The potential of treated municipal wastewater irrigation to cause aggregate instability and pore sealing on Banks Peninsula soils.

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Abstract

The discharge of Treated Municipal Wastewater (TMW) into surface waters can degrade water quality and represents a waste of potentially valuable irrigation water and plant nutrients. While the application of TMW to soil can enhance plant growth, TMW containing high sodium (Na) concentrations can degrade soil structure resulting in decreased permeability and increased runoff. TMW from Banks Peninsula, New Zealand is currently discharged into Akaroa Harbour, however, a legal injunction requires that discharge into water be discontinued and therefore land application is being investigated. This thesis aimed to determine whether TMW application to Banks Peninsula soils would result in significant degradation to soil structure. The TMW contained 40 mg/L Na. Soil columns (0.1m x 0.19m) containing the Pawson Silt Loam and intact lysimeters (0.5m x 0.7m) containing both Pawson Silt Loam and Barry's soil (a silt loam) were irrigated with a total volume of TMW of up to 1500 mm. The TMW irrigated onto the soil columns was spiked with Na up to 325 mg/L. Infiltration occurred unimpeded on all the soils, indicating that irrigating TMW would not degrade soil structure in the short term. Irrigation with TMW resulted in a significant increase in Na in the soil profile (from 330 mg/kg to 1760 mg/kg), however, there was sufficient native Ca and Mg in these soils (6540 mg/kg and 4130 mg/kg) to offset this increased Na. It is likely that in the long term, lime or gypsum will need to be added to maintain soil structure.

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Introduction

Discharge of Treated Municipal Wastewater (TMW) into waterways can have serious environmental and human health risks (Holeton et al., 2011). Alternatively, TMW can be irrigated onto land rather than discharged into waterways. TMW has economic value as an irrigation resource, due to both the water content and the nutrients in the TMW.

Treated municipal wastewater

Domestic sewage and wastewater is created from a variety of sources, including homes, businesses and industries (EPA, 2016). In developed countries, sewage treatment plants (STPs) remove contaminants from the municipal wastewater. In particular, STPs separate the solid and liquid fractions, with many potential contaminants, such as heavy metals, remaining in the sewage sludge (Duan et al., 2010; EPA, 2016).

Municipal wastewater has elevated levels of sodium (Na), nitrogen (N), phosphorus (P), potassium (K), sulfur (S), and dissolved organic carbon (DOC), compared to other supplies of water (Suarez and Gonzalez-Rubio, 2017). The specific chemical and biological nature of the municipal wastewater is dependent on the area that produced the treated municipal wastewater (TMW) (Elia et al., 1983). Depending on the treatment, TMW can contain harmful bacteria and viruses (Monarca et al., 2000). The properties of TMW depend on its provenance, with TMW from industrial areas having distinct chemical and biological properties compared to TMW from residential areas (Mattsson et al., 2016). The village of Duvauchelle at Banks Peninsula NZ produces municipal wastewater from residential sources. Table 1 shows the chemical nature of the TMW from the Duvauchelle treatment plant.

Table 1. Concentrations of chemical species found in TMW from the Duvauchelle treatment plant. Numbers in brackets show standard error (n = 3).

Chemical Species	Average concentration (mg/L)
NH_4^+	0.72 (0.00)
NO_3^-	34 (0)
NO_2^-	0.86 (0.5)
Ca	34 (1)
K	16 (0)
Mg	15 (0)
Na	92 (1)
P	6.1 (0.4)
S	13 (0)

Benefits of wastewater application to land

The properties of TMW make it difficult to deal with because of the environmental and health risks (Dotan et al., 2017). TMW is a potential resource, rather than a problem, due to the benefits, which are discussed by Patterson (2001). The benefits include irrigation, and fertilization from the nutrients in the TMW. The detrimental effect of discharging into a river or stream on water quality has also created appeal when considering alternatives to discharge of TMW into waterbodies (Vogeler, 2009). The irrigation of TMW onto soil instead of discharge into waterways is a practice that could reduce the environmental impact and improve agriculture and ecological processes (Sonune and Ghate, 2004).

Less eutrophication / algal blooms

Irrigation of TMW onto land as opposed to discharge into a waterway has positive effects on the water quality, as irrigating TMW onto land decreases the scale of algal blooms that would otherwise occur (Herath, 1997). Irrigation of TWM onto land creates a risk of groundwater contamination with nitrate and possibly eutrophication on adjacent freshwater bodies (Kathijotes, 2011).

Benefits of TMW as irrigation value

Despite TMW having negative environmental effects when discharged into waterways, these impacts are not felt when TMW is irrigated onto soil. The TMW properties, which cause the negative impacts in waterways, are beneficial when applied to soil. Irrigation increases the dry matter yield of plants, especially in dry climates when compared to scenarios with no irrigation (Goh and Bruce, 2005). Capra and Scicolone (2004) stated that TMW is an available water resource for irrigation in countries where agriculture is the largest water user. Irrigation with TMW allows the water to be reused for growth. One of the reasons for this is that in some parts of the world, particularly in the Mediterranean regions, it is more difficult to meet agricultural demands for water from conventional resources (Capra and Scicolone, 2004).

TMW contains nutrients that can be beneficial for plant growth. By irrigating with TMW, plants can receive the benefits from these nutrients without the need to apply fertilizers, thus reducing costs (Allegre et al., 2004). Reusing TMW can increase crop yields, as the sewage adds nutrients to the soil (Gebrehanna et al., 2014). TMW contains elevated concentrations of N, P, S, and K (Ozenc and Ozenc, 2015; Zhen et al., 2015). In New Zealand, the most common nutrient added to soil as fertilizer is P (Bolan et al., 1990). Most soil P is unavailable for plant uptake (Busato et al., 2017). Only P in a solution chemical species is mobile in soil and able to be taken up by plants (Yang and Post, 2011). Guo et al. (2000) stated that the primary source of soil P is in mineral form, such as apatite minerals, which contain tricalcium phosphate, which is only sparingly available to plants and rapidly immobilised in soil (Nezat et al., 2008; Yang and Post, 2011). Phosphorus in TMW is water-soluble, so all of the P is available for plant uptake. Loganathan et al. (1996) found that New Zealand superphosphate has high levels of cadmium (Cd), which is applied to soils with the superphosphate. The Cd accumulates in the soil and is taken up by plants, presenting an economic and human health risk (Al Mamun et al., 2017). By irrigating with TMW, which has had heavy metals removed, nutrients can be applied to the soil, which lowers the need for fertilizer and reduces the amount of Cd applied to the soil.

Contaminant breakdown in soil

While the chemical nature of TMW is beneficial as irrigation, there are also biological properties. Mandal et al. (2007) reported that microorganisms which can be harmful waterways are rendered harmless in soils. This study showed that various plants have been shown to have antibacterial capabilities and have been able to reduce the amount of toxic bacteria which remains in the TMW and is irrigated onto the soils with the TMW (Mandal et al., 2007). Plants have been used to clean organic pollutants because of this (Chaudhry et al., 2005). Microorganisms are killed in the rhizosphere, and the resulting carbon chemicals are used to stimulate plant growth (Alagic et al., 2015). By irrigating with TMW, harmful microorganisms present in the TMW do not enter waterways, and may be destroyed in the soil.

Risks

While irrigating with TMW has several benefits, there are some drawbacks. Irrigation onto soil places waste remaining in TMW into the soils. This can lead to potential environmental hazards on land, such as nutrient leaching from, and toxic contamination of soil, plants and groundwater.

Nutrient leaching and groundwater contamination

An increase in irrigation can cause an increase of leaching loss from soil due to the larger volume of water in the soil (Houlbrooke et al., 2003). TMW can also add nutrients to the soil, which are of a mobile, and therefore leachable, chemical species. This application of water and excess nutrients can result in excessive leaching and consequent contamination of groundwater (Sepaskhah and Tafteh, 2012). The loss of nutrients from the soil due to leaching represents an economic loss of nutrients that would otherwise enhance plant growth. Groundwater contamination has health risks due to the increase of chemicals, such as nitrate in the water (Chen et al., 2016). Studies that monitor groundwater quality from TMW irrigation examine DOC, N and P in various chemical species and major ions (de Miguel et al., 2014).

Plant and soil exposure to toxic agents

Despite treatment, TMW can still contain nutrients, xenobiotics, heavy metals and pathogens (Akpor and Muchie, 2011). Pharmaceuticals and heavy metals can accumulate in municipal wastewater treatment plants (Table 2), which creates the risk of soil and plant contamination due to irrigation with TMW (Bair et al., 2016). Contaminants may include organic chemicals, such as herbicides, fungicides and insecticides, which are discharged with the TMW, and can cause environmental issues (Wang et al., 2010). The application of some heavy metals, such as copper (Cu) and zinc (Zn), can be beneficial to plants as micro-nutrients (Spark and Swift, 2008). An excess application of these metals can lead to soil accumulation and toxicity (Rouphael et al., 2008). However, the sewage treatment process is effective at removal of heavy metals and organic chemicals, such as pharmaceuticals, from raw sewage, leaving the TMW with minimal toxic agents, (Table 1) (Camargo et al., 2016).

Table 2. Soil concentration ranges and regulatory guidelines for some heavy metals (Wuana and Okieimen, 2011).

Metal	Soil concentration range (mg/kg)	Regulatory limits (mg/kg)
Pb	1.00 – 69 000	600
Cd	0.10 – 345	100
Cr	0.05 – 3 950	100
Hg	<0.01 – 1 800	270
Zn	150 – 5 000	1 500

Applying TMW to land can create an environment that is favourable to waterborne pathogens, some that produce toxins (Akpor and Muchie, 2011). Adding nutrients can encourage bacterial growth (Xu et al., 2012). These organisms can in turn produce toxins, which poison plants, animal and essential microorganisms, as well as competing with the microorganisms (Kalb et al., 2015).

Faecal coliform and *E. coli* in sewage are examples of this (Fremaux et al., 2008). Primary sedimentation of the sewage treatment process is only likely to remove the heavier parasite eggs and so the resulting TMW will still be high in bacterial, fungal and viral numbers, (Table 1) (Health, 1992). Secondary biological treatments are able to reduce these numbers, such as chlorine disinfection and ultra-violet (UV) radiation.

Aggregate stability and infiltration rate

The long-term application of TMW onto soil can have a negative effect on the soil's hydraulic conductivity and aggregate stability. Soil aggregates disperse because of high concentrations of monovalent cations, such as Na^+ (Cannon et al., 2012). TMW has high concentrations of Na (92 mg/L for the Duvauchelle treatment plant), which is why there is a risk that irrigating with this TMW can cause soil dispersion and aggregate instability (Mojid and Wyseure, 2013). Aggregate stability is important for soil sustainability and crop production (Amezketa, 1999). Tang et al. (2011) showed that agrichemicals, including some fungicides and bactericides, also reduce aggregate stability. The agrichemicals do this by killing microbes in the soil. The experiment showed that bacterial and fungal activity was important to the formation of soil aggregates, and that aggregate stability was compromised when fungi and bacteria were killed. Any fungicide and bactericides that were in the DOC when the TMW was irrigated onto land could damage the aggregate stability. If the soil aggregates break down (de-flocculate), then the several aspects of the soil are affected. The clay particles are dispersed, which affects various properties of the soils, such as the aeration and infiltration (Vogeler, 2009) as well as the cation exchange capacity (CEC) (Teh, 2012). A reduction in the CEC results in an inability of the soil to remove ions from the soil solution and prevent them from leaching (Hartmann et al., 1998). Dispersion of clays has major effects on the quality and use of a soil as it reduces the soil's productivity (Voelkner et al., 2015). The ion buffer can contain Ca^{2+} and magnesium (Mg^{2+}) ions that help to prevent soil aggregate instability (Cannon et al., 2012).

Soil dispersion causes clogging of the soil pores, which decreases the soil's infiltration and aeration rate (Dikinya et al., 2006). Irrigation reduces the soil infiltration rate, as an application of water (Gardiner and Sun, 2002).

Soil erosion

A decrease in the soil infiltration rate can lead to ponding and surface runoff (Gardiner and Sun, 2002). These in turn have negative consequences, such as soil erosion and pollution transfer (Dehotin et al., 2015). Erosion of soils by water can transfer all pollutants that were present in the soil at the time of erosion. This can lead to pollution and eutrophication of waterways with all the pollutants and nutrients from the soil, not just those present in the TMW (Prasuhn, 2011). Studies have shown that irrigation with TMW can cause greater sediment loss than water irrigation (Flanagan and Canady, 2006). High aggregate stability has been shown to cause resistance to soil erosion (Annabi et al., 2011). This is also true for runoff, and the soil infiltration rate and aggregate stability are closely related to soil erosion and runoff (Jitander et al., 2012). The smaller aggregates have a higher infiltration rate, which leads to less runoff and erosion (Goebel et al., 2012).

Cultural aspects

The discharge of TMW into water bodies has cultural implications. Maori are supportive of TMW not being discharged into the Akaroa harbour (Te Runanga o Ngai Tahu, 2015, July; Xu et al., 2012). They are also supportive of the decision to decline the Christchurch City Council's application to discharge TMW into the Akaroa harbour (Te Runanga o Ngai Tahu, 2015, July).

Case of Duvauchelle

The wastewater scheme at Duvauchelle was built in 1988 (Christchurch City Council, 2015). There have been minor upgrades to the plant in 1996 and again in 2002 to service the 250 residents. The treatment process has primary and secondary treatment, then UV disinfection (Christchurch City Council, 2015). The primary treatment removes solids from the raw sewage by screening and sedimentation, and is also known as mechanical treatment (World Bank Group, 2015). Up to 70% of waste can be removed by this treatment (Sonune and Ghate, 2004). Secondary treatment removes most (85%) of the remaining suspended solids and biodegradable organic material (World Bank Group, 2015). The primary and secondary treatments remove most of the waste from the raw sewage (Sonune and Ghate, 2004). Many larger treatment plants require tertiary treatment because of the large volumes of waste, but smaller ones, like the Duvauchelle plant service a small community with a smaller waste generation, and therefore do not require a treatment plant with tertiary treatment (Sonune and Ghate, 2004). Anastasi et al. (2013) reported that the UV irradiation is a favourable treatment form, as it leaves no chemical residue that other treatment forms such as chlorination, do, because it is a physical treatment process. The UV radiation is emitted from mercury-amp vapour lamps, but has been shown to be less effective as a disinfection technique than chlorination. The resource consent for the treatment plant's discharge into the Akaroa harbour (CRC 102952) expires in 2023 (Christchurch City Council, 2015). The Christchurch City Council is required to develop a list of wastewater treatment and disposal options (Christchurch City Council, 2015).

Other studies

The use of TMW for crop irrigation is common throughout the world for an alternative water source. In Israel 50%, of crops are irrigated with recycled water (Suarez and Gonzalez-Rubio, 2017). Australia uses TMW irrigation because of the benefits of water conservation, water recycling and surface and groundwater contamination prevention (Muyen et al., 2011). However, many parts of Australia, especially the parts that need the most irrigation, have low rainfall, high evaporation and low leaching rates. This causes soil salinity due to the high concentration of Na in the TMW (Muyen et al., 2011). An increase in the saturated hydraulic conductivity and soil water retention was recorded from TMW irrigation onto wheat and potato fields in Bangladesh (Mojid and Wyseure, 2013). This TMW was recorded to have high levels of Ca^{2+} and Mg^{2+}, as well as Na^{+} and K^{+}. Jenkins and Russell (1994) found that heavy metals can be present in municipal wastewater from household washing products, and other sources. The study found that these metals are at a low concentration and make up 0.5% of metals in the wastewater. Lado et al. (2005) reported that effluent contains high levels of Na, which affects the soils chemical and hydraulic properties. Suarez et al. (2006) showed that soil's physical properties were affected by irrigation containing heightened levels of Na. There was a reduction in the infiltration rate. Buckland et al. (2002) found that Na adversely affected the soil aggregate stability. Many studies have shown that Na in TMW causes a reduction in infiltration rate. Vergine et al. (2017) found that TMW irrigation onto crops has been found to be a suitable alternative to fresh water. The study from Italy found that the TMW did not adversely affect the tomato or broccoli crops through faecal contamination. Irrigation with TMW aided the growth of these crops, and the study concluded that TMW was a suitable source for irrigation. Suarez and Gonzalez-Rubio (2017) conducted a study on an Arlington sandy loam soil in California USA. They found that the TMW had a high sodium adsorption ratio (SAR) and DOC, which affected the soil's physical properties. This was a reduction in the infiltration rate and aggregate stability of the soil.

Based on previous studies, it was hypothesised that the Na in the TMW irrigation would degrade the soil structure. Sodium was expected to accumulate in the soil, increasing the soil's SAR. This in turn would decrease the infiltration rate and aggregate stability, and increase the bulk density of the soil.

Aim

The objective of this thesis is to identify likely short and long-term environmental effects of Na in TMW irrigation of the soils of Banks Peninsula. This thesis also aims to determine how water infiltration, and travel (hydraulic conductivity and leaching) through soil is affected by soil aggregate stability, and how Na causes soil dispersion, leading to a decrease in infiltration rate.

Background

Production of TMW

Coarse solids are removed in the initial (primary) stage of wastewater treatment through the use of screens and settling ponds (Figure 1) (EPA, 2016). This process removes grit and sediment from the wastewater. Organic matter is removed in secondary treatment (EPA, 2016). The wastewater passes through a series of filters that bacteria grow on. The bacteria remove nutrients from the water as it passes (Christchurch City Council, 2016). This process is sometimes combined with the primary stage. Pathogens are removed from the wastewater in the tertiary stage. UV light is often used to kill off the pathogens at this stage (Watercare, 2010).

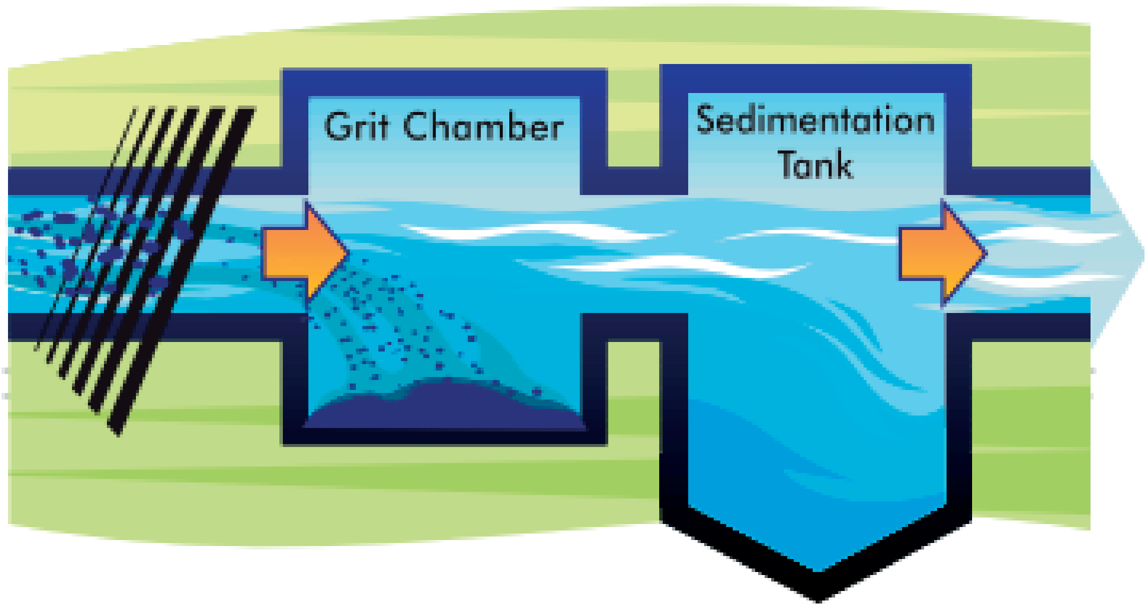

Figure 1. Diagram of screens and settling tanks in primary wastewater treatment. Solids are trapped by the screens or sink to the bottom of the settling tanks while the liquid wastewater passes through, (EPA, 2016).

After the treatment process, the treated municipal wastewater (TMW) is discharged from the plant. One common place of discharge is into a waterway, such as a river or ocean (Christchurch City Council, 2016). The Christchurch City Council discharge TMW off New Brighton beach 3 km from the shore. Other methods of discharge include the irrigation of TMW on to land (EPA, 2016).

An increase in water quality is desirable for several reasons as fresh water is one of the most important resources available (Widomski, 2014). Algal blooms can cause toxins, which can be harmful to human health, such as cyanobacteria (Eynard et al., 2000). The toxins that these produce can cause rapid variations in the water quality, which can greatly affect human uses of water, such as drinking water, as some toxins produced by waterborne micro-organisms are potent enough to kill humans (Harris and Smith, 2016). An increase in the algae population in a waterbody can also cause a critical decrease in the dissolved oxygen levels in the water (Gebrehanna et al., 2014). This has major effects on the health of the ecosystem as dissolved oxygen is essential for sustaining aquatic life (Bailey and Ahmadi, 2014).

Discharge of TWM into waterways adds nutrients such as N and P to the water, which increases eutrophication, and induces algal blooms (Bae et al., 2015). The main nutrients contributing to eutrophication, and so algal blooms are N and P, because of the Redfield ratio, which are present in TMW (Flynn, 2010). The Redfield ratio is a ratio of C, N and P, (106:16:1), which is the chemical composition of phytoplankton and algae (Tett et al., 1985). Discharging of TMW into waterbodies can prevent the water from meeting designated uses, such as recreational, freshwater drinking supply or stock water (Carey and Migliaccio, 2009). By not discharging into waterbodies with a designated use, the water uses can be protected, meeting the quality limits needed for the water use, as the TMW would not have contaminated the waterbody.

Chemistry of TMW

Municipal wastewater comprises of waterborne solids and liquids that are discharged into sewers that have formed from human waste and industrial sources (Sonune and Ghate, 2004). This can be from industrial and domestic activities (Sonune and Ghate, 2004). Jenkins and Russell (1994) found that heavy metals can be present in municipal wastewater from household washing products, and other sources. The study found that these metals are at a low concentration and make up 0.5% of metals in the wastewater.

Xenobiotics surround human life. They can be found in polymers to purify drinking water, soaps, detergents and cosmetics (Awad and Ghany, 2015; Laha et al., 2009; Lee et al., 1998). These chemicals can be discharged into effluent, and end up in TMW (Awad and Ghany, 2015). There are many industrial and domestic materials that can be found in TMW from a variety of sources, such as pesticides, prescription and non-prescription drugs, as well as the aforementioned products (Alvarez et al., 2005). The degraded residue chemicals of the original organic chemicals from the various sources are also found in TMW. Antibiotic-resistant bacteria from hospitals can be discharged into municipal treatment plants (Hocquet et al., 2016). Wastewater treatment plants are a sink for antibiotics, and a variety of bacteria, including antibiotic resistant bacteria from hospitals and other healthcare facilities (Sigala and Unc, 2012). This allows a breeding ground for these bacteria to grow. There are many other xenobiotics found in TMW. Some of these include endocrine-disrupting compounds (EDCs), from common household chemicals, such as the contraceptive pill (Mastrup et al., 2005). These have cast doubt on the safety of reusing TMW (Xu et al., 2009). Once the EDCs have been released into the environment, they have a large disruptive effect on the reproductive, immune, neurological and developmental systems of fauna (Cooper and Kavlock, 2001).

Nitrogen and P occur at elevated concentrations in TMW (Carey and Migliaccio, 2009). TMW can have other elements such as organic C, Na, K, calcium (Ca), Mg, sulphur (S), Zn and boron (B) (Mojid and Wyseure, 2013). From the treatment plant the chemicals can be discharged into the environment with the TMW.

