence of the Gorge Road platform (where the Tertiary and basement sediments are present near the surface as shown in Figure 3.5).
- The gravel thickness appears to be deepest under the Moffat Creek catchment which may reflect a paleochannel or geological structure aligned in a southwest direction.
- Gravel thickness in the Waituna Creek catchment is variable, generally ranging from < 5 m thick in the upper part of the catchment, to between 10 - 20 m across the middle reaches before thinning again in the lower reaches to < 10 m (Figure 3.5).
- The limited thickness of Quaternary gravels present in the Carrans Creek catchment is likely to limit the extent of the unconfined aquifer in this area resulting in the formation of an extensive wetland complex.
- The catchment drainage, geology and possible paleochannels in the lignite measures and gravels are orientated in a predominately southwest direction. This does not exactly mirror surface topography and does not match the overall catchment boundary orientation, which is largely north-south. This indicates ground and surface water catchments may not entirely align and that catchment hydrology may be influenced by the underlying geology.

The depth of the Quaternary cover sequence was mapped using data from Environment Southland containing the depth of the tertiary contact (Gore Lignite Measures) in bore logs within the catchment and within 30 km of the catchment boundary (251 sites after QA/QC). The Tertiary contact depth was interpolated into a surface using an inverse distance weighted method in ArcGIS (Figure 3.4). Quaternary cover sequence depth across the catchment is presented in Figure 3.5 and summarised in Table 3.1. The average depth of the Quaternary cover sequence is 11.3 m BGL in the Waituna Catchment, with the shallowest cover across the centre of Waituna Creek Catchment associated with the Waikiwi Terrace, and to the south of Carran Creek Catchment, where there is significant peat development.

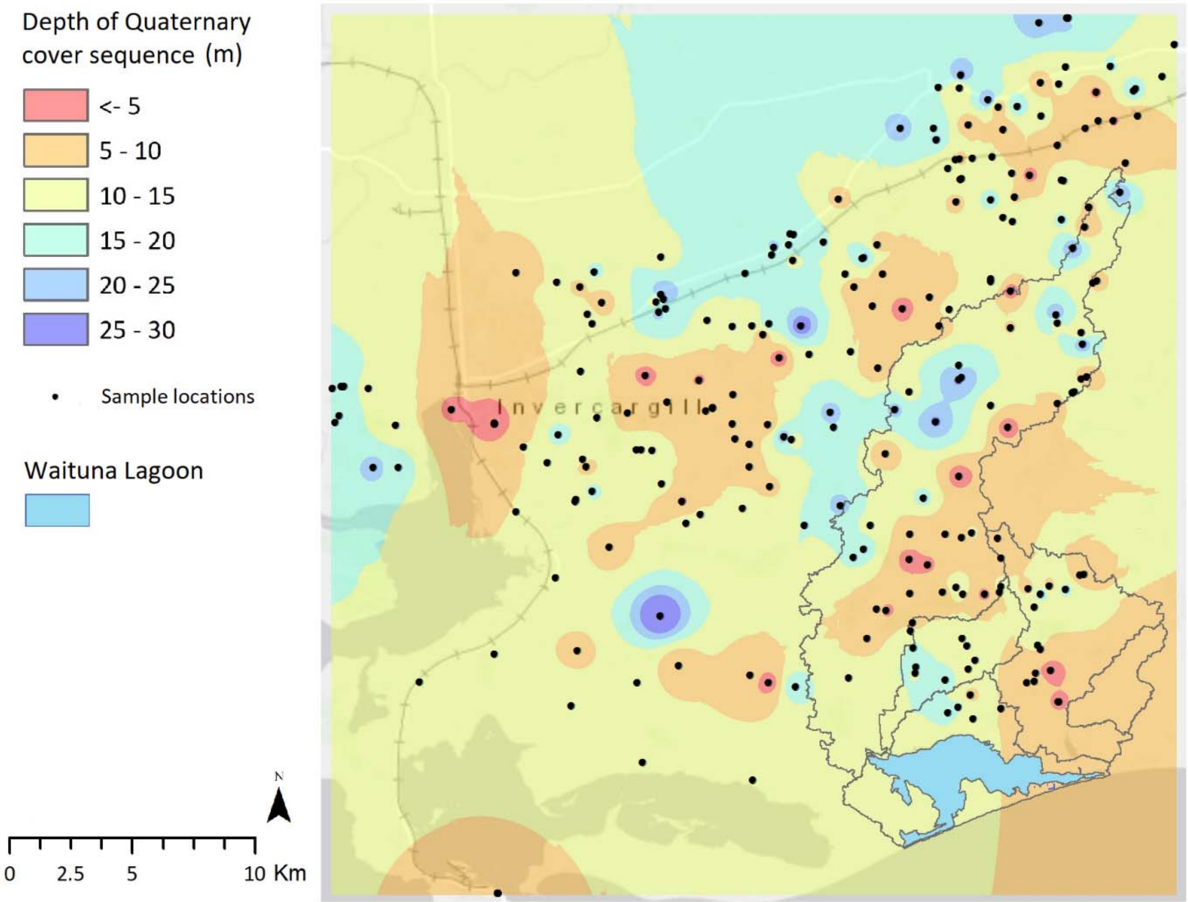


Figure 3.4: Bore locations used to produce an integrated surface showing the depth of Quaternary cover sequence in the Waituna Catchment.

Table 3.1: Depth of Quaternary cover sequence (meters BGL) by catchment.

	Waituna Creek	Moffat Creek	Carran Creek	Craws Creek	Direct Contribution	Waituna Catchment
Area (ha)	11143.2	1562.9	4215.3	828.4	2295.5	20616.5
Average (m BGL)	12.0	12.8	8.2	7.0	12.1	11.3
Minimum (m BGL)	2.6	7.1	1.5	5.0	6.5	1.5
Maximum (m BGL)	28.1	19.9	19.8	9.3	16.4	28.1
Range	25.5	12.8	18.2	4.3	9.9	26.5

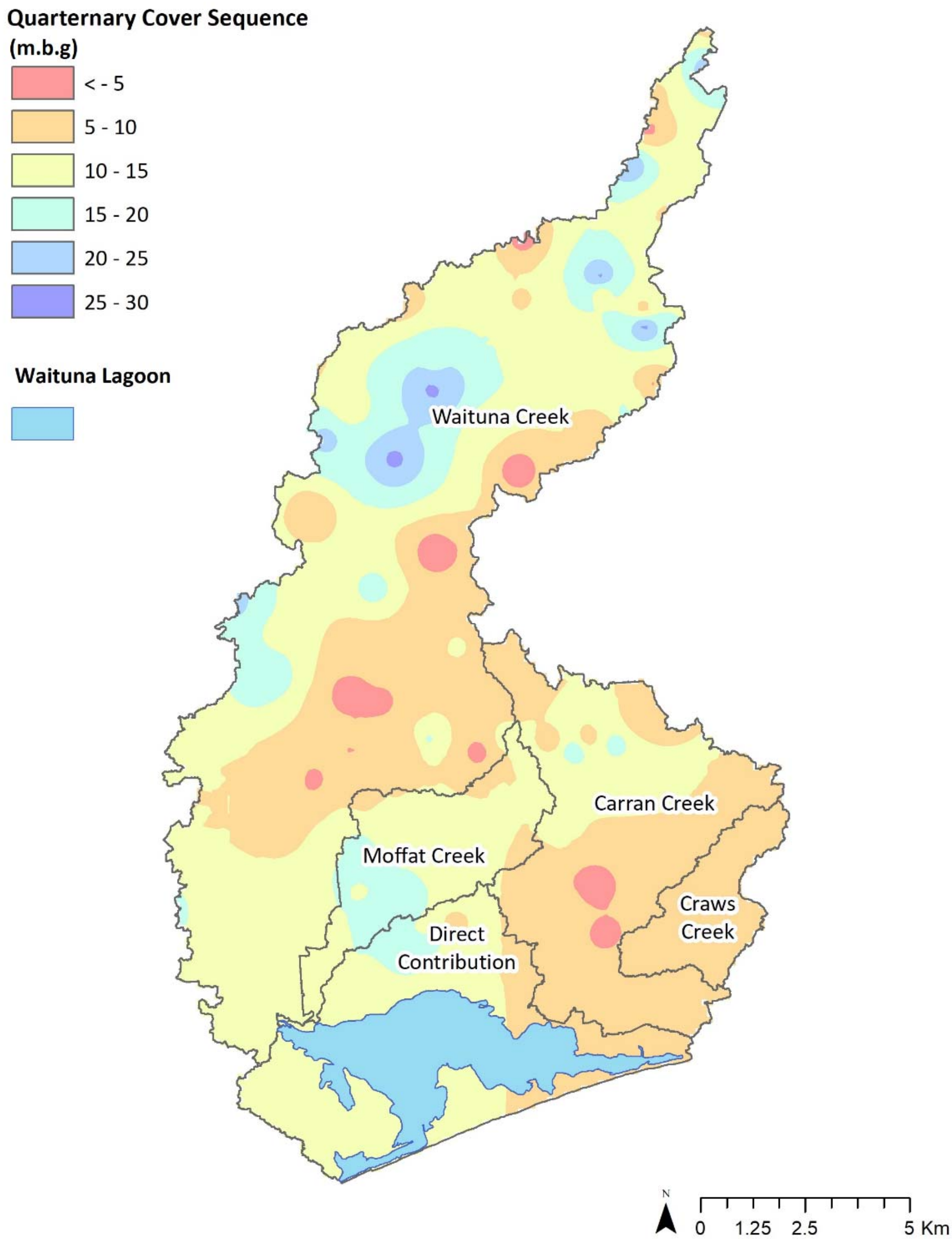


Figure 3.5: Interpolated Quaternary cover sequence depth (m BGL).

3.4.2 Water table

Water table across the Waituna Catchment varies according to the depth below the surface and permeability of both Quaternary gravels and basement rock (GLM and older basement terranes). Groundwater levels were provided by Environment Southland for 39 bores with location data within or near the Waituna Catchment. This data, along with median stream and lagoon level formed part of a regional static water level layer produced by Hughes (Environment Southland, unpublished). This analysis subsequently formed part of an assessment of regional unsaturated zone lag times (Wilson et al., 2014).

Mean and median standing water levels are taken from the interpolated layer and are presented in Figure 3.7. A summary of water table data for each sub-catchment is provided in Table 3.2, including estimated mean and median water table depth varying in a manner consistent with the hydrogeological setting. Immediately obvious is the overall shallow nature of the water table which is within 10 m BGL across the entire catchment with respective mean and median values of 2.75 and 2.24 m BGL (Figure 3.7).

In accordance with elevation and the thickness of the Quaternary cover sequence, water table depth is shallowest for areas south of the Q5 paleoshoreline. The median groundwater table is deepest for the Waituna Creek catchment and shallowest across the short and low lying Moffat Creek catchment. The zone of direct contribution, surrounding the Waituna Lagoon, has a median static water table of 1.84 m BGL. Natural state, Craws Creek has the least amount of variation with an estimated range of 0.9 m as is consistent with its relatively undisturbed state.

Table 3.2: Water table in meters below ground level (m BGL)) from the interpolated ground and surface water data (Hughes, unpublished; Environment Southland, 2015).

	Waituna Creek	Moffat Creek	Carran Creek	Craws Creek	Direct Contribution	Waituna Catchment
Mean	3.21	1.51	2.07	1.91	1.99	2.75
Median	2.56	1.29	1.96	1.92	1.84	2.24
Standard Deviation	2.20	0.85	0.62	0.13	0.68	1.93
Minimum	0.45	0.50	0.50	1.64	0.50	0.45
Maximum	9.79	4.75	3.97	2.54	4.45	9.79
Range	9.34	4.25	3.47	0.90	3.95	9.34

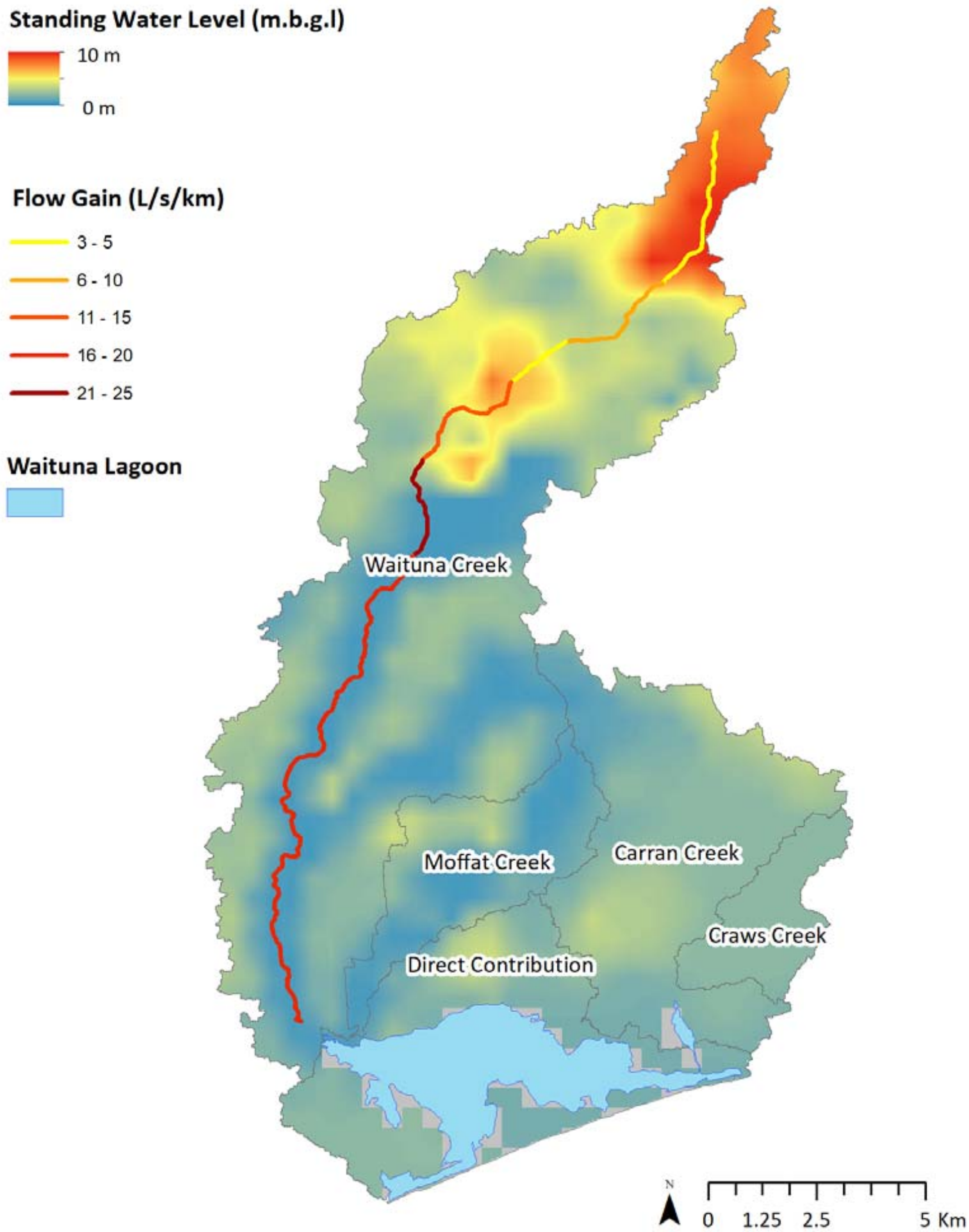


Figure 3.6: Water table depth for the Waituna Catchment (modified from Hughes, unpublished data; Environment Southland). Flow gain is shown for Waituna creek only (after Wilson, 2011).

3.4.3 Surface water-groundwater interaction

Baseflow to streams is a direct discharge of groundwater from a hydrologically connected aquifer. Streams may either gain water from or lose water to a local aquifer. Unless a stream runs dry, baseflow is always a component, albeit potentially a minor one, of the flow in a stream. Baseflow may deliver N and/or P loads depending on the unsaturated zone and aquifer composition.

All of the Waituna catchment is known to be gaining, however, there is a large variation in the longitudinal input of groundwater as baseflow into the system (Wilson, 2011; Rissmann et al., 2012; Rekker and Wilson, 2016). Environment Southland completed a gauging programme in Waituna Creek to characterise the surface water - groundwater interaction in 2011 (Wilson, 2011; Rissmann et al., 2012). The indicative flow gain is presented in Figure 3.7. Upstream of Drakes Hill Road, the average low flow gain is ~5 L/s/km, however, baseflow contribution increases significantly downstream of Waituna Road to an average low flow gain of ~20 L/s/km. Surface water-groundwater interaction has not been quantified in the Moffat Creek or Carran Creek catchments, however available gauging data from the mid to upper reaches of the catchments indicates combined flows are less than 20 L/s (Wilson, 2011; Rissmann et al., 2012).

3.4.4 Groundwater discharge to the lagoon

Direct groundwater discharge via seepage, occurs to the Waituna Lagoon and is focused around its margins but may also occur as seeps or springs within the inner lagoon (Wilson, 2011; Rissmann et al., 2012). Direct groundwater discharge is thought to be predominantly sourced from aquifers that fringe the lagoon and are hosted by the ‘zone of direct contribution’. Hydrogeological investigations indicated a higher degree of hydrological connectivity between western aquifers than those fringing the eastern portion of the lagoon (Wilson, 2011; Rissmann et al., 2012).

In June 2016, further assessment of the extent of direct groundwater discharge was undertaken using conductivity and temperature mapping at a time when the lagoon was open to the sea (Guerin and Wourms, 2016). Areas of direct groundwater discharge seepage were expected to present with lower electrical conductivity and higher temperatures (during winter) than brackish lagoon waters. Measurements were made along 14 x 100 m transects from the shoreline, avoiding areas of surface water inflow, focussed around the northern shoreline of the lagoon, with two transects on the western and northern facing shorelines and identified a significant area (3.22 Ha and 8.91 Ha respectively) of direct groundwater discharge along the north-western shoreline (Figure 3.8). Note that measurements were only taken within 100 m of the shoreline and that there may be other seepage areas that were outside of the extent of the survey. These findings of a significant area of direct groundwater discharge are consistent with earlier studies using benthic flux chambers to assess the location of direct groundwater discharge (Rissmann et al., 2012).

Six seepage samples were taken along the north-western shoreline, from within the zones of lowest electrical conductivity and highest temperatures, using a drive-point piezometer in June 2016. High permeability and low conductivity waters were primarily associated with gravel layers within what appeared to be variable layers of amorphous peat. Gravels were intersected at varying depths but water samples were preferentially taken from the shallowest permeable layer at each site (c. < 0.5 m BGL). Hydrochemical analysis of seepage waters was compared with samples from three bores near the lagoon, seawater, tributary inflows, and lagoon water. This work indicated variation seepage waters to be the product of mixing between local aquifers, predominantly peat type, and to varying degrees seawater (Guerin and Wourms, 2016). The mean concentration of N and P was found to be 2.05 mg/l and 0.62 mg/l respectively, which was consistent with groundwater bores sampled within the zone of direct contribution (Rissmann et al., 2012).

A simple mass balance to estimate the contribution of N and P to the lagoon via direct discharge was undertaken by multiplying the average nutrient concentrations in seepages (n=6), without correcting for dilution by seawater, by an estimated mean annual direct groundwater flux through the bed of the lagoon of 0.46 m^3/s (estimated using a mass balance model by Chris Jenkins, Senior Hydrologist, Environment Southland, pers. comm. in Guerin & Wourms, 2016). This preliminary load assessment was then contrasted against load estimates from the surface water network (Hamilton et al., 2012) - this data was not available when the initial nutrient guidelines were developed for the Waituna Lagoon (Waituna Lagoon Technical Group, 2013). The key findings include indicating a possible, albeit highly uncertain, total P flux of 8,411 t/yr from direct groundwater discharge or an additional P load of c. 50% to the lagoon that was previously unaccounted for. In terms of total ammoniacal nitrogen, the flux was estimated at 4,657 or 2.3% of the total N load (203,762 t/yr). However, it is important to note that this preliminary estimate of the P load associated with direct groundwater discharge is currently associated with a large degree of uncertainty. Specifically, better estimates of the mean annual groundwater flux and nutrient concentrations are required in order to better constrain the magnitude and subsequent importance of this pathway in terms of nutrient transfer from the zone of direct contribution to the lagoon.

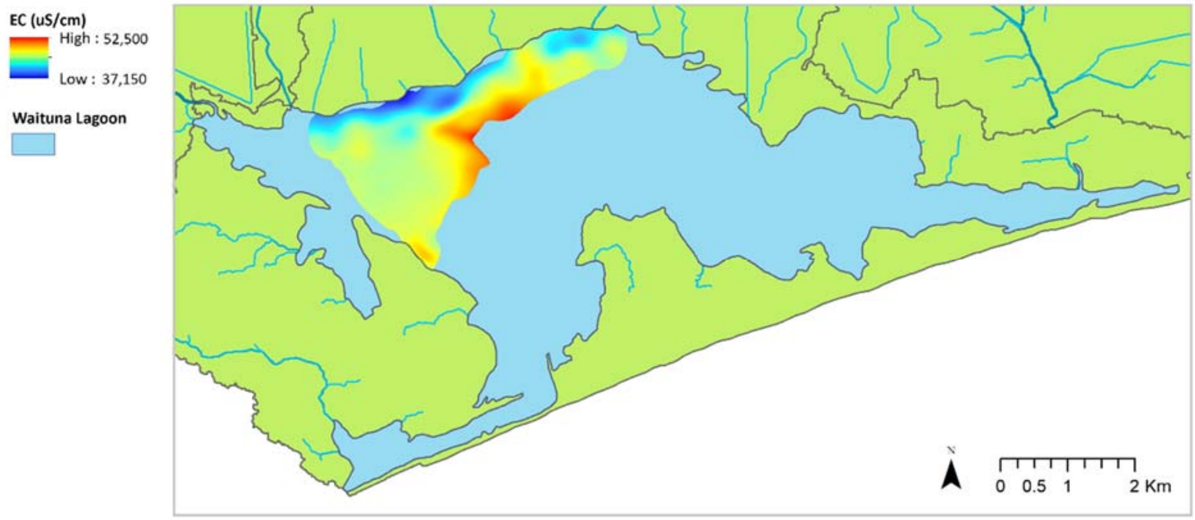


Figure 3.7: Electrical conductivity used to identify seepage zones of groundwater contribution (dark blue) directly to Waituna Lagoon (Guerin & Wourms, 2016). Red areas are associated with higher conductivity more typical of brackish lagoon waters. Transects avoided areas of surface water inflow.

3.5 Soils

Topoclimate South (2001) mapped, classified and described the soil properties for each soil type in Southland along with chemical analysis for each soil series. The survey classified soils according to the New Zealand Soil Classification, which groups soils by their physical and chemical properties into a four-tier hierarchical structure: Order > Group > Subgroup > Soil form. Soil types are also identified by a local series name. The soil mapping was undertaken at a 1:50,000 scale, which in some areas is not sufficient to delineate boundaries between individual soil series and results in polygons with multiple soils.

The soil orders within the Waituna catchment include Brown soils with minor Gley in the north of the catchment, and Gley, Podzol, and Organic soils in the south, with minor Recent soils close to the coast. The general properties of the soil orders are described in more detail below. Variability in soil orders reflects different parent materials, landform ages and controls over the water table.

3.5.1 Brown Soils

Brown soils are mineral soils named due to their dark grey-brown topsoils and brown or yellow-brown subsoils (formed by thin coatings of iron oxides weathered from the parent material). In the Waituna Catchment, Brown soils cover 6,130 ha (32% of the catchment) mostly to the north. They have developed on the high terraces of the southern plains (Kamahi and Waikiwi Terraces) in deep wind-deposited loess. Therefore, the resulting soils are deep with a high soil water holding capacity due to the deep loess parent material. The soil series present in the Waituna Catchment are Waituna (Typic Allophanic Brown), Mokotua (Mottled-acidic Orthic Brown) and Woodlands (Mottled Firm Brown) (Figure 3.9) (Topoclimate South, 2001). In addition, some minor Waikiwi, Firm Typic Brown soils can also be found within the catchment.

These soils have silt loam textures, rapid (Waituna) to moderate over slow permeability (Waikiwi, Mokatua, Woodlands) and are well- (Waituna, Waikiwi) to imperfectly- drained (Mokatua, Woodlands). Due to their relatively good internal drainage and low organic carbon content, soils in this zone have moderately low to low reduction potential. The oxidising nature of these soils is primarily a feature of good drainage but also partly reflects the high proportion of loess parent materials, derived from base poor (and electron donor poor) siliceous felsic rocks. P-retention in these soils is moderate to high minimising the risk of P-leaching.

As Brown soils are typically well-drained, artificial drainage densities are low to moderate depending on the extent of imperfectly drained Woodlands and Mokatua soils. Drainage is typically installed to improve the slow permeability of the subsoils.

3.5.2 Gley soils

Gley soils, along with Organic, represent the original extent of wetlands prior to agricultural development. In the Waituna Catchment, Gley soils cover an extent of 3,545 ha (18.4%) and are developed on the floodplains of small streams in fine alluvium from rewashed loess, or old marine terraces. The mineral soils are strongly affected by waterlogging, resulting in anoxic and reducing conditions producing soils with light grey subsoils, usually with reddish-brown mottles. The organic matter content in the topsoil is elevated reflecting their origin in historical wetlands. The soil series present in the Waituna Catchment are Titipua (Orthic Peaty Gley), Dacre (Recent, Acidic, Gley), Tisbury (Orthic Acidic Gley) and Jacobs (Recent Sandy Gley) (Figure 3.9) (Topoclimate South, 2001).

These soils all have horizons with slow permeabilities (< 4mm/hr) and are poorly drained. Poor aeration occurs when the soils are wet, which may be for most of the year in the absence of artificial drainage. This results in subsoils that are acidic (pH < 5.5), and have a moderately high reducing potential. The redoximorphic features of mottling and gleying are indicative of reducing conditions. P-retention in these soils is moderate, minimising the risk of P-leaching.

Artificial drainage is used extensively in these soils to prevent waterlogging, which occurs due to the combination of flat topography and poor soil drainage (Crops for Southland, 2002; Pearson 2015a). Bypass flow via artificial drainage can reduce soil residence time reducing the potential for denitrification to occur. In soil types that have restricted drainage, lateral flow may occur along slowly permeable layers within the soil profile. However, the spatial extent of lateral flow is limited by the artificial drainage network.

3.5.3 Podzol soils

Podzol soils are strongly acid mineral soils that typically have a bleached horizon immediately beneath the topsoil. A key characteristic of these soil is an organic-rich A/O horizon as organic carbon is a critical feature of these soils. The podzol soils within the Waituna Catchment cover an area of 2,392 ha (12.4%) and were formed in shallow- to moderately- deep loess deposits overlying alluvial and old marine terrace deposits. The soil series present in this unit are Tiwai (Pan Humic Podzol) and Kapuka (Pan Fill Podzol), with minor Ashers (Pan Fill Podzol) (Figure 3.9) (Topoclimate South, 2001).

Podzol soils have a moderately developed structure and loamy silt/silt loam textures with gravels typically found below 40 cm depth. The soils are imperfectly drained, with slowly permeable subsoils that may cause short-term waterlogging after heavy rain. Subsurface mottling occurs in the clay-bound underlying gravels reflecting the slow permeability of these soils and moderate reduction potential. The upper subsoils are characterised by the accumulation of complexes of iron and organic matter, indicative of podzolised soils. Crops for Southland (2002) reports the P-retention in these soils as high, minimising the risk of P-leaching. However, P loss from soils undertaken by

AgResearch for Waituna noted Podzols as having an elevated P leaching risk relative to other mineral soils (McDowell and Monaghan, 2015). The aluminium (Al) and iron (Fe) sesquioxides that sorb P are most unstable under reducing conditions and low pH.

Artificial drainage density is moderate in these soils to prevent short-term waterlogging in wetter months (Pearson, 2015a). Bypass flow, via the artificial drainage network, can reduce soil residence time reducing the potential for denitrification to occur. In soils with pans which restrict deep drainage, lateral flow can occur along slowly permeable layers within the soil profile. However, the spatial extent of lateral flow is limited by the artificial drainage network.

3.5.4 Organic Soils

Organic soils are formed in the partly decomposed remains of wetland plants forming peat. There is some mineral material present, but the soils are dominated by organic matter (50-90%). In Waituna Catchment, Organic soils cover an area of 6,860 ha (35.6%) and occur as raised bogs (up to 6 metres deep) overlying gravels or marine sediments with poor permeability. In Carran Creek, the Tertiary basement is close to the surface with negligible alluvial material, while Moffat Creek, has thick deposits of slowly permeable marine terraces. The soil series present in the Waituna catchment are Otanomomo (Fibric Melanic Organic) and Invercargill (Mesic Acidic Organic) (Figure 3.9) (Topoclimate South, 2001). The peat of the Otanomomo soil shows weak to moderate decomposition of organic matter, while the Invercargill soils are moderate to strongly decomposed (Crops for Southland, 2002).

Near the land surface, the peat is typically loose and fibrous grading to denser, amorphous peat with depth. Organic soils are generally structureless with a low bulk density. They have very poor internal drainage which means they are prone to waterlogging, particularly where the water table is shallow (Crops for Southland, 2002). This also limits air movement through the soils resulting in very poor aeration and anoxic waters. The soils have a high reduction potential due to their high organic carbon content and high water table. This combination results in water that is strongly reducing.

Organic soils have a low P-retention making them susceptible to P-leaching. This occurs because of the low mineral content in the soils limiting the ability to sequester or sorb P out of solution and the strongly reducing conditions in the soil (see section 1.2.6). Organic soils are extremely acidic which limits their versatility for agricultural use, without improved drainage and acidity (i.e. liming).

3.5.5 Recent soils

Recent soils are weakly developed, showing limited signs of soil-forming processes. The Riverton soil series (Typic Sandy Recent) is the only Recent soil in the catchment with an extent of 353 ha (1.8%) (Topoclimate South, 2001). Riverton soils are formed into coastal dunes of wind-blown sand. The soil has a loamy sand texture resulting in a well-drained soil with rapid permeability (> 72mm/hr). As a result, it is well aerated and has a low reduction potential. Due to the limited amount of weathering, P-retention in this soil is very low (<5%), increasing the risk of P-leaching (Crops for Southland, 2002).

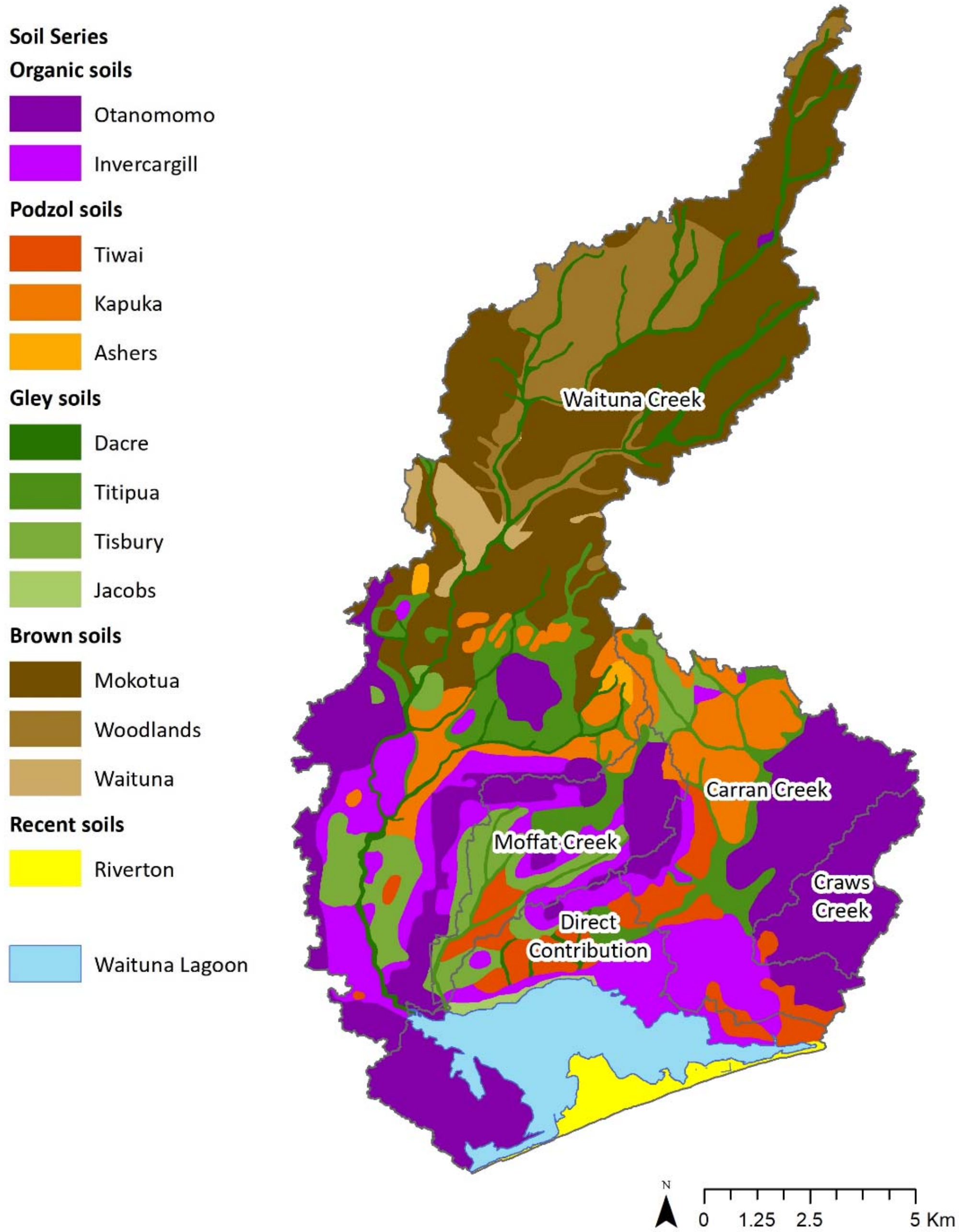


Figure 3.8: Soil order and series within the Waituna Catchment. Soil series are displayed by the dominant soil within a mapped area (data from Topoclimate South, 2001 and NZLRI (DSIR,1978)). Brown soil extent is broadly correlated with maximum sea level.

3.6 Soil influence over water quality

Soil order can be limiting when attempting to understand the relationship between gradients in landscape features such as slope, geology and geomorphic setting and attendant gradients of physical and chemical soil properties (Rissmann et al., 2016a; Rissmann, 2016 d; Beyer et al., 2016a; Beyer et al., 2016b). Therefore, to understand the landscape controls that are relevant to water quality outcomes, the soil attributes are better indicators of processes than NZSC order alone.

Soil texture, permeability and drainage class are the critical attributes determining the pathway water takes across the landscape and the strength of soil zone reduction, respectively (Pearson 2015a; Pearson 2015 b; Rissmann et al. 2016). Areas characterised by slow permeability (<4 mm/hr) in surficial or shallow subsoils are more prone to overland flow than soils with high permeability[8]. Areas of well-drained soils with moderate to high infiltration rates favour deep drainage. The proportion of slowly permeable soils in the Waituna Catchment is 36%, with an additional 61% of the catchment with moderate over slow drainage (Figure 3.10). The proportion of well and moderately well-drained soils is small (<5 %), with most of the soils classified as imperfectly drained (41%), poorly drained (18.4 %) or very poorly drained (35.6 %) (Figure 3.10). Poorly drained soils are often strongly reducing and therefore have higher denitrification rates than well-drained soils, however, poorly drained soils have a greater tendency to leach P and are also more prone to overland flow (McDowell et al. 2003; Simmonds et al., 2015).

The microbial (M) load, free and sorbed N and P, organic carbon (C), and associated carbon to nitrogen (C:N) and carbon to phosphorus (C:P) ratios are important indicators of sediment quality. Developed soils have a higher potential M, N and P load along with C characterised by smaller C:N and C:P ratios relative to areas of indigenous land cover with undeveloped/natural state soils. The contaminant content associated with 1 kg of agriculturally derived sediment, therefore, can be large relative to 1 kg of sediment derived from a natural state setting (Cooper and Thomsen, 1988). Recognition of this difference in sediment quality is very important in terms of internal eutrophication of receiving environments, such as rivers, lakes, lagoons and estuaries.

Soil P-retention (or anion exchange capacity) varies with soil age and parent material. According to Topoclimate South (2001), 50% of the Waituna catchment has moderate P-retention (30-60%), 11% has a high P-retention (60-90%), and the remaining 39% of the catchment area is very low (0-10%) (Figure 3.11). However, the Podzol soils, as noted in section 3.5.3, are more likely to have a low P-retention with a high risk of P leaching (McDowell and Monaghan, 2015; Simmonds et al., 2015). The ability of soils to sorb P has implications for P-leaching, the P-content of sediment lost via farm drainage and/or the P-content of stream banks that are eroded during peak flow events (Simmonds et al., 2015).

[8] It is anticipated that those areas with soils of slower infiltration rates have been modified by artificial drainage.

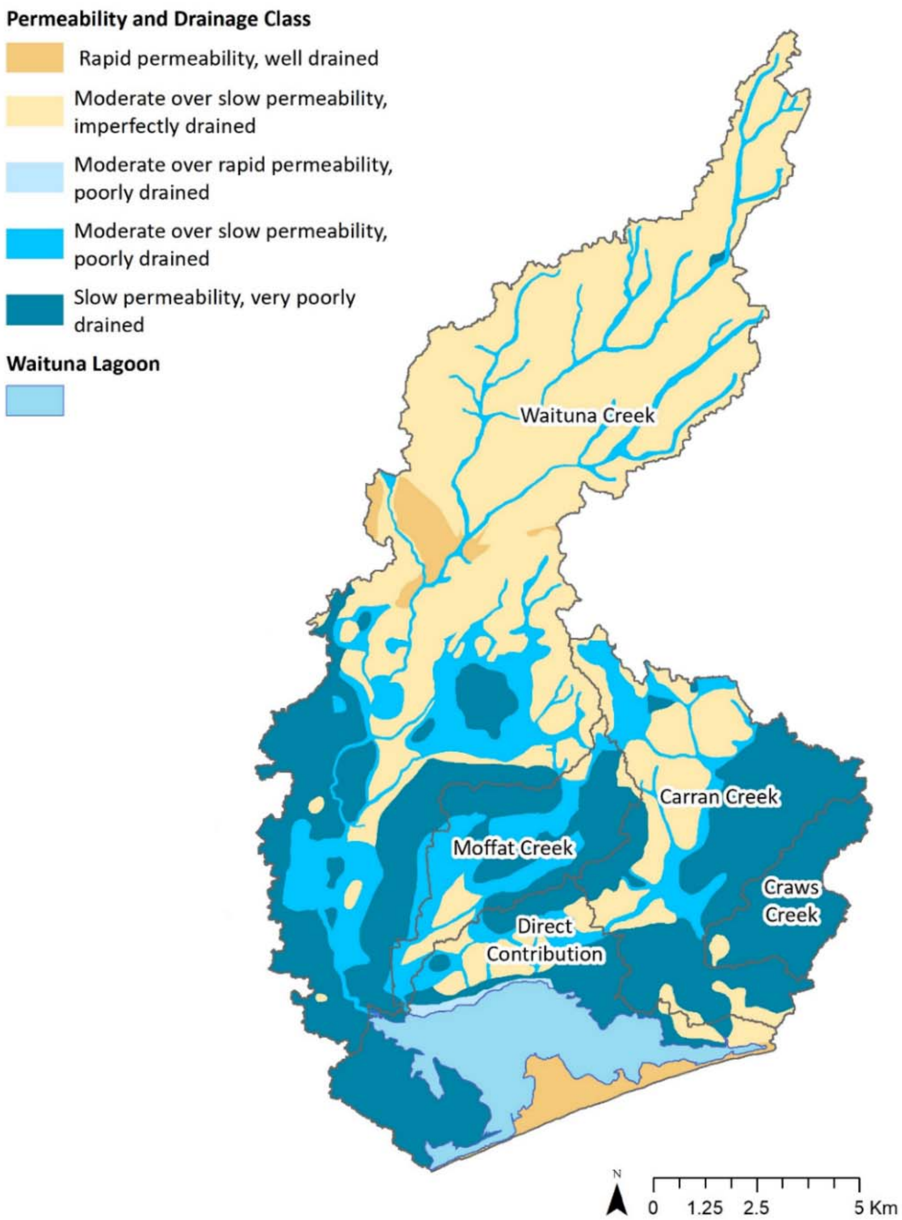


Figure 3.9: Soil drainage and permeability (data from Topoclimate South, 2001).

