Soil and grapevine responses to irrigation with treated municipal and winery wastewaters

by

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SUMMARY

In recent years, water scarcity and the ongoing drought have had serious implications for the agricultural industry in the Western Cape. The present study investigated the sustainability of two different types of wastewater for use as alternative irrigation water for grapevine production. The first objective was to assess the long-term effects of treated municipal wastewater irrigation on soils and grapevines in commercial vineyards in the Coastal region. The second objective was to investigate the use of in-field fractionally applied winery wastewater with raw water for grapevine irrigation under different climatic conditions.

To assess the impact of treated municipal wastewater irrigation on soil and grapevines, a long-term trial was conducted in commercial vineyards in the Coastal region of the Western Cape. Cabernet Sauvignon and Sauvignon blanc grapevines were irrigated using treated municipal wastewater from the Potsdam wastewater treatment works for 11 years. Grapevines were either rainfed (RF), irrigated with treated municipal wastewater *via* a single dripper line (SLD) or received twice the volume of wastewater *via* a double dripper line (DLD). Irrigation using treated municipal wastewater increased soil pH and electrical conductivity (EC_e). Furthermore, an accumulation of chloride (Cl^-) was observed in the topsoil, probably due to the chlorine-disinfection process that is carried out as part of the treatment process at the wastewater treatment works. Appreciable amounts of sodium (Na^+) and potassium (K^+) also accumulated in the topsoil due to wastewater irrigation. However, this did not result in enhanced uptake by grapevines. The near-saturation hydraulic conductivity (K_{ns}) at the surface of the soil decreased as the EC_e in the topsoil increased, with the lowest K_{ns} recorded for the DLD treatments. The irrigation reduced water constraints throughout the growing season compared to RF conditions, particularly in the case of Cabernet Sauvignon. Consequently, the SLD and DLD grapevines produced stronger vegetative growth and higher yields compared to RF. The present study indicated that, with proper management, grapevines can be irrigated successfully using treated municipal wastewater.

Previous research has indicated that soil type and winter rainfall have a pronounced effect on salt accumulation where winery wastewater is used for irrigation. The present study investigated the short-term effects of irrigation using in-field fractionally applied winery wastewater with raw water on different soil types under different climates. Suitable experiment sites were identified in the Coastal, Breede River and Lower Olifants River wine production regions, due to their vast difference in climate. Within each region, two plots of differing soil textures were selected. One season of irrigation using fractionally applied winery wastewater with raw water did not have a pronounced effect on soil EC_e or soil organic carbon content (SOC). Variable amounts of plant nutrients were supplied to grapevines *via* the irrigation water. High K^+ concentrations in the wastewater resulted in an accumulation in the soil and a subsequent increase in extractable potassium percentage (EPP´). Under the prevailing conditions, irrigation using in-field fractionally

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applied winery wastewater did not have adverse effects on grapevine vegetative growth, yield or grape juice characteristics. However, further research is needed to assess the sustainability of this particular practice over the long-term.

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OPSOMMING

Die onlangse waterskaarste en voortdurende droogte in die Wes-Kaap hou ernstige gevolge in vir die provinsie se landbousektor. Die studie het die volhoubaarheid van twee tipes afvalwater vir gebruik as alternatiewe besproeiingswater vir wingerdproduksie ondersoek. Die eerste doel was om die langtermyn effekte van besproeiing met behandelde munisipale afvalwater op grond en wingerdstokke in kommersiële wingerde in die Kusstreek te evalueer. Die tweede doel was om die gebruik van in-veld fraksioneel toegdediende kelderafvalwater met vars water vir wingerdbesproeiing te ondersoek onder verskillende klimaatstoestande.

‘n Langtermyn proef was uitgevoer in kommersiële wingerde in die Kusstreek om die impak van besproeiing met behandelde munisipale afvalwater op grond en wingerdstokke te evalueer. Cabernet Sauvignon en Sauvignon blanc wingerdstokke was vir 11 jaar besproei met behanddelde munisipale afvalwater afkomstig van die Potsdam afvalwatersuiweringsaanleg. Wingerdstokke was óf droëland verbou (RF), besproei met behandelde munisipale afvalwater deur ‘n enkel drupperlyn (SLD) óf het twee keer die volume afvalwater ontvang deur ‘n dubbele drupperlyn (DLD). Besproeiing met behandelde munisipale afvalwater het die pH en elektriese geleiding (EC_e) van die grond verhoog. ‘n Opeenhoping van chloried (Cl^-) was in die bogrond opgelet. Dit was waarskynlik as gevolg van die chloor-ontsmettingsproses wat as deel van die behandelingsproses by die suiweringsaanleg uitgevoer is. Noemenswaardige hoeveelhede natrium (Na^+) en kalium (K^+) het ook in die bogrond akkumuleer as gevolg van die afvalwaterbesproeiing. Dit het egter nie verhoogde opname deur die wingerdstokke veroorsaak nie. Die naby-versadigings hidroliese geleiding (K_{ns}) by die grondoppervlak het afgeneem met ‘n toename in die EC_e van die bogrond. In vergelyking met die RF toestande het afvalwaterbesproeiing die waterspanning van wingerdstokke verminder, veral in die geval van Cabernet Sauvignon. Gevolglik het die SLD en DLD behandelings sterker vegetatiewe groei en hoër opbrengste weerspieël in vergelyking met die RF. Die resulte van hierdie studie het aangedui dat wingerdstokke, mits dit korrek bestuur word, suskesvol besproei kan word met behandelde munisipale afvalwater.

Vorige navorsing het aangedui dat grondtipe en winterreënval ‘n merkbare effek het op soutakkumulasie waar kelderafvalwater gebruik word vir besproeiing. Hierdie studie het die korttermyn effekte van besproeiing met in-veld fraksioneel toegediende kelderafvalwater met vars water op verskillende grondtipes onder verskillende klimaatstoestande ondersoek. Op grond van klimaatsverskille is geskikte eksperimentele persele geïdentifiseer in die Kus-, Breederivier- en Benede Olifantsrivier wynproduksiestreke. Binne elke streek is twee persele met verskillende grondteksture gekies. Een seisoen van besproeiing met fraksioneel toegediende kelderafvalwater het nie ‘n noemenswaardige effek of grond EC_e of organiese koolstof (SOC) gehad nie. Wisselende hoeveelhede plant voedingstowwe is aan wingerstokke verskaf deur die

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besproeiingswater. Hoë K^+ konsentrasies in die afvalwater het 'n akkumulasie van K^+ in die grond veroorsaak en het ook die ekstraheerbare kalium persentasie (EPP´) van die grond verhoog. Onder die heersende toestande het besproeiing met in-veld fraksioneel toegediende kelderafvalwater nie negatiewe effekte op vegetatiewe groei, opbrengs of mos eienskappe gehad nie. Verdere navorsing is egter nodig om die volhoubaarheid van hierdie praktyk oor die langtermyn te evalueer.

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BIOGRAPHICAL SKETCH

Karla Hoogendijk was born in the Northern Suburbs of Cape Town on 2 November 1993. She began her school career at De Tyger Primary in Parow and matriculated from Tygerberg High School, Parow in 2011. She obtained her BScAgric degree in 2015 from Stellenbosch University, majoring in Soil Science and Viticulture. After a year of working in the agricultural industry, Karla enrolled for the MScAgric degree in Soil Science at Stellenbosch University and started her field work with the ARC Infruitec-Nietvoorbij in 2017.

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CHAPTER 1: IRRIGATION OF AGRICULTURAL CROPS WITH MUNICIPAL AND WINERY WASTEWATER – BACKGROUND, PROJECT OBJECTIVES AND LITERATURE REVIEW

1.1. BACKGROUND

Water is an essential resource required for agricultural crop production. Typically, the Western Cape of South Africa has a temperate Mediterranean climate which is characterised by warm, dry summers and mild, wet winters with rainfall mostly occurring between May and August. The mean annual rainfall in the province varies from *ca*. 300 mm to over 900 mm in some regions (Botai *et al*., 2017). The climate of the Western Cape is particularly suitable for the production of grapes and supports a very productive wine industry (Du Plessis & Schloms, 2017). However, fresh water resources are generally limited in grape growing areas. Sustainable grape production in the province is therefore highly dependent on winter rainfall and the application of irrigation in drier areas. In this regard, inconsistent rainfall and periodic droughts can severely impact the wine industry. During the 2014 to 2017 hydrological years, the province experienced its worst drought since 1904 (Botai *et al*., 2017). The City of Cape Town introduced level 6B water restrictions in February 2018 under which daily domestic water consumption was limited to 50 ℓ per person per day. In addition, the agricultural sector in the province had to reduce its consumption by an average of 60% of its normal water quota. Some regions were more severely affected than others as producers in the Lower Olifants River region only received 13% of their normal allocation (World Wildlife Foundation, 2018). Furthermore, the South African wine grape harvest amounted to *ca*. 1.2 million tonnes in 2018, which was 15% smaller than in 2017 and the smallest crop in more than ten years (Vinpro, 2018). Conserving water and improving water use efficiency is therefore of cardinal value to the wine industry. The reuse of effluents and wastewater may present a potential solution to relieve pressure on fresh water sources and provide alternative irrigation water during drought periods.

It should be noted that the term wastewater as used throughout this study can have different definitions (after Raschid-Sally & Jayakody, 2008):

- **Treated wastewater** is wastewater that has been subjected to one or more physical, chemical or biological processes at wastewater treatment works (WWTW) or on-site treatment facilities to reduce its pollution or health hazard and can be reused under controlled conditions for beneficial purposes such as irrigation.
- **Municipal wastewater** is usually a combination of the following:
  - Domestic effluent consisting of *blackwater* (excreta, urine and associated sludge) and *greywater* (kitchen and bathroom wastewater)
  - Water from commercial establishments and institutions, including hospitals
  - Industrial effluent and

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- Storm water and urban runoff.
- **Winery wastewater** is effluent generated by wineries during the winemaking process as well as other wine production processes such as cleaning and cooling.

The use of wastewater for irrigation of agricultural crops is common practice in many arid and semi-arid countries around the world. It is especially suitable in countries that have a Mediterranean climate as wastewater irrigation helps to mitigate the effects of dry spells during summer while significant rainfall during the winter months can leach excess amounts of salts from the soil applied *via* wastewater irrigation.

Thus far, the use of wastewater for irrigation in South Africa is very limited and very little is known about the use of treated municipal wastewater as irrigation water in the Western Cape. Despite a lack of information on the long- and short-term impacts of its use in the region, it is estimated that *ca*. 2 000 ha of arable land (including vineyards) in the Swartland and surrounding areas are being irrigated with treated municipal wastewater from the City of Cape Town's (CoCT) Potsdam WWTW in Milnerton as well as the Malmesbury municipality (Myburgh, 2018). In an area where grapevines are often grown under dryland conditions, the availability of additional irrigation water (in the form of treated municipal wastewater) has been extremely valuable to producers in the region.

A previous project (Project K5/1881) investigated the use of winery wastewater diluted to predetermined levels of chemical oxygen demand (COD) for irrigating grapevines under field conditions in the Breede River region of the Western Cape. The project was initiated and funded by the Water Research Commission (WRC) of South Africa and co-funded by Winetech, THRIP and the Agricultural Research Council (ARC). It was concluded that irrigation with winery wastewater diluted to 3 000 mg/ℓ COD did not adversely affect the vegetative growth and yield of grapevines under the prevailing conditions. With the exception of sodium (Na^+) and potassium (K^+), winery wastewater irrigation had no significant effect on the accumulation of elements in the sandy soil used in the vineyard field trial. However, the accumulation of Na^+ and K^+ remains a concern. Pot experiments revealed that both soil type and rainfall had an impact on the accumulation of salts in the soils (Mulidzi, 2016) and it was therefore suggested that a field study be launched to investigate the effect of these parameters on vineyard performance under field conditions. The great range of different climatic conditions within the Western Cape made it possible to investigate the effect of climatic factors on the potential re-use of winery wastewater for irrigating vineyards. Diluting the winery wastewater to pre-determined COD levels prior to irrigation was considered to be impractical at commercial level and it was suggested that winery wastewater should be augmented in-field by applying a percentage of the irrigation requirement as undiluted winery wastewater followed by raw water for the remainder of the irrigation requirement. This would not only be a more practical approach but may allow more sufficient

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leaching of applied salts and help to mitigate unpleasant odours that are associated with winery wastewater.

Therefore, this study aimed to investigate the impact of irrigation using treated municipal wastewater and fractionally applied winery wastewater on soil chemical and physical properties as well as grapevine performance and yield under field conditions in the Western Cape.

1.2. PROJECT OBJECTIVES

This project formed part of a long-term study performed by the Soil and Water Science division of the ARC Infruitec-Nietvoorbij which assessed the use of treated municipal wastewater for irrigating grapevines in the Coastal region of the Western Cape. This study also formed part of a greater research project (Project K5/2561//4) funded by the WRC, Winetech and the ARC. The aim of the project is to predict if irrigation of vineyards with winery wastewater of a particular quality will be sustainable for a given soil/climate combination.

The specific objectives of this study were:

- To determine the effect of long-term irrigation with treated municipal wastewater on soil chemical and physical properties as well as grapevine responses of commercial vineyards in the Coastal region of the Western Cape.
- To determine the short-term effects of in-field fractionally applied (augmented) winery wastewater with raw water on soil chemical and physical properties as well as grapevine responses under different climatic conditions within the Western Cape.

1.3. LITERATURE REVIEW

1.3.1. Introduction

Wastewater is produced in large quantities as a by-product of various industrial, commercial, agricultural and domestic activities. These waters are often poorly treated and discharged into fresh water bodies, causing serious environmental pollution (Odjadjare & Okoh, 2010). The reuse of wastewater for irrigation in agriculture is a possible solution to this problem. It is already a common practice in many arid and semi-arid countries around the world where fresh water resources are scarce or not suitable to be used for irrigation (Levy *et al*., 2014). It has been suggested that 1.5% to 6.6% of the 301 million ha total global irrigated area is irrigated with wastewater (Sato *et al.*, 2013). One advantage of irrigating with wastewater is that it often contains essential plant nutrients and organic matter which can enhance crop productivity (Chen *et al*., 2013 and references therein). Many farmers in urban and peri-urban areas of developing countries deliberately use undiluted wastewater for irrigation as it is a cost efficient and reliable source of plant nutrients (Qadir *et al*., 2010). This not only assists farmers to generate greater revenues to support their families and enhance local or regional economic activity, but also supplies communities with fresh fruits and vegetables that might not be available if farmers were unable to

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irrigate with wastewater (Qadir *et al.*, 2007). Other advantages include savings in water and fertiliser costs as well as reduced contamination of natural water resources. However, irrigation with wastewater also has its drawbacks as it often contains appreciable amounts of inorganic salts that can lead to increased soil salinity, which may not only negatively impact plant growth but also have adverse effects on soil structural integrity and associated soil physical properties. In addition, applying large amounts of soluble salts to soils may lead to groundwater contamination and further diminishing of fresh water reserves. The presence of heavy metals, harmful pharmaceuticals and pathogens, like *Escherichia coli*, in wastewater can have serious health impacts on consumers. The latter is of even more importance where irrigated crops are to be consumed raw. The importance of public perception and acceptance should also not be underestimated as consumers may not be in favour of buying fresh produce which was irrigated with wastewater.

The water scarcity experienced in the Western Cape in recent years has warranted the search for alternative water sources to use for irrigation. At the same time, population growth and development in the Western Cape has created a need for new and improved strategies for managing municipal wastewater (Jovanovic, 2008). Only a small percentage of treated municipal wastewater produced by the CoCT is reused for summer irrigation. If this can be increased, a larger portion of fresh water can be freed for domestic supply (Jovanovic, 2008). The wine industry in the Western Cape makes a significant contribution to the economy of the region and provides a large number of employment opportunities (Howell, 2016). Wineries consume large amounts of raw water through wine production processes and in return generate large volumes of wastewater. The management requirements of winery wastewater differ from other effluents such as municipal wastewater in that it has a very variable composition and its generation varies significantly over a daily, monthly and annual cycle (Christen *et al*., 2010).

1.3.2. Municipal wastewater

1.3.2.1. Volumes of municipal wastewater generated

Information regarding the volume of municipal wastewater that is generated worldwide tends to be scarce, outdated and/or inconsistent (Sato *et al*., 2013). The United Nations World Water Development Report (2017) estimated that approximately 3.14×10^{11} m^3 of municipal wastewater is generated globally each year. Half of this volume is produced by Brazil, China, India, Indonesia, Japan, Russia and the United States (Mateo-Sagasta *et al.*, 2015). Information on South African wastewater production is limited. According to the FAO AQUASTAT database (2017), 3.54×10^9 m^3 of municipal wastewater was produced in South Africa in 2009. This estimation is 3.42×10^8 m^3 more than was reported for the year 2000 (Sato *et al*., 2013). The city of Cape Town has 23 WWTWs that has the capacity to treat approximately 3×10^8 m^3 of wastewater annually (CoCT, 2017). In 2007/2008, 2.18×10^8 m^3 of treated effluent was discharged from 16 of the WWTWs around Cape Town (Department of Water Affairs [DWA], 2010). During the 2015/2016 financial

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year, approximately 2.27 x 10^8 m^3 of wastewater was collected at and discharged by WWTWs around Cape Town (CoCT, 2016). After sufficient treatment, most of the treated municipal wastewater is discharged into rivers, canals, wetlands, aquifers or the sea. Five percent of the city's used potable water is treated and recycled for non-potable uses, mainly irrigation (CoCT, 2017). This recycled water is redistributed to over 160 consumers in the vicinity of the city, which includes schools, sports centres, golf courses, farms, industries and commercial developments. The water is also used for irrigating municipal parks and flowerbeds in the city. The Potsdam WWTW near Milnerton has the largest capacity to generate recycled effluent, being able to produce 1.67 x 10^7 m^3 of treated effluent annually. The DWA recorded that 1.28 x 10^7 m^3 of municipal wastewater was treated and recycled at the Potsdam WWTW in 2007/2008. The City of Cape Town has operated this wastewater reuse system for several decades and is one of the few local authorities that regards the reuse of treated municipal wastewater as a vital component of its integrated water management plan (Adewumi *et al*., 2010).

1.3.2.2. Quality of municipal wastewater

Knowledge on the quality of irrigation water is critical for understanding its management for long-term use and productivity (Tak *et al*., 2012). For the purpose of this review, water quality will refer to the characteristics and composition of water that could influence its suitability for irrigating agricultural crops. The evaluation of irrigation water is normally based on its chemical and physical properties (Ayers & Westcot, 1985; Angelakis *et al*., 1999; Hussain & Al-Saati, 1999; Pedrero *et al*., 2010; Tak *et al*., 2012) as well as its microbiological content (Angelakis, 1999; WHO, 2006; Qadir *et al*., 2007; Pedrero *et al*., 2010). There are numerous indicators to assess the chemical quality of water. These include (i) salinity of the irrigation water as measured by electrical conductivity (EC_w) (Ayers & Westcot, 1985; Hussain & Al-Saati, 1999; Tak *et al*., 2012); (ii) sodium adsorption ratio (SAR) as it can lead to the deterioration of soil physical properties (Ayers & Westcot, 1985; Hussain & Al-Saati, 1999; Laurenson *et al*., 2012; Tak *et al*., 2012; Müller & Cornel, 2017); as well as (iii) pH and alkalinity as it affects the solubility and availability of plant nutrients and toxic metals. Water with an inherently low pH can also be corrosive to pipelines, sprinklers and control equipment (Ayers & Westcot, 1985). In some instances, residual sodium carbonate (RSC) is used to describe the precipitation and dissolution of alkaline earth carbonates (Hussain & Al-Saati, 1999). A fourth indicator of water quality is COD and biochemical oxygen demand (BOD) and they are used to describe the organic matter in wastewater (Pescod, 1992; Paranychianakis *et al*., 2010) along with total dissolved solids (TDS) and total suspended solids (TSS) (Metcalf & Eddy, Inc. *et al*., 2007). The last indicator of water quality is specific ion toxicity with special reference to Na^+, chloride (Cl^-) and boron (B^{3+}) (Ayers & Westcot, 1985; Pescod, 1992; Pedrero *et al*., 2010), as well as other ions such as sulfate (SO_4^{2-}) (Hussain & Al-Saati, 1999; Tak *et al*., 2012), and various trace elements and heavy metals (Stevens *et al*., 2004). In addition, municipal wastewater often contains appreciable amounts of essential plant nutrients including

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nitrogen (N), phosphorus (P) and K^+ (Gupta *et al*., 1998; Yadav *et al*., 2002; Ryan *et al*., 2006; Rusan *et al*., 2007; Singh *et al*., 2012).

Many wastewater quality criteria are based on health risks pertaining to the exposure of farmers, workers and consumers to pathogens (Qadir *et al*., 2010). Therefore, the microbiological quality of municipal wastewater is of utmost importance. The World Health Organisation (WHO) released revised guidelines in 2006 for the safe use of wastewater, excreta and greywater in the agricultural context (WHO, 2006). These guidelines replaced the previous version (WHO, 1989) which stipulated maximum values of faecal coliforms (FC) and helminth eggs allowed in wastewater destined for irrigation use. The new guidelines comprise health-based targets and the standard metric of disease is expressed as Disability-Adjusted Life Years (DALY's). However, FC is still commonly used as health-based criteria of municipal wastewater, along with the presence of *E. coli* as it is the most representative species to determine the occurrence of faecal contamination (Paranychianakis *et al*., 2010). South African legislation for irrigation water quality (Table 1.1) also uses FC to determine the degree of faecal contamination. These quality variables vary greatly depending on the quality of the water supply, the nature of the wastes that are added during use and the degree to which the wastewater has been treated (Pedrero *et al*., 2010). The following sections of this review will highlight the fundamental differences between raw and treated municipal wastewater.

Table 1.1. General Authorisations for legislated limits for pH, electrical conductivity (EC_w), chemical oxygen demand (COD), faecal coliforms (FC) and sodium adsorption ratio (SAR) for wastewater used for irrigation in South Africa (Department of Water Affairs, 2013).

Parameter	**Maximum irrigation volumes (m^3/day)**		
	< 50	**< 500**	**< 2 000**
pH	6–9	6–9	5.5–9.5
EC_w (dS/m)	2	2	0.7–1.5
COD (mg/ℓ)	5 000	400	75
FC (per 100 mℓ)	1 000 000	100 000	1 000
SAR	< 5	< 5	Other criteria apply

1.3.2.2.1. Raw, untreated municipal wastewater

Raw municipal wastewater usually contains a combination of domestic and industrial effluents as well as storm water or run-off (Qadir *et al.,* 2010). Elevated levels of metals, metalloids and compounds of a volatile or semi-volatile nature are often found in industrial effluents, while domestic effluents commonly have a high pathogenic load (Qadir *et al.,* 2010). Pathogenic organisms such as viruses, bacteria, protozoa and helminth eggs, are commonly found in sewage effluents and pose a serious threat to workers and the end consumers of crops that have been irrigated with untreated wastewater (Iannelli & Giraldi, 2010). Frequent exposure to untreated wastewater containing skin irritants and heavy metals such as cadmium (Cd), lead (Pb) and mercury (Hg) is associated with numerous chronic health effects (Dickin *et al.,* 2016). A number

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of studies have also shown that the salinity of untreated wastewaters is generally higher than conventional water used for irrigation. Similarly, Rana *et al.* (2010) reported that the EC_w of sewage effluents were 51% higher than that of well waters that were used for irrigation. The high EC_w of untreated wastewaters can be ascribed to the exploitation of water by industries that discharge contaminated water back into sewerage systems (Gupta *et al.,* 1998). Khan *et al.* (2015) reported that untreated wastewater of an industrial nature had EC_w values that were four times higher than that of well waters. A study by Jung *et al.* (2014), however, found that even though EC_w values for untreated wastewater was higher than for ground water that was used for irrigation, it was still within the permissible limits to use as irrigation water (Pescod, 1992). The BOD and COD values of untreated sewage waters have been found to be much higher than that of conventional irrigation water (Yadav *et al.,* 2002; Abegunrin *et al.,* 2016, Tripathi *et al.,* 2016). The influent quality of untreated wastewater to WWTWs varies on a daily, monthly and seasonal basis (Metcalf & Eddy, Inc. *et al.*, 2007), therefore it is nearly impossible to describe a “typical” composition of wastewater.

1.3.2.2.2. Treated municipal wastewater

The objective of wastewater treatment is to reduce its risk for polluting the environment and to prevent health hazards through consumption of or contact with contaminated water. Lower levels of heavy metals (De la Varga *et al.,* 2013) and harmful pathogens (Kiziloglu *et al.,* 2008; De Sanctis *et al.,* 2017) have been reported for municipal wastewaters that have been subjected to some form of treatment process. The quality of treated wastewater will vary according to its degree of treatment. Wastewater treatment stages are traditionally categorised as preliminary, primary, secondary, tertiary and quaternary or advanced (Iannelli & Giraldi, 2010). Preliminary treatments aim to remove materials such as sand, oil, grease and grit that could interfere in subsequent stages of treatment. Primary treatment is usually a sedimentation process where primary sludge is separated from effluent. Effluents that have been subjected to preliminary and primary treatment processes indicate measurable reductions in BOD, TSS and total organic carbon (TOC) (Metcalf & Eddy, Inc. *et al.*, 2007). Kiziloglu *et al.* (2008) reported lower EC_w values for preliminary treated wastewater when compared to untreated wastewater and even lower still for wastewater that underwent primary treatment. Secondary treatment processes aim to reach quality standards that would allow municipal wastewater to be safely returned to the natural environment. This is obtained by removal operations such as biological treatments (*e.g.* oxidation ponds & activated sludge) and chemico-physical treatments (*e.g.* flocculation & clarification). Wastewater is subjected to tertiary treatment processes to further decrease TSS after secondary biological treatments (Metcalf & Eddy, Inc. *et al.*, 2007). A disinfection stage is also typically included as part of tertiary treatments (Iannelli & Giraldi, 2010). Considerable reductions in EC_w, turbidity, suspended solids (SS), COD, BOD, as well as metal concentrations in wastewater that was subjected to tertiary treatment processes have been reported by Rekik *et al.* (2017). Quaternary treatment is only used in instances where reclaimed water of a very high quality is required and

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includes processes such as reverse osmosis and nanofiltration which removes nearly all compounds possibly present in wastewater, including small ions (Metcalf & Eddy, Inc. *et al.*, 2007). Nanofiltration membranes have been found to reduce ion concentrations in biologically treated municipal wastewater (Bunani *et al.,* 2013). Thus, wastewater quality clearly increases with higher degrees of treatment.

1.3.2.3. Effect of irrigation with municipal wastewater on soil properties

1.3.2.3.1. Soil chemical properties

pH: Changes in soil pH due to irrigation water occur slowly over time as soils are usually strongly buffered against pH fluctuations (Ayers & Westcot, 1985). Irrigation with municipal wastewater can have variable effects on soil pH depending on the composition and pH of the specific wastewater used. Irrigation using municipal wastewater has been shown to increase (Qian & Mecham, 2005; Gwenzi & Munondo, 2008; Hermon, 2011; Lado *et al.,* 2012), decrease (Shahalam *et al.,* 1998; Rattan *et al.,* 2005, Keser 2013; Abunada & Nassar, 2015) or not have any significant effect on soil pH (Stevens *et al.,* 2003; Duan *et al.,* 2010; El-Nahhal *et al.,* 2013). Indeed, Rusan *et al.* (2007) reported inconsistent trends in soil pH where soils were irrigated with treated municipal wastewater for different amounts of time. This is most likely due to strongly buffered soils.

Electrical conductivity: The accumulation of water soluble salts in the plant root zone can cause salinity problems and could lead to a reduction in crop yield (Ayers & Westcot, 1985). Electrical conductivity as measured from a soil saturation extract (EC_e) is the most commonly used parameter for estimating soil salinity. It is easily measured and considered to be a practical index of the total ionised salt concentration in aqueous samples (Rhoades *et al*., 1999). Most studies that have investigated the effect of municipal wastewater irrigation on soils have reported significant increases in EC_e as a result of wastewater irrigation (Gupta *et al.,* 1998; Panahi Kordlaghari *et al.,* 2013; Bedbabis *et al.,* 2015; Andrews *et al.,* 2016; Ganjegunte *et al.,* 2016; Nicholás *et al.,* 2016). In most cases, increases in soil EC were attributed to greater concentrations of salts and TDS in the irrigation waters. A study by El-Nahhal *et al.* (2013) found no significant difference in EC_e where the EC_w of treated wastewater and fresh water used for irrigation was comparably low. In contrast, Hassanli *et al.* (2008) reported that irrigation with treated municipal effluent that had low salinity decreased soil EC_e significantly. Results from another study indicated a greater increase in EC_e for heavier clay soils that were irrigated with treated municipal wastewater when compared to lighter textured soils (Adrover *et al.*, 2017). This can possibly be explained by a positive correlation between clay content and soil EC (Sudduth *et al.* 2005). Ayers and Westcot (1985) highlighted the importance of leaching salts from the root zone in instances where poor quality water is used for irrigation as it can reduce soil-water availability to crops and affect yields.

Organic matter and organic carbon: The addition of organic matter (OM) to soils through the application of municipal wastewater can have important effects on soil nutrient storage and structure (Jaramillo & Restrepo, 2017). The positive effects of OM on soil physical properties include, amongst others, reduced bulk density, increased water holding capacity and increased aggregate stability (Tisdall & Oades, 1982; Oades, 1984). It also effects cation exchange capacity (CEC), buffer capacity, enzymatic activity, availability of contaminants as well as increases TOC (Jaramillo & Restrepo, 2017 and references therein). Irrigation with municipal wastewater has been shown to increase soil OM (Mohammad & Mazahreh, 2003; Kiziloglu *et al*., 2007; Al-Omron *et al*., 2012; Bedbabis *et al*., 2014) and soil organic carbon (OC) (Rattan *et al*., 2005; Gwenzi & Munondo, 2008; Ghosh *et al*., 2012; Mojid & Weysure, 2013; Andrews *et al*., 2016). It has, however, also been concluded that irrigation with treated municipal wastewater has no significant effect on soil OM (Pedrero & Alarcón, 2009; Chen *et al*., 2015; Pérez *et al*., 2015). Herpin *et al*. (2007) reported a decrease in soil OM after application of wastewater irrigation. This is most likely due to the stimulation of microbial growth as a result of labile C and N supplied through the wastewater (Tarchouna *et al*., 2010).

Nitrogen: Wastewater contains variable amounts of ammonium (NH_4^+), nitrate (NO_3^-) and organic N depending on the degree of treatment. The sum of these are often referred to as total N (Durán-Álvarez & Jiménez-Cisneros, 2016). Sources of N in municipal wastewater include food scraps, body exudates, N-containing cleaning chemicals and personal hygiene products, faeces and urine (Patterson, 2003). The beneficial addition of N to soils through irrigation with municipal wastewater have been widely documented. Rusan *et al.* (2007) and Kamboosi (2017) attributed increases in soil total N to elevated amounts of N found in treated wastewater used for irrigation. Increased soil NO_3^- levels in soils irrigated with wastewater in a pot experiment where untreated and treated municipal wastewater were compared with rainwater, were reported by Thapliyal *et al*. (2013). It has been suggested that the addition of N through wastewater irrigation can help reduce the need for additional N fertilisation (Chen *et al*., 2013a). However, where N levels in wastewater exceed the requirements of cultivated crops, excessive uptake by plants (Tak *et al*., 2012) and N leaching to groundwater sources (Kim & Burger, 1997; Candela *et al*., 2007) might pose possible risks. This can be prevented by scheduling irrigation on the basis of crop water use and by cultivating crops that have high N requirements (Stewart *et al*., 1990).

Phosphorus: A study conducted in the United Kingdom by Comber *et al.* (2013) showed that the main contributors of P to domestic sewage was natural diet, food additives, automatic dishwashing detergents, laundry products, P addition to reduce Pb levels in drinking water, food waste and personal care products. Other sources of P in municipal wastewater include urban run-off, agricultural run-off and industrial discharge. The use of P-rich municipal wastewater for irrigation has been shown to increase soil P levels (Mohammad & Mazahreh, 2003; Adrover *et* al., 2012; Omidbakhsh *et al*., 2012). A study by Meena *et al*. (2016) found that available P increased by

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114% in soils that were irrigated with treated sewage water for 40 years. Similar results were obtained by Kiziloglu *et al*. (2008) and Rana *et al*. (2010) where untreated wastewater was used for irrigation. The increase in P in soils irrigated with wastewater can be attributed to greater P contents in the irrigation water as well as added OM to which P can adsorb (Bedbabis *et al*., 2015). Other researchers, however, have reported that wastewater irrigation had little or no effect on soil P despite irrigation waters having had high P concentrations (Midrar *et al*., 2004; Heidarpour *et al*., 2007).

Potassium: Although K^+ concentrations in municipal wastewaters are considered to be quite low when compared to wastewaters from an agricultural processing origin (Arienzo *et al.*, 2009a), numerous authors have reported an increase in soil K^+ due to irrigation with K-rich municipal wastewater (Kiziloglu *et al*., 2008; Galavi *et al*., 2010; Bedbabis *et al*., 2015; Alghobar & Suresha, 2016; Kamboosi, 2017). The use of municipal wastewater as an alternative K^+ fertiliser is particularly suitable as soluble and exchangeable forms of K^+ are increased more rapidly than conventional fertilisers (Arienzo *et al.*, 2009a) and K^+ is immediately available (Levy & Torrento, 1995). Irrigation with K-rich wastewaters also holds possible benefits in terms of soil fertility where soil K^+ is low (Howell, 2016), but long-term application can have negative impacts on soil chemical and physical properties (Laurenson *et al*., 2012). The extent to which wastewater irrigation impacts the soil K^+ levels are inherent to the K^+ levels of the particular wastewater used. A study by Pedrero and Alarcón (2009) has shown that irrigation with wastewater with relatively low levels of K^+ had little or no effect on soil K^+.

Calcium: Ca^{2+} is not only an essential plant nutrient, but it also plays a role in the structural stability of soils (Wuddivira & Camps-Roach, 2007) and the buffering of soil pH (Bache, 1984). Kiziloglu *et al.* (2007), Galavi *et al.* (2010), Rana *et al.* (2010) and Thapliyal *et al.* (2011) have reported Ca^{2+} increases in soils irrigated with wastewaters of varying degrees of treatment. The addition of Ca^{2+} through wastewater irrigation does not only increase plant available Ca^{2+}, but wastewaters that contain appreciable amounts of Ca^{2+} and Mg^{2+} can indirectly assist in the amelioration of excessive Na^+ application by reducing SAR (Howell, 2016). Some studies have found that well waters contained greater amounts of Ca^{2+} than wastewaters and therefore had a greater impact on soil Ca^{2+} levels (Neilsen *et al.,* 1991; Heidarpour *et al.,* 2007). In another study, however, Laurenson (2010) reported that soil Ca^{2+} levels decreased under treated wastewater irrigation despite irrigation water having high amounts of Ca. This could possibly be explained by Ca^{2+} uptake by plants, excessive losses through leaching or transformations of Ca^{2+} in soil (Abdelrahman *et al.,* 2011).