Aggregate stability

There are several parameters of the soil that determine whether a soil will have stable aggregates. These aggregates are made up of various soil particles, such as organic matter, sands, silts and clays, (Figure 2). The main reason that soil aggregates bind together is due to a balance between Van der Waals forces and electrostatic forces (Atmuri et al., 2013). The Van der Waals attraction causes the soil particles and material to clump together if they come into close contact (Atmuri et al., 2013). The soil solution has many free flowing ions in it. These ions attract and repel each other depending on their charge type (Ferrari et al., 2015). In most New Zealand soils, cations predominately bind to the soil, while anions are leached down and out of the soil profile. If the electrostatic repulsion of the cations that bind to the soil particles is larger than the Van der Waals attraction forces, then the soil particles will disperse (Ruckenstein and Manciu, 2003). This is because the cations in the soil solution form a shell around the soil particles, (Figure 3). The cation charges repel each other. If the repulsion forces are too large to overcome, then the soil particles will disperse, (Figure 4). If the Van der Waals attraction is larger than the electrostatic repulsion, then the soil will form aggregates.

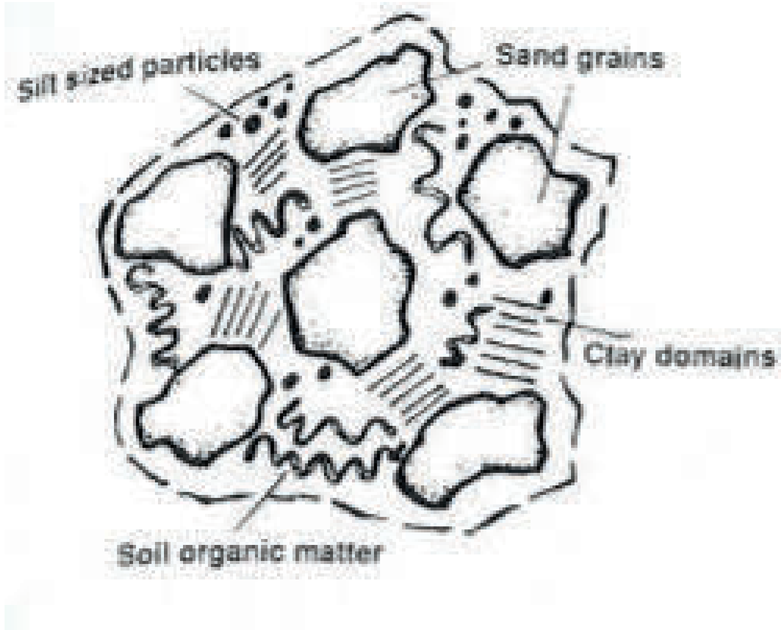

Figure 2. Diagram showing the structure of a soil aggregate. The sand and silt are held together by the clays and organic matter *(McLaren and Cameron, 1996).*

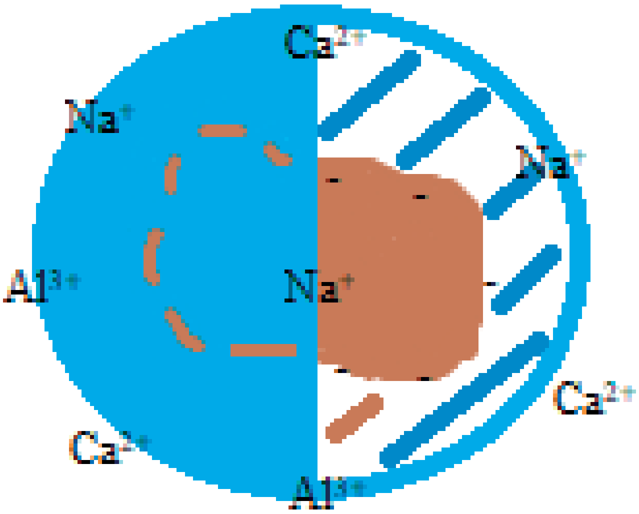

Figure 3. Diagram showing cations forming a shell around a soil particle with a net negative surface charge.

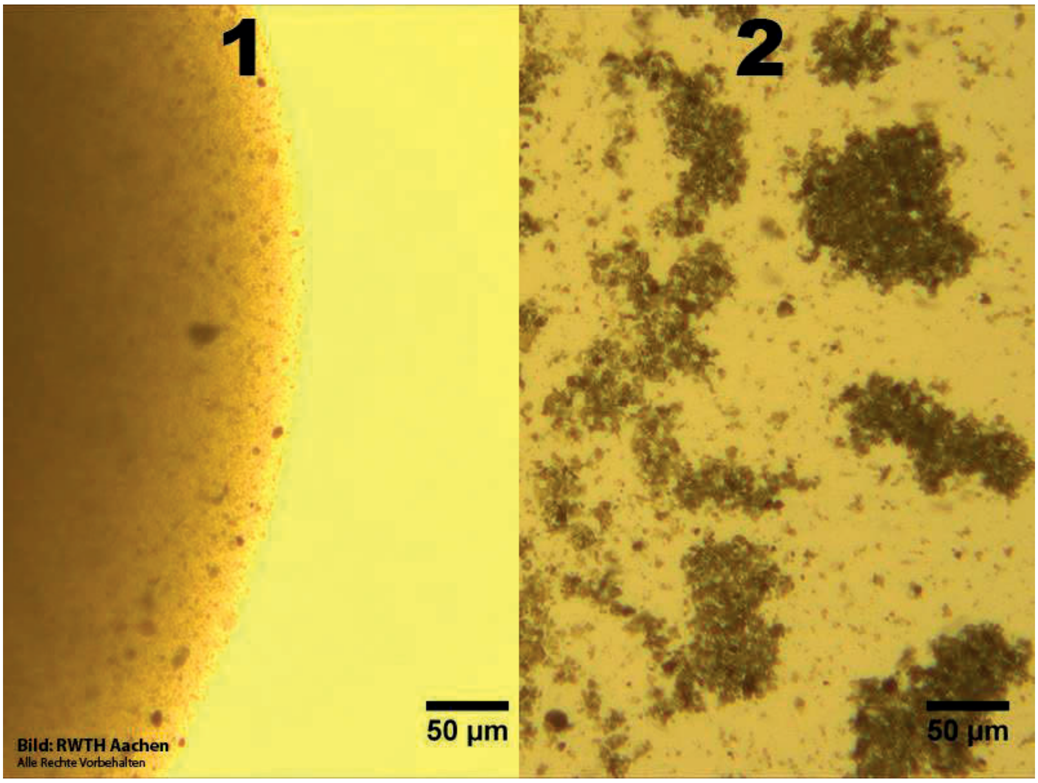

Figure 4. Photo showing (1) dispersed material and (2) flocculated material. (1) shows that the material is evenly dispersed throughout the solution, while (2) shows the material (the dark particles) is clumping together in the solution.

Aggregate structure

A soil aggregate is a group of primary soil particles that bind to each other more strongly than they adhere to other particles surrounding them (Kemper and Rosenau, 1986). Aggregates adhere to each other with slightly weaker bonds than those that hold the original aggregate together. This is how primary particles, (clays, silts, sands and organic matter) form micro aggregates, which then form larger macro aggregates.

In soil, inorganic matter comprises sands, silts and clays made up of mineral compounds such as silica and aluminium (Belver et al., 2012). Soil organic matter is made up of decaying biomass from microorganisms, plant and animal death, as well as soil microbes (Kogel-Knabner, 2002). There are several hierarchical orders to soil aggregates (Amezketa, 1999). Micro aggregates in the < 2 mm diameter range consist of clay and organic materials. These micro aggregates combine to form the next order micro aggregates of the < 250 mm diameter range. These micro aggregates then combine to form macro aggregates that are > 250 mm. The macro aggregates finally bind to each other to form clods that are in the mm - cm range, (Figure 5).

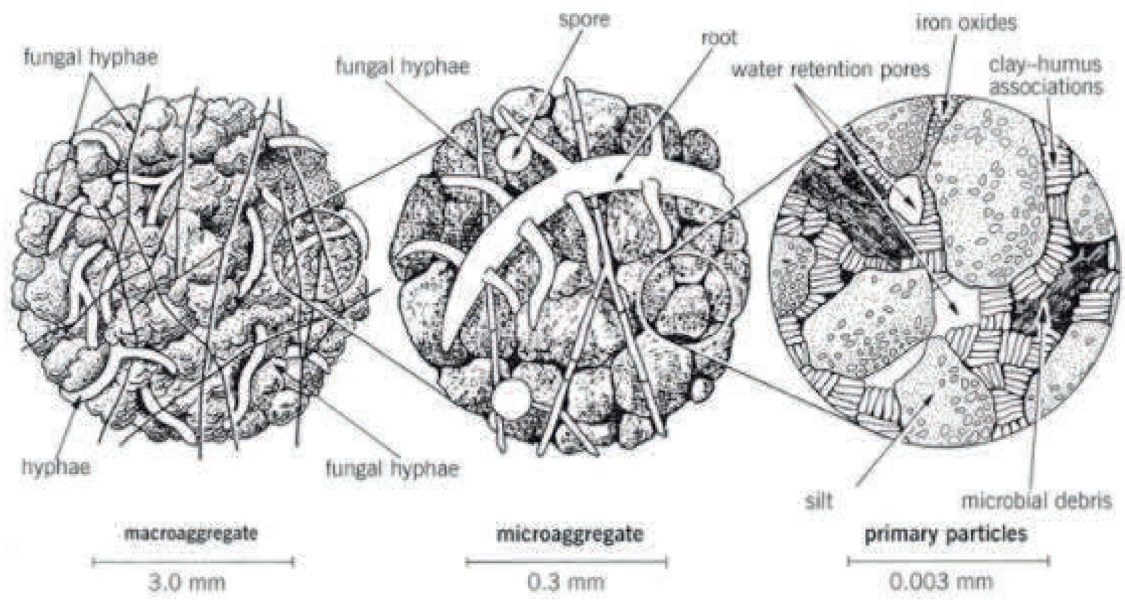

Figure 5. Diagram showing the hierarchical structure of soil aggregates. Macro-aggregates, micro-aggregates and primary particles (Dubbin, 2001).

Soil aggregation is an ongoing process, which is caused by the physical forces involved (Amezketa, 1999). The balancing of the attractive forces holding aggregates together and the repulsive forces that disperse the soil determines the soil's stability. If the equilibrium of the forces favours the repulsive forces then the aggregates are dispersed. If the attractive forces are greater, then the aggregates are stable. Multi charged cations, such as Ca^{2+}, Fe^{3+} and Al^{3+} provide forces that cause aggregation to occur.

The Gouy-Chapman model

Electrostatic forces can be modelled by the Gouy-Chapman model, which measures the concentration of ions at a distance from the surface of a soil particle, (Figure 6). The Gouy-Chapman model explains why a soil has aggregate stability (Figure 7), or dispersion of soil particles, (Figure 8), as it can be used to show the electrostatic forces, and whether they will repel the soil particles enough to cause dispersion. The Gouy-Chapman layer is an area around the surface of a colloid or aggregate, that has an imbalance in the concentration of charges, to where there is an equal concentration of anions and cations (Lamperski and Bhuiyan, 2003). A soil aggregate has a net negative charge on its surface, which attracts cations and repels anions (Rotenberg et al., 2007). This is why there is a cation shell around each of the soil particles. The double layer is the length from the surface of a colloid or aggregate that has a greater concentration of one type of free floating charge (Lamperski and Bhuiyan, 2003). Essentially, the double layer is the cation shell. The shell is not entirely cations though. Figure 6 shows that the concentration of anions increases with the radius from the colloid surface, as the cation concentration decreases. The end of the double layer is where the concentration of the anions matches that of the cations. The Gouy-Chapman layer is the double layer and it has a greater cation concentration than anion, because of the aggregate negative charge (Lamperski and Bhuiyan, 2003), (Figure 9). The greatest concentration of cations is close to the aggregate surface because that is where there is the most electrostatic attraction for cations, and the most repulsion for the anions. The aggregate surface charge attracts the cations and repels the anions, resulting in this double layer (Rotenberg et al., 2007). The cations and anions become more mixed as the distance from the soil particle surface increases as the electrostatic attraction to the cations decreases as the charges become more balanced (Lamperski and Bhuiyan, 2003), (Figure 6 and Figure 9). The cations have greater repulsion as the cations on the surface increase. This also lessens the repulsion of the anions to the surface of the aggregate, and increases electrostatic attraction between the free anions and cations in the soil solution. The double layers repel each other, causing the soil particles to be pushed apart (Ferrari et al., 2015). Soil particles can flocculate if the double layer is small (Atmuri et al., 2013). This is because the soil particles are able to overcome the electrostatic repulsion and come close together. This allows the Van der Waals forces to dominate and cause the soil particles to clump together (Ruckenstein and Manciu, 2003). When the double layer is too long, the soil particles are unable to become close enough to clump, resulting in dispersion of the soils (Ruckenstein and Manciu, 2003).

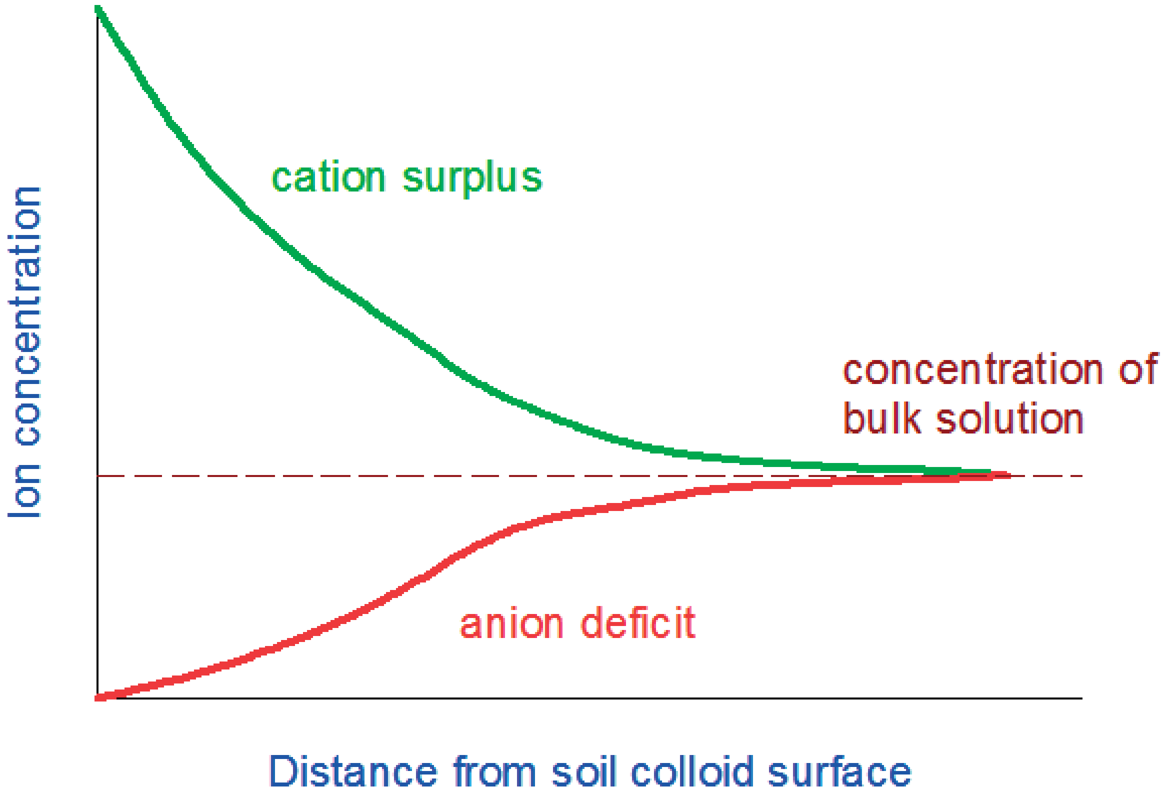

Figure 6. Concentrations of anions (red) and cations (green) over the double layer (Gouy-Chapman layer) from the soil colloid surface (McLaren and Cameron, 1996).

Figure 7. Photo showing stable soil aggregates.

Figure 8. Photo showing dispersed soil material.

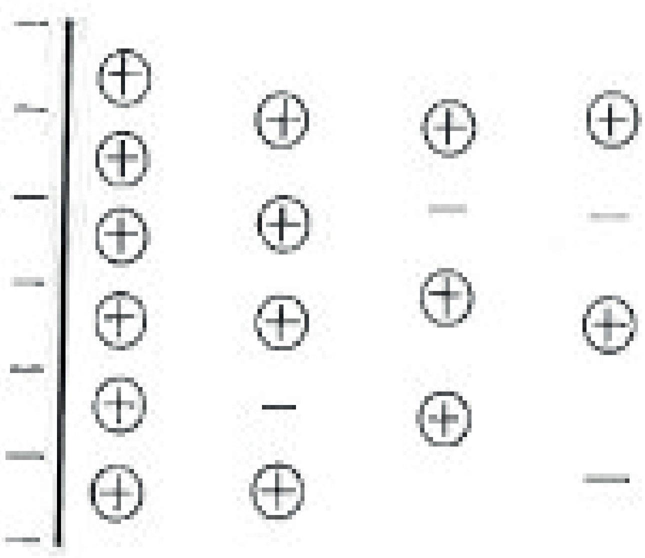

Figure 9. Diagram showing the ions over distance from the surface of the soil aggregate (McLaren and Cameron, 1996).

Ionic strength

The ionic strength and ionic radius of the ions has a major effect on the stability of soil aggregates. It also goes a long way to explain why certain elements are able to stabilize a soil and the aggregates, while others cause clay dispersion. Each soil aggregate has a fixed surface charge on it and the double layer balances this net charge (Rotenberg et al., 2007; Ruckenstein and Manciu, 2003). The higher the surface charge, the more opposite charge is needed in the double layer, provided by the cations in the soil solution to balance it

(Lamperski and Bhuiyan, 2003). A higher charged cation will require fewer atoms to balance the charge, as a bi-charged cation will neutralize two negative charges on the surface of the aggregate, while a mono-charged cation will only neutralize one negative charge each, (Figure 9). A double charged cation only requires half the atoms to balance the surface charge that a mono-charged cation requires. This means that the higher the cation charge, the shorter the double layer is. This is why Ca^{2+} is more stabilising to soil aggregates than K^{+} and why Mg^{2+} is more stabilising than Na^{+} (Cannon et al., 2012).

The hydrated radius

The hydrated radius of ions in solution also helps to explain why some ions and minerals will cause dispersion while others stabilise a soil. Water is a dipolar molecule, as the electronegativity on the oxygen atom is greater than that of the hydrogen atoms (Fries et al., 1995). This causes the electrons on the water molecule to be drawn to surround the oxygen atom more than the hydrogen atoms, (Figure 10) (Guggemos et al., 2015). This means that it is able to surround an ion (such as a cation), because of electrostatic attraction, (Fries et al., 1995). The higher the electronic density of the ion, the more water molecules will be attracted to the ion (Ghosh and Islam, 2011). The distance from the centre of the ion to the edge of the water molecule furthest from the ion, that is still attracted to the ion, is the hydrated radius of the ion (Tansel, 2012), (Figure 10). Electronic density decreases as the periods increase, (going down the periodic table). This is why Na^{+} is more destabilising than K^{+} and why Ca^{2+} is more stabilising than Mg^{2+}. By combining the ionic strength with the hydrated radius, Na^{+} is shown to be the most destabilizing of the four common cations (Na^{+}, K^{+}, Mg^{2+} and Ca^{2+}), as it has a low ionic strength and a large hydrated radius. Calcium is the most stabilizing as it has a small hydrated radius and a large ionic strength. The SAR is a measurement that is used to determine if a water-mass is suitable to apply to a soil without it causing a breakdown in aggregate stability (Rahimi et al., 2000; Wright and Rajper, 2000), (Equation 1). It is a ratio of the concentration of Na^{+} (since it is the most destabilising of the common cations) and the combined concentrations of Ca^{2+} and Mg^{2+}, as they are the most common stabilising cations. If the SAR is low enough, then the soil aggregates are stable. If not then the soil particles are likely to disperse (Tajik et al., 2003). This is because a high SAR indicates that there are more Na^{+} ions present than Ca^{2+} or Mg^{2+} ions. The Na^{+} ions dominate the Gouy-Chapman layer, causing the aggregates to be unstable, as there is a long double layer, (Figure 11).

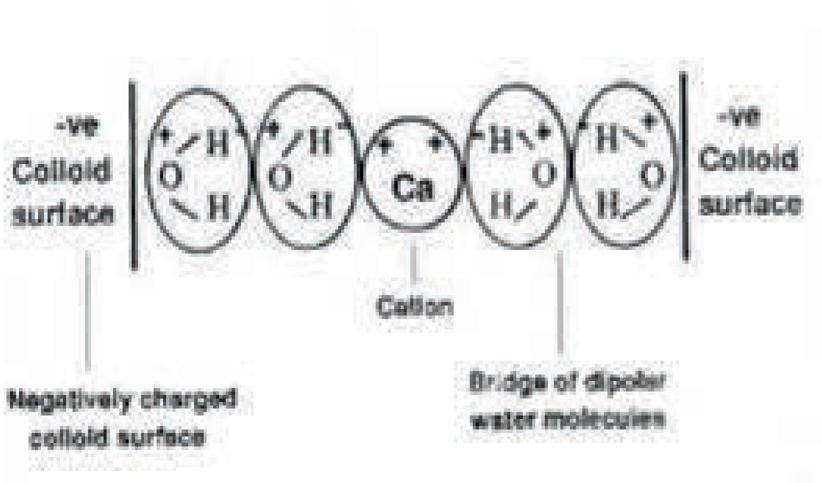

Figure 10. Diagram showing water molecules surrounding a cation. The water molecules increase the distance between a cation and the soil particle surface, increasing the double layer (McLaren and Cameron, 1996).

$$SAR = \frac{[Na]}{\sqrt{[Ca] + [Mg]}}$$

Equation 1. Sodium adsorption ratio equation. A comparative measurement of the concentrations of Na, Ca and Mg.

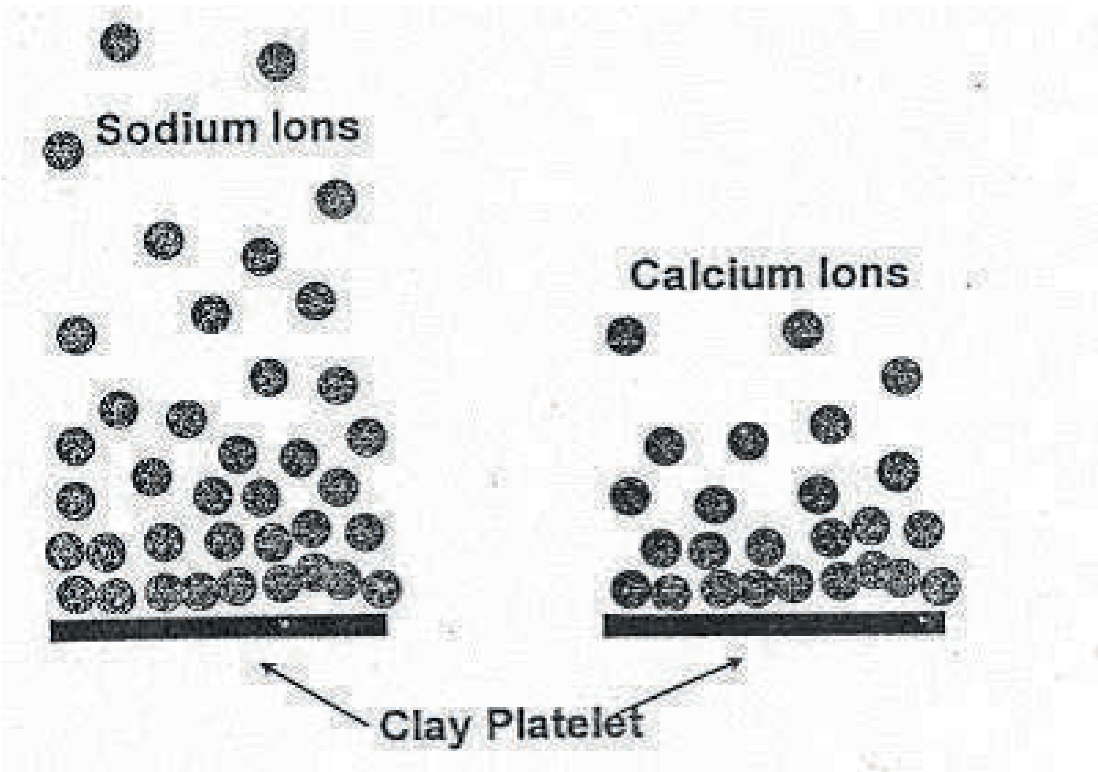

Figure 11. Diagram showing the difference in length of the double layer when it is dominated by Na^{+} and by Ca^{2+}.