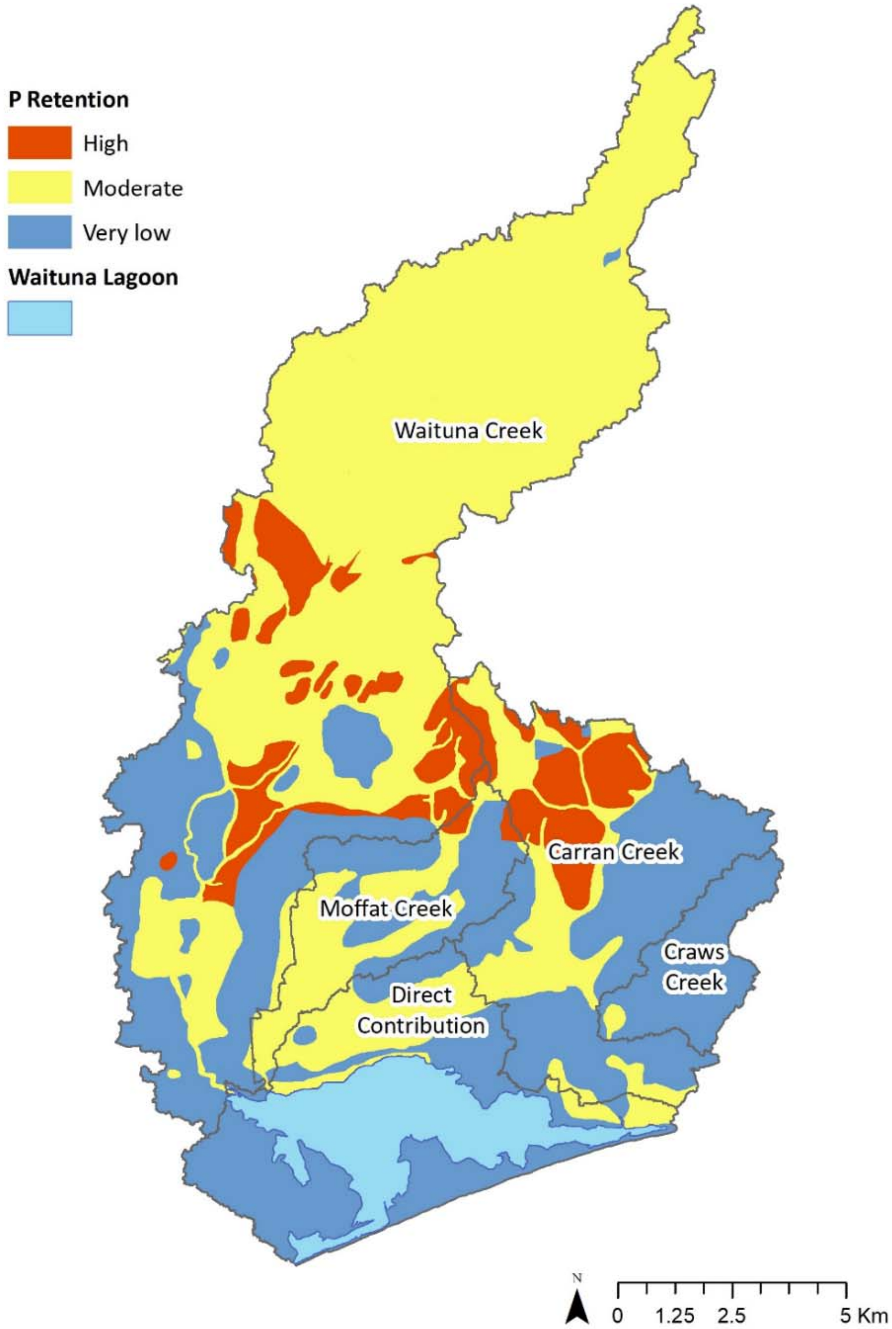


Figure 3.10: Phosphorus retention of Waituna catchment soils (high 60-90%, moderate 30-60% and very low 0-10% (data from Topoclimate South, 2001). Note the areas mapped as High P retention are more likely to be low due to Al and iron Fe sesquioxides that sorb P being unstable under reducing conditions and low pH.

4 Hydrology and Flow Path Analysis

4.1 Introduction

The transport of water and contaminants to the surface water network is facilitated by baseflow from connected aquifers, lateral soil zone flow and overland flow (Figure 4.1). For aquifers, soil drainage must percolate to depth through the subsoil and any undifferentiated sediments overlying the shallow water table aquifer. In some settings, stream flow may also infiltrate the shallow riparian aquifer.

At certain times of the year and/or in response to the natural and modified properties of the surficial soil environment, water may bypass the soil matrix. Bypass flow via macropores may be associated with cracking soils, artificial drainage (mole-pipe drainage) and/or relic features such as root traces or burrows made by soil fauna. Within developed landscapes with slowly permeable and/or poorly drained soils, subsurface artificial drainage as ditch or pipe and in cohesive soils as mole-pipe (tile) drainage is by far the most important bypass mechanism (Pearson, 2015a; Monaghan, 2014). These engineered structures constitute by far the dominant pathway for soil water drainage, with lateral matrix flow considered to be relatively minor (Monaghan, 2016).

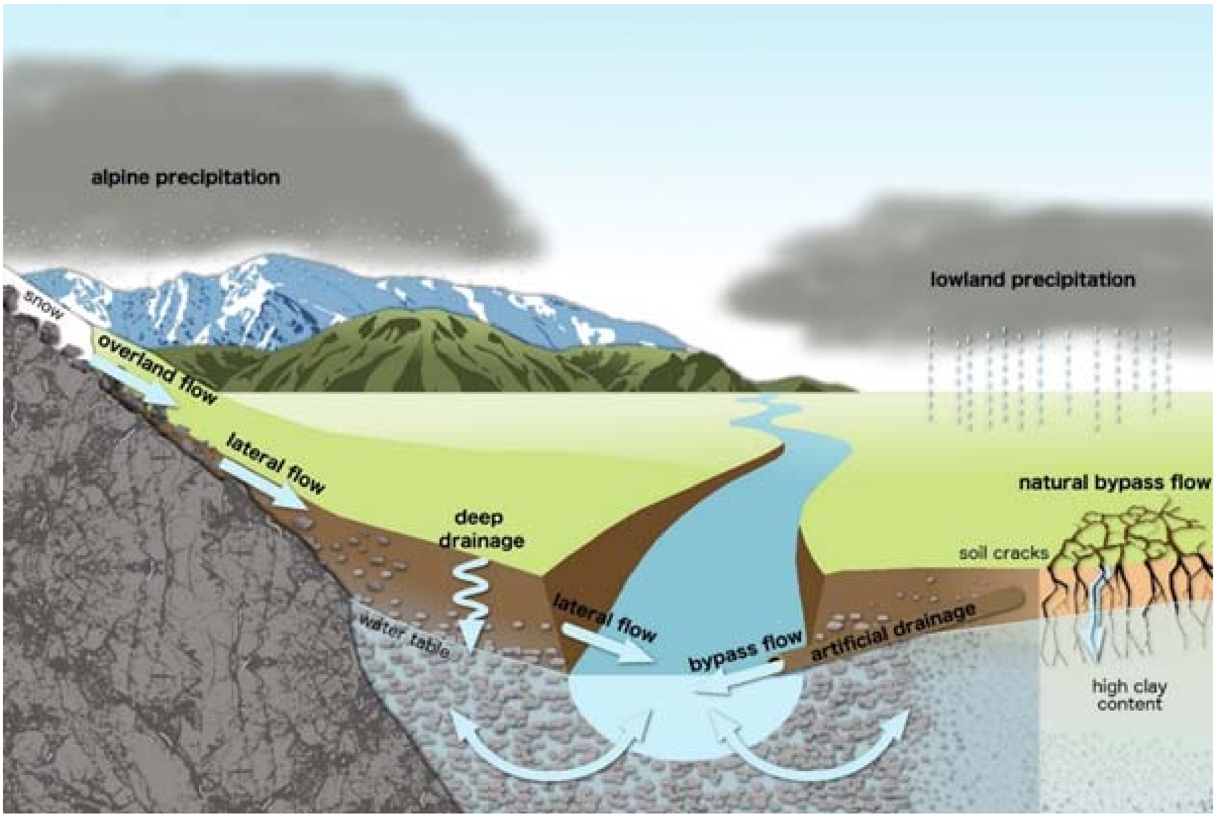


Figure 4.1: Summary of hydrological flow pathways identified in Southland during the 'Physiographics of Southland' Project.

The hydrological layer from the science component of the 'Physiographics of Southland' (Rissmann et al, 2016) provides general spatial information over the water source (origin of streams) and recharge mechanism, and identifies the Waituna Catchment as a lowland catchment recharged by local precipitation. However, the regional hydrological layer does not include an assessment of gradients in overland flow (OLF) nor the likely density of artificial drainage or any high-resolution depiction of surficial and soil compartment flow pathways.

Accordingly, a key focus of this project was to incorporate fine-scale flow path information into the hydrological process-attribute layer (H-PAL) for the Waituna Catchment. As such, this section details the background and technical development of hydrological layers depicting gradients in surficial overland flow risk, mapping of fine-scale surficial flow paths, the location and likely density of artificial drainage, the location of higher rates of deep drainage to shallow aquifers, unconfined aquifer thickness and catchment water table. The evolution of a fine scale hydrological layer for the Waituna Catchment incorporates new analysis as well as outputs from existing work within the catchment (Wilson, 2011; Rissmann et al., 2012; Rekker and Wilson, 2015; Gurrein and Wourms, 2016; Pearson 2015a; Pearson 2015b; Marapara and Jackson, 2017; Environment Southland, unpublished data).

4.2 Surface Water Catchments

The Waituna catchment features three main tributaries which flow into the Waituna Lagoon – Waituna Creek, Moffat Creek and Carran Creek. Craws Creek drains an area of undeveloped peat wetland and is a tributary of Carran Creek. In addition, there is a fifth subcatchment, a zone of direct contribution, that fringes the Waituna Lagoon. These subcatchments all drain into Waituna Lagoon, and ultimately into Foveaux Strait (Wilson, 2011; Rissmann et al., 2012; Rekker and Wilson, 2016). The relative areas of these subcatchments are presented in Table 4.1.

Table 4.1: Waituna subcatchment areas and proportion.

	Area (Ha)	% of Catchment
Waituna Creek	11,152	57.8
Moffat Creek	1,545	8.0
Carran Creek	4,285	22.2
Craws Creek	810	4.2
Direct Contribution	2,296	11.9
Waituna Catchment	19,278	100

4.3 Overland Flow

Overland flow (OLF, or surficial runoff) is cited as a key pathway for land-based contamination to enter waterways (Deakin et al., 2016; Goldsmith and Ryder, 2013; Orchiston et al., 2013; Curran Cournane et al., 2011; McKergow et al, 2007; McDowell, 2006; Smith and Monaghan, 2003). Where conventional hydrological assessments refer to overland flow as purely surface runoff we note a significant component of 'A' and/or 'O' horizon water in event runoff (Rissmann et al., in prep; Inamdar et al., 2011). Therefore the contribution of overland flow commonly includes both surface and shallow 'A' and 'O' horizon waters under event conditions (see Inamdar et al., 2011 and references therein).

Overland flow is driven by two main mechanisms relating to the hydrological status and character of the soil. Specifically, saturation excess and infiltration excess OLF events. Saturation excess events occur when soils approach saturation and there is little capacity for precipitation to infiltrate the deeper soil zone resulting in OLF (Srinivasan et al., 2002). The second type of OLF occurs when precipitation intensity exceeds the rate at which water can infiltrate the soil, and is referred to as infiltration excess or Hortonian runoff (Horton, 1940). Infiltration excess OLF is difficult to predict as it can occur at any time of the year under varying soil moisture conditions. The likelihood of infiltration excess OLF occurring can be increased by animal treading damage, or heavy cultivation restricting soil permeability. On flat land, infiltration excess OLF will result in surface ponding (Needelman et al., 2004).

In reality, OLF may be driven by both permeability and saturation limitations, although it is apparent that the highest frequency of runoff events within the Waituna Catchment, and Southland in general, occur during the cooler months, May through November, when soil water content is elevated (Smith and Monaghan, 2003; McDowell et al. 2005; Monaghan et al, 2016). High densities of artificial drainage likely restrict the frequency and the duration of OLF by dramatically increasing the hydraulic conductivity and intrinsic permeability of the soil zone. However, bare paddocks associated with wintering by dense mobs of stock are known to reduce the permeability of soil thereby increasing the risk of surficial runoff and entrainment of contaminants (Drewry and Paton, 2000; Monaghan et al., 2016; Pearson et al., 2016).

In the Waituna catchment, two separate analyses of overland flow have been completed – a broad scale assessment of surficial runoff risk (Pearson, 2015b) and fine-scale convergent zone modelling (Marapara and Jackson, 2017). The surficial runoff risk layer identifies areas where there is a higher likelihood of saturation excess surficial runoff occurring within the catchment, whilst convergence modelling presents areas where surficial runoff will converge at a sub-farm scale. The methodology for the broad scale overland flow risk is presented below, whilst the convergence zone modelling is presented in Section 5.

4.3.1 Overland Flow Risk

Overland flow risk was assessed by Pearson (2015b) by identifying areas where there is a higher likelihood of saturation excess overland flow occurring across the Southland region. The assessment used soil and topographical information in GIS to spatially show overland flow risk for the region. Overland flow risk is increased in areas where soils have poor internal drainage and are structurally vulnerable to slaking and dispersion, or in areas where there is sufficient slope to generate runoff. The GIS layer was created by firstly combining soil texture and slaking/dispersion characteristics of the soil to calculate a hydrologic index, which was subsequently multiplied by a slope factor and expressed as a percentage of effective annual rainfall (Pearson, 2015b). The hydrological index represents the likelihood of overland flow occurring due to the soil properties, while the slope index indicates whether the topography is a significant factor. Figure 4.2 shows the majority of the catchment has undulating topography, therefore the overland flow risk is mostly associated with the hydrological properties of the soils. The risk assessment was independent of land use management practices or vegetation cover, though it was noted that these factors do have a significant impact on overland flow occurrence (Pearson, 2015b). The output of this assessment is the percentage of effective rainfall that will likely runoff and is presented in Figure 4.3.

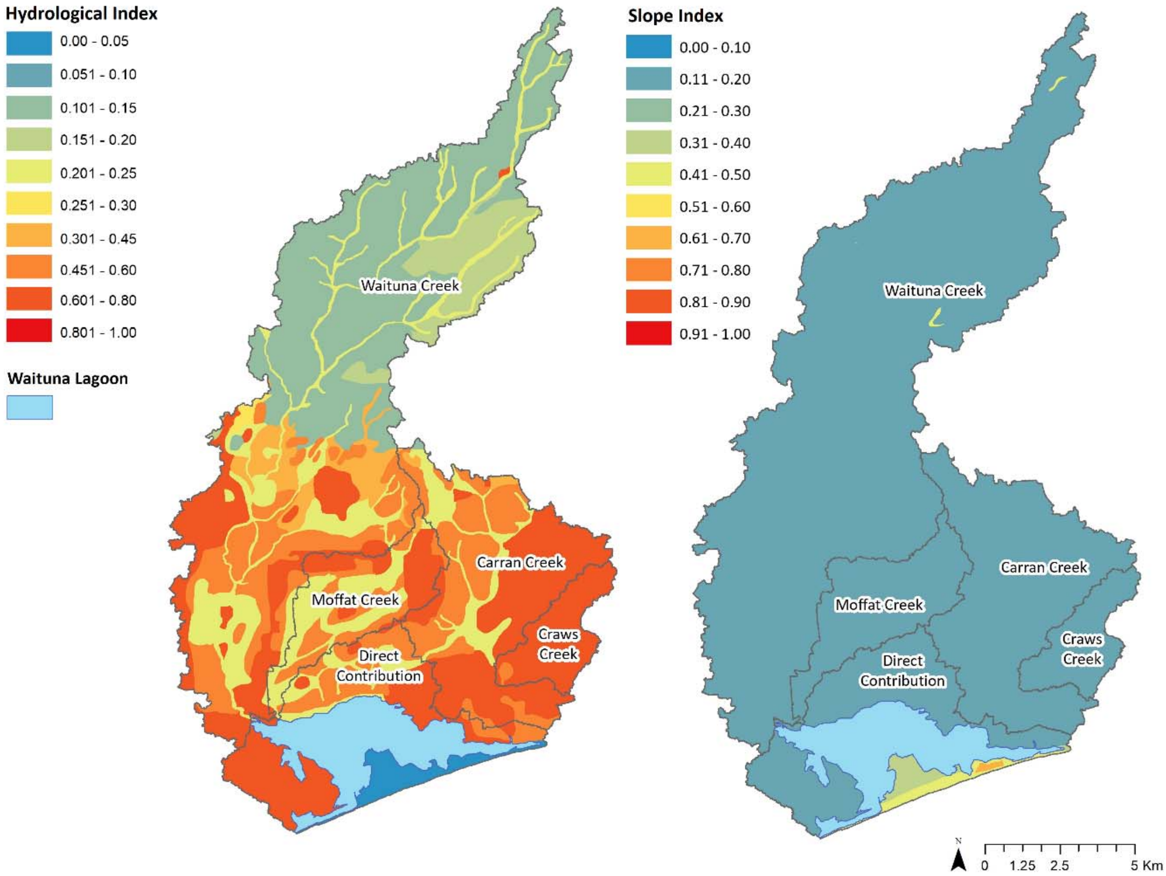


Figure 4.2: Hydrological index and slope index for the Waituna catchment (Pearson, 2015b). The difference in the hydrological index of the northern and southern portions of the Waituna Catchment is an important feature.

A summary of overland flow risk according to Pearson (2015b) across the Waituna Catchment is presented in Table 4.2. Compared to wider Southland region, overland flow risk is relatively low across the Waituna Catchment, with less than 4% effective rainfall likely to runoff across 48% of the catchment, and less than 10% of effective rainfall likely to runoff across 72% of the catchment. Overland flow risk is highest in Craws Creek, however as this area is predominantly natural state runoff does not have a high contaminant load (source limited). Carran Creek has the highest risk and potential contaminant source for the transportation of contaminants to the lagoon, followed by the Waituna Creek catchment. Again it is important to note that although % of effective rainfall occurring as overland flow is relatively low, it does not take much runoff to significantly impact on water quality.

Table 4.2: Overland flow risk by total area (Ha) and percentage of subcatchment.

	Waituna Creek		Moffat Creek		Carran Creek		Craws Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
0-2 %	4,096	37							334	15	4,430	23
2-4 %	3,400	30	639	41	523	12	0.2	<0.1	296	13	4,858	25
4- 6%	676	6	11	1	50	1					737	4
6-8 %	913	8	267	17	216	5			341	15	1,737	9
8-10 %	721	6	187	12	979	23	24	3	270	12	2,157	11
10-12%	1,346	12	441	28	5,516	59	785	97	1054	46	5,359	28
Total area	11,152	-	1,545	-	4,285	-	810	-	2,296	-	19,278	-

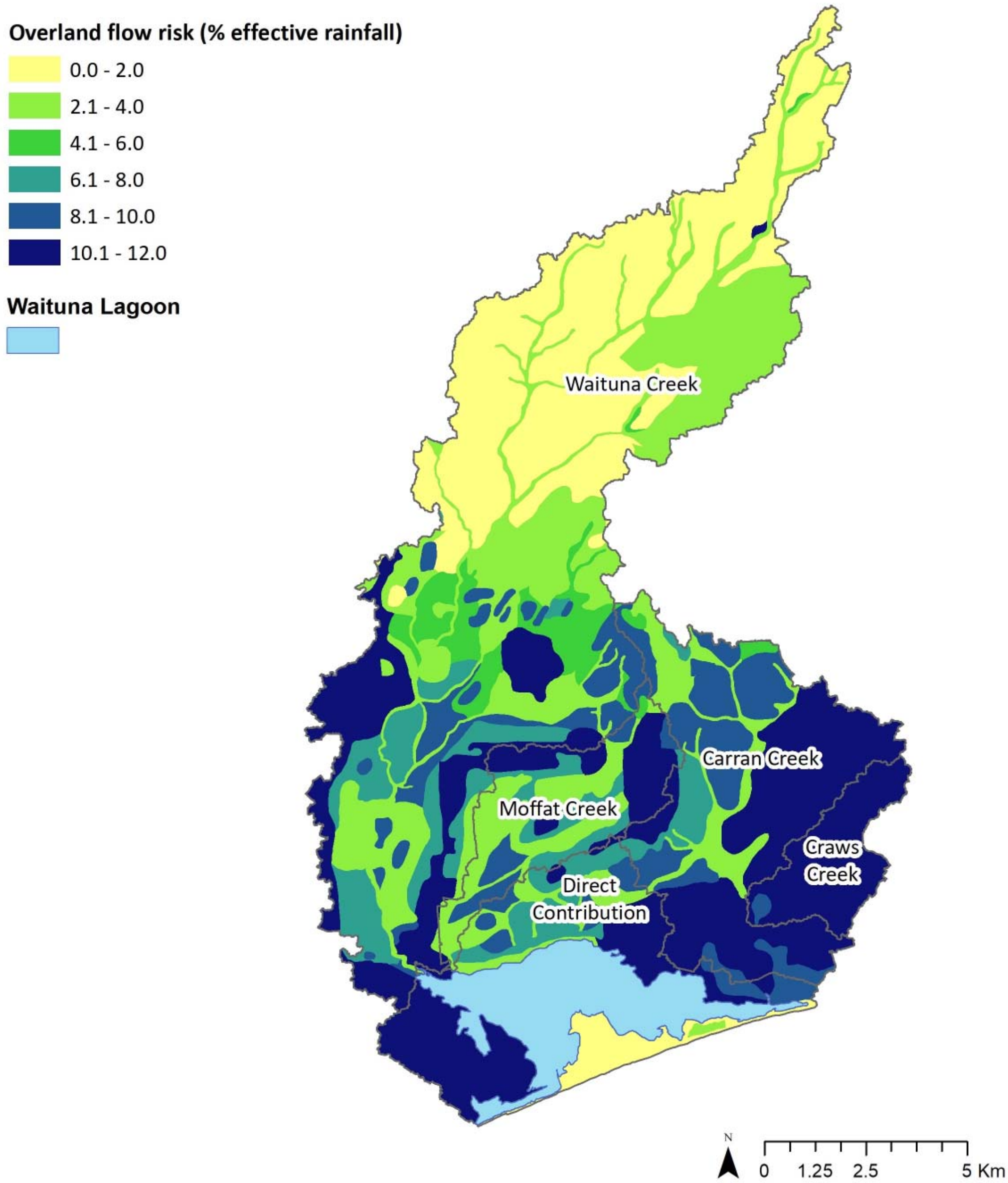


Figure 4.3: Overland flow risk for the Waituna catchment (Pearson, 2015b).

4.4 Artificial Drainage

As land use has intensified in the Waituna catchment, more land has been drained using mole-pipe (tile) drainage, thus modifying the natural state flow paths within the catchment. Tile drainage is a major pathway of water flow and is recognised as a major contaminant transport pathway (Hallbrooke and Monaghan, 2009; Monaghan et al., 2010). Tiles can convey particulate (sediments), microbes and nutrients to waterways, however typically at lower quantities of particulate and microbial contaminants than surficial runoff due to contact and attenuation by the soil zone (Monaghan and Smith, 2004). Open channels are also important sinks and sources of contaminants with fluctuating water tables playing an important role in attenuation processes and contaminant flux to surface and groundwater (Monaghan, 2014).

4.4.1 Artificial Subsurface Drainage Density

Subsurface drainage across Southland was estimated by Pearson (2015a) using the soil properties of permeability and drainage class, combined with land cover and topographical information to produce a framework to estimate drainage density for the Southland region.

Subsurface (tile) drains are typically installed in one of two arrangements in Southland including conventional and contour patterns (Pearson, 2015a). Conventional drainage is used in conjunction with open ditches when the land surface is a constant slope (minimal undulations) to lower the water table to a uniform depth. This type of drainage is typical on Organic soils. Contour drainage is used most on undulating or sloping land, or where wetter areas of a paddock are present and drain into an open waterway (or ditch). The tiles/pipes are laid in hollows or swales and follow the natural contour of the landscape. The depth that the tiles are installed varies depending on the depth of the water table and the amount of fall necessary to drain the area. Tiles are typically found at 60 - 80 cm depth and between 20 - 100 m apart, and mole drains are typically ploughed at 45 cm depth and can be as close as 2 m apart (Houlbrooke and Monaghan, 2009).

Areas where artificial drainage is likely to be present was identified using the Land Cover Database, version 4.1 (LCDB4.1) and via selection of areas where land cover was classified as ‘high producing exotic grassland’, ‘low producing grassland’ or ‘short-rotation cropland’ to identify the area of Southland likely in agricultural production. Areas not in agricultural production are either in a natural state or under other land uses (such as forestry) and were deemed unlikely to have artificial subsurface drainage. For the Waituna Catchment, this classification was modified by using Department of Conservation (DOC) estate and QEII covenants to identify areas of natural state areas with little to no artificial drainage. The permeability and drainage class of the drainage density categories are summarised in Table 4.3.

Table 4.3: Artificial Drainage Density Categories (Source: Pearson, 2015a)

Drainage Density	Drainage Pattern	Permeability	Drainage Class
Very High	Conventional	Slow	Very poorly drained
High	A mix of conventional and contour (slope dependent)	Slow	Poorly drained
		Moderate	Poorly drained
		Rapid	Poorly drained
Moderate	A mix of conventional and contour (slope dependent)	Slow	Imperfectly drained
		Moderate	Imperfectly drained
		Rapid	Imperfectly drained
		Slow	Moderately well drained
Low	Contour and soak hole (flat topography only)	Moderate	Moderately well drained
		Slow	Well drained
Very low to none	Typically feeder drains from other areas	Moderate	Well drained
		Rapid	Well drained

Artificial subsurface drainage density is moderate to very high across 81% of the catchment (Figure 4.4, Table 4.4). Moffat Creek catchment has the highest percentage of very high artificial drainage density (44.5%) of all of the subcatchments, followed by Carran Creek (23.4%). Craws Creek, a tributary of Carran Creek, has the largest percentage of the catchment in a natural undrained state (82.4%), followed by the area of direct contribution to Waituna Lagoon (57.3%). The south of the Waituna catchment has the highest density of artificial drainage associated with the moderately to slowly permeable, very poorly to poorly drained Organic and Gley soils.

Table 4.4: Artificial subsurface drainage density by area and percentage of subcatchment.

	Waituna Creek		Moffat Creek		Carran Creek		Craw Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
Very high	1,680	15	688	44	1,004	23	121	15	330	14	3,702	19
High	2,985	27	648	42	532	12	0.2	0.03	232	10	4,396	23
Moderate	5,872	53	191	12	1,075	25	21	3	399	17	7,538	39
Very low to none									19	1	19	0.1
None (not agricultural)	615	6	18	1	1,675	39	667	82	1,316	57	3,624	19
Total area	11,152	-	1,545	-	4,285	-	810	-	2,296	-	19,278	-

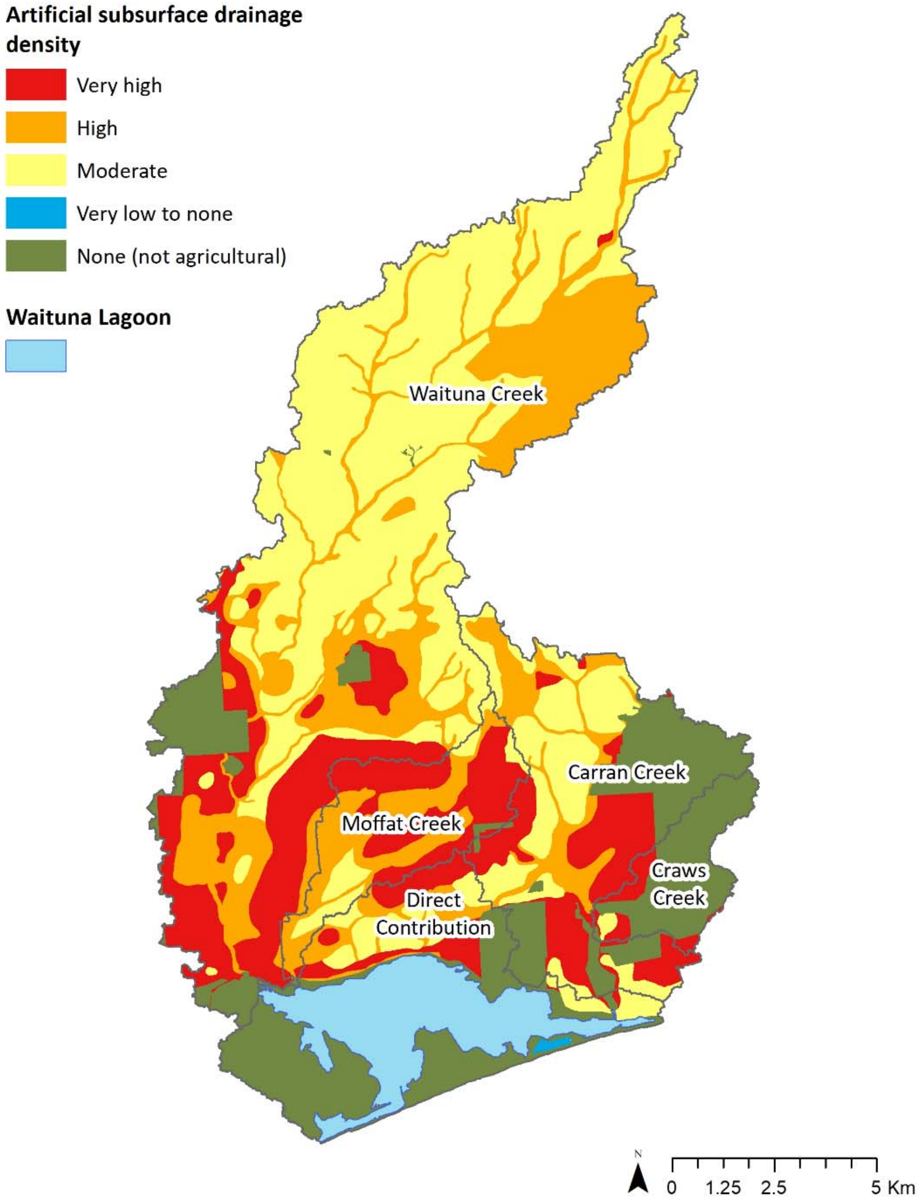


Figure 4.4: Subsurface artificial drainage density (modified from Pearson 2015a).

4.4.2 Drainage Line Mapping

The location of open ditch and subsurface drains were mapped for the Waituna Catchment (Figure 4.5). This was compiled by using the artificial drainage layer (Pearson 2015a) and the digitised river network associated with the River Environment Classification (RECv3; NIWA). The RECv3 identifies both natural and artificial waterways. To digitise the open ditch network, a slope layer generated from the LiDAR was used to identify features consistent with that of known open ditch drains and compared manually to RECv3. Subsurface drains were digitised by identifying straight lines that ran across paddocks on the slope layer. Subsurface drains that leave no visible depression in the land surface will not be identified using this method. Note that many natural drainage channels have been extensively modified and are now presented as open drains. This assessment focused on agricultural land and potential drains alongside roads were not included. This layer could be improved with ground truthing and verification from the local community. The Department of

Conservation is currently working to improve the mapping of drain outlets; however, the outputs of this work were not available at the time of reporting.

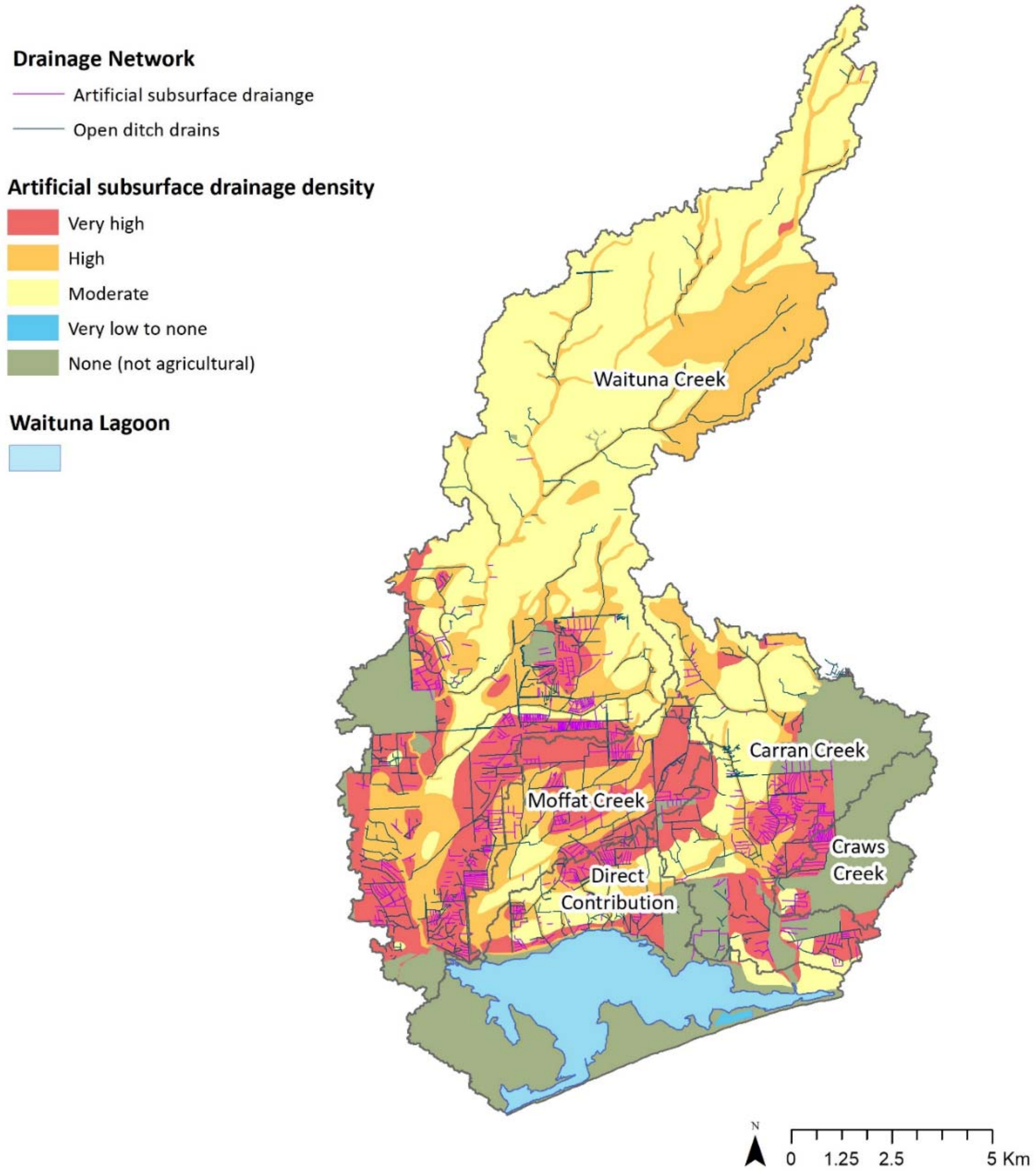


Figure 4.5: Open ditch and artificial subsurface drainage network in Waituna Catchment.

4.5 Soil Zone

Deep drainage or lateral soil zone flow has a strong influence on the composition and quality of surface water and groundwater. This section summarises the information available regarding the likelihood of deep drainage, lateral soil zone flow and bypass flow within the Waituna Catchment.

4.5.1 Deep Drainage

Deep drainage occurs from the percolation of rainfall through the soil zone to underlying aquifers. Deep drainage tends to be highly effective at excluding microbes and sediment and variably effective at retaining P depending on substrate composition and the thickness of the unsaturated zone. Deep drainage to an aquifer, therefore, delivers primarily N and/or P depending on the composition of the soil and aquifer substrates.

A qualitative low-resolution indication of deep drainage was used by Hughes et al. (2016) for the 'Physiographics of Southland' project. The classification combined an assessment of rainfall recharge by Chanut (2014) with the artificial subsurface drainage classes categorised in Section 3.2.1 by Pearson (2015a) to infer whether deep drainage was likely to be high to low in a given area (Table 4.5).

Table 4.5: Potential for deep drainage in Southland (Hughes et al., 2016).

Land surface recharge (% rainfall) (Chanut,2014)	Artificial drainage density (Pearson, 2015a)	Deep drainage class
< 20 %	Very high, High	Low
20-40 %	Moderate	Moderate
> 40 %	Very low to none, Low, Low (slope)	High

In the Waituna Catchment, the areas with the highest likely deep drainage contribution occur in the north of the Waituna Creek catchment in areas with low artificial drainage density (Figure 4.6, Table 4.6). Non-agricultural areas, with natural state hydrology, exist predominantly in Carran Creek catchment and the area of direct contribution to the lagoon. The internal drainage of these areas is likely low, as very poorly and poorly drained soils have accumulated peat under a naturally high water table. In contrast, areas of well-drained soils close to the coast (e.g., Riverton soil) have a high contribution to deep drainage. The extent of well-drained soils in the catchment is very small.

Deep drainage across the Waituna Catchment was investigated at a higher resolution in terms of travel time by Wilson et al. (2014) and is discussed in the following section.

Table 4.6: Deep drainage potential by total area (Ha)and percentage of the catchment.

	Waituna Creek		Moffat Creek		Carran Creek		Craws Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
Developed land												
High	5,864	53	190	12	1,054	25	6	1	418	18	7,533	39
Moderate	2,999	27	648	42	531	12			232	10	4,409	23
Low	1,680	15	687	44	882	21	98	12	330	14	3,676	19
Natural State												
High									334	15	334	2
Moderate	21	0.2	2	0.2	44	1			49	2	117	1
Low	594	5	16	1	963	22	616	76	931	41	3,120	16
Total Area	11,152		1,545		4,285		810		2,296		19,278	

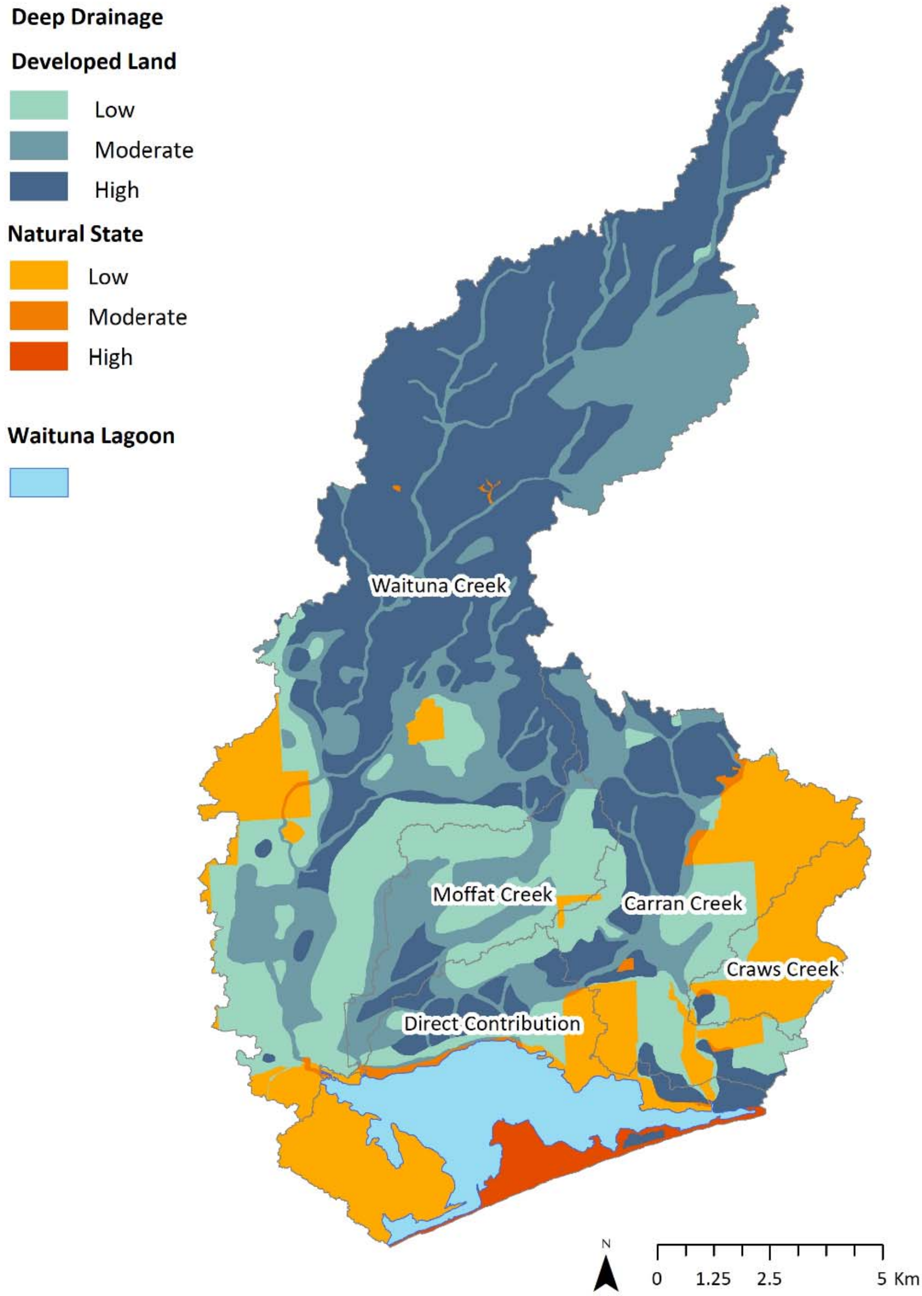


Figure 4.6: Indication of deep drainage in Waituna Catchment (modified from Hughes et al., 2016).

4.5.2 Vadose Travel Time and Saturated Zone Lag

Wilson et al. (2014) developed a methodology for modelling nitrate (NO_3^-) transport from input at the surface through to mixing with the shallow groundwater zone. This model consisted of two components: 1) the time it takes for nitrate to travel from the ground surface through the unsaturated vadose zone ('soil zone'), and 2) the time taken to travel through the saturated zone and mix with the upper part of the shallow aquifer. This assessment did not consider the presence or otherwise of subsurface artificial drainage nor attenuation by denitrification. As such, it represents the transit time of soil water to underlying aquifers and does not assess the relative component transported laterally to stream via subsurface drainage nor the removal of NO_3^- by natural processes. As such it is considered a 'maximum' transit or lag time.

The unsaturated and saturated zone recharge rates were calculated using regional rainfall recharge and potential evapotranspiration models (Rushton Model) from soil hydraulic properties and climate data (National Climate Database, 2017; Environment Southland, 2017). To model the percolation of water through the unsaturated (vadose) zone they calculated the pore water content and transit time using two methods (Brooks-Corey and Van Genuchten; Stankovich and Lockington, 1995), which produced similar results ($r^2 = 0.98$). The methodology was built on steady-state assumptions which reduced the scale of their model to the regional scale (500 m x 500 m) and years of travel time. The mean transit time (> 6,450 sites) was less than a year for both models, and when spatially interpolated for the region about 80% of the region was found to have a vadose transit time of less than one year. When comparing the standard error for vadose zone transit times, 90% of the region has a standard error of less than a year, which is less than the variation caused by annual fluctuations in rainfall recharge.