Magnesium: Although it is widely reported that municipal wastewater is a viable source of Mg^{2+} in soils (Gwenzi & Munondo, 2008; Samaras *et al*., 2009; Thapliyal *et al.,* 2013; Bedbabis *et al*., 2015), other studies have shown no such effect (Pedrero & Alarcón, 2009; Duan *et al*., 2010). In some instances, irrigation with municipal wastewater has even resulted in decreased soil Mg^{2+}

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levels (Abdelrahman *et al.,* 2011). Neilsen *et al*. (1991) reported a decrease in soil Mg^{2+} levels when compared to sites irrigated with well water even though the two irrigation waters (well water & municipal wastewater) had similar Mg^{2+} contents. It is highly likely that high Na^{+} and K^{+} contents in the wastewater resulted in a decrease in soil Mg^{2+} by mass exchange.

Sodium: Na^{+} is considered to be one of the most hazardous elements present in municipal wastewaters. These ions are not easily removed by conventional WWTW therefore, the salinity of reclaimed water is usually 1.5–2 times more than that of municipal drinking water (Chen *et al.,* 2013a). The accumulation of excessive amounts of Na^{+} in the soil can lead to the development of sodic soil conditions. These conditions are often characterised by swelling and dispersion of clays, clogging of soil pores, surface crusting, obstruction of water infiltration and increased runoff (Tak *et al.*, 2012). These conditions restrict the movement of water into and throughout the soil profile, thereby limiting water availability to active growing plant roots. The Na^{+} content of water and soils are most commonly characterised by its SAR (Na^{+} relation relative to Ca^{2+} & Mg^{2+}), therefore the SAR is often used to predict the sodicity hazard of irrigation water (Tak *et al.*, 2012), as depicted in Table 1.2. An abundance of Na^{+} in irrigation waters can have detrimental effects on soil structure (see Section 1.3.2.3.2). Numerous authors have reported increases in exchangeable Na^{+} and/or SAR where soils were irrigated with municipal wastewater (Kiziloglu *et al*., 2007; Galavi *et al*., 2010; Morugán-Coronado *et al*., 2011; Kallel *et al*., 2012). A study conducted by Andrews *et al*. (2016) in a humid region where soils were irrigated with municipal wastewater for over 50 years showed that the SAR at sites irrigated with wastewater increased due to elevated amounts of Na^{+} in the irrigation water as opposed to greater amounts of Ca^{2+} or Mg^{2+}. The high Na^{+} content in these wastewaters are most likely a result of an increase in the use of water softeners in the region. Netzer *et al*. (2014) reported that the addition of NPK fertilizer to treated municipal wastewater decreased the soil Na^{+} concentration of a clay soil under a Mediterranean climate. It is likely that the cations in the NPK fertilizer competed with Na^{+} for adsorption sites on the soil's exchange complex. Previous studies have suggested that when saline water is used for irrigation, irrigation volumes should exceed actual evapotranspiration (ET) to promote the leaching of salts and maintain soil salinity under threshold values for specific crops (Ayers & Westcot, 1985; Dudley *et al*., 2008). However, Netzer *et al*. (2014) concluded that this practice accelerated the accumulation of Na^{+} and increased SAR in a clay loam soil.

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Table 1.2. Guidelines for the assessment of sodium hazard of irrigation water based on sodium adsorption ratio (SAR_W) and electrical conductivity of irrigation water (EC_W) (after Ayers & Westcot, 1985).

	Potential for water infiltration problem	
	Unlikely	Likely
SAR_W	EC_W (dS/m)	
0–3	> 0.7	< 0.2
3–6	> 1.2	< 0.4
6–12	> 1.9	< 0.5
12–20	> 2.9	< 1.0
20–40	> 5.0	< 3.0

Chloride: There are much fewer references in the literature pertaining to the effect of municipal wastewater irrigation on soil Cl^- content as this ion is only essential to plants in very small quantities (Tak *et al*., 2012). It can, in fact, be toxic to plants if it is present in the soil at high concentrations. Tertiary treated municipal wastewater often contain high amounts of Cl^- as it is used in disinfection processes to remove harmful pathogens from wastewater prior to reuse (Asano & Levine, 1996). Increases in soil Cl^- due to irrigation with treated municipal wastewater were reported by Hogg *et al*. (1997), Pedrero and Alarcón (2009) and Bedbabis *et al*. (2015). Segal *et al*. (2011) suggested that Cl^- is a good indicator to estimate salt load since (i) it has a strong correlation with EC, (ii) it has a low relative uptake rate (ratio between uptake & supplied), (iii) it is an anion with low adsorption rate and high mobility in the soil and (iv) it is an anion that often occurs in wastewater. Although both Cl^- and Na^+ are considered to be the principal elements contributing to soil salinity (Chen *et al*., 2013a), a study by Netzer *et al*. (2014) concluded that Cl^- is more easily leached from the soil when compared to Na^+. Therefore, it is likely easier to manage than high soil Na^+ contents.

Trace elements: B^{3+}, copper (Cu^{2+}), iron (Fe^{2+}), manganese (Mn^{2+}), molybdenum (Mo^{2+}) and zinc (Zn^{2+}) are micronutrients or trace elements that are essential to plant growth, but compared to macronutrients such as N, P and K^+, are required in much smaller quantities. Trace elements are normally present in municipal wastewater at relatively low concentrations as a result of industrial wastewater contamination (Tak *et al*., 2012). Depending on the source and the specific ion content of the water, municipal wastewater irrigation has been shown to increase soil B^{3+} (Neilsen *et al*., 1991; Qian & Mecham, 2005; Pedrero & Alarcón, 2009; Lado *et al*., 2012), Cu^{2+} (Gupta *et al*., 1998; Kiziloglu *et al*., 2007; Meena *et al*., 2016), Fe^{2+} (Galavi *et al*., 2010; Singh *et al*., 2012; Bedbabis *et al*., 2015), Mn^{2+} (Kiziloglu *et al*., 2008; Omidbakhsh *et al*., 2012; Bedbabis *et al*., 2015), Mo^{2+} (Galavi *et al*., 2010) and Zn^{2+} (Samaras *et al*., 2009; Rana *et al*., 2010; Thapliyal *et al*., 2013; Meena *et al*., 2016). However, there are inconsistencies in the literature on the effect of wastewater irrigation on soil trace elements. Mohammad and Mazahreh (2003) reported increased Fe^{2+} and Mn^{2+} levels due to wastewater irrigation, but it had no significant effect on Cu^{2+} and Zn^{2+},

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whereas, Rusan *et al*. (2007) found that soil Fe^{2+}, Mn^{2+} and Zn^{2+} were not consistently affected by wastewater irrigation and Cu^{2+} was completely unaffected.

Heavy metals: Heavy metals such as arsenic (As), Cd, chromium (Cr), Pb, Hg and nickel (Ni) can be present in municipal wastewater at varying levels depending on the source and degree of treatment of the wastewater. They are usually present at low concentrations, but long-term application of reclaimed water can lead to heavy metal build-up in soils over time (Chen *et al.*, 2013b). Chipasa (2003) and Qdais and Moussa (2004), however, reported that heavy metals can be removed effectively from the wastewater stream by conventional treatment processes and be concentrated in the sewage sludge or solid phase. This was confirmed by Christou *et al*. (2014) and Bedbabis *et al*. (2015) who concluded that irrigation with tertiary treated wastewater had no significant effect on soil heavy metal content. In contrast, many authors have reported heavy metal accumulation in soils irrigated with treated municipal wastewater (Rattan *et al*., 2005; Rusan *et al*., 2007; Ghosh *et al*., 2012; Bao *et al*., 2014; Meena *et al*., 2016). It is important to note that many of these studies did not indicate to what extent the wastewater had been treated before application of irrigation. This is important as heavy metals are often only removed from wastewater during tertiary treatment stages (Metcalf & Eddy, Inc. *et al*., 2007). It is, however, evident that irrigation with untreated municipal wastewater or wastewater that has undergone very little treatment increases soil heavy metal content (Shariatpanahi & Anderson, 1986; Liu *et al*., 2005; Rana *et al*., 2010; Singh *et al*., 2012; Abunada & Nassar, 2015; Aydin *et al*., 2015; Meng *et al*., 2016).

1.3.2.3.2. Soil physical properties

Hydraulic conductivity: The hydraulic conductivity (K) of a soil refers to its ability to conduct water within its volume (Lado & Ben-Hur, 2009) and is influenced by the pore geometry of the soil as well as the density and viscosity of the conducted liquid (Hillel, 2004). Lin *et al*. (2003) reported that the kinematic viscosities of secondary treated municipal wastewater and that of pure water were similar over temperatures ranging between 15°C and 35°C. This indicates that the effects of the wastewater's density and viscosity on the soil's K, was insignificant. However, the pore geometry of soils can be altered through various clogging processes. Factors that lead to soil clogging can be classified as physical, chemical and biological (Rice, 1974). Many studies have linked a change in K to the physical clogging of soil pores by suspended solids (SS) that is supplied through irrigation with municipal wastewater (Metzger *et al*., 1983; Vinten *et al*., 1983; Levy *et al*., 1999; Viviani & Iovino, 2004; Sepaskhah & Sokoot, 2010; Gharaibeh *et al*., 2016). Suspended particles in the wastewater is filtered through the soil pores as the water moves through the soil profile. This brings forth the accumulation of these particles in the topsoil and could lead to a reduction in the intrinsic K of the irrigated soil (Lado & Ben-Hur, 2009). Vinten *et al*. (1983) have reported greater reductions in K for soils having a finer texture due to accumulation of coarse SS at the soil surface. Similar results have also been reported by Lado and Ben-Hur (2010) who

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attributed the greater entrapment of SS to the large proportion of small pores in a clay soil and evidently leading to a reduction in saturated hydraulic conductivity (K_S). Levy *et al*. (1999) emphasised the importance of the organic load of the irrigation water on *K*, concluding that greater reductions in *K* are likely to be found where greater concentrations of SS are present in wastewater used for irrigation.

Chemical clogging of soil pores occurs as a result of changes in soil swelling and clay dispersion (Lado & Ben-Hur, 2009). Chemical interactions between the dissolved salts in the irrigation water and the soil can lead to decreased pore diameters and could consequently decrease permeability (Rice, 1974). The occurrence of chemical clogging in soils are highly dependent on the soil's clay mineralogy, exchangeable sodium percentage (ESP) and exchangeable potassium percentage (EPP) of the irrigated soil as well as the EC_w of the irrigation water. It has previously been reported that *K* decreases with an increase in ESP and a decrease in EC_w of the percolating solution (Quirk & Schofield, 1955). An ESP of 15% and higher was proposed by the United States Salinity Laboratory (1954) as the level at which soil structure will be adversely affected, yet many studies have reported soil structure degradation and decreases in K_S at ESP lower than 15% (Summer, 1993 and references therein). Lado and Ben-Hur (2010) reported a significant reduction in K_S following the irrigation of a calcareous loamy soil with municipal wastewater that had undergone reverse osmosis. This result was ascribed to soil swelling and clay dispersion caused by the soil's high ESP (11%) and the low EC_w of the wastewater (0.2 dS/m). On the contrary, Balks *et al*. (1998) concluded that although higher ESP lead to a greater tendency of soil dispersion, it did not necessarily cause a decrease in K_S.

Clay dispersion during wastewater irrigation can also occur as a result of the interaction between the dissolved organic molecules in the wastewater and the clay particles of the soil (Lado & Ben-Hur, 2009). Tarchitzky *et al*. (1999) reported that this interaction lead to increased dispersion of clay into suspension and attributed it to the adsorption of negatively charged dissolved organic molecules to the positively charged edges of clay particles- preventing edge-to-face association of the clay particles. The biological clogging of soil pores occurs when micro-organic biomass, such as algae, bacterial growth and their by-products reduce the pore diameter (Rice, 1974). Studies conducted by Vandevivere and Baveye (1992) and Magesan *et al*. (2000) showed that *K* decreased in soils irrigated with wastewater with high C: N ratios, confirming that the addition of growth substrates (such as C) increased the activity of micro-organisms and accelerated the pore clogging process. It is, however, important to note that the majority of these studies made use of laboratory experiments to investigate *K*. Thus, the results of these studies do not necessarily represent field conditions and cannot be compared to in-situ measurements directly. Therefore, a real need in the literature for studies that investigate the effect of municipal wastewater irrigation on in-field determined *K* exists.

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Infiltration rate: Infiltration is the process by which water enters the soil by downward flow through all or part of the soil surface (Hillel, 2004). Infiltration rate is therefore the surface flux at any rate or pressure at which water is supplied to the soil (Hillel, 2004). Cohesion between soil particles decreases during wetting and leaching cycles which results in instability of the soil structure (Lado *et al*., 2004a). The exposure of the soil's surface to impacts from water droplets (either through rainfall or irrigation) can therefore lead to the development of a seal on the soil surface which could result in a reduction in infiltration rate (Assouline, 2004). Agassi *et al*. (1981) found that seal or crust formation arises from two possible mechanisms (i) a physical dispersion of soil aggregates caused by the impact of water droplets and (ii) a chemical dispersion which is dependent on the soil's ESP as well as the EC_w of the applied water. In general, soils that have been irrigated with municipal wastewaters tend to have higher ESPs and are susceptible to decreases in infiltration rate when solutions with low EC_w, such as rainwater, infiltrates the soil. This phenomenon has been demonstrated by Assouline and Narkis (2011) and Bedbabis *et al*. (2014). In both studies, greater decreases in infiltration rate in soils irrigated with treated municipal wastewaters were observed when compared to rainfed or fresh water irrigated treatments. Cook *et al*. (1994) conducted a study on a highly permeable volcanic ash soil in New Zealand and reported a decrease in infiltration rate of nearly 50% after three years of wastewater irrigation. Gharaibeh *et al*. (2007) compared the infiltration rate of vertisols that were irrigated with treated municipal wastewater for 2, 5 and 15 years, respectively, to a rainfed control treatment. Their results indicated lower infiltration rate for the soils irrigated with wastewater for 2 and 5 years. The decrease in infiltration rate was attributed to clay swelling that diminished the cracks that are often present in vertisols. However, the soils that were irrigated for 15 years had higher infiltration rates compared to the control. The soils of this treatment were characterised by high ESP (16.6%) and exhibited a greater percentage of large cracks which allowed dispersed material to pass through the soil surface and increase the infiltration rate (Gharaibeh *et al*., 2007).

Bulk density: The application of municipal wastewater irrigation can have adverse effects on soil bulk density due to the high concentration of SS that is present in most wastewaters. The accumulation of SS in soils through wastewater irrigation can result in decreased macro-porosity and increased micro-porosity, ultimately affecting bulk density (Tunc & Sahin, 2015). Many authors have observed lower bulk densities in soils due to municipal wastewater irrigation (Mathan, 1994; Vogeler, 2009; Mojiri, 2011; Mojid & Wyesure, 2013). A study by Tunc and Sahin (2015) showed that even though bulk density was lower in wastewater irrigated soils compared to those irrigated with groundwater, the differences were considered to be insignificant for any practical purposes. Other researchers have indicated increased bulk densities following irrigation with municipal wastewater (Hassanli *et al*., 2008; Azouzi *et al*., 2016). Conversely, some studies have concluded that municipal wastewater irrigation had little or no effect on bulk density (Abedi-Koupai *et al*., 2006; Bardhan *et al*., 2016; Urbano *et al*., 2017).

Porosity: As soil porosity and bulk density are inversely related, opposite trends for porosity has been reported to that of bulk density. Some researchers reported improved porosity in wastewater irrigated soils (Mathan, 1994; Shahalam *et al*., 1998; Vogeler, 2009; Mojid & Wyesure, 2013), whilst others observed decreased porosity (Coppola *et al*., 2004; Aiello *et al*., 2007).

Water retention: There exists a strong correlation between soil water content and pore size distribution (Tunc & Sahin, 2015). Macro-pores are responsible for drainage and aeration, meso-pores facilitate water conduction, while micro-pores supply water retention (Hillel, 2004). An increase in the volume of micro-pores due to wastewater irrigation has been reported to enhance the water retention capacity of soils (Mojid & Wyesure, 2013; Tunc & Sahin, 2015). Gharaibeh *et al*. (2007) ascribed the increased water retention of wastewater irrigated soil to an improvement in the aggregate stability of the soils.

Aggregate stability: Generally, soil aggregate stability and OM content are closely related (Tisdall & Oades, 1982). The addition of OC through municipal wastewater irrigation has been linked to improved aggregate stability in soils (Vogeler 2009; Tunc & Sahin, 2015). Conversely, Schact and Marschner (2015) reported lower aggregate stability in soils that were irrigated with treated municipal wastewater when compared to soils irrigated with fresh water.

1.3.2.4. Effect of irrigation with municipal wastewater on agricultural crops

1.3.2.4.1. Plant water relations

The high concentration of salts in municipal wastewater can influence the water relations and gas exchange of irrigated crops (Paranychianakis *et al*., 2004). Salinity negatively effects the water absorption capacity of plants and could result in water stress (Gómez-Bellot *et al*., 2015). Saline soil conditions can cause systematic accumulation of salts (primarily Na^+ & Cl^-) in the aerial parts of plants, which in turn can affect plant metabolic processes if ions are not compartmentalised within the cell vacuoles (Gómez-Bellot *et al*., 2013). Plants adapt to these osmotic stresses by exercising osmotic adjustment which maintains a positive turgor that is required for the opening of stomata and cell enlargement (Álvarez *et al*., 2012). Severe water losses are prevented by decreasing the aperture of stomata (Gómez-Bellot *et al*., 2015). These effects were demonstrated by Gómez-Bellot *et al*. (2013) in a study on *Euonymus japonica* (Japanese spindle) shrubs irrigated with treated municipal wastewater. The midday stem water potential (Ψ_S) of *E. japonica* irrigated with wastewater (EC_w 4 dS/m) were significantly lower than those irrigated with low salinity fresh water (EC_w <0.9 dS/m). Similar results were reported for *Viburnum tinus* L. (laurustinus) by Gómez-Bellot *et al*. (2015). In contrast, Nicolás *et al*. (2016) reported that treated municipal wastewater did not affect the Ψ_S of mandarin trees after six years of irrigation, despite the fresh water control having consistently lower salinity. Gonçalves *et al*. (2017) reported similar results for the predawn stem water potential (Ψ_{PD}) and Ψ_S of sugarcane. A study by Paranychianakis *et al*. (2004) investigating the effect of municipal wastewater irrigation on one-year-old Sultanina grapevines reported that Ψ_S was unaffected by wastewater irrigation, but Ψ_{PD} was significantly reduced in comparison with fresh water irrigated grapevines. They ascribed this reduction to the osmotic effect caused by the accumulation of salts in the rootzone. The unaffected Ψ_S was considered to be the result of grapevines' isohydric behaviour which controls water use and helps maintain the minimum leaf water potential (Ψ_L) at a constant value (Winkel & Rambal, 1993; Paranychianakis *et al*., 2004). Compared to the use of saline groundwater, irrigation with treated municipal wastewater has also been reported to enhance the water use efficiency (WUE) of forage maize (Alkhamisi *et al*., 2011). In contrast, Balkhair *et al*. (2014) reported significantly lower WUE for eggplant and okra irrigated with treated municipal wastewater.

1.3.2.4.2. Plant chemical status

Nitrogen: The use of municipal wastewater irrigation as a source of N for plant production has been well documented. McCarthy (1981) reported adequate levels of N in petioles of Shiraz grapevines that were irrigated with treated municipal effluent at a rate of 45 ℓ of effluent/grapevine/week, compared to grapevines irrigated with 135 ℓ potable water/grapevine/week. Neither of the treatments received additional N fertilisation. In the same experiment, grapevines irrigated with 135 ℓ effluent/grapevine/week did not exhibit excessive vegetative growth nor did it reduce fruitfulness (McCarthy, 1981). Similar findings were reported for the leaves of apple (Neilsen *et al*., 1989b), olive (Bedbabis *et al*., 2010), sweet cherry (Neilsen *et al*., 1991) and sugarcane (Gonçalves *et al*., 2017). In contrast, Neilsen *et al*. (1989a) found that municipal wastewater irrigation had no significant effect on the petiole N content of Riesling grapes when compared to grapevines irrigated with well water. This is in agreement with results presented by Martínez *et al*. (2013) for melon leaves and Urbano *et al*. (2017) for lettuce. Libutti *et al*. (2018) recorded lower concentrations of NO_3^- in tomato fruits that were irrigated with secondary treated agro-industrial wastewater when compared to tomatoes irrigated with groundwater. Using the same irrigation water sources, they reported higher concentrations of NO_3^- in the broccoli heads of wastewater irrigated broccoli plants. A study by Vergine *et al*. (2017) concluded that treated municipal wastewater can be used as a viable alternative to N fertiliser to enhance both the production and growth rate of cultivated fennel.

Phosphorus: Lal *et al*. (2015) conducted research on the effect of sewage water irrigation on the P uptake of different cropping sequences (food grain, agroforestry, fodder & vegetable production). They found that P uptake was improved by 30% as a result of sewage water irrigation. As a result, sewage water irrigation could supply 20% to 40% of the crops' P requirements (Lal *et al*., 2015). Higher P concentrations in the leaves of olive trees under municipal wastewater irrigation, compared to fresh water irrigation, have been reported (Bedbabis *et al.*,2010; Bourazanis *et al*., 2016; Bedbabis & Ferrara, 2018). However, Segal *et al*. (2011) and Petousi *et al*. (2015) concluded that municipal wastewater had no significant effect on the P concentration of

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olive leaves after four and three years of irrigation, respectively. Increased P content of barley (Rusan *et al*., 2007) and cabbage (Kiziloglu *et al*., 2008) due to municipal wastewater irrigation have also been reported.

Potassium: Numerous authors have reported increased K^+ concentrations in crops that were irrigated with municipal wastewater. These crops include olive (Bedbabis *et al*., 2010; Bourazanis *et al*., 2016; Bedbabis & Ferrara, 2018), apple (Neilsen *et al*., 1989b), sweet cherry (Neilsen *et al*., 1991), grape (Neilsen *et al*., 1989a), barley (Rusan *et al*., 2007) and cabbage (Kiziloglu *et al*., 2007). Conversely, studies by Koo and Zekri (1989), Zekri & Koo (1993), Pedrero and Alarcón (2009) and Pedrero *et al*. (2012) showed that irrigation with treated municipal wastewater did not affect K^+ levels in citrus leaves. In a study by McCarthy (1981), greater K^+ accumulation was observed in the petioles of Shiraz grapevines that were irrigated with fresh water when compared to those irrigated with sewage effluent, even though the water sources had similar K^+ concentrations.

Calcium: Enhanced Ca^{2+} levels in plants under municipal wastewater irrigation has been reported for Riesling grapevine petioles (Neilsen *et al*., 1989a), olive fruits (Batareseh *et al*., 2011; Bourazanis *et al*., 2016), maize roots (Khaskhoussy *et al*., 2013), cabbage heads (Kiziloglu *et al*., 2008) and tomato fruits (Libutti *et al*., 2018). Neilsen *et al*. (1991) reported lower Ca^{2+} concentrations in the leaves of sweet cherry trees that were irrigated with municipal wastewater when compared to trees irrigated with well water. They ascribed this effect to high K^+ levels in the applied wastewater which may have had an antagonistic effect on the uptake of Ca^{2+} and Mg^{2+}. In melon (Martínez *et al*., 2013), citrus (Zekri & Koo, 1993; Pedrero & Alarcón, 2009; Pedrero *et al*., 2012), olives (Petousi *et al*., 2015) and lettuce (Urbano *et al*., 2017), no significant effect of municipal wastewater irrigation was found on the Ca^{2+} concentrations of the leaves.

Magnesium: The use of municipal wastewater as an alternative source of irrigation water has been shown to increase the Mg^{2+} content of several crops, including olive leaves (Bourazanis *et al*., 2016; Bedbabis & Ferrara, 2018) and fruits (Batareseh *et al*., 2011), grapevine petioles (McCarthy, 1981), citrus leaves (Zekri & Koo, 1993; Morgan *et al*., 2008; Pedrero & Alarcón, 2009), maize leaves (Khaskhoussy *et al*., 2013), sugarcane leaves (Gonçalves *et al*., 2017) and cabbage heads (Kiziloglu *et al*., 2008). In contrast, other studies have reported that municipal wastewater irrigation had little or no effect on the Mg^{2+} levels of olive leaves (Petousi *et al*., 2015), citrus leaves (Koo and Zekri, 1989; Pedrero *et al*., 2012) and lettuce (Urbano *et al*., 2017). Other authors have reported lower concentrations of Mg^{2+} in the petioles of Riesling grapes (Neilsen *et al*., 1989a) and sweet cherry leaves (Neilsen *et al*., 1991) that were irrigated with municipal wastewater compared to those irrigated with well water. This could possibly be due to a K-Mg antagonism within the plant where wastewater contained appreciable amounts of K^+.

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Sodium: The uptake of Na^+ by plants as a result of municipal wastewater irrigation have been widely investigated. Netzer *et al*. (2014) studied the effect of treated municipal wastewater irrigation on Superior Seedless grapevines. Results showed greater accumulations of Na^+ in the xylem sap, trunk wood, bark and leaves of grapevines irrigated with the wastewater compared to those irrigated with fresh water. Similar findings have been reported for Shiraz petioles (McCarthy, 1981), olive leaves (Bedbabis & Ferrara, 2018), citrus leaves (Koo & Zekri, 1989; Zekri & Koo, 1993), root vegetables such as radish and carrots (Zavadil, 2009), maize roots (Khaskoussy *et* al., 2013), tomatoes and broccoli (Libutti *et al*., 2018) as well as cabbage heads (Kiziloglu *et al*., 2008). Grewal and Maheshwari (2013) reported increased Na^+ concentrations in pea and celery shoots of 54% and 19%, respectively, due to treated municipal wastewater irrigation. Alternatively, numerous studies have shown that municipal wastewater irrigation had little or no effect on Na^+ levels in crops (Pedrero & Alarcón, 2009; Bedbabis *et al*., 2010; Segal *et al*., 2011; Pedrero *et al*., 2012; Martínez *et al*., 2013; Petousi *et al*., 2015).

Chloride: Few studies have investigated the effect of municipal wastewater irrigation on the uptake of Cl^- by plants. Greater Cl^- concentrations were reported for the leaves of olive (Bedbabis & Ferrara, 2018) and citrus trees (Zekri & Koo, 1993; Pedrero & Alarcón, 2009; Pedrero *et al*., 2012) that were irrigated with treated municipal wastewater. Conversely, Bedbabis *et al*. (2010) and Segal *et al*. (2011) reported that irrigation with treated municipal wastewater had no significant effect on the accumulation of Cl^- in olive leaves when compared to trees that were irrigated with well water.

Trace elements: Asgari and Cornelis (2015) reported that although treated municipal wastewater irrigation had no significant effect on the Cu^{2+} and Zn^{2+} levels of wheat grains and maize kernels when compared to potable water, greater concentrations of Cu^{2+} was found in wheat grains, while maize kernels exhibited elevated Zn^{2+} concentrations. Conversely, Aydin *et al*. (2015) reported no significant effect on Cu^{2+} but higher Zn^{2+} concentrations in wheat grains irrigated with treated municipal wastewater compared to well water. Similar results were reported by Chung *et al*. (2011) for brown rice irrigated with treated wastewater. Gatta *et al*. (2018) concluded that although higher concentrations of Cu^{2+}, Fe^{2+}, Mn^{2+} and Zn^{2+} were observed in globe artichokes irrigated with treated municipal wastewater, the concentrations were still within permissible limits as prescribed by the WHO. Numerous authors have reported an increase in B^{3+} concentrations of citrus leaves following municipal wastewater irrigation (Zekri & Koo, 1993; Morgan *et al*., 2008; Nicholás *et al*., 2016). In contrast, Pedrero *et al*. (2012) reported that although the municipal wastewater used for irrigation had high B^{3+} levels, no significant increases were observed in lemon leaves. In contrast to fresh water irrigated olive trees, increased levels of Mn^{2+} and Zn^{2+} were measured in the leaves of olive trees that were irrigated with treated municipal wastewater for two years (Bedbabis *et al*., 2010; Bourazanis *et al*., 2016) and 10 years (Bedbabis & Ferrara, 2018). In addition to Mn^{2+} and

Zn^{2+}, Batareseh *et al*. (2011) also reported higher Cu^{2+} and Fe^{2+} concentrations in the leaves of olive trees irrigated with municipal wastewater.

Heavy metals: In India, Singh *et al*. (2010) investigated the contamination of vegetables with trace elements and heavy metals under municipal wastewater irrigation. It was found that heavy metal concentrations were several folds higher in wastewater irrigated vegetables compared to those irrigated with fresh water. Moreover, compared to cash crops, mean concentrations of heavy metals in wheat and rice were observed. However, due to higher dietary consumption of wheat and rice, these crops pose a greater risk to human health (Singh *et al*., 2010). Bao *et al*. (2014) reported significantly higher concentrations of Pb in wheat grains and maize kernels that were irrigated with municipal wastewater for 30 and 40 years compared to crops irrigated with well water. Yet, municipal wastewater irrigation did not affect the levels of Cd, Cr and Hg in these crops (Bao *et al*., 2014). Similar results were reported by Aydin *et al*. (2011) for wastewater irrigated wheat. In the latter study, Pb increases exceeded the permissible limit of 0.2 mg/kg dry weight as prescribed by Turkish Food Codex, whilst no translocation of Cd from soil to plants were observed and the translocation of Cr and Ni were insignificant. In contrast, Abunada and Nassar (2015) reported that municipal wastewater irrigation did not affect the Pb concentrations of alfalfa compared to well water. However, following five years of municipal wastewater irrigation, a steady increase in the Pb concentration of alfalfa was observed, but it was still below the permissible limit of 9 mg/kg which is enforced by China's State Environmental Protection Administration (SPEA). Shahalam *et al*. (1998) concluded that municipal wastewater irrigation caused no difference in the Cd and Pb concentrations of alfalfa and radish, compared to the same crops irrigated with fresh water. Conversely, greater Cd and Pb concentrations were reported for vegetables irrigated with municipal wastewater by Shariatpanahi and Anderson (1986). Lal *et al*. (2013) reported that although municipal wastewater irrigation increased the concentrations of Cd, Cr, Ni and Pb in lemongrass, it was still within permissible limits and the accumulation of heavy metals was not reflected in the essential oils extracted from the plants. Batarseh *et al*. (2011) studied the effect of treated municipal wastewater irrigation on the heavy metal uptake of olive trees in the Mediterranean country of Jordan. Even though the wastewater used for irrigation contained high amounts of Ni and Pb, the concentration of these elements in both the leaves and fruits were considerably low. In contrast, the wastewater had very low Cr levels, but significant quantities of Cr accumulated in the leaves and to a far lesser extent in fruits (Batarseh *et al*., 2011). These results do not only suggest that the uptake of heavy metals by olive trees is a selective process, but that it is often independent of the concentration of heavy metals in the applied wastewater.

1.3.2.4.3. Yield and biomass production

It has been reported that the high nutrient content (especially N and P) present in municipal wastewater can lead to an increase in yield and biomass production of crops under wastewater irrigation. Increased yields have been reported for olive (Charfi *et al*., 1999; Bedbabis *et al*., 2010;

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Bedbabis *et al*., 2015; Bourazanis *et al*., 2016), apple (Neilsen *et al*., 1989b), grape (Neilsen *et al*., 1989a; Mendoza-Espinosa *et al*., 2008), tomato (Cirelli *et al*., 2012), cucumber (Safi *et al*., 2018), lettuce (Zavadil *et al*., 2009; Urbano *et al*., 2017; Vergine *et al*., 2017), artichoke (Gatta *et al*., 2016), fennel (Lonigro *et al*., 2016; Vergine *et al*., 2017), cauliflower (Kiziloglu *et al*., 2008; Tripathi *et al*., 2016), rice (Jung *et al*., 2014), sunflower seeds (Papadopoulos & Stylianou, 1991) and lemongrass (Lal *et al*., 2013) under municipal wastewater irrigation. Similar results were reported for the biomass production of *Panicum maximum* grass (Abdoulkader *et al*., 2015), maize (Alkhamisi *et al*., 2011; El-Nahhal *et al*., 2013), barley (Rusan *et al*., 2007) and willow trees (Nissim *et al*., 2015). In contrast, lower yields as a result of wastewater irrigation were reported for alfalfa (Shahalam *et al*., 1998), cabbage (Balkhair *et al*., 2013), okra and aubergine (Balkhair *et al*., 2014) as well as snow peas and celery (Grewal & Maheshwari, 2013). Decreased yields were attributed to the accumulation of toxic elements in the stems and leaves of plants (Balkhair *et al*., 2014) and the salinization of the irrigated soil (Shahalam *et al*., 1998). Pedrero *et al*. (2012) compared the use of secondary treated municipal wastewater and a mix of tertiary treated municipal wastewater and fresh water for irrigation in a lemon tree orchard. They reported lower yields for trees under secondary treated wastewater irrigation and greater yields under the mixed irrigation water. The reduction in yield was probably caused by the salinity induced by the lesser treated wastewater. In numerous studies, crops irrigated with municipal wastewater did not have significantly different yields compared to crops irrigated with fresh water (McCarthy, 1981; Aiello *et al*., 2007; Segal *et al*., 2011; Martínez *et al*., 2013; Netzer *et al*., 2014; Ganjegunte *et al*., 2017; Libutti *et al*., 2018), suggesting that municipal wastewater may not adversely affects plant growth. Moreover, due to the nutrient supply through wastewater, similar yields could be obtained without the application of additional fertilisers. Other studies have indicated greater yields and biomass production of crops irrigated with municipal wastewater in comparison with crops produced under dryland conditions (Wang *et al*., 2007; Ayoub *et al*., 2016; Gonçalves *et al*., 2017). Therefore, the use of municipal wastewater for irrigation in water scarce regions may significantly increase crop productivity when no alternative water sources are available.

1.3.2.4.4. Crop and product quality

The quality of crops and their processed products can be assessed in terms of their general qualitative characteristics, which is dependent on the specific crop in question (*e.g.* phenolic composition of oils, juice characteristics of fruits & physical appearance of cut flowers). The irrigation of olive trees in semi-arid regions using treated municipal wastewaters have been the subject of many research studies in recent years. Gharsallaoui *et al*. (2011) investigated the effect of secondary treated municipal wastewater irrigation on the oil quality of Chemlali olives in Tunisia. It was concluded that the number of polyphenols in olive oil was enhanced in comparison with well water irrigated plots, but the olives were much more sensitive to oxidation after harvest. In agreement with these findings, Bedbabis & Ferrara (2018) reported increased total phenols as well as free acidity in Chemlali olives that were irrigated with treated municipal wastewater for 10 years. However, earlier studies on the same cultivar (Bedbabis *et al*., 2009; Bedbabis *et al*., 2015) as well as Koroneiki (Bourazanis *et al*., 2016) showed no such effects.

Similar to that reported by Morgan *et al*. (2008) for oranges, the results of a comparative three-year study on lemon under secondary and tertiary treated municipal wastewater irrigation (Pedrero *et al*., 2012), showed increased levels of greater titratable acidity (TA) and total soluble solids (TSS) in the extracted juice. This could be explained by some plants' adaptability to salinity. Under these conditions, increasing their production of secondary metabolites including organic acids, proteins and sugars, enhances the quality and market value of the final product (Pedrero *et al*., 2012). Koo and Zekri (1989) reported lower TA and TSS for Hamlin and Valencia oranges under municipal wastewater irrigation compared to fresh water and ascribed this decrease to higher soil water content due to the application of excessive amounts of wastewater irrigation.