Other soil properties affecting soil aggregate stability

Many factors influence soil aggregate stability. These are soil properties and parameters that vary from soil to soil. The aggregate stability is dependent on these parameters. The main effect on soil aggregate stability of these properties is their effect on the Gouy-Chapman model and the double layer. These properties can be divided into categories, such as biological, which is the soil organic and biological parameters and how they affect aggregate stability, mineral, which are the effect that the minerals in the soil have on aggregate stability and chemical factors. Chemical factors are the chemical properties that a soil has that affects soil aggregate stability. They are caused by a combination of mineralogy and organic composition of the soil.

Biological factors

Heavy vegetation cover has been shown to have positive effects on soil aggregate stability. The most stable aggregates have been found to be under forest or scrub, as these plants protect and enhance organic C. The canopy protects the soil from the raindrop effect and the roots stabilize the soil (An et al., 2010; Wilmshurst, 1997). This increases the aggregate stability of the soil. The plant roots are able to shape the micro-aggregates, and stabilise them by holding them in place, (Figure 12). Aggregation occurs because of the compressive and drying action of plant roots (Amezketa, 1999). The soil material is pressed together, which helps to form aggregates. Plants uptake soil moisture through the root system (Markewitz et al., 2010). Removal of moisture from the soil decreases the hydrated radius, and stabilizes the soil aggregates, (Figure 10). Mucilage released from the biomass of the soil (roots and microbes) sticks quickly to soil particles and is therefore able to stabilise soil aggregates (Morel et al., 1991). Soil aggregates have been most commonly found in the topsoil (the A horizon). Macro-aggregates have high C:N ratios, which indicates a high organic C turnover, which indicates topsoil (An et al., 2010). The organic matter in the soil has a strong correlation with aggregate stability. It is the glue that bonds the micro-aggregates together into macro-aggregates (An et al., 2010). This has been shown as the macro-aggregates have a higher organic C contribution than that of the micro-aggregates. A soil with low natural C has been shown to have low natural stability (Annabi et al., 2011). An application of C onto these soils, (such as a silty loam soil) has been shown to increase the stability of the aggregates. Bacterial and fungal activity are important in aggregate formation (Tang et al., 2011). These microorganisms therefore increase aggregate stability (Tang et al., 2011). The higher the organic C levels are in the soil, the more stable the aggregates are (Tajik et al., 2003). Soil organic matter contains most of the cation exchange capacity, as the deprotonation of the functional groups of the organic matter leaves negative charge (Oorts et al., 2003; Tan and Dowling, 1984). This is the variable charge in the soil, and forms the bulk of the CEC. The more organic matter there is, the more deprotonation there can be, and so the more CEC and aggregate negative surface charge there can be. The chemicals in the organic matter, are able to interact to form long chain molecules (such as esters and polysaccharides), which can trap and bind aggregates (Six et al., 2000), (Figure 13).

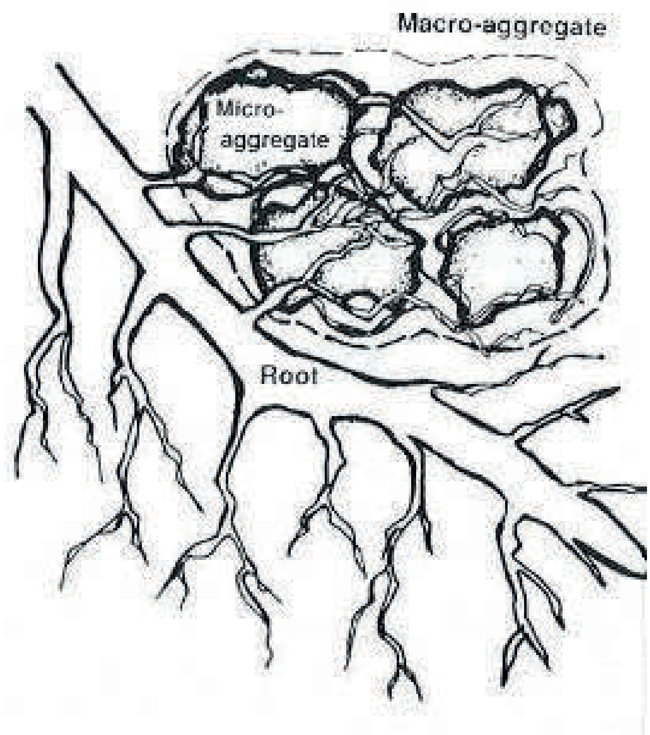

Figure 12. Diagram showing how plant roots are able to hold, shape and bind micro-aggregates into macro-aggregates (McLaren and Cameron, 1996).

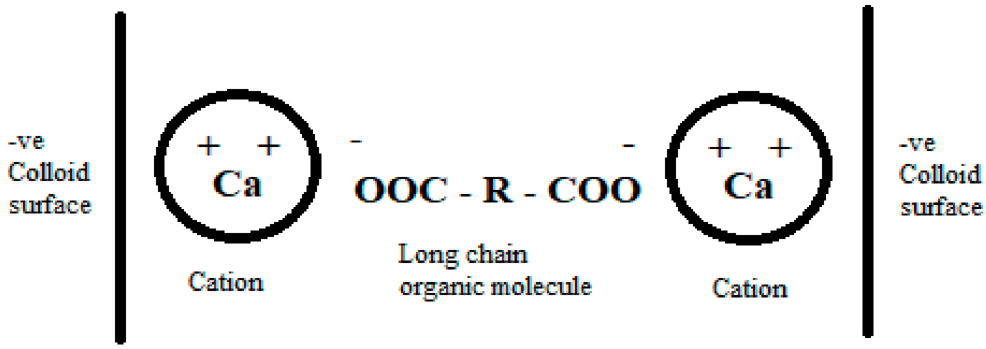

Figure 13. Diagram showing how soil organic matter can form bridges, connecting soil colloids to stabilize soil aggregates. Other methods include forming ester bonds between the micro-aggregates (McLaren and Cameron, 1996).

Mineralogy factors

Clay content is important for soil aggregate stability. Soils with low clay content have been shown to have low natural stability (Annabi et al., 2011). The clays act in a similar way to that of the soil organic matter when it comes to the enhancement of aggregate stability. The best texture is a mixture of sands, silts and clays, in addition to the organic matter. This is because the organic matter and the clays are able to form bonds through electrostatic attraction between the silts and sands that give the aggregates shape and structure. Sands and silts have no significant surface charge. Therefore they do not increase the CEC, as they add no net negative charge to the soil particle's surface (Nelson et al., 1999). However, they are able to give shape and structure to the aggregates, as they form the bulk of the material (Figure 2). Clays do have negative charge because of isomorphic substitution, therefore they contribute to the CEC (Sposito et al., 1999). In isomorphic substitution, elements of similar size are exchanged in the neutral structure of the clays, such as aluminium and silica for lower charged elements, leaving an overall net negative permanent charge. The aluminium silicates are also able to act as a glue, similar to the soil organic matter, and hold the aggregates together. A high clay content therefore increases the stability of the soil aggregates. The natural mineralogy of the soils affects the stability of the aggregates (Rotenberg et al., 2007). A soil, which has naturally high concentrations of Ca and Mg, has greater aggregate stability than that of a soil with naturally high geogenic Na.

Chemical factors

The CEC is the negative charge on the surface of a soil colloid. A high CEC is therefore going to increase the stability of the aggregates, as there will be more attraction to the cations, drawing them closer to the colloid's surface, and shortening the Gouy-Chapman layer (Zhang and Horn, 2001). The pH of the soil does not directly affect the stability of the soil aggregates. It does, however, affect the chemical species of various minerals in the soil. For example, calcium carbonate reacts to form bicarbonate and Ca^{2+} ions when exposed to acidic conditions, such as those in most New Zealand soils (McLaren and Cameron, 1996). This reaction releases Ca^{2+} ions into the soil solution, which can then balance the surface charge on the soil colloids, and shorten the Gouy-Chapman layer. A low pH is desirable to increase the stability of the soil aggregates because of this. The hydrated radius of the ions is dependent on the presence of water. This means that a low moisture level in the soil increases the stability of the aggregates. Without water, there is no hydrated radius, which will shorten the Gouy-Chapman layer.

Raindrop explosive effect

Soil is bombarded by raindrops, which, in the moment of contact, disrupts the soil particles, which disturb the soil structure (Boroghani et al., 2012). The explosive power of rain drops is more disrupting to soil structure than volume of rain (Farres, 1985). The energy of the raindrops affects the breakdown of structured units. Splash energy can be dissipated by detachment of micro-units from the larger structure.

The raindrop effect has been found to have a positive effect on sediment delivery (Beuselinck et al., 2002). The slash effect of raindrops is the first stage of soil erosion (Farres, 1985). Once the soil particles have been disturbed, and are no longer part of the soil aggregates, they can be moved by runoff, which is also provided by the rainfall (Boroghani et al., 2012). Raindrop energy is able to remove discreet soil particles and micro aggregates from their original location (Farres, 1985).

The raindrop effect contributes to sediment sorting and crust formation (Eldridge, 1998). The breakdown of soil aggregates into their smaller primary particles allows the soil to fill pore spaces and become more densely packed (Hairsine et al., 1999). The energy of the falling raindrop breaks the soil aggregates, as well as transferring kinetic energy in a sufficient amount for particle transportation. This allows the particles to be reorganised and sorted which fills soil pores, increases bulk density and compaction, and forms surface crusts *(Farres, 1985).*

Wetting and drying

Wetting can reduce aggregate stability (Kemper and Rosenau, 1986). The moisture gets in between the gaps within the aggregates and the primary material. The water then forces the particles apart by increasing the hydrated radius (Figure 10). When dry soil is wetted quickly, such as in a flood event, air present inside of the soil aggregate can become trapped and compressed inside the aggregate (Hillel, 2008). The air causes a pressure build up, which leads to a literal explosion of the aggregate due to air-slaking. This process is capable of reducing a well aggregated soil to a mass of mud that forms a surface crust upon drying. Irrigation can cause dispersion of the soils through a breakdown in the aggregate stability. This dispersion of soil fills soil pores, resulting in a lower macro porosity on irrigated blocks compared to control (Vogeler, 2009). This means that irrigation can cause a reduction in the soil's infiltration rate (Bjorneberg and Aase, 2000).

Hydrophobicity

High soil C sites caused TMW to increase hydrophobicity (Vogeler, 2009). Soil hydrophobicity causes runoff (Mataix-Solera and Doerr, 2004). This has benefits as a reduction in the soil moisture level can increase aggregate stability due to a shortening in the hydrated radius. Runoff is, however, an environmental hazard. Abraham et al. (2017)

reported that surface runoff causes erosion and contamination of waterways. Contamination of waterways occurs because the runoff brings soil nutrients and toxic chemicals from the surface of the soil to waterways.

Freeze-thaw

Freeze-thaw cycles have been shown to affect aggregate stability and size distribution (Wang et al., 2012). Freezing only and the freeze-dry process increase the soil aggregate stability in the majority of cases (Dagesse, 2013). Conversely the freeze-thaw process lowered the strength of soil aggregates compared to the control samples. The freeze process can increase aggregate stability (Dagesse, 2011). This is because ice crystal growth within interaggregate pore spaces has a desiccating effect on the aggregates, which contributes to aggregate stability. The desiccating effect of the freezing process increases aggregate stability. However, the additional effect of the thawing process, and the additional moistening that comes with it, is responsible for the decrease in aggregate stability (Dagesse, 2013).

Pores

Soil aggregate shape

Soil pore shape is a critical factor for soil fertility (RingroseVoase, 1996). This is because it affects root growth by providing pathways for expansion through the soil. Soil aggregates have a spherical like but irregular shape due to the complex relationship between the variety of materials that form the aggregates, and the forces that form them (Munkholm et al., 2016). The effects of the Gouy-Chapman layer causes a sphere to be the most desirable shape, as it has all the soil particles as close to the centre of the aggregate as possible (Lamperski and Bhuiyan, 2003). The spherical shape of the aggregates means that no matter how they are arranged, there will be gaps between them, (Munkholm et al., 2016). These gaps are the soil pores.

Infiltration and aeration

Soil pores affect aeration and water infiltration (RingroseVoase, 1996). Pores provide pathways for water to be redistributed and soil drainage. Water moves into soil pores because of gradients in water content and potentials (Amer, 2012). Pores provide conduits that infiltration can occur through (Watanabe and Kugisaki, 2017). The size of the pores is important when determining the infiltration and aeration rates. Large pores result in greater infiltration and aeration, which promotes greater plant growth (Kemper and Rosenau, 1986). Larger pores have been shown to have more of an influence on the infiltration rate than the total porosity of the soil (Amer, 2012). Larger pores provide greater conductivity than narrow pores when both soils have the same total porosity.

Dispersion and packing

The soil bulk density and the total soil porosity are closely linked. This is because the pores are empty space without soil material. Therefore the lower the bulk density, the higher the total soil porosity (Vogeler, 2009). Dispersion of soils causes the bonds between the various parts of the soil aggregates to break, splitting the aggregate apart. The smaller aggregates, and soil particles that are formed when the larger aggregates break up, fill pore spaces, which contributes to surface crusting (Figure 14) (Amezketa et al., 2003).

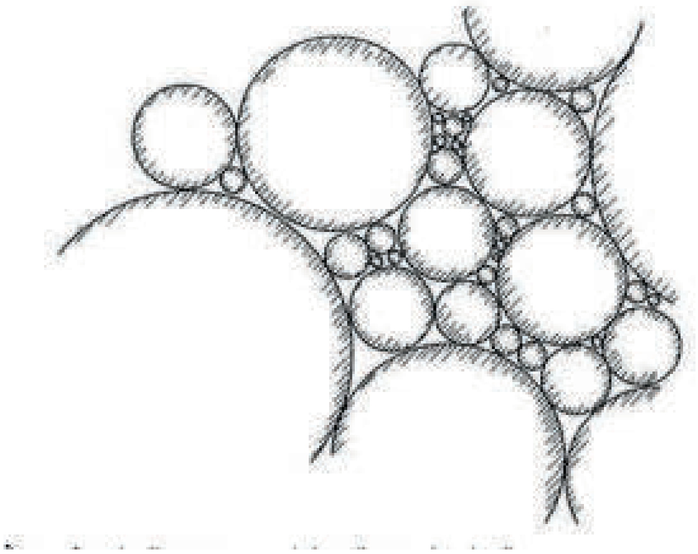

Figure 14. Diagram showing particle packing (Hillel, 1998).

Surface sealing and crust

When soil dispersion occurs, a crust can form on the surface of the soil (Roth, 1997). The formation of a soil crust has several negative effects on the soil, and the use of the soil for beneficial human uses (Castilho et al., 2011). Crusts form as the soil porosity decreases (Jakab et al., 2013). This is expected as the pores are filled with material, forming the crust. Soil crusts prevent infiltration, which leads to an increase in runoff and an increase in soil erosion (Castilho et al., 2011). Gypsum has been recorded to have an effect in reducing soil crust formation (Borselli et al., 1996). This is because gypsum ($CaSO_4$) is able to stabilise soil aggregates, which increase the soil porosity, and remove the soil sealing and crust.

Materials and Methods

Site description

On the 28th of August 2014, a site visit was made to Duvauchelle Golf Course (Barry's soil) and the Takamatua Peninsula (Pawson silt loam). Soil pits were opened with a view to ascertain whether the soils would be suitable for lysimetry, namely that they would have an adequate permeability to allow significant through-flow of water. Soil pits revealed both soils to be imperfectly drained (some mottling) but no evidence of a fragipan, perched water, or impermeably (reduced iron). The mean (standard deviation) of the size fractions for these soils are: coarse sand 1.2 (0.2)%, fine sand 44.5 (0.9)%, silt 28.1 (2.1)% and clay 24.0 (2.2%) (Anon, 1939b). The location of these sites are shown below, (Figure 15).

Figure 15. Locations where the lysimeters were excavated and of the ongoing field trial where TMW is being irrigated onto NZ native vegetation.

The site at the Duvauchelle field trail where the soil columns were collected was a field 500 m from the Duvauchelle treatment plant and 500 m from the Duvauchelle Bay. The area is on a gentle incline leading to a steep hill. The field is next to a farmhouse and near the residential section of Duvauchelle town. The hill to the north of the site adds colluvium to the soil. The associated colluvium included small semi-weathered stones and other material. The site was fenced off seven months prior to the collection of columns. The field was previously part of a sheep farm.

The area receives an average annual rainfall of 1,200-1,400 mm/yr (Macara, 2016). The average daily air temperature in summer is 20-22 ºC and the average daily air temperature in winter is 3-5 ºC. The site receives an average of 1,950-2,000 annual sunshine hours.

Soil type

Table 3. Soil types on the Banks Peninsula. Brackets represent standard errors (Anon, 1939a; Griffiths, 2012; Trangmer, 1986).

Soil type		Barry's soil		Pawson silt loam			
Location		Duvavchelle Golf course		Takamatue Peninsula / Duvauchelle test site			
Clay %		10.8 (0.4)	8.7 (1)	11.2 (0.5)	8 (1.3)	9.8 (0.9)	8.3 (0.7)
Silt %		28 (0.6)	22.1 (1.8)	29 (0.9)	22.5 (2.5)	25.4 (1.8)	23.5 (1.6)
Sand %		61.1 (1)	69.3 (2.7)	59.8 (1.4)	68.5 (3.5)	64.8 (2.8)	68.3 (2.2)
Horizons		A	B	Ah	AB	Bw	Bg
Depth (m)		0 - 0.3	0.3 - 1.2	0 - 0.2	0.2 - 0.28	0.28 - 0.39	0.39 - 0.6
Colour	Matris	10TR3/1	2.5Y6/2	Dull brown	Dull dark yellowish brown	Dull lighter yellowish brown	Dull dark creamy grey and yellowish brown
	Mottles	-	10YR5/6	-	-	-	-
Texture		Silty loam	Silty loam	Silt loam	Silt loam	Silt loam	Silt loam
Consistence		Friable	Friable	Friable	Loose	Loose	
Structure		Nutty	Nutty	Crumb	Fair Crumb		

Soil columns

Collection

Twenty intact soil column samples were collected from the Duvauchelle, Pawson silt loam, test plot (43º45’08.4” S, 172.º56’35.8” E) on the 28th of January 2016. The soil columns were 250 mm long with a diameter of 190 mm. Two parallel trenches were dug at the site. There was a 200 mm space between the two trenches. The trenches were 120 mm deep. The lysimeters were cut into the soil using an edger by shaving off soil outside of the lysimeter as the lysimeter was pressed into the soil, (Figure 17). The cores were cut and pressed 100 mm into the ground. The mini-lysimeters had a cutting ring, which made a 5 mm gap between the soil column and the edge of the lysimeter.

Figure 16. Field trial location at Duvauchelle, where soil columns were taken on 28th January 2016. Photo shows trial plots with native vegetation on 28th January 2016.

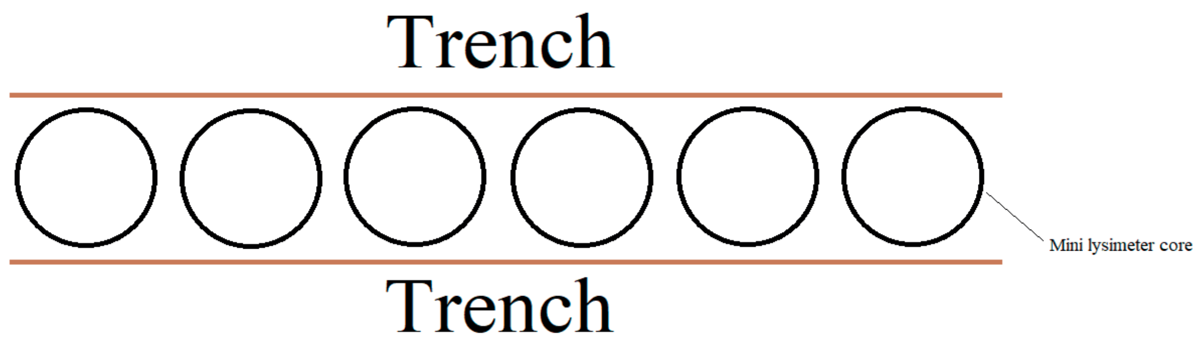

Figure 17. Diagram of the soil column collection method. Lysimeter cores were place on the ground in single file with a trench dug on either side. The cores are then dug into the ground in the same way full sized lysimeters are, (Figure 18).

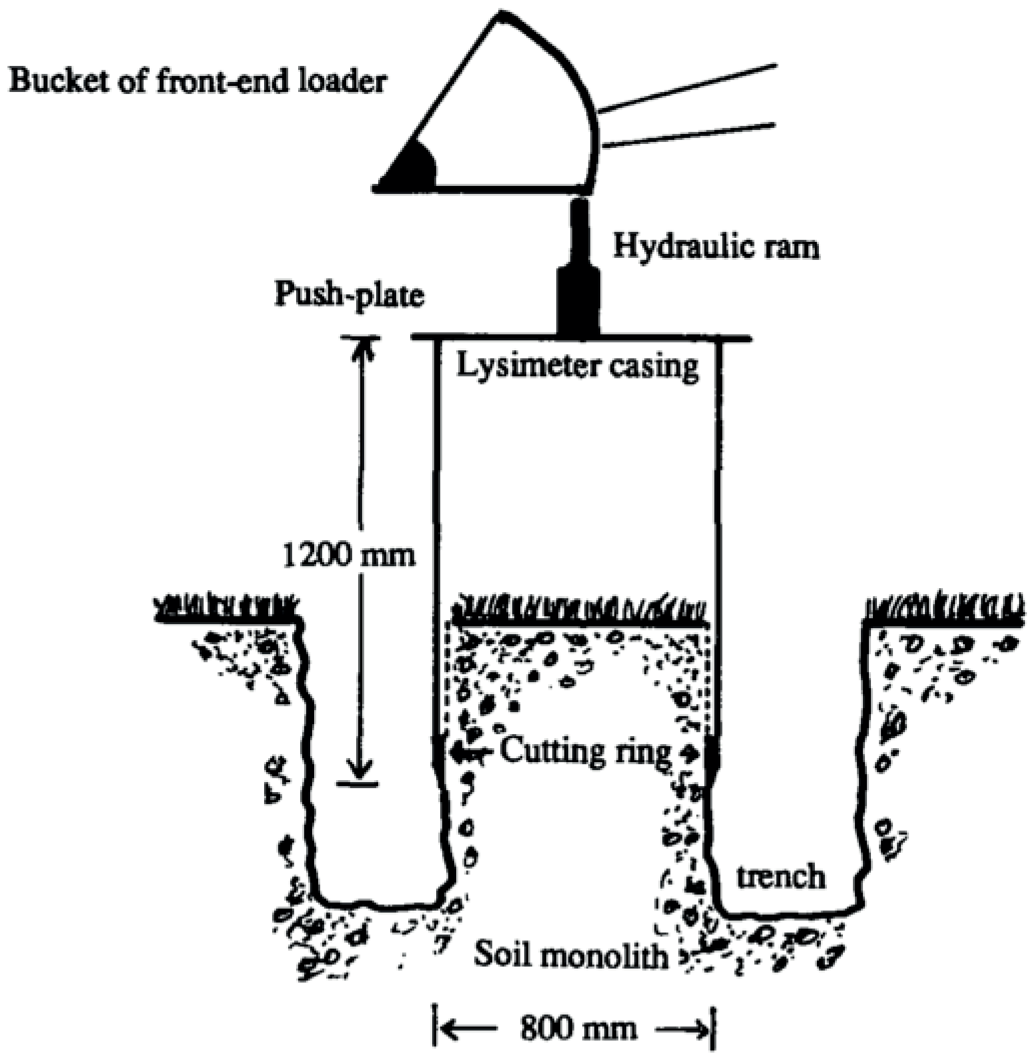

Figure 18. Diagram showing a lysimeter sampling method with a cutting ring (Cameron et al., 1992).