The mixing depth of the shallow saturated component was calculated from the mean annual rainfall recharge divided by the effective porosity at each site. The model calculates vertical flow through the uppermost portion (c. 1 m) of an unconfined aquifer that is recharged from the land surface. The model implies vertical infiltration, and any horizontal flow velocity is equivalent to depth. This rule applies to all areas except those near surface water bodies as groundwater flow lines converge and diverge around them depending on the water table, and this can alter flow rates. The saturated zone output transit time should, therefore, be considered as a maximum groundwater mixing time.

The vadose and shallow saturated mixing time rasters were combined to produce the total time lag, which is presented in Figure 4.7. The regional vadose zone time lag model has been validated, firstly by comparing the soil drainage model with estimated aquifer recharge values, and secondly by comparing modelled total travel time with age tracer data from GNS Science (Environment Southland, unpublished data). However, it should again be emphasised that the calculations did not consider artificial drainage (Section 4.3.1), macropore flow or paleochannel flow (Section 4.5.2). Further, poorly drained soils with high densities of drainage limit groundwater recharge (Rekker and Wilson, 2016). This is evidenced by independent eigen modelling by Burbery, (2012) for the Waituna Catchment and from water balance studies undertaken by AgResearch at Tussock Creek research station in Southland.

The maximum total lag for the Waituna Catchment from the work of Wilson et al. (2014) is summarised in Table 4.7 with mean travel time is less than one year for 66% of Waituna catchment. Only the north of Waituna Creek catchment has total lags >1 year (max 2.5 years) (Figure 4.7). However, as noted above the saturated zone component of the total lag is considered a maximum, where zones of preferential drainage occur, paleochannels or equivalent, transit times are likely much shorter. Overall, this work indicates little if any meaningful vertical lag time across the majority of the Waituna Catchment.

Table 4.7: Total lag (vadose zone and saturated zone) in years (from Wilson et al. 2014).

	Waituna Creek	Moffat Creek	Carran Creek	Craws Creek	Direct Contribution	Waituna Catchment
Area	10,891	1,544	4,261	809	1,322	18,022
Mean travel time	0.63	0.33	0.33	0.25	0.27	0.50
Min. travel time	0.07	0.10	0.12	0.19	0.11	0.07
Max. travel time	2.36	0.99	0.80	0.51	0.68	2.36
Range	2.29	0.89	0.68	0.32	0.57	2.29

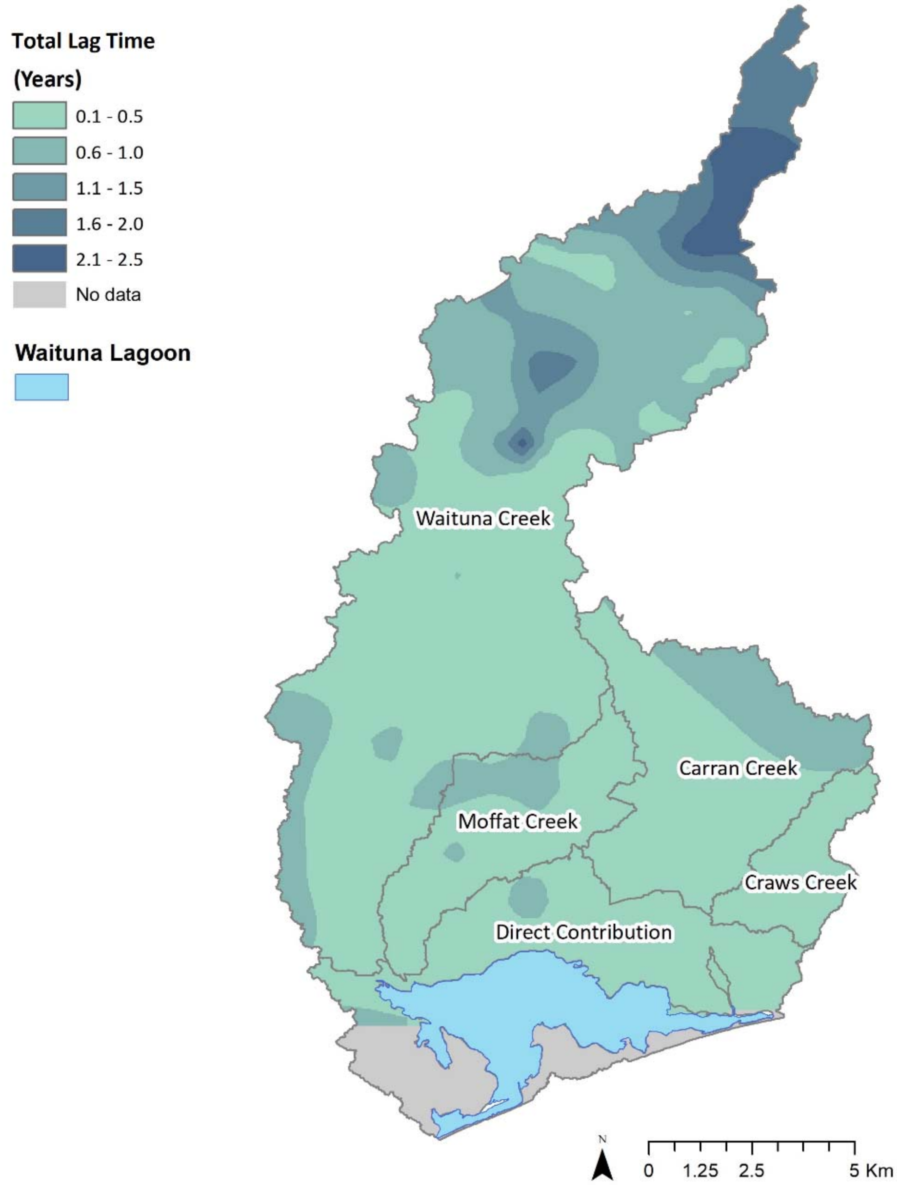


Figure 4.7: Total lag from land surface to mixing with shallow groundwater in years (modified from Wilson et al., 2014).

4.5.3 Lateral Soil Zone Matrix Flow

Lateral soil flow occurs when water infiltrating through the soil intersects a layer with lower vertical permeability. Water flows along the contact between the higher and lower permeability layers (e.g. within a permeable topsoil overlying a slowly permeable subsoil). Water may accumulate to form a perched water table. Areas with a high potential for lateral flow were identified across Southland Hughes et al. (2016) as:

- Hill country and alpine areas where thin sloping soils overlie slowly permeable bedrock, and
- Soils in lowland areas identified as having moderate to highly permeable topsoil overlying a slowly permeable subsoil. These soils were also assessed as having very low to low artificial drainage.

Due to the soil types and artificial drainage density in the Waituna Catchment, no areas within the catchment were identified as having an elevated potential for lateral flow according to the definition of Hughes et al., (2016). Consequently, no further analysis of lateral soil zone matrix flow has been included in this analysis.

4.5.4 Natural Macropore Bypass

Macropore bypass may be locally important with networks of cracks or soil pedogenic structures that propagate to depth (subsoil or underlying aquifer) and bypass much of the natural filtering or attenuative capacity of the soil zone. Macropore bypass may conduct large quantities of nutrients or microbial contaminants to underlying aquifers or mole-pipe drainage networks with little attenuation.

Natural macropore bypass was assessed for Southland during physiographic mapping (Rissmann et al., 2016; Rissmann et al., freshwater sciences shrink-swell soils). The work identified areas where there was a strong disconnect between soils mapped as poorly draining, and hence reducing, and the occurrence of strongly oxidising and high NO_3^- groundwaters directly beneath the soil zone. Time series groundwater level data beneath areas of cracking soils also showed a rapid response to recharge, consistent with bypass of the soil matrix during autumn recharge. Soil cracking in response to soil moisture deficit was be associated with clay-rich soils of elevated Base Saturation (BS) and pH formed in base-rich parent materials (e.g., limestone, mafic and ultramafic parent materials). These parent materials favour the generation of expansive clays or so-called smectites that shrink in response to soil moisture deficit and rehydrate upon wetting.

As soil parent materials within the Waituna Catchment are base-poor, the clays that form are unlikely to be expansive and as such bypass via the cracking of soils is unlikely to be an important mechanism governing contaminant transport. Rather, macropores associated with subsurface-artificial drainage are more likely an important mechanism controlling soil zone bypass.

5 Fine Scale Hydrological Pathways

5.1 Introduction

All major contaminants, N, P, M, and S, are subject to periodic transport over land during rainfall events. Hence surficial runoff is cited as a key pathway for land-based contamination to enter waterways (McDowell, 2006; Edwards et al., 2008; Deakin et al., 2016). However, surficial runoff generally accumulates or converges before reaching a surface waterway or recharging groundwater (Helmers et al., 2005). When mapped, the enhanced knowledge of areas of convergence can assist in identifying and prioritising spatially where contaminant rich runoff can be mitigated.

Interception and mitigation interventions that will be most effective across all contaminants are those that target the peak runoff events and slow the rate at which water leaves the landscape. Mitigating options, such as peak runoff control (PRC) structures (engineered retention dams), work by reducing the velocity of the peak flow and holding back water to allow for the settlement of sediment, and other contaminants, out of the water column (Marttila and Kløve, 2010; McDowell et al., 2012). PRC structures provide one option for intervention as they are suitable across a wide range of land use types, including agricultural areas, artificial surfaces, forests and semi-natural areas, and can be engineered to match the hydrological setting (NWRM, 2013). PRC structures are designed to increase retention times within an existing drainage ditch (ideally 24- 48 hours) to allow contaminants to settle out of solution. Research has found the PRC method to decrease suspended solid loads by 86% of the storm flows load, for nitrogen by 65%, and phosphorus by 67% (Marttila, 2010; Marttila and Kløve, 2010; McDowell et al., 2012). The reduction in nitrogen export reflects the important contribution of ammoniacal and organic nitrogen forms that are transported with sediment, especially during periods of peak flow (Section 2.8). The capacity of the existing drainage network for retaining water needs to be assessed when the catchment is wettest.

Other mitigation options that can be considered on high flow accumulation areas include forested wetlands, swamps and marshes that are effective at nutrient and sediment retention (Fisher and Acreman, 2004; Marapara, 2016; Mitsch and Mander, 2017). Wetlands were observed to reduce N by 16% on poorly drained soils with some rolling topography in Southland (McDowell et al., 2017). For P, it is important to note that the effectiveness of wetlands is minimal when silt accumulation increases, and when sediment-bound P dissolves under reducing conditions (Ballantine and Tanner, 2010; McDowell et al., 2017). Other complementary strategies that can be considered for adoption include sediment traps, detainment bunding, restricted grazing, variable width buffers, and riparian fencing (McDowell et al., 2017)

In this section, areas of surficial runoff accumulation are mapped at a fine scale (paddock). This is undertaken by comparing the method developed by Marapara and Jackson (2017) for convergent zone modelling with other hydrological tools to provide specific target areas for applying mitigation strategies. The aim is to identify the most effective way to identify high-risk areas for contaminant transport from an individual property. This method can only be used to assess hydrological pathways and needs to be used in conjunction with physiographic knowledge and land intensity to identify areas of high risk.

5.2 Convergent Zone Modelling

Convergent zone modelling was completed by Marapara and Jackson (2017) as part of an Environment Southland and Envirolink funded project (1703-ESRC277). The Envirolink project aimed to evaluate whether existing information can be used to reliably map areas where runoff converges at the paddock scale. The flow and convergence of water transporting contaminants is a function of climate, topography, soil properties, geology, proximity to water bodies, land use and land cover type (Marapara and Jackson, 2017).

A combination of poorly permeable and/or compacted soils, high rainfall, low evapotranspiration, large runoff contributing areas and poorly permeable bedrock results in high incidence of fast-moving overland flow and rapid throughflow, raising the risk of flow convergence before reaching surface water bodies. In contrast, rapidly permeable soils overlying rapidly permeable bedrock and receiving flow from uphill areas have the capacity to absorb and store much of this fast-moving overland flow, reducing flow convergence risk.

The Land Utilisation and Capability Indicator (LUCI) model was used to complete the convergence modelling. The model evaluates flow accumulation based on physical principles of hillslope flow, deriving information on permeability and storage capacity of elements within the landscape from soil and land cover data. Overland flow attenuation is interpreted as a reduction in the flow reaching surface water bodies during large rainfall events. Based on the permeability and storage information, LUCI considers volumetric constraints on readily and total available plant water, infiltration capacity, maximum drainage rate, and drainable water holding capacity (the capacity of the soil to hold water between field capacity and complete saturation) (Marapara, 2016). LUCI then discretises units within the landscape based on the similarity of their hydraulic properties and spatially explicit topographical routing.

The following layers were used as inputs to LUCI:

- Digital elevation model (8 m resolution)
- Waituna catchment boundary
- Stream network (RECv3)
- Precipitation data (average annual values)
- Gridded evapotranspiration
- Landcover data (LCDB4.1)
- New Zealand Fundamental Soils Layer

Using gridded annual rainfall and evaporation data inputs, the average annual flow rates for Waituna catchment were calculated within LUCI. Water was routed through the landscape using an algorithm within LUCI that considers the constraints on infiltration, drainage and available water. The direction of the routing was enforced by hydrological conditioning (filling sinks/pits) within the digital elevation model of the landscape prior to running the model.

All land use or soil types that absorb water, provide significant mitigation and are treated as areas of high existing value (sinks), and areas that are intercepted by these features are considered to be mitigated (Jackson et al., 2013). Poorly or slowly permeable areas, where a large amount of unmitigated flow directly routes to water bodies, are identified as convergence zones. These areas identify where intervention approaches will capture the largest amount of surficial runoff. The output produced by LUCI shows areas which accumulate five times more water than provided by effective precipitation, and are classified as high convergence, while areas accumulating more than twenty times the effective precipitation volume are classified as very high convergence. This output is displayed by subcatchment in Figures 5.1 to 5.4, however, due to the resolution of the data, it is best viewed using an interactive map viewer. This allows the user to zoom to areas of interest and combine the output with other supporting information, such as slope, land cover, tiles and open ditches when assessing and prioritising areas for intervention (see Section 5.2.1).

The convergence zone modelling output showed that areas of high convergence are widespread in the catchment. The areas providing mitigation are those associated with land covered by deciduous hardwoods, indigenous forest, exotic forest, manuka or kanuka, and flax (Figure 5.1-5.4). The area of high and very high convergence is summarised in Table 5.1 by sub-catchment. Across the whole catchment, approximately 20% of the area is considered to have either high or very high convergence through which the majority of surficial runoff and potential soil zone drainage is thought to occur. Craws Creek has the lowest proportion of convergence zones, whereas the proportion of areas with very high convergence zones are fairly evenly distributed across the remaining subcatchments. However, it is important to note that soil drainage class and infiltration rate are critical when considering the risk that a given area of convergence poses over water quality outcomes. For example, there is little evidence of significant surficial runoff across the northern portion of the Waituna Catchment, unless a very high-intensity storm event occurs.

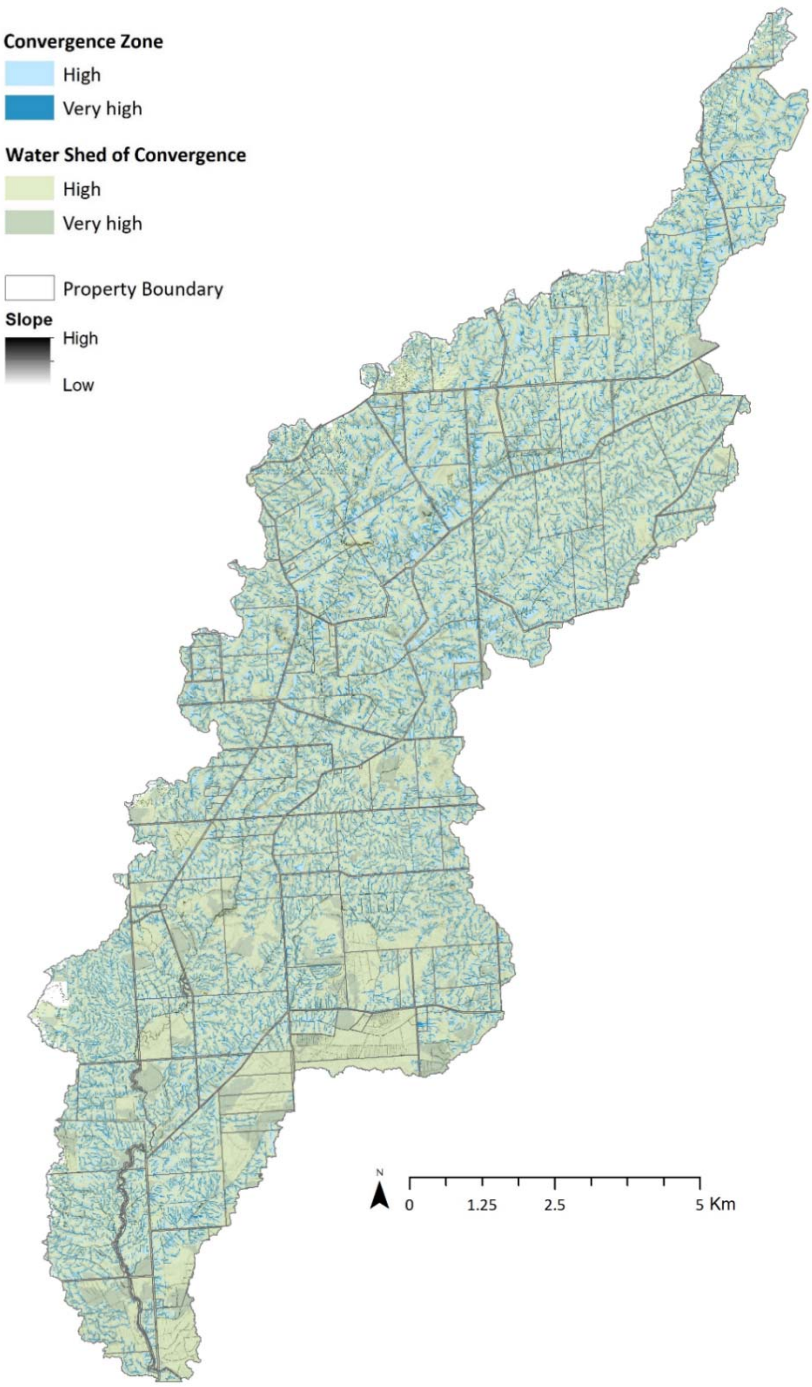


Figure 5.1: Waituna Creek convergent zones.

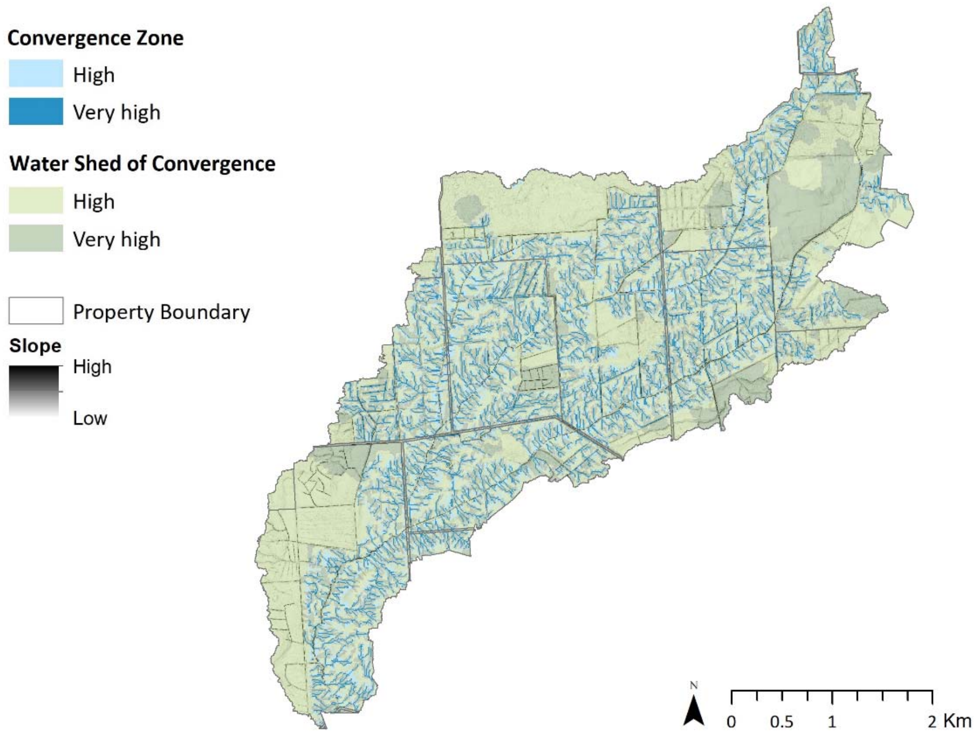


Figure 5.2: Moffat Creek convergence zones.

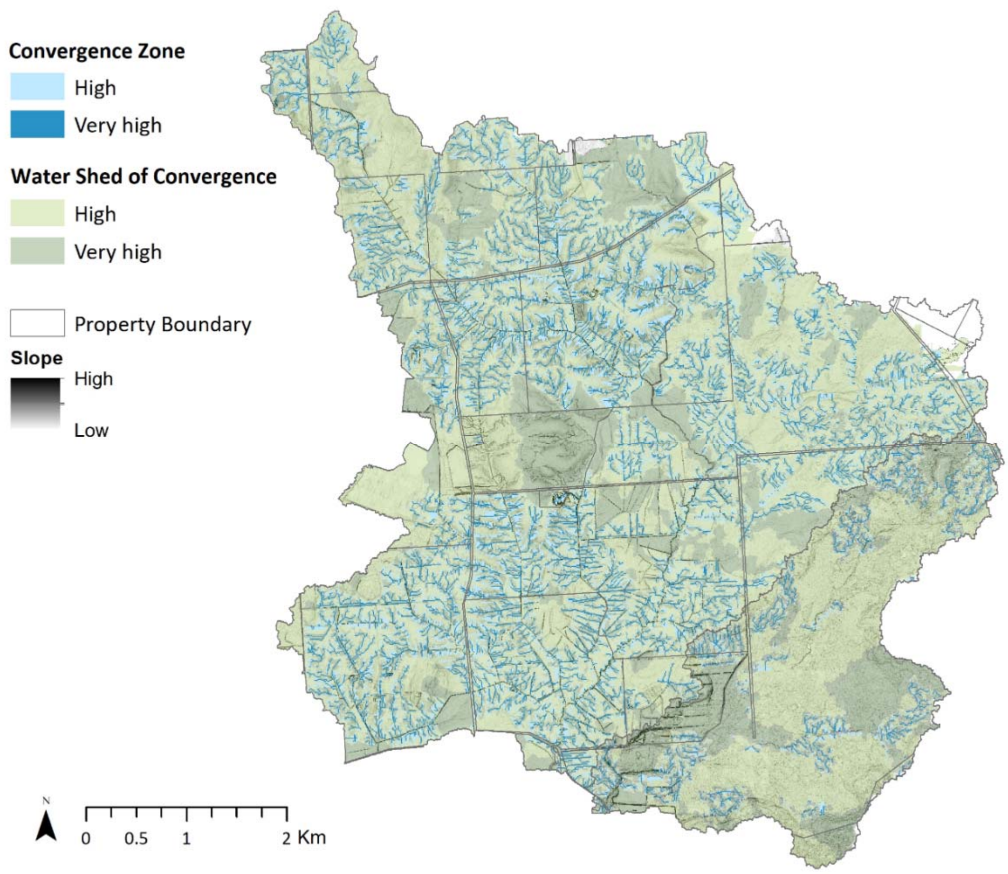


Figure 5.3: Carran Creek convergence zones (note Craws creek to the southeast of the catchment with a low proportion of convergence zones due to the extent of natural state wetlands).

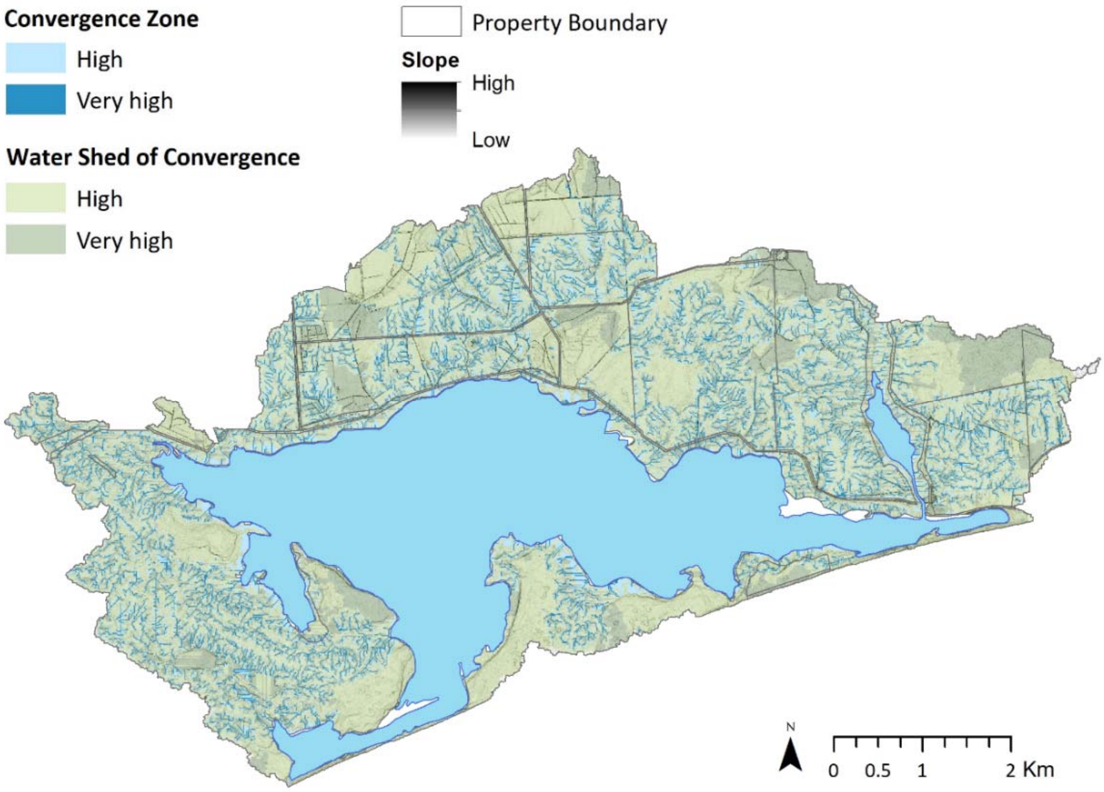


Figure 5.4: Convergence zones in the area of direct contribution to Waituna Lagoon.

Table 5.1: Convergence zones by total area (Ha)and percentage of subcatchment.

	Waituna Creek		Moffat Creek		Carran Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
High	1554.5	13.9	144.7	9.4	376.3	8.8	247.1	10.8	2322.7	12.0
Very high	978.7	8.8	126.5	8.2	283.1	6.6	189.0	8.2	1577.4	8.2
Total area of high to v. high flow convergence	2533.3	22.7	271.2	17.6	659.4	15.4	436.2	19.0	3900.1	20.2

5.3 Stream Strahler Order

The branching nature of a river and its tributaries are known as the Strahler stream order. It is used to define the size of a stream based on the hierarchy of the tributaries flowing to a point of interest. If two tributaries of the same order combine, the next 'order' in the sequence is used to define that streams size (Figure 5.5).

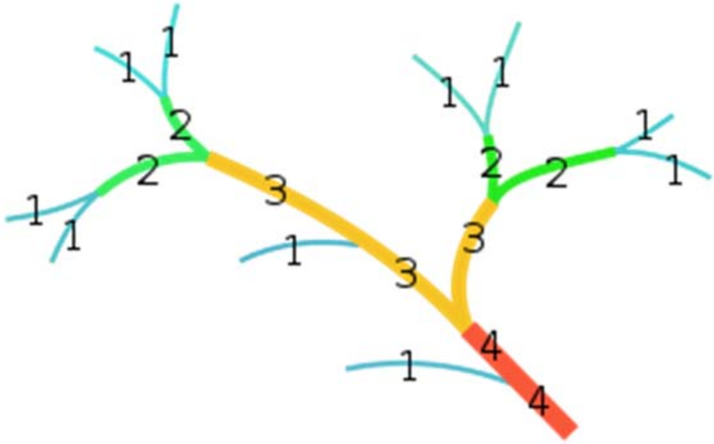


Figure 5.5: Diagram of the Strahler stream order. (image from https://en.wikipedia.org/wiki/Strahler_number#/media/File:Flussordnung_(Strahler).svg).

As very high convergent zones are widespread across the catchment, it is potentially more effective to prioritise areas within first and second Strahler orders for targeted mitigation strategies first. This is because smaller, low-cost investments can be made across the catchment, which has a lower risk of failure, compared to mitigations in higher Strahler orders, which require much larger investment and have a proportionally higher risk of failure. At small (farm) scale, mitigation strategies can target discharge points leaving a property in the available Strahler order(s) (Marapara and Jackson, 2017). LUCI was configured to include vector analysis to produce Strahler stream orders that show a hierarchy of flow accumulation for the catchment (Figure 5.6, Marapara and Jackson, 2017).

Approximately 60%[9] of the Waituna catchment occurs within Strahler order 1 with an additional 23% in order 2 (Table 5.2). Convergent zones within 1st and 2nd stream orders are seen as key areas for intervention mitigations, such as bunding, ponds or peak runoff control structures and if widely adopted could reduce the flashiness and peakedness of surficial runoff events discharging to the stream network. Work is currently underway at Land and Water Science to identify where Peak Runoff Control structures are best placed in the Waituna Catchment using a hierarchical ranking classification (Couldrey et al., 2018). The classification prioritises sites by annual runoff volume, overland flow potential, and potential contaminant load from various land uses.

[9] Analysis by Marapara and Jackson (2017) was undertaken using an earlier version of the catchment boundary (REC1) and as a result there are areas of the catchment with no data. However, given the proximity to the catchment boundary these areas are likely to be order 1.

Table 5.2: Stream Strahler order by subcatchment.

	Waituna Creek		Moffat Creek		Carran Creek		Craws Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
1	6,447	57.8	677	43.8	2,399	54.8	496	61.1	1,636	71.3	11,159	59.4
2	2,246	20.1	613	39.7	1,097	25.8	201	24.8	360	15.7	4,316	23.0
3	1,084	9.7	255	16.5	503	11.4	108	13.3	102	4.4	1,944	10.3
4	408	3.7			245	7.0	0.1	0.02	30	1.3	683	3.6
5	617	5.5							72	3.2	689	3.7
6									3	0.1	3	0.01
No data	350	3.2			41	1.1	5	0.7	93	4.0	483	2.5
Total	11,152	100	1,545	100	4,285	100	810	100	2,296	100	19,278	100

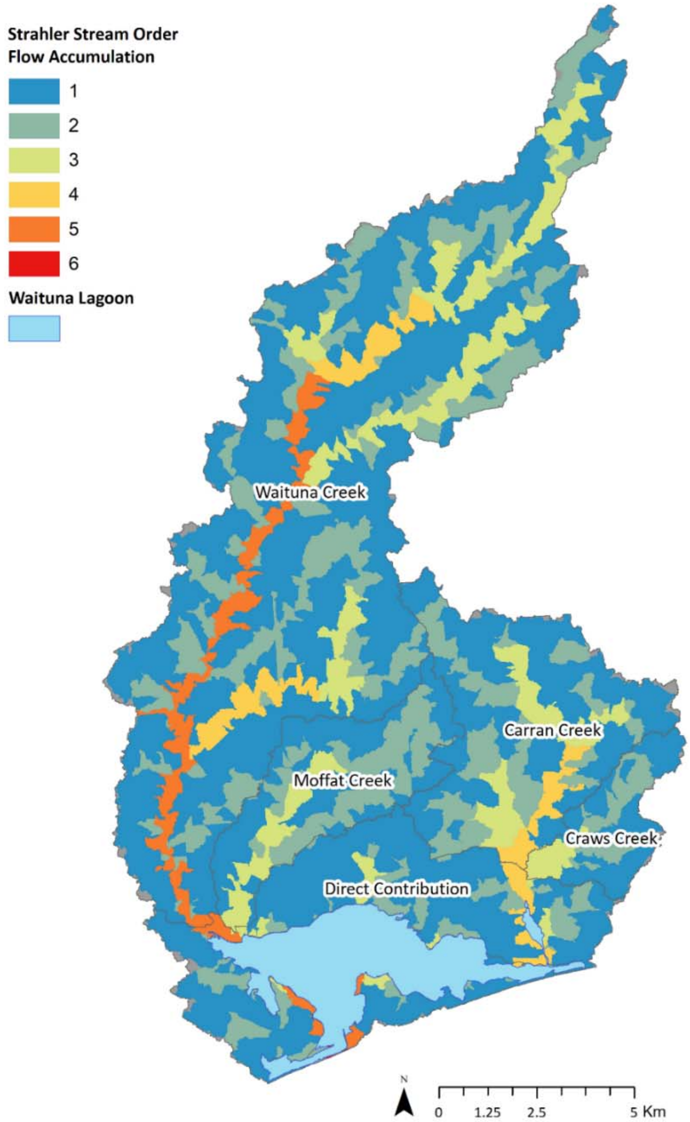


Figure 5.6: Strahler stream orders that show a hierarchy of flow accumulation for Waituna Catchment (Marapara and Jackson, 2017). 1st order flow convergence (accumulation) areas are considered most suited to small-scale mitigations.

5.3.1 Farm Scale Example

An example property within the Waituna catchment was selected to demonstrate the outputs at the farm scale. The property was selected as it contained a range of slope gradients, stream Strahler orders, a mix of land covers, and mapped subsurface and open drains (Figure 5.7).

The integration of the convergence zones with stream Strahler order and the drainage network for the property is shown in Figure 5.8. The subsurface drainage network aligns well with the areas of highest convergence in swales (depressions) in the landscape and the interception point with open drains. Areas with the largest convergence zone length are likely to transport the highest contaminant load, however, this is very dependent on the land cover/use of the paddock at the time of an event. Areas with no convergence zones coincide with areas of land cover types (such as forests or shelterbelts) that are not high producing exotic grassland. The LUCI model considers these areas as mitigating due to the land cover type, however actual land cover may differ from the input data to the model. The LUCI model[10] outputs can be improved by incorporating finer scale land cover, riverlines and soil data specific to a farm instead of national data sets.

The property has a large proportion of the area in Strahler order >3, which means any intervention strategies are best placed up catchment of these areas, prior to attempting mitigations in these higher order (3-8) convergence zones.

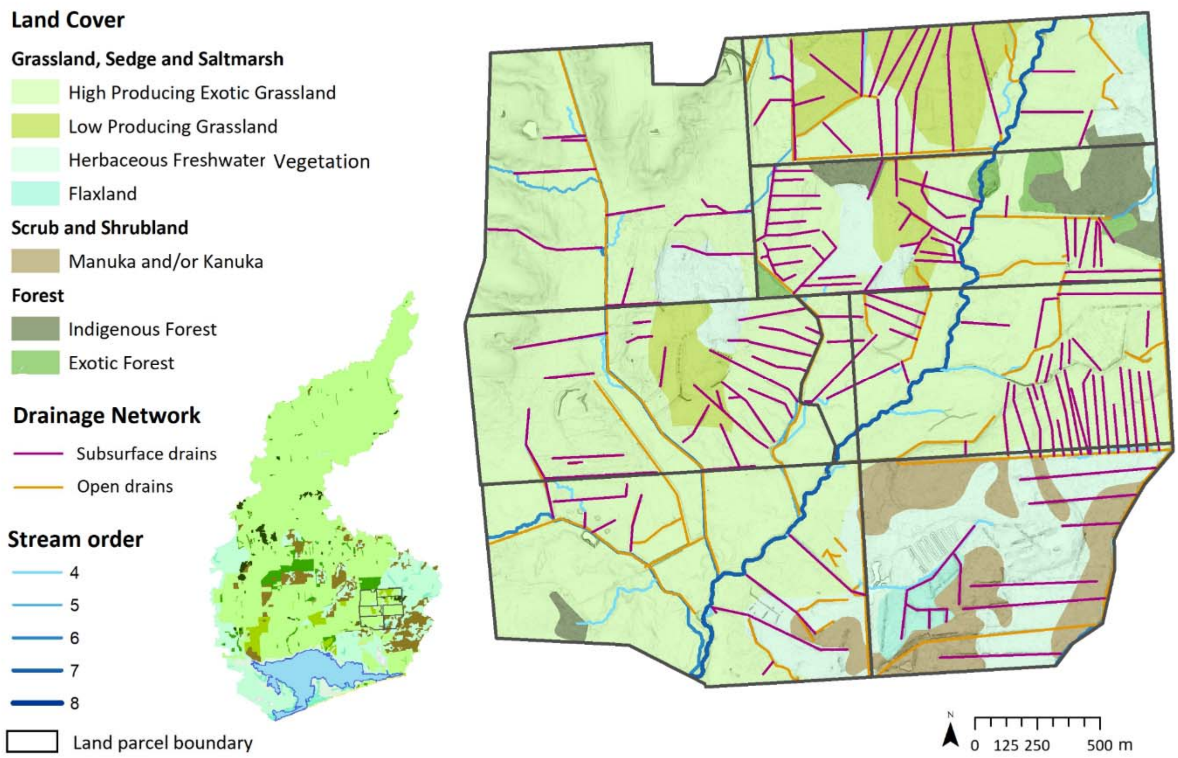


Figure 5.7: Land cover, drainage network and stream order to demonstrate fine scale outputs.

[10] Access to the LUCI model is on request through Victoria University and is still in development.

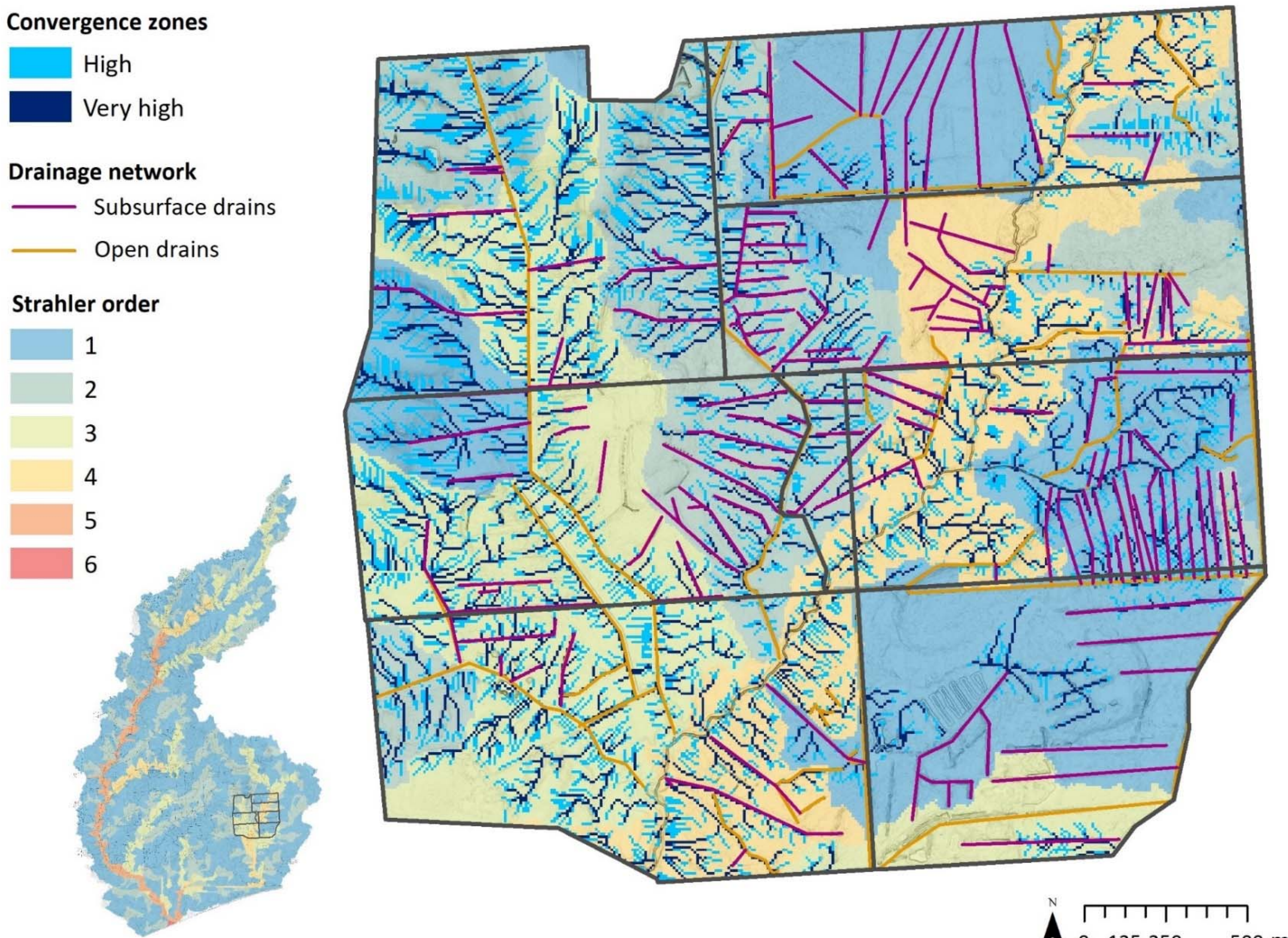


Figure 5.8: Farm scale example of convergence zones and stream Strahler order for a Waituna property.