Neilsen *et al*. (1989a) reported an increase in the must pH and TSS of Riesling grapes that were irrigated with treated municipal wastewater. However, TA was not affected, and it did not limit the production of high quality wine. In another study, Mendoza-Espinosa *et al*. (2008) reported no significant difference in the TA, TSS and pH of Cabernet Sauvignon and Merlot grapes that were irrigated with either secondary treated municipal wastewater or groundwater. McCarthy and Downton (1981) reported elevated concentrations of N, P, K^+, Na^+, Cl^- and Mg^{2+} in wines produced from Shiraz grapes that were irrigated with municipal wastewater compared to those under fresh water irrigation. The wines produced from wastewater irrigated grapevines also exhibited higher anthocyanin and phenolic contents. Although these wines had very high K^+ concentrations as well as a high pH, these levels were considered acceptable for red wine production according to Australian standards (McCarthy & Downton, 1981).

Hasan *et al*. (2014) investigated the effect of municipal wastewater irrigation on the quality of *Gladiolus communis* flowers. The influence of irrigation with 100% potable water, 100% municipal wastewater and different ratios of the two distinct water sources were compared. It was found that utilising either potable or municipal wastewater only, resulted in flowers of lower quality. Plants irrigated with a combination of 75% potable water and 25% wastewater had the longest spikes, larger spike diameter and more cormels per corm.

The quality of crops irrigated with municipal wastewater can also be evaluated in terms of their microbial safety as it is one of the most important factors affecting the use of wastewater for crop irrigation (Chen *et al*., 2013a). Libutti *et al*. (2018) observed greater numbers of FC on the plant itself, compared to the crop products of tomato and broccoli that were irrigated with treated municipal wastewater. Yet, the numbers recorded did not differ from those recorded for fresh water irrigated tomatoes and broccoli (Libutti *et al*., 2018). Similar results have been reported elsewhere for tomatoes (Shahalam *et al*., 1998; Aiello *et al*., 2007; Forslund *et al*., 2012; Gatta *et al*., 2015;

Lonigro *et al*., 2016; Orlofsky *et al*., 2016) as well as lettuce (Urbano *et al*., 2017; Intriago *et al*., 2018), artichokes (Gatta *et al*., 2016), strawberries (Christou *et al*., 2016) and lemons (Pedrero *et al*., 2012). It is important to note that the aforementioned studies investigated the use of (in most cases, tertiary) treated municipal wastewater rather than untreated municipal wastewater. Studies that investigated the use of untreated or partially treated municipal wastewater as an irrigation source for vegetables reported greater microbial contamination (Rosas *et al*., 1984; Minhas *et al*., 2006; Rai & Tripathi, 2007). Cirelli *et al*. (2012) reported significant microbial contamination of tomatoes under tertiary treated municipal wastewater irrigation only where fruits were in direct contact with the soil or plastic mulch. Recommendations were made by Minhas *et al*. (2006) to minimise the microbial contamination of vegetables grown under municipal wastewater irrigation, including (i) avoiding direct contact between wastewater and vegetables; (ii) exposing vegetables to adequate amounts of sunlight to enhance the die-off of pathogens and (iii) removing the outer layers of vegetables exposed to wastewater where possible (*e.g.* the outmost leaves of cabbage).

1.3.3. Winery wastewater

1.3.3.1. Volumes of wastewater generated during winemaking

Information regarding the volumes of water used by wineries tends to be scarce and inconsistent (Howell & Myburgh, 2018). A survey conducted by Sheridan *et al*. (2005) revealed that the volume of raw water used by wineries significantly increased with the amounts of grapes crushed. Their report indicated that approximately 2 m^3 of raw water is used to crush one tonne of grapes. Gabzdylova *et al*. (2009) reported that 2–3 m^3 of raw water is needed to process one tonne of grapes. In comparison, Howell & Myburgh (2018) estimated that the Lutzville Vineyards winery uses approximately 2.1 m^3 of raw water to process one tonne of grapes. According to South African Wine Industry Information and Systems (SAWIS) (2017), an annual average of 1.43 million tonnes of grapes were crushed from 2015 to 2017. If the value reported by Sheridan *et al*. (2005) is correct, the South African wine industry may require up to 2.86 million litres of raw water annually.

Like the usage of raw water, information on the volumes of wastewater produced by wineries are limited and highly variable. According to Van Schoor (2005) and others cited therein, medium to large wineries may produce wastewater in excess of 15 000 m^3 each year, whereas smaller wineries generate less than 15 000 m^3 annually. The Spanish wine industry produces approximately 18 x 10^6 m^3 of wastewater annually, which is six times more than the French or Italian wine industries (Bustamante *et al*., 2005). According to Gabzdylova *et al*. (2009), New Zealand wineries generate *ca.* 7.5 ℓ of wastewater per 750 mℓ bottle of wine, which equates to *ca.* 3.8 x 10^5 m^3 of wastewater annually. Others have estimated that 0.2–4 ℓ of wastewater is generated in the production of a litre of wine (Welz *et al*., 2016 and references therein). Kumar *et al*. (2006) reported that wineries generate 3–5 m^3 of wastewater per tonne of grapes crushed,

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whereas Oliviera and Duarte (2016) reported that one tonne of crushed grapes produced approximately 1.65 m^3 of wastewater at Portuguese cellars. The Lutzville Vineyards winery produces *ca*. 1.1 m^3 of wastewater per tonne of grapes crushed (Howell *et al*., 2015). However, it is estimated that 50% of the wastewater at this winery is lost to evaporation (Kriel, 2008).

Different winemaking techniques can also alter the volume of wastewater produced by wineries. It was reported that French wineries generate 0.359 m^3 of wastewater by off-skin white wine production and 0.357 m^3 of wastewater by rosé and thermo-vinification of red wines, per tonne of grapes crushed (Bories & Sire, 2010). Crushing one tonne of grapes generated a fairly low volume of 0.262 m^3 of wastewater when on-skin solid-phase vinification of red wines were implemented (Bories & Sire, 2010). It is estimated that brandy distillation in California produces 0.76 m^3 of wastewater per tonne of grapes crushed (Vaugh & Marsh, 1953). The volumes of winery wastewater produced vary throughout the year in relation to the different processes taking place in the cellar (Arienzo *et al*., 2009b). Various stages of winemaking activities and their associated effect on wastewater generation is presented in Table 1.3. The majority of wastewater generated by wineries is a result of cleaning operations within wineries (Van Schoor, 2005), and it usually takes place during the harvest period (Buelow *et al*., 2015b; Langinestra, 2016) which is typically between January and April in the southern hemisphere.

Table 1.3. Typical wine production stages and their relation to wastewater generation (Van Schoor, 2005 and references therein).

Stage	Winemaking activities	Duration (weeks)
1. Pre-harvest	Bottling takes place and tanks are washed out with Na or K hydroxide. Other equipment is also washed in preparation of the harvest period.	1–4
2. Early harvest	Wastewater generation increases drastically during this period and reaches 40% of the maximum weekly rate measured at peak. White wine production dominates harvest activities.	2–3
3. Peak harvest	Wastewater generation and harvest activities reach their peak.	3–14
4. Late harvest	Wastewater generation decreases to 40% of the maximum weekly flow and red wine production dominates harvest activities. Distillation of ethanol may take place.	2–6
5. Post-harvest	Pre-fermentation activities come to a close and maximum usage of hydroxide occurs.	6–12
6. None harvest	Wastewater volume is at its minimum (< 30% of the peak weekly flow). Wastewater quality depends on daily activities.	10–20

1.3.3.2. Quality of wastewater generated by wineries

In contrast to the availability of information on volumes of winery wastewater generated, there are many reports on the quality thereof (Howell & Myburgh, 2018). Although wine production is not generally considered to be a polluting industry, winery wastewater usually has a high organic load, low pH, variable salinity and variable nutrient levels, all of which can have a potentially negative impact on the receiving environment (Mosse *et al*., 2011). As mentioned previously, winery wastewater is mainly produced through cleaning processes and is therefore primarily composed of wine, grape juice, suspended solids (during harvest period) and cleaning agents (*e.g.* sodium hydroxide [NaOH] & potassium hydroxide [KOH]) (Mosse *et al*., 2011).

Numerous parameters may be used to evaluate the quality of winery wastewater, but pH, EC, SAR, K^+, Na^+, Cl^- and COD are considered to be the most important (Howell & Myburgh, 2018). Winery wastewater is usually acidic, and the pH can be below 3 (Howell & Myburgh, 2018). Therefore, many wineries add lime ($CaCO_3$) to wastewater to increase pH levels in line with the legal or crop requirement (Van Schoor, 2005). A recent review reported a pH range of 2.5 to 12.9 for winery wastewater (Ioannou *et al.*, 2015 and references therein), whereas, Mosse *et al*. (2011) and others cited therein, reported a range of 3 to 12. Many winery wastewaters have a high pH due to the use of alkaline products in cleaning processes (Bustamante *et al*., 2005). Mahajan *et al*. (2009) reported a pH increase of two units when comparing winery wastewater samples collected during harvest to samples collected during the post-harvest period. As for EC_w, ranges of 0.8 to 3.1 dS/m (Mosse *et al*., 2011 and references therein) and 0.92 to 2.31 dS/m (Oliviera & Duarte, 2016) have been reported. The annual mean monthly EC_w of wastewater sampled at a winery in the Breede River region ranged between 0.7 dS/m and 2.2 dS/m (Howell *et al*. 2016b). The SAR of winery wastewater can range from 6.5 to 15 (Laurenson & Houlbrooke, 2012 and references therein). A range of 0.33 to 33.1 was reported by Mulidzi *et al*. (2009b), whereas Howell *et al*. (2016b) measured an annual range of between 2.4 and 9.0 at a winery in the Breede River region.

Winery wastewater has inherently high K^+ concentrations due to the high concentrations of K^+ present in grape juice and wine (Mosse *et al*., 2011). Concentrations ranging between 29 and 353 mg/ℓ (Bustamante *et al*., 2005) and 3 and 410 mg/ℓ (Laurenson & Houlbrooke, 2012 and references therein) have previously been reported. Sheridan *et al*. (2011) studied the effluent parameters of two cellars near Stellenbosch and reported that the K^+ concentrations at both wineries ranged between approximately 20 mg/ℓ and 220 mg/ℓ. Howell *et al*. (2016b) reported a greater range for a winery in the Breede River region (44 mg/ℓ to 506 mg/ℓ). Sodium based cleaning products (particularly NaOH) can lead to an appreciable accumulation of Na^+ in winery wastewater (Mosse *et al*., 2011; Sheridan *et al*., 2011; Laurenson & Houlbrooke, 2012; Conradie *et al*., 2014; Howell *et al*., 2016b). Concentrations of Na^+ in winery wastewater may range from 173 mg/ℓ to 238 mg/ℓ (Laurenson *et al*., 2012). Bustamante *et al*. (2005) reported a range of between 7 mg/ℓ

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and 470 mg/ℓ for wineries in Spain. Howell *et al*. (2016b) reported annual mean monthly Na^{+} concentrations between 76 mg/ℓ and 224 mg/ℓ at a winery in the Breede River region. Far fewer studies have investigated Cl^{-} concentrations in winery wastewater. Buelow *et al*. (2015b) reported Cl^{-} concentrations in untreated winery wastewater ranging from 3.35 mg/ℓ to 143 mg/ℓ at wineries in California. Following physicochemical and biological treatment, the range was 2.79 mg/ℓ to 115 mg/ℓ. Other inorganic constituents often found in winery wastewater include heavy metals, N, P (Laurenson & Houlbrooke, 2012), as well as Ca^{2+} and Mg^{2+} (Mosse *et al*., 2011). Estimates of the amount of N and P present in wastewater generated at small wineries are presented in Table 1.4. Welz *et al*. (2016) reported N values (compromised by different ratios of NH_3/NH_4^{+} and NO_3^{-}) of up to 176 mg/ℓ at a winery that generates approximately 5 000 m^3 of wastewater annually. Similarly, Vlyssides *et al*. (2005) compared the total N and P content of wastewater produced from red and white wine production, reporting 71 mg/ℓ and 67 mg/ℓ of N and 8.5 mg/ℓ and 7 mg/ℓ of P, respectively. Ranges of 10 mg/ℓ to 415 mg/ℓ and 2.1 mg/ℓ to 280 mg/ℓ have been reported for total N and total P, respectively (Ioannou *et al*., 2015 and references therein).

Table 1.4. Estimated nitrogen (N) and phosphorous (P) loadings of untreated wastewater from small wineries (adapted from Haran-Smith & Gibberd, 2009).

Period	Tonnes grapes crushed	N in winery wastewater (kg)	P in winery wastewater (kg)
Harvest	100	24.5	7
	300	73.5	21
	500	122.5	35
Pre-/ post-harvest	100	3.75	1.5
	300	11.25	4.5
	500	18.75	7.5

Values for COD vary greatly as a result of the different organic constituents present in winery wastewater. According to Ruggieri *et al*. (2009), the majority of wastes produced in cellars are organic, therefore high COD values are to be expected in winery wastewaters. The COD of winery wastewater can range between 320 mg/ℓ to 296 119 mg/ℓ (Mosse *et al*., 2011 and references therein). A more recent study reported a COD range of 320 mg/ℓ to 49 105 mg/ℓ (Ioannou *et al*., 2015 and references therein). In a survey of ten different South African wineries, the COD values ranged from 3 370 mg/ℓ for a winery near Paarl to 47 024 mg/ℓ for a winery in the Olifants River region (Mulidzi *et al*., 2009b). In a year-long study in Stellenbosch at wineries that store wastewater in settling basins before irrigation, Welz *et al*. (2016) reported a COD range of 28 mg/ℓ to 7 265 mg/ℓ (average: 905 mg/ℓ) at one winery and a range of 675 mg/ℓ to 76 900 mg/ℓ (average: 10 906 mg/ℓ) at another. Whereas, the wastewater sampled at wineries using waste stabilisation ponds exhibited COD concentrations of 470 mg/ℓ to 13 730 mg/ℓ, with an average of approximately 5 000 mg/ℓ (Welz *et al*., 2016). The annual mean monthly COD levels of wastewater at a winery in the Breede River region ranged from 1 815 mg/ℓ to 13 286 mg/ℓ (Howell *et al*., 2016b). In a study

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conducted by Mosse *et al*. (2012), the COD of untreated winery wastewater decreased from 13 000 mg/ℓ to 610 mg/ℓ following treatment by means of a sequencing batch reactor. Many studies have reported values of BOD which is estimated as 66% of the COD (Van Schoor, 2005). The variability of winery wastewater quality is a result of differences in winemaking techniques and processes, volumes of water used as well as the overall winery design (Mosse *et al*., 2011). It is also possible that spikes of low quality wastewater can be caused by process interruptions (Howell & Myburgh, 2018). These include power failures, fires, floods, storms, over- or underloading of treatment systems, temporary unavailability of wastewater holding dam capacity and the absence of trained personnel (Howell & Myburgh, 2018 and references therein). Irrigation with winery wastewater is subject to the same legal requirements as municipal wastewater (Table 1.1) and the intended water use has to be registered with the DWA (Van Schoor, 2005).

1.3.3.3. Origins of winery wastewater

Virtually all the steps of the winemaking process require water inputs (Conradie, 2015). Wastewater is thus generated from the receival of grapes at the cellar, to bottling of the final product (Devesa-Rey *et al*., 2011). The four major components that contribute to wastewater production in a winery are (i) sub-product residues, such as stems, skins, sludge, lees and tartar; (ii) product loss, *e.g.* spillage of wine and must during winemaking processes; (iii) products used for the treatment of wine, *e.g.* fining agents and filtration earths; as well as (iv) cleaning and disinfecting agents, such as NaOH and/or KOH used to wash materials and equipment (Conradie *et al*., 2014 and references therein). Table 1.5 summarises typical winemaking processes related to winery wastewater production and their contribution towards wastewater quantity and quality, as well as possible effects on the legal wastewater quality parameters.

The organic matter present in winery wastewater arises mainly from the grapes and wine (Mosse *et al*., 2011). Grape marc (skins & pips) is produced after grapes are destemmed and pressed (Devesa-Rey *et al*., 2011). Even though grape marc is kept separated from the wastewater reticulation system, residues on cellar floors and in the grape press can contribute to higher COD levels and variations in pH (Van Schoor, 2005). Lees that form on the bottom of wine tanks and barrels following fermentation can also contribute to the organic load of winery wastewater (Conradie, 2015). Colin *et al*. (2005) and Sheridan *et al*. (2011) reported that ethanol contributes largely to the COD of winery wastewater. Specific winemaking methods as well as grape varietals can affect the quality of winery wastewater (Vlyssides *et al*., 2005; Welz *et al*., 2016). Bories and Sire (2010) reported that off-skin winemaking techniques (as used in the production of white and rosé wines), generate wastewaters that are rich in sugars, while classical winemaking techniques (as used for red wine production) generate wastewaters with high levels of ethanol. Other organic compounds present in winery wastewater include organic acids (acetic, tartaric, malic, lactic, propionic), esters and polyphenols (Mosse *et al*., 2011). High concentrations of sugars (glucose, fructose & maltose) may also be prevalent during the harvest period (Welz *et al*., 2016).

Table 1.5. Primary winemaking processes related to winery wastewater quantity and quality and possible effects on legal wastewater quality parameters (Van Schoor, 2005).

Winery operation	Contribution to total wastewater quantity	Contribution to wastewater quality	Effect on legal wastewater quality parameters
Cleaning water			
Alkali washing (removal of K-bitartrate) and neutralisation	Up to 33%	Increase in Na^+, K^+, COD, pH Decrease in pH	Increase in EC, SAR, COD Variation in pH
Rinse water (tanks, floors, transfer lines, bottles, barrels *etc.*)	Up to 43%	Increase in Na^+, P, Cl^-, COD	Increase in EC, SAR, COD Variation in pH
Process water			
Filtration with filter aid	Up to 15%	Various contaminants	Increase EC and COD
Acidification and stabilisation of wine	Up to 3%	H_2SO_4 or NaCl	Increase EC and COD Decrease in pH
Cooling tower waste	Up to 6%	Various salts	Increase EC and COD
Other sources			
Laboratory practices	Up to 5% to 10%	Various salts, variation in pH, *etc.*	Increase EC and COD

Inorganic constituents in winery wastewater are highly dependent on the composition of cleaning products used in cellars, however high K^+ concentrations are present as a result of high concentrations in grape juice (Arienzo *et al*., 2009a; Mosse *et al*., 2011). A variety of cleaning agents are used in cellars and include NaOH, KOH, sodium metasilicate (Na_2SiO_3), trisodium phosphate (Na_3PO_4), sodium carbonate (Na_2CO_3), acids such as phosphoric acid (H_3PO_4) and peracetic acid compounds, quaternary ammonium compounds, hydrogen peroxide (H_2O_2), ozone and sulphur (S) compounds (Welz *et al*., 2016). Cleaning agents are expected to contribute large amounts of Na^+ and K^+ to the inorganic fraction of wastewater, but some PO_4^{3-} and NH_3 or NH_4^+ may also be present in cleaning products (Welz *et al*., 2016). Lesser amounts of Ca^{2+} and Mg^{2+} may also be present in winery wastewater as they are naturally present in grape juice (Mosse *et al*., 2011). Some metals such as aluminium (Al^{3+}), Cu^{2+} and Pb can be present in winery wastewater due to the exposure of water to the surface of metal piping, tanks, soldering and brass fittings (Kumar *et al*., 2009). The organic and inorganic constituents of winery wastewater do not only vary with differences in winemaking procedures over time, but also between individual wineries (Mosse *et al*., 2012).

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1.3.3.4. Effect of irrigation with winery wastewater on soil properties

1.3.3.4.1. Chemical properties

pH: Irrigating soils with acidic winery wastewater may result in a reduction of soil pH (Laurenson *et al*., 2012). However, some studies have reported an increase in soil pH where winery wastewaters with a low pH was applied (Mulidzi *et al*., 2015; Shilpi *et al*., 2018). A pot experiment conducted by Mulidzi *et al*. (2015) investigated the effects of winery wastewater irrigation on the pH of four differently textured soils commonly found in the Western Cape. Over the course of four simulated irrigation seasons the $pH_{(KCl)}$ of all the soils increased, despite the wastewater having a slightly acidic pH (ranging from 4.9 to 6.0). The increased soil pH was attributed to the decarboxylation and hydrolysis of organic/bicarbonate anions arising from the organic salts present in the wastewater (Mulidzi *et al*., 2015). Similarly, the $pH_{(H2O)}$ of a silty clay loam soil that received winery wastewater for *ca.* 30 years was higher than adjacent soils that were unirrigated (Mosse *et al*., 2012). Unfortunately, the pH of the wastewater was not reported for this study. In contrast, Quale *et al*. (2010) reported that irrigation with winery wastewater had no significant effect on the pH of soil with a clay content between 50% and 60%. Hirzel *et al*. (2017) reported similar results for two vineyard soils in California. Variations in the pH of winery wastewater throughout the season can also affect the pH of receiving soils. In a study where winery wastewater was irrigated onto land, Mahajan *et al*. (2009) reported a lower soil pH during the harvest period (pH 4) when compared to the post-harvest period (pH 6).

Electrical conductivity: The high salt content of winery wastewater has been shown to increase the EC_e of irrigated soils (Kumar *et al*., 2006; Mahajan *et al*., 2009; Quale *et al*., 2010; Gray, 2012; Mosse *et al*. 2012; Hirzel *et al*., 2017; Howell *et al*., 2018; Shilpi *et al*., 2018). Conversely, Kumar *et al*. (2014) reported a decrease in EC_e over time at pasture and woodlot sites that were irrigated with winery wastewater. In a pot experiment that simulated field conditions, soil EC_e was unaffected by irrigation with either potable water, municipal wastewater or winery wastewater, regardless of soil type (Laurenson, 2010).

Organic matter and organic carbon: Irrigating soil with winery wastewater rich in OC may increase the TOC of the soil (Kumar *et al*., 2006; Kumar *et al*., 2009). However, a study by Quale *et al*. (2010) showed that winery wastewater irrigation had no significant effect on the TOC of soil, despite the wastewater having high TOC. Howell *et al*. (2018) conducted a field trial where a sandy, alluvial vineyard soil in the Breede River region was irrigated with winery wastewater diluted up to a COD of 3 000 mg/ℓ. Their results indicated inconsistent trends with regards to soil OC as affected by the dilution of winery wastewater. This could be a result of insufficient OC levels in the wastewater to affect soil fertility or the OM present in the wastewater, which could have increased TOC, decomposed upon aeration between irrigation applications Howell *et al*. (2018). Mulidzi (2001) reported that many highly permeable soils may not be suitable for irrigation with winery

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wastewater rich in OM as it is not retained in the soil and may lead to the pollution of natural water resources.

Nitrogen: Very few studies have investigated the effect of winery wastewater on soil N levels. Nevertheless, Kumar *et al*. (2006) and Hirzel *et al*. (2017) reported higher soil N concentrations in soils irrigated with winery wastewater compared to the control. Results of a pot experiment performed in a glasshouse by irrigating crops with various dilution ratios of winery wastewater indicated that soil N was increased up to a dilution ratio of 1:1 winery wastewater to tap water, where after soil N decreased as the amount of wastewater in the dilution increased (Shilpi *et al*., 2018). The seasonal variation of N levels in winery wastewater may also have an impact on the N levels of soil onto which it is irrigated. Mahajan *et al*. (2009) reported slightly higher soil N during the harvest period than during the pre-/post-harvest period when soils were irrigated with winery wastewater throughout the year.

Phosphorous: Information concerning the effect of winery wastewater irrigation on soil P levels are scarce and inadequate. According to Mulidzi *et al.* (2009a) land application of undiluted winery wastewater increased soil P levels, however seasonal P fluctuations were observed in the different soil horizons. In a more recent study, Mulidzi *et al*. (2016) investigated the effect of irrigation with diluted winery wastewater on the chemical properties of four soils with different parent material and clay content. The $pH_{(KCl)}$ of both shale- and granite-derived soils increased to the optimum range of P availability as a result of winery wastewater irrigation. Initially, the $pH_{(KCl)}$ of the aeolian sand was above the optimum range, but relatively high Na^+ levels increased the available P as the $pH_{(KCl)}$ increased. Available P decreased in the alluvial sand as $pH_{(KCl)}$ increased beyond the optimum range. The results indicate that P availability may only be enhanced by winery wastewater irrigation if the shift in $pH_{(KCl)}$ is towards the optimum range. It is, however, important to note that this study was performed in the absence of rainfall or crops, therefore it represents a worst-case scenario.

Potassium: Irrigation with K-rich wastewater can enhance overall soil fertility (Smiles & Smith, 2004; Mosse *et al*., 2011). According to Arienzo *et al*. (2009b), soluble and exchangeable forms of K^+ can be increased more rapidly by irrigating with wastewater than by applying conventional inorganic fertilisers to soil. The preferential use of winery wastewater as a source of K^+ over conventional fertilisers can also serve as an efficient recycling strategy in areas where soils exhibit K^+ deficiencies (Howell & Myburgh, 2018). Numerous authors have reported increased levels of soil K^+ as a result of winery wastewater irrigation (Kumar *et al*., 2006; Kumar *et al*., 2009; Quale *et al*., 2010; Gray *et al*., 2012; Mosse *et al*., 2012; Howell *et al.* 2018; Mulidzi *et al*., 2015; Hirzel *et al*., 2017). Mosse *et al*. (2013) conducted a field study where grapevines in an established vineyard were irrigated with either lake water (control) or artificial winery wastewater that had levels of high K^+, high K^+ plus wine, low K^+ or Na^+. Soil K^+ and Na^+ increased consistently with the amount of salts applied. Potassium accumulated primarily in the topsoil, however, the addition of

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wine to the irrigation water enhanced K^+ transport to deeper layers. It is likely that the presence of dissolved organic compounds in the wine facilitated the transport of K^+ to the subsoil. Caution should be taken when irrigating with K-rich winery wastewater as the application of excessive amounts of K^+ can influence the uptake of other nutrients, such as Ca^{2+} and Mg^{2+} (Morris & Cawthon, 1982).

It should also be noted that K^+ can have soil dispersal effects similar to Na^+ due to its large hydrated ion size and affinity to clay minerals (Levy & Feigenbaum, 1996; Arienzo *et al*., 2009a; Laurenson *et al*., 2010a). High levels of K^+ in winery wastewater can therefore have deleterious effects on soil structure, which will be discussed in section 1.3.3.4.2. Some studies have indicated that soils generally exhibit a greater binding affinity towards K^+ when equal amounts Na^+ and K^+ are present in the irrigation water (Laurenson, 2010; Laurenson *et al*., 2011; Mosse *et al*., 2013). This could possibly be explained by the presence of clay minerals (*e.g.* illite) that contain specific bindings sites for K^+ within its structural layers (Laurenson *et al*., 2010a). As Na^+ normally has greater effects on soil dispersion when compared to K^+, the presence of K^+ in winery wastewater may facilitate the leaching of Na^+ and subsequently limit the effects of dispersion (Laurenson & Houlbrooke 2011). Therefore, it might be advisable for wineries to shift from Na-based cleaning agents to K-based products (Laurenson & Houlbrooke 2011).

Calcium: In general, the amount of Ca^{2+} present in winery wastewater is substantially less when compared to other elements. Therefore, irrigation with winery wastewater rarely has an impact on the levels of soil Ca^{2+}. Mulidzi *et al*. (2015) and Howell *et al*. (2018) reported that irrigation with winery wastewater diluted up to 3 000 mg COD/ℓ had little or no effect on soil Ca^{2+} due to small amounts present in the wastewater. Similarly, Quale *et al*. (2010) reported fairly constant soil Ca^{2+} levels for a clay soil over four consecutive irrigation seasons. Other studies have indicated a build-up of Ca^{2+} in the soil as a result of long-term winery wastewater irrigation. Kumar *et al*. (2006) reported that pastures irrigated with winery wastewater for a century had significantly greater Ca^{2+} levels compared to controls. An accumulation of Ca^{2+} was also observed in a silty clay loam soil that received winery wastewater for 30 years (Mosse *et al*., 2012). Where winery wastewater contains significant amounts of Ca^{2+} it can be beneficial in reducing SAR and PAR and therefore ameliorate risks associated with clay dispersal (Laurenson, 2010).

Magnesium: Similar to Ca^{2+}, Mg^{2+} is also present in winery wastewater at low concentrations and its impact on soil Mg^{2+} levels are often negligible (Laurenson, 2010; Gray, 2012; Howell *et al*., 2018; Mulidzi *et al*., 2015). In a field study where a vineyard was irrigated with winery wastewater for *ca.* 21 years, no significant difference in soil Mg^{2+} concentrations were observed when compared to a vineyard soil that received only well water (Hirzel *et al*., 2017). The authors attributed the lack of Mg^{2+} accumulation to high Na^+ levels that caused displacement of divalent cations in the soil. In contrast, Kumar *et al*. (2006) reported substantially increased soil Mg^{2+} as a result of a 100 years of winery wastewater irrigation. Mosse *et al*. (2012; 2013) also reported increases in soil Mg^{2+} due irrigation with Mg-rich winery wastewater. Conversely, Quale *et al*. (2010) reported reduced soil Mg^{2+} concentrations following four years of winery wastewater irrigation.

Sodium: The accumulation of excessive amounts of exchangeable Na^{+} in soils may result in a breakdown of soil aggregates and lead to changes in numerous soil physical properties including *K*, infiltration rate and soil aeration (Laurenson *et al*., 2010b). Irrigation with winery wastewater has been shown to increase soil Na^{+}. Kumar *et al*. (2009) reported elevated levels of Na^{+} in vineyard and pasture soils due to application of Na-rich winery wastewater. Similar results were reported by Gray (2012), Hirzel *et al*. (2017) and Howell *et al*. (2018). In a pot study where four differently textured soils were irrigated with diluted winery wastewater, the risk of Na^{+} accumulation increased linearly with an increase in clay content (Mulidzi *et al*., 2015). Likewise, Na^{+} accumulations were observed in surface and subsurface soils that were irrigated with Na-based artificial winery wastewater (Mosse *et al*., 2013). A study by Mosse *et al*. (2012) revealed that although soil Na^{+} levels increased after 30 years of winery wastewater irrigation, the accumulation was relatively small considering the high concentration of Na^{+} present in the wastewater. This could be explained by significant leaching of the applied Na^{+} from the soil profile. Winter rainfall in combination with over-irrigation with winery wastewater can lead to considerable Na^{+} leaching beyond soil depths of 90 cm (Mulidzi, 2016).

Chloride, trace elements and heavy metals: The effects of winery wastewater irrigation on soil Cl^{-}, trace element and heavy metal concentrations are not well documented. Mosse *et al*. (2012) reported appreciable increases in soil B^{3+}, Cu^{2+}, Fe^{2+} and Zn^{2+} as a result of long-term winery wastewater irrigation. Similar trends were reported for Mn^{2+}, but samples were highly variable. In contrast, in a field study near Rawsonville, Howell *et al*. (2018) could not relate soil Cl^{-}, Cu^{2+}, Fe^{2+} and Zn^{2+} concentrations to different levels to which winery wastewater was diluted. However, after three and four seasons in which diluted winery wastewater was used for irrigation, an increase in soil B^{3+} levels were observed up to a depth of 150 cm.

1.3.3.4.2. Physical properties

Hydraulic conductivity: Irrigating with winery wastewater that is rich in OM may lead to the clogging of soil pores (Mosse *et al*., 2011). However, no references in the literature could be sourced to support this statement. It is therefore assumed that chemical clogging of soil pores (as discussed in section 1.3.2.3.2.) is the primary concern related to the effect of winery wastewater irrigation on *K*. The influence of exchangeable Na^{+} on *K* has been widely documented (Laurenson *et al*., 2012) and it is generally accepted that soil *K* will continue to decline as ESP increases (Menneer *et al*., 2001). Research pertaining to the effect of K^{+} on soil *K*, however, is limited (Arienzo *et al*., 2009a). As K^{+} is often the most abundant cation present in winery wastewater, it is of the utmost importance to be aware of potential soil degradation risks associated with K^{+}. The effects of K^{+} on

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infiltration may be similar to Na^+ or Ca^{2+} (Arienzo *et al*., 2009a). Quirk and Schofield (1955) reported that K^+ and Na^+ have an equally deleterious effect on *K.* Chen *et al*. (1983) reported an improvement in *K* of two soils when EPP was close to 20, but it negatively impacted a third soil and EPP > 20 reduced *K* in all three soils. This was attributed to the differences in mineralogical properties between the soils. Buelow *et al*. (2015a) investigated the effect of solution SAR and PAR on the *K* of three soils dominated by either montmorillionite, vermiculite or kaolinite clays. They reported that vermiculite and kaolinite dominated soils showed greater reductions in *K* at PAR ≥4 due to the presence of minerals with high K-fixation potential. They recommended the use of K-based cleaning agents in wineries if soils where wastewater is to be applied, are dominated by montmorillionite clays. Arienzo *et al*. (2012) conducted a laboratory study where soils containing predominantly smectite was percolated with different SAR and PAR solutions. Reductions in *K* were observed across all the treatments, but in PAR solutions the reductions were significantly smaller compared to SAR solutions, indicating greater soil stability in the presence of K^+ relative to Na^+. Similar results were reported by Kumar *et al*. (2009). Levy and Van der Watt (1990) concluded that K^+ can neither be grouped with Ca^{2+} or Na^+ with regards to its effect on soil hydraulic properties, but it rather has an intermediate effect on such properties, between those of Ca^{2+} and Na^+. It is also known that the effect of SAR/PAR on soil dispersion is highly dependent on the electrolyte concentration of the percolating solution. Shainberg *et al*. (1981) reported that clay dispersion is unlikely to occur if the EC_w remains below a critical flocculation concentration (CFC). When both EC_w and SAR were relatively high, soil structure was unaffected, but a decrease in EC_w enhanced clay dispersion and decreased *K*. Winery wastewaters that are high in Na^+ or K^+ generally also have a high EC_w which can mitigate the effects of Na^+ and K^+ on *K* and aggregate stability (Arienzo *et al*., 2009a).

It should be noted that the abovementioned studies used laboratory techniques with disturbed or repacked cores and "artificial winery wastewater" to simulate the effects of winery wastewater irrigation on soil physical properties. Howell and Myburgh (2014), however, conducted a first-of-its-kind study in which the effect of diluted winery wastewater irrigation on the near-saturation hydraulic conductivity (K_{ns}) of four different soils were investigated in a vineyard setup. Following three years of irrigation, the results indicated substantial reductions in the K_{ns} of shale-derived, alluvial and aeolian soils as the level of winery wastewater dilution decreased, *i.e.* as the COD increased. However, it is important to note that these soils received no fresh water irrigation or rainfall and it was in the absence of crops that could possibly absorb excess amounts of K^+.

Infiltration rate: Only a single source could be found in the literature in which the effect of winery wastewater irrigation on soil infiltration rate was estimated. In a laboratory experiment three different soil types (loamy sand, clay loam & clay) obtained from sites irrigated with winery wastewater were repacked into soil columns and irrigated with either winery wastewater or water which was treated through reverse osmosis (Kumar *et al*., 2006). The lowest infiltration rates were observed in the clay soil, followed by the clay loam and finally the loamy sand. Control soils irrigated with reverse osmosis water had even lower infiltration rates due to the low ionic strength of the solution which facilitated soil dispersion and ultimately clogged soil pores. In a soil column study using three South African soils, Levy and Van der Watt (1990) reported a decrease in infiltration rate as K^+ in the exchangeable phase increased. This phenomenon was more pronounced in soils dominated by illite and less so in kaolinitic soils where iron oxides could have had a stabilisation effect. A furrow-irrigated clay loam soil (dominated by illite & chlorite) which was irrigated with water having either low EC_w and SAR, moderate EC_w and SAR or high EC_w and SAR showed significant reductions in infiltration rate when irrigated with solutions having moderate or high EC_w and SAR due to the formation of a hard-setting, apedal surface soil layer (Emdad *et al*., 2004).