Set up

Liquefied petroleum jelly was injected into the gap between the soil column and the lysimeter to eliminate the edge-flow effect (Cameron et al., 1992). Columns were set up in an array with funnels and containers that allowed the leachates from the columns to be collected for analysis (Figure 19). Gauze was placed under each of the lysimeters and held in place with rubber bands, (Figure 20). A solution of Westminster Weed Killer G360 (40 mL, 1% glyphosate) was applied to each of the columns to remove all vegetation from the cover of the columns. The laboratory was located in the Burns building on the Lincoln University campus (43º38’34.2” S, 172º28’11.5” E).

Figure 19. Photo of soil column set up in lab. Soil columns are set up above drainage collection buckets.

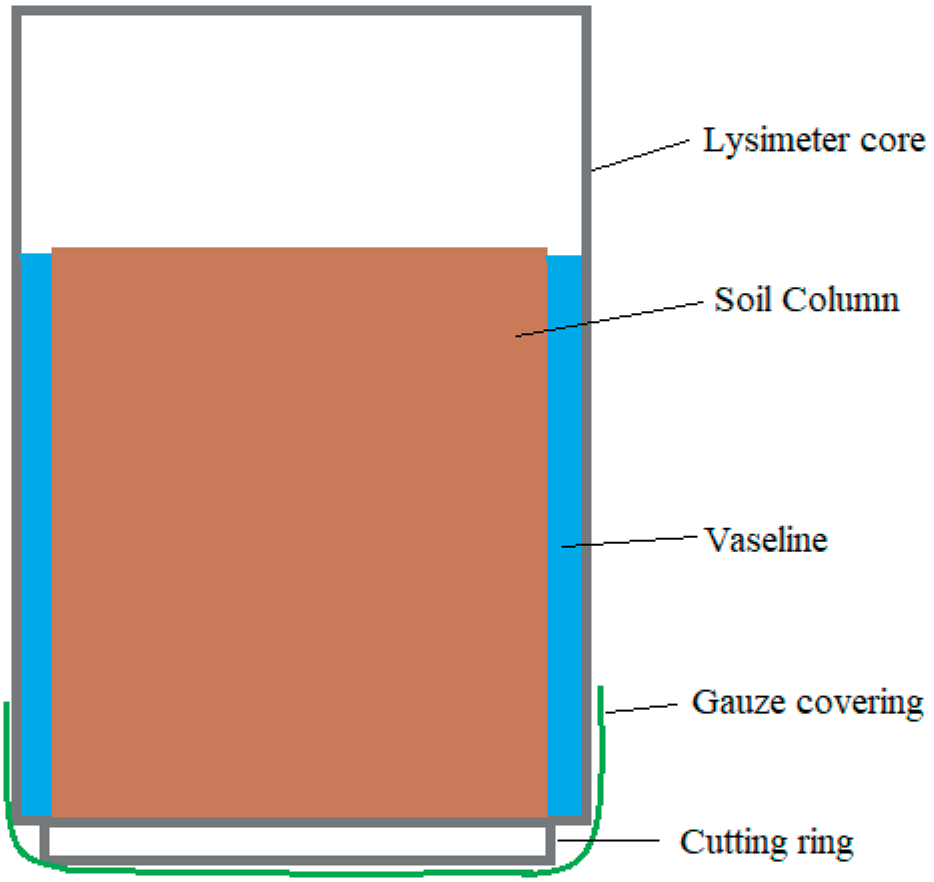

Figure 20. Cross section diagram of the soil columns set up. Space above the soil column allows for irrigation and prevents loss of TMW from overflow. The gauze covering prevents the loss of soil from the column.

Hydraulic conductivity

Before irrigation began, an initial hydraulic conductivity at soil saturation (K_{sat}) was determined for each of the soil cores. The soil columns were left in water for three days to reach a fully saturated state. The soil columns were then removed from the water tubs and placed over the funnels and collection buckets as described in the set up above, (Figure 19). The soil columns were allowed to reach a steady state of drainage, with a constant head height of 50 mm before drainage samples were collected over 120 seconds in triplicate, (Figure 21). The saturated hydraulic conductivity was redone at the end of the irrigation period before the lysimeters were deconstructed using the same method as the initial hydraulic conductivity measurement.

Figure 21. Left: Soil column under a constant head of 50 mm of water, Right: Steady state of drainage being collected from the soil column.

Irrigation

The trial consisted of three treatments and a control, each with five core samples. The control was water with a Na concentration of 8 mg/L, SAR value of 2.0. Treatment 1 was irrigated with TMW from the Duvauchelle sewage treatment plant, containing a Na concentration of 40 mg/L, SAR value of 10. Treatment 2 was irrigated with TMW from the Duvauchelle sewage treatment plant, with a concentration of 260 mg/L of Na, SAR value of 86. Treatment 3 was irrigated with TMW from the Duvauchelle sewage treatment plant, with a concentration of 325 mg/L of Na, SAR value of 170. The soil columns each received 20 mm/wk (1040 mm/yr) of their respective treatment irrigation for the first five weeks. The irrigation rate was then increased to 80 mm/wk (4160 mm/yr) from week 6 to week 19. The irrigation rate was then increased again to 140 mm/wk (7280 mm/yr) for week 20 to week 22, when the experiment was concluded. Each sample received the same volume of their respective treatment irrigation as the rest of the samples, four days a week. On the first day of the week, all soil columns received an application of control water, at the same volume as the other days of the week. This was because saline soils need an application of fresh water to cause soil dispersion, and a reduction in infiltration (Diamantis and Voudrias, 2002). The control water irrigation on all soil columns was to simulate rainfall, which often causes soil aggregate dispersion on saline soils. The experiment consisted for a total of twenty two weeks.

Leachates

Leachates were collected on Mondays before the first irrigation. This gave adequate time for the irrigation to leach into the collection buckets. The leachate volumes were measured and a 30 mL sample was frozen for analysis. The samples were diluted once by a factor of 10 with deionized water, and again with a potassium chloride solution (77.06 mmol/L). The samples were then analysed for Na using atomic absorption flame emission spectroscopy, using a Shimadzu AA-670.

Infiltration

The infiltration rate and soil sorptivity of the columns was periodically measured. These dates were the 24th of May, the 4th of July, the 26th of September and the 23rd of November, 2016. Two disk permeameters measuring soil pores with a radius of 0.30 cm and smaller, and a radius of 0.07 cm and smaller, and a radius of 15.5 mm were used. The maximum pore radius that was absorbing water during the infiltration period for the different head pressures was calculated using (Equation 2) (Nimmo, 2004). The head height from the permeameter was measured at time intervals and recorded. The 0.07 cm pore radius infiltration head-heights were recorded once every two minutes until a constant rate was reached. The 0.30 cm pore radius infiltration head-heights were recorded once every 30 seconds until a constant rate was reached. A table was set up to determine the soil sorptivity (C_2). The infiltration of the columns was determined for each time interval using Equation 3. Where I is the infiltration (cm), V is the volume of water that has infiltrated into the soil column, and r is the radius of the disk perimeter (cm). This real world scenario was compared to the model scenario shown by Equation 4. Where I is the infiltration (cm), C_1 is a constant related to the soil hydraulic conductivity (cm/s), t is the time of infiltration (s), and C_2 is a constant related to the soil sorptivity ($cms^{-1/2}$). Nonlinear curve fitting was used to determine the values of C_1 and C_2 by least square difference.

$$r = \frac{-2T\cos\theta}{\rho g Z}$$

Equation 2. Calculation for the soil pore radius

Where r is the radius of the soil pores (cm),

T is the surface tension of water (72.8 dyn/cm),

θ is the contact angle of the infiltrometer,

ρ is the density of water (0.998 g/cm^3),

g is the force of gravity (981 cm/s^2),

Z is the head pressure (cm).

$$I = \frac{V}{(\pi r^2)}$$

Equation 3. Infiltration rate as a function of volume of water and surface area

$$I = C_1 t + C_2 \sqrt{t}$$

Equation 4. Infiltration rate as a function of hydraulic conductivity and soil sorptivity.

Deconstruction

Bulk density

Core samples were collected from each of the columns for bulk density analysis. The corer had a length of 79 mm and a diameter of 27 mm. The sample's volume was measured, and the samples were dried at 30 °C until a constant weight was obtained. The bulk density (ρ_B) was calculated.

Chemical analysis

Soil samples were collected for the chemical analysis. Samples were passed through a 5 mm stainless steel sieve. The sieve was washed and dried between samples. The samples were dried using a soil drying oven. The samples were analysed using inductively coupled plasma optical emission spectrometry (ICP-OES). This was done to determine the soil concentration of the four major cations (Na, Mg, K and Ca).

Concentrations of Cd, B, Ca, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, P, S and Zn were determined using Inductively Coupled Plasma Optical Emission Spectrometry (ICP-OES Varian 720 ES - USA) in soils (Kovács et al., 2000) and in plants (Simmler et al., 2013; Valentinuzzi et al., 2015). Extraction and digestion solution and method blanks were analysed in triplicate as part of standard quality control procedure for the analysis and were as below the ICP-OES's detection limit for all metals. Recoverable concentrations of the CRMs were within 93% - 110% of the certified values.

Lysimeters

Lysimeter experiment

Two intact lysimeters were collected from the golf course at Duvauchelle on the 18th of September 2014. These lysimeters were taken to Lincoln University and irrigated with water (10 mm per day) until drainage stabilised in late October 2014. This demonstrated that the intact cores would drain and therefore be suitable for the full experiment. In November 2014, a further 10 lysimeters were taken from the golf course in Duvauchelle (43°44'53.06"S, 172°55'41.44"E) and six were taken from a paddock containing cattle (43°47'33.11"S, 172°57'16.96"E) between Takamatua and Akaroa, (Figure 15). Each lysimeter cylinder was placed on the soil surface, and gently tapped into the soil, while the soil surrounding the cylinder was excavated, (Figure 22). Molten Vaseline petroleum jelly was poured around the edge of the intact soil core before removal to the Lincoln University lysimeter facility.

Figure 22. Photo showing the stages of lysimeter sample collection. (1) the lysimeter is placed on the sample soil. (2) the lysimeter is dug into the ground. (3) the lysimeter is ready to be romoved from the sample site.

The lysimeters, replete with intact soil cores, were installed at the Lincoln University lysimeter paddock (43°38'53.54"S, 172°28'7.69"E) in December 2014. The original vegetation was left upon the lysimeters. The Duvauchelle lysimeters were covered with a fescue / browntop mixture, while the Takamatua lysimeters were dominated by perennial ryegrass. A decision was taken not to remove and re-sow the pasture because this would have resulted in significant topsoil disturbance and consequent flush of N through the soil profile.

Between December 2014 and April 22nd 2015, the lysimeters were irrigated with 2 L (10 mm) of water per day. The lysimeters started to drain in February, 2015 and by March, 2015, similar volumes of leachate were obtained for all lysimeters. On the 22nd of April, 2015, effluent application of the lysimeters began. TMW was collected by the Christchurch City Council and delivered to Lincoln University in a 1000 L tank. Samples of the stored effluent were taken weekly. The tank was refilled as needed. There were three replicates of five treatments:

1) Barry's soil. Control (no effluent application)
2) Barry's soil. TMW added at ca. 500 mm / yr (0.4 L/day, 5x per week)
3) Barry's soil. TMW added at ca. 1000 mm / yr (0.75 L/day, 5x per week)
4) Barry's soil. TMW added at ca. 2000 mm / yr (1.5 L/day, 5x per week)
5) Pawson silt loam. Control.
6) Pawson silt loam. TMW added at ca. 1000 mm/yr (0.75 L/day, 5x per week)

Drainage volumes were measured weekly or more often following high rainfall events. Pasture was harvested periodically, typically every three weeks, during the growing season. The photos below show PhD student, Minakshi Mishra measuring pasture growth and Dr Maria Jesus Gutierrez-Gines irrigating effluent and collecting drainage, (Figure 23).

Figure 23. Opposite: The installed lysimeters showing the six Pawson silt loam soil cores (front-left) and the 12 Barry's soil cores (rear-right). Top left: Effluent application. Top right: Drainage collection. Bottom: Destructive sampling of the lysimeters at the conclusion of the experiment. 16th of November, 2016.

Infiltration Measurement

After the lysimeter leachate sampling was concluded, the soil infiltration rate was taken for each of the lysimeters, on the 1st of November, 2016. The infiltration rate was taken using a CSIRO disk permeameter. A small section of the lysimeter was cleared of pasture (large enough to fit the disk permeameter) down to the soil using scissors. The soil was levelled, where need be, with a flat blade paint scrapper and any remaining pasture was removed down to soil surface level using the scissors to allow for good contact between the disk permeameter and the soil. The infiltration rate of the soil was taken at three different tensions, which measured soil pores with a maximum radius on 0.01 cm, 0.05 cm and 0.50 cm. The volume of water infiltrated into the soil at -10.0 cm pressure was taken once every two minutes until steady-state was reached (Figure 24). The volume of water infiltrated into the soil at -3.0 cm pressure was taken once every 1 minute for 20 minutes until steady-state was reached. The volume of water infiltrated into the soil at -0.3 cm pressure was taken once every ten seconds until the water in the disk permeameter had infiltrated into the soil (maximum of three minutes). The -10.0 cm infiltration was taken first, followed immediately by the -3.0 cm infiltration, then the -0.3 cm infiltration.

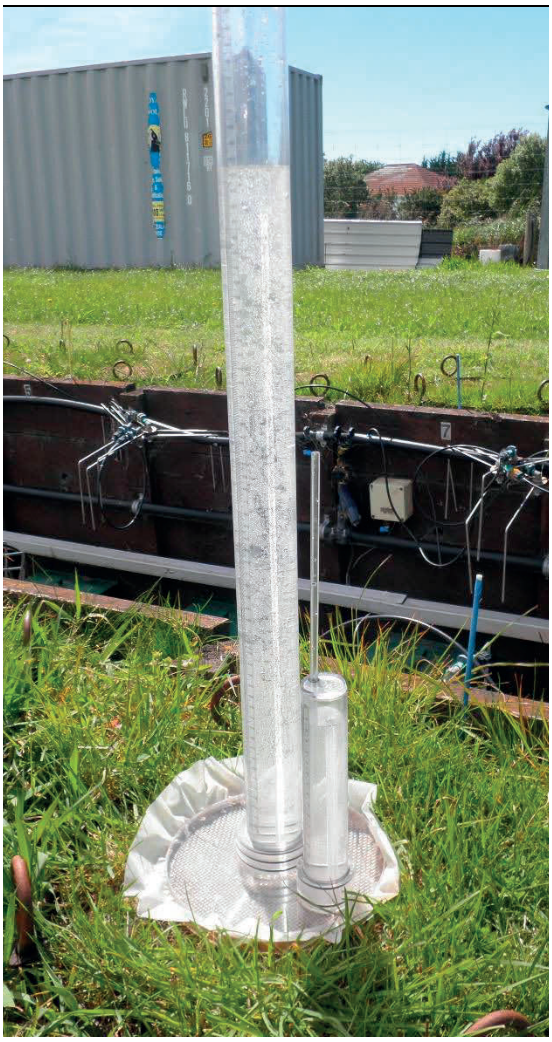

Figure 24. Opposite: Pasture cleared from the lysimeter surface in preparation for the disk permeameter, Opposite bottom: Infiltration rate data collection using a disk permeameter, Above: Water reserve and bubble tower set to -10.0 cm pressure.

Deconstruction

On the 16th of November, the lysimeters were deconstructed. Soil samples from 0-15 cm, 15-30 cm, 30-45 cm, and 45 – 60 cm depth were taken and stored for chemical analyses. At the same time, separate soil samples were collected for bulk density analysis. These samples were collected from 0-3 cm, 3-15 cm, 15-30 cm, 30-45 cm, and 45 – 60 cm depth. These samples were collected using soil corers. The bulk density and chemical analysis of the soil samples were analysed the same as the soil column samples.

Statistical analysis

Data were analysed using Minitab® 17 (Minitab Inc, State College, Pennsylvania, USA) and Microsoft Excel 2013. The ANOVA with Fisher's Least-Significance-Difference post-hoc test was used to assess the effects of different treatments. The significance level for all statistical analyses was $P<0.05$.

Results and Discussion

Duvauchelle TMW

Table 4 shows the properties of the effluent wastewater were analysed. This data was similar to the data the Christchurch City Council had provided over the past five years.

Soil columns

The drainage from the soil columns remained equal across the different treatments. The drainage also remained the same over time for each of the different treatments, (Figure 25). The data showed that 80% of the cumulative irrigation was recovered as drainage. The soil columns were of barren soil, which meant that 20% of the irrigation was lost to evaporation. The Na that was leached from the treatment soil columns all shared similar trends, (Figure 26). There was an initial low leaching rate that increased after 400 mm of irrigation onto the soil columns. The initial trend cumulative Na leached from the treatment soil columns was a simple curve relationship. The relationship between Na in drainage water and irrigation changed after 100 mm of irrigation to a straight-line relationship. The cumulative Na leached from the other treatment soil columns had similar relationships with the cumulative irrigation. There was an initial simple curve relationship prior to 400 mm of cumulative irrigation, followed by a straight-line relationship after 400 mm. This change in the relationship between the cumulative Na leached and the applied irrigation indicates that the treatments all reached equilibrium after approximately 400 mm of irrigation. The simple curve relationship indicated that the soil Na concentration was increasing to reach saturation. As the soil Na concentration increased, Na was being absorbed by the soil at a slower rate due to the soil becoming saturated. The straight line relationship that followed the simple curve relationship indicated that the soil had reached Na saturation. Because there was less Na being absorbed by the soil, more was being leached from the soil, which was shown in the data, (Figure 26).

Table 4. Chemical analysis of Treated Municipal Wastewater (TMW) from Duvauchelle treatment plant, and soils used in lysimeter and soil column experiments. Brackets represent standard deviation of the mean (n=54, except trace elements n=9).

	Treated Municipal Wastewater	Barry's soil (Duvauchelle)	Pawson Silt Loam (Takamatua peninsula)
pH	7.5	5.2	4.8
EC (uS/cm)	423 (40)	-	
Total suspended solids (g/m3)	32	-	-
NH_4^+-N (mg/L)	0.49 (0.15 – 0.80)	10.1 (7.5)	11 (6.8)
NO_3^- -N (mg/L)	18 (7.5)	17.1 (13.2)	4.4 (1.1)
NO_2^--N (mg/L)	0.86 (0.09)	-	-
Total C (%)	-	4.4 (0.6)	5.4 (0.3)
Total N (%)	<25	0.38 (0.05)	0.48 (0.03)
Al (mg/L)	0.43 (0.11 – 1.7)	32731 (1418)	34903 (3699)
B (mg/L)	0.10 (0.04)	-	
Ca (mg/L)	59 (12)	6770 (393)	5852(187)
Cd (mg/L)	<0.001	-	-
Cu (mg/L)	0.04 (0.03)	7.7 (0.2)	5.1 (1.4)
Fe (mg/L)	0.96 (0.25 – 3.6)	20155 (2852)	16806 (4098)
K (mg/L)	22 (5.0)	4491 (346)	4008 (365)
Mg (mg/L)	19 (5.5)	4251 (76)	3575 (463)
Mn (mg/L)	0.06 (0.03)	624 (9)	496 (50)
Na (mg/L)	95 (21)	290 (10)	374 (30)
P (mg/L)	11 (5.0)	1046 (30)	599 (125)
S (mg/L)	25 (11)	490 (21)	430 (5)
Zn (mg/L)	0.17 (0.11)	68 (3)	62 (7)
Sodium Accumulation Ratio (SAR)	15 (2.6)	-	-

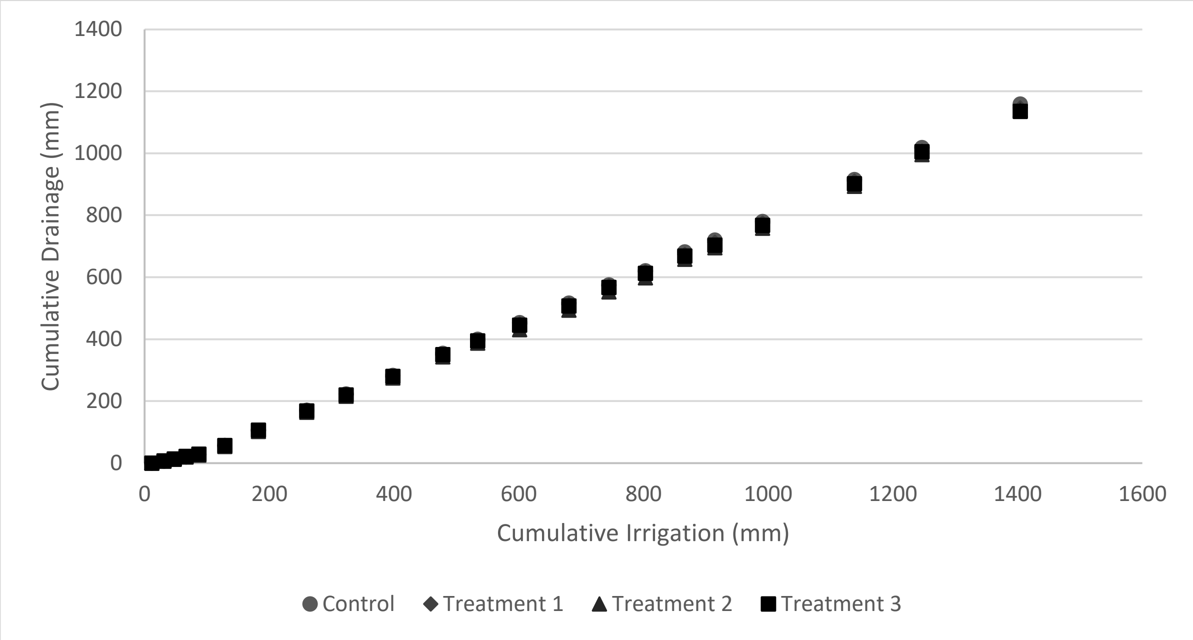

Figure 25. Cumulative drainage volume from soil columns against the cumulative irrigation the soil columns received.