5.4 Riverlines

The River Environment Classification v3.0 layer (RECv3), produced by NIWA, contains information on a series of spatial attributes for New Zealand's river network system. Despite multiple revisions of this layer, the REC does not always pick up small streams and there are some errors in flow direction in some river systems. To improve the riverline layer for the Waituna Catchment, a series of river lines were generated using the LiDAR-derived DEM that has up to 1 m resolution. By using the high-resolution DEM, riverlines were identified that conform to topographic features in the landscape better than RECv3.

The riverlines were created by first filling the depressions in the DEM and running the 'Flow Direction' and 'Flow Accumulation' tool in ArcGIS. Values in the flow accumulation raster that were greater than 1000 were selected – this identified cells that had more than 1000 cells flowing into it and assigned them a value of 1. All other cells were assigned (by default) 'no data' values. From here a stream network could be extracted and analysed further using the 'Flow Length, 'Stream Link' and 'Stream Order' tools. The resulting output is a river network system that is denser[11] and more topographically correct than the current RECv3 layer (Figure 5.9).

The first, second, and third order streams from the output coincide with the zones of convergence consistent with the LUCI model output (Section 5.2). A further comparison of this layer with the RECv3 should be conducted to identify ephemeral discharges to waterways and a method to differentiate or refine high and very high areas of convergence. Initial mitigation strategies to reduce the contaminants directly entering the waterways should be targeted in the catchment collectively at the first and second order streams. Identification of suitable sites to install the mitigation structures/strategies can be derived from the above layers. Although these layers can stand alone, the resolution can be further refined from evaluation and comparison by using a combination of the fine-scale hydrological pathway layers, allowing for optimal site identification. Further structures in higher order streams could follow upon successful mitigated sediment/contaminant losses from structures installed in the first and second order streams.

[11] Many of the 1st and 2nd order riverlines generated are likely to be ephemeral or intermittent.

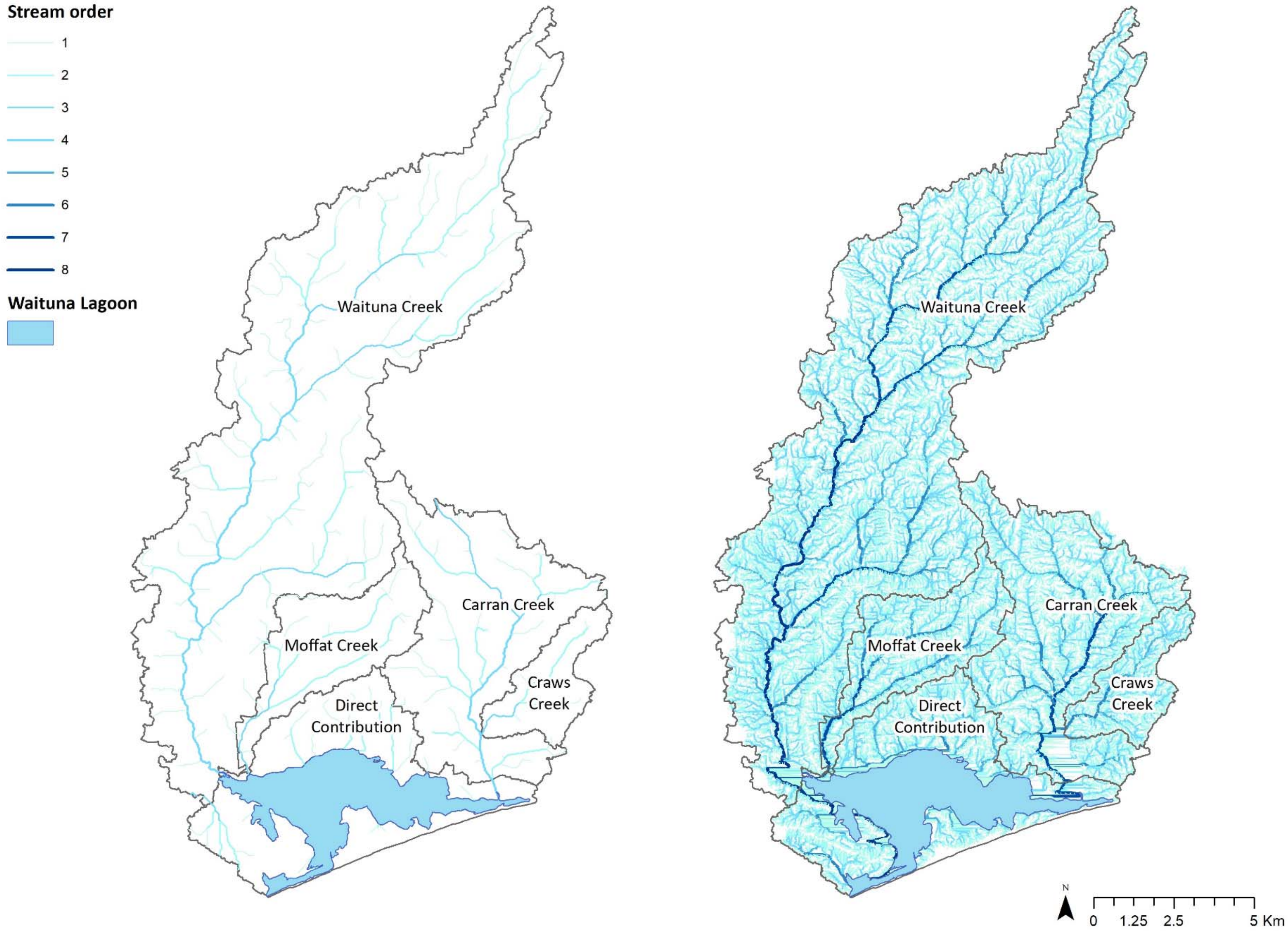


Figure 5.9: Riverlines for Waituna catchment from RECv3 (left) and high-resolution DEM output (right).

5.5 Convergence and Flow Direction

To identify the direction of flow paths, a flow direction raster was generated from the LiDAR-derived 1 m DEM using the ‘Create Watershed’ tool in Global Mapper (v19.1, Blue Marble - Geographics). The tool was processed at a 1 by 1 m scale and symbolised with arrows rotated according to the direction water flows (Figure 5.9). Each square metre of the catchment has an arrow indicating where precipitation will accumulate and flow. The magnitude and size of the arrow are user-defined to increase at intervals of 100 m^2, up to 500 m^2. This output is intended to be viewed at the paddock scale for Waituna Catchment as the arrows generated become too dense at the catchment scale. However, the symbology can be modified to display the main contributing areas proportional to the scale. The riverlines (see Section 5.4) and the flow direction within a paddock in Waituna Creek subcatchment is shown in Figure 5.9. This output can be used in conjunction with physiographic mapping to spatially locate areas where contaminants enter surface waterways.

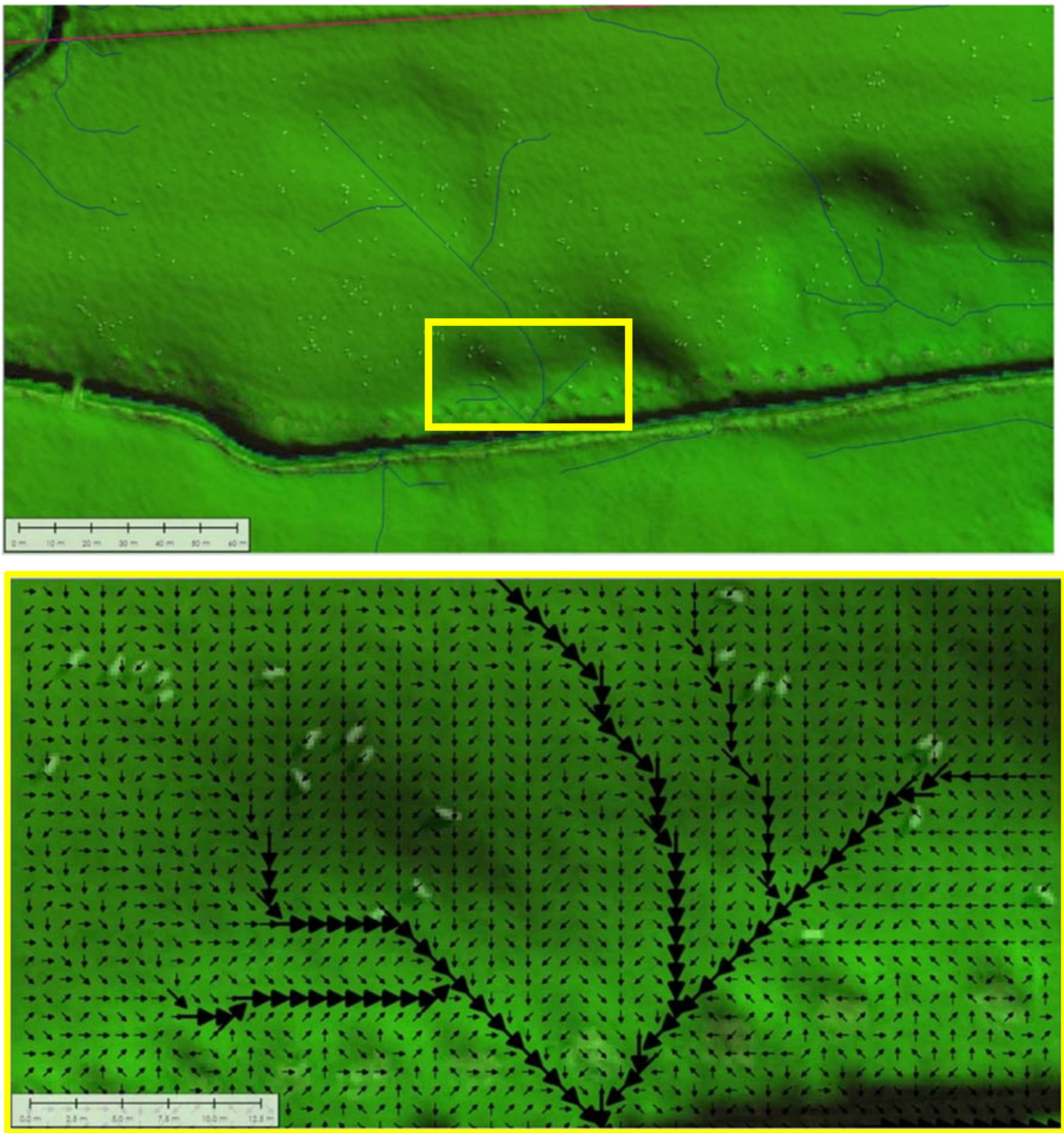


Figure 5.10: Flow direction and convergence at the paddock scale. The top image shows the riverlines generated in Section 5.4. The bottom image illustrates flow convergence with 1 arrow for every 1m^2.

6 Physiographic Application for Waituna Catchment

6.1 Introduction

Water quality can vary spatially across the landscape, even when there is similar land uses or pressures in a catchment. These differences occur because of the natural spatial variation in the physical landscape, which alters the composition of the water through coupled physical, chemical and biological processes. The water composition (of dissolved and particulate constituents) provides information about its origin, the pathway it has travelled and the processes to which it has been subjected. Of most significance to surface water quality are the processes occurring in the soil zone and shallow hydrologically connected groundwater (Figure 6.1). Identifying, mapping, and classifying these landscape features across an area forms the basis of the physiographic approach, making it possible to accurately predict the water chemistry of surface water and shallow groundwater.

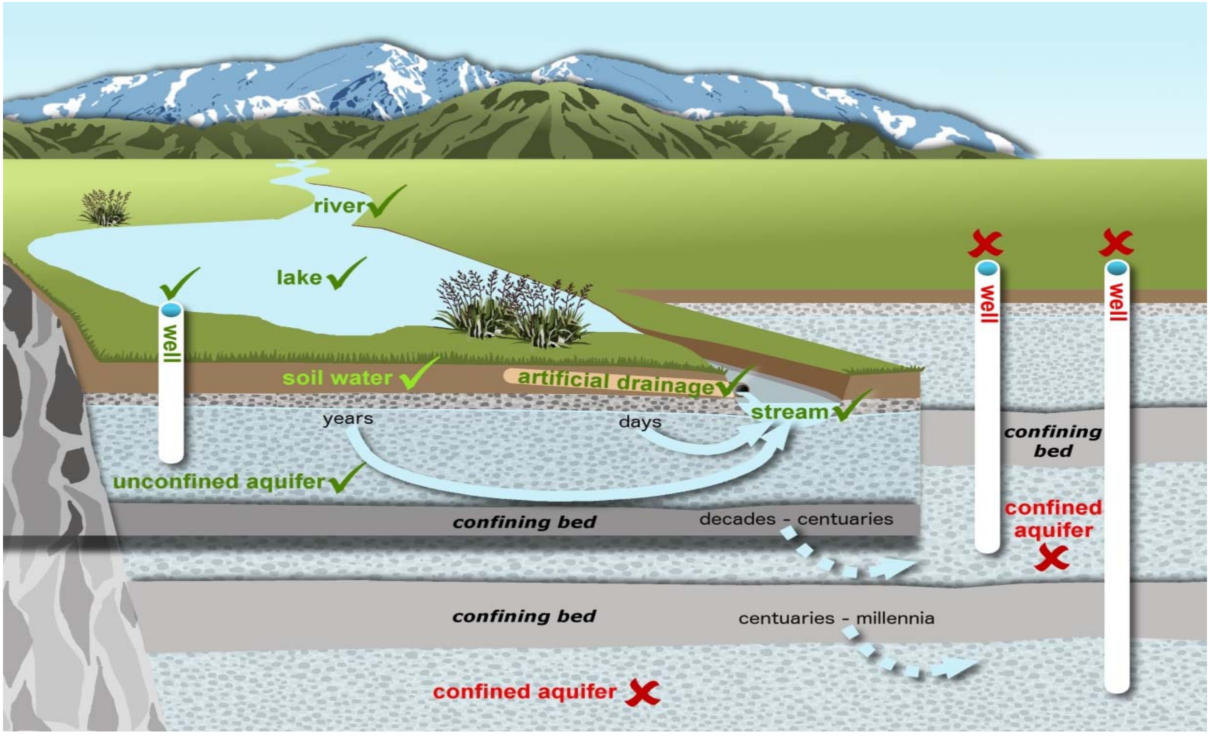


Figure 6.1: Illustration of the connectivity of water resources, including soil water, surface and shallow groundwater (Rissmann et al. 2016). The green tick marks show the hydrologically connected settings included in the physiographic approach, red crosses identify settings that are excluded.

6.1.1 Physiographic Method Overview

Identifying the landscape features which control the spatial variation in water quality is the basis of the physiographic method (Rissmann et al., 2016; Rissmann et al., 2018). Landscape features can account for more than twice the variability in water quality than land use alone (Johnson et al., 1997; Hale et al., 2004; Dow et al., 2005; King et al., 2005; Shiels, 2010; Becker et al., 2014). Natural **gradients** in landscape features, which we term **attributes**, govern the variation in the key **processes** that determine water composition, water quality outcomes and risk. For example, it is widely recognised that soil drainage class (an attribute) is a primary landscape control over the magnitude of denitrification (a redox process) (Webb et al., 2010; Rissmann, 2011; Tratnyek et al., 2012; Killick et al., 2015; Beyer et al., 2016). Soil drainage is also a critical control over the pathway water takes through the landscape (a hydrological process), with the risk or potential for overland flow increasing as soil drainage becomes increasingly poor (Figure 6.2).

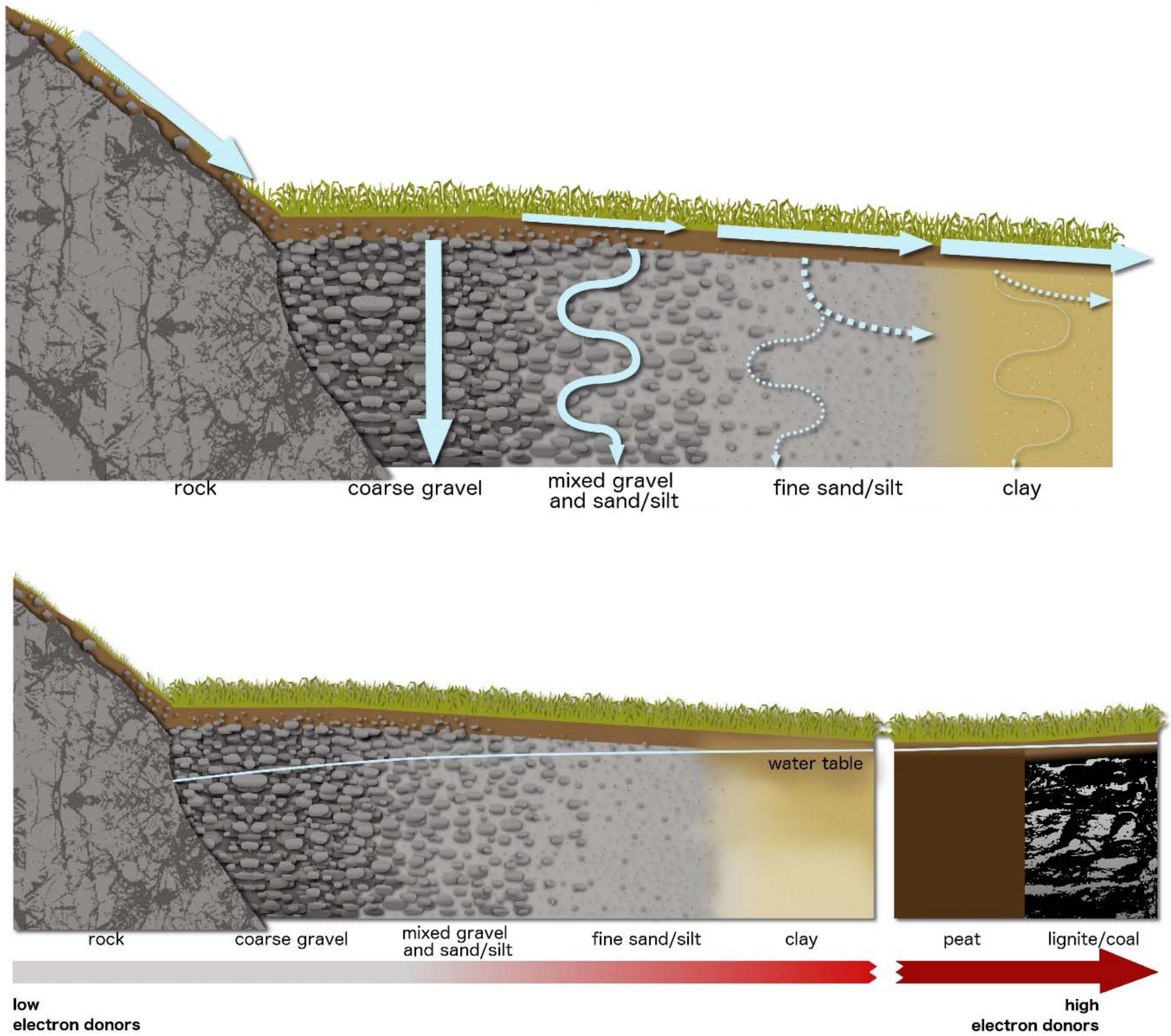


Figure 6.2: Example of an attribute gradient for hydrological (top) and redox (bottom) processes. The hydrological gradient governs the pathway water takes across the landscape. Redox represents the combined influence of soil (unsaturated zone) and geological attributes over redox signatures in water.

The physiographic method involves mapping the gradients in the processes governing variation in water composition as individual **process-attribute layers (PAL)** using GIS mapping software. National and international literature provides evidence that most differences in water quality outcomes (i.e., N, P, S and M), for a given land use pressure, across a landscape can be explained through the combination of **hydrology** and **redox** processes alone (Rissmann et al., 2018). Other processes that control variation in water composition are **atmospheric** and **weathering** (erosion and deposition). These processes are necessary for a detailed understanding of water composition (e.g. hydrochemical facies and mineral saturation indices) and are useful for estimating parameters such as pH, alkalinity, major and trace ion abundances including heavy metals.

The ultimate aim is to produce a number of classed PALs that depict the spatial coupling of key water quality attributes and attendant process signals within the water. Across the Waituna

Catchment, this includes consideration of the process-attribute gradients governing the physical (clarity, turbidity, sediment concentration), chemical (nitrogen, phosphorus, pH, sodium) and microbial (*E. coli*) composition of surface water and shallow groundwater. Each PAL is subsequently classified into categories according to the water quality data – letting the water determine the classes. Each classed PAL can operate as a stand-alone platform for estimating spatial variation in the key processes of interest (i.e. hydrological, redox processes). When these PALs are combined they self-classify, through the identification of areas with common properties. Areas with common properties produce waters that are compositionally similar. Importantly, areas with similar PAL assemblages also tend to have similar fine-scale flow path architecture. Accordingly, through the combination of each PAL, it is possible to estimate ('predict') with considerable confidence the steady state composition of surface and shallow groundwater (Rissmann et al., 2016, 2018; Rissmann et al., in prep). The accuracy of each delineated PAL is verified by assessing the relationship between data from water quality monitoring sites and individual PAL using statistical methods including principal component analysis[12] (PCA) and hierarchical cluster analysis[13] (HCA).

6.1.2 Background

The science behind each of the four PALs is explained in detail in Part 1 of the 'Physiographics of Southland' report (Rissmann et al., 2016) although the method has been subsequently refined and evolved over the last 18 months by the authors of this report (Rissmann et al., 2018). The science component of Environment Southland's 'Physiographics of Southland' project delivered a series of regional scale GIS layers of the landscape controls over the composition and quality of surface and shallow groundwaters across the Southland region (Rissmann et al., 2016) Figure 6.3 shows the resolution of these process-attribute layers for the Waituna catchment used in the 'Physiographics of Southland'. It is important to note the science component of the 'Physiographics of Southland' project differs from the 'Physiographic Zone Map' (after Hughes et al., 2016) produced for Environment Southlands' Water and Land Plan (Environment Southland, 2016). This project for Living Water is an evolution of the original science methodology (Rissmann et al. 2016). The work is founded upon the integration and mapping of the key landscape gradients that govern spatial and temporal variation in water quality outcomes. As such, this work does its best to identify and integrate at a high resolution the inherent spatial controls that drive variation in water quality across the Waituna catchment.

[12] PCA - eigenvector-based multivariate analyses used to show relatedness and explain variation in a dataset.
[13] HCA – groups samples with similar characteristics.

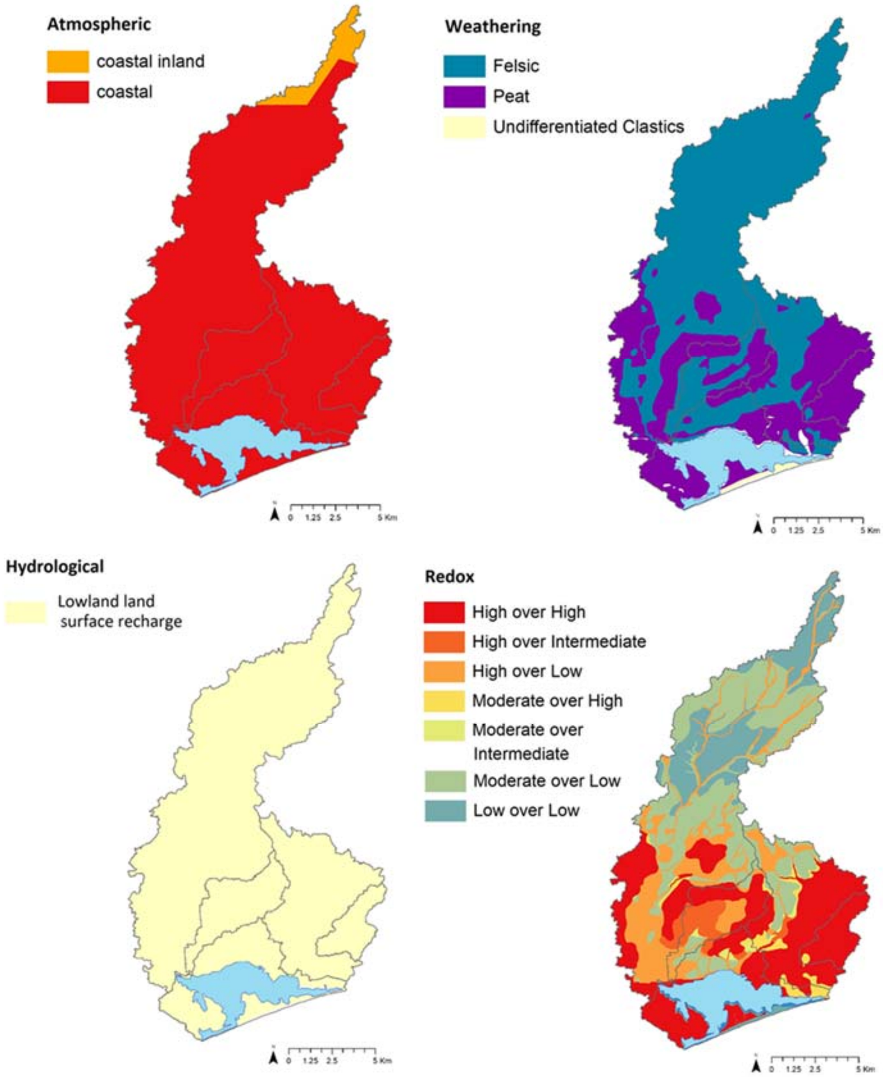


Figure 6.3: Resolution of Environment Southland's four key process-attribute layers for the Waituna catchment. Hydrology and redox layers were used to inform the physiographic zones (Physiographics of Southland project, Rissmann et al. 2016).

6.2 Objectives

In this section, hypotheses as to the attributes which govern water composition and quality are proposed and tested. Specifically, hypotheses as to the coupled spatial relationship between the process signals in water and a combination of spatial gradients in landscape attributes are developed and tested for each of the main process influencing water composition – atmospheric, hydrological, redox and weathering processes.

Throughout this work, our focus is on the broader process signals with testing of more specific outcomes, such as spatial variation in N, P, S and M a by-product of mapping the key process-attribute gradients. For example, the redox indicators DO and Fe^{2+} (occasionally SO_4^-) are the key signals of redox processes occurring within soil and aquifer systems. Dissolved Organic Carbon (DOC) is also assessed as it is a good proxy of the abundance of solid-state organic carbon, the main electron donor governing redox evolution. Nitrate is also considered a key redox indicator but is not, on its own, a sufficient indicator of redox succession (Beyer and Rissmann, 2016; Beyer et al., 2016). As such, the focus is on multiple process signals that provide insight over the role of gradients in landscape attributes within the soil and shallow aquifer that govern the biogeochemical and biophysical processes determining the composition and quality of water. Where possible, the use of water quality indicators as process signals has been avoided.

Following testing and iterative refinement of hypotheses for each process family, the key landscape attributes governing spatial variation in a process signal were extracted and mapped as individual process-attribute layers (i.e. atmospheric, hydrological, redox and weathering processes). Each process-attribute layer was subsequently classified, subdivided into classes, using water compositional data to identify meaningful thresholds in the landscape attributes that correspond to distinct process signals (e.g., a change in surface water DO may be associated with a change in soil drainage class from well to imperfectly drained).

Once classified, each process-attribute layer was joined to produce a combination of classed process-attribute layers. Combining the PALs identifies 'physiographic units' within a catchment or region that share similar process-attribute classes and as such produce compositionally similar waters. Due to the similar assemblages of process classes within a physiographic unit, water quality outcomes are expected to be similar for a broadly equivalent land use pressure. The unioning of classed process-attribute layers is important when considering spatial variation in analytes controlled by more than one process-attribute gradient or when wanting to assess the broader, hydrochemical, composition of water. For example, TN (Total Nitrogen) concentration in a stream is likely to be governed by both redox and hydrological process-attribute gradients. In another example, alkalinity is expected to be controlled by weathering, hydrological and redox process-attribute classes. Accordingly, by combining classed process-attribute layers a more accurate estimate of water quality contaminants is possible due to the influence of multiple processes on water composition.

Testing of the performance of the physiographic units is hypothesis based. Specifically, hypotheses as to the process-attribute gradients governing variation in N, P, S and M are formulated and the spatial relationship between classed process-attribute layers and water measures are assessed using graphical and multivariate means. However, given the large number of classes and the small number of monitoring sites within the Waituna Catchment, a predictive modelling tool (Eureqa (Version 0.98 beta) [Software]) has been incorporated to test the relevance of our hypotheses. Specifically, Eurequa utilises a genetic (evolutionary) programming approach and Machine Intelligence (MI) to test millions of potential models per second and converging on the simplest, most accurate ones that explain the data (Schmidt and Lipson, 2009). The MI approach makes no *a priori* assumptions about the dataset and instead fits models to the data dynamically. To prevent overfitting, the MI approach utilises cross-validation, splitting the data set into two parts: one to train the model, the other to validate its accuracy. The models are presented as mathematical equations, with the software enabling the user to evaluate the sensitivity and magnitude of response in a positive or negative direction of a given attribute over the key variable of interest. The Waituna Catchment Physiographic Model is detailed in Section 7. A model-independent assessment of the uncertainty of the model is also provided in Section 7.

6.3 Site information for hypothesis testing and mapping

There are 17 sites in the Waituna catchment with surface water monitoring data collected by Environment Southland. Figure 6.4 shows the location of the monitoring sites and capture areas which contribute to the monitoring point. Monitoring points downstream of other capture zones include the upper catchment area.

Water quality monitoring data was selected between the years 2012 and 2016 and median values calculated. Median values are used to remove the bias of high flow events in the data record. Of the sites, 16 have suitable data to test hypotheses of controls over water composition (Table 6.1). However, not all analytes used in this assessment were routinely monitored. Site 14 in Carrans Creek was removed as the drain was likely stagnant (very low DO) and not a true representation of local hydrology for the capture zone associated with this site. Site 6 in Moffat Creek was used selectively in the hypothesis testing as the predominant land use in the capture area is the harvesting of peat, which has no agricultural inputs (i.e. fertilizer, animal wastes). A summary of hydrological pathways for the sites is provided in Figure 6.5 and was used to aid hypothesis interpretation.

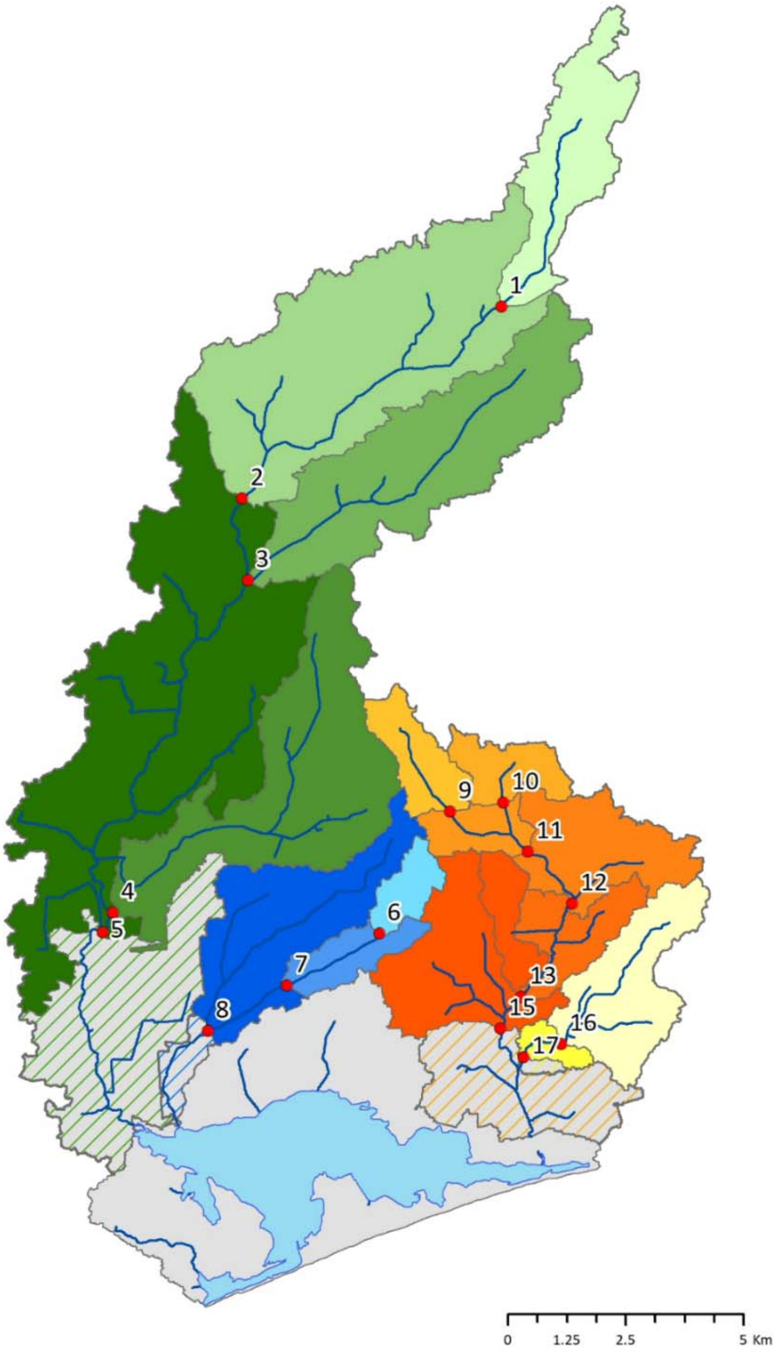


Figure 6.4: Surface water monitoring sites and capture zone within Waituna catchment. Waituna creek catchment is represented in green, Moffat creek catchment in blue, Carran creek catchment in orange (includes Craws creek) and Craws creek catchment in yellow. Hatched areas show unmonitored areas within the subcatchments, coloured as identified above. Grey area is the unmonitored zone of direct contribution to Waituna Lagoon.

*Table 6.1: Surface water monitoring sites in Waituna Catchment. The number of samples for the site is collected between the years 2012 and 2016 (inclusive). The main sites for each subcatchment are highlighted in grey. *Sites excluded from the validation and testing dataset due to strong land use signature.*

Site No.	Site Name	Easting	Northing	No. of samples
	Waituna Creek			
1	Waituna Creek 1m upstream Rimu Seaward Downs Road	1266605	4851793	24
2	Waituna Creek 1m upstream Waituna Road	1261099	4847710	58
3	Waituna Creek NE tributary 10m upstream Waituna Creek Confluence	1261223	4845969	25
4	Waituna Creek SE trib 20m u/s Waituna Creek Confluence	1258355	4838917	22
5	Waituna Creek at Marshall Road	1258129	4838488	143
	Moffat Creek			
6*	Moffat Creek Sth branch 1.2km u/s Miller Road	1264016	4838470	13
7	Moffat Creek 20m u/s Hanson Road	1262043	4837367	14
8	Moffat Creek at Moffat Road	1260369	4836394	90
	Carran Creek			
9	Carran Creek west branch d/s Waituna Gorge Road	1265517	4841056	13
10	Carran Creek east branch u/s Waituna Gorge Road	1266646	4841244	13
11	Carran Creek 1km d/s Waituna Gorge Road	1267164	4840209	13
12	Carran Creek 3km u/s Waituna Lagoon Road	1268105	4839101	12
13	Carran Creek 800m u/s Waituna Lagoon Road	1267026	4837117	12
14*	Carran Creek drain 800m u/s Waituna Lagoon Road	1266988	4837201	13
15	Carran Creek at Waituna Lagoon Road	1266584	4836448	88
	Craws Creek			
16	Carran Creek tributary 1km u/s Waituna Lagoon Road	1267881	4836121	13
17	Carran Creek Trib at Waituna Lagoon Rd	1267080	4835836	45

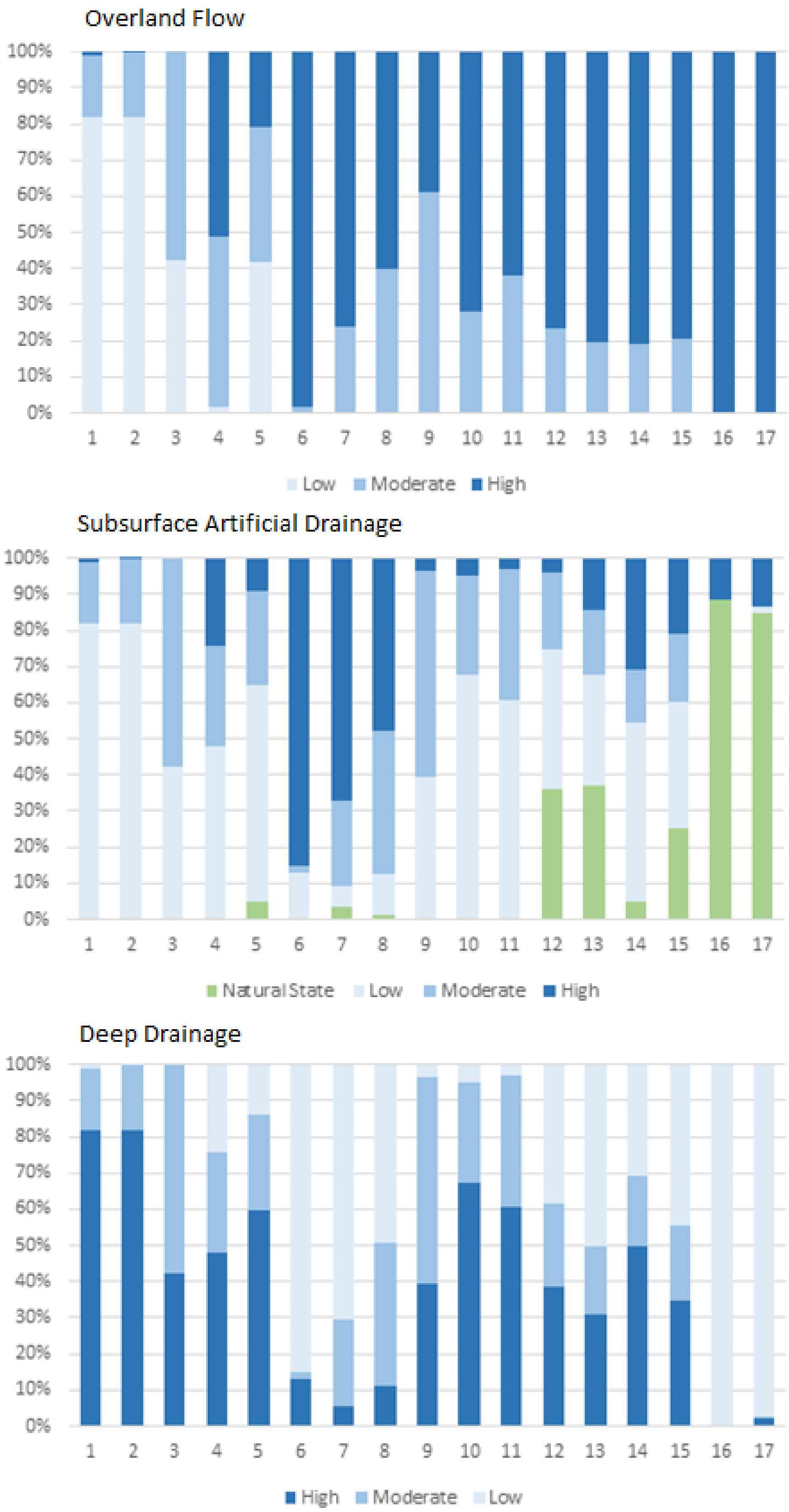


Figure 6.5: Hydrological pathways of overland flow, subsurface artificial drainage and deep drainage for water quality monitoring sites in Waituna catchment.

6.4 Atmospheric Process-Attribute Layer (A-PAL)

The concentration and composition of marine-derived salts across each sub-catchment is important for understanding water source (where it is recharged) and gradients in water conductivity and major ion composition. The 'Physiographics of Southland' identified the most prominent controls over variation in the chemical signature (and precipitation volume) are altitude and secondly, distance from the coast expressed as latitude (Rodway et al., 2016; Rissmann et al., 2016). The Southland A-PAL was produced from precipitation samples collected at sites over Southland. The density of sample sites is not sufficient to refine the Waituna catchment, and thus a method to assess atmospheric loading was developed for the catchment.

6.4.1 Relevance

Spatial variation in the concentration of sodium (Na), chloride (Cl) and the stable isotopes of water (i.e., $\delta^{18}O$-H_2O and $\delta^{2}H$-H_2O, V-SMOW) are considered the key signals of precipitation source. The atmospheric process-attribute layer (A-PAL) defines the loading of precipitation signals, prior to redistribution by the hydrological network. Knowledge of the precipitation signatures of water, prior to redistribution, is important when attempting to understand the ultimate source of surface and shallow groundwater at any given point.

6.4.2 Atmospheric Process Attribute (A-PAL) Hypotheses

Gradients in the concentration of Na and Cl in precipitation, soil water, surface and groundwater of the Waituna Catchment will be controlled by the wet and dry deposition of marine aerosols (salts; Rodway et al., 2016; Beyer et al., 2016; Rissmann et al., 2016e). Specifically, Na and Cl concentration in these waters will decline in a northerly direction away from the coast and with increasing altitude. The lowest Na and Cl concentrations will occur in the far north and the highest concentrations will occur in the most southern coastal portions of the catchment. The stable isotopes of water $\delta^{18}O$-H_2O (‰, V-SMOW) and $\delta^{2}H$-H_2O (‰, V-SMOW) will also exhibit a north-south spatial gradient, becoming increasingly negative with distance from the southern coast and with increasing altitude, reflecting Rayleigh type fractionation typical of meteoric rainout (Clark and Fritz, 1997). Accordingly, there will be a clear N-S gradient in Na, Cl, $\delta^{18}O$-H_2O and $\delta^{2}H$-H_2O. A smaller east-west gradient of increasing Na, Cl and more positive $\delta^{18}O$-H_2O and $\delta^{2}H$-H_2O is likely in the shorter run more southern sub-catchments of Moffat and Craws creek (a tributary of Carran creek).