Bulk density, *porosity, aggregate stability and water retention*: No information pertaining to the effect of winery wastewater irrigation on soil bulk density, porosity, aggregate stability or water retention could be sourced. However, Emdad *et al*. (2004) reported greater bulk density and reduced aggregate stability in a clay loam soil irrigated with medium- to high SAR (10–30) solution. In a silt loam soil where dairy factory effluent was irrigated onto pastures in New Zealand for over 22 years, a reduction in bulk density and increased porosity was observed (Sparling *et al*., 2001). Similarly, Hati *et al*. (2007) reported lower bulk density, increased porosity and enhanced soil water retention of a vertisol irrigated with distillery effluent in India. It should be noted that changes in soil physical properties are difficult to quantify as they usually only occur over the long term and they are often variable and difficult to measure accurately, repeatedly and precisely (Hawke & Summers, 2006).

1.3.3.5. Effect of irrigation with winery wastewater on agricultural crops

1.3.3.5.1. Plant water relations

Information regarding the effect of winery wastewater irrigation on plant water relations are very limited. Only a single reference on this subject could be sourced from the literature. A field trial was conducted in commercial vineyards in the Breede River region where grapevines planted on sandy alluvial soil were irrigated with river water or winery wastewater which was diluted to COD concentrations ranging between 100 mg/ℓ and 3 000 mg/ℓ (Howell *et al*., 2016a). Under the prevailing conditions, the authors reported that winery wastewater (regardless of the level of dilution) did not affect grapevine water status compared to a river water control. It should, however, be noted that winery wastewater diluted to 3 000 mg/ℓ COD had a low EC_w and the pH and SAR of the irrigation water was considered to be acceptable for vineyard irrigation under these circumstances.

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1.3.3.5.2. Plant chemical status

Nitrogen and phosphorous: Fourie *et al*. (2015) investigated the effect of diluted winery wastewater irrigation on *Avena sativa* L. *cv*. Palinup (oats) and *Pennisetum glaucum* L. *cv*. Babala (pearl millet) planted as interception crops in commercial vineyards during winter and summer, respectively. Nitrogen levels in oats seemed to vary between different dilution concentrations of wastewater, but no clear trends were observed. These results indicated that P levels of both crops were unaffected by the application of winery wastewater. Similarly, Howell *et al*. (2016a) reported that N and P levels in the leaf blades of Cabernet Sauvignon grapevines could not be related to different dilution levels of winery wastewater. This is most likely due to insignificant concentrations of N and P present in the wastewater (Howell *et al*., 2015).

Calcium and magnesium: Irrigation of agricultural crops with winery wastewater may have variable effects on crop Ca^{2+} and Mg^{2+} levels. In a field study of commercial Sauvignon blanc and Cabernet Sauvignon vineyards in California, winery wastewater irrigation did not affect the Ca^{2+} concentrations of grapevine leaves, but Mg^{2+} levels were enhanced by 12% to 50% compared to grapevines irrigated with well-water (Hirzel *et al*., 2017). In contrast, Howell *et al*. (2016a) reported reduced levels of Ca^{2+} and Mg^{2+} in the leaf petioles of Cabernet Sauvignon grapevines. The levels decreased as the level of winery wastewater dilution decreased and can possibly be explained by high K^+ levels in the wastewater having an antagonistic effect towards both Ca^{2+} and Mg^{2+}. When K-rich artificial winery wastewater was applied to established Shiraz vineyards, Ca^{2+} concentrations in leaf petioles decreased significantly, but Mg^{2+} levels were unaffected (Mosse *et al*., 2013). The effect of winery wastewater irrigation on the chemical status of oats and pearl millet, established as cover crops in a commercial vineyard, was assessed by Fourie *et al*. (2015) in a field trial with different dilution levels of winery wastewater. According to their results, different levels of winery wastewater dilution did not affect the Ca^{2+} levels of either crops or the Mg^{2+} levels of oats, however, Mg^{2+} concentrations of pearl millet varied between treatments, but no definite trends were observed. Conversely, Shilpi *et al*. (2018) reported enhanced Ca^{2+} and Mg^{2+} levels in irrigated sunflower and maize plants as dilution levels of winery wastewater decreased.

Potassium and sodium: As wineries are usually surrounded by vineyards, grapevines make logical recipients for wastewater irrigation (Mosse *et al*., 2011). Therefore, a number of recently published studies have investigated the effect of winery wastewater irrigation on the K^+ and Na^+ levels of grapevine petioles and leaves. Compared to a control treatment irrigated with well-water, the leaf tissue of Sauvignon blanc and Cabernet Sauvignon grapevines irrigated with winery wastewater exhibited reduced K^+ levels and minor accumulations of Na^+ that would not be detrimental to grapevine health (Hirzel *et al*., 2017). Shiraz grapevines irrigated with Na-rich simulated winery wastewater accumulated greater amounts of Na^+ in leaves and petioles compared to control grapevines irrigated with lake water (Mosse *et al*., 2013). When irrigated with simulated winery wastewater containing either low or high K^+ concentrations, Shiraz petioles had greater K^+ levels compared to the control. Grapevines irrigated with a solution high in K^+ mixed with wine did not respond in the same way, suggesting that the organic matter present in wine may reduce plant available K^+ (Mosse *et al*., 2013). Similarly, Howell *et al*. (2016a) reported that the K^+ concentrations in petioles of Cabernet Sauvignon leaves were largely unaffected by winery wastewater irrigation, despite high levels of K^+ in the 0–60 cm soil profile, indicating that additional K^+ supplied *via* wastewater was not taken up by plants. This could be attributed to winery wastewater being applied too late in the season as most of the K^+ is taken up by grapevines prior to véraison, with little uptake taking place from five weeks after harvest (Conradie, 1981). During one of the growing seasons, K^+ levels in petioles were slightly reduced, possibly due to competition from pearl millet planted between grapevine rows as a summer interception crop (Howell *et al*., 2016a). The same study showed that despite high amounts of Na^+ applied *via* the irrigation water, Na^+ levels in Cabernet Sauvignon petioles did not increase significantly.

Since winery wastewater often contains appreciable amounts of K^+ and Na^+, it is important to be aware of the relative sensitivity or tolerance of irrigated crops towards these elements. Myburgh and Howell (2014a) investigated the potential of halophytic fodder beet (*Beta vulgaris* L. *cv.* Brigadier) to absorb Na^+ if irrigated with water containing Na^+ levels that resemble that of winery wastewater. Their results indicated that fodder beet absorbed up to 38% of the Na^+ that was applied through the irrigation water. As both the leaves and tubers of the plant can be harvested for fodder, absorbed Na^+ will effectively be removed from the soil completely, making this plant a promising interception crop to limit Na^+ accumulation where winery wastewater is used for irrigation (Myburgh & Howell, 2014a). In addition, pearl millet has shown some ability to consume K^+ applied through application of winery wastewater (Fourie *et al*., 2015). As for grapevines, uptake of K^+ and Na^+ are largely influenced by rootstock and clone selection (Downton, 1977), therefore, selecting appropriate cultivars to receive winery wastewater may play an important role in ensuring long term sustainability of its reuse (Mosse *et al*., 2013).

Trace elements and heavy metals: Information regarding the effect of winery wastewater irrigation on the plant uptake of trace elements and heavy metals is very limited. Fourie *et al*. (2015) reported that the micro-element concentrations of oats and pearl millet was unaffected by winery wastewater irrigation. In a pot study with different plant species, the Fe^{2+} concentrations of phalaris (*Phalaris aquatica*) roots and Cr concentrations of phalaris and lucerne (*Medicago sativa*) roots were reduced following irrigation with Na-rich synthetic winery wastewater (Mosse *et al*., 2010). No responses in metal contents were observed for barley (*Hordeum vulgare*) or millet (*Pennisetum glaucum*). Schoeman (2012) reported that irrigation with different dilution levels of winery wastewater did not affect the As, Cr, Cd, Hg or Pb levels of Cabernet Sauvignon grape must as these metals were present in winery wastewater at very low concentrations.

1.3.3.5.3. Yield and biomass production

The presence of macronutrients in winery wastewater may be able to enhance yield and biomass production of irrigated crops. Irrigating pearl millet with diluted winery wastewater under field conditions increased the dry matter production (DMP) when compared to a river water control (Fourie *et al*., 2015). Similarly, the seed yield of soy beans and grain yield of wheat was enhanced by irrigating with distillery effluent compared to soy beans and wheat irrigated with fresh water (Hati *et al*., 2007). In contrast, Mosse *et al*. (2010) reported significant reductions in the root and shoot biomass production of barley, millet, lucerne and phalaris crops with an increase in winery wastewater concentration. Lucerne was particularly sensitive to winery wastewater irrigation, experiencing a sharp reduction in biomass between a winery wastewater concentration of 0% to 10% and little difference was observed with further increases in winery wastewater concentration. Shilpi *et al*. (2018) reported drastic reductions in the DMP of sunflower and maize roots and shoots as a result of irrigation with undiluted winery wastewater. Their research did, however, indicate that diluting winery wastewater with 75% fresh water could enhance the shoot DMP of sunflower and maize plants. In the only study of its kind, Cabernet Sauvignon grapevines irrigated with winery wastewater diluted up to 3 000 mg/ℓ COD displayed no adverse effects on cane mass, yield, berry mass or bunch mass when compared to grapevines irrigated with river water (Howell *et al*., 2016a). This research indicates that winery wastewater might be a viable alternative source of water for irrigating grapevines under the specific growing conditions of this study. The DMP of fodder beet was also unaffected by irrigation with Na^+ enriched water when compared to a control treatment irrigated with fresh water (Myburgh & Howell, 2014a).

1.3.3.5.4. Crop and product quality

As wineries are often surrounded by vineyards, most studies have focussed on the effect of winery wastewater irrigation on grape juice characteristics and wine quality. This is particularly important as the irrigation of grapevines with K-rich wastewater may lead to elevated K^+ concentrations in grape must which can result in wines with high pH and poor colour (Jackson & Lombard, 1993), reduced tartaric/malic acid ratio (Mpelasoka *et al*., 2003), as well as impact fermentation properties and other wine sensory attributes including taste, bitterness and sourness (Kumar *et al*., 2014). Where a commercial Sauvignon blanc vineyard in the Napa Valley of California was irrigated with either winery wastewater or well water, sensorial analysis of the wines indicated no significant differences between the two treatments, despite the winery wastewater having 55 times the K^+ content of the well water (Hirzel *et al.*, 2017). While the TSS of grape must was unaffected, increased pH and lowered TA was observed for wastewater irrigated grapes, which was attributed to a high EC_e (2.38 dS/m) and Na^+ accumulation (SAR = 21) in the soil. Howell *et al*. (2016a) reported that irrigation with winery wastewater augmented to different COD levels did not affect TSS and TA of Cabernet Sauvignon grapes compared to a river water control. However, the mean juice pH was reduced with a decrease in level of wastewater dilution (*i.e.* greater COD) and was

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linearly related to the amount of K^+ applied *via* the irrigation water as well as the mean juice K^+. Since the juice pH was below the threshold value of 3.8 at which wine colour might be affected (Kodur, 2011 and references therein), no negative impacts on wine colour was observed. Furthermore, sensorial analyses of the wines indicated no off-odours or off-tastes that could be associated with the wastewater, thus implying that winery wastewater irrigation did not negatively affect wine quality. In a parallel study, when the bunches were directly sprayed with diluted winery wastewater, off-odours of a wastewater-like nature and a decrease in the spicy character of the wine was observed (Schoeman, 2012). In a field study where an established Shiraz vineyard was irrigated with different types of simulated winery wastewater, Mosse *et al*. (2013) reported wastewaters containing high K^+ concentrations and wine produced the lowest juice K^+ when compared to high Na^+ and high K^+ wastewaters. Further analyses of the must at harvest indicated that TA, TSS, pH and anthocyanin content was not affected by irrigation with simulated winery wastewater.

1.3.4. Conclusions

Large volumes of municipal and winery wastewater are produced each year. As water is a scarce resource, the reuse of wastewater for irrigation may present several advantages for the agricultural sector, including recycling of essential plant nutrients, fertiliser savings, addition of organic material, reduced pressure on fresh water sources through water savings and reduced environmental contamination. However, high salt loads, with specific reference to Na^+ and K^+, can have detrimental effects on soil physical properties, crop yields and quality. It is therefore essential to implement measures to limit damages caused by salinity. The current drought situation in the Western Cape has forced producers to find alternative irrigation water sources to sustain crop yield and quality. Since grapevines are an important agricultural crop in the Western Cape, a study was launched to investigate the possible use of treated municipal and winery wastewater for vineyard irrigation. No known studies have investigated the impact of irrigation of grapevines with treated municipal wastewater under South African conditions. Since grapevines are considered to be natural recipients of winery wastewater, more research is required to investigate the potential effects of its irrigation on grapevine health and productivity.

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CHAPTER 2: LONG-TERM EFFECTS OF IRRIGATION WITH TREATED MUNICIPAL WASTEWATER ON SOILS AND GRAPEVINES IN COMMERCIAL VINEYARDS IN THE COASTAL REGION

2.1. INTRODUCTION

In recent years the Western Cape of South Africa has been exposed to frequent water shortages and below-average rainfall that has led to the worst drought the province has experienced in the last 114 years. The ongoing drought has been particularly detrimental to the wine industry as water constraints experienced during the current season may impact grapevine growth and yield of the following seasons. As a result, water scarcity has become an increasingly more important challenge to the agricultural sector in the region. Farmers and growers have had to improve their water use efficiency, irrigation techniques and irrigation scheduling. In areas that experienced severe water shortages more profitable vineyards were prioritised and received more irrigation water and less profitable vineyards were removed.

Water restrictions imposed by the authorities and the limited supply of fresh water that can be stored on farms, have emphasised the need for alternative irrigation water sources. Treated municipal wastewater has been used as an alternative source of irrigation water in many arid and semi-arid countries. For example, *ca*. 50% Israel's irrgation water is treated municipal wastewater (Levy *et al*., 2014). It is particularly suitable as an irrigation water source in Mediterranean countries that have limited fresh water supplies during the warmer months and high rainfall during winter which can facilitate the leaching of salts applied *via* wastewater irrigation. Thus far, no studies have assessed the feasibility of using treated municipal wastewater for vineyard irrigation under South African conditions. Despite this, it is reported that *ca*. 2 000 ha of vineyards in the Swartland and surrounding regions are irrigated with treated municipal wastewater (Myburgh, 2018). This wastewater is supplied by the City of Cape Town's Potsdam wastewater treatment works (WWTW) and the Malmesbury municipality.

The possible benefits of using treated municipal wastewater for irrigation are as follows. It is a source of additional water which can improve vegetative growth and yield potential. Since domestic sources often contain high amounts of macro-elements, nutrients such as nitrogen (N), phosphorous (P) and potassium (K^+) can be recycled if applied *via* the irrigation water. In addition, the presence of organic compounds in treated municipal wastewater may have positive effects on soil structural stability. Furthermore, reusing large volumes of treated municipal wastewater in a beneficial and environmentally responsible way can be a sustainable waste disposal management strategy. This will also limit the pollution of natural water bodies where wastewater is often deposited.

On the negative side, municipal wastewaters usually have high salt loads which can affect the physical, chemical and biological properties of the soil. Sodium (Na^+) and K^+ are particularly detrimental in terms of soil structural stability and increasing soil salinity which in turn affects crop water availability. The presence of large amounts of monovalent cations may result in clay dispersion that can subsequently clog soil pores and limit water movement into and throughout the soil. In addition, irrigation using K^+-rich wastewaters may lead to excessive K^+ uptake by grapevines which can potentially have a negative effect on wine quality (Laurenson *et al*., 2012). Furthermore, corrosive metals such as iron (Fe^{2+}) and manganese (Mn^{2+}) are often present in municipal wastewater due to an influx of industrial wastewater and can lead to the clogging of irrigation equipment. The presence of heavy metals, pathogens and pharmaceutical compounds may also limit the use of treated municipal wastewater, since some of these elements can accumulate in plants and ultimately enter the biological food chain.

The objective of this study was to assess the effects of long-term irrigation with treated municipal wastewater on soil chemical and physical properties, as well as grapevine responses of commercial vineyards in the Coastal region of the Western Cape. This study formed part of a long-term study performed by the Soil and Water Science division of the ARC Infruitec-Nietvoorbij to assess the use of treated municipal wastewater for irrigating grapevines in the Coastal region of the Western Cape.

2.2. MATERIALS AND METHODS

2.2.1. Site selection and vineyard characteristics

The field trial was carried out in commercial vineyards at Boterberg farm near the town of Philadelphia in the Western Cape (-33.401661°, 18.334810°). The trial commenced around flowering (November) and continued until dormancy (June) in the 2017/18 season. The farm is located 12.4 km from the Atlantic Ocean, situated *ca.* 130 m above sea level and has a mean February temperature (MFT) of 22.1°C (Myburgh, 2011a). The region has a Mediterranean climate and is classified as a class III climatic region according to its growing degree days (GDD) from September to March (Winkler, 1974). Three experiment sites were selected on the farm on different landscape positions (Figs. 2.1 & 2.2). The first site was in a *Vitis vinifera* L. *cv*. Sauvignon blanc vineyard located on the shoulder of a hill. The second and third sites were in two *V. vinifera* L. *cv.* Cabernet Sauvignon vineyards that were situated on a backslope and a footslope, respectively (Table 2.1). The grapevines were planted 2.75 m x 1.2 m and trained onto a moveable five strand lengthened Perold trellis. Vertical shoot positioning (VSP) was implemented to prevent the development of a sprawling canopy. The vineyard was managed according the grower's normal viticultural practices in terms of cover crop and fertiliser management.

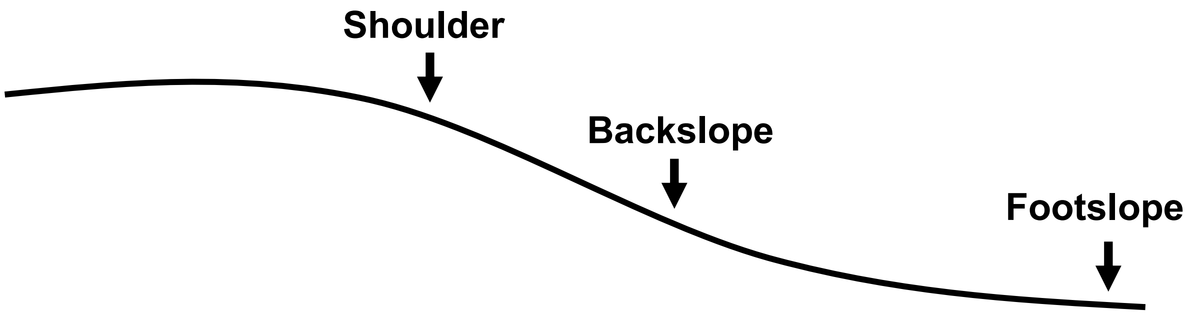

Figure 2.1. Relative landscape positions of the experiment sites at Boterberg farm.

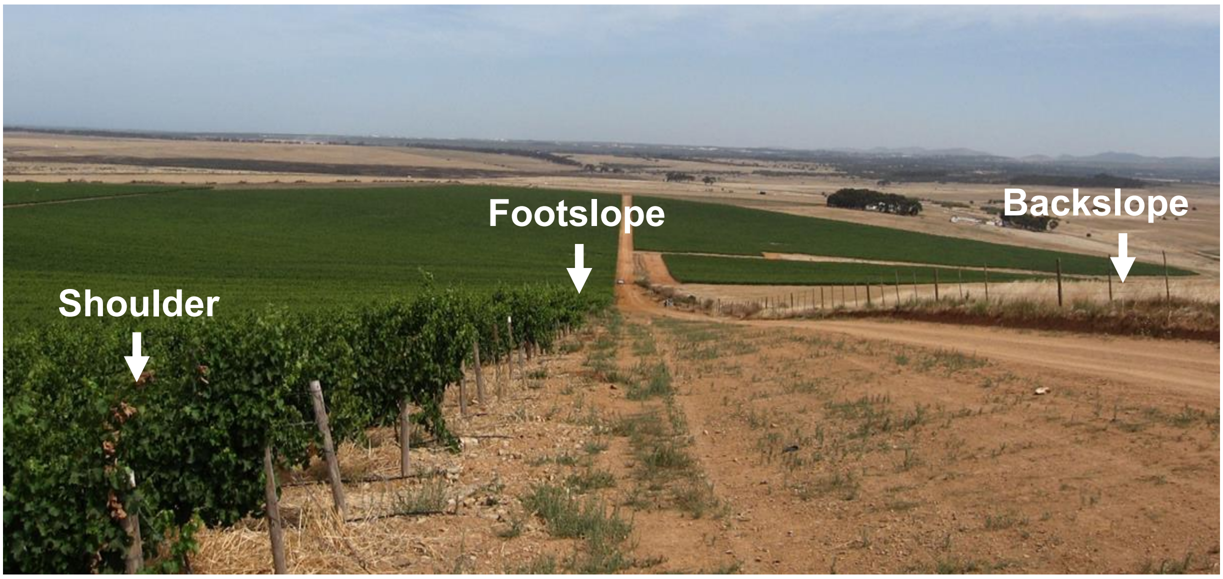

Figure 2.2. Actual localities of the experiment vineyards at Boterberg farm.

Table 2.1. Vineyard characteristics of the experiment sites at Boterberg farm.

Landscape position	Scion cultivar	Rootstock	Planting date
Shoulder	Sauvignon blanc	99R	2000
Backslope	Cabernet Sauvignon	99R	2002
Footslope	Cabernet Sauvignon	99R	2001

2.2.2. Irrigation treatments and application

The three main experiment sites were divided into three plots, each receiving a different irrigation treatment. Each of the treatment plots consisted of one row of 15 experiment grapevines, as well as a buffer row of grapevines on each side and at least two buffer grapevines at each end of the experiment rows. The first treatment was rainfed (RF), *i.e.* farmed under dryland conditions. This was considered as a control treatment. The second treatment was drip irrigated with treated municipal wastewater *via* a single dripper line (SLD) on the grapevine row. Drippers were spaced 1 m apart and had a flow rate of 2.3 ℓ/h. The volume of water applied, and the irrigation frequency was according to the grower's normal irrigation schedule. The third treatment had a double dripper line (DLD) which supplied double the volume of wastewater on the grapevine row. The three experiment sites were irrigated separately according to the grower's irrigation schedule. Water

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meters were installed in the dripper lines of the SLD plots at the beginning of the study period to measure applied irrigation volumes. The volumes of water applied to the DLD plots were calculated as twice the volume applied at the SLD plots for the respective landscape position. Irrigation commenced between September and November of each year until May or June of the next year, when the first winter rains began. Irrigation volumes, as well as rainfall data were documented each month for the duration of the study period.

2.2.3. Irrigation water origin and quality

Plots were irrigated with treated municipal wastewater that originated from the Potsdam WWTW in Milnerton, near Cape Town. The wastewater is pumped *via* an intricate pipe network to the farm. The Potsdam WWTW uses the activated sludge method along with chlorination and an ultraviolet (UV) disinfection stage to treat raw municipal wastewater to achieve chemical standards that allow the safe use of the wastewater for irrigation purposes (Olujimi *et al*., 2016; www.aurecongroup.com/projects/water/potsdam-wastewater-treatment-works). A sample of the treated municipal wastewater was taken on the farm annually at the beginning of each year from the start of the bigger study period in 2006 until 2018 (ARC, unpublished data). Wastewater samples were analysed by a commercial laboratory (Bemlab, Strand) for pH, electrical conductivity (EC_w), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), P, K^+, calcium (Ca^{2+}), magnesium (Mg^{2+}), Na^+, chloride (Cl^-), bicarbonate (HCO_3^-), sulfate (SO_4^{2-}), boron (B^{3+}), Fe^{2+}, copper (Cu^{2+}), Mn^{2+} and zinc (Zn^{2+}). The sodium adsorption ratio (SAR) was calculated as follows:

$$SAR = Na^+ \div [(Ca^{2+} + Mg^{2+}) \div 2]^{0.5} \quad \text{(Eq. 2.1)}$$

where Na^+ is the sodium concentration (mmol/ ℓ), Ca^{2+} is the calcium concentration (mmol/ ℓ) and Mg^{2+} is the magnesium concentration (mmol/ ℓ).

Similarly, the potassium adsorption ratio (PAR) was calculated as follows:

$$PAR = K^+ \div [(Ca^{2+} + Mg^{2+}) \div 2]^{0.5} \quad \text{(Eq. 2.2)}$$

where K^+ is the potassium concentration (mmol/ ℓ). Total nitrogen (total-N) was calculated as the sum of the NH_4-N and NO_3-N concentrations. The amounts of elements applied *via* treated municipal wastewater irrigation was calculated each year according to methods described by Howell (2016). The amounts of elements present in the treated municipal wastewater was presumed to be relatively constant throughout the year. Therefore, the annual wastewater samples and total irrigated volume was used to calculate the amounts of elements applied during each season.

2.2.4. Soil chemical properties

Baseline soil samples were taken in 2006 before wastewater irrigation commenced (ARC, unpublished data). Following 11 years of irrigation with treated municipal wastewater, samples were taken at budbreak (September) of the 2017/18 season. Soils were sampled in 30 cm increments to a depth of 90 cm in all plots, and up to 180 cm in soils where it was possible to sample deeper. Soil chemical analyses were carried out by a commercial laboratory (Bemlab, Strand). The $pH_{(KCl)}$ was determined in a suspension of 1 M potassium chloride (KCl) at 25°C. Soil electrical resistance of the saturated paste extract (R_s) was determined according to methods presented by Jones Jr. (1999). Thereafter, R_s was converted to electrical conductivity (EC_e) by means of the following equation:

$$EC_e = (0.25 \div R_s) \times 1\,000 \quad \text{(Eq. 2.3)}$$

where EC_e is the electrical conductivity of the saturated paste extract (dS/m), 0.25 is the constant for the Bureau of Soils electrode cup (Richards, 1954), R_s is the electrical resistance of the saturated paste extract (ohm) and 1 000 is the conversion factor used to convert millimhos/cm to dS/m.

Basic cations (Ca^{2+}, Mg^{2+}, K^+, Na^+) were extracted with 1 M ammonium acetate at pH 7. The cation concentrations in the extracts were determined by inductively coupled plasma optical emission spectrometry (ICP-OES) using a spectrometer (PerkinElmer Optima 7300 DV, Waltham, Massachusetts). The amounts of soluble cations were not determined, therefore the amount of exchangeable cations, which is the extractable minus the soluble amounts (Richards, 1954), could not be calculated. As a result, the cation exchange capacity (CEC) could not be calculated. The majority of South African laboratories solely determine extractable cations due to the laborious process of determining exchangeable cations and CEC (Conradie, 1994). Most laboratories calculate the sum of extractable cations to obtain an estimated CEC, which is referred to as the S-value (Howell, 2016). Subsequently, the exchangeable sodium percentage (ESP) and exchangeable potassium percentage (EPP) of the soil could not be calculated. However, the extractable sodium percentage (ESP´) was calculated by means of the following equation:

$$ESP' = (Na^+ \div S) \times 100 \quad \text{(Eq. 2.4)}$$

where Na^+ is the extractable sodium ($cmol^{(+)}/kg$) and S is the S-value ($cmol^{(+)}/kg$), *i.e.* the sum of Ca^{2+}, Mg^{2+}, K^+ and Na^+. Similarly, the extractable potassium percentage (EPP´) was calculated as follows:

$$EPP' = (K^+ \div S) \times 100 \quad \text{(Eq. 2.5)}$$

where K^+ is the extractable potassium ($cmol^{(+)}/kg$) and S is the S-value ($cmol^{(+)}/kg$), *i.e.* the sum of Ca^{2+}, Mg^{2+}, K^+ and Na^+.

Bray II P and K^+ was determined by extraction with 0.03 M ammonium fluoride (NH_4F) in 0.01 M hydrochloric acid (HCl). The P and K^+ concentrations in the extract were determined in the same manner as the basic cations. Soil organic carbon (SOC) was determined using methods described by Walkley and Black (1934). Soil chloride (Cl^-) was not determined at the beginning of the larger study period. During the 2017/18 season, soil Cl^- was determined volumetrically *via* titration of a

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0.1 M potassium nitrate (KNO_3) soil extract with 0.043 M silver nitrate ($AgNO_3$) using potassium dichromate (K_2CrO_4) as an indicator (Chapman & Pratt, 1961).

2.2.5. Soil physical properties

2.2.5.1. Soil texture

Soil textural characteristics were determined for each 30 cm soil increment as described in Section 2.2.4, except for the shoulder DLD plot where samples deeper than 60 cm did not contain enough soil to perform the analyses. Particle size distribution was determined using the hydrometer method (Van der Watt, 1966).

2.2.5.2. Near-saturation hydraulic conductivity

Mini disk infiltrometers (Decagon Devices, Pullman, WA) were used to measure near-saturation hydraulic conductivity (K_{ns}) of the soils in October 2017 (Fig. 2.3). Measurements were replicated five times in each treatment plot at each of the three landscape positions. Measurements were carried out on the grapevine row where a thin layer of fine sand was added to the soil surface to ensure a level surface and adequate contact between the base of the infiltrometer and the soil surface (Köhne *et al*., 2011). The treated municipal wastewater used for irrigation was also used for the K_{ns} measurements. The EC_w and SAR of the irrigation water were 1.3 dS/m and 3.8, respectively. A suction head of 2 cm was maintained for each measurement. Fixed time intervals between measurements were chosen to allow a water level decrease of at least 1 mℓ to minimise reading errors. The K_{ns} values (mm/h) were calculated by means of the following equation:

$$K_{ns} = \{[(V_i - V_f) \div 1000] \div 0.001521\} \times 60 \div \Delta t \quad \text{(Eq. 2.6)}$$

where V_i is the initial volume reading (mℓ), V_f is the final volume reading (mℓ), 0.001521 is the area (m^2) of the sintered stainless steel disk at the bottom of the infiltrometer and Δt is the difference in time between measurements (min).

Figure 2.3. Measurement of near-saturation hydraulic conductivity using mini disk infiltrometer.

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2.2.6. Soil water content

Soil water content (SWC) was measured once a month for the duration of the study using a calibrated neutron probe (Fig. 2.4A). One access tube (50 mm Ø class 4 Polyvinyl chloride [PVC]) was installed on the grapevine row at each of the treatment plots at the beginning of the study period using a 50 mm custom built steel auger. Measurements were taken in 30 cm increments up to a depth of 90 cm. Before and after neutron count readings were recorded, five standard count readings were taken while the probe was standing on the neutron probe case (Fig. 2.4B). Count ratios were calculated by determining the ratio between the actual neutron probe readings at each depth and the mean of the ten standard count readings (Moffat, 2017). Subsequently, the count ratios were calibrated against volumetric soil water content (θ_V).

In order to establish θ_V, gravimetric soil water content θ_g was determined by taking three replicate soil samples at each of the treatment plots on the same day neutron probe readings were taken. Soil samples were collected over the 0–30 cm, 30–60 cm and 60–90 cm soil layers. Soils were sampled using a Viehmeyer soil auger on the grapevine row within close proximity of the neutron probe access tubes. The sampled soils were placed in individual metal cans and sealed, where after the samples were weighed on a laboratory balance (Sartorius Excellence E2000D, Göttingen, Germany) at the ARC Infruitec-Nietvoorbij Irrigation laboratory. The samples were then oven-dried at 105°C for 16 hours in the cans with the lids removed. Following this, the containers were removed from the oven, the lids were closed, and samples were placed in a desiccator containing copper sulfate ($CuSO_4$) crystals.

Once cooled down, samples were weighed again and θ_g was calculated using the following equation:

$$\theta_g = (m_w - m_d) \div m_d \qquad \text{(Eq. 2.7)}$$

where m_w is the initial mass of wet soil (g), m_d is the mass of dried soil (g). The θ_g of each plot was determined as the mean of the three gravimetric samples. Subsequently, θ_V was calculated as follows:

$$\theta_V = \theta_g \times \rho_b \qquad \text{(Eq. 2.8)}$$

where ρ_b is soil bulk density. A ρ_b of 1.65 g/cm^3 was used for the calculation, which is the mean ρ_b of over 70 soils in the Western Cape as determined by Van Huysteen (1989). Soil water content for each soil layer was calculated as follows:

$$SWC = \theta_V \times d \times 100 \qquad \text{(Eq. 2.9)}$$

where d is the depth of the soil layer (dm). The SWC for the respective soil layers were summed to obtain the SWC of the 90 cm soil profile.

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Figure 2.4. Performing actual neutron probe readings (A) and standard counts with the instrument on its carry case (B).

2.2.7. Grapevine water status

Grapevine water potential was measured by means of the pressure chamber technique (Scholander *et al.,* 1965), according to guidelines described by Myburgh (2010). During the 2017/18 season midday stem water potential (Ψ_S) was measured at each treatment plot in three mature, unscathed leaves located opposite a bunch (Fig. 2.5). The leaves were covered in aluminium bags (Choné *et al*., 2001; Myburgh, 2010) for a minimum of one hour before measurements were carried out (Howell, 2016). Mean Ψ_S per treatment at each of the landscape positions was calculated. Measurements were carried out at pea size (November 2017), at véraison (December 2017) and prior to harvest (February 2018).

Figure 2.5. Mature leaf covered in aluminium bag for determination of midday stem water potential.

2.2.8. Vegetative grapevine measurements

2.2.8.1. Leaf chemical status

At véraison of the 2017/18 growing season, 30 mature leaves opposite a bunch were collected per treatment plot at each of the landscape positions. Petioles were immediately removed from

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the leaf blades and the leaves were placed into paper bags. The samples were then dried in a fan oven at 60°C for 24 hours. The chemical status of the dried leaf blades was determined by a commercial laboratory (Bemlab, Strand). Leaf N was determined according to the methods described by Horneck & Miller (1998) by means of a nitrogen analyser. An ICP-OES spectrometer (PerkinElmer Optima 7300 DV, Waltham, Massachusetts, U.S.A.) was used to determine P, K^+, Ca^{2+}, Mg^{2+}, Na^+, Cl^- Mn^{2+}, Fe^{2+}, Cu^{2+}, Zn^{2+} and B^{3+} according to methods described by Isaac and Johnson (1998).