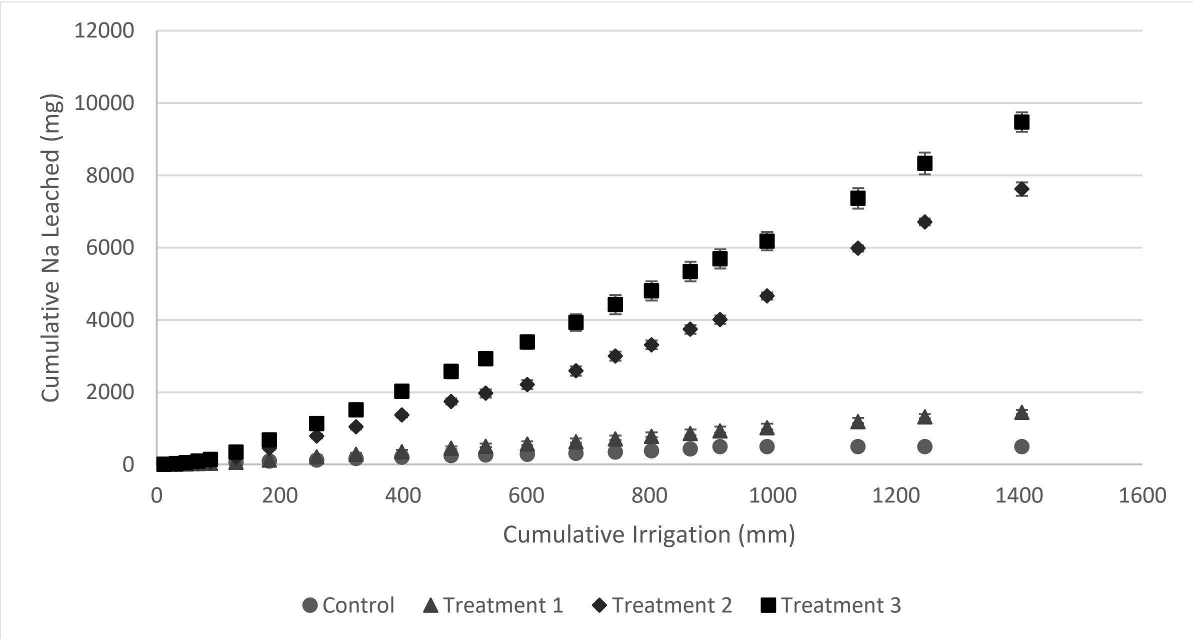

Figure 26. Cumulative Na Leached from soil columns as a function of Cumulative Irrigation.

Each of the soil column groups were significantly different for the final cumulative leached Na, [$p<0.001$]. The control soil columns leached a total of 488 mg of Na. The soil columns irrigated with the Treatment 1 40 mg Na TMW had a total cumulative leached Na mass of 1435 mg. The soil columns irrigated with the Treatment 2 260 mg Na TMW had a total cumulative leached Na mass of 7620 mg. The soil columns irrigated with the Treatment 3 325 mg Na TMW had a total cumulative leached Na mass of 9473 mg. These values were proportional to that of the Na concentrations in the different TMW treatments. The leached Na increased as the treatment irrigation increased.

The data showed that the major factor affecting the leachate concentration changed from the volume of applied irrigation, to the Na concentration of the TMW. The p-values and statistical groups for the first ten weeks are shown in the table below, Table 5. This table shows the point when the treatments became significantly different from each other with the accumulated Na leached (mg), at Week 10. Initially there was minimal Na leached from the soil columns for all the treatments suggesting Na was accumulating in the soil. As the soil columns became salt saturated, more salt was leached per week. This was when the data separated into different statistical groups. Treatment 3 soil columns became significantly different between when approximately 300 and 400 mg of Na had been applied. The soil columns irrigated with the treatment 2 TMW also became significantly different between when approximately 300 and 400 mg of Na had been applied. The soil columns from the Treatment 1 irrigation became significantly different when approximately 350 and 450 mg of Na had been applied. This gives an indication of where the soil had reached a stage of salt saturation. This caused the Na that was being applied to no longer be trapped by the soil, but stay mobile in the TMW and be lost due to leaching. The soil columns that had not reached the same stage of saturation were still absorbing most of the Na that was being applied, which caused the columns that were leaching to become significantly different. Figure 27 shows the mass of Na leached from the soil columns as a function of the mass of Na applied to the soil. After the initial application of Na which was adsorbed by the soil, the relationship of the mass of Na irrigated and leached was a 1:1 relationship. This means that the same mass of Na that was irrigated was also leached, and the Na was no longer being accumulated by the soil. The soil columns irrigated with the straight effluent accumulated 50 mg of Na in the soil. The treatment 2 soil columns accumulated 1480 mg of Na in the soil and the treatment 3 soil columns accumulated 1665 mg of Na in the soil. Figure 27 shows that the cumulative Na leached from the columns was consistent with the other treatments as a factor of irrigation.

Table 5. The statistical groups of the soil columns, from Na leached (mg) data, for the first ten weeks. Letters represent statistical groups for each treatment, and show when each treatment became significantly different. The point where each treatment began leaching Na indicates when soil Na saturation occurs.

	P-value	Control Group	Treatment 1 Group	Treatment 2 Group	Treatment 3 Group
Wk 1	0.54	2.5 (0.8)[a]	2.7 (1)[a]	3.7 (1.4)[a]	3.9 (1.5)[a]
Wk 2	0.016	8.8 (0.4)[a]	7.4 (1)[a]	11.5 (2.8)[a]	24 (6.7)[b]
Wk 3	0.002	15.9 (2)[a]	13.3 (1.1)[a]	21.1 (4.6)[a]	44.2 (8.6)[b]
Wk 4	0.001	23.2 (2.6)[a]	26.8 (3.5)[a]	42.4 (9.7)[a]	87.8 (24.4)[b]
Wk 5	0	28.7 (1.7)[a]	35.3 (4.6)[ab]	70.2 (11.7)[b]	135.5 (29)[c]
Wk 6	0	72.3 (8.3)[a]	71 (9.4)[a]	189.6 (26.9)[b]	335.5 (39.9)[c]
Wk 7	0	119.2 (22.2)[a]	125.4 (25.2)[a]	460.6 (47.9)[b]	738.3 (74.9)[c]
Wk 8	0	149.7 (27.4)[a]	217.5 (39.2)[a]	807.2 (55.1)[b]	1195.6 (65.9)[c]
Wk 9	0	192.4 (31.4)[a]	288.3 (46.1)[a]	1045.3 (81.5)[b]	1629.7 (48.1)[c]
Wk 10	0	231.3 (34.4)[a]	359.7 (52.2)[b]	1375.6 (86.1)[c]	2206.7 (138.8)[d]

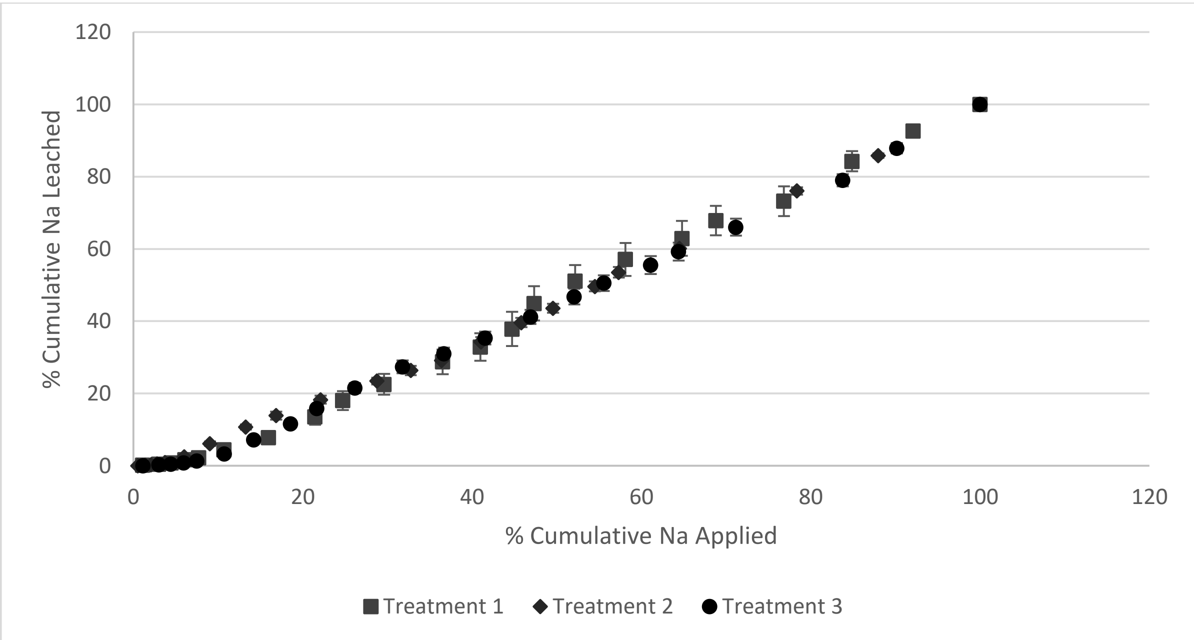

Figure 27. Cumulative Na Leached compared to Cumulative Na applied from soil columns as a percentage of total Na leached and applied.

Prior to 400 mm of irrigation, there was a lack of Na in the leachates. Sodium that was irrigated onto the soil in the TMW during this period Na accumulated in the soil. Once the soil Na saturation was reached, the leachate Na concentration increased (after 400 mm). The slope of the graphs in Figure 26 indicates that the soil would have an increased concentration of Na as the TMW Na concentration increased.

Soil chemistry

The concentrations of the common cations in the soil columns at the conclusion of the experiment are shown in (Figure 28). The soil chemistry data was consistent with the leachate concentration data. The leachate data indicated Na accumulated in the soil, which was reflected in the soil chemistry data. The leachate data also indicated that more Na would be in the higher treatments, which was confirmed by soil analysis (Figure 28). There was no significant difference in the concentrations in Ca [$p=0.174$], K [$p=0.213$] or Mg [$p=0.422$] between treatments. The soil Na concentrations were significantly different between each of the different treatments [$p<0.001$]. The greatest concentration of Na and the soil was from the treatment 4 (325 mg/L Na). This treatment had a soil Na concentration average of 1754 mg/kg of soil. The lowest concentration was from the control columns with an average of 331 mg/kg. The columns from treatment 1 (40 mg/L Na) had a soil concentration average of 427 mg/kg, and the columns from treatment 2 (260 mg/L Na) had an average Na concentration of 1098 mg/kg. The soil concentration of other elements are shown below, Table 7. There were no significant differences in concentration, between the different treatments. Of the four common cations, Na^{+}, K^{+}, Mg^{2+} and Ca^{2+}, only the soil Na concentration was significantly different between the different TMW treatments. The soil Na concentration of the soil columns increased as the Na concentration in the treatment water increased. This indicates that the soil Na saturation concentration is dependent on the concentration of the irrigation water and Na does not accumulate independently of the input concentration. The soil concentration of Ca and Mg did not change between the different treatments. This indicates that the irrigation of the soil did not cause significantly increased leaching of Ca or the Mg. The increase in the Na soil concentration is likely to cause ion displacement, which could cause a decrease in Ca or Mg soil concentration over time.

Table 6. Total mass of Na leached from the soil columns after irrigation. Brackets indicate standard error (n=3).

Treatment	Control	Treatment 1	Treatment 2	Treatment 3
Final Na Leached (mg)	486 (22)^{a}	1580 (70)^{b}	7530 (50)^{c}	10300 (140)^{d}

Table 7. Soil concentration for elements in the soil columns. Brackets represent standard error (n=5). There was no significant difference between the different treatments.

	Control	Treatment 1 (40 Na)	Treatment 2 (260 Na)	Treatment 3 (325 Na)
Al	30230 (170)	30620 (172)	30340 (419)	30540 (300)
Cr	17 (0.34)	18 (0.15)	17 (0.28)	18 (0.14)
Cu	8.3 (0.54)	8.5 (0.31)	9.0 (0.75)	9.0 (0.39)
Fe	19310 (720)	18760 (510)	18400 (550)	18950 (670)
Mn	495 (378)	506 (34)	520 (32)	462 (36)
Ni	6.6 (0.08)	6.6 (0.09)	6.8 (0.13)	7.3 (0.55)
P	797 (48)	818 (32)	817 (42)	888 (2.4)
Pb	14 (0.47)	14 (0.61)	14 (0.34)	14 (0.41)
S	399 (14)	408 (9.8)	407 (18)	414 (4.1)
Zn	65 (1.7)	66 (0.95)	66 (1.0)	68 (0.59)

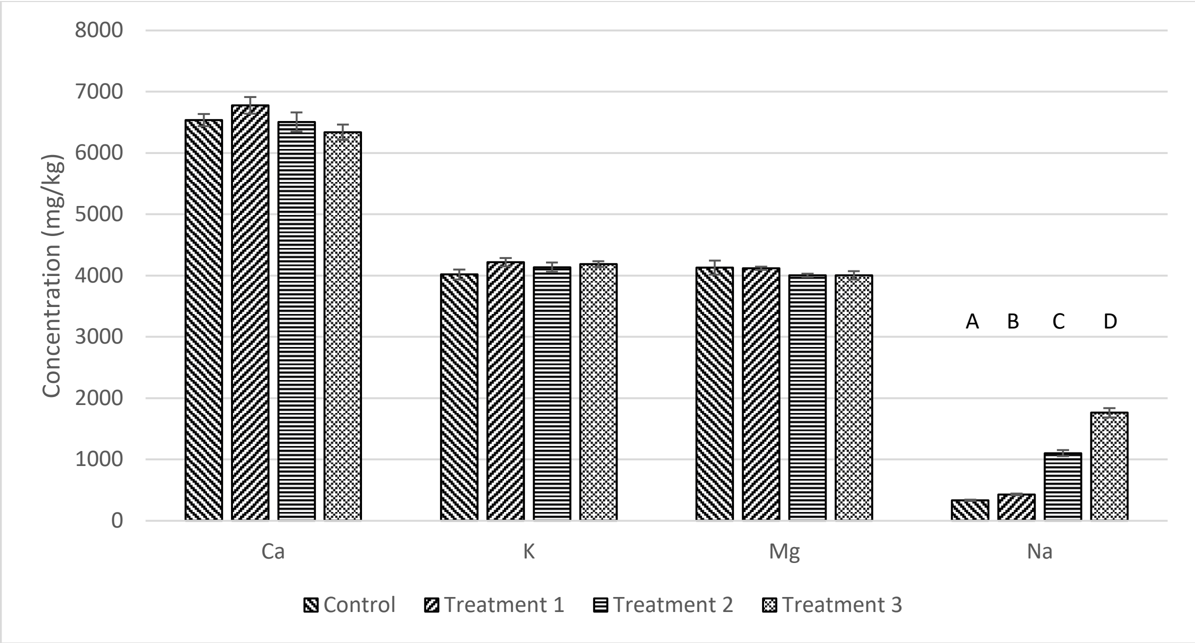

Figure 28. Concentration of the common cations in soil for the lab soil columns following 1400 mm irrigation of treatment solution. There was no significant difference between the different treatments for the concentrations of Ca, K or Mg. Letters represent statistical groups for Na concentrations.

A loss of divalent cations such as Ca^{2+} and Mg^{2+}, to be replaced by mono-charged cations such as Na^{+} can cause problems in the soil, as it can result in a lengthening in the Gouy-Chapman layer, which can lead to soil dispersion and a loss in infiltration. Because there was no significant leaching of the multi-charged cations, there is less risk of the soil aggregates becoming unstable and the soils from dispersing.

The soil concentrations of the Ca, Mg and Na were calculated into tonnes per hectare (ton/ha) in the top 10 cm of soil. There was 6.36 ton/ha Ca, 4.02 ton/ha Mg and 0.32 ton/ha Na in the soils. This Na was the naturally occurring Na in the soil. The final Na mass from both TMW accumulation and naturally occurring soil Na from the highest Na treatment was 1.71 ton/ha Na.

There was a significant difference between the pH of the soil columns. The control soil columns had the lowest pH, and the pH increased as the salt concentration in the TMW irrigation increased, (Figure 29). The higher pH of the higher treatment soil columns indicates that H^{+} has been leached from the soil. Other overseas studies found a decrease in infiltration rate due to irrigation of high SAR water. This was not observed in this trial. The soils that were examined in those overseas trials had naturally occurring Na, Ca and Mg concentrations that were of the same order of magnitude. This was not the case with this experiment, (Emdad et al., 2004; Suarez et al., 2006). The naturally occurring Na was lower than that of the naturally occurring Ca and Mg.

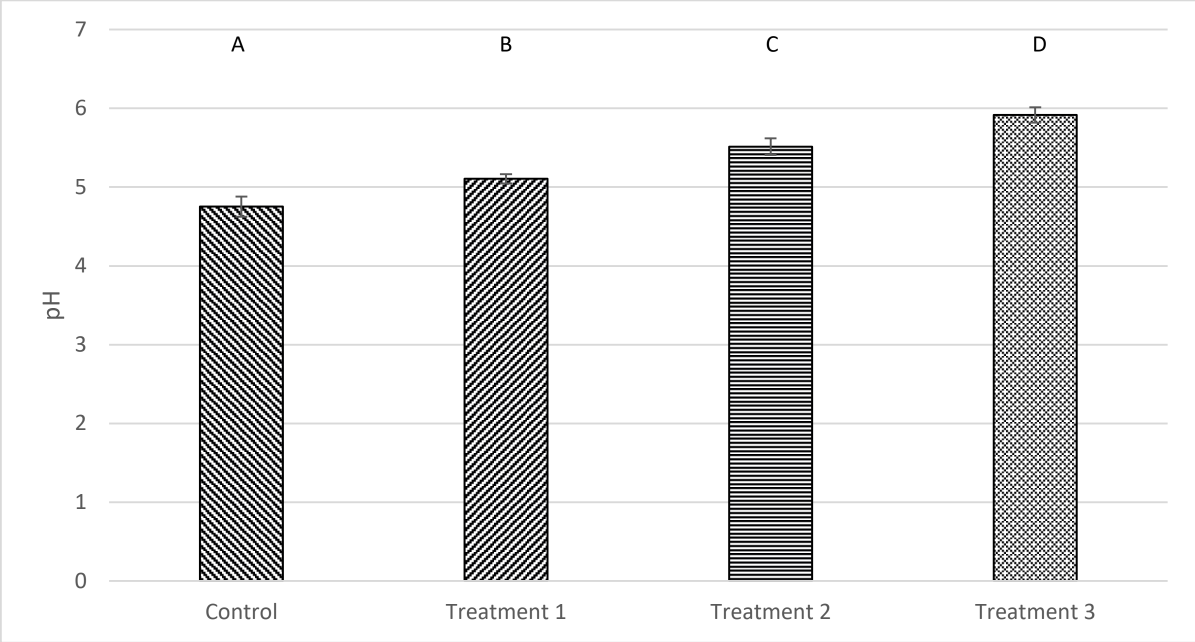

Figure 29. pH of the soil columns after irrigation with TMW. Letters represent significant differences.

Mass balance

The mass balance shows the change in the soil Na as a result of the experiment. The data shows that each of the different treatments resulted in a different soil Na, Table 8. The control water caused a loss of soil Na. Approximately 250 mg of Na was leached from each of the soil columns, which was a loss of 150 mg Na / kg soil per control soil column. The straight effluent (Treatment 1) resulted in no change in the soil Na concentration. The same mass of Na that was applied to the soil was also leached. The soil columns that were irrigated with the Treatment 2 and 3 TMW accumulated Na in the soil.

Table 8. Soil Na mass balance from soil columns after irrigation with 1400 mm of treatment solution. Table compares mass of Na applied to soil by irrigation, mass of Na leached from the soil columns, and Na in soil. Brackets represent standard error (n=5). Values with the same letter are not significantly different at the statistical level.

	Na applied (mg)	Na leached (mg)	Difference (mg)	Final soil mass (mg)	Final soil concentration (mg/kg)
Control	243	488 (41)^{a}	-245	1134 (82)^{e}	332 (12)^{e}
Treatment 1	1430	1435 (72)^{b}	-5.0	1464 (83)^{f}	428 (12)^{f}
Treatment 2	9467	7620 (187)^{c}	1846	3773 (287)^{g}	1102 (49)^{g}
Treatment 3	11465	9473 (273)^{d}	1992	5725 (256)^{h}	1760 (74)^{h}

SAR

The sodium absorption ratio unsurprisingly increased as the concentration of the treatment increased, (Figure 30). This result was consistent with the Ca, Mg and Na soil concentration across the soil columns, as the Ca and Mg soil concentrations were consistent across the different treatments, but the Na soil concentration increased as the treatment Na concentration increased. The SAR values in the soil columns were under a value of 18 for each of the treatments. This indicates that the soil aggregates are not expected to disperse, which means the soil infiltration rate was not expected to decrease. The soil columns irrigated with the treatment 3 TMW showed a SAR value close to that the 18, indicating that they are at the most risk of aggregate dispersion. The soil chemistry data showed that the SAR of the soil was not high enough to cause soil dispersion. This indicated that the soil density and infiltration may have no significant difference between the treatments.

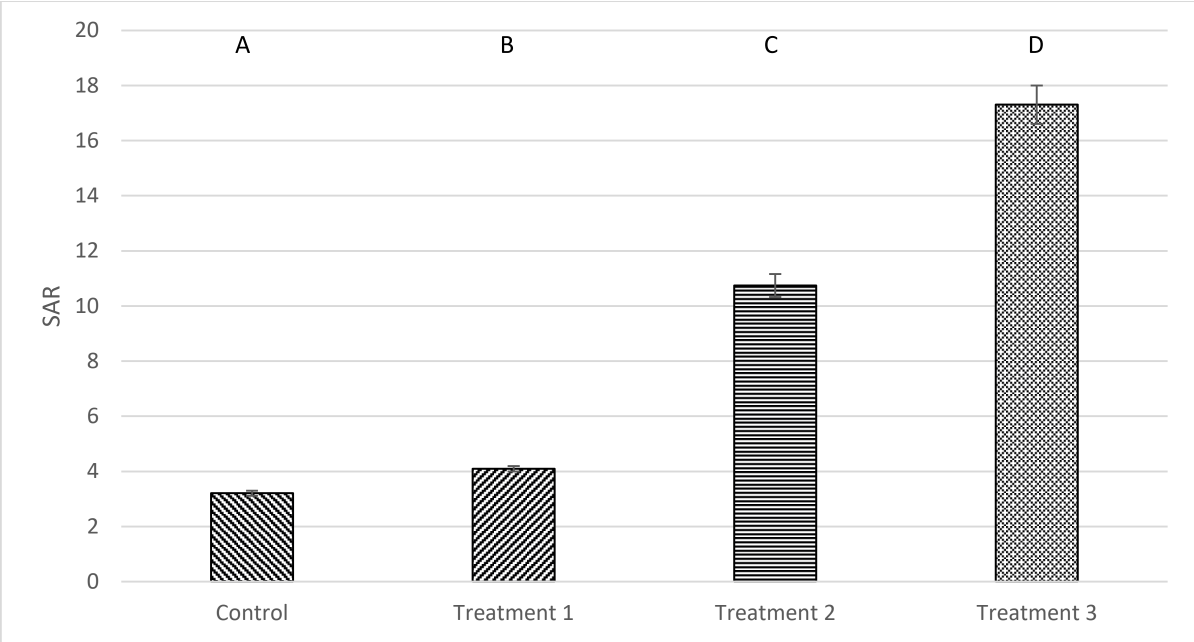

Figure 30. SAR values of soil after irrigation of 1400 mm of treatment solution. Letters represent the statistical groups.

Hydraulic conductivity

The initial and final hydraulic conductivity of the soil columns under soil saturation conditions are shown below, (Table 9). There was no significant difference in the data. The different treatments did not cause a difference in the hydraulic conductivity at complete soil saturation [p=0.715]. There was also no difference between the initial and final saturated hydraulic conductivity [p=0.069]. The K_{sat} values are at a negative pressure of 0 cm, which includes infiltration through all soil pores regardless of their radius. There was no significant difference between the saturated hydraulic conductivity values between either the initial and final tests, or between the different treatments. This indicates that the water flow rate was not affected by the treatments. If soil aggregates had dispersed, then the soil pores may become filled with soil particles and the water flow rate would decrease. There was no significant difference in the hydraulic conductivity. This indicated that the Na in the TMW had no effect on the soil aggregate stability. If the Na had affected the soil aggregate stability then the aggregate stability would have decreased, and soil aggregates would have been dispersed causing a decrease in the soil porosity. The infiltration rate and the hydraulic conductivity would therefore decrease, which would be visible in the K_{sat} data, as the final measurements for the treatment soil columns, particularly the higher treatments, would have lower values than that of the initial tests. There was no significant difference between the different tests or treatments, which shows that the total soil porosity did not change over the course of the experiment.