Detailed hypotheses include:

1. The bulk of Na and Cl in the waters of the catchment are derived from the wet (precipitation – rainfall, hail, snow) and dry (salts carried in air) deposition of marine aerosols. The Na and Cl concentrations of a given surface water capture zone will vary according to the centroid northing and altitude value for a capture zone.
2. The concentration of Na and Cl in soil and groundwaters will be significantly higher than that in precipitation, reflecting the evapotranspirative concentration of both ions within the soil zone.
3. There will be a clear N-S gradient of decreasing Na and Cl concentrations for precipitation collected within the catchment reflecting the control of altitude and distance from the coast over marine aerosol deposition.
    - For long run sub-catchments, such as Waituna Creek and Carran Creek, the concentration of Na and Cl at their southernmost monitoring points will be lower

than that of local groundwaters. This reflects the redistribution of water with low Na and Cl from the north to the south of the catchment.

- o Short run catchments such as Moffat Creek will show little variation in Na and Cl concentration relative to local groundwater.

4. Groundwaters, not influenced by the stream network, will mimic the N-S decline in Na and Cl concentrations observed in precipitation.

6.4.3 Atmospheric Hypothesis Testing

Precipitation, soil water, surface and ground waters from the Waituna Catchment fall predominantly along the Seawater Dilution Line (SWDL) (Figure 6.6). This indicates a dominance of both species associated with wet and dry deposition of marine aerosols. Small deviations from the SWDL indicates enrichment or depletion in each ion relative to seawater.

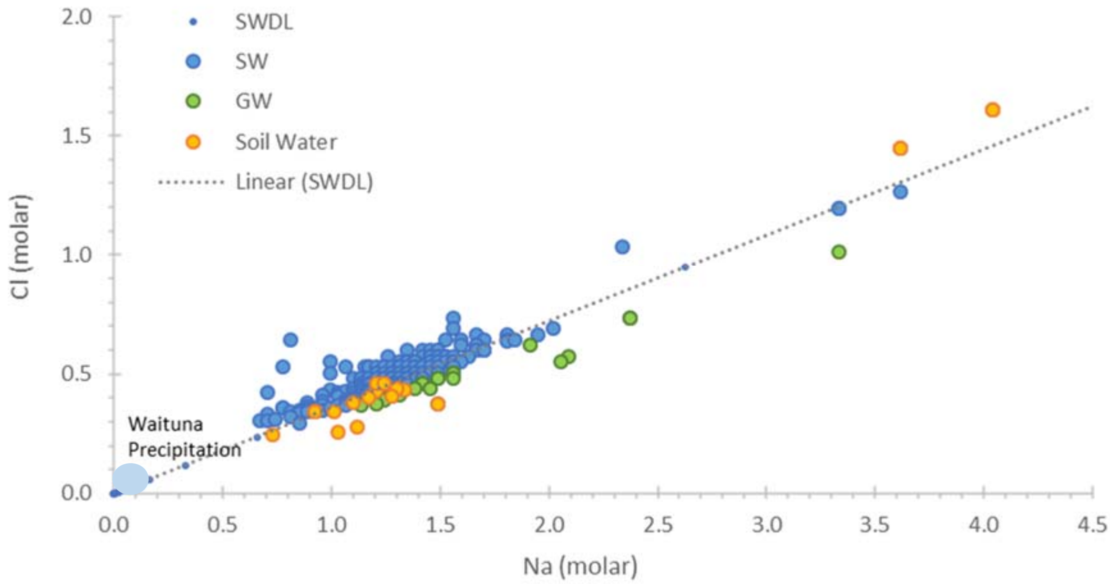


Figure 6.6: Waituna precipitation, soil water, surface (SW) and groundwater (GW), molar Na and Cl relative to seawater. SWDL = Seawater Dilution Line.

Evidence for a predominantly marine aerosol source for Na is further supported by calculation of the enrichment or deficit of Na relative to seawater and is assessed for the Waituna Catchment using the following equation (1):

$Fc = (C/Cl)_{sample}/(C/Cl)_{seawater}$ (Equation 1)

where Fc is the Fractionation Factor, and C is the concentration of a given species (e.g. Na). Fc is determined in relation to Cl since Cl is considered a conservative species. An $Fc > 1$ indicates an excess of the given species over the marine source, $Fc < 1$ indicates a deficit of the given species relative to the marine source and an $Fc = 1$ indicates unity with respects to seawater. The mean and median Fc for Na within Waituna surface waters is at unity, indicating a dominant marine source.

Magnesium (Mg) and sulphate (SO_4^-) are also important marine aerosols but occur in lower concentrations than Na and Cl in precipitation. Assessment of the Fc for Mg and SO_4^- indicates significant enrichment relative to seawater indicating a significant terrigenous source for SO_4^- and to a lesser degree Mg (Table 6.2). The most likely sources of terrigenous SO_4^- and Mg in the Waituna

Catchment are geogenic (weathering and biogeochemical cycling) and anthropogenic (fertilisers and animal wastes).

Table 6.2: Fractionation factors for sodium, sulphate and magnesium in Waituna catchment.

	Fc Na	Fc SO_4	Fc Mg
Valid Cases	293	293	293
Mean	1.0	3.1	1.8
Median	1.0	3.3	1.8
Coefficient of Variation	0.1	0.5	0.2
Minimum	0.7	0.1	0.8
Maximum	1.3	6.7	2.6
Range	0.6	6.6	1.7

Evapotranspirative enrichment of marine-derived Na and Cl within the soil zone is an important mechanism controlling the concentration of Na and Cl in soil, surface and ground waters within the Waituna Catchment. Precipitation collected from within the catchment has a median Na and Cl concentration of 8.8 and 13.7 mg/L, respectively (Table 6.3; Figure 6.7). These concentrations are slightly less than half that of the median values for soil (including tile drain waters) and groundwater. Median groundwater Na shows a slight enrichment relative to soil water probably due to greater interaction with aquifer materials and the subsequent addition of weathering-derived Na (Table 6.3).

Table 6.3: Summary statistics for sodium (Na) and chloride (Cl) in precipitation, soil water and groundwater (GW) in Waituna catchment.

	Na (mg/L)			Cl (mg/L)		
	Precip.	Soil Water	GW	Precip.	Soil Water	GW
Valid Cases	6	30	24	6	21	28
Mean	9.9	18.2	22.9	16.0	35.8	35.7
Median	8.8	18.5	23	13.7	34	32
Coefficient of Variation	0.8	0.3	0.3	0.8	0.3	0.4
Minimum	1.1	10.3	12.7	2.4	20.6	19.9
Maximum	19.4	32	44	33.9	67	94
Range	18.3	21.8	31.3	31.5	46.4	74.1

The pathway of Na and Cl enrichment for the Waituna Catchment is shown in Figure 6.7. Soil suction cup waters from the zone of direct contribution, show extreme enrichment which is thought to be due to wicking of moisture by peat moss. However, Na and Cl concentration in the peat moss declines with depth (0.05 – 0.65 m), producing a median concentration of ~50 mg/L Cl and ~30 mg/L Na, which is consistent with expected values for recharge waters within the zone of direct contribution. Enrichment and subsequent averaging of variation in the concentration of marine-derived salts with depth is a key process occurring within the soil zone (Clark and Fritz, 1997; Rissmann et al. 2016b and c).

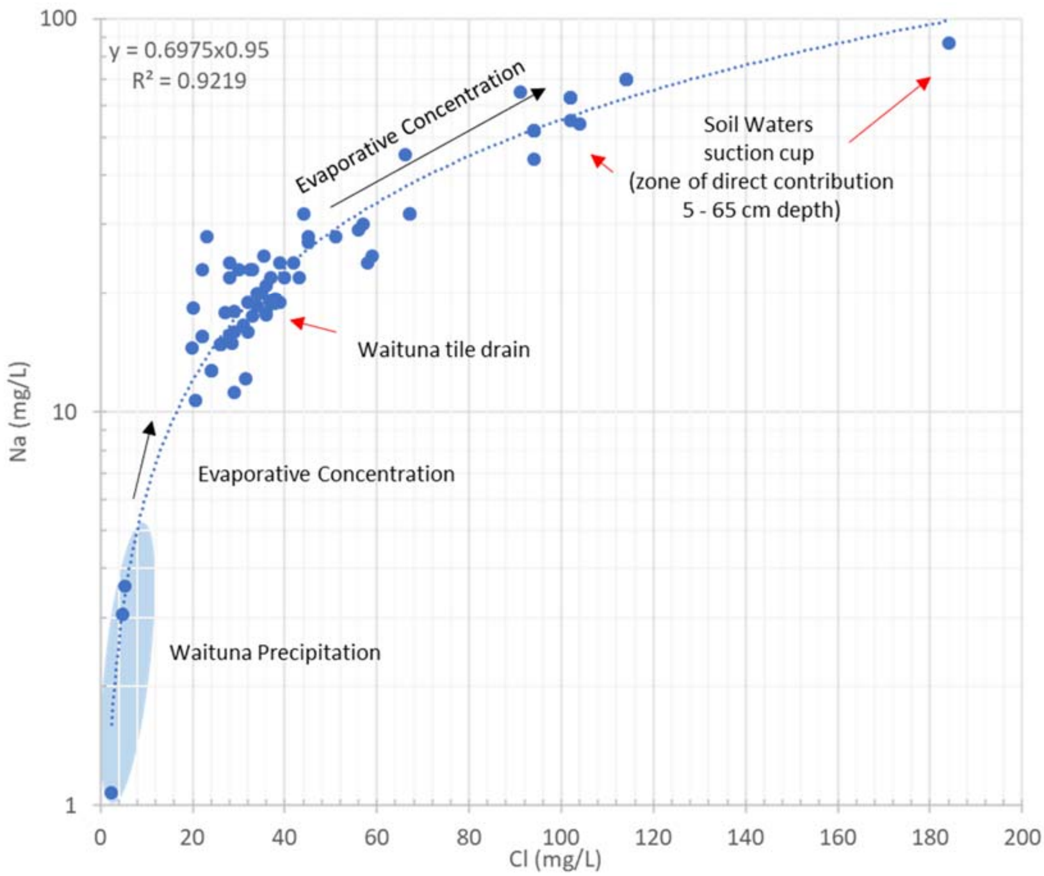


Figure 6.7: Evapotranspirative enrichment of Na and Cl relative to precipitation within the Waituna catchment.

Median surface water data from monitoring points within the Waituna sub-catchments show the change in Na and Cl concentrations down the catchments with the lowest concentrations of Na and Cl at the most southern monitoring point (Figure 6.8). The lower Na and Cl values reflect the export of low Na and Cl waters from the northern portions of each subcatchment. Moffat and Craws Creeks show little longitudinal variation in Na and Cl concentration, reflecting the short run of these subcatchments.

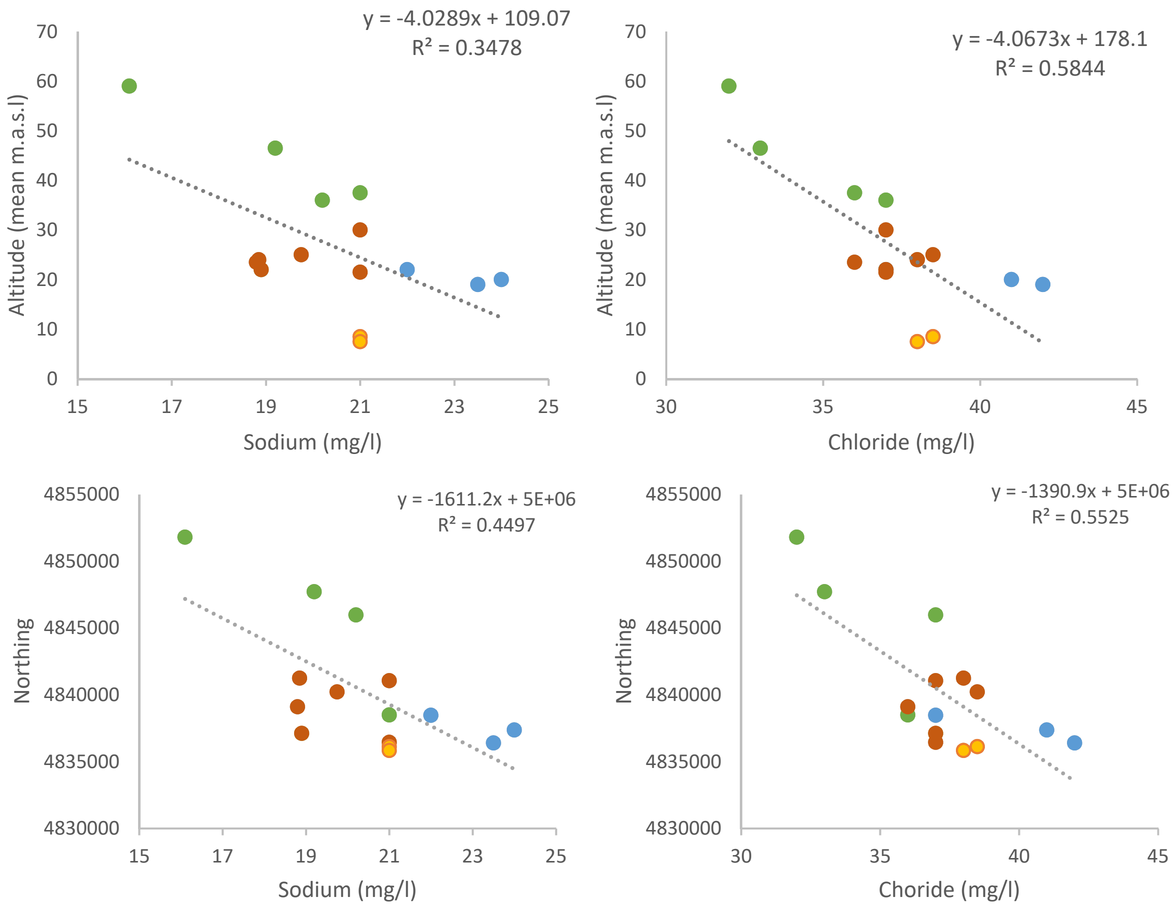


Figure 6.8: Altitude and northing of Waituna subcatchments by sodium and chloride concentration where Waituna creek is shown in green, Moffat creek in blue, Carran creek in orange and Craws creek in yellow.

Principal Component Analysis (PCA) was used to demonstrate the role of distance from the southern coast and altitude over marine aerosol concentration on log-transformed and z-scored data for Na and Cl measures for Waituna streams (Table 6.3). There are two principal components that are significant (variance >1) which explains 87% of the variation in the data. The coefficients show for component 1 (53% of variation) – altitude and northing are highly correlated, and Na and Cl are highly correlated. As altitude or northing increases, Na and Cl concentrations decrease. Component 2 (33% of variation) shows a relationship with easting with more eastern locations having a higher Na or Cl concentration.

PCA was also run on the smaller subset of $\delta^{18}O$-H_2O and $\delta^{2}H$-H_2O samples (Table 6.4). As hypothesised, PCA indicates the majority of variance in Na, Cl, $\delta^{18}O$-H_2O and $\delta^{2}H$-H_2O is associated with altitude and distance from the coast along a south-north gradient. This shift coincides with the general topographic gradient of the catchment and the regional evaluation of Na and Cl gradients (Rodway et al., 2016).

Table 6.4: PCA for Waituna surface water subcatchments.

Variances

Component	Variance	Proportion	Cumulative proportion
1	2.654	0.531	0.531
2	1.685	0.337	0.868
3	0.491	0.098	0.966

Coefficients	Component		
N = 589	1	2	3
Altitude (masl)	-0.536	0.334	-0.280
Easting (X)	0.127	-0.633	-0.758
Northing (Y)	-0.545	0.271	-0.400
Sodium (ppm)	0.445	0.455	-0.333
Chloride (ppm)	0.449	0.456	-0.277

Table 6.5: PCA for Waituna surface water subcatchments with the addition of the isotopes of water.

Coefficients	Component		
N = 32	1	2	3
Altitude (m RSL)	-0.420	-0.195	-0.483
Easting (X)	0.149	0.690	-0.029
Northing (Y)	-0.424	-0.106	-0.512
⍰^{18}O-H_2O	0.391	0.171	-0.569
⍰^{2}H-H_2O	0.442	0.100	-0.423
Sodium (ppm)	0.351	-0.513	-0.036
Chloride (ppm)	0.387	-0.416	0.016

The drivers of variation in Na and Cl concentration identified above were tested using PCA on the groundwater dataset for the catchment and surrounding bores. Table 6.6 shows that the northing and easting control the same amount of variation (87%) in the groundwater Na and Cl data compared to the median stream data (Table 6.4). Across the Waituna catchment, the proximity to the coast or northing is the most significant factor in influencing the precipitation signature for both Na and Cl (57% of variation). However, the easting gradient occurs for Na only, accounting for an additional 30% of the variation. This gradient in Na concentration is likely a result of weathering of the geological substrate (higher Na in the west associated with mineral substrates to low in the east as the geology becomes peat dominated) rather than marine aerosol loading.

Table 6.6: PCA on groundwater data.

Variances (n=44)

Component	Variance	Proportion	Cumulative proportion
1	2.274	0.568	0.568
2	1.199	0.300	0.868
3	0.328	0.082	0.950
4	0.200	0.050	1.000

Coefficients (n=44)

	Component			
	1	2	3	4
Northing	-0.558	0.319	-0.642	0.418
Easting	0.098	-0.866	-0.488	0.043
Sodium (ppm)	0.546	0.372	-0.591	-0.463
Chloride (ppm)	0.618	0.098	0.020	0.780

6.4.4 Mapping Method

The atmospheric or marine aerosol loading for the Waituna Catchment was assessed based upon recent shallow groundwater samples at 54 locations within the Waituna Catchment and a 15 km radius (Environment Southland data, 2014 – 2016). Groundwater chemistry was used as a proxy for direct measures of precipitation due to the lack of samples for the catchment. The groundwater data was quality assessed and controlled (QA/QC) by removing bores from the dataset which were identified as agriculturally influenced, confined groundwaters, or hydrologically connected to the surface water network. The resulting dataset (44 points) was clustered using Hierarchical Clustering Analysis (HCA - Ward's algorithm on log-transformed, z-scored Na and Cl data) to identify groups with similar variance. Cluster 1 was identified as water with a high marine aerosol load, while cluster 3 has the lowest marine aerosol load. A summary of the Na and Cl concentrations for each cluster is summarised in Table 6.7. The clusters were interpolated using the 'IDW (Spatial Analyst)' tool in ArcGIS (Figure 6.9). To produce the A-PAL, the interpolated data was classed according to the HCA clusters and is presented in Figure 6.10.

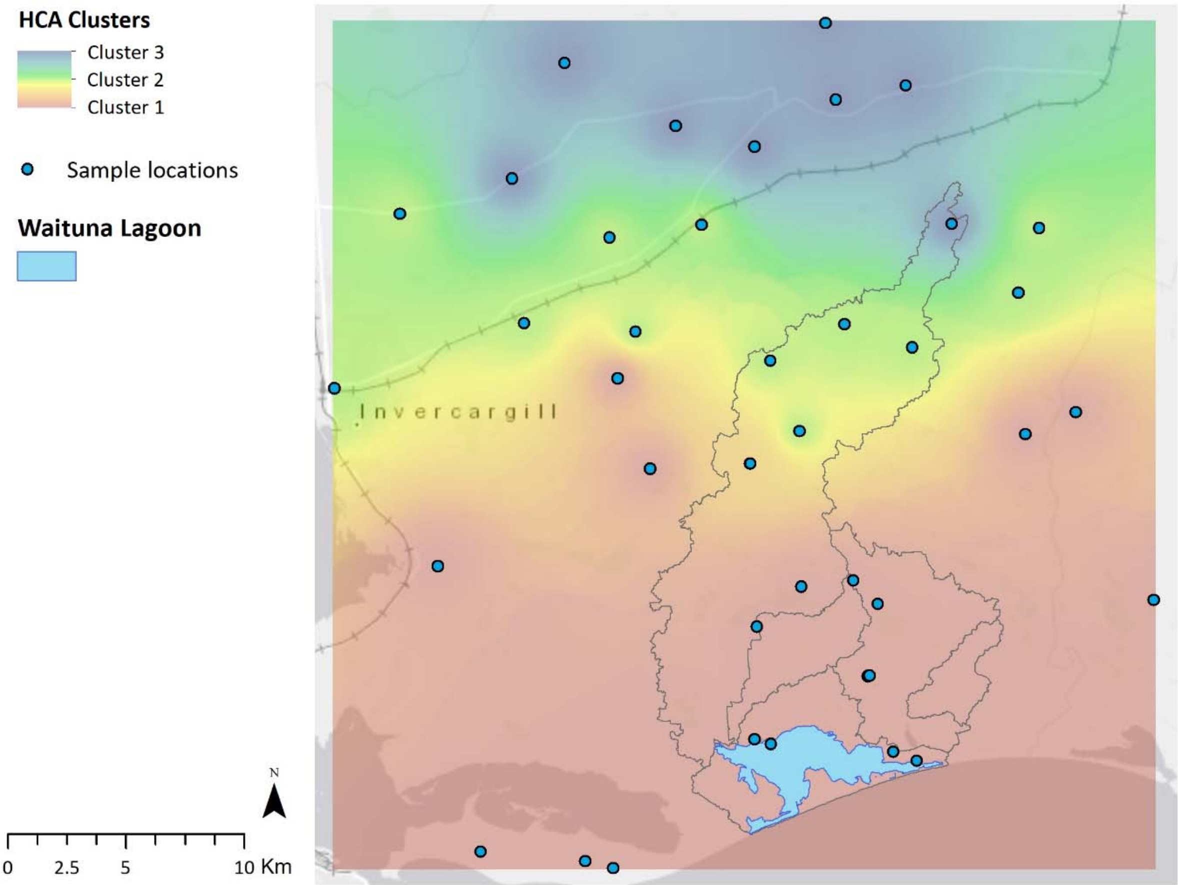


Figure 6.9: Sample locations and cluster interpolation of shallow groundwater samples.

Table 6.7: Summary statistics for marine aerosol loading clusters.

	Cluster 1	Cluster 2	Cluster 3
Aerosol loading class	High	Moderate	Low
Number of samples	21	15	8
Sodium (mg/L)			
Sodium - mean	26.7	20.1	19.9
Sodium - median	27.0	21.0	18.2
Sodium - minimum	21.0	13.4	16.6
Sodium - maximum	34.0	24.0	28.0
Sodium - range	13.0	10.7	11.4
Chloride (mg/L)			
Chloride - mean	47.4	27.9	21.1
Chloride - median	50.0	28.0	21.0
Chloride - minimum	36.0	22.0	19.3
Chloride - maximum	59.0	35.0	23.0
Chloride - range	23.0	13.0	3.7

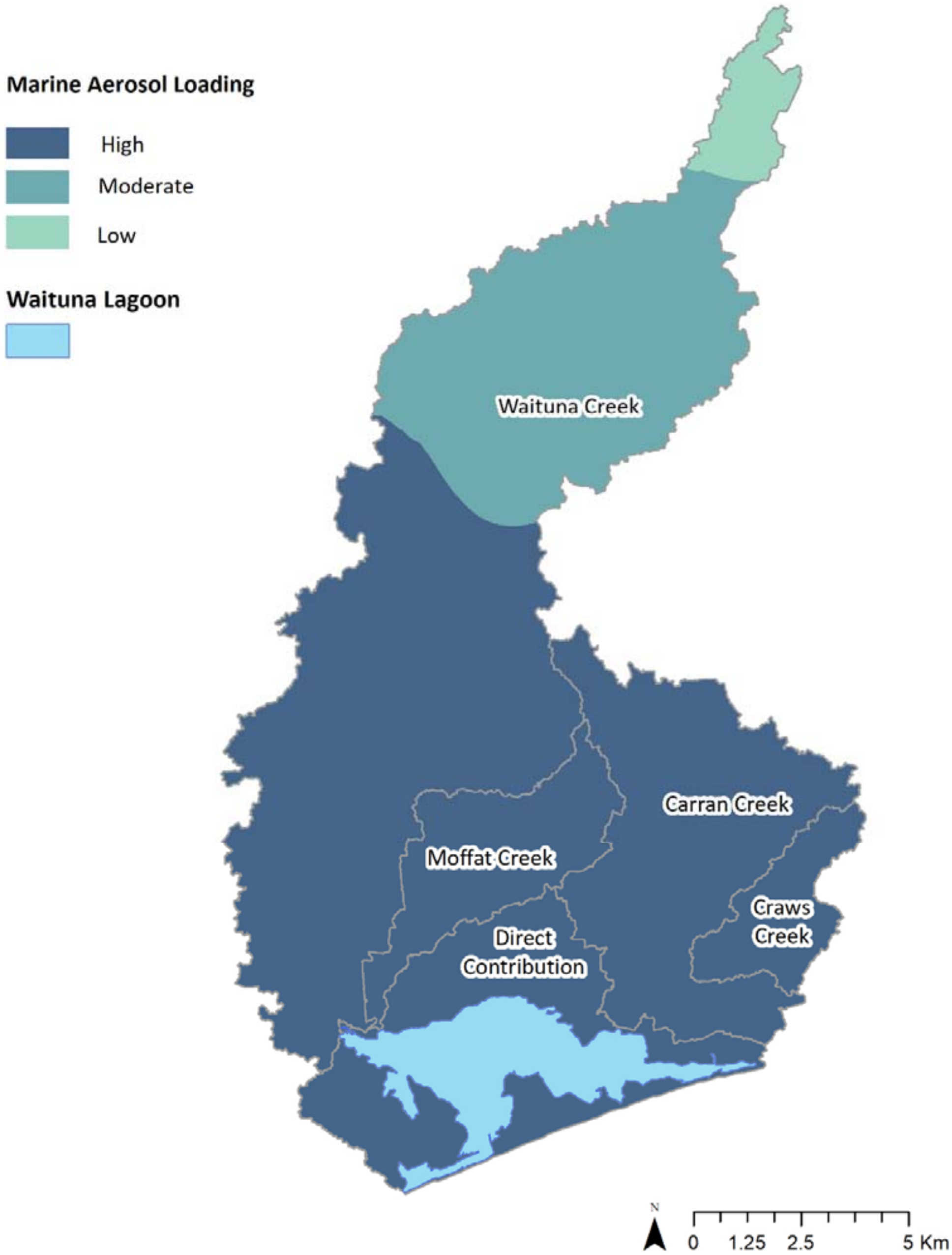


Figure 6.10: Atmospheric Process-Attribute Layer (A-PAL) for the Waituna Catchment.

6.4.5 A-PAL Catchment Summary

As the catchment area is small (~19,000 ha) with low relief (< 80 m), variation in precipitation composition is a minor driver of water quality variation within the Waituna Catchment. Waituna Creek is the only subcatchment to exhibit a range of marine aerosol loading due to the length of the catchment (distance from the coast) when compared to Moffat and Carran Creek which are comparatively short and coastal. Table 6.8 provides a summary of the marine aerosol loading by catchment.

Table 6.8: Atmospheric Process-Attribute Layer for Waituna subcatchments.

	Waituna Creek		Moffat Creek		Carran and Craws Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
High	6328.5	56.7	1545.4	100	4287.8	100	2296.3	100	14458.0	74.9
Moderate	4304.8	38.6	-		-		-		4304.8	22.3
Low	528.1	4.7	-		-		-		528.1	2.7

6.5 Hydrological Process-Attribute Layer (H-PAL) – Transport layer

The purpose of the Hydrological PAL (H-PAL) is to characterise the source of surface and groundwaters, which includes identification of the mechanism by which water is recharged, the degree of hydrological connectivity to the broader hydrological network and identify smaller scale variation in the landscape attributes governing the likely path water takes across the landscape.

For the Waituna Catchment, variation in soil hydrological properties and attendant modification by artificial drainage is the primary driver of spatial variation in hydrological processes as all water, including shallow groundwater, is derived from local precipitation with no evidence of a distal water source associated with Hill or Alpine recharge domains (Figure 6.1; Hughes, 2003; Wilson, 2011; Rissmann et al., 2012)

Identifying variation in hydrological attributes is critical to understanding the hydrological response and as such the likely pathway(s) water will take – overland flow (OLF), lateral soil drainage mediated by artificial sub-surface drainage and/or deep drainage to the underlying aquifer. As discussed in Section 4, the pathway water takes exerts a strong control over the type and magnitude of water quality issues making it an important feature of any study attempting to better understand the natural controls over water quality. Here, it is important to remember that it is the initial ('natural state') hydrological properties of a soil that determine the degree of modification by artificial drainage for a given land use intensity (see Sections 2 and 4).

6.5.1 Relevance

Water is the vehicle that transports land use derived contaminants from the land to water and the perennial hydrological network is the key distributor. Drainage from the northern portion of a catchment may be a key control over water composition and quality at its most distal sampling point. In addition to the perennial stream network, finer grained variation in topography and soil hydrological properties determine the pathway water takes to the stream channel. Specifically, deep drainage or 'vertical percolation' of water through the soil to underlying aquifers; lateral drainage, where water mainly moves horizontally through the soil zone, commonly in association with subsurface artificial drains to an open drain or surface water body, and; OLF that results in water running off across the land surface directly to open ditches or natural waterways.

The pathway water takes from the land to stream is a strong influence over the type of water quality outcomes (e.g. sediment vs. nitrate) as well as the magnitude of export. Specifically, it is widely recognised that the export of sediment, nutrients and microbes generally increases across the deep > lateral > overland (or surficial) pathway continuum in intensively farmed catchments (Figure 6.11). Therefore, when attempting to understand the spatial variation in water quality within a distributed hydrological network it is important to recognise the source of water and the probable hydrological pathways water has taken to the stream channel. The H-PAL layer produced here is designed to be used in conjunction with finer scale flow path mapping (i.e., paddock scale mapping of runoff pathways to identify discharge points).

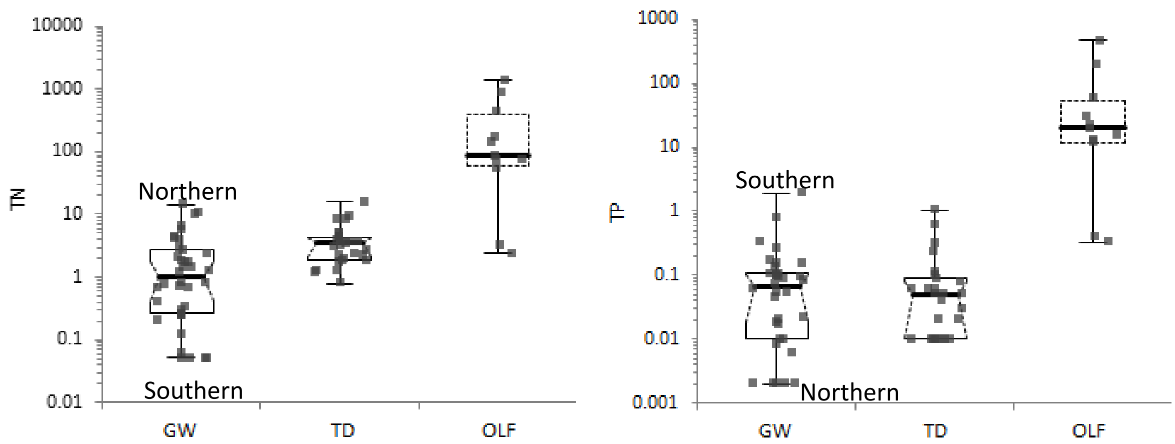


Figure 6.11: Concentration of total nitrogen (TN) and total phosphorus (TP) in groundwater (GW), tile drains (TD) and overland flow (OLF) from the Waituna catchment (Environment Southland Data).

6.5.2 Hydrological Process Attribute (H-PAL) Hypotheses

The general hypotheses for the generation of the H-PAL layer are as follows:

1. Water Source: All freshwater water within Waituna catchment is derived from local precipitation with no evidence of Hill or Alpine sourced water.
2. Hydrological Pathway: For the larger modified area of the catchment, soil drainage class (class 1 – 5; very poorly drained to well drained) plays an important role over the pathway water takes, specifically:
    - Lateral drainage mediated by mole-pipe drainage will become increasingly important as soil drainage class decreases from well > moderately well > imperfectly > poorly > very poorly drained.
    - The percentage of precipitation occurring as overland flow (OLF) will increase as drainage class decreases.
    - Vertical drainage to the aquifer will dominate in areas of well-drained soils.

The role of soil drainage over the flow path will be evident in:

1. An increase in the median turbidity and TSS concentrations of streams as the proportion of poorly drained soils increases within the capture zone of a monitoring point for the intensively farmed portion of the catchment, reflecting greater sediment export via subsurface artificial drainage and OLF.
2. An increase in median sediment and microbial export as the proportion of poorly drained soils increases within the capture zone of a monitoring point for the intensively farmed portion of the catchment, reflecting greater sediment export via subsurface artificial drainage and OLF.

6.5.3 Hypothesis Testing

Water Source

Within a cloud fractionation of ^2H-H_2O (‰, V-SMOW) and ^{18}O-H_2O (‰, V-SMOW) isotopes occurs as an air mass moves from the source (ocean) to higher altitudes. Specifically, precipitation becomes increasingly more negative with distance from the coast and with altitude. This fractionation occurs under equilibrium conditions preserving the relationship between $\delta^{18}O$ and δ^2H, resulting in a meteoric water line with a slope close to 8 (Clark and Fritz, 1997). As fractionation is strongly controlled by temperature, surface and groundwater data should be compared to the Local Meteoric Water Line (LMWL). Southland's LMWL, shown in Figure 6.12, is established from 908 surface water and groundwater samples from across the region (Environment Southland Data).

Water sampled from within the Waituna Catchment fall at the low altitude, coastal end of the LMWL for Southland which is consistent with the location of the catchment (Figure 6.12). Event flow samples for Waituna Creek at Marshall Rd show a more negative signal which is consistent with high-intensity rainfall events. The median stable isotopes of water for Waituna Catchment surface water monitoring sites show there is variation within the subcatchments with the most coastal/southern sites have more positive values, while the more northern, higher altitude, catchment of Waituna Creek, produce the most isotopically light waters (Figure 6.13).

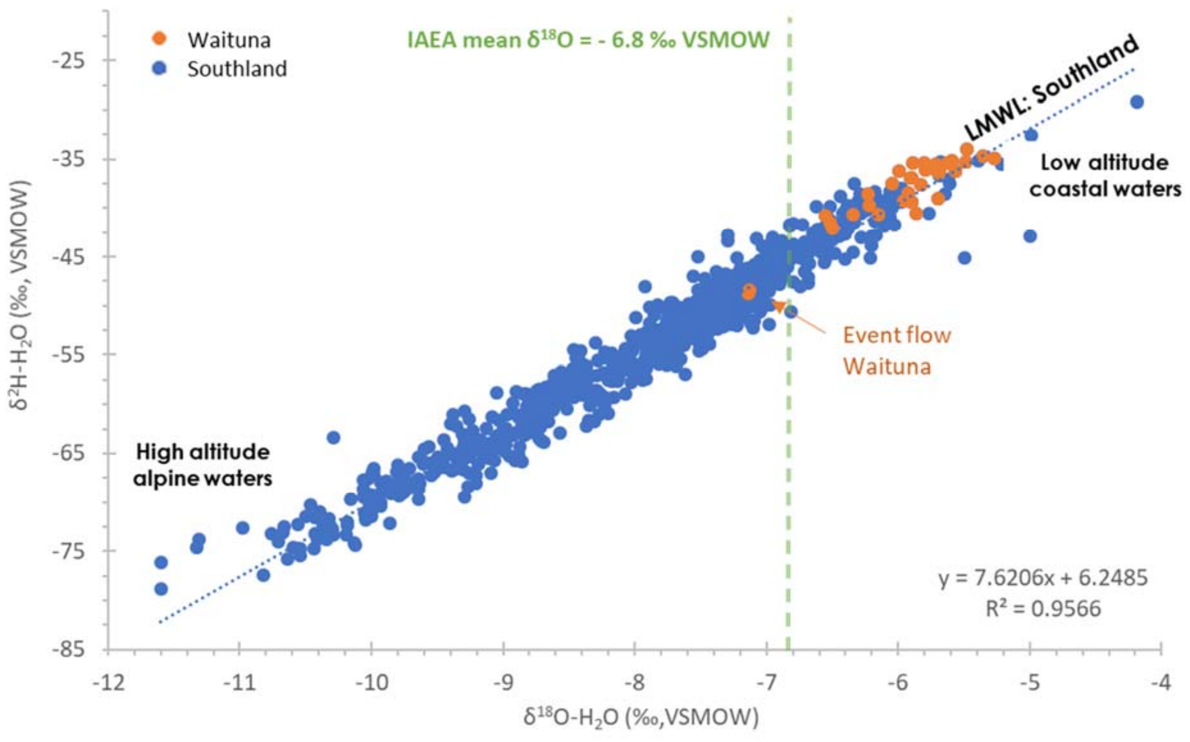


Figure 6.12: Plot of the stable isotopes of water for Southland surface waters and groundwaters (blue, n = 908 samples) with Waituna Catchment surface waters identified in orange. LMWL = Local Meteoric Water Line for Southland (Environment Southland Data).

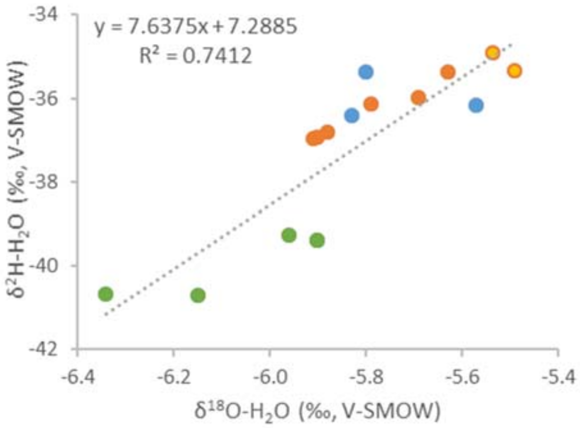


Figure 6.13: Median stable isotopes of water for Waituna Catchment surface water monitoring sites showing Waituna Creek (green), Moffat Creek (blue), Carran Creek (orange) and Craws Creek (yellow).

Hydrological Pathway

As there is no direct measure of the hydrological pathway, the classification is demonstrated by proportionally weighting the capture zone area by pathway and soil drainage (Figure 6.14). In developed areas, as soil drainage becomes increasingly more poorly drained, overland flow and artificial drainage increase, whilst the proportion of deep drainage decreases. Natural state areas are not expected to be artificially drained.

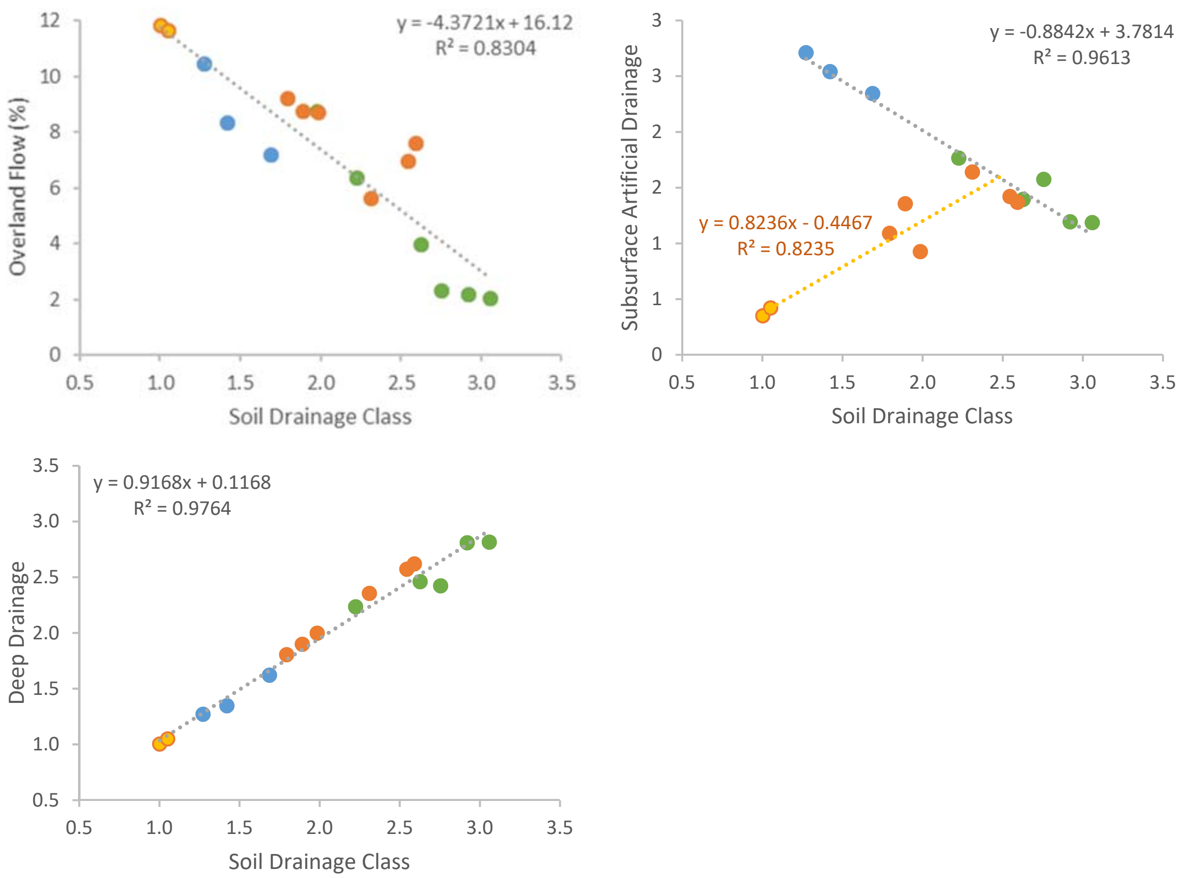


Figure 6.14: Relationship between overland flow, subsurface artificial drainage, and deep drainage by drainage class (1 – very poorly drained to 5 – well drained) - proportionally weighted for the subcatchments Waituna Creek (green), Moffat Creek (blue), Carran Creek (orange) and Craws Creek (yellow).