2.2.8.2. Growth characteristics

Prior to budbreak of the 2017/18 season, trunk circumference and cordon length were quantified. The average number of fruiting canes per grapevine were also counted. During the 2017/18 ripening period, five shoots were collected per treatment plot at each of the experiment sites (*i.e.* 5 x 9 = 45 shoots in total) for the analyses of canopy characteristics. Five grapevines per treatment plot from each of the landscape positions were randomly selected from which one shoot was selected per grapevine. Shoots were selected from spurs that were in close proximity to the crown of the grapevine. Shoots were cut off at the base, placed in plastic bags and transported to the laboratory for analyses. Secondary shoots were separated from the primary shoots. The length of primary and secondary shoots was measured, and the number of secondary shoots were counted. The number of internodes on primary shoots were also counted. The average length of shoots and internodes per treatment were calculated for each of the experiment plots. The mean diameter of primary shoots was estimated by measuring the shoot diameter with a digital caliper at three points, namely at the top, middle and bottom of each primary shoot. Leaves were separated into primary and secondary leaves. Leaves were counted, and the total leaf area was measured using an electronic surface area meter (LI-COR Model 3100C, Nebraska, U.S.A.). Leaf area per grapevine (m^2) was calculated by multiplying the total leaf area per shoot by the calculated number of shoots per grapevine. The leaf area index was calculated by dividing the leaf area per grapevine by the plant spacing (Mehmel, 2010).

Over the last five years of the bigger study period, *i.e.* 2014 to 2018, grapevine vigour was quantified by measuring pruning mass in the dormant period (ARC, unpublished data). Cane mass was determined in the vineyard at each of the experiment plots after pruning using a hanging balance.

2.2.9. Yield and its components

2.2.9.1. Bunch and berry mass at harvest

At harvest of the 2017/18 season, ten randomly selected bunches were picked from each plot at the three landscape positions. The bunches were weighed using an electronic balance to determine bunch mass. A sample of 100 berries was obtained by picking ten berries from each of

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the ten bunches. The berry samples were weighed in the laboratory to determine mean berry mass.

2.2.9.2. Yield

All the bunches of the experiment plots were picked by hand and counted using a mechanical counter at harvest. The objective was to harvest the grapes at 24°B, but due to logistical constraints this was not always possible. A top loader mechanical balance was used to weigh the grapes and obtain the total mass per plot at each of the experiment sites. Grape mass per grapevine (kg/grapevine) was calculated by dividing the total grape mass per treatment by the number of grapevines per treatment. This was then converted to yield (t/ha).

2.2.10. Grape juice characteristics

A representative sample of bunches were selected at harvest and gently crushed to extract juice from the berries. The juice was poured through a fine sieve and collected in 50 mℓ sample tubes. The samples were analysed for total soluble solids (TSS) using a handheld refractometer (Atago PAL 1, Tokyo, Japan). Total titratable acidity (TTA) and pH was determined at the Department of Viticulture and Oenology of the University of Stellenbosch using an automatic titrator (Metrohm 785 DMP Titrino, Herisau, Switzerland).

2.2.11. Statistical analysis

Calculations of means and standard deviations (SD) were carried out using Microsoft Office Excel 365 version. Relationships between variables were calculated by means of linear regression at the 95% confidence level using Statgraphics®.

2.3. RESULTS AND DISCUSSION

2.3.1. Rainfall

The monthly rainfall for the 2017/18 season measured from July 2017 to June 2018 in relation to the long-term mean (LTM) measured over the 12-year study period is shown in Figure 2.6. The rainfall during the 2017/18 season was below the LTM for most of the season and only exceeded the LTM during August 2017 as well as April and May 2018 (Fig. 2.6). An average of 86 mm was measured during the summer months of September to March over the course of the study period, whilst only 39 mm was measured during the summer of 2017/18. The average winter rainfall from May to August was 160 mm. The LTM annual rainfall measured from July to June of each year was 253 mm while only 118 mm was measured from July 2017 to June 2018.

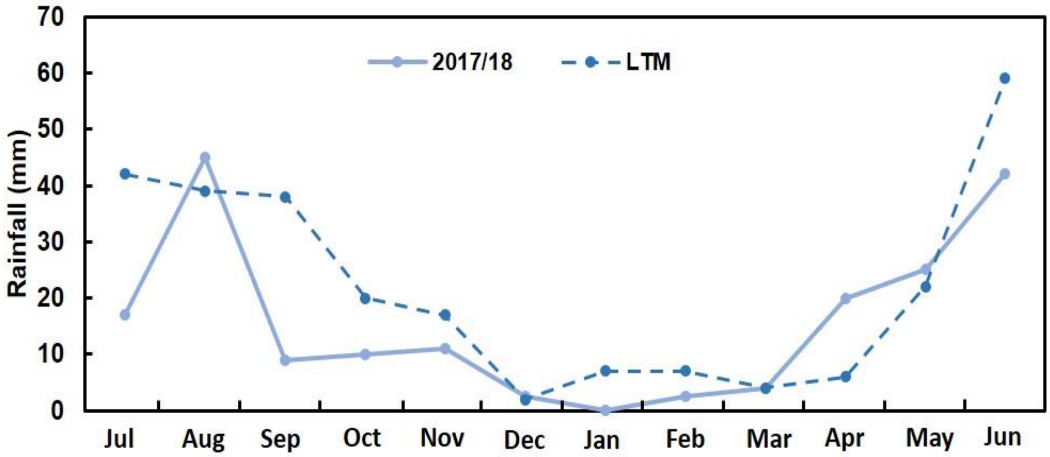

Figure 2.6. Monthly rainfall (mm) from July 2017 to June 2018 in relation to the long-term mean (LTM) measured over the 12-year study period at Boterberg farm (ARC, unpublished data).

2.3.2. Irrigation

2.3.2.1. Volumes of irrigation water applied

The total amounts of treated municipal wastewater applied at the SLD and DLD plots over the course of the study period are presented in Appendix 1. The total amount of irrigation water applied during the 2017/18 season (September 2017 to May 2018) in relation to the seasonal LTM is presented in Table 2.2. Below average rainfall experienced during the 2017/18 growing season warranted the application of greater volumes of irrigation water. On average, the shoulder and backslope sites received similar volumes of irrigation water throughout the season, whereas the plots at the footslope site received considerably greater amounts of water. During the 2017/18 growing season, the footslope SLD plot received a larger volume of irrigation water compared to the DLD plots of both the shoulder and backslope sites. This was due to the grower's irrigation scheduling.

Table 2.2. Volume of treated municipal wastewater (mm) applied during the 2017/18 growing season and long-term seasonal mean (mm) irrigated at each of the experiment sites under single (SLD) and double line drip (DLD).

Landscape position	Treatment	2017/18	Long-term seasonal mean
Shoulder	SLD	246.5	145.9
	DLD	493.0	291.7
Backslope	SLD	221.1	154.3
	DLD	442.2	308.6
Footslope	SLD	624.9	269.6
	DLD	1249.8	539.1

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2.3.2.2. Quality of irrigation water

pH: The pH range of the treated municipal wastewater varied between 6.7 and 8.0 throughout the 12-year study period (Table 2.3). This was between values of 6.2 and 9.8 previously reported for treated municipal wastewater in Australia (Stevens, 2009), but lower than pH values of 8.5 to 9.0 reported for secondary treated municipal effluents in Botswana (Emongor & Ramolemana, 2004). The pH variation was within the range of 6.5 to 8.4 which is recommended for irrigation water (Howell & Myburgh, 2013 and references therein). Irrigation water with a pH outside of this range may result in nutritional imbalances, or may contain toxic ions (Ayers & Westcot, 1985).

Table 2.3. Summary of the pH and electrical conductivity (EC_w) levels of the treated municipal wastewater used for vineyard irrigation at Boterberg farm from 2006 to 2018.

Parameter	Unit	Min.	Max.	Mean
pH		6.7	8.0	7.1
EC_w	dS/m	0.7	1.2	0.9

EC_w: The EC_w of the treated municipal wastewater (Table 2.3) was well within the range of 0.2 dS/m to 2.9 dS/m reported by Stevens (2009) and similar to values of 0.9 dS/m to 1.6 dS/m reported by Laurenson *et al*. (2012). However, the mean EC_w exceeded the critical value of 0.8 dS/m which is the salinity threshold for water used to irrigate grapevines (Van Zyl, 1981). The EC_w range measured for the irrigation water fell within the range of 0.7 dS/m to 3.0 dS/m at which salinity problems in terms of crop water availability might occur in sensitive crops (Ayers & Westcot, 1985). However, no reduction in vegetative growth of grapevines is expected at the maximum measured EC_w (1.2 dS/m) (Ayers & Westcot, 1985). Both the pH and EC_w ranges of the treated municipal wastewater were within the legislated limits to irrigate up to 500 m^3 of wastewater per day as prescribed by the General Authorisations (Department of Water Affairs [DWA], 2013).

Nitrogen: The mean total-N value measured over the course of the study period (Table 2.4) was considerably lower than the range of 8.0 mg/ℓ to 30.7 mg/ℓ reported previously (Laurenson *et al*., 2012). The maximum of 16.0 mg/ℓ was a result of high NH_4-N levels in the wastewater during the 2011/12 growing season (data not shown). However, the NH_4-N levels of the irrigation water was on average quite low. Similarly, NO_3-N levels were well below values of 6.7 mg/ℓ to 29.3 mg/ℓ previously reported for secondary treated municipal effluent (Emongor & Ramolemana, 2004). As a result, the mean total-N level was below the critical value of 5 mg/ℓ at which crops sensitive to N (such as grapevines) might be affected (Howell & Myburgh, 2013 and references therein). Therefore, an over-supply of N through treated wastewater irrigation was not a concern.

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Table 2.4. Summary of the total nitrogen (Total N), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N) and phosphorous (P) levels present in the treated municipal wastewater used for vineyard irrigation at Boterberg farm from 2006 to 2018.

Parameter	Unit	Min.	Max.	Mean
Total N	mg/ℓ	1.0	16.0	4.3
NH_4-N	mg/ℓ	0.1	12.7	2.1
NO_3-N	mg/ℓ	0.0	5.6	2.4
P	mg/ℓ	0.1	9.5	3.2

Phosphorous: The P levels of the treated municipal wastewater ranged between 0.1 mg/ℓ and 9.5 mg/ℓ (Table 2.4) and was similar to the range of 2.7 mg/ℓ to 12.8 mg/ℓ reported by Laurenson *et al.* (2012). However, the P concentration in the wastewater consistently exceeded the long-term critical value of 0.05 mg/ℓ which demarcates a risk for algal blooms and bio-fouling of the irrigation equipment (Howell & Myburgh, 2013 and references therein).

Calcium: The levels of Ca^{2+} in the wastewater varied between 33.4 mg/ℓ and 67.3 mg/ℓ throughout the 12-year study period (Table 2.5). This range was considerably higher than values of 6 mg/ℓ to 16 mg/ℓ that was reported by Chen *et al*. (2013a), but it was similar to values of 31 mg/ℓ to 70 mg/ℓ reported by Andrews *et al*. (2016). There are no South African guidelines for Ca^{2+} concentrations in irrigation water (Department of Water Affairs & Forestry [DWAF], 1996). The determination of Ca^{2+} levels is important since it is used to calculate the SAR. Where irrigation water contains appreciable amount of Ca^{2+} it may help to reduce the SAR and PAR and as a result, mitigate the impacts of Na^+ and K^+ on soil structural stability.

Table 2.5. Summary of the calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+) and sodium (Na^+) levels present in the treated municipal wastewater used for vineyard irrigation at Boterberg farm from 2006 to 2018.

Parameter	Unit	Min.	Max.	Mean
Ca^{2+}	mg/ℓ	33.4	67.3	46.4
Mg^{2+}	mg/ℓ	6.1	11.6	8.5
K^+	mg/ℓ	14.8	32.6	20.3
Na^+	mg/ℓ	100.7	173.6	120.9

Magnesium: Similar to Ca^{2+}, there are no guidelines available for Mg^{2+} levels in irrigation water (DWAF, 1996) and it can also play a positive role in decreasing the SAR. However, crops that are irrigated with Mg-rich water may be affected by Mg-induced Ca^{2+} deficiencies, but the Ca-Mg ratio is not regularly used for evaluation due to insufficient data (Ayers & Westcot, 1985). Nevertheless, the Mg^{2+} levels in the treated municipal wastewater used in this study was relatively low and ranged from 6.1 mg/ℓ to 11.6 mg/ℓ over the course of the study period (Table 2.5).

Potassium: Mean K^+ levels of the irrigation water were 20.3 mg/ℓ (Table 2.5). This was below the ranges of 23 mg/ℓ to 25 mg/ℓ reported by Laurenson *et al*. (2012) in Australia and 22 mg/ℓ to 37.4 mg/ℓ reported by Paranychianakis *et al*. (2006) in Greece. Since K^+ concentrations in municipal wastewater are often relatively low compared to other constituents, it is generally not reported (Stevens, 2009). Subsequently, the South African Water Quality Guidelines (DWAF, 1996) omitted a legal limit for K^+ concentrations in irrigation water. Previous studies have reported that increased K^+ concentrations in soils may lead to a reduction in the hydraulic conductivity (*K*) and infiltration rate (IR) of soils (Quirk & Schofield, 1955; Levy & Van der Watt, 1990). According to Arienzo *et al*. (2009a), K^+ can have a broad spectrum of possible effects on infiltration, ranging from being similar to Na^+, to being similar to Ca^{2+}. Furthermore, Levy and Van der Watt (1990) concluded that K^+ had an intermediate effect relative to Na^+ and Ca^{2+} on soil hydraulic properties. Given the K^+ concentration of the treated municipal wastewater used in this study, it is not expected that K^+ supplied *via* wastewater irrigation would have a negative impact on the soil hydraulic properties.

Sodium: The variation in Na^+ levels of the irrigation water was between 100.7 mg/ℓ and 173.6 mg/ℓ (Table 2.5) and therefore lower than the range of 208 mg/ℓ to 264 mg/ℓ reported previously for treated municipal wastewater that have undergone secondary treatment *via* the activated sludge method (Paranychianakis *et al*., 2006). In contrast, the Na^+ levels were considerably higher than the range of 40 mg/ℓ to 70 mg/ℓ reported by Chen *et al*. (2013a). The mean Na^+ concentration of 120.9 mg/ℓ exceeded the critical value of 100 mg/ℓ which is the legal limit for irrigating grapevines in South Africa (Howell & Myburgh, 2013 and references therein). Grapevines are considered to be moderately sensitive to foliar injury from Na^+ (Howell, 2016). Consequently, a Na^+ concentration of 115 mg/ℓ in the irrigation water is considered the upper threshold for overhead irrigation (DWAF, 1996). As the experiment grapevines in this study were irrigated by means of drippers below the canopy, the leaves were not wetted with irrigation water. However, increasing Na^+ levels in the soil *via* wastewater irrigation may have adverse effects on soil structure (Rengasamy & Olsson, 1991).

SAR: The mean SAR of the treated municipal wastewater also met the criteria stipulated by the General Authorisations for irrigating up to 500 m^3 per day (Table 2.6). The SAR only exceeded the criteria of 5 during the 2017/18 season and was measured as 5.5. According to the SAR values of 0 to 10 presented by Van Zyl (1981), the treated municipal wastewater had a low sodium hazard. Furthermore, the SAR was below the threshold value of 20 at which Na^+ toxicities are expected in grapevines (Ayers & Westcot, 1985 and references therein). However, the combination of the relatively low EC_w (mean 0.9 dS/m) and low SAR (mean 4.3) may potentially result in problems with water infiltration (Ayers & Westcot, 1985).

Table 2.6. Summary of the sodium adsorption ratio (SAR) and potassium adsorption ratio (PAR) levels of the treated municipal wastewater used for vineyard irrigation at Boterberg farm from 2006 to 2018.

Parameter	Unit	Min.	Max.	Mean
SAR	$(mmol/ℓ)^{0.5}$	3.0	5.5	4.3
PAR	$(mmol/ℓ)^{0.5}$	0.3	0.6	0.4

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PAR: The PAR variation of 0.3 to 0.6 (Table 2.6) was similar to values of 0.4 to 0.6 reported by Laurenson *et al*. (2012). The PAR has been less widely adopted for wastewater quality evaluation due to the typically low K^+ concentrations present in most wastewaters (Laurenson *et al*., 2012). However, the PAR can be an important measurement to estimate soil dispersion risks where agro-industrial wastewaters are applied (Smiles & Smith, 2004).

Chloride: The mean Cl^- concentration of the treated municipal wastewater was 160.2 mg/ℓ but ranged between 111.2 mg/ℓ and 218.2 mg/ℓ throughout the 12-year study period (Table 2.7). The Cl^- levels present in the irrigation water was well below the threshold value of 700 mg/ℓ at which toxicity problems in grapevines might occur (Van Zyl, 1981).

Table 2.7. Summary of the chloride (Cl^-), bicarbonate (HCO_3^-) and sulfate (SO_4^{2-}) levels present in the treated municipal wastewater used for vineyard irrigation at Boterberg farm from 2006 to 2018.

Parameter	Unit	Min.	Max.	Mean
Cl^-	mg/ℓ	111.2	281.2	160.2
HCO_3^-	mg/ℓ	142.1	242.0	203.0
SO_4^{2-}	mg/ℓ	54.0	276.0	84.4

Bicarbonate: The levels of HCO_3^- in the irrigation water ranged between 142.1 mg/ℓ and 242.0 mg/ℓ (Table 2.7) which is higher than the values of 50 mg/ℓ to 100 mg/ℓ reported by Chen *et al*. (2013a). However, the mean HCO_3^- concentration measured throughout the study period was similar to secondary treated municipal effluents in Botswana (Emongor & Ramolemana, 2004). It should be noted that high levels of HCO_3^- in irrigation water may have negative impacts on crops, soils and irrigation equipment (Howell, 2016). The addition of water rich in HCO_3^- and carbonate (CO_3^{2-}) may increase HCO_3^- in the soil solution and result in the precipitation of insoluble Ca^{2+} and Mg^{2+} carbonates when the soil dries out (Van Zyl, 1981).

Sulfate: The SO_4^{2-} levels in the irrigation water varied between 54 mg/ℓ and 276 mg/ℓ (Table 2.7) and was similar to values of 66 mg/ℓ and 192 mg/ℓ previously reported for municipal wastewater treated *via* the activated sludge method in California (Pescod, 1992 and references therein). There are currently no guidelines available for the permissible levels of SO_4^{2-} in irrigation water (DWAF, 1996). However, it is important to measure SO_4^{2-} levels in irrigation water as waters containing high levels of both Ca^{2+} and SO_4^{2-} may result in the precipitation of gypsum ($CaSO_4$) in irrigation equipment and subsequent clogging of equipment (Du Plessis *et al*., 2017).

Boron: Levels of B^{3+} in the irrigation water ranged between 0.18 mg/ℓ and 0.50 mg/ℓ with a mean value of 0.27 mg/ℓ over the course of the study (Table 2.8). Although B^{3+} is considered an essential plant nutrient, it can be toxic at reasonably low concentrations. Grapevines have been classified as sensitive (Ayers & Wetscot, 1985; DWAF, 1996) to highly sensitive (Van Zyl, 1981) towards B^{3+} toxicities. A maximum B^{3+} concentration of between 0.5 mg/ℓ and 0.75 mg/ℓ has been suggested by Ayers and Westcot (1985) for water used for irrigating grapevines. With regards to these

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thresholds, the treated municipal wastewater used for the present study did not hold any risks in terms of B^{3+} toxicity.

Table 2.8. Summary of the boron (B^{3+}), copper (Cu^{2+}), iron (Fe^{2+}), manganese (Mn^{2+}) and zinc (Zn^{2+}) levels present in the treated municipal wastewater used for vineyard irrigation at Boterberg farm from 2006 to 2018.

Parameter	Unit	Min.	Max.	Mean
B^{3+}	mg/ℓ	0.2	0.50	0.27
Cu^{2+}	mg/ℓ	0.0	0.06	0.02
Fe^{2+}	mg/ℓ	0.0	0.34	0.10
Mn^{2+}	mg/ℓ	0.0	0.08	0.04
Zn^{2+}	mg/ℓ	0.0	0.21	0.05

Copper: Concentrations of Cu^{2+} in the treated municipal wastewater varied from being completely absent to a maximum concentration of 0.06 mg/ℓ (Table 2.8). According to Ayers and Westcot (1985), Cu^{2+} can be toxic to some plants at levels between 0.1 and 1.0 mg/ℓ which is higher than was measured in the irrigation water in this study. Therefore, no Cu^{2+} toxicities were expected.

Iron: The Fe^{2+} levels ranged between 0.00 mg/ℓ and 0.34 mg/ℓ with a mean value of 0.10 mg/ℓ throughout the 12-year study period (Table 2.8). The Fe^{2+} concentration in the wastewater never exceeded the critical value of 5 mg/ℓ which is the recommended maximum concentration of Fe^{2+} in irrigation water used for irrigation of grapevines (Van Zyl, 1981). In addition, the measured Fe^{2+} levels were below the value of 1.5 mg/ℓ at which Fe precipitation and the clogging of drip irrigation systems might occur (DWAF, 1996).

Manganese: Levels of Mn^{2+} in the irrigation water varied from being absent to a maximum concentration of 0.08 mg/ℓ (Table 2.8). Ayers and Westcot (1985) recommended a maximum level of 0.2 mg/ℓ, whilst South African guidelines recommend not to exceed 1.5 mg/ℓ as Mn^{2+}, like Fe^{2+}, may cause the clogging of irrigation pipelines (DWAF, 1996).

Zinc: The maximum concentration of Zn^{2+} measured in the treated municipal wastewater was 0.21 mg/ℓ, whereas the mean over the course of the study was 0.05 mg/ℓ (Table 2.8). A maximum level of 2 mg/ℓ is recommended for grapevines under continuous irrigation on all soils (Van Zyl, 1981).

2.3.2.3. Amounts of elements applied

Nitrogen: The amounts of NH_4-N, NO_3-N and total-N applied at each of the experiment plots from 2006 to 2017 are presented in Appendices 2.1, 2.2 and 2.3, respectively. During the 2017/18 growing season, the amounts of NH_4-N applied were well below the LTM (Table 2.9). This was due to a low concentration of NH_4-N present in the wastewater (data not shown). In contrast, the NO_3-N applied during the 2017/18 season was similar to the mean measured over the 12-year study period (Table 2.9). However, the low amounts of NH_4-N applied resulted in low amounts of total-N applied (Table 2.9). With the exception of the footslope site, all the plots received total-N well below the LTM. The plots at the footslope site received greater amounts of N compared to the shoulder and backslope sites due to higher irrigation volumes applied (Table 2.2). The total-N applied at the DLD plot of the footslope site was comparable to the estimated 22.6 kg/ha applied *via* 100 mm of municipal wastewater irrigation (Laurenson *et al*., 2012). According to Saayman (1981), an annual N loading of *ca*. 50 kg/ha is required where grapevines produce 10 tonnes of grapes per hectare. Therefore, the amount of N applied *via* treated municipal wastewater irrigation at all the experiment plots appeared to be inadequate to supply the annual N requirements of grapevines.

Table 2.9. Amounts of ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N) and total nitrogen (Total-N) applied *via* treated municipal wastewater (kg/ha) used to irrigate vineyards during the 2017/18 growing season in relation to the long-term mean (LTM).

Landscape position	Treatment	NH_4-N	LTM	NO_3-N	LTM	Total-N	LTM
Shoulder	SLD	0.7	3.0	2.5	2.8	3.2	5.8
	DLD	1.4	6.1	5.1	5.6	6.5	11.7
Backslope	SLD	0.6	3.2	2.3	2.9	2.9	6.1
	DLD	1.2	6.4	4.6	5.7	5.8	12.2
Footslope	SLD	1.7	3.5	6.4	5.7	8.2	9.2
	DLD	3.5	6.9	12.9	11.5	16.4	18.4

Phosphorous: The amount of P applied *via* treated municipal wastewater irrigation at each of the experiment plots from the beginning of the study period is presented in Appendix 2.4. During the 2017/18 growing season the amounts of P applied varied between 1.3 kg/ha at the backslope SLD plot and 7.1 kg/ha at the footslope DLD plot (Table 2.10). The amounts of P applied at all the experiment plots during 2017/18 were lower than the LTM due to a low P concentration in the municipal wastewater (data not shown). The mean amounts of P applied *via* treated municipal wastewater at the shoulder and backslope DLD plots were similar, whilst the amounts applied at the footslope DLD plot was higher than the estimated 8.2 kg/ha P applied through 100 mm of municipal wastewater irrigation (Laurenson *et al*., 2012). Grapevines require *ca*. 0.7 kg P per tonne of grapes produced (Saayman, 1981). Based on this recommendation, only the footslope DLD plot received sufficient amounts of P *via* treated municipal wastewater irrigation during the 2017/18 season to produce a grape yield of 10 t/ha under the prevailing conditions. However, the mean amounts of P applied over the course of the study period indicated that DLD plots reached the abovementioned requirements during most years. In contrast, SLD plots received inadequate amounts of P *via* wastewater irrigation during all the years of the study. Previous studies have shown improved grapevine P nutritional status as a result of treated municipal wastewater irrigation (McCarthy, 1981; Neilsen *et al*., 1989; Paranychianakis *et al*., 2006). Higher mobility within the soil profile (Laurenson *et al*., 2012) and greater plant utilisation (Sakadevan *et al*., 2000) have been reported for P applied *via* municipal wastewater irrigation.

Table 2.10. Amounts of phosphorous (P), potassium (K^+) and calcium (Ca^{2+}) applied *via* treated municipal wastewater (kg/ha) used to irrigate vineyards during the 2017/18 growing season in relation to the long-term mean (LTM).

Landscape position	Treatment	P	LTM	K^+	LTM	Ca^{2+}	LTM
Shoulder	SLD	1.4	4.2	57.2	28.2	125.2	64.3
	DLD	2.8	8.3	114.4	56.4	250.4	128.6
Backslope	SLD	1.3	4.3	51.3	30.8	112.3	67.4
	DLD	2.5	8.6	102.6	61.6	224.6	134.8
Footslope	SLD	3.6	5.3	145.0	53.6	317.4	113.6
	DLD	7.1	10.5	290.0	107.3	634.9	227.2

Potassium: The amounts of K^+ applied annually *via* treated municipal wastewater irrigation over the 12-year study period ranged between 11.8 kg/ha at the footslope SLD plot and 390.2 kg/ha at the footslope DLD plot (Appendix 2.5). During the 2017/18 growing season the amounts of K^+ applied through the irrigation water varied between 51.3 kg/ha at the backslope SLD and 290.0 kg/ha at the footslope DLD plot (Table 2.10). The mean amounts of K^+ applied at the shoulder and backslope SLD plots were similar to the estimated amount of 29.4 kg/ha applied *via* 100mm of municipal wastewater irrigation (Laurenson *et al*., 2012). Grapevines have an annual requirement of *ca.* 3 kg K^+ per tonne of grapes produced (Saayman, 1981). Based on this norm, during the 2017/18 growing season all of the experiment plots were supplied amounts of K^+ *via* treated municipal wastewater in excess of what is required to produce grape yields of 10 t/ha. On average, between 0.8 kg/ha and 77.3 kg/ha K^+ was applied in excess each year. An over-supply of K^+ to grapevines can have numerous implications. Since grape berries are considered to be a strong sink for K^+ (Mpelasoka *et al*., 2003), excessive application may lead to an accumulation of K^+ in the berries. This in turn may have negative impacts on wine quality. A high concentration of K^+ in grape juice may lead to a reduction in the concentration of tartaric acid in the juice and result in increased juice, must and wine pH (Saayman, 1981; Mpelasoka *et al*., 2003; Kodur, 2011). Consequently, the increased pH may lead to the development of unstable musts and wines, as well as reduce the colour quality of red wines (Somers, 1975; McCarthy & Downton, 1981; Mpelasoka *et al*., 2003). The application of excessive amounts of K^+ may also reduce the juice N content (Saayman, 1981) and subsequently increase the risk of stuck fermentations during winemaking (Bell & Henschke, 2005; Malherbe *et al*., 2007). Excessive K^+ in the soil can also reduce the uptake of Ca^{2+} and Mg^{2+} by grapevines due to an antagonistic interaction between K^+ and these cations (Morris & Cawthon, 1982). Furthermore, it must be noted that the supplied K^+ will only be beneficial for grapevine nutrition for a short period after harvest as K^+ absorption decreases during the post-harvest period (Conradie, 1981b).

Calcium: The amounts of Ca^{2+} applied through treated municipal wastewater irrigation over the duration of the study period is presented in Appendix 2.6. During the 2017/18 growing season, the amounts of Ca^{2+} applied varied between 112.3 kg/ha at the backslope SLD plot and 634.9 kg/ha at the footslope DLD plot (Table 2.10). The amounts applied during the 2017/18 season were considerably higher than the LTM due to a high concentration of Ca^{2+} present in the wastewater (data not shown). According to Saayman (1981), grapevines annually require *ca*. 2 kg Ca^{2+} per tonne of grapes produced. Based on this recommendation, the treated municipal wastewater supplied more than adequate amounts of Ca^{2+} to all of the experiment plots throughout the 12-year study period if the grape yields amounted to 10 t/ha. The application of excess Ca^{2+} may also be beneficial in mitigating the possible negative effects of Na^{+} applied *via* wastewater irrigation due to its role in decreasing the SAR.

Magnesium: A minimum of 6.9 kg/ha Mg^{2+} was applied at the footslope SLD plot and a maximum of 130.8 kg/ha was applied at the footslope DLD plot over the course of the 12-year study period (Appendix 2.7). During the 2017/18 season, the amounts of Mg^{2+} applied *via* treated municipal wastewater irrigation varied between 16.8 kg/ha at the backslope SLD plot and 95.0 kg/ha at the footslope DLD plot (Table 2.11). According to Conradie (1981a), grapevines require an annual amount of 0.6 kg Mg^{2+} for each tonne of grapes produced. Therefore, the treated municipal wastewater was able to supply more than adequate amounts of Mg^{2+} to meet the grapevine's requirements.

Table 2.11. Amounts of magnesium (Mg^{2+}), sodium (Na^{+}) and chloride (Cl^{-}) applied *via* treated municipal wastewater (kg/ha) used to irrigate vineyards during the 2017/18 growing season in relation to the long-term mean (LTM).

Landscape position	Treatment	Mg^{2+}	LTM	Na^{+}	LTM	Cl^{-}	LTM
Shoulder	SLD	18.7	11.6	292.8	174.0	388.0	214.2
	DLD	37.5	23.3	585.7	347.9	776.0	428.3
Backslope	SLD	16.8	12.4	262.7	183.9	348.0	228.1
	DLD	33.6	24.9	525.3	367.7	696.0	456.2
Footslope	SLD	47.5	20.3	742.4	322.1	983.6	375.3
	DLD	95.0	40.5	1484.8	644.2	1967.2	750.6

Sodium: The amounts of Na^{+} applied during the 2017/18 season varied between 262.7 kg/ha at the backslope SLD plot and 1 484.4 kg/ha at the footslope DLD plot (Table 2.11). Over the course of the 12-year study period, more than 6 t/ha Na^{+} was applied at the footslope DLD plot (Appendix 2.8), with an average of 644.2 kg/ha applied each year. Since Na^{+} is not considered an essential element for grapevine growth (Winkler *et al*., 1974), no threshold value with regard to amount of Na^{+} applied to vineyards exist (Howell, 2016). However, excessive application of Na^{+} may reduce vegetative growth, yield and suppress Ca^{2+} uptake (Myburgh & Howell, 2014c and references therein), as well as reduce soil *K* and IR (Halliwell *et al*., 2001).

Chloride: The total amounts of Cl^{-} applied *via* irrigation with treated municipal wastewater from 2006 to 2017 varied between 2 182.06 kg/ha and 7 040.20 kg/ha (Appendix 2.9). The mean amount of Cl^- applied annually ranged between 214.2 kg/ha at the shoulder SLD plot and 750.6 kg/ha at the footslope DLD plot (Table 2.11). During the 2017/18 growing season, *ca*. 2 t/ha Cl^- was applied *via* wastewater irrigation at the footslope DLD. The application of excessive amounts of Cl^- to soils can impact grapevine water relations negatively, since grapevines have to take up water at high osmotic potential. Furthermore, Na^+ may also have a direct toxic effect (Saayman, 1981) that can affect vegetative growth (Myburgh & Howell, 2014c).

Bicarbonate: Annual amounts of HCO_3^- applied *via* treated municipal wastewater irrigation from 2006 to 2017 ranged between 140.6 kg/ha to 3 046.3 kg/ha (Appendix 2.10). The total amount applied at the footslope DLD plot over the course of study period amounted to almost 10 t/ha, whilst *ca*. 1 t/ha was applied at this plot during the 2017/18 growing season (Table 2.12). The large amounts of HCO_3^- applied through the irrigation water is alarming as it may lead to the precipitation of insoluble Ca- and Mg-carbonates when the soils dry out, resulting in the removal of Ca^{2+} and Mg^{2+} from the soil solution (Van Zyl, 1981). This, in turn will increase relative Na^+ levels and subsequently lead to higher SAR levels which may have an impact on soil physical properties (Van Zyl, 1981).

Table 2.12. Amounts of bicarbonate (HCO_3^-), and sulfate (SO_4^{2-}) applied *via* treated municipal wastewater (kg/ha) used to irrigate vineyards during the 2017/18 growing season in relation to the long-term mean (LTM) and standard deviation (SD).

Landscape position	Treatment	HCO_3^-	LTM	SO_4^{2-}	LTM
Shoulder	SLD	586.2	293.7	157.8	118.8
	DLD	1172.4	587.4	315.5	237.6
Backslope	SLD	525.8	316.0	141.5	118.8
	DLD	1051.6	632.0	283.0	237.6
Footslope	SLD	1486.0	516.6	399.9	189.3
	DLD	2972.0	1033.2	799.9	378.6

Sulfate: Irrigation with treated municipal wastewater annually applied between 45.4 kg/ha and 1 329.1 kg/ha SO_4^{2-} over the course of the 12-year study period (Appendix 2.11). On average, *ca*. 126.2 kg/ha and 284.6 kg/ha SO_4^{2-} was applied during the 2017/18 growing season at the SLD and DLD treatments, respectively (Table 2.12).

Trace elements: The amounts of B^{3+}, Cu^{2+}, Fe^{3+}, Mn^{2+} and Zn^{2+} applied *via* treated municipal wastewater irrigation was extremely low (Appendices 2.12, 2.13, 2.14, 2.15 & 2.16) and will therefore not be discussed further.

2.3.3. Soil chemical properties

2.3.3.1. $pH_{(KCl)}$

The $pH_{(KCl)}$ of the 0-30 cm soil layer increased at all of the landscape positions following 11 years of treated municipal wastewater irrigation (Fig. 2.7). At the shoulder site, the DLD plot had the highest topsoil pH at the end of the study period, followed by the SLD and RF plots (Fig. 2.7A), although the differences between the three treatments were relatively small. In contrast, the topsoil pH of the SLD and DLD plots at the backslope site was similar, whilst the RF was considerably lower (Fig. 2.7B). The topsoil pH of the backslope SLD plot increased by approximately 2.3 units compared to the baseline before irrigation commenced. Similar results were obtained at the footslope site (Fig. 2.7C). At this site, the topsoil pH of the RF treatment remained unchanged, but decreased with depth when compared to the baseline.