Table 9. Hydraulic conductivity of soil columns at soil saturation prior to irrigation with TMW, and before deconstruction of the soil columns. Brackets indicate standard error. Letters represent statistical groups.

Hydraulic Conductivity (cm/s)	Control	Treatment 1	Treatment 2	Treatment 3
Initial	0.0063 (0.0018)[a]	0.0037 (0.0011)[a]	0.0039 (0.0009)[a]	0.0033 (0.0014)[a]
Final	0.0025 (0.0015)[a]	0.0032 (0.0012)[a]	0.0019 (0.0011)[a]	0.0043 (0.0026)[a]

Infiltration and soil sorptivity

0.07 cm soil pore radius and less

The soil sorptivity of the smaller pores decreased over the course of the experiment. The soil sorptivity is a physical property of the soil, which describes the soil's ability to be able to absorb fluid. The higher the soil sorptivity, the greater the infiltration rate. There was a significant difference between the amount of irrigation applied (tests over time) [$p<0.001$]. The letters that accompany the data show the different statistical groups that each test time belongs to, (Figure 31). The different statistical groups showed that the soil columns continued to decrease over the course of the experiment as irrigation continued to be applied. There was no significant change in the soil sorptivity between the different treatments. This was consistent for each of the measurements done after different irrigation volumes [$p=0.412$]. This indicated that there was no difference due to the Na that was applied. The infiltration rates showed that there was no significant difference between the soil columns as a result from the different treatments of TMW. This indicates that breakdown in the soil pores with a radius of 0.07 mm or less was not because of the Na in the TMW. The infiltration rate of the soil columns was significantly lower after irrigation with a greater volume compared with earlier tests after irrigation with a lower volume. This indicates that the volume of water caused soil dispersion, which filled pore space with a radius of 0.7 mm or less. The soil columns had bare soil, which makes them more vulnerable to raindrop erosion. The irrigation was a fine spray, but the pressure of the irrigation caused some soil dispersion. After each irrigation event, there was ponding on the surface of the soil columns due to the volume of irrigation. This led to a small amount of overland flow each day, due to the daily application of irrigation. Overland flow causes transportation and soil erosion (Mahdi et al., 2017). The overland flow effect of the irrigation caused soil particles to fill pore spaces, which resulted in lower irrigation rates after increased irrigation volumes had been applied (Wang et al., 2013). Irrigation onto soils that have vegetation cover is expected to be less affected by this phenomenon than bare soil.

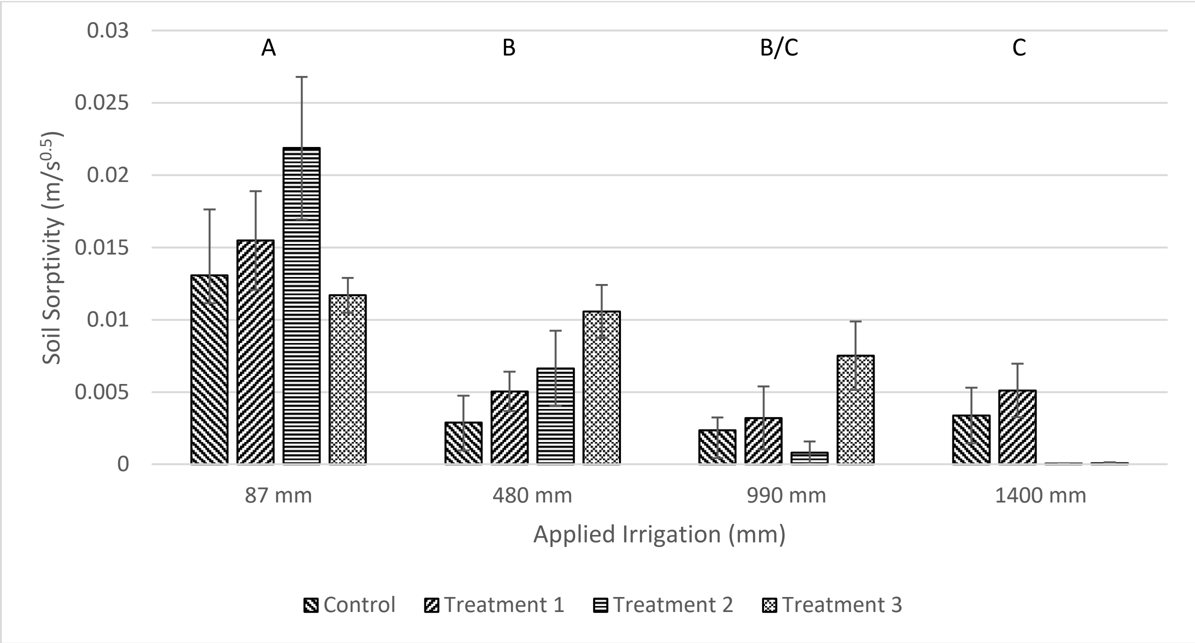

Figure 31. Soil sorptivity of lab soil columns under a -2.0 cm suction (soil pore radius of 0.7 mm and less) after different volumes of irrigation had been applied. Letters show which statistical group each irrigation volume group belongs to.

0.30 cm soil pore radius and less

As with the previous -2.0 cm pressure, there was no significant difference between the treatments [p=0.402]. There was also no significant difference between the different volumes of irrigation [p=0.742]. This indicates that the infiltration rate and the soil sorptivity of the soil columns did not change over the course of the experiment. This in turn indicates that the soil pores, with a radius of 3 mm and less, did not fill with material and become blocked. Large pores have a greater infiltration rate compared with smaller pores. There was no significant difference in the soil sorptivity for the test, which included the larger pores as well as the smaller ones. The irrigation, not the salt concentration caused the soil aggregates to break down. This breakdown was only minor, as only the smaller soil pores, radius 0.7 mm and less, rather than the 3 mm to 0.7 mm soil pore radius, were affected. Over time, the smaller soil pores will become completely sealed, and the larger pores will begin to fill with material as more soil aggregates disperse. The change in the soil sorptivity was due to the irrigation volume rather than the salt in the TMW. A vegetation cover will reduce the raindrop and erosion effects of the water. This in turn will stop the degradation of soil aggregates, stop the sealing of soil pores and stabilise the soil sorptivity and infiltration rate.

Other experiments

Other studies have found that an increase in Na concentration (SAR) in irrigation correlated with a decrease in soil infiltration rate on a Kobase silty clay loam from the Tongue River, north of Miles City Montana USA (Suarez et al., 2006). Irrigation with SAR values of 10-30 water at 238-261 mm has been shown to cause degradation of soil structure, such as aggregate instability and the formation of a surface crust in the soils of Tehran University farm in Karaj, Iran (Emdad et al., 2004).

Irrigation onto bare soil decreases the infiltration rate of the soil. This result was shown in the infiltration rate of the soil columns, (Figure 31), and also in other studies, (He et al., 2017). In other studies the water droplet impact effect caused soil dispersion, which caused a seal to form on the surface of the soil. This in turn caused the decrease in the soil's infiltration rate as shown in the study done by King and Bjorneberg (2012).

Bulk density

The bulk density of the lab soil columns at the conclusion of the experiment are shown below, (Table 10). There was no significant difference between the different treatments [p=0.229]. The bulk density of the soil is related to the total soil porosity. The more pores (essentially empty space) that are in the soil, the lower the bulk density. The bulk density measurements were consistent with the soil's infiltration ability, through soil pores, due to the application if TMW irrigation and the soil chemistry data. The soil SAR was too low to cause soil dispersion, and the soil pores were unchanged across the different treatments. There was no significant difference in the bulk density of the soil between the different treatments. This indicates that there was no significant change in the soil pores, as the volume of pore space was similar in each of the soil columns across the different treatments. The cations in the TMW did not significantly affect the porosity or the overall infiltration rate. The soil's density was unchanged across the different treatments, which indicated that soil material had not filled the pores, making the soil denser.

Table 10. Bulk Density (g/cm^3) of soil columns following the irrigation of 1400 mm of solution. There were no significant differences between the different treatments.

	Control	Treatment 1	Treatment 2	Treatment 3
Bulk Density (g/cm^3)	0.95	0.93	0.98	0.96
	0.99	0.98	0.93	0.88
	0.99	0.94	1.0	0.97
	1.0	1.0	0.96	0.93
	0.90	0.98	0.98	0.89
Average (g/cm^3)	0.97	0.97	0.97	0.93
Standard Error	0.02	0.02	0.01	0.02

Lysimeters

There was no ponding on any of the lysimeters after irrigation, or any other visible signs of soil degradation or surface sealing on the lysimeters. Drainage occurred throughout the experiment, which shows that the TMW was being absorbed by the soil and passing through the entire soil profile, (Figure 32 and Figure 33). The infiltration into the soil, and travel down the profile is evidence that there is no surface seal present, and the soil porosity has not changed. Soil porosity decreases, and surface seals form because of soil aggregate breakdown. Because there was no surface seal, and no decrease in soil porosity, there is evidence that the soil aggregates have not degraded.

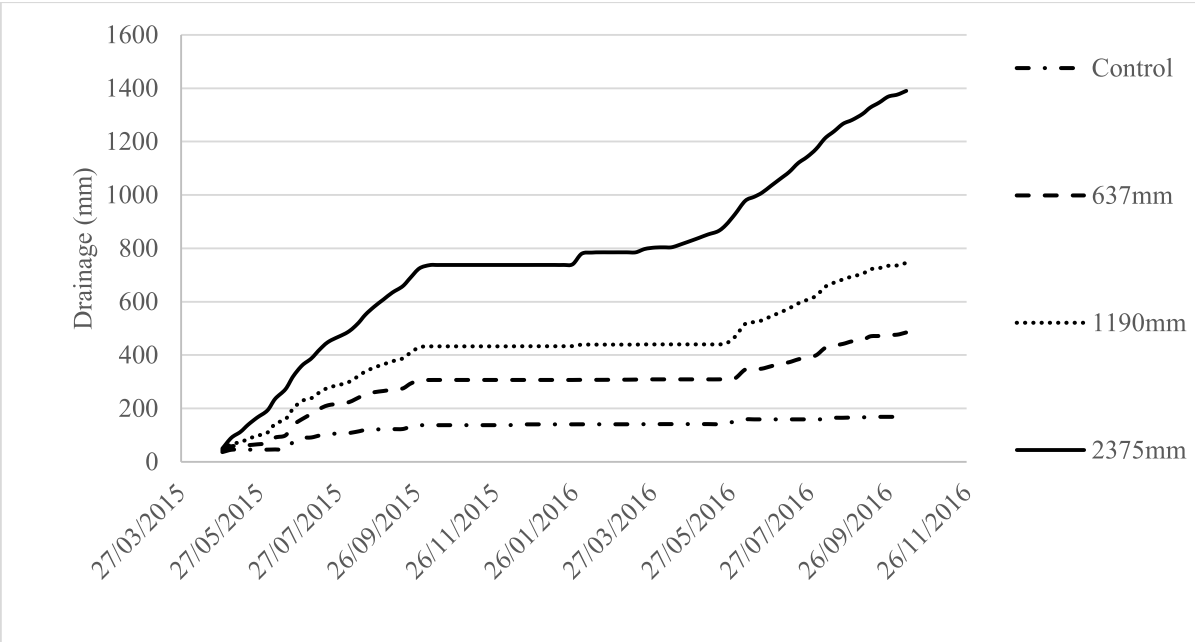

Figure 32. Cumulative drainage from the Duvauchelle lysimeters (Barry's soil) over time for the different treatments.

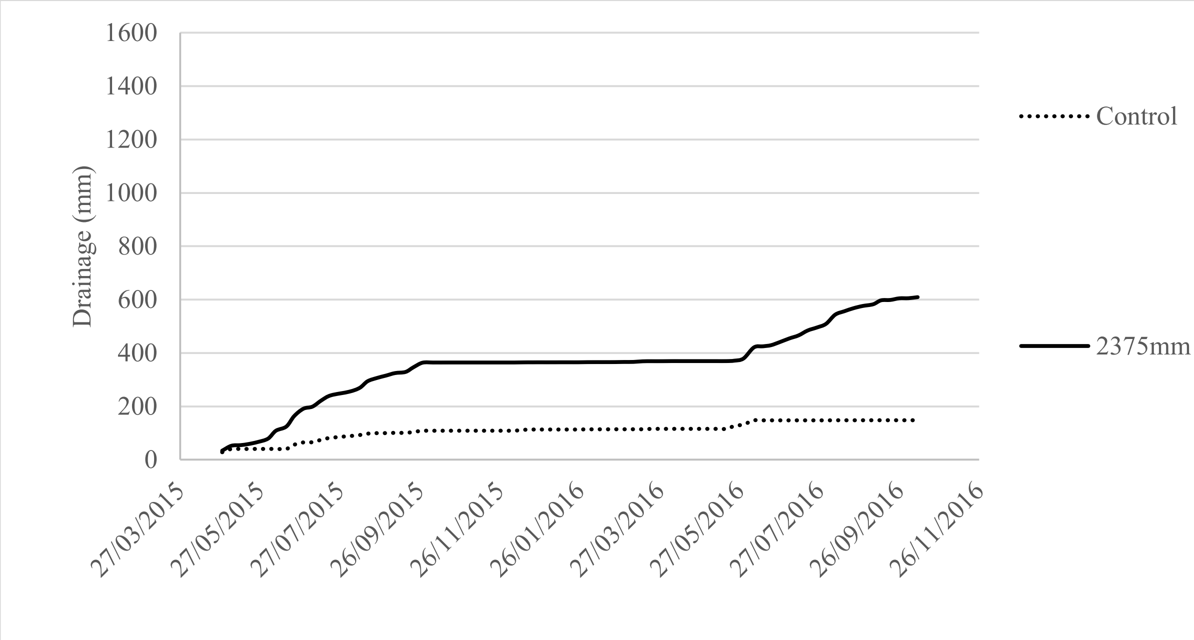

Figure 33. Cumulative drainage from the Akaroa lysimeters (Pawson silt loam) over time for the different treatments.

The treatments all had a larger drainage than that of the control lysimeters. This was unsurprising as the control lysimeters were not irrigated, and only exposed to rainfall. This drainage from the lysimeters increased as the volume of treatment increased. The Duvauchelle lysimeters had the most drainage over the winter months. This increase in drainage also occurred in the Akaroa lysimeters. There was little to no drainage from any of the lysimeters over the summer months. This indicates that there is little to no risk of contaminant, either pathogen or nutrient leaching over these months.

Soil chemistry

The leachate chemistry data was analysed along with the chemistry of the soil and the pasture. The results can be seen below, (Table 11).

Table 11. Mass of Na in the TMW, pasture, soil and leachates over the lysimeter experiment. Values in brackets represent the standard error (n=3). Values with the same letter in each soil type were not significantly different.

	Irrigation Na (kg/ha equiv.)	Average Pasture Na (mg/kg)	Pasture Na (kg/ha equiv.)	Na leached (kg/ha equiv.)	Soil Na (0 – 60 cm) (kg/ha equiv.)
Barry's soil					
Control	5	2243 (475)^{a}	10 (3)^{a}	45 (6)^{a}	2492 (76)^{a}
500 mm	605	2256 (241)^{a}	13 (3)^{a}	159 (18)^{b}	2840 (137)^{ab}
1000 mm	1131	2651 (159)^{a}	23 (3)^{ab}	264 (23)^{b}	2980 (106)^{b}
2000 mm	2256	3109 (308)^{a}	45 (6)^{b}	412 (61)^{b}	3113 (122)^{b}
Pawson silt loam					
Control	5	2525 (198)^{a}	13 (1)^{a}	30 (0)^{a}	2428 (181)^{a}
1000 mm	1131	4038 (273)^{b}	50 (2)^{b}	232 (32)^{b}	2610 (239)^{a}

Sodium

Both the treatments and depths were significantly different [$p<0.001$] for the Duvauchelle soils. The control lysimeters were in group A, the 500 mm lysimeters were in group B, the 1000 mm lysimeters were in group B and C, and the 2000 mm lysimeters were in group C. The soil concentration of Na increased as the treatment volume increased, as expected. The data showed that the soil concentration of Na was similar across the different depths for the Duvauchelle control lysimeters, but the treatment lysimeters had a greater concentration of Na at the surface of the lysimeter, (Figure 34). This Na was seen to be leaching down through the soil profile for the treatment lysimeters. Most of this leaching was expected to have happened during the winter months when leaching from the lysimeters occurred. This could mean that Na accumulated in the soil in the summer months is leached into groundwater in winter once the Na has been transported down the soil profile. Like the Duvauchelle lysimeters, the Akaroa lysimeters were in different statistical groups with respect to both the treatment and the soil depth [$p<0.001$]. The 0 to 15 cm depth of the treatment lysimeters were in group A, the 15 to 30 cm depth of the treatment lysimeters were in group B. The rest of the samples were all in group C. This indicated that the TMW irrigation caused Na accumulation in the soil. The Na did not leach all the way down the soil profile, as the Na concentration in the lower parts of the treatment lysimeters was of the same statistical group as the control lysimeters. Na can cause aggregate dispersion, and a decrease in infiltration. Significantly more Na was added to soil than was taken up by the pasture, (Table 11). Some of the Na was lost as leachate, while the rest accumulated in the soil. The mass of Na that accumulated in the soil can be seen above for both the Duvauchelle and Akaroa lysimeters, (Figure 34). The Na concentrations of the pasture from the Duvauchelle lysimeters were not significantly different. There was a significant difference between the Na pasture concentrations. This indicated that the soils from Akaroa were Na deficient for complete pasture Na saturation. This meant that the TMW irrigation added Na to the soil, which was taken up by pasture. The treatment lysimeter pasture accumulated significantly more Na from the soil than the control lysimeter pastures.

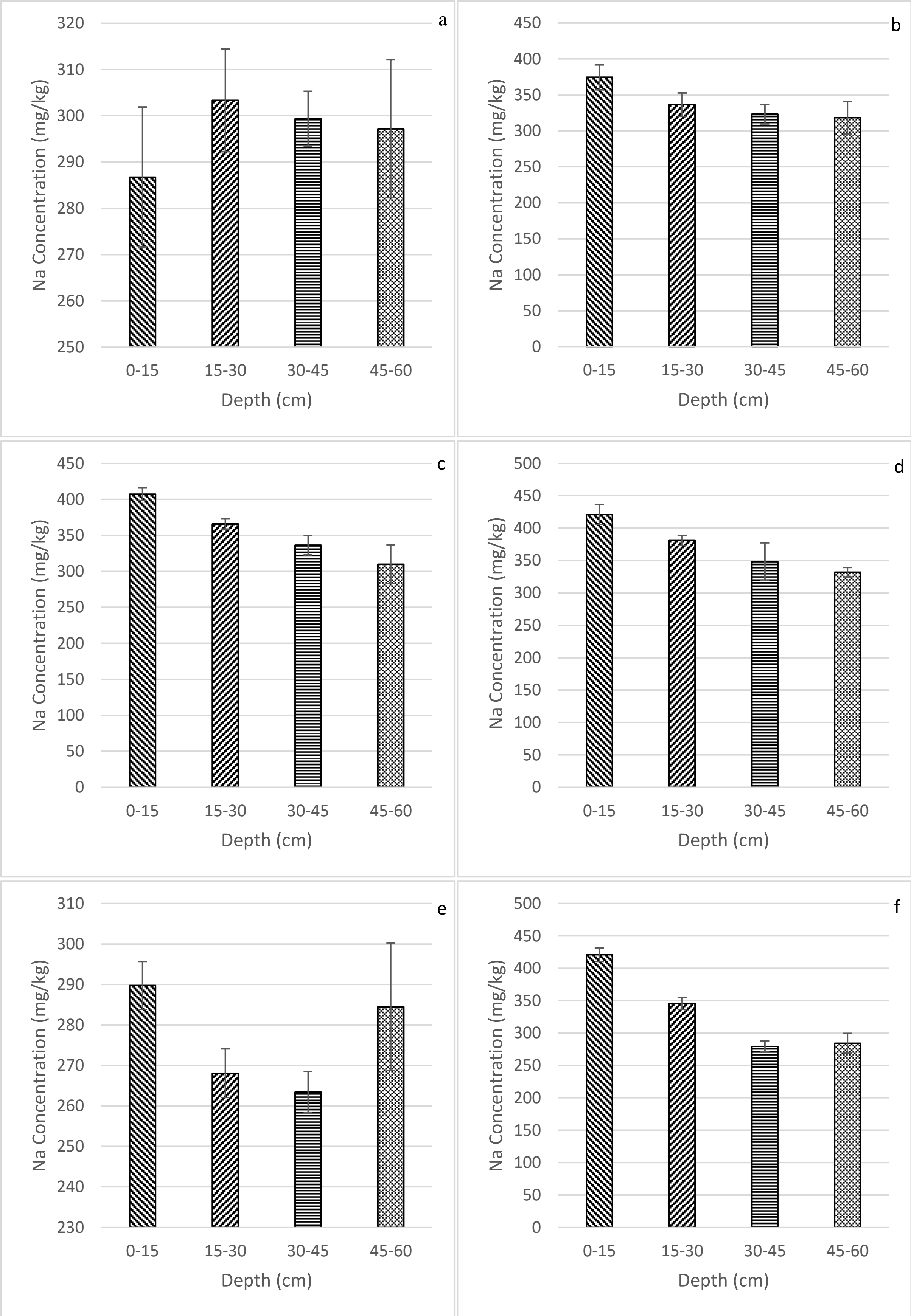

Figure 34. Soil Na concentration of the field lysimeters at different depths. Graphs represent different irrigation rates of TWM and different soil types.

Calcium

There was no significant difference between the different treatment lysimeters in terms of Ca soil concentration for the Duvauchelle lysimeters [p=0.671]. There was a difference between the different depths [p=0.011], (Table 12). This indicated that the treatment irrigation did not cause Ca to be leached from the soil. The soil samples between 0 and 45 cm depth were in one statistical group, while the samples at a depth of 45 to 60 cm were in a different statistical group. The data from the Akaroa lysimeters was similar to that of the Duvauchelle lysimeters (Table 13). There was no significant difference between the treatments [p=0.104], but there was a difference between the different depths [$p<0.001$]. The soil samples between 0 and 15 cm were of one statistical group, and the rest were in another statistical group. The lack of a significant difference between the treatments indicated that the irrigation was not causing the Ca to leach, and the Na in the TMW was not causing the soil's natural Ca to become mobile. Small amounts of Ca were added to the soil due to irrigation with the TMW, (Table 15). This Ca was not significant, however, as there was no significant difference between the different treatments for either the Duvauchelle or the Akaroa lysimeters, for soil Ca concentration. The Ca in the TMW was not significant as it was unable to significantly increase the soil concentration.

Table 12. Soil concentration of Ca in lysimeters after irrigation with TMW onto the Barry's soil. Brackets represent standard errors (n=3). There were no significant differences between the different treatments.

Depth (cm)	Control	Treatment 1	Treatment 2	Treatment 3
0-15	5555 (68)	5852 (108)	5847 (14)	5907 (185)
15-30	6038 (49)	5785 (83)	5852 (82)	5848 (102)
30-45	5935 (193)	5665 (152)	5966 (404)	5759 (209)
45-60	5496 (630)	4972 (373)	5565 (525)	5108 (246)

Table 13. Soil concentration of Ca in lysimeters after irrigation with TMW onto the Pawson silt loam. Brackets represent standard errors (n=3). There were no significant differences between the different treatments.