Hypotheses as to the role of soil drainage class over the pathway water takes to the stream network were assessed by PCA on log-transformed and z-scored data for soil drainage class and TSS (Table 6.9). The results of PCA for the developed portions of the catchment indicate a strong control by soil drainage class over TSS concentration. Specifically, higher TSS concentrations for capture zones with larger areas of poorly to very poorly drained soils (Figure 6.15). Similar PCA results were obtained when soil drainage class was swapped for other measures of soil hydrological status such as soil organic carbon content, soil chroma and percentage mottling - soil drainage class is considered a master variable with respects to these attributes. Turbidity shows a similar albeit slightly weaker relationship with soil drainage class, increasing as the proportion of poorly drained soils increases within the capture zone of a monitoring site. Clarity was not included as it is a measure of the transmission of light which is strongly influenced by tannins derived from wetlands.

Table 6.9: PCA for Waituna surface water subcatchments.

Variances

N = 987

Component	Variance	Proportion	Cumulative proportion
1	1.143	0.381	0.381
2	1.043	0.348	0.729
3	0.814	0.271	1.000

Coefficients

	Component		
	1	2	3
Clarity	0.009	-0.902	-0.432
TSS	0.705	0.312	-0.637
Soil Drainage Class	0.709	-0.299	0.639

6.5.4 Mapping Method

As water quality is strongly controlled by hydrology and soil hydrological response, the first step in the refinement process was to discriminate between areas likely to exhibit natural state versus a modified hydrological response. This was achieved by incorporating land parcel boundaries for Department of Conservation estate and areas protected by QEII covenants for the catchment. The property boundaries were used to infer where the local hydrological properties of the soil zone are likely to be in a natural state, rather than land cover previously used by Pearson (2015a).

The hydrological flow pathways for overland flow, artificial drainage, and deep drainage described in Sections 4.2 – 4.4 were incorporated into the H-PAL by refining and classing each individual layer as follows. The artificial drainage assessment identified drainage densities for the Waituna Catchment ranging from moderate to high and very high (Section 4.2.1). This was scaled to the Waituna Catchment by reclassifying the classes into natural state (none), low, moderate, and high and subsequently validated against maps of the digitised artificial drainage network (Section 4.3.2). Deep drainage (Section 4.4.1) for the Waituna Catchment was reclassified according to the artificial drainage classification; where deep drainage is low, the artificial drainage density is high, moderate where the artificial drainage density is moderate and high, where the artificial drainage densities are low. For natural state areas, the soils internal drainage properties where used where very poorly drained peats are classified as low, poorly to imperfectly drained areas as moderate and well drained as high. Overland flow susceptibility was incorporated into the H-PAL by identifying the percentage thresholds of overland flow risk and aligning these with the classes for artificial drainage and deep drainage. This resulted in the generation of three OLF classes for the catchment: <2% of effective precipitation, 2-6% of effective precipitation, and >6% of effective precipitation as overland flow. The low artificial drainage, high deep drainage class has an overland flow risk from <2 up to 12% due to the change in soil hydrological properties and slope and was therefore split into three separate classes. The resulting H-PAL classification is presented in Figure 6.15 and Table 6.10.

Table 6.10: Flow pathway classification for Waituna Catchment.

Artificial drainage	**Deep drainage**	**Overland flow**
Natural State		
No artificial drainage	High deep drainage	<2% effective rainfall as overland flow
No artificial drainage	Moderate deep drainage	2-6% effective rainfall as overland flow
No artificial drainage	Low deep drainage	>6% effective rainfall as overland flow
Developed Land		
Low artificial drainage	High deep drainage	<2% effective rainfall as overland flow
Low artificial drainage	High deep drainage	2-6% effective rainfall as overland flow
Low artificial drainage	High deep drainage	>6% effective rainfall as overland flow
Moderate artificial drainage	Moderate deep drainage	2-6% effective rainfall as overland flow
High artificial drainage	Low deep drainage	>6% effective rainfall as overland flow

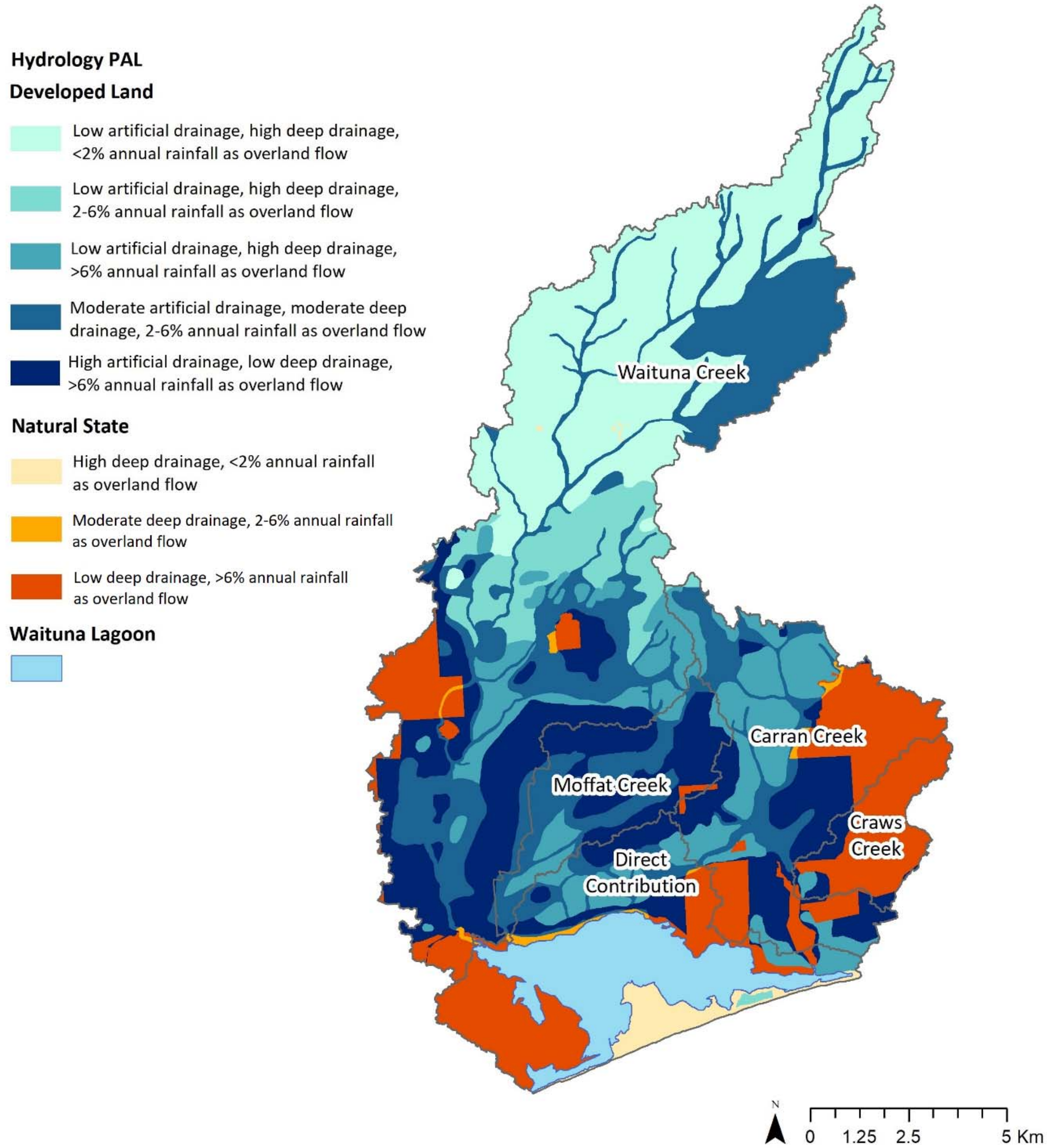


Figure 6.15: Hydrological Process-Attribute Layer (H-PAL) for the Waituna catchment.

6.5.5 Catchment Summary

Waituna Creek has the largest variation in hydrological flow pathways from the north to the south of the catchment (Table 6.11). Areas classed as low overland flow, low artificial drainage, and high deep drainage are located to the north of Waituna Creek catchment only. Moffat and Carrans Creeks have a higher proportion of poorly drained soils resulting in a high proportion of artificial drains and overland flow (Table 6.11).

Table 6.11: Hydrological Process-Attribute Layer for Waituna subcatchments.

	Waituna Creek		Moffat Creek		Carran Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
Developed Land										
High deep drainage, Low artificial drainage, < 2% annual rainfall as overland flow	4,090	36.7							4,090	21.2
High deep drainage, Low artificial drainage, 2-6% annual rainfall as overland flow	1,051	9.4			13	0.3	19	0.8	1,082	5.6
High deep drainage, Low artificial drainage, > 6% annual rainfall as overland flow	716	6.4	191	12.4	1,062	24.8	399	17.4	2,369	12.3
Moderate deep drainage, Moderate artificial drainage, 2-6% annual rainfall as overland flow	3,000	26.9	648	41.9	532	12.4	232	10.1	4,412	22.9
Low deep drainage, High artificial drainage, > 6% annual rainfall as overland flow	1,680	15.1	688	44.5	1,004	23.4	330	14.4	3,702	19.2
Total developed land	10,537	94.5	1,527	98.8	2,610	60.9	980	42.7	15,654	81.2
Natural State										
High deep drainage, < 2% annual rainfall as overland flow	6	0.1					334	14.6	341	1.8
Moderate deep drainage, 2-6% annual rainfall as overland flow	584	5.2	16	1.0	1,646	38.4	936	40.8	3,182	16.5
Low deep drainage, > 6% annual rainfall as overland flow	25	0.2	2	0.2	29	0.7	45	2.0	102	0.5
Total natural state	615	5.5	18	1.2	1,675	39.1	1,316	57.3	3,624	18.8
Total area	11,152	100	1,545	100	4,285	100	2,296	100	19,278	100

6.6 Redox Process-Attribute Layer (R-PAL)

The redox potential of soil and subsurface geology defines spatial gradients in the biogeochemical controls over nitrogen (N) and phosphorus (P) attenuation and mobility. Redox processes also influence other aspects of water composition including the concentration of the dissolved forms of oxygen (DO), manganese (Mn^{2+}), iron (Fe^{2+}) and sulphate (SO_4^{2-}). The key drivers of redox potential within the Waituna Catchment are soil drainage class (soil zone redox) and the organic carbon content of shallow aquifers (Rissmann, 2011; Rissmann et al., 2012 and 2013). In the majority of studies investigating the biogeochemical controls over water quality outcomes redox is routinely identified as a major driver of variation in water quality outcomes and hydrochemical composition (McMahon and Chapelle, 2008; Rissmann, 2011; Rissmann et al., 2012).

6.6.1 Relevance

Redox is recognised as one of the most important processes controlling variation in water quality, both nationally and internationally (Moldan and Cerny, 1994; McMahon and Chapelle, 2008; Tratnyek et al., 2012). Within the Waituna Catchment and from studies of redox dynamics around the world, redox processes are known to control the form, and hence the mobility, of both N and P in addition to driving the process of denitrification through which NO_3 (nitrate) nitrogen is removed from water (Rissmann, 2011; Rissmann et al., 2012; Rismann and Hodson, 2013). It is important to note that whilst redox processes influence the mobility and form of P, they do not result in the removal of P. Redox processes also control the concentration of dissolved oxygen (DO), dissolved manganese (Mn^{2+}), dissolved iron (Fe^{2+}), sulphate (SO_4^{2-}) and the production of greenhouse gases such as the oxides of nitrogen (N_xO), hydrogen sulphide (H_2S) and methane (CH_4). Ammoniacal forms of nitrogen are also produced under reducing conditions in areas with high organic carbon content (Ponnamperuma, 1972; Moldan and Cerny, 1994; McMahon and Chapelle, 2008; Tratnyek et al., 2012). Heavy metal solubility and mobility are also significantly influenced by redox processes.

In low temperature systems, redox evolution is mediated by bacteria and follows a successional sequence of Terminal Electron Accepting Processes (TEAPs): $O_2 > NO_3 > Mn^{4+} > Fe^{3+} > SO_4^{2-} > CO_2$ (Figure 6.16). Successional evolution of TEAP results in the transformation of O_2 to H_2O, NO_3 to gaseous N_XO (nitrous oxides) and/or N_2 (inert dinitrogen gas), Mn and Fe oxides and oxyhydroxides to dissolved Mn^{2+} and Fe^{2+}, and SO_4^- (sulphate) and CO_2 (carbon dioxide) to gaseous H_2S (hydrogen sulphide) and CH_4^+ (methane), respectively. The reduction of CO_2 to CH_4^+ represents the end point of the microbially mediated redox succession and is associated with the most reducing environments and subsequently waters. Peat aquifers are one example of where the abundance of organic carbon and the absence of appreciable ferric oxides and oxyhydroxides results in the evolution of CO_2 reducing conditions. Wetland soils are also known to produce CH_4 through redox succession (Ponnamperuma, 1972). In other settings where electron donors are less abundant and/or drainage is good redox succession may be limited, with the system poised at O_2 reduction. In these systems, leached nitrate is able to accumulate without being removed. Across Southland, the full range of TEAP is observed, however, Fe^{3+} reduction is by far the most common (Beyer and Rissmann, 2016; Rissmann et al., 2016).

Importantly, although an electron donor source is the primary control over redox succession, soils are not typically limited with respects to electron donors with an often-abundant supply of organic carbon from the upper horizons of the soil. Under these conditions, the frequency and duration of soil saturation is the key control over soil reduction potential (Seitzinger et al., 2006; Clough et al., 1998). As soil pores become saturated (at any level in the soil profile) the supply of meteoric oxygen is ‘shut-off’ kick-starting the chain of TEAP succession. The role of saturation of the soil over redox progression reflects the c. 10,000-fold reduction in the diffusivity of molecular O_2 through water relative to air-filled pores. Accordingly, the degree and duration of soil saturation exerts a key control over soil zone reduction with the indicators of soil reduction including low chroma colours, ‘gleying’ and ‘percent mottling,’ increasing along the soil drainage continuum (Ernstsen et al., 2006; Beyer et al., 2016; Webb et al., 2010). Due to the strong correlation between soil saturation and soil colour (chroma), percent mottling and iron pan formation these visual indicators of redox evolution are often used to assess soil drainage class (Milne et al., 1995). As organic soils are the result of organic matter accumulation under saturated conditions, redox succession is often advanced.

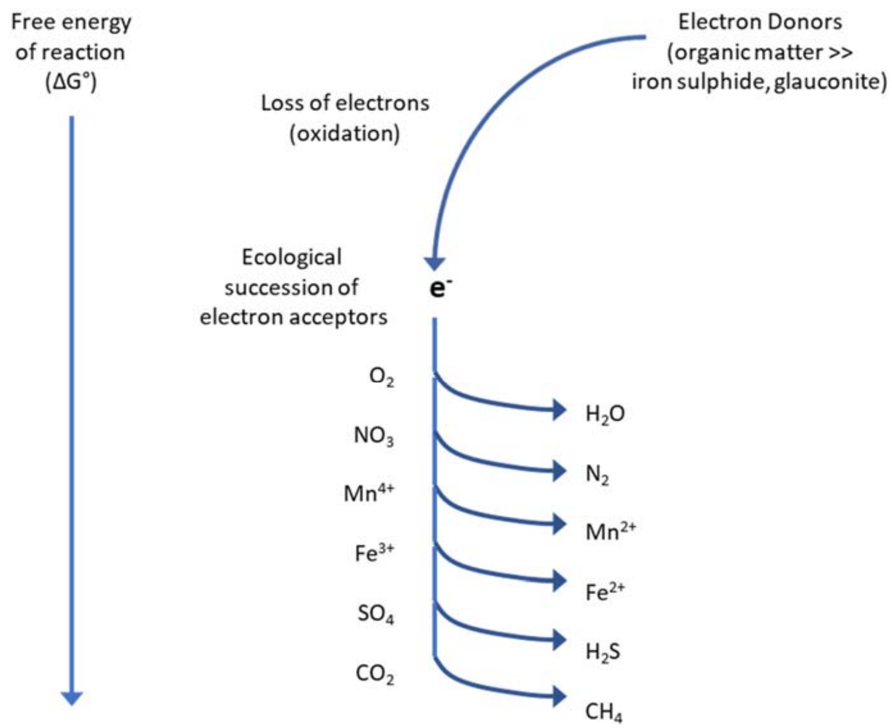


Figure 6.16: The ecological succession of terminal electron acceptors in natural waters (modified from McMahon and Chapelle, 2009).

6.6.2 Redox Process Attribute Layer (R-PAL) Hypotheses

The hypotheses for the process-attribute gradients governing redox processes within the Waituna Catchment are as follows:

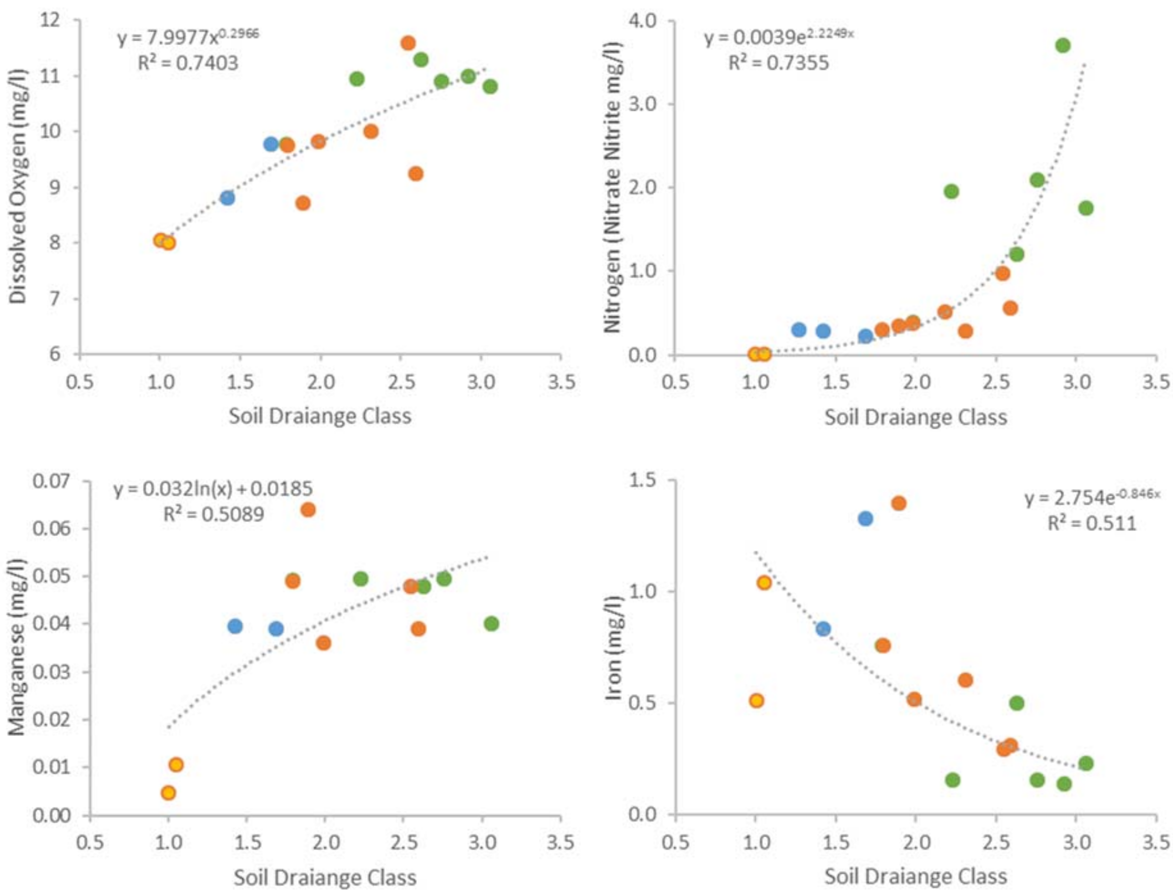
1. As the groundwater contribution to streamflow is small in the Waituna Catchment, the majority of water supplied to the stream will originate from lateral soil drainage and lower frequency surficial runoff. As such, redox signatures in the streams of the catchment will primarily be controlled by the drainage class of soils within the capture zone of a surface water monitoring point, specifically:
    - DO and NNN will decrease as the proportion of poorly drained soils increases;
    - Dissolved Fe^{2+} will increase as the proportion of poorly drained soils increases. This is important because the reductive dissolution of the oxides and oxyhydroxides of iron strongly influence phosphorus retention.
    - Soil organic carbon will increase as the proportion of poorly drained soils increases.
    - Dissolved organic carbon concentration will increase as the proportion of poorly drained soils increases.
2. The reductive dissolution of the oxides and oxyhydroxides of Fe (ferric; Fe^{3+}) and Mn (Mn^{4+}) will increase as the proportion of organic soils and aquifers increase within a capture zone of any given monitoring point:
    - This will be reflected in an increase in Fe^{2+} and to a lesser degree Mn^{2+} as the proportion of organic soils and aquifers increase within a capture zone

3. The role of reductive dissolution of oxides and oxyhydroxides within soils and aquifers over P concentration and mobility will be reflected in a positive correlation between P (DRP and TP) and Mn^{2+} and Fe^{2+} concentrations in surface waters.
4. Aquifer redox conditions will only be of importance under low flows (baseflow), with a decline in DO and an increase in reduced species as aquifer level and subsequently, baseflow contribution wanes (Rissmann et al., in prep).

6.6.3 Hypothesis Testing

Soil drainage class

As hypothesised the figures below show a strong relationship between soil drainage class and median concentrations of redox-sensitive species and is particularly strong for DO and NNN (Figure 6.17). More specific discussion of the landscape controls over both DRP and TP are provided in Section 7.



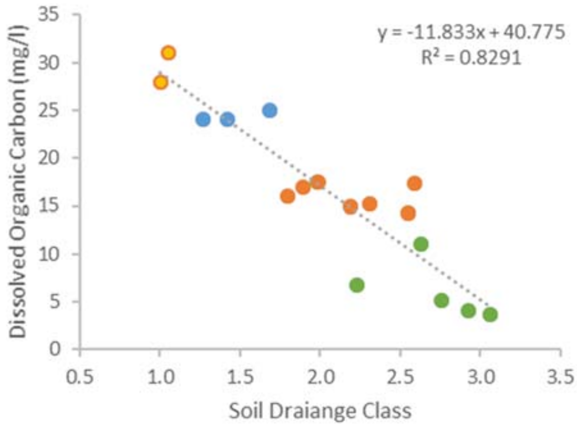
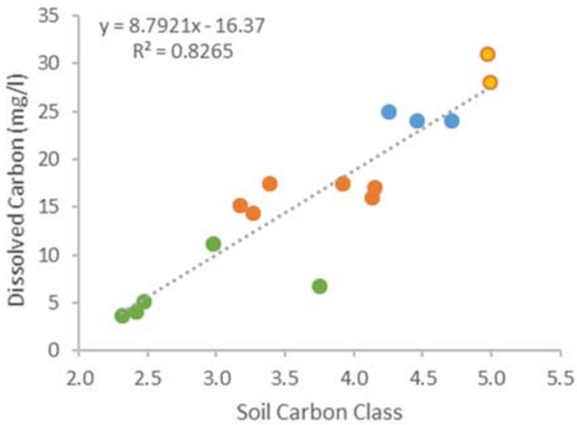


Figure 6.17: Dissolved oxygen, nitrogen (NNN), manganese, iron, and dissolved organic carbon by soil drainage class (1 – very poorly drained to 5 – well drained) proportionally weighted for the subcatchments Waituna Creek (green), Moffat Creek (blue), Carran Creek (orange) and Craws Creek (yellow).

Carbon content

Organic carbon is typically not limited in pastoral soils and is correlated with soil drainage class. Dissolved carbon in streams reflects the proportion of carbon in soils and aquifers (Figure 6.18). Soil organic carbon content exerts an important control over the stability of the sesquioxides of Al, which are an important mineral governing P-retention. As organic carbon content increases, acidity increases, reducing the stability of Al sesquioxides and reducing the ability of soils and aquifers to retain P.



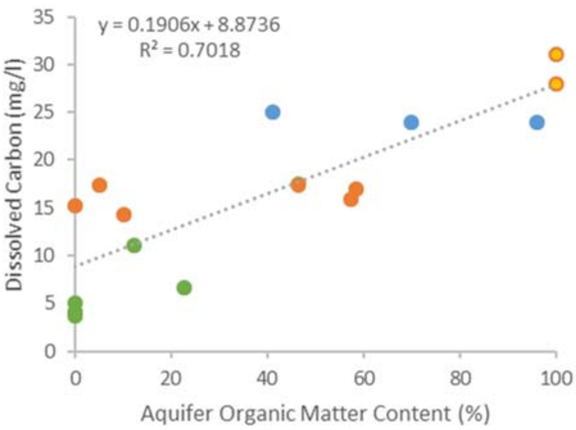


Figure 6.18: Dissolved carbon by soil carbon class (1. <2%, 2. 2-4%, 3. 4-10%, 4. 10-20%, 5. >20%), and aquifer organic matter (%) proportionally weighted for the subcatchments Waituna Creek (green), Moffat Creek (blue), Carran Creek (orange) and Craws Creek (yellow).

Indirectly controlled

As noted above, soil drainage class and associated soil organic carbon content play an important role over the retention of P. The sesquioxides of Fe and Al are the main minerals governing P retention in soil but also aquifers. As soils become more reducing (poorly drained) the sesquioxides of Fe are reduced and dissolved, resulting in lower volume and surface area for P-retention. Reductive dissolution of Fe sesquioxides is associated with low chroma (gleying) colours and redox segregations (mottles). The sesquioxides of Al are less stable under acidic conditions with soil zone and aquifer acidity increasing as the proportion of organic carbon increases. The combination of reductive dissolution of the sesquioxides of Fe and the destabilisation of the sesquioxides of Al play a dominant role over P-retention. The relationship between soil drainage class and soil organic carbon content and both DRP and TP are shown below (Figure 6.19).

These figures show a strong correlation for DRP, although TP shows more scatter. The latter is thought to reflect the concentration of phosphorus species, especially the particulate P-fraction, at or near the land surface and as such its susceptibility to transport via OLF. In comparison, DRP is considered more mobile, existing in the dissolved form. More discussion as to the landscape controls over both DRP and TP are provided in Section 7.

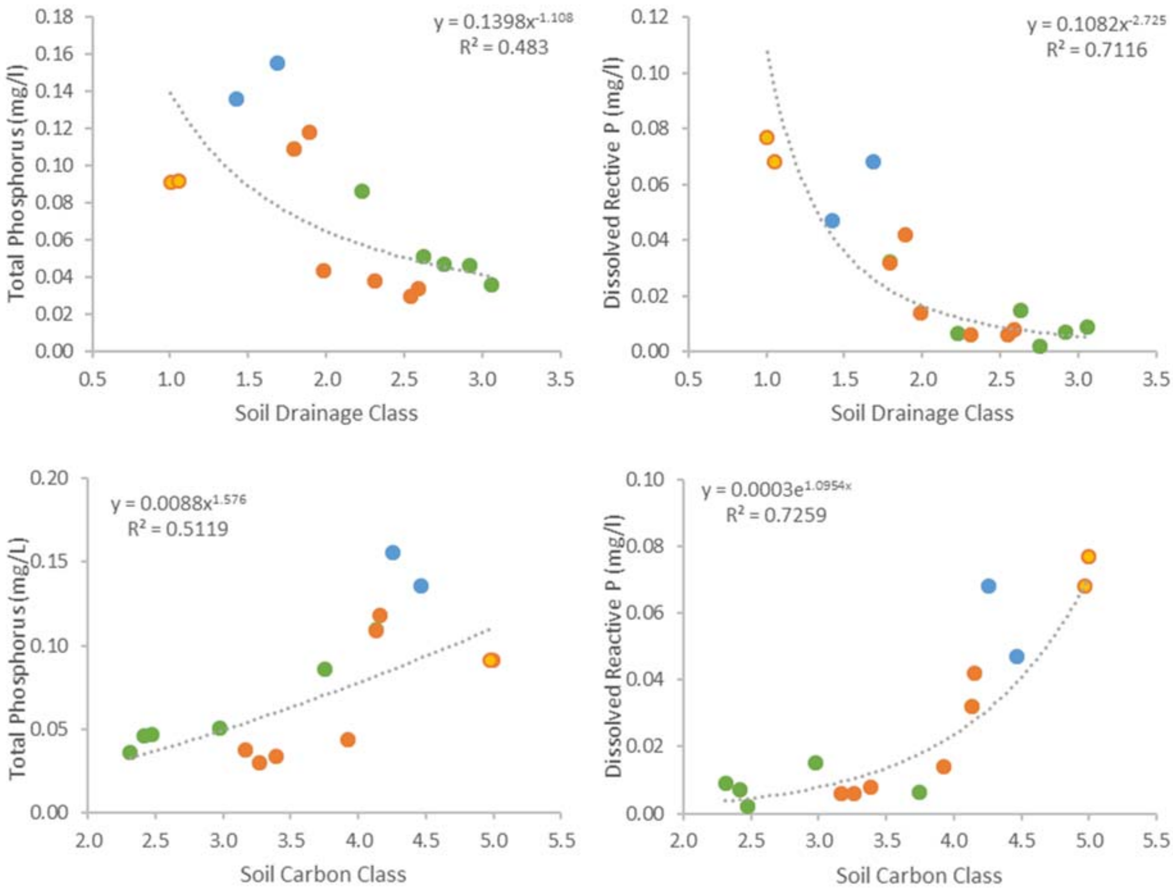


Figure 6.19: Total phosphorus and dissolved reactive phosphorus by soil drainage class (1 – very poorly drained to 5 – well drained) and by soil carbon class (1. <2%, 2. 2-4%, 3. 4-10%, 4. 10-20%, 5. >20%) proportionally weighted for the subcatchments Waituna Creek (green), Moffat Creek (blue), Carran Creek (orange) and Craws Creek (yellow).

6.6.4 Mapping Method

The R-PAL for the Waituna catchment was developed by classifying the soil (Topoclimate South, 2001; NZ Land Resource Inventory, DSIR, 1968) and geological (QMAP; Turnbull and Allibone, 2003) layers by their reduction potential. The soil reduction potential (SRP) is primarily influenced by soil drainage class as organic carbon is seldom a limiting factor[14]. Drainage class is typically assessed using the percentage of low chroma colours (indicative of waterlogging) and redoximorphic features (reduction of iron and manganese oxides) according to Milne et al. (1995). The soil reduction potential was therefore assigned a score of 1 to 5 (low to high) associated with the drainage class and carbon content of the soil. For mixed soil polygons, the SRP was proportionally weighted by the extent of the soil series within the polygon. The resulting soil reduction potential classification is presented in Table 6.12. The SRP for the Waituna Catchment is shown in Figure 6.20.

The geological reduction potential (GRP) is strongly influenced by the abundance of organic carbon within an aquifer (Rissmann, 2011; Rismann and Hodson, 2013). Aquifers hosted in materials containing a significant proportion of peat, lignite or other organic materials have an elevated organic carbon content and tend to be strongly reduced, whereas alluvial aquifers low in organic carbon are oxidising. The classification of GRP by Rissmann (2011) was modified for the Waituna Catchment by extending the 3 class classification to a 5 class classification on the basis of significant differences in the organic carbon content of geological substrates. The geological composition was scored 1 to 5 (low to high) based on organic carbon of the substrate. The resulting aquifer reduction potential classification is presented in Table 6.12. The GRP for the Waituna Catchment is shown in Figure 6.20.

To combine the soil and aquifer redox potential into a single classification the scores were averaged and ranked according to Table 6.12. However it is important to note, the combined reduction potential represents the reduction potential of deep drainage waters only (Figure 6.21). Water bypassing the soil matrix through artificial drainage and overland flow has less contact with the soil and none with the geological substrates which significantly lowers reduction potential, and as such is expected to show a lower cumulative reduction signature. Therefore, the reduction potential expressed as SRP over geological RP is a better descriptor for defining and communicating physiographic units.

[14] Soils in the Waituna catchment all soils have carbon contents greater than 5% in the upper profile (Topoclimate South, 2001).

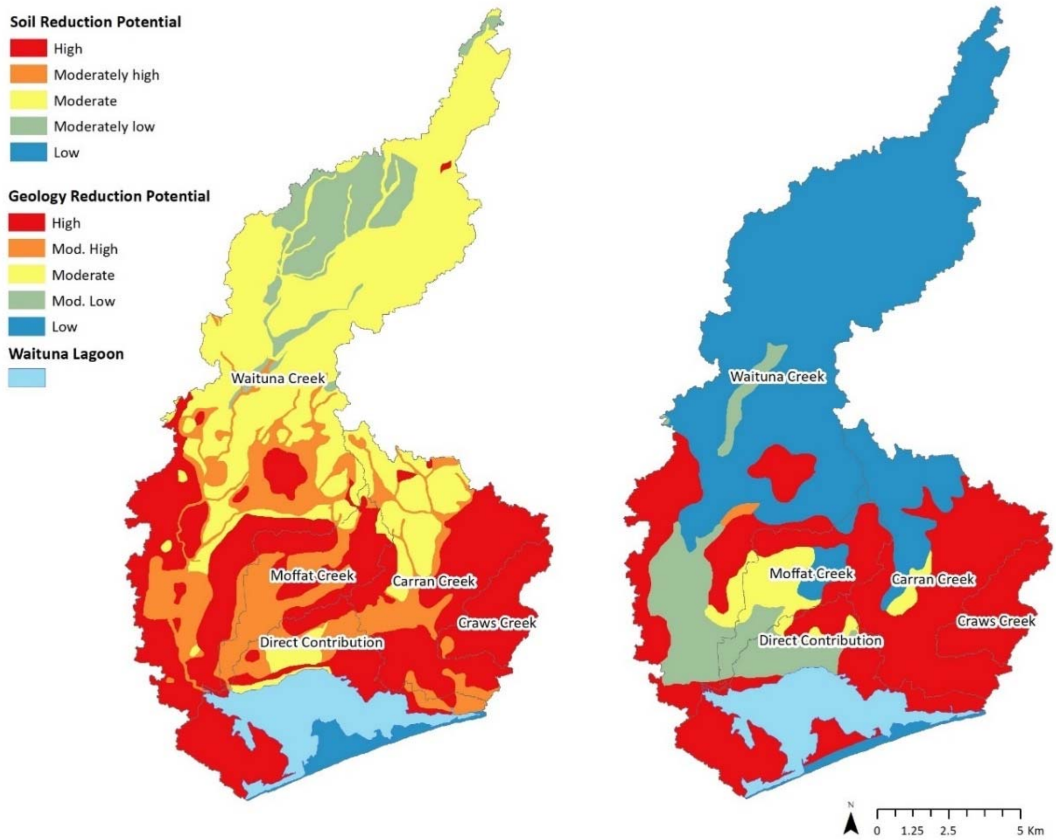


Figure 6.20: Soil (left) and aquifer (right) reduction potential.

Table 6.12: Soil and aquifer R-PAL classification.

	Soil drainage class	Soil carbon (%)	SRP class	Aquifer Composition	GRP class	Combined RP class
High	Very poorly drained	>20	4.6 - 5.0	Peat	5	4.3 -5.0
Mod. high	Poorly drained	10-20	3.6 - 4.5	Contains lignite	4	3.3- 4.25
Moderate	Imperfectly drained	4-10	2.6 - 3.5	Marine terraces	3	2.3- 3.25
Mod. low	Moderately well drained	2-4	1.6 - 2.5	Alluvial gravels with minor peat	2	1.8 - 2.25
Low	Well drained	<2	1.0 - 1.5	Alluvial gravels	1	1.0 - 1.75

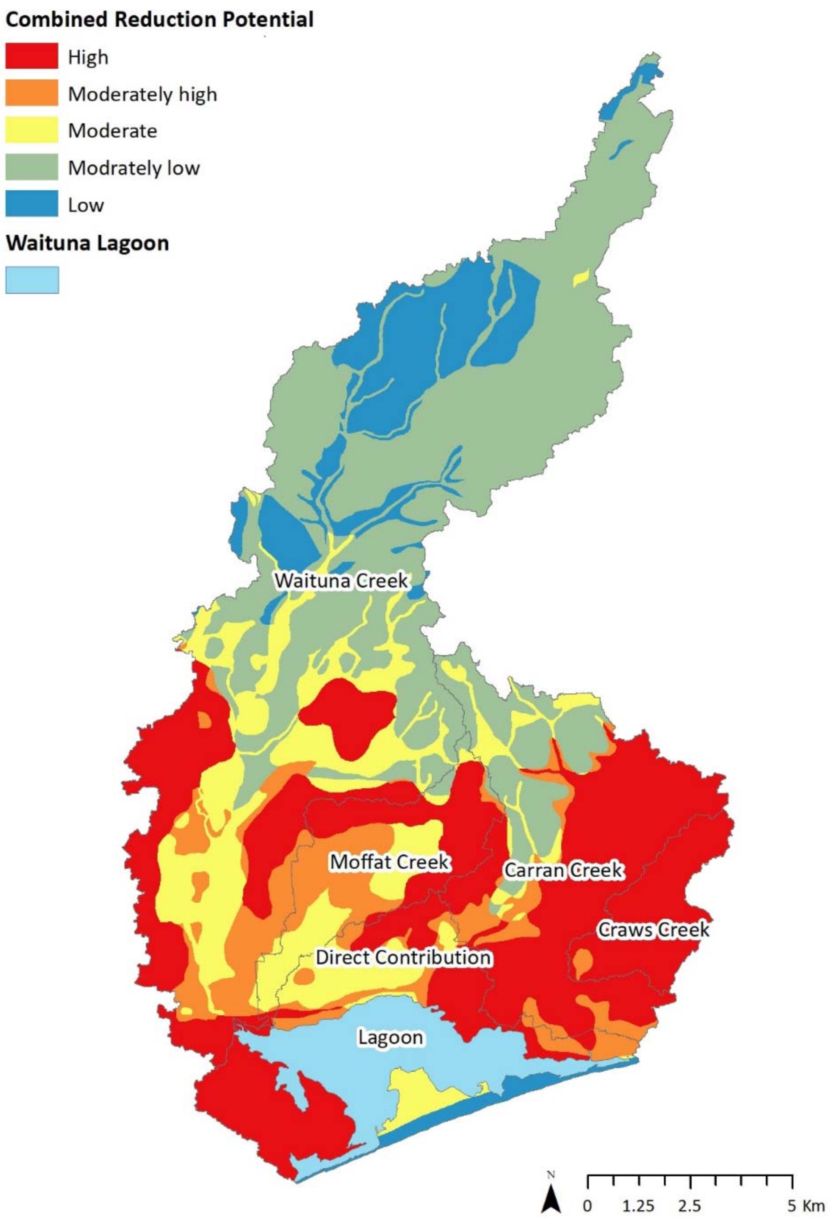


Figure 6.21: Redox Process-Attribute Layer (R-PAL) for Waituna Catchment associated with the combination of both soil and shallow aquifers.

6.6.5 R-PAL Catchment Summary

Soils and aquifers in the north of Waituna Catchment have a low to moderately low reduction potential, whilst in the south soils and aquifers exhibit a moderate to high reduction potential (Table 6.13).

Groundwater by in large will reflect the sum of both soil and aquifer reduction potential, except for where bypass occurs (through paleochannels). This is because soil zone drainage is superimposed upon aquifers. Soil reduction potential is inherited and may not be greatly modified if underlying aquifers are not reducing. However, surface water is strongly influenced by the hydrological pathway water has taken to the stream, resulting in a minimal reduction in surficial runoff even if soils have a high reduction potential.

Table 6.13: Soil, geological (aquifer) and combined reduction potential for the Waituna catchment.

	Waituna Creek		**Moffat Creek**		**Carran Creek**		**Craws Creek**		**Direct Contribution**		**Waituna Catchment**	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
Soil												
High	2,264	20	702	46	2,630	61	785	97	1,262	55	6,858	36
Mod. high	2,057	18	650	42	559	13			277	12	3,544	18
Moderate	6,230	56	190	12	1,095	26	26	3	404	18	7,919	41
Mod. low	607	5									607	3
Low									353	15	353	2
Geological												
High	1,713	15.4	590	38.2	3,143	73.3	810	100	1,568	68.3	7,013	36.4
Mod. high	46	0.4									46	0.2
Moderate	146	1.3	451	29.2	108	2.5			108	4.7	812	4.2
Mod. low	1,207	10.8	222	14.4					431	18.8	1,860	9.6
Low	8,040	72.1	282	18.3	1,035	24.1			190	8.3	9,547	49.5
Combined												
High	1,670	15	573	37	2,802	65	785	97	1,261	55	6,306	33
Mod. high	663	6	457	30	408	10	24	3	298	13	1,827	9
Moderate	1,032	9	213	14	126	3			462	20	1,833	10
Mod. low	7,241	65	300	19	948	22	1	0.1	84	4	8,573	44
Low	552	5							190	8	742	4
Total area	11,152		1,545		4,285		810		2,296		19,278	

6.7 Weathering Process-Attribute Layer (W-PAL)

The age (degree of weathering) and composition of landforms and associated soils play an important role in determining the composition of soils and aquifer materials, which in turn influence water composition and quality. Spatial variance in Ca, Mg, pH and alkalinity profiles of regional soil, soil water, surface water and soil-influenced groundwaters are represented by the weathering PAL.

Geomorphic age is also important, as older soils and aquifers are more weathered and have less capacity to neutralise acidity produced within the soil zone. The pH (acidic or basic) of water is a critical determinant of ecological function and species diversity. These are but two examples of the many different controls that substrates of different age and composition exert over water quality outcomes.