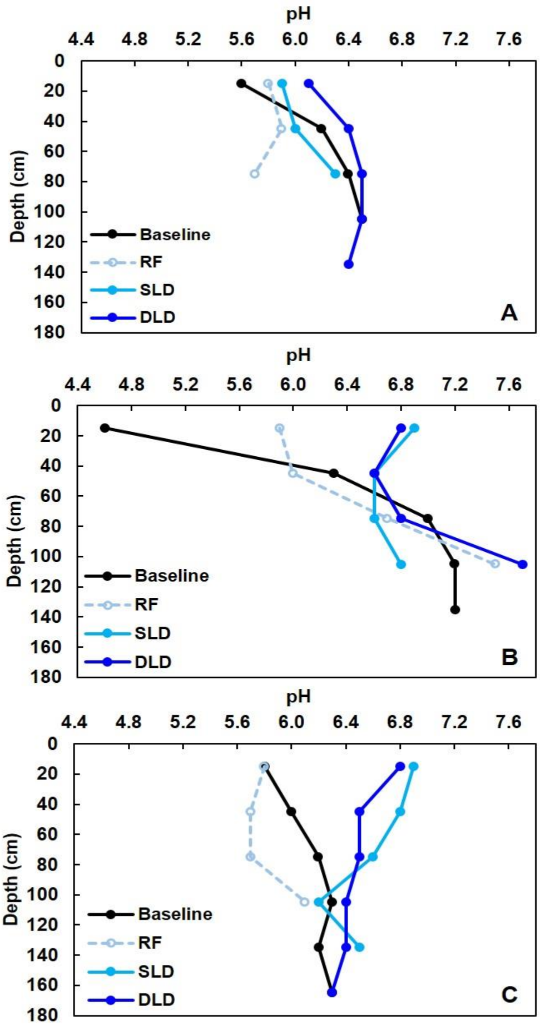

Figure 2.7. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil pH on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

On average, the topsoil pH of the SLD and DLD plots increased by 1.3 units after 11 years of irrigation with treated municipal wastewater (Fig. 2.8). The increase in pH was most likely due to the pH of the irrigation water which varied between 6.7 and 8.0 over the course of the study period. The decarboxylation and hydrolysis of organic acids and bicarbonate anions present in the wastewater may also have contributed to the increased pH (Li *et al.*, 2008). The increased pH did not cause concern as it remained near neutral and would therefore have little effect on biological functioning (Schipper *et al.*, 1996). In addition, the pH at all of the experiment plots were within the

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recommended range of 5.0 to 7.5 to sustain optimal grapevine growth (Saayman, 1981). Similar results have been reported by Sparling *et al*. (2006) and Schipper *et al*. (1996) where soils were irrigated with secondary and tertiary treated municipal wastewater, respectively. In contrast, Xu *et al*. (2010) reported a decrease in soil pH of *ca.* 1.1 units in a 150 cm soil profile following 20 years of irrigation with treated municipal wastewater.

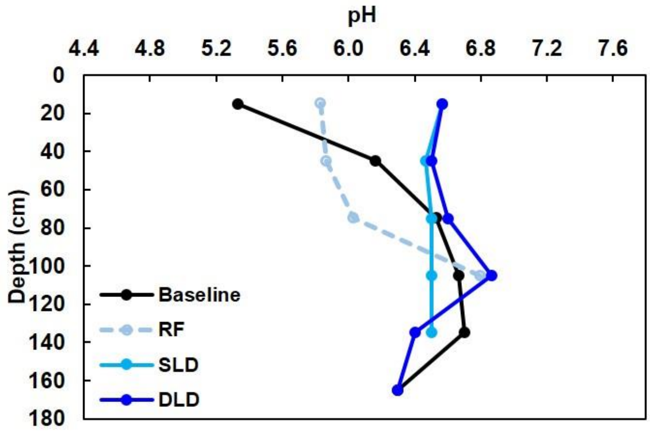

Figure 2.8. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean soil pH across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.2. Electrical conductivity

The EC_e of the 0-30 cm soil layer increased in all of the treatments at each of the landscape positions compared to the baseline values before irrigation commenced (Fig. 2.9). The topsoil EC_e of the shoulder DLD plot was nearly three times higher than the baseline value (Fig. 2.9A). In addition, it was the highest EC_e measured across all three landscape positions. However, no clear trends could be observed in the subsoil between the treatments of the shoulder site. At the backslope site, the SLD treatment had the highest topsoil EC_e. However, the differences between the treatments and the baseline measurement was relatively small (Fig. 2.9B). An increase in EC_e with soil depth was also evident at this site. The topsoil EC_e of the footslope DLD plot increased from 0.17 dS/m before irrigation with treated municipal wastewater began to 0.56 dS/m following 11 years of irrigation (Fig. 2.9C).

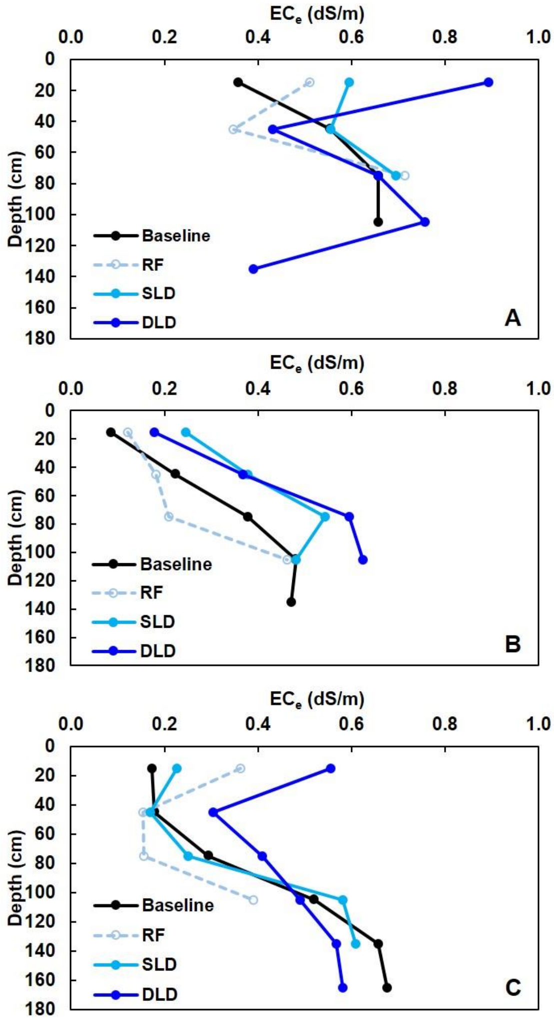

Figure 2.9. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil electrical conductivity (EC_e) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

The mean topsoil EC_e increased with the amount of treated municipal wastewater applied (Fig. 2.10). However, there were no clear trends in the deeper soil layers that could be related to the different irrigation treatments compared to the baseline. The increased EC_e of the topsoil indicates an accumulation of salts at the soil surface. The EC_w of the treated municipal wastewater ranged between 0.7 dS/m and 1.2 dS/m which could explain the increased EC_e. The accumulation of salts at the soil surface is most likely a result of high evapotranspiration during the irrigation season which concentrated the salts in the upper parts of the root zone (Rhoades *et al*., 1973). Similar results were reported for vineyard soils in Great Western, Australia which were irrigated with treated municipal wastewater for at least five years (Hermon, 2011). The accumulation of salts in the soil profile is of concern as a progressive increase in soil salinity can result in grapevine nutrient deficiencies (McCarthy, 1981; Paranychianakis *et al*., 2006). However, the relatively small increase in EC_e after 11 years of wastewater irrigation suggests that winter rainfall could have leached some of the applied salts beyond the measured depth. In a laboratory study where rainfall

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cycles were simulated, the EC of the drainage water was considerably higher than that of the input water (Laurenson, 2010), indicating a loss of salts from the soil during rainfall events. Therefore, regular rainfall events may help to alleviate high soil EC_e where municipal wastewater containing high levels of salts are used for irrigation.

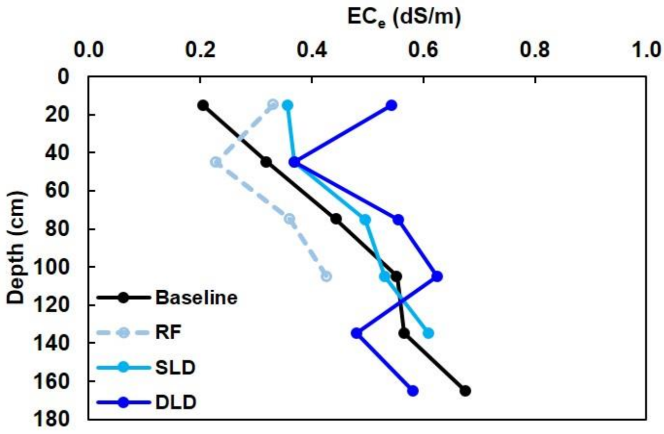

Figure 2.10. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean soil electrical conductivity (EC_e) across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.3. Phosphorous (Bray II)

Bray II P increased in the 0-30 cm soil layer at all of the experiment plots following 11 years of treated municipal wastewater irrigation (Fig. 2.11). The soil P content at the shoulder site was highest at the SLD, followed by the RF plot whilst the P content at the DLD plot was approximately half that of the SLD (Fig. 2.11A). The SLD and RF plots at the shoulder site exceeded the norm of 30 mg/kg P recommended for grapevines in soils containing more than 15% clay (Conradie, 1994). The Bray II P concentration in the 0-30 cm soil layer of the SLD and DLD plots at the backslope site was slightly less than the RF (Fig. 2.11B), however all plots met the norm of 30 mg/kg. The RF plot at the footslope site had the highest P content in the 0-30 cm soil layer compared to the other treatments (Fig. 2.11C). The P content of the RF plot increased from 7 mg/kg measured before the study commenced to 46 mg/kg 11 years thereafter.

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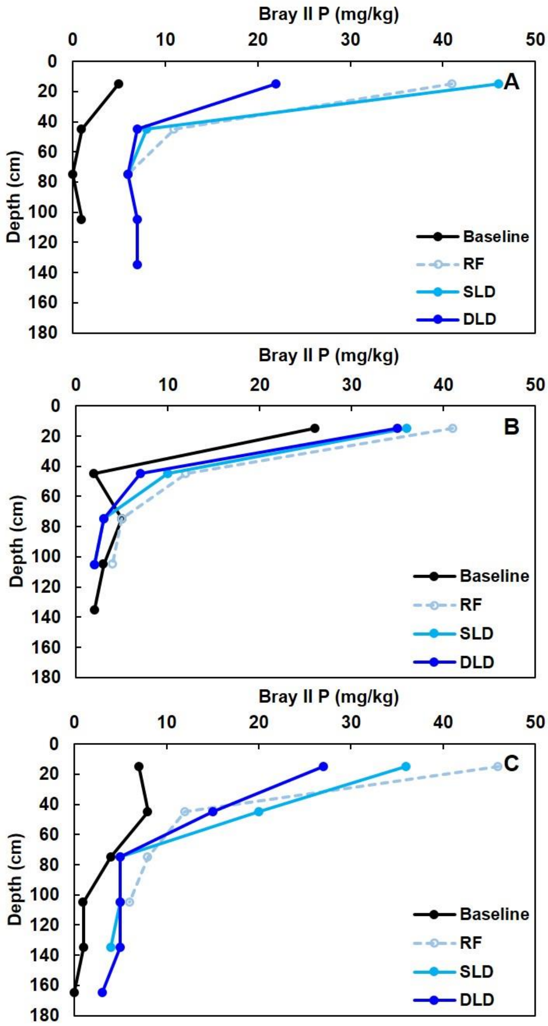

Figure 2.11. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil Bray II extractable phosphorous (P) content on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

On average, after 11 years of irrigation, the soil contained 42.6 mg/kg, 39.3 mg/kg and 28.1 mg/kg Bray II P at the RF, SLD and DLD treatments, respectively, whilst the baseline value was 12.7 mg/kg. An accumulation of P in the topsoil was evident as the concentration decreased sharply in the 30-60 cm soil layer for all of the treatments and remained at relatively constant levels in the 60-90 cm soil layer (Fig. 2.12). The 0-30 cm soil layers of the RF and SLD treatments consistently exceeded the norm of 30 mg/kg P for soils with a clay content of at least 15%. In contrast, the DLD treatments were on average only 2 mg/kg below the norm. The accumulation of P in the topsoil of the RF treatments could possibly be explained by the application of P fertiliser by the grower and the low vigour that is expected of grapevines that are grown under dryland conditions which would absorb very small amounts of P from the soil. Conversely, the high vigour that is expected of over-irrigated grapevines is reflected by the lower soil P content at the DLD plots, despite the application of additional P *via* treated municipal wastewater irrigation.

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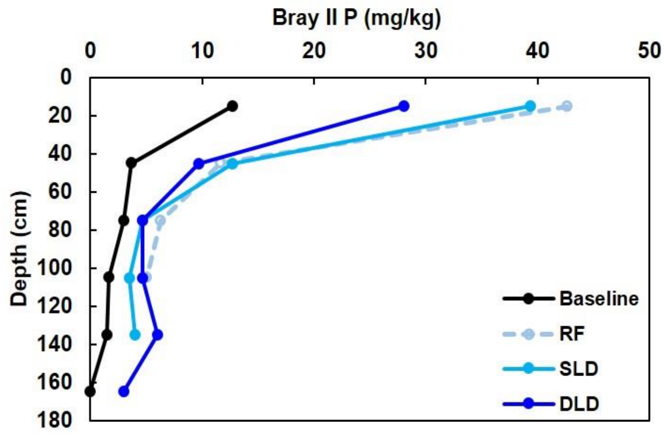

Figure 2.12. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean Bray II extractable phosphorous (P) content across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.4. Potassium (Bray II)

The Bray II K^+ content of the 0-30 cm soil layer at the shoulder site increased in all of the treatment plots compared to the baseline values (Fig. 2.13A). The slightly increased K^+ content at the RF plot of the shoulder site was probably due to the application of K^+ fertiliser by the grower, whereas the high K^+ content at the DLD plot may be the result of low mobility of K^+ in the soil and its retention by clay minerals (Pérez *et al*., 2015), as well as an over-supply of K^+ *via* treated municipal wastewater irrigation. At the backslope site, the K^+ content of the 0-30 cm soil layer did not differ substantially between treatment plots but increased in all of the plots compared to the baseline (Fig. 2.13B). The K^+ content of the SLD and DLD plots were maintained in the deeper soil layers, whereas a gradual decrease up to 90 cm was observed at the RF plot. Bray II K^+ levels in the 90-120 cm soil layer at the backslope site decreased in all of the treatment plots compared to the baseline. This was probably due to K^+ uptake by grapevines from deeper soil layers, or the leaching of K^+ beyond the measured depth. The K^+ content of the topsoil layer at the footslope increased under DLD compared to the baseline (Fig. 2.13C). Similar to the backslope site, K^+ levels beyond 90 cm decreased below the baseline levels in all of the treatments.

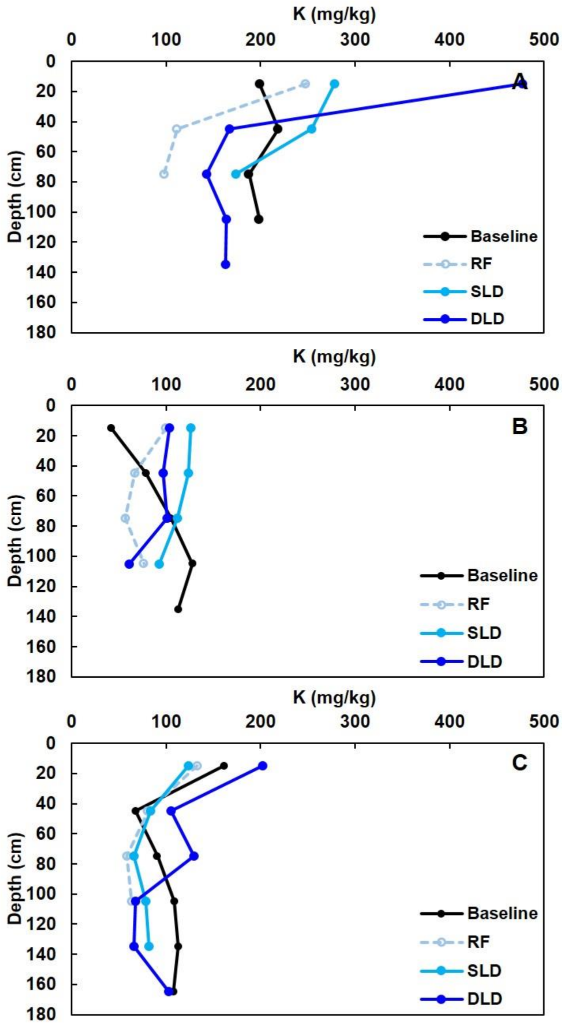

Figure 2.13. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil Bray II extractable potassium (K) content on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

On average, the Bray II K^+ content of the 0-30 cm soil layer increased by 26 mg/kg, 42 mg/kg and 127 mg/kg for the RF, SLD and DLD treatments, respectively (Fig. 2.14). An accumulation of K^+ in the topsoil due to municipal wastewater irrigation has previously been reported by Heidarpour *et al*. (2007), Kiziloglu *et al*. (2007) and Singh *et al*. (2012). The high K^+ content under DLD is of concern since an over-supply of K^+ to grapevines may result in excessive K^+ uptake. This may lead to musts with high pH and malate concentrations, as well as poor colour in red wines (Mpelasoka *et al*., 2003). In addition, an accumulation of K^+ in the soil may have deleterious effects on soil structure (Laurenson *et al*., 2012) and may negatively impact soil *K* and IR due to its effects on clay dispersal (Arienzo *et al*., 2009a). However, clay dispersion is highly dependent on the electrolyte concentration of the infiltrating water (Shainberg *et al*., 1981). Therefore, the high salinity that is often associated with wastewater might mitigate the negative effects of K^+ on aggregate stability and *K* (Arienzo *et al*., 2009a). However, as long as the EC_w of the treated

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municipal wastewater remains above the critical coagulation value, no soil dispersion is expected to occur (Abedi-Koupai *et al*., 2006).

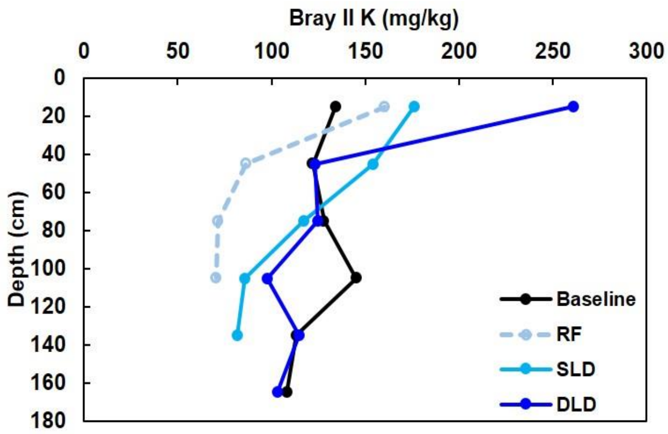

Figure 2.14. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean Bray II extractable potassium (K) content across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.5. Extractable cations

2.3.3.5.1. Calcium

Soil Ca^{2+} did not show any consistent trends at the various landscape positions that could be related to the different irrigation treatments (data not shown). The mean soil Ca^{2+} levels of the 0-30 cm soil layer increased only slightly for all of the treatments when to the baseline (Fig. 2.15). The lack of substantial response could be explained by the small amounts of Ca^{2+} applied *via* treated municipal wastewater, the uptake of Ca^{2+} by grapevines and the leaching from the soil profile through deep percolation. Similar results were presented by Duan *et al*. (2010) for a sandy clay loam soil after one year of irrigation with secondary treated municipal wastewater. Irrigating golf course fairways with a clay loam topsoil with treated municipal wastewater for four to five years also did not affect soil Ca^{2+} (Qian & Mecham, 2005).

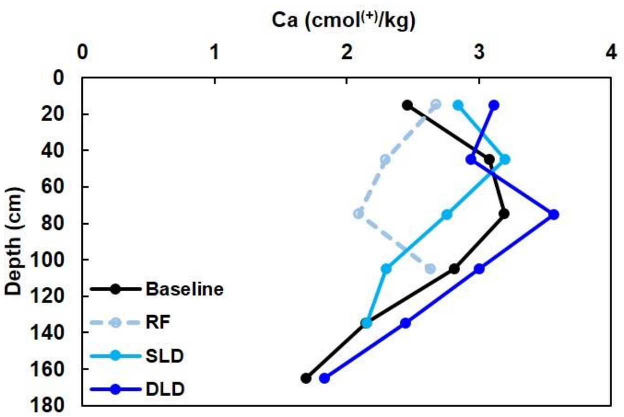

Figure 2.15. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean extractable calcium (Ca) across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.5.2. Magnesium

Similar to Ca^{2+}, soil Mg^{2+} levels did not show any consistent trends at the various landscape positions that could be related to the different irrigation treatments (data not shown). The mean soil Mg^{2+} concentrations of the 0-30 cm soil layer at the RF and SLD treatments decreased when compared to the baseline values, whilst the DLD remained at relatively unchanged after 11 years of irrigation using treated municipal wastewater (Fig. 2.16). This is probably due to small amounts of Mg^{2+} supplied to the soil and the uptake of Mg^{2+} by grapevines which depleted soil Mg^{2+} levels over the long-term. Neilsen *et al*. (1991) attributed the reduction of soil Mg^{2+} following five years of municipal wastewater irrigation to mass exchange by Na^+ and K^+.

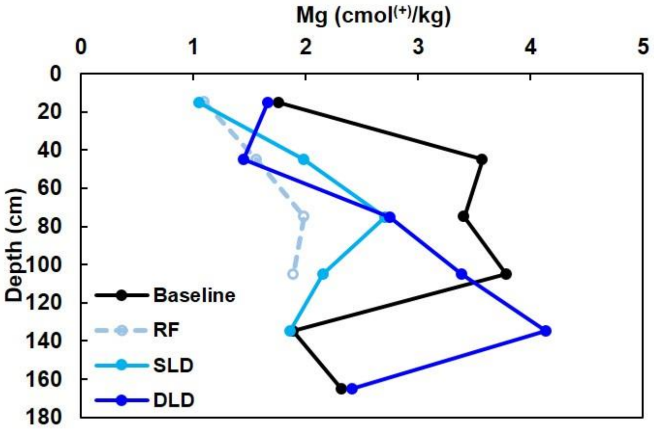

Figure 2.16. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean extractable magnesium (Mg) across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.5.3. Potassium

Soil extractable K^+ followed similar trends as was observed for Bray II K^+ (data not shown), therefore it will not be discussed. Since exchangeable K^+ was not determined, the extractable potassium percentage (EPP´) was calculated as opposed to the exchangeable potassium percentage (EPP). Conradie (1994) recommended a ratio of 3% to 4% for exchangeable K^+ to other cations. The baseline EPP´ in the 0-30 cm soil layer of the shoulder site already exceeded this norm and a further increase in all of the treatments was observed over the course of the study period (Fig. 2.17A). These results were expected, as the topsoil at this particular site had a high clay content which could retain a high amount of extractable K^+. At this site, the EPP´ of the DLD plot was *ca*. 2% higher than the SLD plot and 3% higher than the RF plot, indicating a steady increase in EPP´ due to irrigation with treated municipal wastewater. The EPP´ of the 0-30 cm soil layer of the backslope site increased in all the treatment plots compared to the baseline with the RF plot having the highest EPP´ followed by the SLD and DLD plots (Fig. 2.17B). The accumulation of extractable K^+ at the RF plot could be explained by low vigour and grape production that is associated with grapevines grown under dryland conditions and the subsequent

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lower uptake of K^+ from the soil. The topsoil EPP´ of the backslope DLD plot remained similar to the baseline and the SLD plot only increased by 0.8% following 11 years of wastewater irrigation. The lack of response to wastewater irrigation in terms of EPP´ at this site might be explained by the uptake of K^+ by grapevines and the leaching of excess K^+ from the soil profile as the clay content at this site is considerably lower than the shoulder site (Appendix 3). At the footslope site, the baseline EPP´ of the 0-30 cm soil layer was 10% and increased by 1% and 2% at the SLD and DLD plot, respectively, the RF plot remained unchanged (Fig. 2.17C). The higher EPP´ was expected at the DLD plot of this site due to the larger volume of K-containing wastewater applied at this site. The EPP´ decreased with depth in all of the treatment plots, but increased in relation to the baseline in the 30-60 cm soil layer and reached levels below the baseline at a depth of 90 cm. The higher EPP´ in the 30-60 cm layer could be explained by the higher clay content of this layer. The decrease in deeper layers may be due to a combination of grapevine K^+ uptake and K^+ leaching from the soil profile.

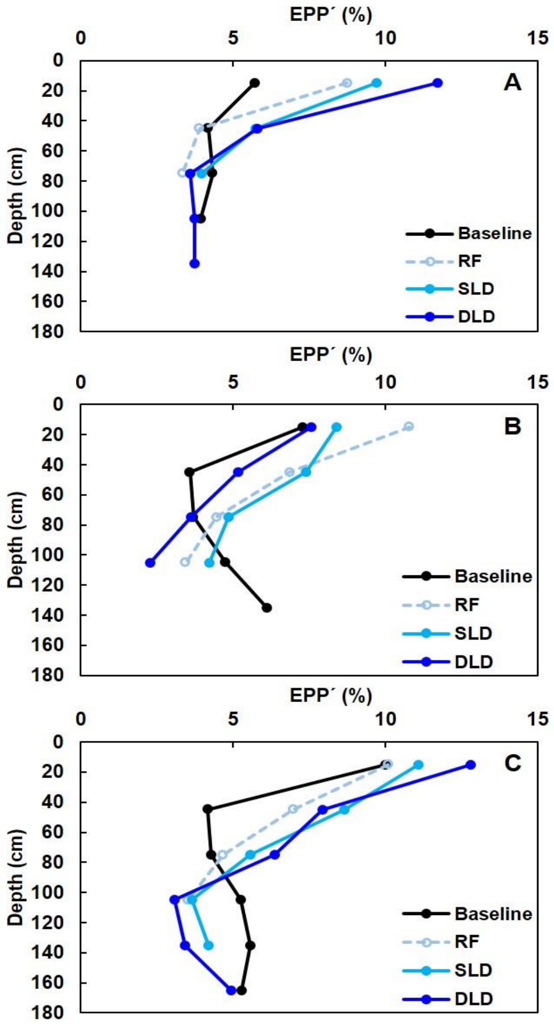

Figure 2.17. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil extractable potassium percentage (EPP´) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

The mean EPP´ for the baseline exceeded the recommended norm by far and increased substantially in all of the treatments over the course of the study period (Fig. 2.18). There was little difference between the topsoil EPP´ of the RF and SLD treatments but the DLD was 2% higher than the other treatments. The high extractable K^+ at all of the experiment sites are concerning as it could lead to greater K^+ uptake by grapevines which may ultimately result in unstable wines with high pH (Gawel *et al*., 2000; Mpelasoka *et al*., 2003). In addition, high amounts of exchangeable K^+ have been associated with reduced *K* (Quirk & Schofield, 1955; Chen *et al*., 1983).

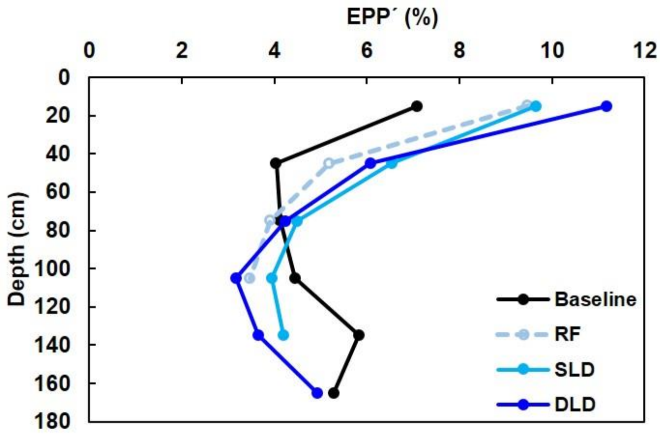

Figure 2.18. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean soil extractable potassium (EPP´) across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.5.4. Sodium

Soil extractable Na^+ followed similar trends as was observed for ESP´ (data not shown), therefore only the latter will be discussed. The ESP´ of the 0-30 cm soil layer at the shoulder site decreased at the RF plot, but levels similar to the baseline were maintained at the SLD and DLD plots (Fig. 2.19A). An increase in ESP´ was observed with depth in all of the plots. However, the values were similar between treatment plots and lower than the baseline. An increase in ESP with depth in three Israeli soils irrigated with treated municipal wastewater with SAR similar to the present study, was attributed to an increase in clay content at deeper soil layers (Levy *et al*., 2014). The reason for the high baseline ESP´ values in the subsoil layers of the shoulder site could be explained by increased weathering of clay minerals due to soil preparation. Irrigation with treated municipal wastewater increased the ESP´ throughout the soil profile at the backslope site (Fig. 2.19B). However, little difference in Na^+ accumulation could be seen between the SLD and DLD plots. The values measured at the RF plot of the backslope site at the end of the study period remained comparable to the baseline. Similar results were observed where olive orchards were irrigated with treated municipal wastewater and compared to a rainfed control treatment (Ayoub *et al*., 2016). The ESP´ at the footslope site followed a similar trend to the backslope site (Fig. 2.19C) with the RF plot remaining largely unaffected and the SLD and DLD plots increasing substantially

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following 11 years of irrigation with treated municipal wastewater. However, the SLD plot at the footslope site had greater ESP´ values in the subsoil compared to the DLD plot, indicating more accumulation of salts in the subsoil under SLD. The increase in ESP´ at the backslope and footslope sites are particularly concerning as the SAR of the irrigation water largely remained below 5, which is the critical limit for wastewater used for irrigation (DWA, 2013). Similarly, Levy *et al*. (2014) reported ESP levels in sandy clay subsoils (≥15% clay) reaching between 6% and 16% following ten years of irrigation with treated municipal wastewater having SAR of 3 to 5. They attributed the accumulation of Na^+ to a possible lack of chemical equilibrium between the SAR of the treated wastewater, the SAR of the soil solution and the ESP of the subsoil layers. The replacement of exchangeable Na^+ (applied *via* wastewater irrigation) in the topsoil by Ca^{2+} originating from the dissolution of calcium carbonate during the rainy season and the subsequent leaching of Na^+ to deeper soil layers where it is re-adsorbed to the soil exchange complex was proposed as a possible mechanism for the Na^+ accumulation (Levy *et al*., 2014). Results presented by Myburgh (2018) indicated that Na^+ accumulation in the soil profiles of these sites were highly dependent on the winter rainfall, implying that salts will accumulate in the soil during winters with low rainfall and leach to deeper layers following higher rainfall. The winter rainfall preceding the collection of the soil samples in 2017 was 138 mm (data not shown), which was lower than the LTM of 160 mm and could therefore contribute to the accumulation of Na^+ at the wastewater irrigated sites.

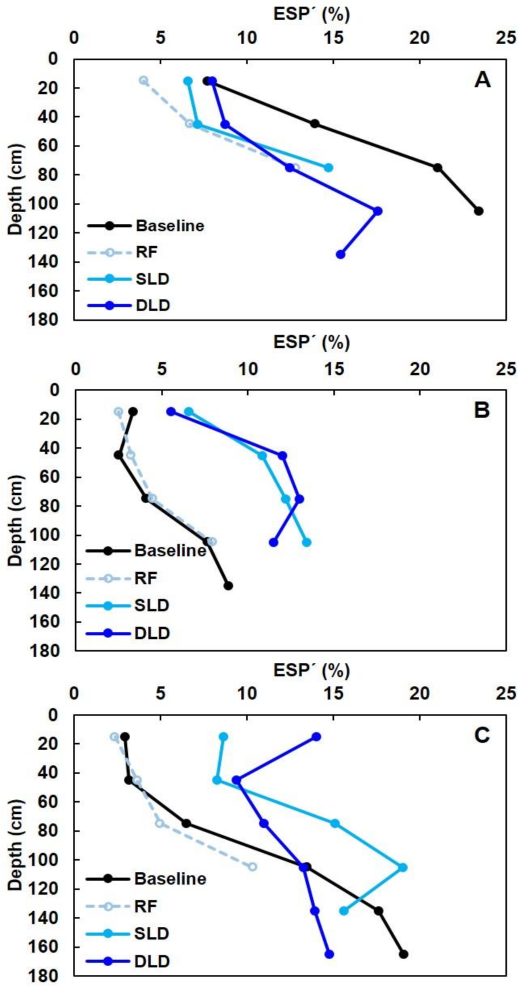

Figure 2.19. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil extractable sodium percentage (ESP´) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

On average, the topsoil ESP´ increased by 1% and 3% for the SLD and DLD treatments, respectively, whereas the RF treatment was 3% lower compared to the baseline (Fig. 2.20). A steady increase in ESP´ with depth was evident for all of the treatments. This is probably due to high Na^+ levels present at the beginning of the study period as the subsoil ESP´ decreased in relation to the baseline with the exception of the SLD at 60 cm and 150 cm as well as the DLD at 150 cm soil depth. Deeper than 60 cm the mean ESP´ of the SLD plots were higher than the DLD. This could possibly be explained by the larger volumes of irrigation water applied at the latter plots which facilitated the leaching of more Na^+ from the soil profile. The importance of leaching salts from the soil profile have been highlighted previously (Hussain, 1981). Rengasamy and Olsson (1993) predicted that Na^+ would accumulate in a soil if the SAR of the applied water is greater than 3 and the leaching fraction is less than 50%. The high ESP´ observed in the subsoil of the SLD and DLD treatments remains a concern as it may reduce the movement of water through the soil profile. Lower macro-porosity due to an accumulation of Na^+ and K^+ in the soil may affect the drainage capacity of soils which in turn limits water percolation and ultimately the leaching of salts

(Prior *et al*., 1992; Halliwell *et al*., 2001). In a soil with a permeable A horizon overlying a moderately draining B horizon, irrigation with treated municipal wastewater caused a reduction in *K* of the soil due to the increase of exchangeable Na^+ in the B horizon which reduced the leaching of salts and lead to increased soil salinity in the A horizon (Stevens *et al*., 2003). Soil permeability problems will also occur if a solution with very low electrolyte concentration, such as rainwater, percolates through the soil (Du Plessis & Shainberg, 1985). Therefore, the application of large volumes of higher salinity water, as is the case at the DLD treatments, might help to mitigate the accumulation of Na^+ at the soil surface and prevent reductions in *K* and IR.

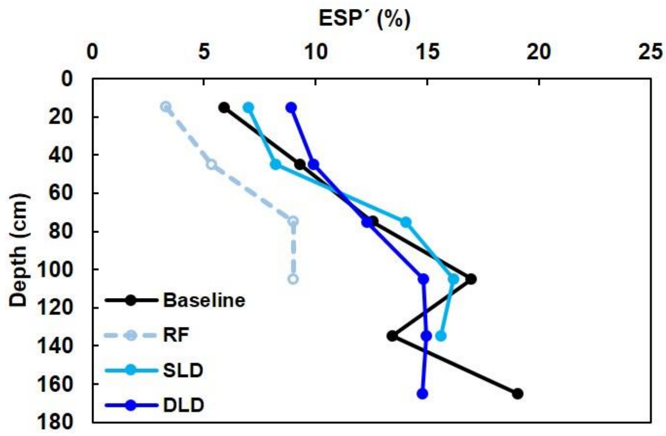

Figure 2.20. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean soil extractable sodium (ESP´) across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

2.3.3.6. Chloride

The soil Cl^- content was not determined at the beginning of the study period; therefore, no baseline value is available. Soil Cl^- concentrations in the 0-30 cm layer increased with the amount of irrigation water applied at all the experiment plots (Fig. 2.21). The topsoil Cl^- content at the shoulder site was 17 mg/kg, 35 mg/kg and 79 mg/kg for the RF, SLD and DLD plots, respectively (Fig. 2.21A). Soil Cl^- levels decreased sharply in the 30-60 cm layer at this landscape position and values were comparable between the three treatment plots. In the 60-90 cm soil layer, the Cl^- concentration of all the treatments increased again, suggesting that the subsoil has inherently high Cl^- levels. Similar results were observed at the backslope site, however Cl^- content increased rather than decreased in the 30-60 cm soil layer (Fig. 2.21B). At the footslope site, the soil Cl^- content of the SLD and DLD plots were consistently higher than the RF plot and reached similar concentrations to what was observed at the shoulder and backslope sites (Fig. 2.21C).