Depth (cm)	Control	Treatment 1
0-15	6770 (227)	7073 (87)
15-30	6062 (172)	6234 (81)
30-45	5712 (91)	5790 (91)
45-60	5844 (195)	6045 (206)

Table 14. Mass of Ca in the TMW, pasture, soil and leachates over the lysimeter experiment. Values in brackets represent the standard error. Values with the same letter in each soil type were not significantly different.

	Irrigation Ca (kg/ha equiv.)	Pasture Ca (mg/kg)	Pasture Ca (kg/ha equiv.)	Ca leached (kg/ha equiv.)	Soil Ca (0 – 60 cm) (kg/ha equiv.)
Barry's soil					
Control	3	3879 (527)^{a}	24 (5)^{a}	20 (5)^{a}	48351 (1620)^{a}
500 mm	371	3373 (216)^{a}	26 (4)^{a}	55 (13)^{a}	46775 (748)^{a}
1000 mm	696	3350 (69)^{a}	39 (3)^{ab}	61 (10)^{a}	47506 (1059)^{a}
2000 mm	1392	3327 (170)^{a}	51 (0)^{b}	92 (18)^{a}	48786 (1433)^{a}
Pawson silt loam					
Control	<1	5581 (396)^{a}	31 (2)^{a}	22 (6)^{a}	53218 (3475)^{a}
1000 mm	696	4890 (183)^{a}	68 (2)^{b}	92 (5)^{a}	49948 (4004)^{a}

Table 15. Soil concentration of Mg in lysimeters after irrigation with TMW onto the Pawson silt loam. Brackets represent standard errors (n=3). There were no significant differences between the different treatments.

Depth (cm)	Control	Treatment 1
0-15	4251 (44)	4314 (68)
15-30	4372 (24)	4461 (125)
30-45	4880 (149)	5070 (92)
45-60	5546 (178)	5976 (35)

Magnesium

The data trends for the Mg soil concentrations were similar to that of the Ca soil concentrations. There was no significant difference [p=0.785] between the different irrigation rates for the Duvauchelle lysimeters. This indicated that the irrigation water and the Na in the TMW did not cause Mg to leach. The lack of leaching in the soil Mg and Ca suggested that the soil structure would not be compromised, as Ca and Mg increase soil aggregate stability. The lysimeters from Akaroa had a significant difference between both the different treatments [p=0.02] and the different depths [p<0.001]. The statistical groups for the depths were 0 to 30 cm in group A, 30 to 45 cm in group B, and 45 to 60 cm in group C. The difference between the different treatments indicates that there has been some accumulation of Mg in the soil due to the irrigation. The lysimeters that were irrigated with the treatment TMW had a slightly higher Mg soil concentration than that of the control lysimeters. This Mg is beneficial as the TMW also caused an accumulation of Na, (Figure 34). The Na decreases the aggregate stability, and the Mg (and Ca) increases the aggregate stability of the soil. The data shows that the greatest concentration of Mg is at the bottom of the soil profile, (Table 16 and Table 15). The TMW is mainly causing a build-up of Na in the top 15 cm of soil, which makes Mg less important for aggregate stability, even though Mg at the soil surface is still several times higher than that of the Na. The greatest concentration of Ca is in the top 15 cm of the soil profile, (Table 12 and Table 13). The Ca concentration is also more than that of the Mg, and more than that of the Na soil concentration, making it more important for aggregate stability than the Mg.

Table 16. Soil concentration of Mg in lysimeters after irrigation with TMW onto the Barry's soil. Brackets represent standard errors (n=3). There were no significant differences between the different treatments.

Depth (cm)	Control	Treatment 1	Treatment 2	Treatment 3
0-15	3718 (140)	3575 (267)	3645 (152)	3472 (175)
15-30	3888 (149)	3581 (212)	3659 (157)	3651 (86)
30-45	4023 (41)	4092 (23)	3876 (133)	4042 (264)
45-60	4094 (34)	4267 (86)	4281 (122)	4107 (177)

There were small amounts of Mg present in the TMW, (Table 17). This Mg was not able to significantly increase the soil concentration in either the Duvauchelle or the Akaroa lysimeters. The Mg and the Ca were important as they can offset the negative effects of Na in the soil. The concentrations of these elements were so low that they were unable to prevent the increase of Na in the soil, which caused the SAR of the lysimeters that were irrigated with the treatment TMW to rise.

Table 17. Mass of Mg in the TMW, pasture, soil and leachates over the lysimeter experiment. Values in brackets represent the standard error. Values with the same letter in each soil type were not significantly different.

	Irrigation Mg (kg/ha equiv.)	Pasture Mg (mg/kg)	Pasture Mg (kg/ha equiv.)	Mg leached (kg/ha equiv.)	Soil Mg (0 – 60 cm) (kg/ha equiv.)
Barry's soil					
Control	<1	2065 (279)^{a}	13 (3)^{a}	6 (1)^{a}	33017^{a}
500 mm	124	1823 (110)^{a}	15 (2)^{a}	21 (7)^{a}	32580^{a}
1000 mm	232	1964 (52)^{a}	23 (1)^{ab}	23 (1)^{a}	32074^{a}
2000 mm	463	1960 (210)^{a}	33 (3)^{b}	50 (17)^{a}	32469^{a}
Pawson silt loam					
Control	<1	2481 (106)^{a}	16 (1)^{a}	5 (1)^{a}	42274 (2734)^{a}
1000 mm	463	2572 (78)^{a}	38 (2)^{b}	30 (2)^{a}	40351 (2596)^{a}

Potassium

There was no significant difference between either the different depths, or the different treatments for the Akaroa samples. This indicated that the TMW had had no effect on the soil. There was no significant difference between the different treatments from the Duvauchelle soils, which indicated that the TMW had no effect on the K concentration on either of the soils. The significant difference in the K concentration in the different depths for the Duvauchelle soils was a result of natural dispersion of K through the soil profile. As with P, more K was added with the TMW than was removed by the pasture, (Table 20). Most of this K accumulated in the soil, with only minor amounts leached. The accumulation of K in soil is insignificant as the soil concentrations were greater than the amount being added. At the highest TMW application rate the pasture took up significantly more K than the controls.

Table 18. Soil concentration of K in lysimeters after irrigation with TMW onto the Barry's soil. Brackets represent standard errors (n=3). There were no significant differences between the different treatments.

Depth (cm)	Control	Treatment 1	Treatment 2	Treatment 3
0-15	3711 (296)	4008 (211)	3899 (223)	4061 (63)
15-30	4248 (124)	4072 (196)	4136 (13)	4259 (101)
30-45	4353 (125)	4333 (14)	4369 (315)	4469 (26)
45-60	4163 (253)	4181 (168)	4340 (136)	4177 (248)

Table 19. Soil concentration of K in lysimeters after irrigation with TMW onto the Pawson silt loam. Brackets represent standard errors (n=3). There were no significant differences between the different treatments.

Depth (cm)	Control	Treatment 1
0-15	4491 (200)	4472 (123)
15-30	4558 (159)	4538 (174)
30-45	4657 (59)	4700 (96)
45-60	4887 (164)	4870 (182)

Table 20. Mass of K in the TMW, pasture, soil and leachates over the lysimeter experiment. Values in brackets represent the standard error. Values with the same letter in each soil type were not significantly different.

	Irrigation K (kg/ha equiv.)	Pasture K (mg/kg)	Pasture K (kg/ha equiv.)	K leached (kg/ha equiv.)	Soil K (0 – 60 cm) (kg/ha equiv.)
Barry's soil					
Control	1	11624 (263)^{ab}	65 (12)^{a}	1 (0)^{a}	34597 (493)^{a}
500 mm	177	8990 (723)^{c}	68 (4)^{a}	2 (0)^{a}	34848 (785)^{a}
1000 mm	331	10349 (510)^{bc}	112 (8)^{a}	3 (0)^{a}	35627 (908)^{a}
2000 mm	662	13060 (1150)^{a}	179 (6)^{b}	4 (1)^{a}	35165 (1134)^{a}
Pawson silt loam					
Control	1	17252 (1847)^{a}	104 (15)^{a}	6 (2)^{a}	40824 (1322)^{a}
1000 mm	331	17933 (518)^{a}	229 (16)^{b}	21 (6)^{a}	37392 (3319)^{a}

pH

There was no significant difference in the soil pH of the Barry's soil lysimeters from the Duvauchelle golf course after the irrigation of TMW [p=0.09], (Table 21). This indicated that the TMW did not cause a significant amount of H^+ to be leached from the soil. This in turn indicated that the additional Na from the TMW had not displaced H^+ from the soil particles. The Pawson silt loam soil did have a significant difference between the different treatments [p=0.027]. The lysimeters that received irrigation had a significantly higher pH than that of the control. This indicated that H^+ had been leached from the soil, which increased the pH of the treated lysimeter.

Table 21. Soil pH on lysimeters after irrigation with TMW onto Barry's soil and Pawson silt loam at different depths in the soil profile, and under different treatments. Brackets represent standard errors (n=3). Letters represent different statistical groups. There were no significant differences between the different treatments of the Barry's soil samples, but there was a significant difference between the different depths. There was a significant difference between both the depths and the treatments for the Pawson silt loam.

Treatment	Depth (cm)	pH
D 0	0-15[a]	5.00 (0.05)
	15-30[b]	5.56 (0.30)
	30-45[c]	6.14 (0.19)
	45-60[c]	6.27 (0.11)
D 0.4	0-15[a]	4.83 (0.04)
	15-30[b]	5.29 (0.14)
	30-45[c]	5.98 (0.27)
	45-60[c]	6.12 (0.17)
D 0.75	0-15[a]	4.91 (0.13)
	15-30[b]	5.42 (0.15)
	30-45[c]	6.09 (0.15)
	45-60[c]	6.08 (0.02)
D 1.5	0-15[a]	4.79 (0.07)
	15-30[b]	5.34 (0.08)
	30-45[c]	5.78 (0.06)
	45-60[c]	5.97 (0.12)
A 0[a]	0-15[a]	5.19 (0.08)
	15-30[ab]	5.34 (0.14)
	30-45[bc]	5.50 (0.15)
	45-60[b]	5.70 (0.20)
A 0.75[b]	0-15[a]	5.34 (0.11)
	15-30[ab]	5.57 (0.14)
	30-45[bc]	5.79 (0.04)
	45-60[b]	5.92 (0.09)

SAR

The SAR from the top 15 cm of the lysimeters showed that there was minimal risk of soil aggregate dispersion. This suggest that there would be no change to the soil's bulk density or the total infiltration rate. Since the soil chemistry and SAR suggests that there will be no dispersion of soil material, there will be no filling and sealing of soil pores. The chemistry data suggests that there will be no decrease in the infiltration rate, and no increasing of the soil density.

Infiltration

0.01 cm soil pore radius and less

There was no significant difference between the different irrigation rates for the Duvauchelle lysimeters [p=0.372]. This negative pressure resulted in only the smallest pores contributing to infiltration rate. The maximum soil pore radius for this negative pressure was 0.15 mm. There was a significant difference between the infiltration of the two irrigation rates from the Akaroa soil columns [p=0.014]. The lack of a significant difference between the different treatments and the control indicated that the soil aggregates from the Duvauchelle lysimeters did not degrade due to the different treatments. At this negative pressure, the soil pores that were infiltrating had a radius of 0.15 mm or less. These soil pores were not filled with dispersed soil material to a significant degree, as there was no significant difference in the infiltration rate. There was a significant difference between the infiltrations from the different Akaroa treatments. This indicated that the soils from Akaroa aggregates dispersed material that filled the soil pores with a radius of 0.15 mm or less. This caused the infiltration from the treatment lysimeters to have a lower infiltration rate than that of the control (Figure 35).

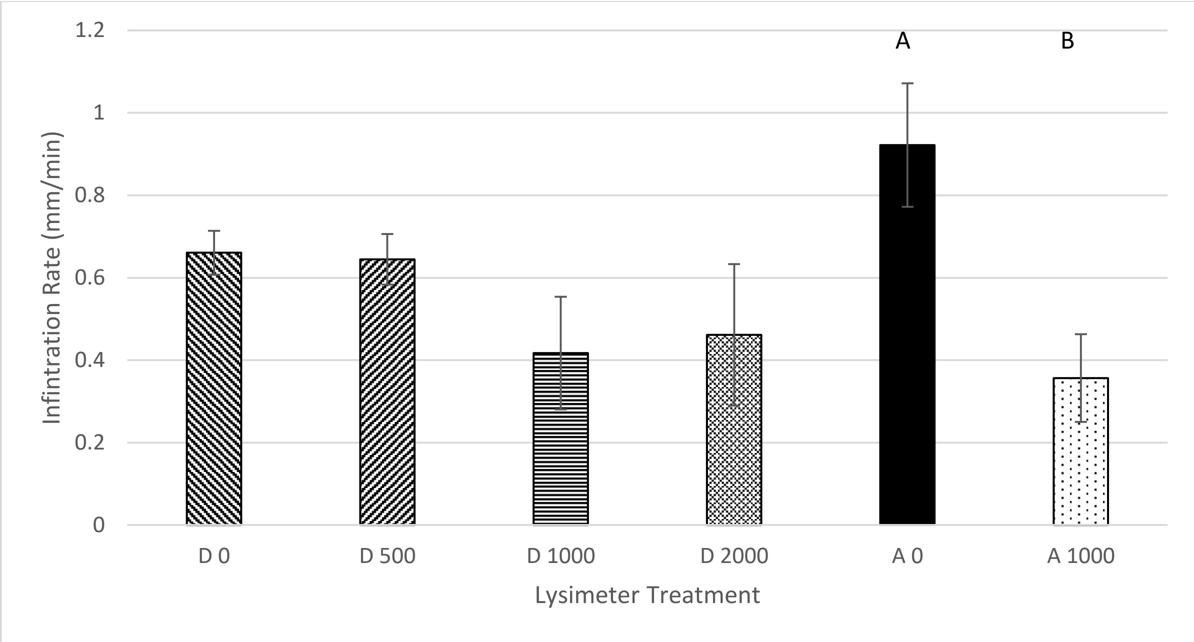

Figure 35. Infiltration rate of Duvauchelle (D) Barry's soil, and Akaroa (A) Pawson silt loam field lysimeters under 10.0 cm of negative pressure. Irrigation rates are shown, 0 (0 mm/yr), 500 (500 mm/yr), 1000 (1000 mm/yr) and 2000 (2000 mm/yr). There was no statistical difference between the different Barry's soil treatments. Letters show the statistical groups for the Pawson silt loam treatments.

0.05 cm soil pore radius and less

The soil pores that were able to infiltrate at a negative pressure of 3.0 cm were of a pore radius of 0.49 mm or less. There was no significant difference between the different irrigation rates of the Duvauchelle lysimeters [p=0.089]. Because there was no significant difference in the infiltration rates of the Duvauchelle lysimeters, the treatment irrigation had not caused the soil pores to become blocked. This indicated that the application of TMW onto this soil will not degrade the soil structure at this pore size. There was no significant difference between the irrigation rates from the Akaroa lysimeters [p=0.194]. As with the Duvauchelle lysimeters, the treatment TMW is unlikely to cause a breakdown in soil structure.

0.50 cm soil pore radius and less

There was no significant difference between the irrigation rates [p=0.788] from the Akaroa lysimeters. There was a significant difference between the irrigation rates [p=0.033] from the Duvauchelle lysimeters. At this negative pressure, all soil pores that had a radius of 4.95 mm or less were infiltrating. The treatment TMW had no effect on the infiltration rate of the Akaroa soils. However, the TMW affected the Duvauchelle soil. The control lysimeters were in one statistical group, while the rest of the Duvauchelle lysimeters were all in another statistical group. This indicated that the irrigation caused a change in the infiltration rate. The control lysimeters received no irrigation, only rainfall. This made the soil hydrophobic, while the treatment soils remained moist because of the constant irrigation (Lichner et al., 2010). The Duvauchelle treatment lysimeters had a greater infiltration rate than that of the control lysimeters, which meant that the treatment TMW increased the soil's infiltration rate, and made the soil pores accept water more readily, (Figure 36). The salt in the treatment water did not affect the infiltration, as soil aggregate dispersion would have filled the soil pores. This would have caused the infiltration rate from the treatment lysimeters to be lower than that of the control lysimeters. The smallest pores would have become sealed before the larger pores, so if there was soil structure degradation, there would have been a difference in the previous infiltration rates that was not present, (Figure 35 and Figure 37).

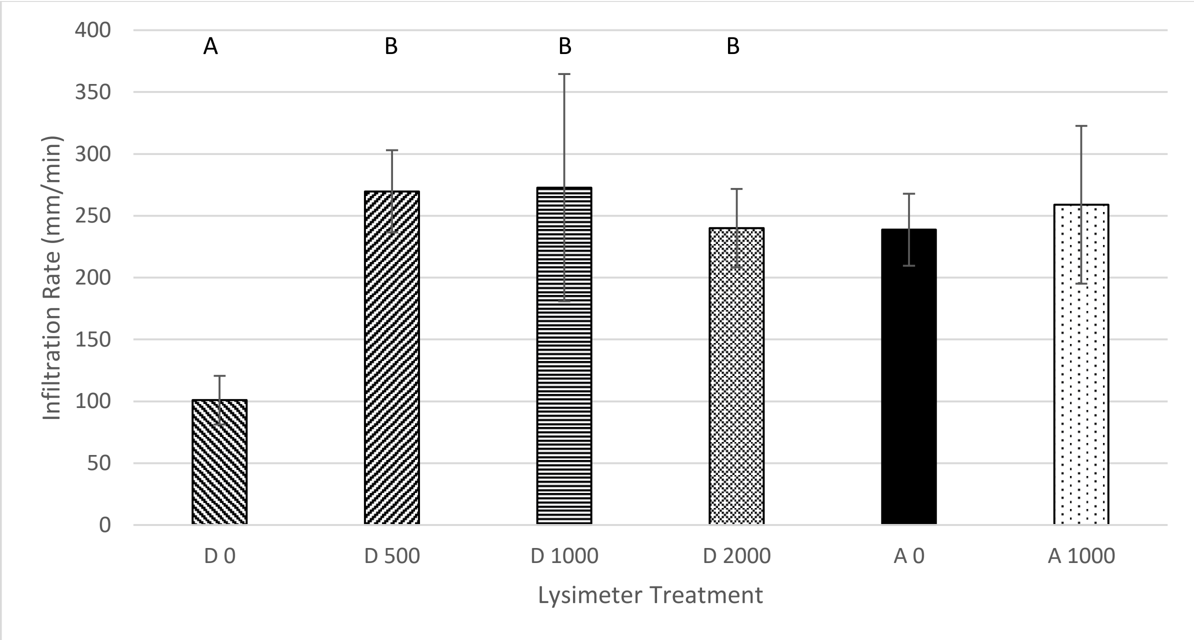

Figure 36. Infiltration rate of Duvauchelle (D) Barry's soil and Akaroa (A) Pawson silt loam field lysimeters under 0.3 cm of negative pressure. Irrigation rates are shown, 0 (0 mm/yr), 500 (500 mm/yr), 1000 (1000 mm/yr) and 2000 (2000 mm/yr). Letters show the statistical groups between the different treatments for the Barry's soil. There was no significant difference between the different treatments for the Pawson silt loam

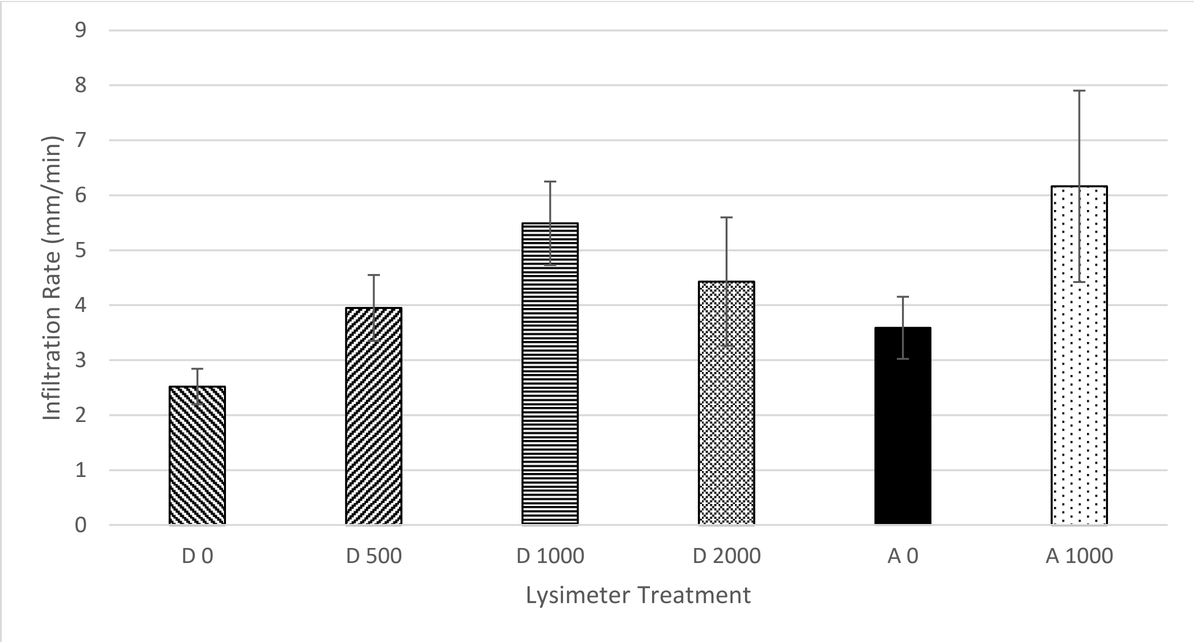

Figure 37. Infiltration rate of Duvauchelle (D) Barry's soil, and Akaroa (A) Pawson silt loam field lysimeters under 3.0 cm of negative pressure. Irrigation rates are shown, 0 (0 mm/yr), 500 (500 mm/yr), 1000 (1000 mm/yr) and 2000 (2000 mm/yr). There was no significant difference between the different treatments for the Barry's soil or the Pawson silt loam.

Other studies

As many other studies have shown, irrigation with TMW can reduce the infiltration rate of the soil (Suarez and Gonzalez-Rubio, 2017). This reduction was present in the lysimeters. As with the soil columns from the laboratory, the reduction of infiltration rate was only present in the data that excluded larger pores. These pores had a smaller pore size (0.15 mm radius and less) than what was measured for the soil column experiment (0.7 mm radius and less). The same reduction in the infiltration rate of the soil was not observed in this trial. The Pawson silt loam soils had no significant difference in the total infiltration rate. The reduction in infiltration rate that was observed when measuring the smallest pores, (Figure 35), did not affect the total infiltration rate, (Figure 36). There was also no significant decrease in the infiltration rate from the Barry's soils as a result of the TMW irrigation.

Bulk density

There was no significant difference between the different treatment groups in soil density [$p=0.424$]. There was, however, a significant difference between the different depths for all the lysimeters [$p<0.001$]. The 0 to 3 cm depths were of a different statistical group to the rest of the depths, which were all of the same statistical group. This difference could have been due to the vegetation and organic material found in the surface of the soil. There was no indication of the treatment lysimeters having a greater bulk density, which would have indicated that the soil porosity would have decreased. This was not evident, indicating that the treatment TMW did not disperse soil aggregates and seal soil pores. There was no significant difference between the different treatments from the Akaroa lysimeters [p=0.387], (Figure 38). This, like the Duvauchelle lysimeters, indicated that the treatment irrigation did not affect the soil in the lysimeter, (Figure 39). There was a significant difference in density between the different depth ranges of the soil [p<0.001]. This was just natural variation from the soil.