6.7.1 Relevance

Weathering reactions determine the pH and hence the alkalinity of natural waters. pH determines the speciation of ammoniacal nitrogen which is toxic to aquatic life, as well as the solubility and toxicity of metals. With regards to ammoniacal nitrogen, the concentration of ammonia increases as pH increases, this is particularly relevant to Waituna Lagoon where the higher pH of seawater increases the risk of elevated ammonia concentrations. Concentrations of pH, alkalinity and calcium (Ca) also exert important controls over the composition and range of floral and faunal communities, including macroinvertebrates (Petrin et al., 2007).

The resultant pH, alkalinity, and Ca concentrations of natural waters is primarily governed by the Acid Neutralisation Capacity (ANC) of soil and geological materials. Typically, the inherent ANC of the geological substrate is *assumed* to be dominated by the dissolution of carbonate minerals. For example, neutralisation of 2 x protons (H^+) by 1 mole of calcite is demonstrated in Equation 2:

$CaCO_3 + 2H^+ \text{-->} Ca^{2+} + CO_2 + H_2O$ (pH <6.4) (Equation 2)

However, the majority of New Zealand soils have pH values <6.4 and little evidence for free carbonate minerals (DSIR, 1968). Studies of the stable isotopes of Dissolved Inorganic Carbon ($\delta^{13}C$-DIC) in carbonate terrains also show that calcite is rapidly depleted in the upper soil zone with $\delta^{13}C$-DIC (V-PDB) signatures indicative of soil derived respiration of CO_2 and not calcite dissolution (Doctor et al. 2009; Rissmann et al., 2016b). Furthermore, silicate minerals typically have a greater overall ANC relative to calcite (per mole of mineral; Paktunc 1999; Weber et al., 2005). However, carbonate minerals, such as calcite, dissolve at rates that are 6 orders of magnitude faster than silicate minerals (Lasaga, 1984). Due to the rapid reaction rate, carbonate minerals in soil or rock of aquifer recharge areas are often rapidly exhausted relative to silicate minerals. As a result, the ANC of silicates dominates over the longer term (Paktunc 1999; Weber et al., 2005). The importance of silicate derived minerals over ANC is further highlighted by their ubiquity.

In silicate dominated environments, such the Waituna Catchment and Southland in general, it is proposed that the ANC the of rock or soil is determined by the abundance of the oxides of sodium (Na), potassium (K), magnesium (Mg), calcium (Ca) and manganese (Mn) or the so-called neutralising ions in primary parent materials (Stumm and Morgan 1996; Weber et al., 2005). The mechanism by which the oxides of Na, K, Mg, Ca and Mn neutralise acidity proceeds via cation–proton exchange reactions that release cations from the structure of silicate minerals; where the number of protons involved in these reactions is equal to the valence of the cation (Casey and Ludwig, 1996; Oelkers, 2001). Accordingly, the abundance of multioxide silicate minerals within a given geological material or soil appears to determine both the ANC and the abundance of Na, K, Mg and Ca ions released to exchange sites.

Silica (Si), aluminium (Al) and iron (Fe) do not reduce acidity during the weathering of silicate minerals (Dove and Crerar, 1990; Lasaga, 1995; Paktunc, 1999; Oelkers, 2001; Weber et al., 2005). Therefore, quartz dominated gravels cemented by the oxides of Fe, such as marine terrace deposits, will typically have a limited ANC. With regards to organic carbon, organic matter has a very low ANC due to low silicate mineral content and the prevalence of organic acids (e.g. fluvic and humic). Under the pH of most natural waters (pH > 4.5), the majority of organic acids will be deprotonated thereby contributing significantly to the acid balance of natural waters.

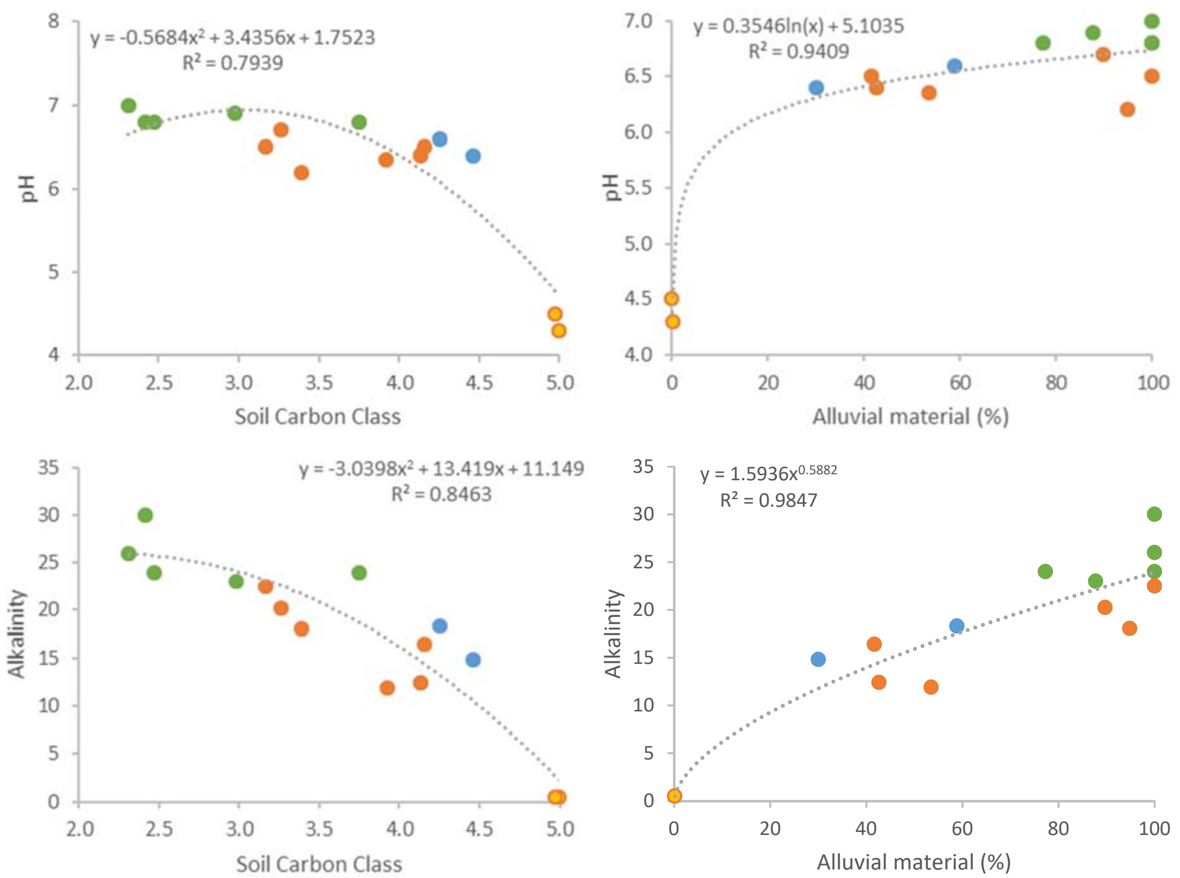
6.7.2 Weathering Process Attribute Layer (W-PAL) Hypotheses

The hypotheses for the process-attribute gradients governing weathering processes within the Waituna Catchment are as follows:

1. The ANC of soils and aquifer material will decrease across the felsic alluvium > quartz gravel > organic carbon continuum, specifically:
    - The pH, alkalinity, and Ca concentration of water will increase as the proportion of felsic alluvium increases within the capture zone of a monitoring point,
    - the pH, alkalinity, and Ca concentration of water will decrease as the proportion of organic carbon increases within the capture zone of a monitoring point.

6.7.3 Hypotheses Testing

As hypothesised the figures below show a strong relationship between soil and aquifer parent material and median pH, Ca^{2+} and alkalinity (Figure 6.22). This variation is consistent with decreasing Acid Neutralising Capacity (ANC) as across the felsic – organic carbon continuum. Notably, natural state catchments formed in peat wetland have the lowest pH and consequently alkalinity of all waters within the catchment. Equivalent areas of developed peat (Moffat Creek) have elevated pH probably reflecting a long history of liming.



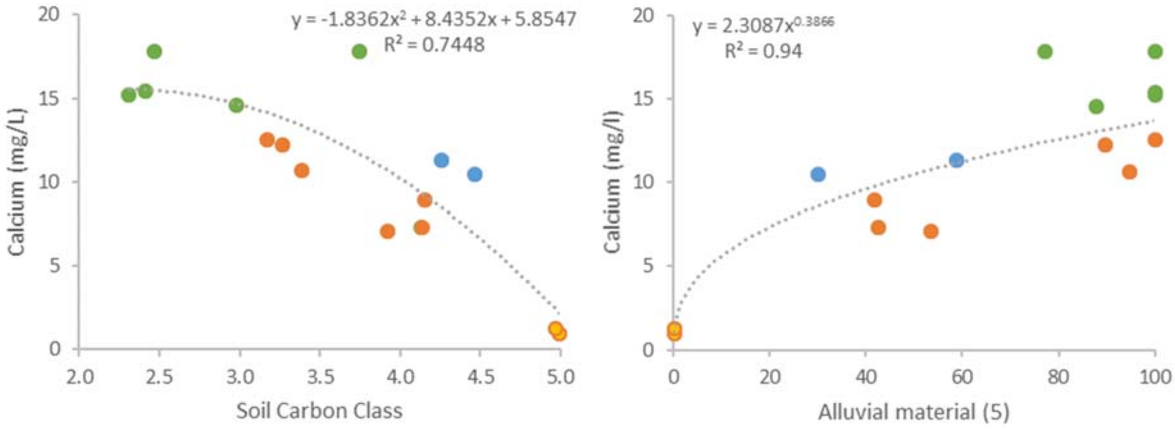


Figure 6.22: Median pH, alkalinity and dissolved calcium by soil carbon class (1. <2%, 2. 2-4%, 3. 4-10%, 4. 10-20%, 5. >20%) and aquifer alluvial material (%) proportionally weighted for the subcatchments Waituna Creek (green), Moffat Creek (blue), Carran Creek (orange) and Craws Creek (yellow).

6.7.4 Mapping Method

The W-PAL for the Waituna Catchment was developed by classifying the soil (Topoclimate South, 2001; NZ Land Resource Inventory, DSIR, 1968) and geological (QMAP; Turnbull and Allibone, 2003) layers by their acid neutralising capacity (ANC). As the ANC within an aquifer is strongly influenced by the parent material of the geological substrate, aquifers hosted in materials containing a significant proportion of peat, or other organic materials have very little mineral content and ANC, whereas alluvial aquifers have a much higher ANC increasing with the amount of carbonate in the substrate. A five class classification on the basis of significant differences in the carbonate content of geological substrates was developed for Waituna. The geological composition was scored 1 to 5 (low to high) based on the composition of the substrate. The resulting ANC classification is presented in Table 6.14. The G-ANC for the Waituna Catchment is shown in Figure 6.23.

The soil acid neutralising capacity is primarily related to the soil pH class. The soil ANC was therefore assigned a score of 1 to 5 (low to high) associated with the pH of the soil. Soil pH is classified as <4.5 very low, 4.5 - 5.2 low, 5.3 - 6.5 moderate, 6.6 – 7.5 high and > 7.6 very high (Table 6.14). For mixed soil polygons, the soil ANC was proportionally weighted by the extent of the soil series within the polygon. The resulting soil ANC classification is presented in Table 6.14. The soil ANC for the Waituna Catchment is shown in Figure 6.23. To combine the soil and aquifer ANC into a single classification the scores were averaged and ranked according to Table 6.14 and is shown in Figure 6.24.

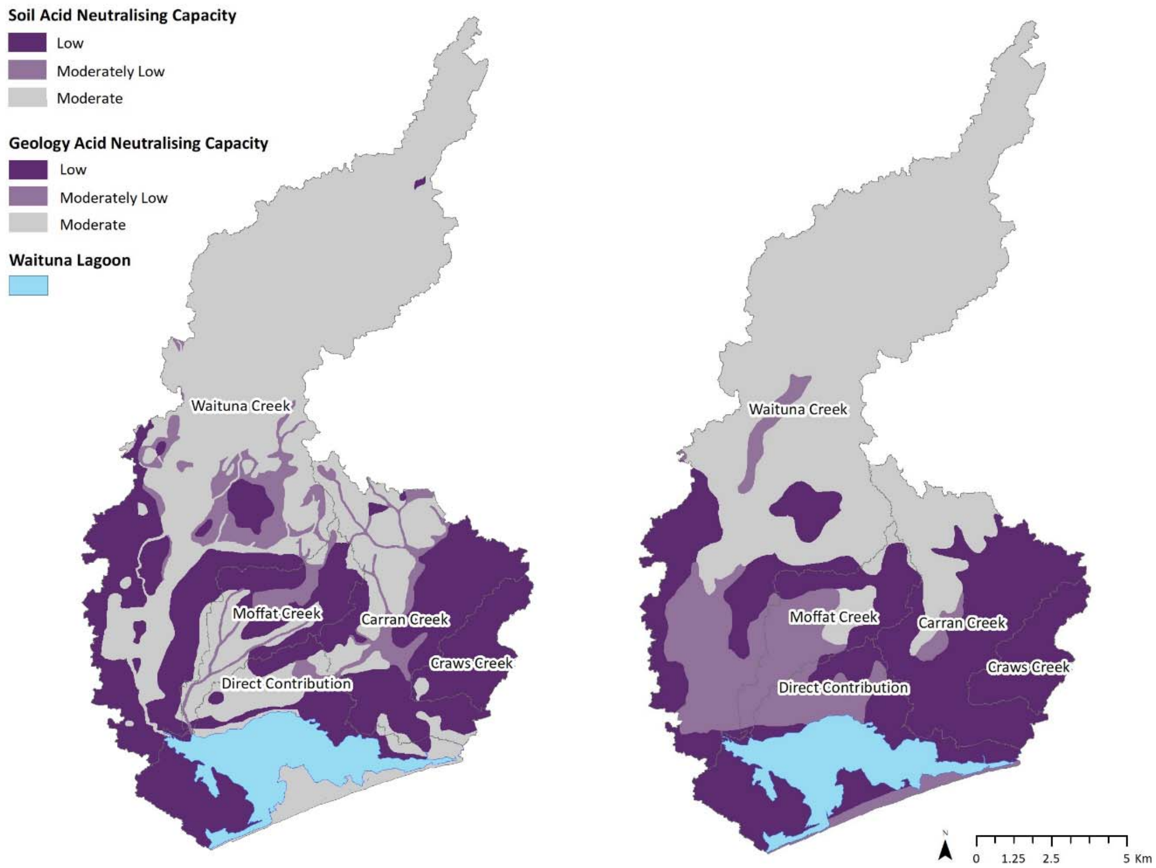


Figure 6.23: Soil (left) and aquifer (right) acid neutralising capacity.

Table 6.14: Soil and aquifer Acid Neutrilising Capacity (ANC) to produce a W-PAL for Waituna Catchment.

ANC	Soil pH	S-ANC class	Aquifer Composition	G-ANC class	Combined ANC class
High	Alkaline (>7.6)	> 4.5	Carbonate	5	> 4.5
Mod. High	Neutral to slightly alkaline (6.6 - 7.5)	3.5 - 4.4	Mafic	4	3.5 - 4.4
Moderate	Acidic (5.3 - 6.9)	2.5 - 3.4	Felsic / lignite	3	2.5 - 3.4
Mod. Low	Strongly acid (4.5 - 5.2)	1.5 - 2.4	Silica (Marine Terraces, Sand, Mixed felsic & peat)	2	1.5 - 2.4
Low	Extremely acid (< 4.5)	< 1.4	Peat	1	< 1.4

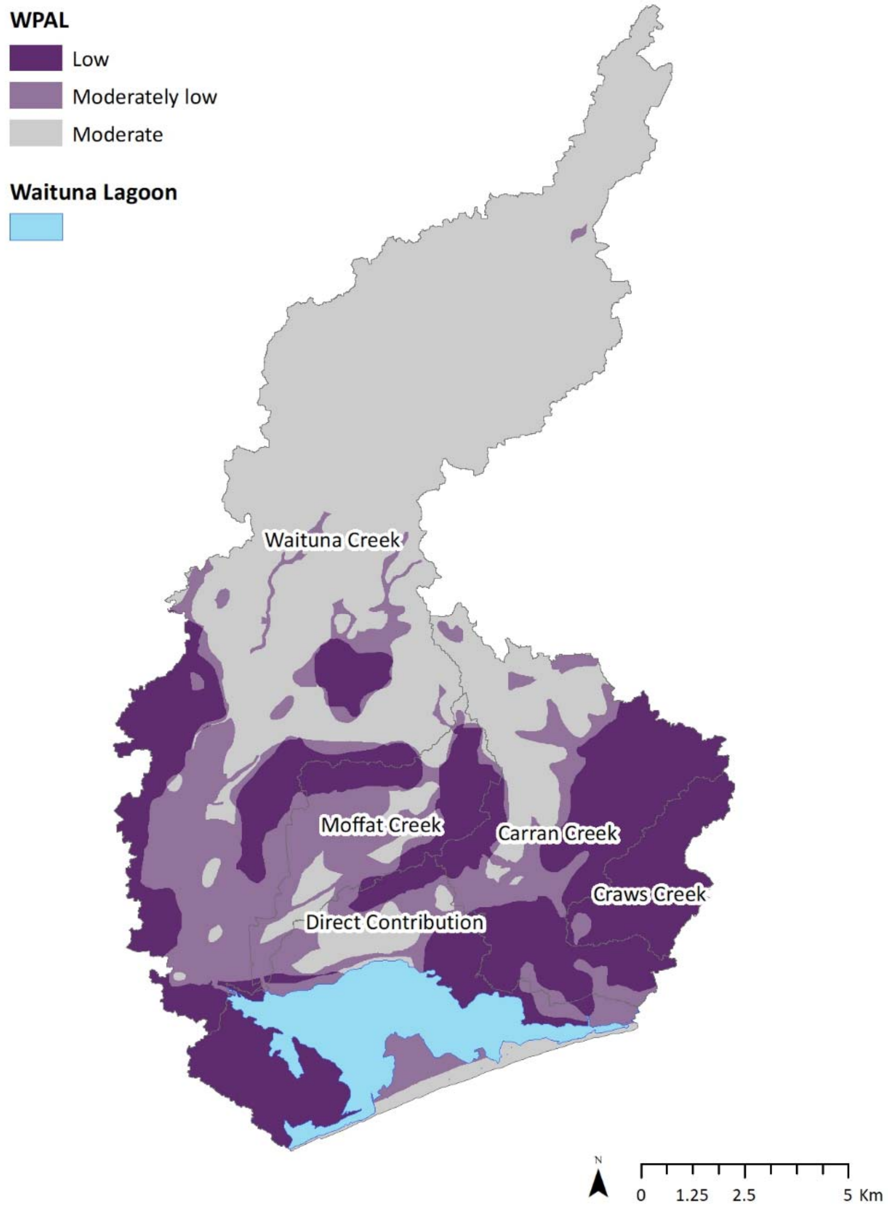


Figure 6.24: Weathering Process-Attribute Layer (W-PAL) for Waituna Catchment.

6.7.5 W-PAL Catchment Summary

Younger geomorphic surfaces dominate the southern catchments, with older geomorphic surfaces towards the north. This divide is a reflection of the movement of the paleoshoreline throughout geological time. This has resulted in soils and aquifers in the north of Waituna Catchment having a moderate acid neutralising capacity, whilst in the south soils and aquifers exhibit a moderately low to low acid neutralising capacity (Table 6.15). Therefore, the pH of drainage waters from the south are inherently lower, more acidic, than those from the north of the catchment.

Table 6.15: Soil, geological (aquifer) and combined acid neutralising capacity for the Waituna Catchment.

	Waituna Creek		Moffat Creek		Carran Creek		Craws Creek		Direct Contribution		Waituna Catchment	
	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)	(Ha)	(%)
Soil												
Moderate	8,181	73	553	36	1,247	29	25.5	3.1			9,981	52
Mod. low	713	6	288	19	407	9			81	4	1,489	8
Low	2,264	20	702	46	2,630	61	785	97	1,262	55	6,858	36
Geological												
Moderate	8,093	73	281	18	1,036	24	1.0	0.1			9,410	49
Mod. low	1,353	12	672	44	107	3			729	32	2,861	15
Low	1,712	15	589	38	3,141	73	809	100	1,567	68	7,009	36
Combined												
Moderate	7,940	71	316	20	986	23	1.0	0.1			9,242	48
Mod. low	1,681	15	724	47	708	17			644	28	3,757	19
Low	1,537	14	503	33	2,590	60	785	97	1,149	50	5,779	30
Total area	11,158		1,543		4,284		811		2,296		19,281	

7 Validation and Physiographic Model

7.1 Introduction

Previous sections have depicted graphically the general reliability and performance of process-attribute layers, on their own, to estimate spatial variation in steady-state water quality measures. Importantly, these sections revealed that the direction (magnitude) of the response of each individual process-attribute layer is consistent with the process level understanding used to map the Southland region (Rissmann et al., 2016; Snelder et al., 2016). However, for the Waituna Catchment, it appears that incorporation and mapping of deep drainage (DD), subsurface artificial drainage (SAD) and % rainfall excess as overland flow (OLF) into the H-PAL provides improved constraint over spatial variation in steady-state sediment and E. *coli* measures.

In the following section, we apply a Machine Intelligence (MI) approach to assess the underlying response and sensitivity of combinations and individual process-attribute layers for the estimation of spatial variation in the steady-state concentrations of key water quality measures (i.e., N, P, S, M*).* The key objectives of MI evaluation of the process-attribute layers developed for the Waituna Catchment are as follows:

- Independently test the validity of the process-level understanding that forms the basis of the hydrology and redox process-attribute layers developed for the Waituna Catchment.
- Produce numerical outputs in the form of simple equations that can be used to estimate the steady-state concentration of contaminants according to the physiographic assemblage within any given capture zone for a surface water site.

7.2 Methodology

We use the MI software Eureqa (Version 1.24.0, Schmidt and Lipson, 2015), which employs a symbolic regression approach based on evolutionary computation to search the space of mathematical expressions while minimising various error metrics (Schmidt and Lipson, 2009). Unlike traditional linear and nonlinear regression methods, symbolic regression searches both the parameters and the form of the equation simultaneously. The input data for the model is first tested for normality and if required the input data is log transformed prior to standardisation (subtracting by the mean and dividing by the standard deviation).

Model development proceeds by the user specifying a function to explain a dependent variable from one or more independent variables (Figure 7.1). For example, the expression TKN = *f*(SRP, OLF) instructs Eureqa to search for a function that explains the steady state, spatial variation in the concentration of Total Kjeldahl Nitrogen (TKN) as a function of the proportion of reducing soils (SRP) and the % rainfall occurring as overland flow with the capture zone of the long-term surface water monitoring sites within the Waituna Catchment.

Testing is based on a small number of monitoring sites (n = 5), the majority of which represent the southernmost location on the 4 main streams and Craws Creek tributary (i.e., Waituna Creek 1m upstream Waituna Road; Waituna Creek at Marshall Road; Moffat Creek at Moffat Road; Carran Creek at Waituna Lagoon Road, and; Carran Creek Trib at Waituna Lagoon Rd). Combined the sum of the area of capture zone equates to 88.1% of the total catchment area. We favour the use of these 5 sites as they are more likely to be representative of the actual flow variable population due to a large number of samples (n = 1,399) including a significant event flow sample set collected by Environment Southland.

Initial numeric expressions are formed by randomly combining mathematical building blocks such as algebraic operators, analytical functions, constant and state variables (Figure 7.1). New equations are formed by recombining previous equations and probabilistically varying their subexpressions (Figure 7.2). The algorithm retains equations that model the data better than others and abandons unpromising solutions (Schmidt and Lipton, 2009). Independent variables that offer little explanatory power, relative to others are also excluded during the evolutionary process.

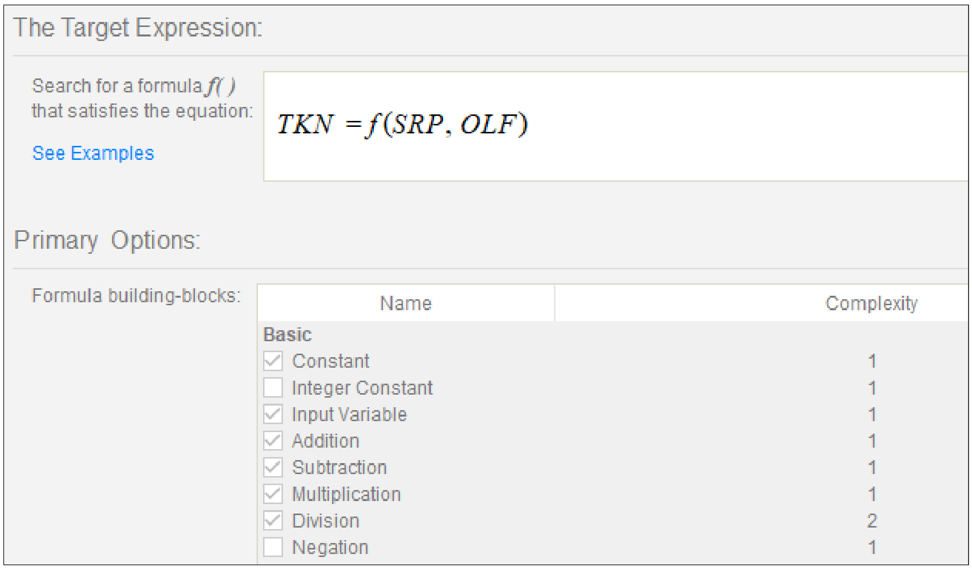


Figure 7.1: Target expression (i.e., TKN = f(SRP, OLF)) and formula building blocks (not all shown; (from Eureqa (v. 0.98 beta; Schmidt and Lipson (2014)).

Eventually, after a large number of iterations, the algorithm terminates returning a set of equations that can be used to estimate the dependent variable (Figure 7.2). Model-independent parameter estimation and uncertainty analysis can then be performed on the outputs using a range of tools.

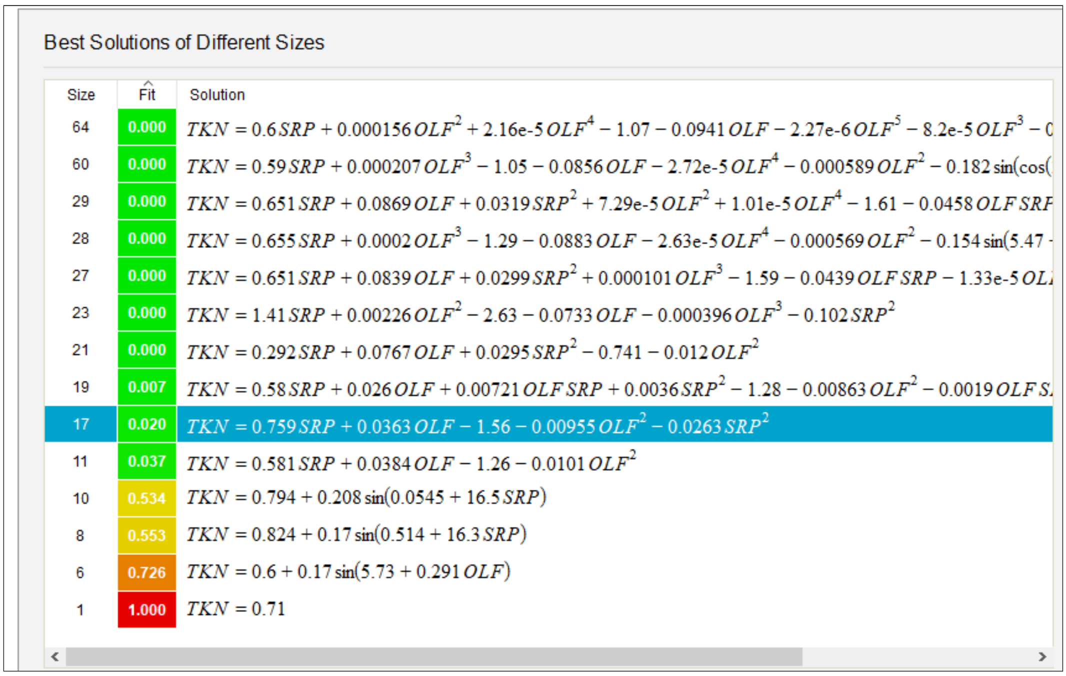
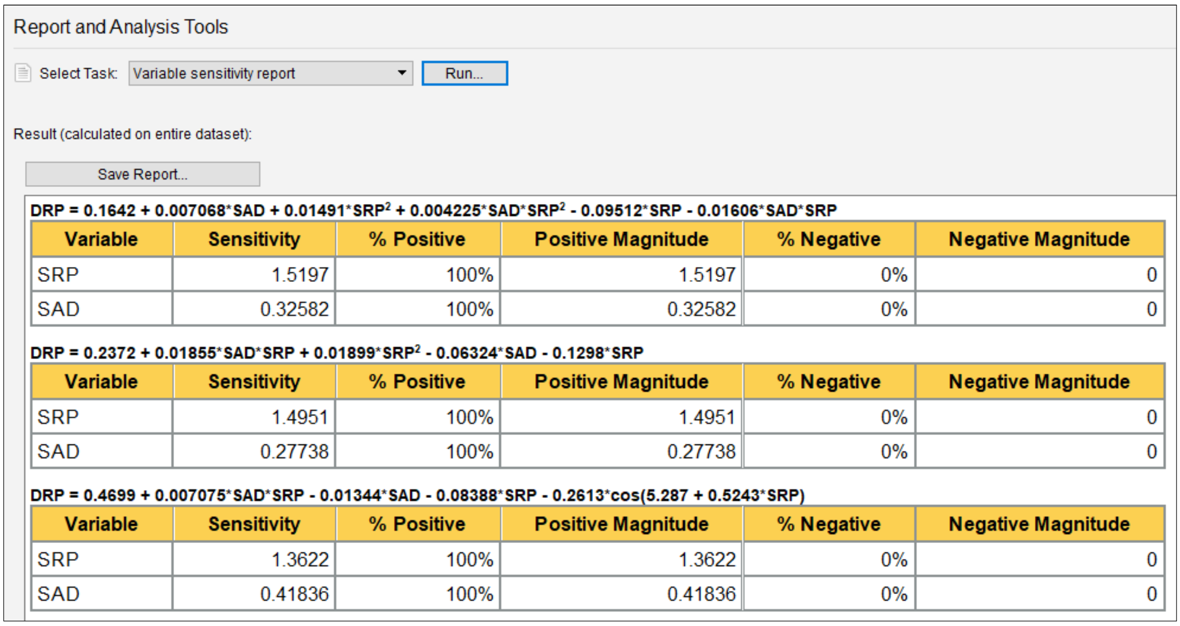


Figure 7.2: Best solutions are retained by the model and the fit (MAE) and level of complexity (size) returned (from Eureqa (v. 0.98 beta; Schmidt and Lipson (2014)).

When applied to the Waituna Catchment, the main value of the MI approach is the ability to specify and test the response and sensitivity of the process-attribute layers to estimate spatial variation in steady-state water quality measures. Specifically, hypotheses based on the process level understanding developed in earlier sections are formulated and used to test the validity of the underlying process-attribute. For example, we may hypothesise that the R-PAL and H-PAL are the most important process-attribute layers for estimating spatial variation in steady-state TN. We then go on to hypothesise the likely relative sensitivity of each respective PAL over the steady-state variation as well as the magnitude (positive or negative) of the response. For example, we may hypothesise that redox (R-PAL) is likely to be the most sensitive process-attribute layer governing steady-state variation in TN, but that OLF is also important but likely to be less sensitive. The validity of these hypotheses is then tested by running the model and evaluating if the model outputs for sensitivity and magnitude are consistent with our underlying process level understanding.

Within Eureqa, sensitivity is defined as the relative impact within a model output that an independent variable has on the dependent variable. The sensitivity score is important when two or more PAL are combined as it provides a measure of the relative influence of each PAL over the target variable (Figure 7.3). Magnitude provides a measure of the size and direction (+/- %) of the response of a model. A positive magnitude is when increases in the independent variable lead to increases in the dependent variable. A negative magnitude is the opposite, when an increase in an independent variable leads to a decrease in the dependent variable. For example, we would expect to see a 100% negative magnitude (decrease) in TN as the proportion of reducing soils and aquifers increases within the capture zone of a monitoring point.


Report and Analysis Tools

Select Task: Variable sensitivity report Run...

Result (calculated on entire dataset):

Save Report...

$DRP = 0.1642 + 0.007068*SAD + 0.01491*SRP^2 + 0.004225*SAD*SRP^2 - 0.09512*SRP - 0.01606*SAD*SRP$

Variable	Sensitivity	% Positive	Positive Magnitude	% Negative	Negative Magnitude
SRP	1.5197	100%	1.5197	0%	0
SAD	0.32582	100%	0.32582	0%	0

$DRP = 0.2372 + 0.01855*SAD*SRP + 0.01899*SRP^2 - 0.06324*SAD - 0.1298*SRP$

Variable	Sensitivity	% Positive	Positive Magnitude	% Negative	Negative Magnitude
SRP	1.4951	100%	1.4951	0%	0
SAD	0.27738	100%	0.27738	0%	0

$DRP = 0.4699 + 0.007075*SAD*SRP - 0.01344*SAD - 0.08388*SRP - 0.2613*\cos(5.287 + 0.5243*SRP)$

Variable	Sensitivity	% Positive	Positive Magnitude	% Negative	Negative Magnitude
SRP	1.3622	100%	1.3622	0%	0
SAD	0.41836	100%	0.41836	0%	0

Figure 7.3: Variable sensitivity and magnitude report for the 3 least complex models for explaining spatial variation in steady-state Dissolved Reactive Phosphorus (DRP) between monitoring sites within the Waituna Catchment. Where SRP = soil reduction potential; SAD = subsurface artificial drainage density. In this example, SRP is the most sensitive landscape attribute followed by SAD. For both landscape variables the steady-state concentration of DRP increases in a positive direction (100%) as the proportion of reducing soils and subsurface artificial drainage increases within the capture zone of a monitoring site. Importantly, the model response is consistent with the process-level knowledge. Removing SAD and running the model with only SRP produces a much more complex model for an equivalent accuracy.

Another key test of the performance of any given PAL to estimate spatial variation in water quality measures is the resultant complexity of the model produced (Figure 7.2 and 7.4). Specifically, if a PAL is a poor estimator of spatial variation, or is not the best estimator, then the complexity of the model will be large. Model complexity is evaluated in terms of a Pareto Chart that ranks solutions by accuracy and complexity (Figure 7.4). For the Waituna Catchment work, we consider the most suitable model for any given water quality measure to be of the lowest complexity that still achieves an R^2 of at least 0.95 accuracy. Complexity is also used to evaluate the importance/relevance of other possible landscape controls over the spatial variation of steady-state water quality measures. For example, does the inclusion of the density of Subsurface Artificial Drainage (SAD) along with soil reduction potential (SRP) redox and OLF reduce or increase the complexity of a model when attempting to estimate TN? Alternatively, the model may be rerun with only one landscape attribute to see if the model complexity increases or decreases.

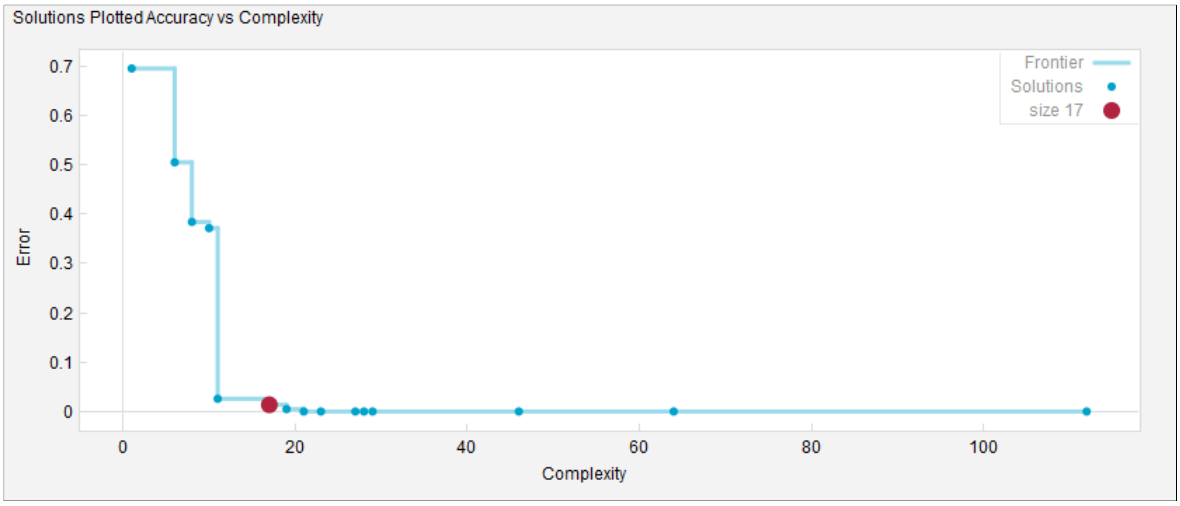


Figure 7.4: Pareto chart of model accuracy versus complexity. The key goal is to select the least complex model that achieves the desired accuracy. As complexity moves beyond a size of 15 the gain in accuracy is small relative to the increase in complexity. A key objective of minimising complexity is to avoid the risk of overfitting the model.

In terms of validation, Eurequa automatically splits the data, using a disjunctive approach e.g. where the training and validation sets are kept separate: e.g. 95% of data are used as training data set and the last 5% for validation, with a random shuffle before splitting the data (Dubčáková, 2011). The equations that evaluate accuracy, complexity, sensitivity and magnitude are available in Schmidt and Lipton (2009, 2014) and online from www.nutonian.com. All models were run on a Desktop PC with a 3.2 GHz Intel Core (TM), i7-6000 processor and 16 GB of RAM.

The flexibility of being able to define a function based on process level understanding is an attractive feature of this approach, avoiding the potential for spatial correlation when there is no process level basis for any causal relationship. For example, in the regional scale 'Physiographics of Southland', there was a strong correlation between precipitation source (i.e., Alpine, Hill, Inland, Inland-Coastal and Coastal) and NO_3^- (as nitrogen) concentration in shallow aquifers and streams despite NO_3-N concentrations in precipitation across all precipitation categories are negligible (often below detection; Rodway et al., 2016). Here the correlation between NO_3-N and precipitation is coincidental and relates to both a land use intensity gradient that is coincident with precipitation source (e.g. lowest intensity in Alpine area and Alpine precipitation category; highest land use intensity in lowland areas and coastal precipitation category and a generalised redox gradient), with more reducing conditions associated with soils and aquifers coincident with the Coastal precipitation category. An ability to exclude such erroneous correlations is appealing when seeking to best explain the underlying controls over spatial variability to water quality outcomes.

7.3 Process Testing and Evaluation of Individual Water Quality Measures

Before selection of process-attribute layers for estimating water quality data we applied a correlation matrix (Pearson-Spearman-Kendall) with a 2-tailed probability to the process-attribute layer scores for each monitoring site and associated capture zone and each of the following water quality measures: TN, NNN, TKN, TP, DRP, TSS, Clarity, Turbidity and *E. coli* (See Appendix 1: https://drive.google.com/drive/folders/1YEjnDrjNkAKIzv112KhFX8p1odW8A66e?usp=sharing). The strongest correlation between the three (i.e., Pearson-Spearman-Kendall) was used. Correlation strength in conjunction with process knowledge was then used to identify and select the most relevant process-attribute layers for each water quality analyte.

7.3.1 Nitrogen

Total Nitrogen (TN)

Total Nitrogen (TN) is a measure of all the nitrogen forms found in water, specifically: organic nitrogen (both dissolved and particulate), ammoniacal nitrogen and both NO_3 and NO_2. The fraction of organic and ammoniacal nitrogen tends to increase in response to poor soil drainage and high organic carbon contents in aquifer systems (Rissmann et al., 2016). Most commonly, NO_3 (an oxidised form of nitrogen) constitutes the largest component of TN in areas of intensive land use, especially where soils are reasonably well drained and aquifers oxidising (Jones et al., 2004; Tratnyek et al., 2012; Rissmann, 2011; Rissmann et al., 2012; Beyer et al., 2016; Environment Southland Data). However, the magnitude of NO_3 (and NO_2) concentrations typically decrease as soil and aquifer reduction potential increases (Mengel, 1996; Jones et al., 2004; Beyer and Rissmann, 2016; Rissmann et al., 2016; Snelder et al., 2016). Given that NO_3 is often the largest component of the Total Nitrogen pool, reduction of NO_3 to gaseous forms significantly lowers the TN concentration of waters.

In terms of the pathway, NO_3 is highly mobile and if not removed via reduction, will migrate rapidly through porous media (Daniels et al., 2012; Cameron et al., 2014; Klefoth et al., 2014; Owens et al., 2016). By comparison, ammoniacal and high molecular weight organic nitrogen species are commonly concentrated at or within the near surface (Rissmann et al., 2012). Within the Waituna Catchment, ammoniacal and organic nitrogen concentrations increase as surficial flow paths are engaged, attesting to the importance of OLF over the export of ammoniacal and organic fractions of nitrogen to surface waters throughout the catchment.

Correlation analysis reveals strong correlations between TN and the redox indicators DO (+0.94), Fe^{2+} (-0.86) and DOC (-0.92) as is consistent with a strong redox control. Total nitrogen and NNN are also strongly positively correlated (+0.97), again indicating the significant contribution NNN makes to TN.

Hypothesis: On the basis of process level knowledge and correlation analysis we hypothesise that TN concentration will decrease as the proportion of reducing soils and aquifers increases (100% negative magnitude) within the capture zone of a monitoring site. Although correlated with % OLF, soil and aquifer reduction potential is the main process governing spatial variation in steady-state TN concentrations. Accordingly, the inclusion of % OLF will not significantly improve the model and will likely increase the underlying complexity.