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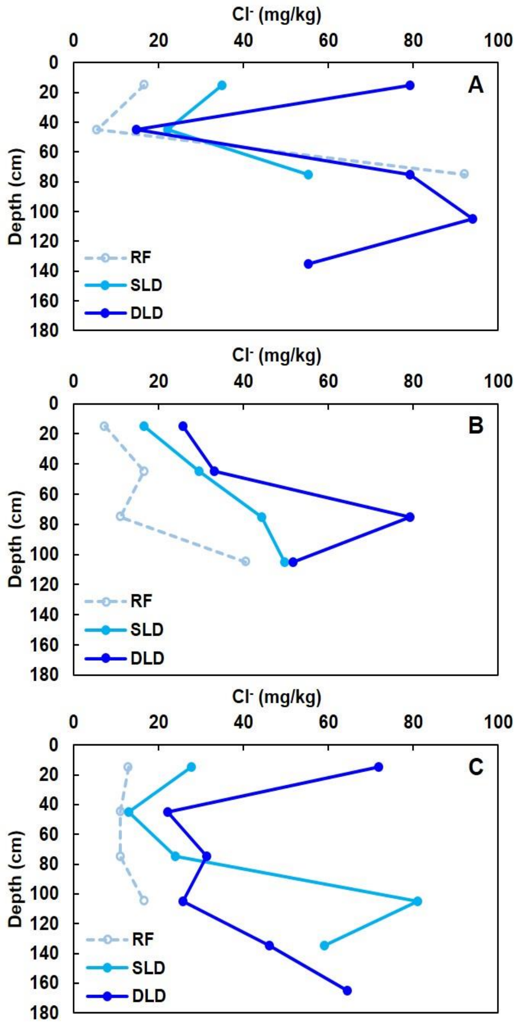

Figure 2.21. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil chloride content (Cl) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of treated municipal wastewater irrigation.

The mean Cl^- content of the topsoil increased with the amount of treated municipal wastewater applied (Fig. 2.22). The high Cl^- levels were to be expected as the wastewater is disinfected by a chlorination treatment at the WWTW, resulting in a mean Cl^- content in the wastewater of 160 mg/ℓ over the 12-year study period. An accumulation of Cl^- seems evident at a depth of 90 cm but cannot necessarily be ascribed to the irrigation water as high Cl^- levels were observed in the subsoil of the RF treatments as well.

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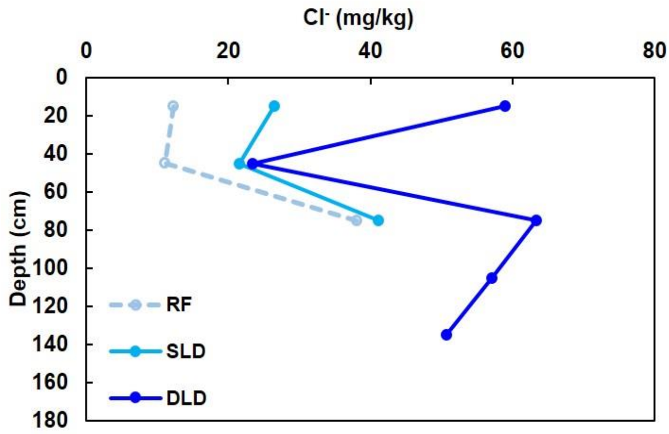

Figure 2.22. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean soil chloride (Cl) content across main experiment sites after 11 years of treated municipal wastewater irrigation.

2.3.3.7. Soil organic carbon

With the exception of the SLD plot at the footslope site, the SOC content of the 0-30 cm soil layer increased in all of the treatment plots at all of experiment sites compared to the baseline values (Fig. 2.23). The accumulation of SOC could not be ascribed to the application of treated municipal wastewater since the RF plots had the highest SOC content at each of the landscape positions. On average, the SOC content was 0.6% at the RF treatment, followed by 0.5% at the DLD and 0.4% at the SLD treatment (Fig. 2.24). An increase in SOC was also observed in the subsoil at the end of the study period, but no clear trend with regards to the different treatments could be seen. It should be noted that the chemical oxygen demand (COD) of the treated municipal wastewater was very low and would therefore not have made a significant contribution towards the SOC content (ARC, unpublished data). The increased SOC content could be explained by the accumulation of organic matter due to the annual establishment of a cover crop. The accumulation of grapevine plant material debris over the course of the 11-year study period also contributed to higher SOC levels. The greater accumulation of SOC at the RF treatments could be explained by a lack of water that is needed to facilitate the decomposition of organic matter. Herpin *et al*. (2007) reported significant reductions in soil organic matter (SOM) due to the stimulation of soil microbial activity where soils were irrigated with secondary treated municipal wastewater. In contrast, an increase in total carbon was reported for the 0-10 cm layer of soils irrigated with treated municipal wastewater for 8 and 10 years when compared to a rainfed control (Xu *et al*., 2010). The results of the present study reflected a balance between accumulation and mineralisation of SOM following irrigation with treated municipal wastewater.

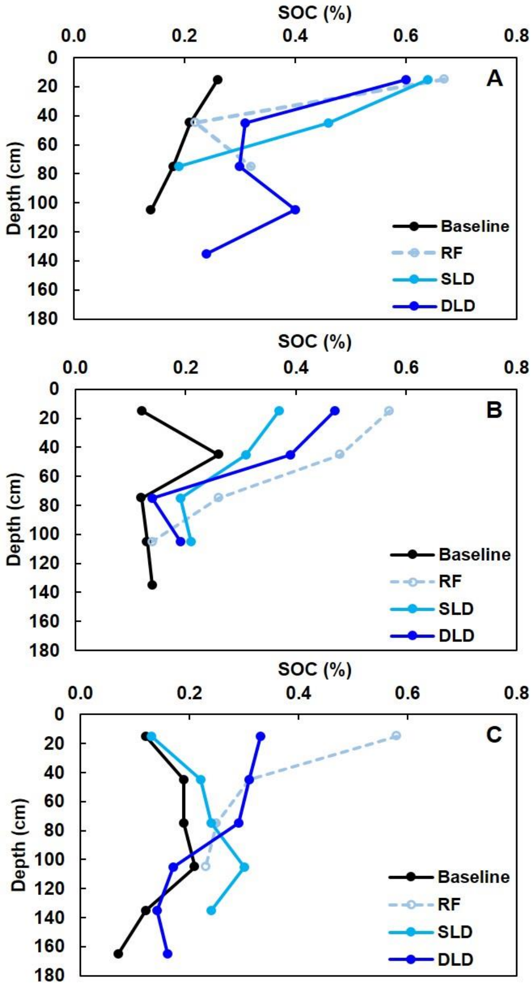

Figure 2.23. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil organic carbon content (SOC) on (A) a shoulder, (B) a backslope and (C) a footslope after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

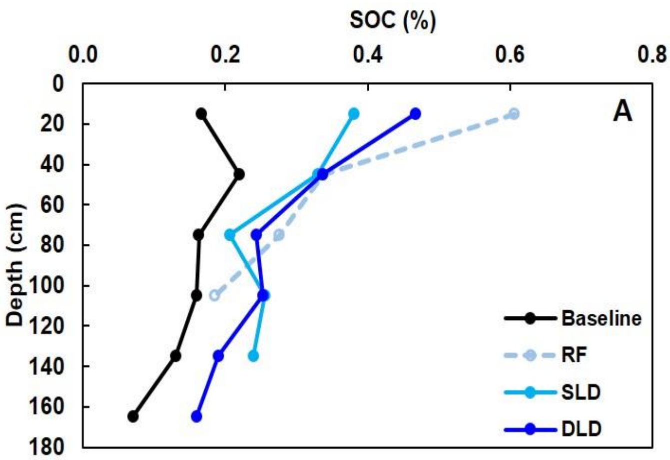

Figure 2.24. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the mean soil organic carbon (SOC) content across main experiment sites after 11 years of wastewater irrigation compared to the baseline before irrigation commenced (ARC, unpublished data).

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2.3.4. Soil physical properties

2.3.4.1. Soil texture

The particle size distribution, soil texture classification and stone fraction per 30 cm depth increment for each of the experiment plots are presented in Appendix 3. The soil texture at the shoulder site was relatively uniform and was predominantly clay to clay loam with a clay content that ranged between 35% and 51%. The soils at this site had a high stone fraction which ranged between 15.3% to 45.8% (Appendix 3), which was mainly due to the presence of the shale parent material which was brought up to the soil surface when the soil was prepared before planting. The backslope site had a sandy loam topsoil with a clay content of 15% to 19% and a sandy clay loam to clay layer. The clay content ranged between 29% and 49% at 90 cm. The soils at the backslope did not contain any stones (Appendix 3). The 0-30 cm layer of the footslope site ranged from being sandy loam at the RF and SLD plots to sandy clay loam at the DLD plot. The clay content of the footslope DLD plot was between 17% and 25% in the 0-30 cm layer and increased with depth to a maximum of 55% at 120 cm. The only presence of stones at the footslope site was in the 30-60 cm soil layer of the SLD plot (Appendix 3). As expected, the soils differed substantially between the three landscape positions. However, soils were relatively uniform between treatment plots at the individual landscape positions.

2.3.4.2. Near-saturation hydraulic conductivity

No clear trend was observed that could explain the effect of treated municipal wastewater irrigation on the K_{ns} at the shoulder site (Fig. 2.25A). However, the results were similar to what was reported by Walker and Lin (2008) for summit landscape positions irrigated with treated municipal wastewater for over 40 years. Similarly, there were little difference between the treatments at the footslope in terms of K_{ns}, despite the slightly lower K_{ns} measured at the DLD plot (Fig. 2.25C). These results were comparable to reports by Sparling *et al*. (2006) and Vogeler (2009) who observed no significant difference in K_{ns} between wastewater irrigated and non-irrigated soils. In contrast, at the backslope site K_{ns} decreased with an increase in the amount of treated municipal wastewater applied (Fig. 2.25B). The K_{ns} at the backslope site was 103 mm/h, 66 mm/h and 38 mm/h for the RF, SLD and DLD plots, respectively. Bedbabis *et al*. (2014) reported a significant decrease in the IR of a sandy soil (5.5% clay) following four years of treated municipal wastewater irrigation. The decrease was also significant with respect to a rainfed control treatment and one irrigated with well water (Bedbabis *et al*., 2014). In contrast, Lado and Ben-Hur (2010) reported improved saturated hydraulic conductivity (K_s) for a sandy soil (12% clay) irrigated with secondary treated municipal wastewater for more than 12 years. Furthermore, a strong correlation could be made between the EC_e of the 0-30 cm topsoil and the K_{ns}. The relationship between EC_e and K_{ns} was best described using a reciprocal-Y logarithmic-X model with which K_{ns} decreased significantly with an increase in EC_e up to an EC_e of 0.4 dS/m where after it was expected to plateau (Fig. 2.26). Conversely, Andrews *et al*. (2016) found no correlation between K_s and EC_e of loam soils irrigated with treated municipal wastewater for over 50 years.

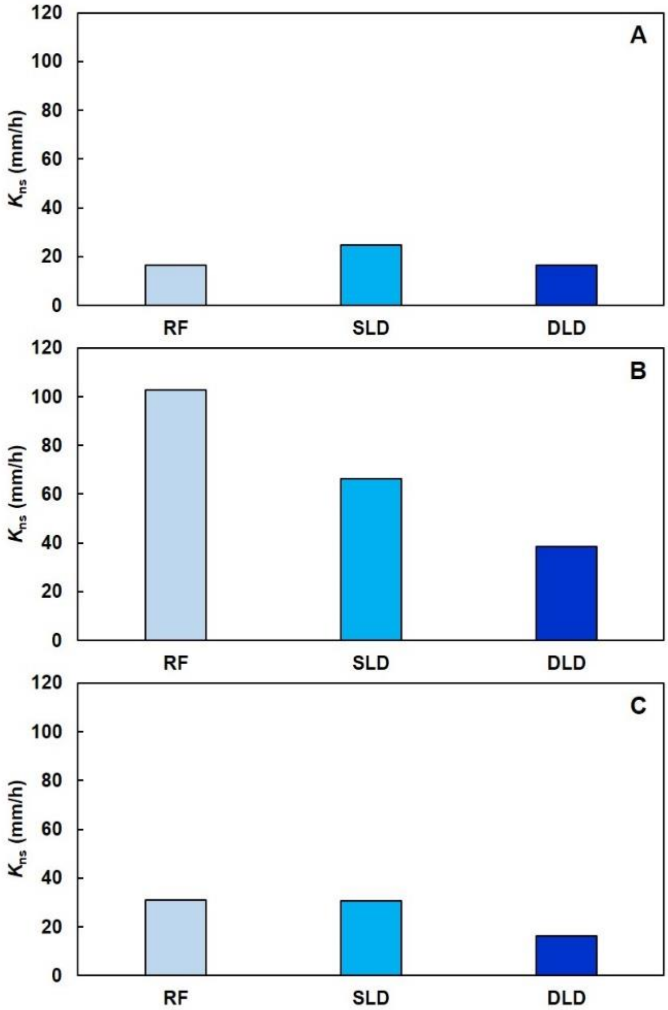

Figure 2.25. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the near-saturation hydraulic conductivity (K_{ns}) (suction = 2 cm) of (A) a shoulder, (B) a backslope and (C) a footslope during the 2017/18 season.

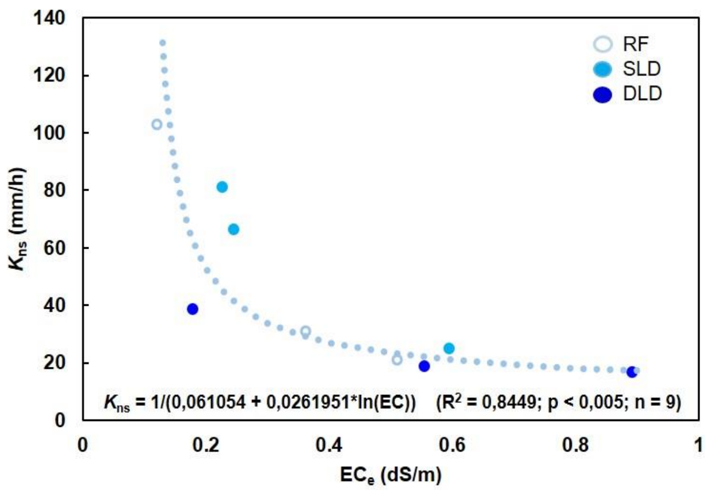

Figure 2.26. Effect of electrical conductivity (EC_e) of the 0-30 cm topsoil layer on near-saturation hydraulic conductivity (K_{ns}) during the 2017/18 season.

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2.3.5. Soil water content

The SWC of the irrigated treatment plots at the shoulder site was consistently higher than the RF plot, however the SLD and DLD plots maintained relatively similar SWC throughout the 2017/18 growing season (Fig. 2.27A). Above average rainfall during August 2017 (Fig. 2.6) resulted in increased SWC for all of the RF treatments (Fig. 2.27). The first irrigation of the season was applied in November 2017 and would therefore explain the slightly increased SWC of the irrigated treatments during that particular period. The SWC at the shoulder site decreased progressively during the summer months until April 2018 when 20 mm rainfall was recorded (Fig. 2.6). Thereafter the winter rainfall period began which increased the SWC at all of the experiment plots. Despite the higher clay content of the shoulder site, this landscape position had lower SWC compared to backslope and footslope sites (Fig. 2.27). This is likely a result of (i) the high stone fraction at the shoulder site (Appendix 3), which decreased the water holding capacity of the soil and (ii) the convex form of the landscape which facilitated the lateral movement of water through the soil to lower landscape positions.

The DLD plot at the backslope site had higher SWC compared to the SLD plot before the irrigation season commenced, where after it decreased to levels below what was measured at the SLD plot and only increased to similar SWC again in the winter of 2018 (Fig. 2.27B). Increased vegetative and reproductive growth under DLD irrigation would have increased the water requirements of grapevines and resulted in greater soil water depletion during the summer months compared to grapevines under SLD. No irrigation was applied at the backslope site during February and April 2018 (data not shown). This could explain the significant decrease in SWC of the irrigated treatments at this site during the post-harvest period (Fig. 2.27B).

The SWC of the irrigated plots at the footslope site followed similar trends as were observed at the backslope, but at lower levels of SWC (Fig. 2.27C). The SWC at the footslope site at the beginning of the season was 165 mm and 155 mm for the DLD and SLD treatment plots, respectively, whereas SWC values at the backslope site was 184 mm and 174 mm for the respective plots during the same time period (Fig. 2.27). The application of higher volumes of irrigation water at the footslope DLD plot was reflected by subtle changes in SWC throughout the season compared to the SLD plot which experienced more severe fluctuations in SWC (Fig. 2.27C). During the harvest period, the SWC of the footslope SLD plot decreased to levels below that of the RF plot. This was most likely due to strong vegetative growth and higher crop load which increased the grapevine water requirement and subsequently depleted soil water to 84 mm (Fig. 2.27C). Similar to the backslope site, no irrigation was applied at the footslope during April 2018 and would explain the significant decrease in SWC of the irrigated plots in early May (Fig. 2.27C). In contrast, the SWC of the RF treatments steadily increased during this time due to substantial rainfall during April, May and June 2018.

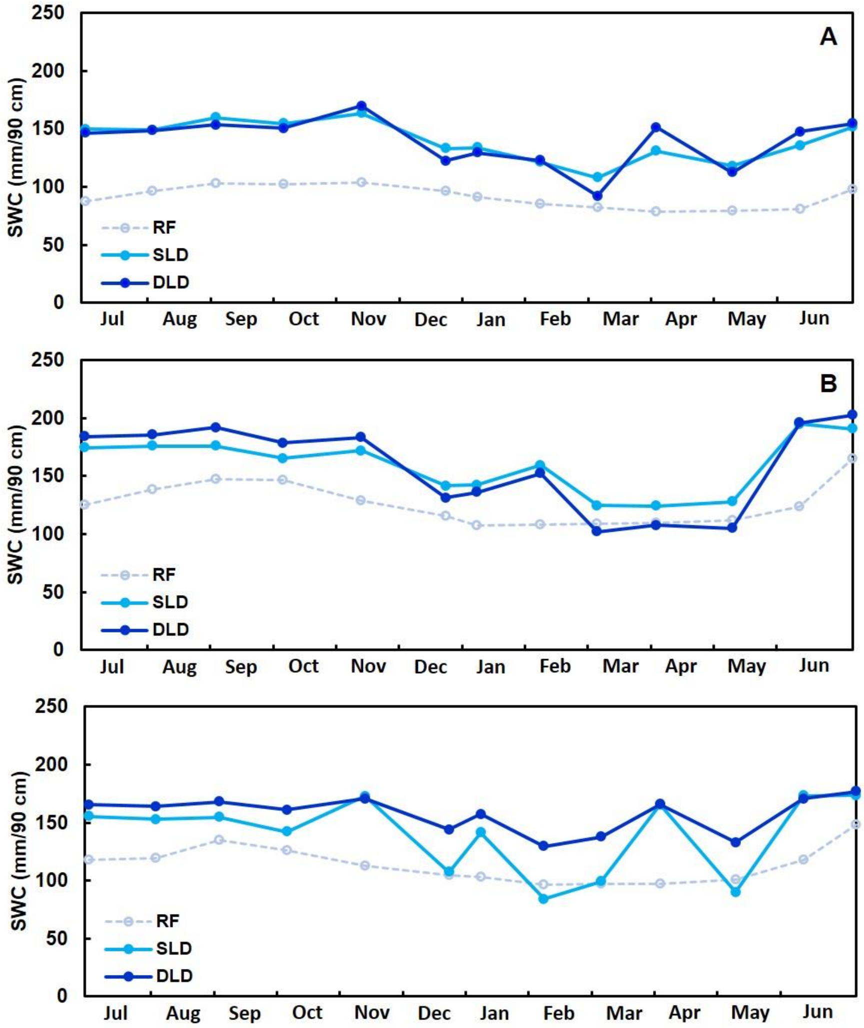

Figure 2.27. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the soil water content (SWC) up to 90 cm soil depth of (A) a shoulder, (B) a backslope and (C) a footslope from July 2017 to June 2018.

2.3.6. Grapevine water status

At the pea size berry stage of the 2017/18 growing season, with the exception of the backslope DLD and footslope SLD plots, the irrigated treatments did not experience any water constraints according to thresholds for water stress levels proposed by Van Leeuwen *et al*. (2009) for Sauvignon blanc and Myburgh *et al*. (2016) for Cabernet Sauvignon grapevines (Fig. 2.28). However, grapevines at the RF plots experienced low water constraints at the shoulder and backslope and moderate constraints at the footslope site during this growth stage (Fig. 2.28). On 18 December 2017 (véraison) all of grapevines at the shoulder site experienced moderate water constraints, with Ψ_S varying between -0.9 MPa and -1.1 MPa (Fig. 2.28A). In contrast, the grapevines of the RF plot at the backslope site was already experiencing severe water constraints (Fig. 2.28B). Prior to harvest, there was little difference between the treatments and all the grapevines experienced severe water constraints, with the exception of the grapevines at the footslope DLD plot (Fig. 2.28C). According to water constraint thresholds, the maximum Ψ_S

measured at the footslope DLD plot fell under Class IV, namely “high water constraints”, which is regarded as ideal to produce quality Cabernet Sauvignon wine on a clay soil (Myburgh *et al*., 2016). The substantially higher Ψ_S measured at the footslope DLD plot during véraison and harvest was most probably a result of the high volumes of irrigation water applied at this plot and subsequent greater SWC (Fig. 2.27C). At the back- and footslope sites, the Ψ_S was consistently higher at the SLD and DLD plots when compared to the RF plots, albeit very slightly. Similarly, Mehmel (2010) reported lower Ψ_S in non-irrigated grapevines when compared to grapevines irrigated with a single and double dripper line in the Swartland and attributed this to greater SWC in irrigated plots. From these results, it is clear that irrigation with treated municipal wastewater was only beneficial in preventing water constraints up until véraison, where after irrigated grapevines experienced similar levels of water stress compared to non-irrigated grapevines. Similar results were reported by Intrigliolo and Castel (2008) for Tempranillo grapevines under rainfed and irrigated conditions during seasons with limited rainfall.

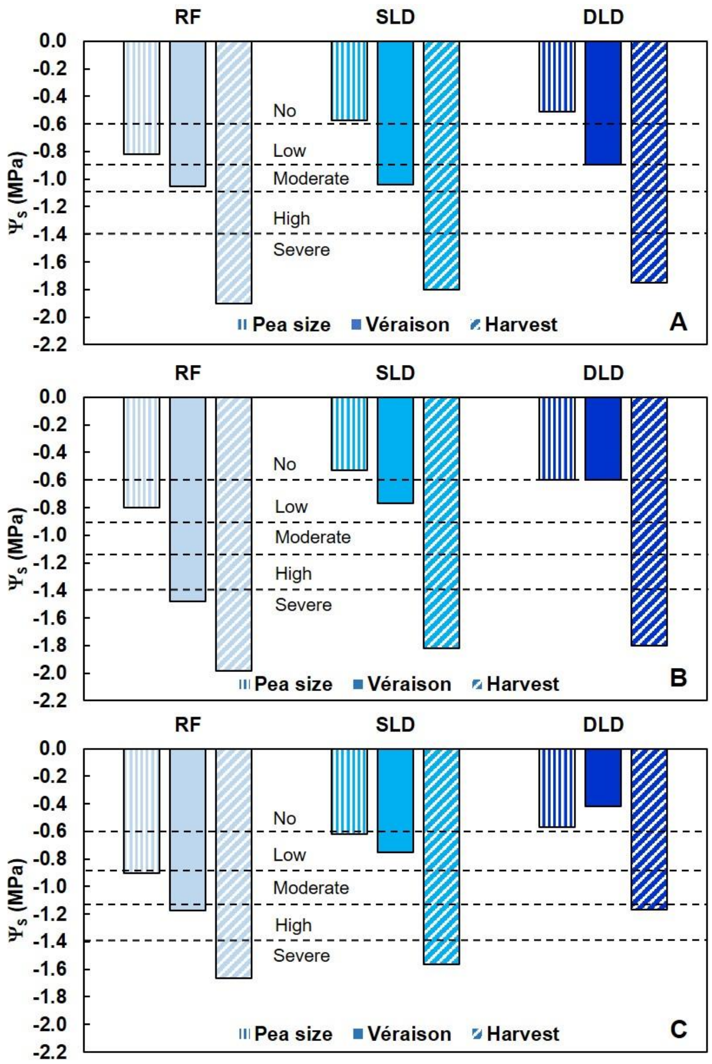

Figure 2.28. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the midday stem water potential (Ψ_S) in (A) Sauvignon blanc on a shoulder and Cabernet Sauvignon on (B) a backslope and (C) a footslope at pea size, véraison and harvest during the 2017/18 season.

2.3.7. Vegetative grapevine measurements

2.3.7.1. Leaf chemical status

Chemical analysis of the leaf blades at véraison of the 2017/18 season revealed that all of the experiment grapevines had levels of N exceeding the recommended norms of 1.6% to 2.7% (Conradie, 1994), except for the grapevines at the back- and footslope RF plots. No substantial differences were observed between treatment plots. However, the leaf N content tended to slightly increase with the amount of irrigation water applied (Table 2.13). The N content of the leaves were not at toxic levels yet, but care should be taken to avoid over-fertilisation that could lead to excessive vegetative growth and reduced fruitfulness (Saayman, 1981 and references therein). The leaf blade P content of all of the grapevines were within the recommended range of 0.14% to 0.55% (Conradie, 1994), except for slightly higher concentrations at the shoulder SLD plot. Irrigation with treated municipal wastewater significantly increased the leaf P concentrations of Sultanina grapevines when compared to grapevines irrigated with fresh water (Paranychianakis *et al*., 2006). The leaf blade K^{+}, Ca^{2+} and Mg^{2+} concentrations of all of the experiment grapevines were within the recommended norms (Conradie, 1994). Furthermore, no trends were observed that could be related to the different irrigation treatments (Table 2.13). In contrast, Neilsen *et al*. (1989) reported increased P, K^{+} and Ca^{2+} and decreased Mg^{2+} levels in the petioles of Riesling grapevines as a result of municipal wastewater irrigation. McCarthy (1981) reported increased leaf petiole Mg^{2+} concentrations in Shiraz grapevines when compared to a fresh water control. Despite the observation of high soil Cl^{-} levels, leaf blade Cl^{-} concentrations at all of the experiment plots were below the recommended threshold value of 0.5% (Beyers, 1962; Christensen, 2005). However, no clear trend relating to the irrigation treatments was observed (Table 2.13). Similarly, no trend occurred with regards to the leaf blade Na^{+} content, either (Table 2.13). In addition, the Na^{+} concentrations were well below the recommended threshold value of 0.25% (Conradie, 1994). This indicated that grapevines did not accumulate excessive amounts of Na^{+} under treated municipal wastewater irrigation. Conversely, Netzer *et al*. (2014) reported significantly greater Na^{+} concentrations in the leaf petioles of table grapes (*Vitis vinifera* L. *cv*. Superior Seedless) under treated municipal wastewater irrigation when compared to grapevines irrigated with fresh water and fertiliser. Furthermore, the petiole Na^{+} increased with an increasing amount of irrigation water applied. With the exception of Fe^{2+} at the backslope RF plot, concentrations of B^{3+}, Cu^{2+} and Fe^{2+} were within the recommended norms for grapevines (Saayman, 1981) and could not be related to the irrigation treatments (Table 2.13). Levels of Mn^{2+} in the Sauvignon blanc leaves of the shoulder site were above the critical value of 300 mg/kg, but still below the toxicity threshold of 1 000 mg/kg (Saayman, 1981). The leaf blade Zn^{2+} concentrations at all of the experiment plots met the recommended norm of 15 mg/kg and was not present at toxic levels (Table 2.13).

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Table 2.13. Nutrient status of Sauvignon blanc leaves on a shoulder, as well as Cabernet Sauvignon leaves on a backslope and footslope, respectively, at véraison during the 2017/18 season.

Landscape position	Treatment	N (%)	P (%)	K^+ (%)	Ca^{2+} (%)	Mg^{2+} (%)	Cl^- (%)	Na^+ (mg/kg)	B^{3+} (mg/kg)	Cu^{2+} (mg/kg)	Fe^{2+} (mg/kg)	Mn^{2+} (mg/kg)	Zn^{2+} (mg/kg)
Shoulder	RF	2.89	0.38	0.98	1.39	0.54	0.14	888	57	8	215	323	47
	SLD	2.94	0.56	0.9	1.66	0.52	0.19	918	59	7	204	350	52
	DLD	2.95	0.7	1.01	1.66	0.55	0.14	966	62	8	262	425	62
Backslope	RF	2.63	0.29	1.02	1.57	0.34	0.08	1089	95	8	506	161	44
	SLD	2.92	0.44	0.93	1.72	0.32	0.03	866	76	8	248	275	49
	DLD	3.12	0.35	0.73	1.94	0.36	0.09	763	73	11	210	232	39
Footslope	RF	2.65	0.29	1.12	1.97	0.33	0.1	703	71	9	199	233	43
	SLD	2.9	0.36	0.81	1.5	0.25	0.07	701	60	8	237	243	46
	DLD	3.01	0.48	1.02	1.8	0.32	0.12	873	67	10	247	243	55

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2.3.7.2. Growth characteristics

2.3.7.2.1. Canopy characteristics

Prior to harvest of the 2017/18 season, the Sauvignon blanc grapevines at the shoulder RF plot showed slight visual water constraints, *i.e.* light green leaves and a less dense canopy compared to the irrigated treatment plots (Fig. 2.29). This was to be expected as the grapevines at this plot experienced severe water constraints at harvest with Ψ_S reaching -1.9 MPa (Fig. 2.28A). In addition, the SWC of the 0-90 cm soil layer of the RF plot was below 90 mm prior to harvest (Fig. 2.27A). The SLD and DLD treatment plots at the shoulder site experienced similar levels of water constraints at harvest, *i.e.* Ψ_S of -1.8 and -1.75, respectively. However, almost no visual symptoms of water stress were observed at these grapevines (Fig. 2.29). Furthermore, the grapevine canopy of the SLD plot appeared to be more dense than that of the DLD plot. The occurrence of actively growing shoots prior to harvest in irrigated Cabernet Sauvignon grapevines near Philadelphia have been reported previously (Mehmel, 2010). This growth is undesirable since active vegetative growth during the ripening period may become a strong sink which can compete with reproductive growth (Smart & Robinson, 1991).

Visual water constraints in the form of yellowing basal leaves in the bunch zone were observed at all the treatment plots on the backslope site prior to harvest of the 2017/18 season (Fig. 2.30). The Ψ_S measured at harvest was similar between the plots and ranged between -1.80 MPa and -1.98 MPa which indicated severe water constraints (Fig. 2.28B). Furthermore, the grapevines at the backslope SLD and DLD plots had denser canopies when compared to the RF plot (Fig. 2.30). This is most likely due to the greater SWC of the irrigated treatments (Fig. 2.27B).

Only the grapevines of the RF plot showed visual signs of water constraints at the footslope site (Fig. 2.31). This could be explained by both low SWC (Fig. 2.27C) and low Ψ_S (Fig. 2.28C) measured at this plot during the harvest period. The canopy in the bunch zone of the DLD plot was visibly more dense than the SLD plot (Fig. 2.31). This is likely a result of considerably higher Ψ_S (Fig. 2.28) and SWC (Fig. 2.27C) at harvest. Excessive shade in the bunch zone of Cabernet Sauvignon grapevines in Stellenbosch resulted in reduced berry mass, bunch mass, yield and skin colour and increased the K^+ concentration, pH and TTA of the grape must (Archer & Strauss, 1989). Densely shaded canopies may also increase the chances of developing Botrytis bunch rot and induce unwanted herbaceous characters in wine (Smart *et al*., 1990 and references therein). Since Cabernet Sauvignon is considered to be a vigorous, low yielding cultivar (Goussard, 2008), it is particularly sensitive to over-irrigation (Bruwer, 2010).

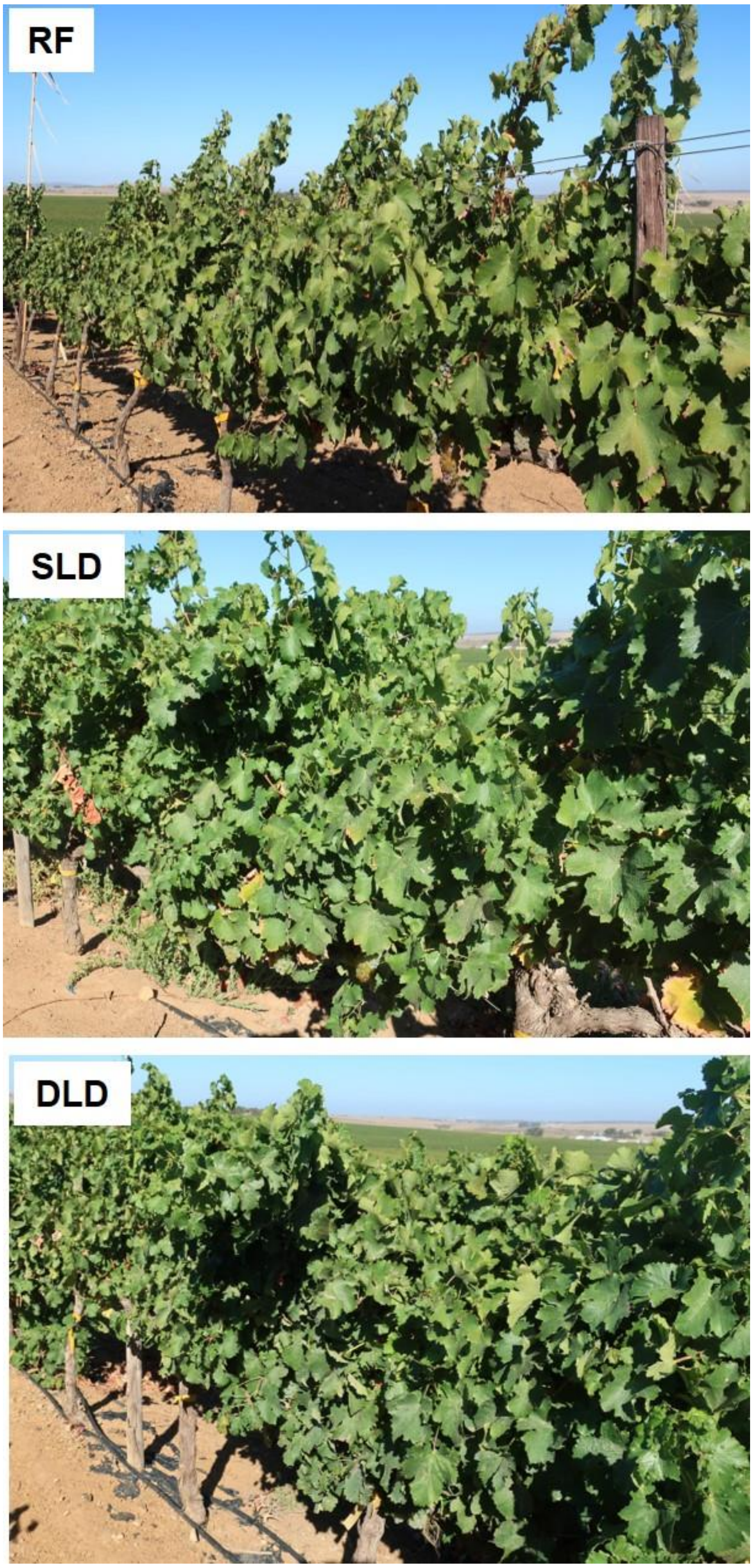

Figure 2.29. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the visual appearance of Sauvignon blanc on a shoulder prior to harvest of the 2017/18 season.