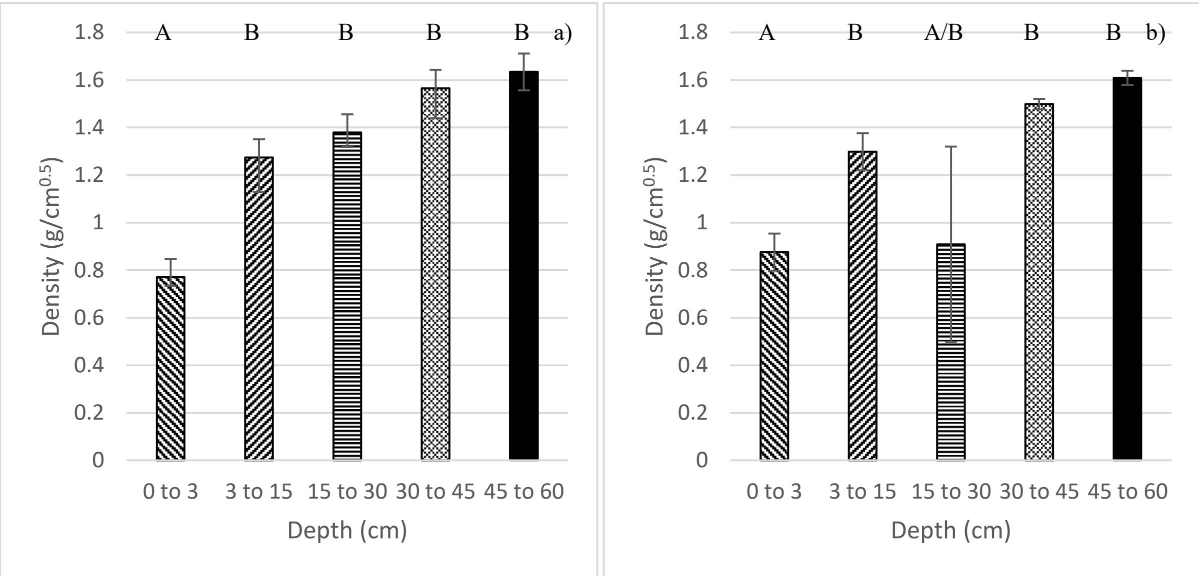

Figure 38. Bulk density of lysimeters under different treatments (a – control, b –1000 mm/yr TMW) for Pawson silt loam after irrigation with TMW. There was no significant difference between treatments. Capital letters represent statistical groups between different depths.

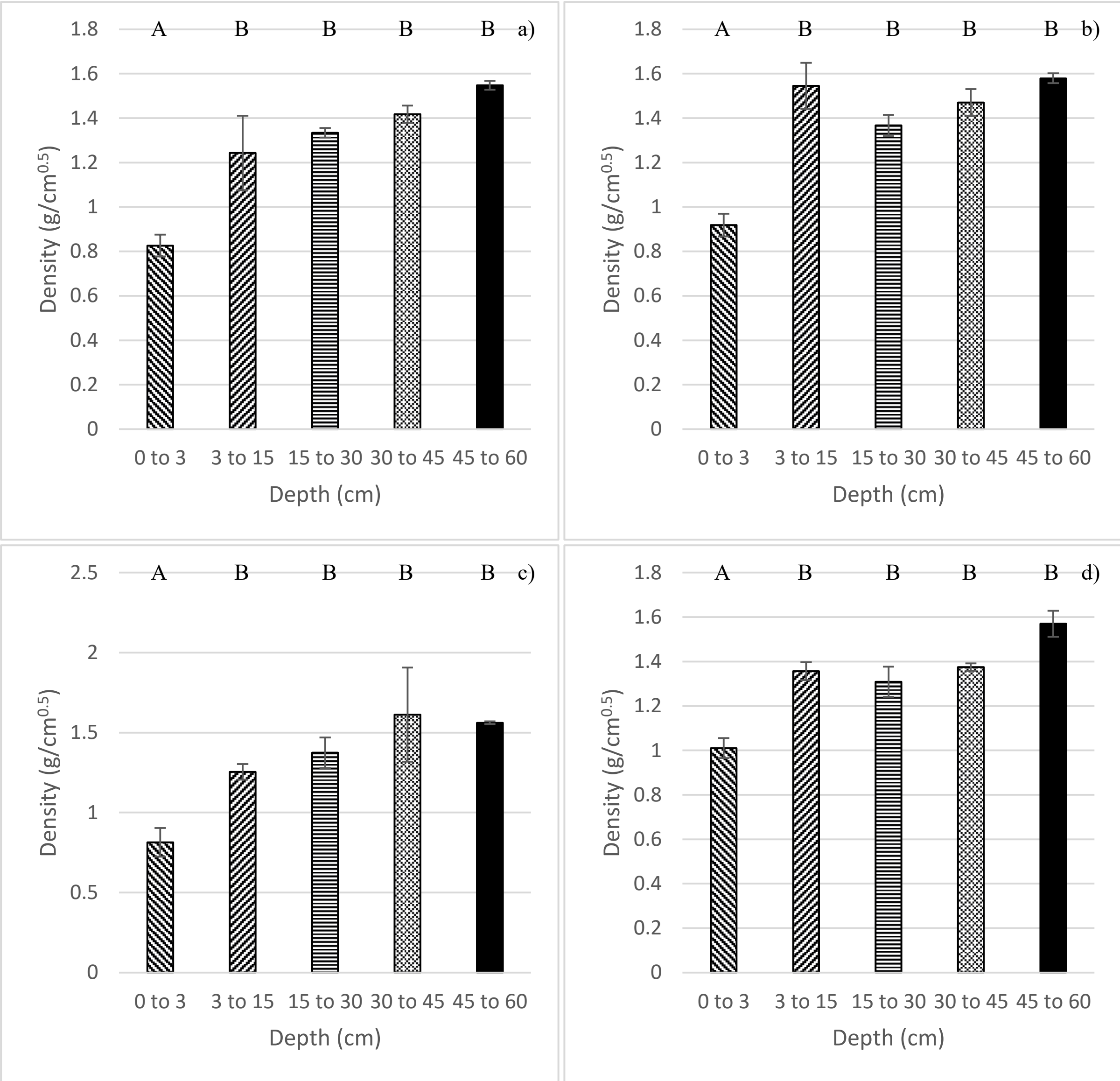

Figure 39. Bulk density of lysimeters under different treatments (a – control, b – 500 mm/yr TMW, c – 1000 mm/yr TMW, d – 2000 mm/yr TMW) for Barry's soil after irrigation with TMW. There was no significant difference between treatments. Capital letters represent statistical groups between different depths.

Summary and Conclusions

Irrigation and Drainage

The soil columns all received the same volume of irrigation, while the treatment columns received increasing concentrations of Na. The field lysimeters, however, received TMW with the same concentration of Na, but the treatment lysimeters received increasing volumes of irrigation. This resulted in different parameters being measured by the two experiments. The lysimeters tested the usefulness of the TMW as an irrigation resource. The growth of the pasture on the lysimeters showed that the TMW irrigation significantly improved the dry matter yield (Table 22). The drainage volumes also increased with the increase in irrigation onto the lysimeters. By comparison, the drainage from the soil columns had no significant difference between the different treatments. The soil was also bare, so dry matter yields were not measured. There was no evidence of ponding on either the lysimeters or the soil columns, which indicates that there was no surface crust forming on either the lysimeters or the soil columns.

Table 22. General parameters from the 21st of May 2015 until the 3rd of October 2016. Values in brackets represent the standard error of the mean (n=3).

Treatment	Total Irrigation (mm)	Total Rainfall (mm)	Total drainage (mm)	Total Evapotranspiration (mm)	Biomass production (t/ha equiv.)
Barry's soil					
Control	0	779	169 (22)^a	610	5.4 (1.0)^a
440 mm/yr	637	779	485 (23)^b	931	6.3 (0.6)^a
825 mm/yr	1190	779	736 (17)^c	1233	8.9 (0.6)^b
1650 mm/yr	2375	779	1375 (11)^d	1779	12.3 (0.2)^c
Pawson silt loam					
Control	0	779	148 (2)^a	631	6.0 (0.3)^a
825 mm/yr	1190	779	609 (32)^b	1360	13.3 (0.7)^b

Infiltration

The data showed that irrigating the TMW onto the target soils had a negligible effect on the infiltration rate. The only significant difference in infiltration rate between the different treatments and the control was observed when the largest soil pores were measured (Figure 36). The infiltration rate was higher for the treatments compared with the control, which indicated that the treatment aided the soil infiltration. The constant irrigation of TMW during the summer months (when the soil is slightly hydrophobic due to lack of rainfall) increases all water infiltration, as the irrigation will help to prevent the soil from becoming hydrophobic. There was a difference between the treatments for the Akaroa lysimeters when measuring the infiltration from the smallest pores (0.15 mm radius and smaller). This indicated that over time, steps may need to be taken to ensure that soil structure and infiltration remain optimal, such as an application of gypsum. The data showed that the total infiltration from the highest treatment of both salt, (Table 23) and TMW volume, (Figure 36), was not significantly different to that of the lesser treatments (lower Na concentration and lower TMW volume). This showed that any potential problems will not arise immediately, and so any annual application of gypsum is not needed immediately.

Table 23. Soil sorptivity ($m/s^{0.5}$) of soil columns under a -0.5 cm suction (soil pore radius 3 mm and less) after different volumes of irrigation had been applied. Brackets indicate standard error. Letters indicate statistical groups.

Volume (mm)	87.05 mm	478.26 mm	991.047 mm	1404.76 mm
Control	0.0033 (0.0006)[a]	0.0028 (0.0009)[a]	0.0036 (0.0008)[a]	0.0036 (0.0007)[a]
Treatment 1	0.0027 (0.0002)[a]	0.0024 (0.0006)[a]	0.0037 (0.0012)[a]	0.0048 (0.0003)[a]
Treatment 2	0.0022 (0.0006)[a]	0.0029 (0.0005)[a]	0.0029 (0.0007)[a]	0.0036 (0.0008)[a]
Treatment 3	0.0022 (0.0002)[a]	0.0021 (0.0005)[a]	0.0025 (0.0008)[a]	0.0045 (7E-05)[a]

Aggregate Stability and Soil Chemistry

The soil Na concentration showed that the risk of aggregate breakdown was low. Sodium from the lysimeters had a low value in the top 15 cm of the soil profile, where most of the Na accumulated in the soil, and where the primary soil aggregates, and infiltration occurs. The increase of Na concentration in the TMW was used to simulate the effects of irrigation after an extended period of time. The data indicated that soil Na accumulation had an insignificant effect on soil structure in the short and medium term.

Smith et al. (2015) stated that studies have shown that the irrigation of Na and K onto soil have had adverse effects on soil's physical properties in Australia and California, USA. Their study showed that the four common cations (Na^+, K^+, Mg^{2+} and Ca^{2+}) were important parameters for measuring irrigation water quality because of the effects these cations had on the soil's physical properties. Sodium is of concern when it has a high concentration in the soil. This is especially true in arid and semi-arid regions of the world where irrigation is required and water is a precious resource (Singh, 2016).

The significant difference in the pH of the soil columns showed that the TMW had caused H^+ to be leached from the soil. This was consistent with the results from the cations' concentration analysis. The Na soil concentration increased as the TMW concentration increased, but there was no significant leaching of Ca, Mg, Al, Fe, or any other element. This indicates that the Na caused H^+ to become mobile in the soil solution, and be leached from the soil. By comparison, there was no significant difference in the lysimeter pH results. One reason for this is that the Na in the TMW had not been sufficient to cause significant H^+ to be leached from the irrigated lysimeters.

Conclusions

The discharge of TMW from the Duvauchelle sewage treatment plant onto the soils of Banks Peninsula as irrigation is unlikely to have an effect, negative or positive, on the soil infiltration rate or aggregate stability. This was because of the high amount of natural Ca and Mg in the soil compared to the natural occurring and accumulated Na from TMW irrigation.

The irrigation of TMW onto the soils of Banks Peninsula is not expected to cause aggregate instability, or decrease the soil's infiltration rate. However, more research is needed to determine the extent of the long term environmental effects of TMW irrigation on these soils. Research is also needed to determine how irrigating onto native plants will influence the potential environmental effects of TMW irrigation on these soils.

Outlook

The project is continuing at a field trial site at Duvauchelle, (Figure 15). This trial is being used to test the effects of irrigation of TMW onto New Zealand native plants, which are native to the Banks Peninsula area.

In July 2015, we planted 1350 native trees (Figure 41), divided into 27 blocks of three different vegetation types, (Table 24). Twelve of the 27 blocks are receiving TNW at a rate of 500 mm during the growing season (October – April), a similar rate to that used on an irrigated dairy farm in Canterbury. Effluent irrigation started in January 2016. Weeds were controlled using a lawnmower. An information board was installed near the roadside describing the aims of the experiment.

Figure 40. The field trial in Piper's Valley Road shortly after planting. The gate is at the top left of the picture.

Table 24. Composition of the three vegetation types used in the experiment. The design of the field plot is shown below.

Vegetation type 1		Vegetation type 2		Vegetation type 3	
Mānuka	*Leptospermum scoparium*	Akiraho	*Olearia paniculata*	Kapuka	*Griselinia littoralis*
Kānuka	*Kunzea robusta*	Puahou	*Pseudopanax arboreus*	Tarata	*Pittosporum eugenioides*
		Karamu	*Coprosma robusta*	Tī kōuka	*Cordyline australis*
		Hall's tōtara	*Podocarpus cunninghamii*	Harakeke	*Phormium tenax*
				Wharariki	*Phormium colensoi*

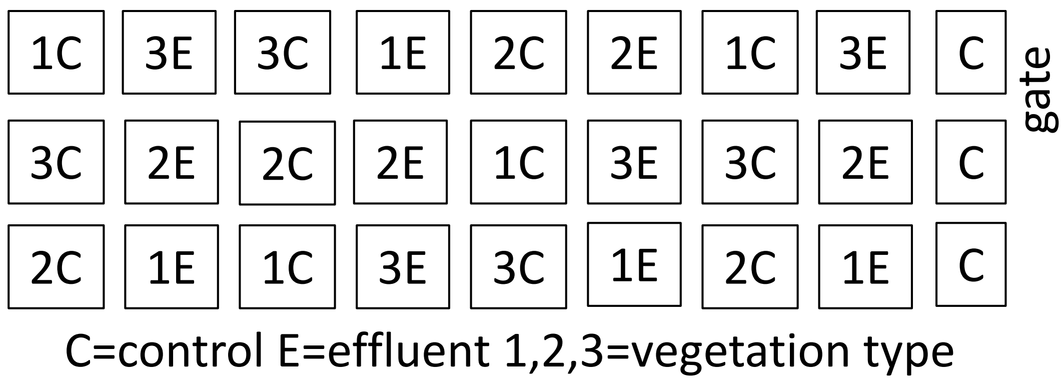

Figure 41. Layout of vegetation block types at the field trial site. Numbers represent vegetation types as stated in Table 24.

In May 2017 the survival of the plants was recorded along with the canopy volume of each individual plant. Soil and plant samples have been taken for chemical analysis. In June 2017, all areas within the plot that were not under native vegetation were planted with silver tussock (*Poa cita*). It is hoped that these tussocks will minimise the need for further weed control at the site.

Many local parties, including the Christchurch City Council, Ngai Tahu and the local residents wished for native species to be planted instead of other vegetation such as pine trees. The field trial is investigating the effects of TMW irrigation, but the experiment is also investigating potential benefits. This includes the use of native plants to produce products that have value, such as manuka honey and natural oils. Large scale irrigation of native species with TMW will prevent the need to discharge the TMW into the Akaroa harbour.

Appendix

Lysimetry and soil columns

Lysimeters are devices that isolate a soil block from its surroundings in order to control variables so that the soil-water-plant conditions can be monitored to determine various terms (Hillel et al., 1969; McIlroy et al., 1963; Tanner, 1967). Lysimeters are usually used to make determinations of the hydrologic equation as they are a convenient way of measuring inputs and outputs of water from a set amount of soil (Tanner, 1967). When a lysimeter has been dug, it is often exposed to natural surroundings so that the soil is affected in a realistic way (McIlroy et al., 1963). Lysimeters allow the measurement of a variety of factors. These include water, gas and fertiliser and nutrient losses, such as N from the soil (Cameron et al., 1992).

Materials and design of a lysimeter

A simple drum lysimeter design is a metal cylinder, 5mm thick steel plate, which insulates the soil inside the lysimeter from the soil outside of it (Cameron et al., 1992). This lysimeter design has a collection base that collects all the drainage into a drum for analysis, (Figure 42).

Other lysimeter designs exist to improve the measure factors that the simple drum lysimeters cannot, such as gasses that are released from the soil.

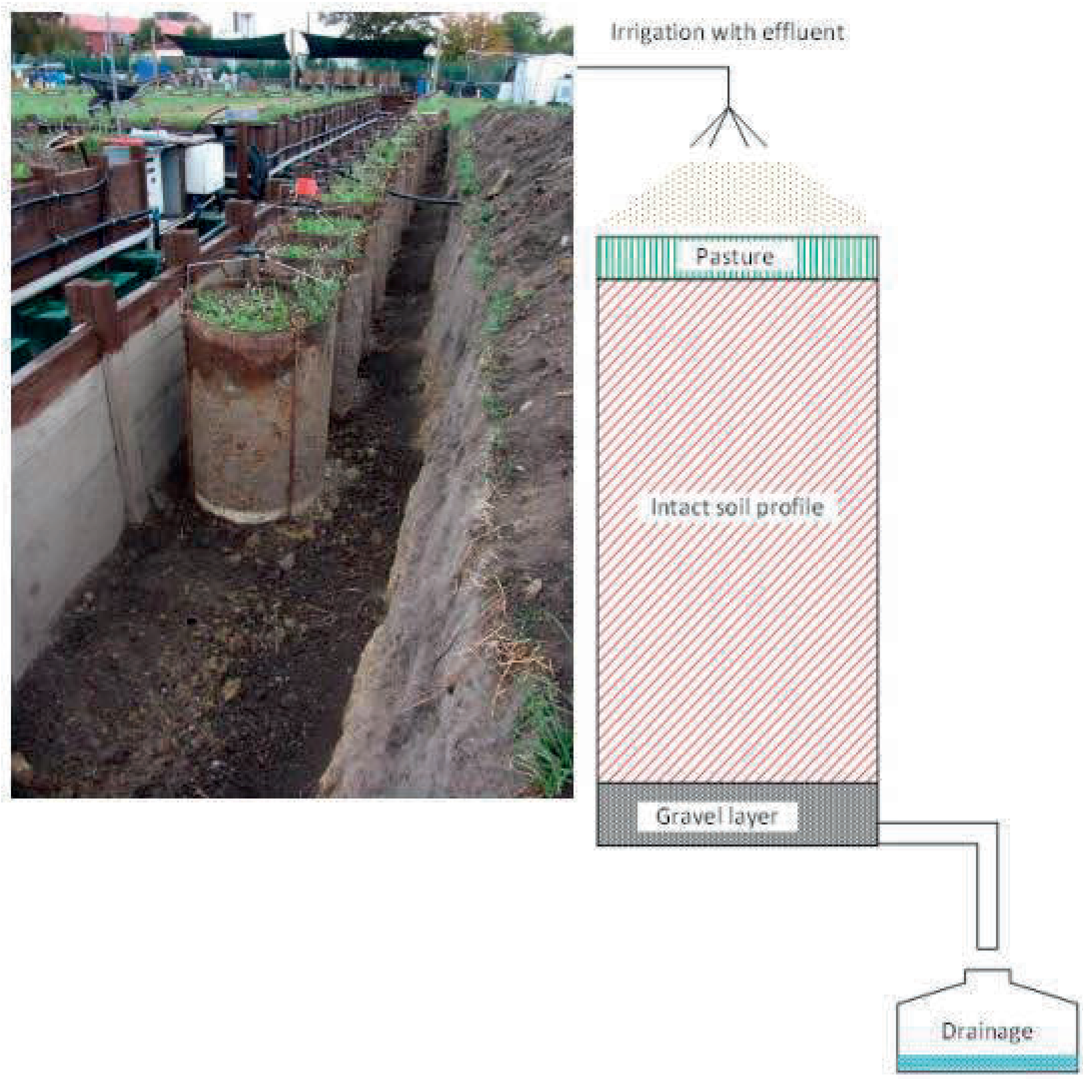

Figure 42. Diagram showing the design of a simple drum lysimeter.

Some more elaborate lysimeter designs are weighed to determine the input and output of liquid and gas from the lysimeter.

How lysimeters are installed

The lysimeters are placed on top of the sample soil. Trenches are dug around the lysimeters so that they can be easily pressed into the soil without damaging the soil or the lysimeter (Cameron et al., 1992). The lysimeter is pressed into the soil until the top of the soil is level with the top of the lysimeter, so that the lysimeter is not causing any obstruction of the sunlight or rainfall, therefore best simulating natural conditions (Hillel et al., 1969). A cutting plate is inserted under the lysimeter to separate it from the soil below so that the lysimeter can be removed from the sample site (Cameron et al., 1992).

Water balance equation

The sum of the mass of water that is initially present in the lysimeter system and the mass of water that is added to the system must be able to be accounted for, either by leaving the system, or being present in the system through the use of water storage in the system. This is represented by the water balance equation, (Equation 5).

$$P + I + Ro = ET + D + \Delta W$$

Equation 5. Water balance equation showing inputs and outputs of water volume in a system.

Where;

P is the mass of the water that is precipitated onto the soil,

I is the mass of the water that is irrigated onto the soil,

R_o is the mass of the water that is runoff on the soil (positive when water is being added by runoff, and negative when water is being removed by runoff),

ET is the mass of water that is lost due to evapotranspiration, from both evaporation from the soil and transpiration from the vegetation,

D is the mass of water that is lost due to drainage from the soil

ΔW is the change in the mass of water content (W) in the isolated soil mass

Here W is related to the amount of water that the mass of soil is capable of storing. Changes can occur due to an increase in the water stored due to runoff, precipitation or irrigation, or due to a decrease in the mass of water stored in the soil, due to drainage, vegetation uptake, or evapotranspiration (Holden, 2008).

Benefits and Limitations

The simple drainage lysimeter has a wide application use, and is common around the world due to its simplicity, (Figure 42) (Aboukhaled et al., 1982).

Due to the large size (2 to 4 m^2 surface for most grasses and field crops) the lysimeter has limitations in determining the evapotranspiration (Aboukhaled et al., 1982). The evapotranspiration can only be measured in the scale of weeks, rather than days or hours.

This is because of the time it takes for the water mass to penetrate through the lysimeter soil and be drained so that equilibrium can be established in the soil water storage, which is necessary for the evapotranspiration to be calculated. Weighed lysimeters have a benefit in accuracy. They allow direct measurements in the evapotranspiration in the time frame of hours, rather than weeks like non-weighed lysimeters (Cameron et al., 1992).

Edge-flow effect

Modern lysimeters include a cutting ring on the inside of the bottom of the lysimeter to leave a gap between the outside of the soil column and the inner edge of the lysimeter steel (Cameron et al., 1992). This allows heated liquid petrolatum to be injected into the gap. The petrolatum fills the gaps between the lysimeter and the soil, cools then solidifies. This prevents preferential water drainage from between the soil and the edge of the lysimeter. Drainage is forced to occur through the soil pores, which produces results that represent the real field scenario.

Soil Columns

Intact Cores

Intact soil columns are a true representation of the soil, as the soil has not been changed in any way. Because the columns have not been disturbed, properties of the soil such as soil porosity and aggregate stability are accurate to the field scenario.

Repacked Cores

Disturbed soil cores often have a lower mean bulk density than that of undisturbed soil cores (Cassel et al., 1973). This is because once the soil has been dug up (prior to repacking), the soil cannot be put back the same way. Therefore, there are gaps between the soil particles that were not previously present. This results in a lower bulk density, and a higher soil porosity (Vogeler, 2009). The change in the bulk density indicates that there are pores in the repacked soil cores that were not present in the undisturbed soil cores (Cassel et al., 1973). This in turn indicates that the results from a repacked soil core will be less accurate than those of an intact soil core.

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