Most Important PALs: R-PAL
Function:
TN = f(R-PAL)
Results:
Model: $TN = 3.824 + 0.0125*RPAL^2 - 0.6872*RPAL$
Coefficients: 2
Complexity: 9
$R^2 = \geq 0.95$
Magnitude: 100% negative response

Conclusion: The model output is of low complexity, obtains an R^2 of 0.95 and supports our process level hypotheses as to the role of soil and aquifer reduction potential over spatial variation in steady-state TN concentrations. Specifically, as the proportion of reducing soils and aquifers increases within the capture zone of a surface water site the steady-state concentration of TN decreases – 100% negative magnitude. The inclusion of OLF in the model greatly increase the complexity of the model (complexity of 25 and 5 coefficients). Estimation of TN using other variables (e.g., Geological Reduction Potential (GRP) on its own produces a significantly more complex model. Interestingly, clarity and TN are strongly positively correlated (≥0.95) and appears a useful indicator of like TN concentrations in Waituna Streams.

Nitrate-Nitrite-Nitrogen (NNN)

Nitrate and NO_2 nitrogen (NNN) production are governed by the mineralisation of organic and inorganic forms of nitrogen within the soil and unsaturated zone. The accumulation of NNN is greatest in oxidising soils with concentrations typically declining as soil zone reduction potential (SRP) increases (Beyer et al., 2016; Beyer and Rissmann, 2016). Correlation analysis supports this process-level understanding with strong relationships between NNN and the redox indicators DOC (-0.95), Fe^{2+} (-0.91) and DO (+0.85).

Hypothesis: Due to the susceptibility of NNN to reduction, the steady state concentration of NNN will decline (100% negative magnitude) as the proportion of reducing soils and aquifers (R-PAL) increase within the capture zone of a monitoring site. Accordingly, the magnitude of response will be strongly negative. Correlation analysis indicates a strongly negative relationship between NNN and %OLF. This we hypothesise reflects a strong spatial correlation between Soil Reduction Potential (SRP) and %OLF. However, we note that OLF is not the process controlling NNN removal.

Most Important PALs: R-PAL
Function:
NNN = f(R-PAL)
Results:
Model: $NNN = 4.15/(0.9409*RPAL - 0.3409) - 0.9431$
Coefficients: 3
Complexity: 8
$R^2 = \geq 0.95$
Magnitude: negative, 100%

Conclusion: The model outputs are of low complexity, obtains an R^2 of 0.95 and supports our process level hypotheses as to the magnitude of influence. Specifically, steady-state NNN concentrations decline as the area of reducing soils and aquifers increases within the capture zone of a monitoring point. Including SAD (a component of H-PAL) in the function does not improve the model. All other PALs or combinations result in additional complexity for an equivalent accuracy (R^2), indicating that the R-PAL is the best estimator of spatial variation in steady-state NNN concentrations.

Total Kjeldahl Nitrogen (TKN)

Although TN declines as the proportion of soil and aquifer reduction potential increases, the organic and ammoniacal forms of nitrogen typically increase (Rissmann et al., 2016; Environment Southland data). This increase reflects the production of ammoniacal and organic nitrogen in anaerobic soils (poorly drained) and aquifers via several mechanisms: organic nitrogen mineralisation (ONM); dissimilatory nitrate reduction to ammonium (DNRA), and; the release of adsorbed ammonium during microbial reduction of iron oxides (MRFeO) (Mengel, 1996; Tratnyek et al., 2012; Richard et al., 2014; Chacón et al., 2017).

Animal wastes also produce significant quantities of ammoniacal nitrogen (Rissmann et al., 2012). However, ammonium is strongly fixed by structured clay (2:1) minerals and large organic molecules are excluded by physical and electrostatic mechanisms resulting in the concentration of both species at or close to the soil surface and within the topsoil (Mengel, 1996; Wilhelm, 2009; Rissmann et al., 2012; Richard et al., 2014; Chacon et al., 2017). Consequently, shallow lateral soil zone flow (mediated by mole-pipe drainage) and overland flow are often important pathways for organic and ammoniacal nitrogen delivery to streams. Correlation analysis supports this process-level understanding with a strongly positive (+0.87) relationship between TKN and both Subsurface Artificial Drainage (SAD) and the % of OLF weighted by developed land (OLF_DL). Both SAD and OLF_DL are strongly positively correlated (+0.9) indicating that these attributes can be used as alternates. Importantly, TKN exhibits positive correlation with other species that are transported by OLF including TP (+0.85), Turbidity (+0.7), TSS (+0.65) and *E. coli* (+0.7).

Hypothesis: Steady-state concentrations of TKN will increase as the %OLF_DL and the proportion of reducing soils and aquifers increase within a given capture zone or sub-catchment. Due to concentration in the near-surface, we hypothesise that %OLF_DL will be the most sensitive estimator of TKN, followed by the proportion of reducing soils and aquifers. Due to the importance of soil and aquifer reduction potential (R-PAL) over TKN, its inclusion will result in the production of a less complex model than if % OLF_DL is used on its own. For both %OLF_DL and R-PAL the magnitude of response will be strongly negative. In short, steady state TKN will increase as the % OLF_DL and the proportion of reducing soils and aquifers increases within the capture zone of a monitoring site.

Most Important PALs: OLF_DL (SAD) component of H-PAL; R-PAL.
Function:
TKN = f(RPAL, OLF_DL)
Results:
Model: $TKN = 1.086*OLF\_DL + 0.2536*RPAL + 0.01199*OLF\_DL^3 + 0.01021*RPAL*OLF\_DL^2 - 1.219 - 0.1018*RPAL*OLF\_DL - 0.1986*OLF\_DL^2$
Coefficients: 3
Complexity: 17
R^2 = ≥0.95
Magnitude: both 100% positive
Sensitivity: OLF = 1.4; SRP = 0.35

Conclusion: The model output is more complex that than for TN, but this is expected given the particle reactive nature of TKN. The model obtains an R^2 of 0.95 and supports our process level hypotheses as to the role of OLF and R-PAL over TKN export to streams. Specifically, OLF is the most sensitive attribute in areas of developed land followed by soil and aquifer reduction potential in governing the spatial variation in steady-state TKN concentrations. However, running the model with only OLF_DL or only R-PAL results in far greater model complexity for an equivalent accuracy (complexity of 29 and 5 coefficients for OLF; >50 and >8 coefficients for R-PAL).

7.3.2 Phosphorus

Phosphorus mobility is known to vary according to the anion exchange capacity of soils and aquifers. In most instances, anion exchange capacity is controlled by the abundance of the sesquioxides of aluminium (Al) and iron (Fe) (Richardson, 1985; McDowell and Monaghan, 2015). In mineral soils and aquifers, amorphous clay minerals play an important role in the retention of P and its removal from solution. In these settings, P mobility is typically low and the main pathway of P transport is associated with overland flow. Within the Waituna Catchment, TP peaks as OLF pathways are engaged (Hodson, R, unpublished data, Environment Southland).

However, as the proportion of soil and aquifer organic carbon increases P-retention typically declines and transport via lateral and deep vertical pathways becomes increasingly important (McDowell and Monaghan, 2015). The increased mobility of P reflects a decrease in the abundance and stability of the sesquioxides of Fe and Al that govern P-retention. The abundance and stability of Fe and Al sesquioxides are controlled by reductive dissolution (redox) and acidity, respectively (Richardson, 1985; Van Hees et al., 2001). McDowell and Monaghan (2015) report extreme P-leaching rates for peat soils within the Waituna Catchment. Rissmann et al. (2012) noted a 50-fold increase in median TP between the alluvial aquifers in the north of the Waituna Catchment and peat bog aquifers in the south and noted that in strongly reducing (beyond Fe^{III}-reduction) peat aquifers that the formation of small (1 – 30 nm), highly-mobile inorganic and organic-P colloids is likely an important mechanism of P supply via baseflow (Ryan and Gschwend, 1990; Rissmann et al., 2012; Rissmann and Lovett, 2016).

Total Phosphorus (TP)

Phosphorus may be found in several forms in freshwaters, including dissolved form (orthophosphate), inorganic form (reactive plus condensed or acid hydrolysable phosphate) and organically bound forms. Total Phosphorus is the sum of reactive, condensed and organic phosphorous and often includes a significant particulate phosphorus (PP) fraction in intensified landscapes. Correlation analysis reveals a strong relationship between TP and %OLF of developed land (%OLF_DL) but only moderately correlated (+0.6) with the % OLF that has not been weighted by the proportion of natural state areas, suggesting a source limitation control. TP is also strongly correlated with clarity (-0.83) and Fe^{II} (+0.94), indicating redox control.

Hypothesis: Steady-state concentrations of TP will increase as the %OLF_DL and the proportion of reducing soils and aquifers increase within a given capture zone or sub-catchment. Due to the concentration at or near the surface of the land we hypothesise that %OLF_DL will be the most sensitive estimator of PP, followed by the proportion of reducing soils and aquifers (R-PAL). Due to the importance of soil and aquifer reduction potential (R-PAL) over TP, its inclusion will result in the production of a less complex model than if % OLF_DL is used on its own. For both %OLF_DL and R-PAL, the magnitude of response will be strongly negative, except for capture zones with a significant natural state component for which PP is source limited.

Most Important PALs: %OLF of developed land component of H-PAL.
Function:
TP = f(OLF_DL, RPAL)
Results:
Model: $TP = 0.01455 + 0.041*RPAL + 0.0145*OLF\_DL^2 + 0.00178*RPAL*OLF\_DL^2 - 0.027*OLF\_DL - 0.0171*RPAL*OLF\_DL - 0.001193*OLF\_DL^3$

Coefficients: 3
Complexity: 13
$R^2 = \geq 0.95$
Magnitude: 75% positive and 25% negative (natural state)

Conclusion: The model output is of moderate complexity reflecting a strongly positively skewed distribution for TP and the tendency for natural state areas to be source limited. However, the model obtains an R^2 of 0.95 and supports our process level hypotheses as to the magnitude of influence. Specifically, median TP concentrations increase as the % OLF_DL and the proportion of reducing soils and aquifers increase within the capture zone of a monitoring site.

Dissolved Reactive Phosphorus (DRP)

Phosphorus that passes through a 0.45 μM filter is considered to be in the dissolved form. However, both organic (inositol hexaphosphate type) and inorganic forms of DRP are strongly sorbed (retained) in the presence of the sesquioxides of Al and Fe. Accordingly, DRP is often concentrated in the upper layer of the soil profile. However, as soils become more poorly drained organic carbon and the duration and magnitude of soil saturation all increase. The latter result in lower pH and more reducing soil profile forms, both of which lower the stability and abundance of the sesquioxides of Al and Fe, both of which are critical to P-retention. As such, P-retention tends to decrease across the soil drainage class continuum. Within the Waituna Catchment, very low concentrations of Fe and Al sesquioxides are correlated with former peat bogs and organic soils. In these areas, P mobility is often greatly enhanced (see McDowell and Monaghan, 2015). However, over time peat soils may develop increased P-retention in response to liming and drainage - both of which may enhance to varying degrees the inherent stability and abundance of the sesquioxides of Fe and Al. As DRP is often concentrated at or near the soil surface, especially in areas of developed land, the % of effective rainfall occurring as OLF is an important hydrological process transporting DRP to stream.

Correlation analysis reveals strong correlations between DRP and the redox indicators DO (-0.72), NNN (+0.97), Fe^{2+} (+0.82) and DOC (+0.97) as is consistent with a strong redox control. In terms of redox control, DRP is strongly positively correlated with soil and aquifer reduction potential (+0.97). However, we note that DRP is not correlated with TSS and Turbidity. Furthermore, although DRP is strongly correlated with % OLF (+0.86) it is only weakly correlated with %OLF_DL (+0.37) suggesting DRP is not source limited in areas of natural state peat. Previous work has noted that DRP is the dominant form of P associated with the natural state peat wetland areas of the catchment whereas Particulate Phosphorus (PP) dominates across the larger developed area (Rissmann et al., 2012, 2013). DRP in strongly reducing, peat wetland systems is often highly mobile and due to negligible loading rates is not concentrated at or near the surface of the soil. For this reason, DRP will behave differently in natural state wetlands compared to developed wetlands where loading rates are higher and organic soils have been modified.

Hypothesis: Steady-state concentrations of DRP in Waituna surface waters will increase as the abundance of reducing soils and aquifers and the % OLF increases for sites with more than 70% of a capture zone associated with developed land. The redox status of soils and aquifers across an area of developed land will be the most sensitive estimator of spatial variation in steady-state DRP concentrations followed by the %OLF_DL. A model including only the R-PAL will be significantly more complex than when both the R-PAL and %OLF_DL are combined, indicating the importance of both processes over the export of DRP to stream.

Most Important PALs: R-PAL (combined soil and geological reduction potential) and %OLF (H-PAL)
Function:
DRP = f(R-PAL, OLF_DL)
Results:
Model: DRP = 0.0168*OLF_DL + 0.01009*RPAL + 0.0006154*OLF_DL^3 - 0.02731 - 0.00557*OLF_DL^2
Accuracy: R^2 = ≥0.95
Sensitivity: R-PAL is the most sensitive attribute (0.89), followed by OLF_DL (0.51)
Coefficients: 3

Complexity: 13
Magnitude: both R-PAL and % OLF are of a positive magnitude (100%) for catchments with >70% of a capture zone associated with developed land.

Conclusion: The model output is of low complexity, obtains an R^2 of 0.95 and supports our process level hypotheses as to the magnitude of influence. Specifically, DRP concentration increases as the proportion of reducing soils and aquifers and % OLF increases within the capture zone of a monitoring site with >70% of a capture zone associated with developed land. In terms of sensitivity, model outputs indicate that the R-PAL is a more sensitive attribute than OLF for the estimation of spatial variation in steady-state DRP across areas of developed land. However, running the model with only the R-PAL as an attribute results in a significantly more complex equation for an equivalent accuracy (i.e., complexity score of 43 relative to 10), reflecting the important role of OLF as a pathway for DRP export to streams in areas of developed land. The inclusion of sites with a large proportion of natural state (>84%; Craws Creek Trib) results in a small negative magnitude (25%). This response is not unexpected and likely reflects the fact that natural state areas are not associated with additional anthropogenic loading.

7.3.3 Suspended Sediment

In addition to land use, we hypothesise that spatial variation in sediment concentration in surface waters is controlled primarily by hydrological process-attribute gradients and climatic forcing[15]. Here we define sediment as the suspended solid phase within a stream that is >2 microns in diameter; that is a heterogeneous mix of organic and inorganic constituents which may include organic carbon, clays (both poorly ordered and structured), silt and sand, bacteria, viruses and both organic and inorganic ions and molecules - including N and P species.

We consider Total Suspended Sediment (TSS) as the best measure of suspended sediment. Absorbance, clarity (black disk) and turbidity (NTU) are measures of the optical properties of water that may be influenced by both dissolved and solid phase constituents. For example, clarity is generally lower in waters with a high dissolved organic carbon concentration and turbidity is influenced by the presence of 'dissolved' colloids that are smaller than the nominal 2 microns used to define TSS and SSC (Davies-Collies and Smith, 2001). Accordingly, the relationship between turbidity, clarity, TSS and SSC is not always simple.

Research from across Southland has noted that the majority of sediment is delivered during OLF events although lateral flow paths mediated by mole-pipe drainage can be important vectors of sediment export (Magesan et al., 1995; Cameron et al., 2014; Monaghan et al., 2002, 2016;). Other studies have noted a significant stream bank component associated with stream bank erosion, including a novel study of sediment source completed within the Waituna Catchment (McDowell et al., 2016). As with other catchments, analysis of Waituna surface water monitoring sites indicate that TSS increases with streamflow, reaching maximum values as surficial pathways are engaged - highlighting the important role of surficial runoff over TSS concentration. Aquifers are not considered an important source of sediment >2 μm, to stream.

Total Suspended Sediment (TSS)

Hypotheses: Based on process level knowledge and correlation analysis we hypothesise that % OLF associated with developed land will be the most sensitive estimator of spatial variation in steady-state TSS concentrations. Specifically, capture zones with a greater proportion of soils susceptible to

[15] We do not consider the temporal climatic aspect in this section.

OLF will be associated with more frequent and larger magnitude exports of sediment to stream. Although natural state areas may have high % OLF (peat bogs) they are considered source limited and as such TSS will show a minor negative magnitude.

Most Important PALs: H-PAL (OLF_DL)
Function:
TSS = f(OLF_DL)
Results:
Model: TSS = 5.561 + 0.4221*OLF_DL^2 - 0.4558*OLF_DL - 0.04664*OLF_DL^3
Accuracy: $R^2 \geq 0.95$
Coefficients: 3
Complexity: 13
Magnitude: TSS is 100%.

Conclusion: The model output is of low complexity, obtains an R^2 of 0.95 and supports our hypotheses as to the magnitude of influence. If sites associated with natural state are included in the model run the complexity increases and the directional magnitude exhibits a minor negative percentage. This response is not unexpected and likely reflects the fact that natural state areas are relatively source limited and have not been hydrologically modified. No other process-attribute layer, or a combination thereof, produces a model of equivalent or lower complexity than that provided by OLF_DL on its own.

Turbidity

Turbidity is not correlated with steady-state clarity but strongly correlated with *E. coli* (+0.97), TSS (+0.97), VSS (+0.97). Turbidity is moderately correlated with TP (+0.6) and is not correlated with DRP. In terms of landscape controls, turbidity is strongly correlated with %OLD_DL (+0.9) but is not correlated with %OL, indicating a strong source control.

Hypotheses: Based on process level knowledge and correlation analysis we hypothesise that % OLF associated with developed land (OLF_DL) will be the most sensitive estimator of spatial variation in steady-state turbidity concentrations. Further, steady-state turbidity will increase as the % OLF_DL increases within the capture zone of a monitoring site. Although natural state areas may have high % OLF (peat bogs) they are considered source limited and as such turbidity will show a minor negative magnitude. No other landscape attribute or combination of landscape attributes will achieve equivalent simplicity and accuracy as OLF_DL.

Most Important PALs: H-PAL (OLF)
Function:
Turbidity = f(OLF_DL)
Results:
Model: Turb = 8.696*OLF_DL + 0.5072*OLF_DL^3 - 7.241 - 0.02805*OLF_DL^4 - 3.251*OLF_DL^2
Accuracy: $R^2 = \geq 0.95$
Coefficients: 4
Complexity: 22
Magnitude: Turbidity is 75% positive and 25% negative (natural state).

Conclusion: The model output is of moderate complexity reflecting a strongly positively skewed distribution for turbidity and a tendency for natural state areas to be source limited. The response of the model supports our process level hypotheses as to the magnitude of influence. Specifically, median TSS concentrations increase as the % OLF_DL within the capture zone of a monitoring site. This interpretation is consistent with the surficial mobilisation of sediment from disturbed soils in areas of developed land and a strong correlation between Turbidity and *E. coli*, TSS and VSS. If sites associated with natural state are included in the model run, complexity increases and the directional magnitude exhibits a minor negative percentage. This response is not unexpected and likely reflects the fact that natural state areas are relatively source limited and have not been hydrologically modified.

Clarity

Hypotheses: Clarity is strongly correlated (-0.9) with Dissolved Organic Carbon (DOC) but is not correlated with TSS or turbidity. Accordingly, we hypothesise that the dominant control over spatial variation in steady-state clarity is the abundance of organic carbon associated with soils and aquifers. As the accumulation of organic carbon and its subsequent dissolution to produce DOC is mediated by redox processes we propose that clarity will decline as the proportion of reducing soils and aquifers increases within the capture zone of a monitoring point. No other landscape attribute, or combination of attributes, will achieve the same degree of simplicity for an equivalent accuracy as the R-PAL.

Most Important PALs: R-PAL
Function:
Clarity = f(R-PAL)
Results:
Solution: Clarity = 2.533 + 0.1092*RPAL2 - 0.9721*RPAL
Accuracy: R^2 = ≥0.95
Coefficients: 3
Complexity: 11
Magnitude: R-PAL is 100% negative.

Conclusion: The model output is of low complexity, obtains an R^2 of 0.95 and supports our hypotheses as to the magnitude of influence. Specifically, steady-state clarity decreases as the proportion of reducing soils and aquifers increases within the capture zone of a monitoring point. This is consistent with the role of redox controls over organic carbon accumulation and subsequent dissolution to produce DOC (fluvic and humic acids). Unlike TSS and turbidity, organic carbon is not source limited in natural state areas.

7.3.4 Microbes

E. coli is a microbial indicator of microbial contamination of water. The *E. coli* concentration in this study is expressed as Colony-Forming Units (CFU). This is a unit of measurement used to determine the number of viable bacteria which grew on the agar medium per 100 ml of water tested. A key feature of time series *E. coli* data is a tendency be strongly positively skewed. The microbial loss to waterways is dependent on land use, climate and landscape attributes that govern the hydrological response.

In addition to land use, we hypothesise that spatial variation in mean *E. coli* concentration in surface waters is controlled primarily by soil hydrological properties and climatic forcing.[16] Principal Component Analysis (PCA) on standardised data for the main surface water monitoring sites indicates that the intensity of rainfall in a 24 and 48-hour period controls a significant portion, 59%, of the variation in *E. coli* abundance (Table 7.1). The second largest source of variation in *E. coli*

[16] We do not consider the temporal climatic aspect in this section.

abundance, 27%, is associated with seasonality. Specifically, higher *E. coli* abundance during the cooler wetter months of the year when soils are wetter and flow (Q) is higher[17].

Table 7.1: Principal Component Analysis (2012 – 2017), 8 surface water sites (including all SOE) for the period 2012 - 2017 (Environment Southland data). Natural state sites excluded.

Component	Variance	Proportion	Cumulative proportion
1	2.455	0.491	0.491
2	1.201	0.240	0.731
3	0.692	0.138	0.870

Coefficients	1	2	3
Season	0.024	-0.826	-0.400
Flow (Q)	0.362	0.412	-0.823
24 Hr Rainfall Intensity	0.566	-0.118	0.323
48 Hr Rainfall Intensity	0.558	0.171	0.240
E-Coli <CFU>	0.486	-0.325	-0.018

A strong response to rainfall intensity and seasonality is a feature of catchment soil hydrological properties over *E. coli* transmission. Specifically, the majority of the soils within the catchment are either slowly permeable (<4 mm/hr) or have a slowly permeable horizon within 1 m of the soil surface. For this reason, slope and soil hydrological properties are considered the most important landscape attributes governing microbial transmission for the majority of the catchment.

Microbial indicator (E. coli)

Steady-state *E. coli* concentrations are strongly positively (+0.9) correlated with the % of OLF occurring within the capture zone of a monitoring site associated with developed land (%OLF_DL). Steady-state *E. coli* is not correlated with OLF if weighting associated with natural state area is removed. This suggests a strong source limitation associated with natural state areas.

Hypotheses: Based on process level knowledge and correlation analysis we hypothesise that steady-state *E. coli* concentrations will increase as the proportion of %OLF_DL as effective rainfall increases within the capture zone of monitoring sites associated with developed land. Specifically, capture zones with a greater proportion of soils susceptible to OLF will be associated with more frequent and larger magnitude exports of animals wastes to stream. No other landscape attribute, or combination of attributes, will achieve the same degree of simplicity for an equivalent accuracy OLF on its own.

Most Important PALs: H-PAL (OLF)
Function:
Mean *E. coli* = f(OLF_DL)
Results:
Solution: *E. coli* = 1182 + 734.4*OLF_DL^2 - 1583*OLF_DL - 71.61*OLF_DL^3
Accuracy: R^2 = ≥0.95
Coefficients: 3
Complexity: 17
Magnitude: *E. coli* is 100% positive.

[17] On the basis of land use mapping we anticipate that variation in land use intensity, with regard to E. coli, to be relatively stable over the 6-year data record (2011 – 2016) used for this assessment.

Conclusion: The model output is of low complexity, obtains an R^2 of 0.95 and supports our hypotheses as to the magnitude of influence. Specifically, steady-state *E. coli* increases as the % rainfall occurring as OLF increases within the capture zone of a monitoring site. If sites associated with natural state are included in the model run, complexity increases (from 19 to 41) and the directional magnitude exhibits a minor negative percentage. This response is not unexpected and likely reflects the fact that natural state areas are source limited.

7.4 Uncertainty Analysis

Models are always simplifications of reality and hence, 'imperfect' (Loucks and Van Beek, 2005; Klein et al., 2016). For the MI approach taken above, we apply uncertainty analysis in an attempt to describe the entire set of possible outcomes. The predictive uncertainty of each model developed above is presented below along with measures of uncertainty and performance (i.e., Mean Square Error (MSE), Root Mean Square Error (RMSE), Nash–Sutcliffe model efficiency coefficient (NSE), correlation coefficient (R) and the coefficient of determination (R^2) (Table 7.2).

Table 7.2: Predictive uncertainty of models for each analyte with measures of uncertainty and performance (i.e., Mean Square Error (MSE), Root Mean Square Error (RMSE), Nash–Sutcliffe model efficiency coefficient (NSE), correlation coefficient (R) and the coefficient of determination (R^2). Units in ppm unless denoted otherwise.

Median Total Nitrogen (TN)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	2.5	2.0	2.9
Waituna Creek at Marshall Road	5	2.2	1.8	2.5
Moffat Creek at Moffat Road	8	1.4	1.1	1.7
Carran Creek at Waituna Lagoon Road	15	1.4	1.1	1.7
Carran Creek Trib at Waituna Lagoon Rd	17	0.7	0.2	1.2

Model	**TN = 3.824 + 0.0125*RPAL^2 - 0.6872*RPAL**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
0.10	0.14	0.96	0.99	0.97

Median Nitrate-Nitrite-Nitrogen (NNN)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	1.75	1.56	1.94
Waituna Creek at Marshall Road	5	1.12	0.97	1.27
Moffat Creek at Moffat Road	8	0.34	0.22	0.46
Carran Creek at Waituna Lagoon Road	15	0.34	0.22	0.46
Carran Creek Trib at Waituna Lagoon Rd	17	0.01	0.00	0.22

Model	**NNN = 4.15/(0.9409*RPAL - 0.3409) - 0.9431**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
0.02	0.06	0.99	0.99	0.99

Median Total Kjeldahl Nitrogen (TKN)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	0.4	0.38	0.52
Waituna Creek at Marshall Road	5	0.7	0.67	0.76
Moffat Creek at Moffat Road	8	0.9	0.87	0.99
Carran Creek at Waituna Lagoon Road	15	0.8	0.72	0.81
Carran Creek Trib at Waituna Lagoon Rd	17	0.65	0.57	0.73

Model	**TKN = 1.086*OLF_DL + 0.2536*RPAL + 0.01199*OLF_DL^3 + 0.01021*RPAL*OLF_DL^2 - 1.219 - 0.1018*RPAL*OLF_DL - 0.1986*OLF_DL^2**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
0.01	0.04	0.95	0.98	0.95

Median Total Phosphorus (TP)

		fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	0.04	0.04	0.04
Waituna Creek at Marshall Road	5	0.05	0.05	0.05
Moffat Creek at Moffat Road	8	0.16	0.16	0.16
Carran Creek at Waituna Lagoon Road	15	0.12	0.12	0.12
Carran Creek Trib at Waituna Lagoon Rd	17	0.09	0.09	0.09

Model	**TP = 0.06369*RPAL + 0.004508*OLF_DL^2 - 0.03053 - 0.02606*OLF_DL - 0.00661*RPAL^2**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
3.34E-07	0.0003	0.99	0.99	0.99

Median Dissolved Reactive Phosphorus (DRP)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	0.01	-1.6E-02	0.03
Waituna Creek at Marshall Road	5	0.02	-5.0E-05	0.03
Moffat Creek at Moffat Road	8	0.06	4.8E-02	0.09
Carran Creek at Waituna Lagoon Road	15	0.05	2.7E-02	0.06
Carran Creek Trib at Waituna Lagoon Rd	17	0.07	9.3E-03	0.07

Model	**DRP = 0.0168*OLF_DL + 0.01009*RPAL + 0.0006154*OLF_DL^3 - 0.02731 - 0.00557*OLF_DL^2**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
0.0008	0.01	0.72	0.88	0.78

Median Total Suspended Sediment (TSS)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	6.0	5.80	6.18
Waituna Creek at Marshall Road	5	7.1	6.94	7.21
Moffat Creek at Moffat Road	8	7.0	6.81	7.18
Carran Creek at Waituna Lagoon Road	15	8.0	7.86	8.14

Model	**TSS = 5.561 + 0.4221*OLF_DL^2 - 0.4558*OLF_DL - 0.04664*OLF_DL^3**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
0.006	0.038	0.99	0.99	0.99

Median Volatile Suspended Sediment (VSS)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	1.5	1.37	1.62
Waituna Creek at Marshall Road	5	1.6	1.50	1.68
Moffat Creek at Moffat Road	8	3.0	2.83	3.08
Carran Creek at Waituna Lagoon Road	15	4.0	3.88	4.07

Model	**TSS = 5.561 + 0.4221*OLF_DL^2 - 0.4558*OLF_DL - 0.04664*OLF_DL^3**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
0.003	0.025	0.99	0.99	0.99

Mean *E. coli* (CFU)

	Site #	Fitted	Lower	Upper
Waituna Creek 1m upstream Waituna Road	2	402	0	829
Waituna Creek at Marshall Road	5	1496	1171	1822
Moffat Creek at Moffat Road	8	1450	901	2000
Carran Creek at Waituna Lagoon Road	15	2716	2299	3134
Carran Creek Trib at Waituna Lagoon Rd	17	247	0	716

Model	***E. coli* = 1182 + 734.4*OLF_DL^2 - 1583*OLF_DL - 71.61*OLF_DL^3**			
MSE	**RMSE**	**NSE**	**r**	**r^2**
145936.00	170.84	0.96	0.98	0.97

7.5 Summary and Limitations

The purpose of this section of work was to:

- Independently test the validity of the process-level understanding that forms the basis of the hydrology and redox process-attribute layers developed for the Waituna Catchment.
- Produce numerical outputs in the form of simple equations that can be used to estimate the steady-state concentration of contaminants according to the physiographic assemblage within any given capture zone for a surface water site.

The above modelling exercise allowed us to further test the validity of the underlying process-level understanding that was used to map the Waituna Catchment. The magnitude of response and the sensitivity of process-attribute layers were consistent with the hypothesised process level controls over water quality. In conjunction with sensitivity and magnitude of response, the relative complexity of the model (as an equation) was seen as a useful tool for evaluating the importance of one or more process-attribute layer over spatial variation in steady-state water quality outcomes.

Numerical equations defining the relationship between process-attribute layers and spatial variation in steady-state water quality measures provide a platform for estimating values for stream reaches with little data or could be used to produce estimates of steady state values for each sub-catchment or the entire catchment. However, the majority of the sites used to calibrate the model are dominated by intensive land use, with the exception of the Carran Creek subcatchment. Accordingly, estimation of the steady-state concentration of water quality contaminants from natural state sites is likely to be significantly overestimated. Accordingly, the model for the Waituna Lagoon Catchment is most suited to the estimation of water quality outputs for areas that are intensively farmed. Incorporation of land parcel scale land use intensity scores is also likely to improve the predictive accuracy of the model for unmonitored portions of the catchment. Ranking of land use intensity scores can then be incorporated into the model as a land use pressure layer.

In terms of scale, the sites used to calibrate the model are all associated with 3rd order or higher streams. Accordingly, it is not possible to asses how the model performs for 1st and 2nd order streams, although at low stream orders the relative area contributing drainage and the relative scale of the geospatial layers (e.g. soil) representing spatial variation in landscape attributes are likely to be important.

Finally, despite the small number of long-term monitoring sites relative to the the number of attribute-process classes we are confident in the performance of the layers, at the process level, to estimate spatial variation in steady-state water quality and propose that the sites used for numerical validation and modelling are the best basis for development of numerical outputs for the Waituna Lagoon Catchment, specifically:

- The 5-capture zones account for 88.1% of the catchment area
- The number of samples (n=1,399) and the effort to measure event flows provides a reasonable representation of the actual population
- Conjunctive validation was used in model development, and;
- Uncertainty measures indicate sound performance.

Ideally, this type of modelling approach would be applied at a regional scale so that a greater number of sites can be used to test the model and produce numerical outputs.

8 Physiographic Units for Water Quality

8.1 Physiographic Map

To obtain a thorough understanding of water composition within the catchment, all four Process-Attribute Layers can be combined to produce landscape classes. However, for the purposes of understanding water quality variation, only the Redox and Hydrological PALs are necessary. Therefore, units which describe the fundamental landscape controls over water quality for the Waituna Catchment are produced by combining Levels 1 and 2 in Figure 8.1. The resulting map is shown in Figure 8.2. For paddock scale resolution, fine-scale surficial flow direction and water routing (Level 3) can be used in conjunction with catchment scale units.

Within each redox family, siblings are used to showing the gradients that exist across the landscape (Figure 8.2). These gradients are also often related to temporal controls and hydrological pathways and allow for specific water quality management strategies to be developed specifically for a family or sibling.

The Physiographic Map can also be overlaid by the finer resolution flow path information, including convergent zones and tile drains for on-farm management.

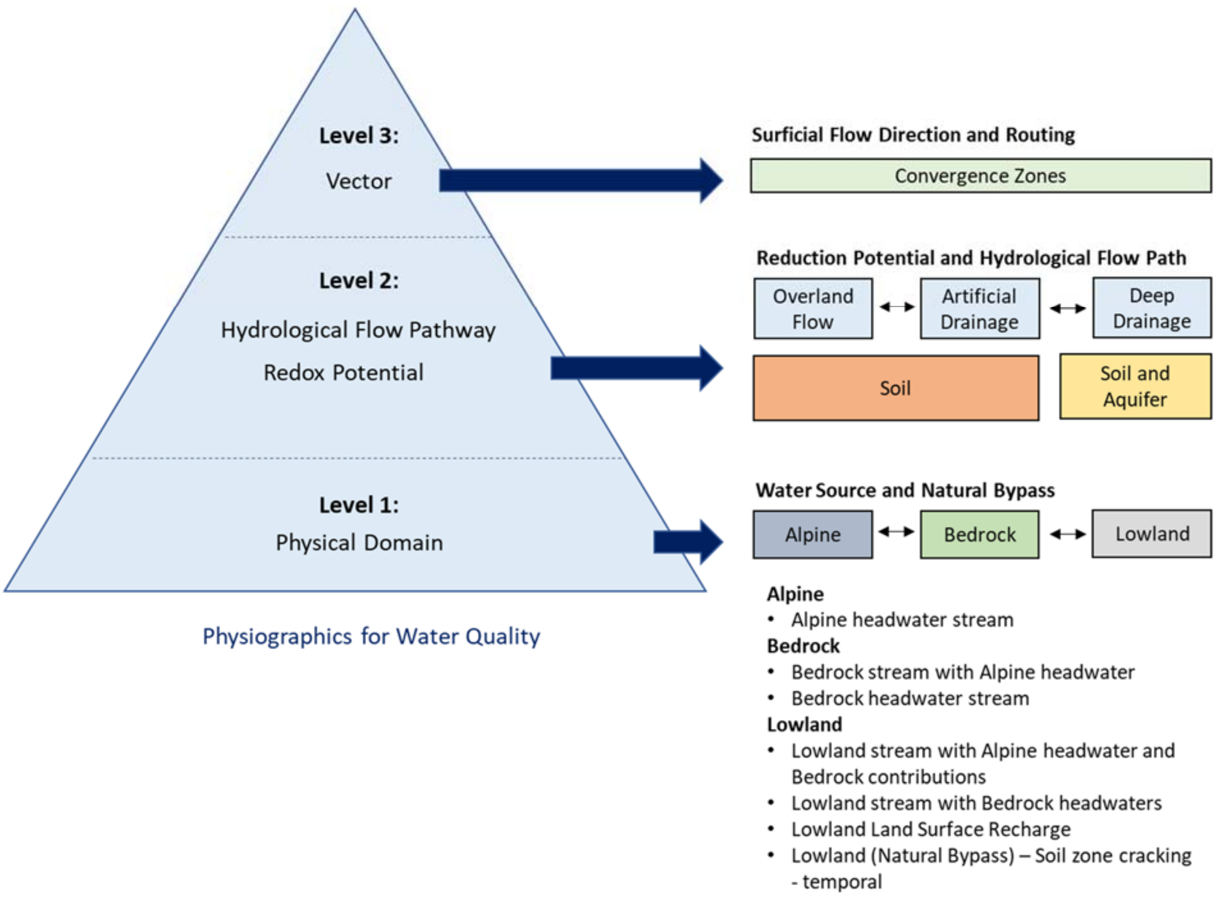


Figure 8.1: Physiographics for water quality.

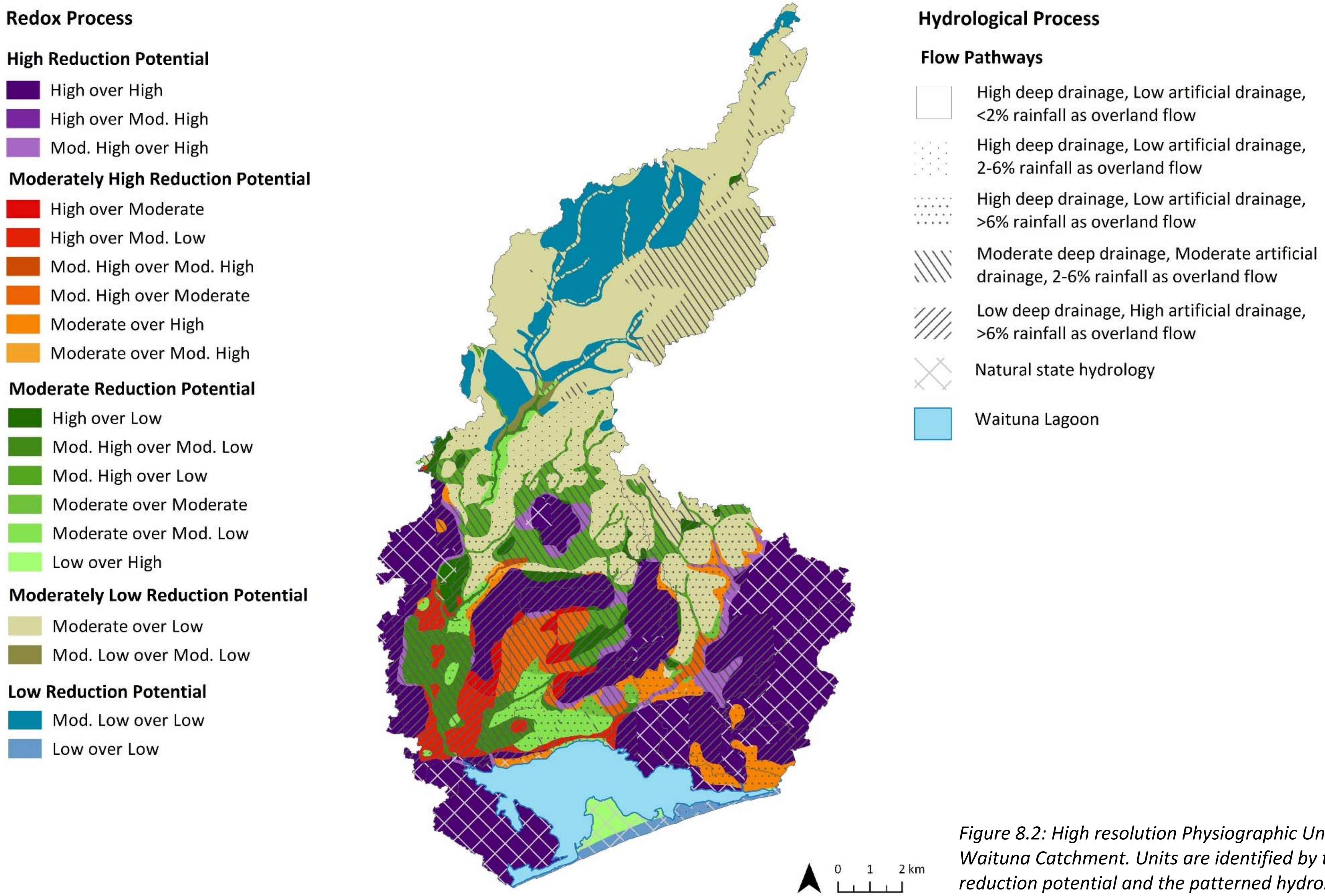


Figure 8.2: High resolution Physiographic Units for Waituna Catchment. Units are identified by the coloured reduction potential and the patterned hydrological flowpath.

9. Summary

This work explores the relationship between landscape attributes and the key processes governing spatial variation in water quality across the Waituna Catchment. From this exploration, maps of process-attribute gradients have been developed for the catchment that can be used to estimate spatial variation in water quality outcomes. This report is largely a technical document defining the physiographic method and testing its relevance and performance. A subsequent report on the application of the work provides more accessible information for Living Water and the Whakamana Te Waituna partners.

The key outputs of this report include:

- Exploration of the relationship between landscape attributes and processes.
- Development of a strong understanding of the processes controlling variation in water quality outcomes across the catchment.
- Production of four key process-attribute layers (A-PAL, H-PAL, R-PAL and W-PAL).
- Testing of the validity and performance of the redox (R-PAL) and hydrological (H-PAL) layers to estimate the main water quality contaminants across the catchment.
- Incorporation of a simple land use pressure layer to further refine estimations of spatial variation in water quality.
- Generation of numerical models for estimating each contaminant for receiving environments without water quality data.
- Combination of the R-PAL and H-PAL to produce a Physiographic Map of the inherent landscape controls over water quality.

The information contained in this report has been summarised in a web-based application, ESRI Story Maps. The figures contained in this report have been provided over a base map of Southland, with main roads and land parcel boundaries to allow the user to easily locate and interrogate areas of interest. Maps have an interactive component allowing the user to view maps at farm or catchment scale.

Access to the Story Map is through the following URL: https://e3s.maps.arcgis.com/apps/MapJournal/index.html?appid=0c0fc1fa5afa423eb63d85bd9a1ec980

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