Figure 2.30. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the visual appearance of Cabernet Sauvignon on a backslope prior to harvest of the 2017/18 season.

Figure 2.31. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the visual appearance of Cabernet Sauvignon on a footslope prior to harvest of the 2017/18 season.

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After 11 years of irrigation with treated municipal wastewater, the mean trunk circumference of the irrigated Sauvignon blanc grapevines at the shoulder site tended to be slightly higher than the RF plot (Table 2.14), indicating that the RF grapevines assimilated less carbon over the course of the study period. Long-term exposure to severe water constraints leads to a progressive decline in stomatal conductance and a decrease in carbon assimilation (Escalona *et al*., 1999). The greater trunk circumference measured at the footslope site could be a result of the slightly concave form of the landscape which facilitated the accumulation of soil water at this site and subsequently lead to greater carbon assimilation. Irrigation with treated municipal wastewater did not affect grapevine cordon length (Table 2.14). The number of spurs per grapevine also remained largely unaffected after 11 years of wastewater irrigation (Table 2.14).

Table 2.14. Mean trunk circumference, cordon length and number of spurs of Sauvignon blanc grapevines on a shoulder and Cabernet Sauvignon grapevines on a backslope and footslope, respectively, under rainfed conditions (RF) and irrigated with treated municipal wastewater *via* single (SLD) or double line drip (DLD) prior to budbreak of the 2017/18 season.

Cultivar	Landscape position	RF	SLD	DLD
		Trunk circumference (cm)		
Sauvignon blanc	Shoulder	14.57	16.08	16.57
Cabernet Sauvignon	Backslope	16.29	18.06	17.34
Cabernet Sauvignon	Footslope	18.71	19.63	21.95
		Cordon length (m)		
Sauvignon blanc	Shoulder	1.25	1.46	1.33
Cabernet Sauvignon	Backslope	1.38	1.26	1.26
Cabernet Sauvignon	Footslope	1.27	1.36	1.34
		Number of spurs per grapevine		
Sauvignon blanc	Shoulder	10	11	10
Cabernet Sauvignon	Backslope	13	15	19
Cabernet Sauvignon	Footslope	16	16	17

With the exception of the grapevines at the backslope site, the length of the primary shoots tended to increase with an increase in amount of irrigation water applied (Table 2.15). Similar results were reported for Cabernet Sauvignon grapevines in the Swartland region (Mehmel, 2010). Shoots shorter than 30 cm produced berries that were low in sugar and phenol concentrations and were poorly coloured, whereas 1.2 m shoots were considered optimal for producing high quality Cabernet Sauvignon grapes (Mehmel, 2010 and references therein). Therefore, the SLD irrigated grapevines exhibited optimal shoot growth since the length of primary shoots varied between 1.11 m and 1.31 m (Table 2.15). In contrast, the primary shoots of the shoulder and footslope DLD treatment plots were 1.5 m and longer, indicating excessive vegetative growth. The DLD plot at the footslope site also had substantially more and longer secondary shoots (Table 2.15). When

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compared to rainfed and severely stressed grapevines, drip irrigated Cabernet Sauvignon grapevines had increased vigour and active shoot growth during the ripening period which induced competition for photosynthetic assimilates and reduced berry sugar content (Tandonnet *et al*., 1999). The elongation of primary shoots was associated with an increase in the length of internodes. In addition, the primary shoot diameter of the Cabernet Sauvignon grapevines increased with an increase in irrigation (Table 2.15). Although this response was not observed for the Sauvignon blanc grapevines at the shoulder site, the number of primary leaves per shoot increased with the amount of irrigation water applied. This was also observed at the Cabernet Sauvignon grapevines of the footslope site. However, the number of secondary leaves per primary shoot did not follow a clear trend at any of the experiment sites. Similarly, the total number of leaves per primary shoot remained largely unaffected by irrigation water application at the shoulder and backslope sites, whereas the footslope site exhibited a slight increase in the amount of leaves as the amount of irrigation water increased (Table 2.15).

Table 2.15. Mean vegetative growth components of Sauvignon blanc grapevines on a shoulder and Cabernet Sauvignon grapevines on a backslope and footslope, respectively, under rainfed conditions (RF) and irrigated with treated municipal wastewater *via* single (SLD) or double line drip (DLD) during the ripening period of the 2017/18 season.

Cultivar	Landscape position	RF	SLD	DLD
		Primary shoot length (m)		
Sauvignon blanc	Shoulder	0.79	1.31	1.52
Cabernet Sauvignon	Backslope	0.52	1.11	1.07
Cabernet Sauvignon	Footslope	0.76	1.27	1.67
		Primary shoot internode length (mm)		
Sauvignon blanc	Shoulder	8.06	8.49	8.56
Cabernet Sauvignon	Backslope	8.81	10.77	12.75
Cabernet Sauvignon	Footslope	9.47	9.90	11.25
		Primary shoot diameter (mm)		
Sauvignon blanc	Shoulder	6.85	6.55	6.76
Cabernet Sauvignon	Backslope	5.99	8.18	8.59
Cabernet Sauvignon	Footslope	5.26	5.95	6.86
		Number of primary leaves per shoot		
Sauvignon blanc	Shoulder	17	25	34
Cabernet Sauvignon	Backslope	9	13	10
Cabernet Sauvignon	Footslope	13	18	28
		Secondary shoot length (m)		
Sauvignon blanc	Shoulder	0.12	0.11	0.08
Cabernet Sauvignon	Backslope	0.08	0.10	0.10
Cabernet Sauvignon	Footslope	0.06	0.17	0.16
		Number of secondary shoots		
Sauvignon blanc	Shoulder	9	19	18
Cabernet Sauvignon	Backslope	2	5	8
Cabernet Sauvignon	Footslope	5	2	13
		Number of secondary leaves per shoot		
Sauvignon blanc	Shoulder	58	96	82
Cabernet Sauvignon	Backslope	11	21	32
Cabernet Sauvignon	Footslope	21	16	79
		Total number of leaves per shoot		
Sauvignon blanc	Shoulder	50	51	56
Cabernet Sauvignon	Backslope	31	34	30
Cabernet Sauvignon	Footslope	32	52	59

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The leaf area per grapevine of the Cabernet Sauvignon grapevines increased with the amount of irrigation water applied (Table 2.16). Similarly, the leaf area of the Sauvignon blanc grapevines at the shoulder site was greater for the irrigated treatments when compared to the RF plot, but little difference could be seen between the SLD and DLD plots (Table 2.16). In fact, the SLD plot had a slightly higher leaf area index (LAI) compared to the DLD plot (Fig. 2.32). The grapevine LAI at the backslope site increased with the amount of irrigation water applied. The grapevines of the footslope DLD plot had excessively high leaf area per grapevine which was also reflected by the LAI (Fig. 2.32). This could result in reduced bud fertility and fruit of poorer quality (Smart *et al*., 1990).

Table 2.16. Mean leaf area of Sauvignon blanc grapevines on a shoulder and Cabernet Sauvignon grapevines on a backslope and footslope, respectively, under rainfed conditions (RF) and irrigated with treated municipal wastewater *via* single (SLD) or double line drip (DLD) during the ripening period of the 2017/18 season.

Cultivar	**Landscape position**	**RF**	**SLD**	**DLD**
		Primary leaf area per shoot (m^2)		
Sauvignon blanc	Shoulder	0.15	0.21	0.32
Cabernet Sauvignon	Backslope	0.06	0.14	0.11
Cabernet Sauvignon	Footslope	0.10	0.18	0.25
		Secondary leaf area per shoot (m^2)		
Sauvignon blanc	Shoulder	0.23	0.43	0.29
Cabernet Sauvignon	Backslope	0.03	0.09	0.12
Cabernet Sauvignon	Footslope	0.07	0.07	0.45
		Total leaf area per shoot (m^2)		
Sauvignon blanc	Shoulder	0.38	0.64	0.61
Cabernet Sauvignon	Backslope	0.09	0.22	0.23
Cabernet Sauvignon	Footslope	0.17	0.25	0.70
		Total leaf area per grapevine (m^2)		
Sauvignon blanc	Shoulder	7.73	13.69	12.45
Cabernet Sauvignon	Backslope	2.32	6.76	8.54
Cabernet Sauvignon	Footslope	5.47	8.18	24.30

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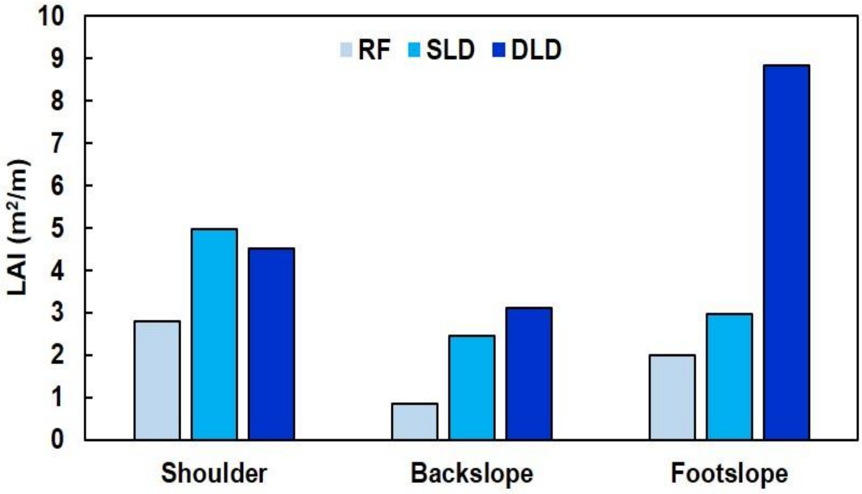

Figure 2.32. Leaf area index (LAI) of Sauvignon blanc grapevines on a shoulder and Cabernet Sauvignon grapevines on a backslope and footslope, respectively, under rainfed conditions (RF) and irrigated with treated municipal wastewater *via* single (SLD) and double line drip (DLD) during the ripening period of the 2017/18 season.

2.3.7.2.2. Cane mass

Irrigation using treated municipal wastewater increased the cane mass of grapevines compared to the RF control (Fig. 2.33). These results were expected since the irrigated plots had higher SWC for most of the season (Fig. 2.27) as well as higher Ψ_S (Fig. 2.28). Similar results were reported for irrigated and non-irrigated Cabernet Sauvignon grapevines in the Swartland region (Mehmel, 2010). In the Coastal region, Conradie *et al*. (2002) reported significantly higher cane mass for Sauvignon blanc grapevines in a soil with higher SWC when compared to grapevines in a drier soil in the same vineyard. According to Williams (2000), reduced shoot growth is one of the first visible symptoms of grapevine water constraints. In this regard, the availability of treated municipal wastewater as an irrigation water source had a positive impact on grapevine vegetative growth in a region where grapevines are traditionally grown under dryland conditions due to a lack of natural fresh water resources. With the exception of the footslope RF plot, the cane mass measured during the 2017/18 season was greater at all of the experiment plots when compared to the mean cane mass of the previous four seasons (Fig. 2.33). This is likely a result of greater rainfall experienced during the 2017/18 season as well as larger volumes of irrigation water applied at the SLD and DLD plots compared to the 2013/14, 2014/15 and 2015/16 seasons (Appendix 1). The foregoing suggests that irrigation with treated municipal wastewater did not pose a salinity hazard to grapevine vegetative growth.

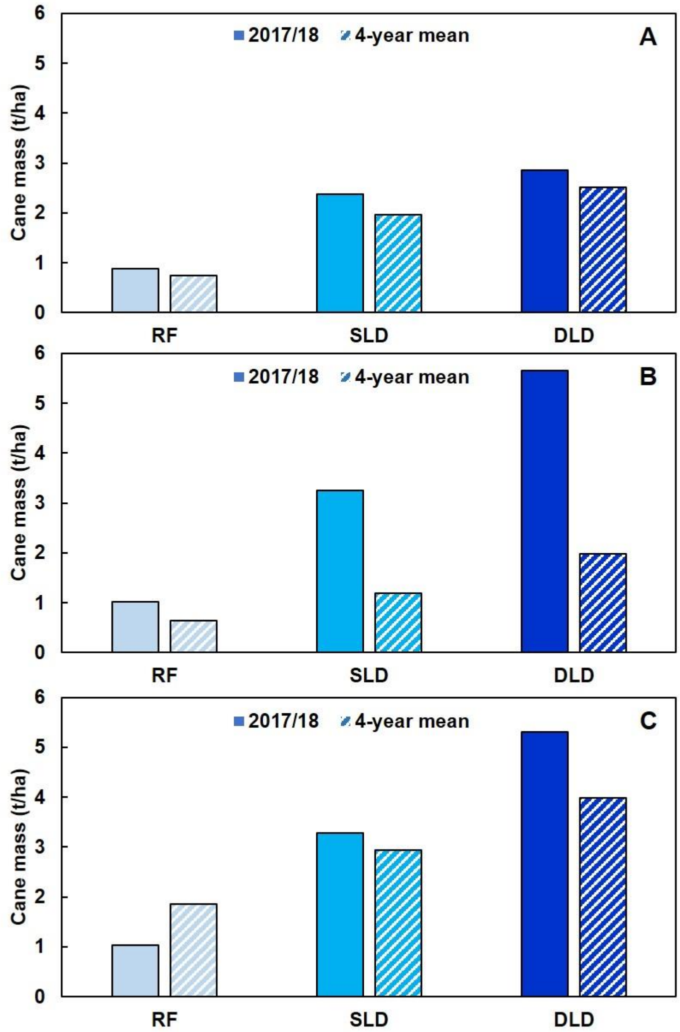

Figure 2.33. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the cane mass in (A) Sauvignon blanc on a shoulder and Cabernet Sauvignon on (B) a backslope and (C) a footslope during the 2017/18 season compared to a 4-year mean (ARC, unpublished data).

2.3.8. Yield and its components

2.3.8.1. Bunch and berry mass at harvest

Irrigation with treated municipal wastewater substantially increased the bunch mass at all three of the experiment sites during the 2017/18 season (Table 2.17). At the shoulder and backslope sites the increased bunch mass was associated with larger berries for the irrigated treatments, whereas the SLD and DLD plots of the footslope site had larger berries as well as more bunches per grapevine compared to the RF plot (Table 2.17). Although the irrigated treatments substantially increased the bunch mass at the shoulder and backslope sites when compared to the RF plots (Fig. 2.34A), the additional irrigation water applied *via* the DLD did not result in higher bunch mass compared to the SLD plots (Table 2.17). In contrast, the bunch mass of the Cabernet Sauvignon grapes at the footslope increased with the amount of irrigation water applied (Fig. 2.34B & Table 2.17). This could be explained by the amount of irrigation water applied at this site throughout the season (Table 2.2) and the subsequent lower water constraints experienced at the DLD plot (Fig.

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2.28). Mirás-Avalos and Intrigliolo (2017) reported that the berry mass of Sauvignon blanc grapes were severely reduced when Ψ_S became more negative. Berries are most sensitive to water deficits during the beginning stages of berry development and may reduce berry size irreversibly (Williams, 2000). Water constraints prior to véraison can result in smaller berries (Van Leeuwen *et al*., 2004). High water constraints that were associated with low soil matric potential during the period from flowering to harvest reduced the berry size of Cabernet Sauvignon grapevines in the Swartland region (Mehmel, 2010). Similarly, water deficits experienced by Shiraz grapevines between flowering and véraison irreversibly reduced berry size (Ojeda *et al*., 2001). Furthermore, water constraints during the period from flowering to berry set may reduce the number of berries set (Hardie & Considine, 1976).

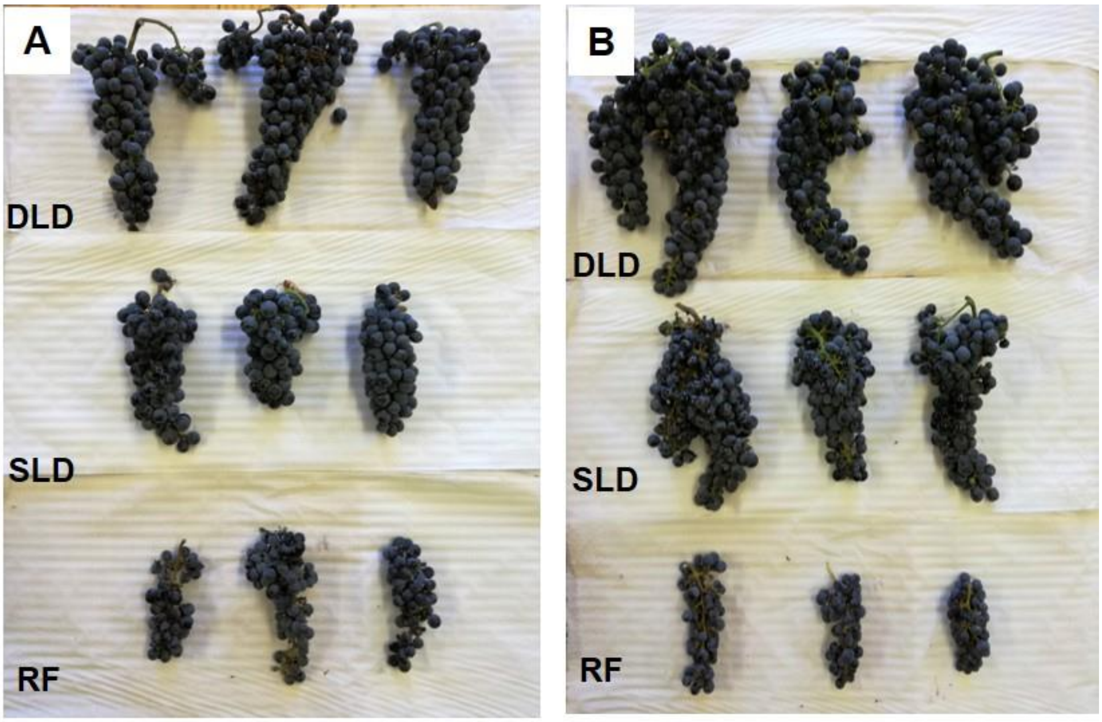

Figure 2.34. Variation in bunch mass of Cabernet Sauvignon grapevines on (A) a backslope and (B) a footslope, under rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) during the 2017/18 season.

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Table 2.17. Yield components of Sauvignon blanc grapevines on a shoulder and Cabernet Sauvignon grapevines on a backslope and footslope, respectively, under rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) during the 2017/18 season.

Cultivar	Landscape position	RF	SLD	DLD
			Berry mass (g)	
Sauvignon blanc	Shoulder	1.34	1.85	1.88
Cabernet Sauvignon	Backslope	0.46	1.11	0.94
Cabernet Sauvignon	Footslope	0.55	0.94	1.19
			Bunch mass (g)	
Sauvignon blanc	Shoulder	91	153	150
Cabernet Sauvignon	Backslope	29	94	134
Cabernet Sauvignon	Footslope	23	95	82
			Number of bunches per grapevine	
Sauvignon blanc	Shoulder	24	27	28
Cabernet Sauvignon	Backslope	43	37	46
Cabernet Sauvignon	Footslope	24	32	36

2.3.8.2. Yield

Irrigation with treated municipal wastewater increased grapevine yield at all of the experiment plots compared to the RF control (Fig. 2.35). During the 2017/18 season, yield followed similar trends as was seen for bunch mass. Therefore, the increased yield can be attributed to bigger bunches in the irrigated treatments. Similar to bunch mass, the yield between the SLD and DLD treatment plots of the shoulder and backslope sites did not differ substantially, whereas the yield at the footslope site increased with increasing amount of irrigation water applied (Fig. 2.35). Similar results were reported for Cabernet Sauvignon grapevines in the Swartland region (Mehmel, 2010). Shiraz grapevines irrigated with 135 ℓ of municipal wastewater per week had more and heavier bunches which resulted in greater yields compared to grapevines irrigated with either 45 ℓ/week municipal wastewater or 135 ℓ/week fresh water (McCarthy, 1981). Similar to what was found for cane mass, results confirmed that irrigation with treated municipal wastewater did not pose a salinity hazard to yield. Low rainfall during the beginning stages of berry development (Fig. 2.6) might help to explain the lower yields measured at the RF treatments compared to the mean yield of the previous four seasons (Fig. 2.35). Whereas, the higher yield measured at the footslope DLD plot is probably a result of high irrigation volumes applied at this plot (Table 2.2).

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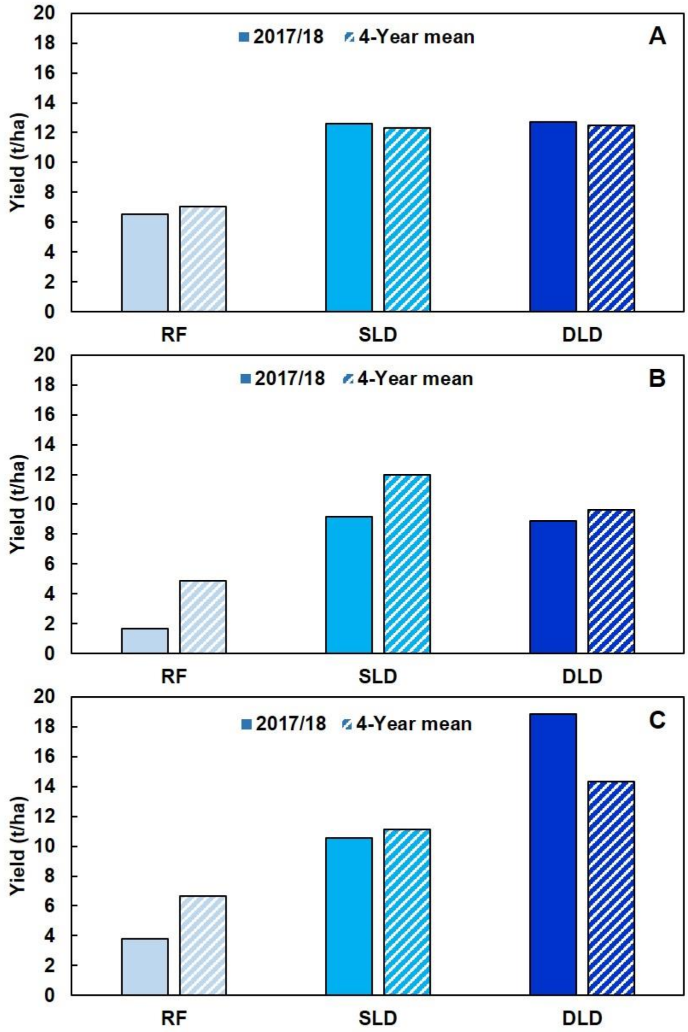

Figure 2.35. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater *via* single (SLD) and double line drip (DLD) on the yield in (A) Sauvignon blanc on a shoulder, as well as Cabernet Sauvignon on (B) a backslope and (C) a footslope during the 2017/18 season compared to a four year mean (ARC, unpublished data).

2.3.9. Grape juice characteristics

2.3.9.1. Total soluble solids

The juice TSS of the Sauvignon blanc grapes at the shoulder site was not affected by the different irrigation treatments, whereas the RF treatment plots of the Cabernet Sauvignon grapevines at the backslope and footslope sites had slightly higher TSS compared to the irrigated treatments (Fig. 2.36). This is likely a result of actively growing shoots and excessive vegetative growth at the SLD and DLD treatment plots during the ripening period which was a stronger sink for photosynthates compared to the ripening grapes (Mehmel, 2010). The higher yields of the irrigated treatments may also have obstucted sugar accumulation due to sink competition (Kliewer & Dokoozlian, 2005). Since all the Cabernet Sauvignon grapes were harvested on the same day (due to logistical reasons), the RF plots probably accumulated more sugars over the ripening period. It has been reported previously that grapevine water constraints enhance berry sugar content in low yielding grapevines, whereas it reduces berry sugar content of high yielding grapevines (Van Leeuwen *et al*., 2009 and references therein). The 2017/18 TSS followed similar trends to the mean of the previous four seasons (Fig. 2.36).

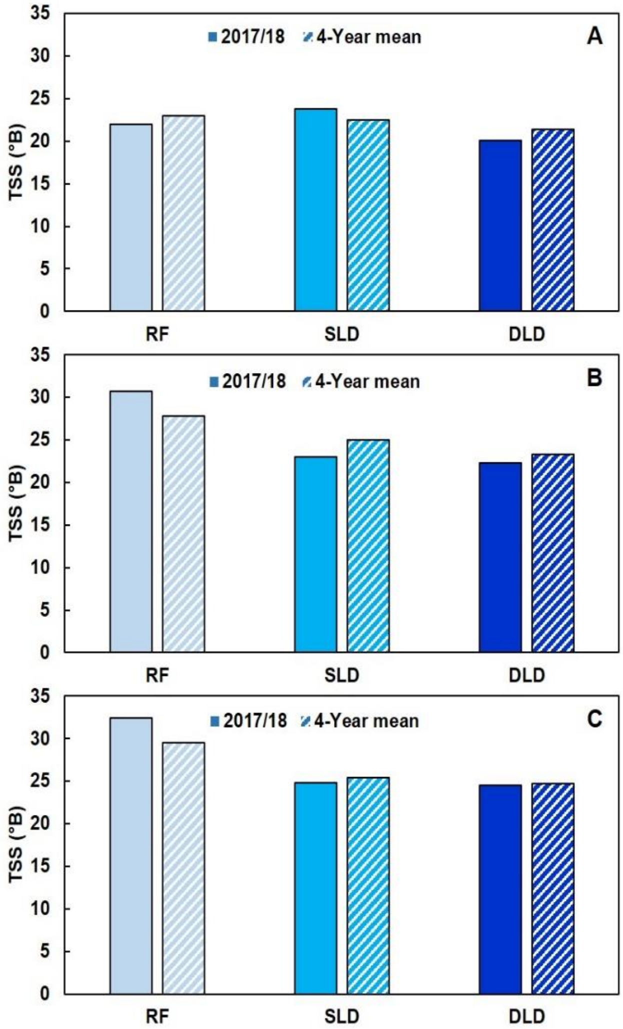

Figure. 2.36. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater via single (SLD) and double line drip (DLD) on the grape juice total soluble solids (TSS) in (A) Sauvignon blanc on a shoulder, as well as Cabernet Sauvignon on (B) a backslope and (C) a footslope during the 2017/18 season compared to a four year mean (ARC, unpublished data).

2.3.9.2. Total titratable acidity

The TTA of the Cabernet Sauvignon grapes at the back- and footslope sites increased with the amount of irrigation water applied (Fig. 2.37). The lower TTA measured at the RF treatment plots may be a result of increased sunlight penetration in the less dense bunch zone which lead to more berry exposure and reduced TTA (Iland, 1989b; Conradie *et al*., 2002). The increase in TTA with increased water application can also be related to less water constraints experienced by the grapevines at the SLD and DLD plots during the ripening period (Fig. 2.28). Mehmel (2010) reported reduced TTA in non-irrigated Cabernet Sauvignon grapevines compared to grapevines irrigated *via* a single or double dripper line. The reduction in TTA was attributed to water constraints experienced during the ripening period, as well as the warm climate of the Swartland region. In contrast, the TTA of the Sauvignon blanc grapes at the shoulder site did not differ between the RF and SLD plots, however TTA increased in the DLD plot (Fig. 2.37). These results were to be expected as the RF and SLD plots at the shoulder site experienced similar water constraints during the ripening period (Fig. 2.28). Similar results were reported by Conradie *et al*. (2002) for Sauvignon blanc grapes in the Coastal region. They reported lower acidity and higher pH for grapevines planted in a Glenrosa soil as a result of water stress. The TTA measured during the 2017/18 season followed similar trends to the mean TTA of the previous four seasons (Fig. 2.37).

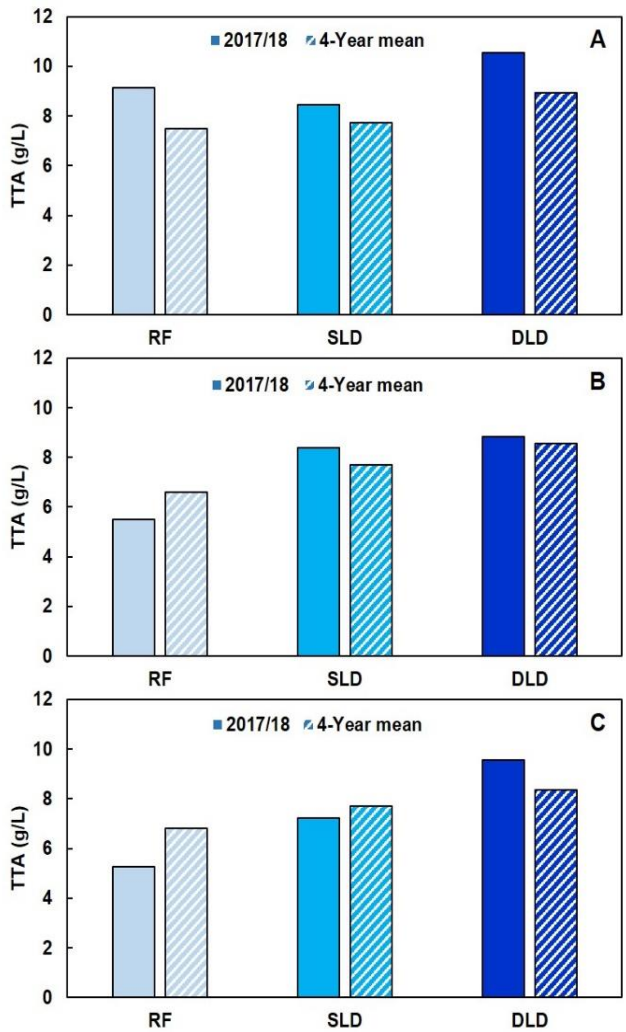

Figure 2.37. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater via single (SLD) and double line drip (DLD) on total titratable acidity (TTA) in (A) Sauvignon blanc on a shoulder, as well as Cabernet Sauvignon on (B) a backslope and (C) a footslope during the 2017/18 season compared to a four year mean (ARC, unpublished data).

2.3.9.3. pH

The different irrigation treatments did not have a significant effect on the juice pH at harvest of either the Sauvignon blanc or Cabernet Sauvignon grapes during the 2017/18 season (Fig. 2.38). Similar results were seen for the mean juice pH of the previous four seasons. With the exception of the RF plots of the back- and footslope sites, the juice pH was within the range of 3.0 g/L to 3.8 g/L recommended by Kodur (2011). High juice pH (*e.g.* > 3.8 g/L) is often associated with high concentrations of K^+ in the juice and may result in wines with poor colour stability and poor taste (Somers, 1975). Previous studies have linked increased berry K^+ concentrations to increased K^+ supply to grapevines (Morris *et al*., 1983; Ruhl, 1989). McCarthy and Downton (1981) reported increased pH in the wines made from Shiraz grapes that received 135 ℓ of municipal wastewater per week compared to grapevines irrigated with the same amount of fresh water. The increased pH was attributed to greater K^+ concentrations and resulted in wines with poor colour, less anthocyanins and a greater "chemical age". Despite the high amounts of K^+ applied *via* treated municipal wastewater irrigation during this study, no detrimental effects with regards to juice quality was observed. In fact, irrigation tended to improve the quality of the must compared to the RF control treatments.

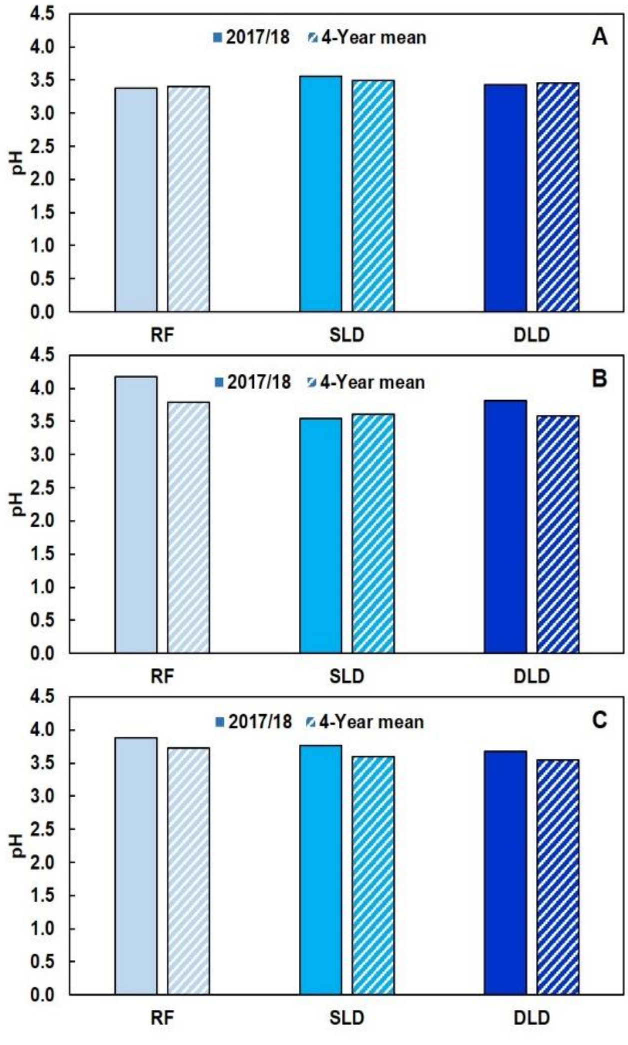

Figure 2.38. Effect of rainfed conditions (RF) and irrigation with treated municipal wastewater via single (SLD) and double line drip (DLD) on the grape juice pH in (A) Sauvignon blanc on a shoulder, as well as Cabernet Sauvignon on (B) a backslope and (C) a footslope during the 2017/18 season compared to a four year mean (ARC, unpublished data).

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2.4. CONCLUSIONS

The quality of the treated municipal wastewater used for irrigation in the present study was acceptable, and met the minimum criteria stipulated by the General Authorisations to irrigate up to 500 m^3 per day. The pH of the topsoil increased by *ca*. 1.3 units over the course of the study period but remained within the recommended range for growing grapevines. The EC_e of the topsoil increased where irrigated with treated municipal wastewater. The accumulation of salts at the soil surface was most likely due to evapotranspiration during the warm summer months. Subsequently, the K_{ns} of the topsoil decreased with an increase in EC_e, therefore problems with infiltration may occur during heavy rainfall events.

The low N content of the wastewater was not sufficient to supply the annual N requirement of grapevines, *i.e.* not even in the case of the DLD. Furthermore, the level of N did not pose any risk for pollution of natural water sources. In contrast, the high P content of the irrigation water could lead to the formation of algal blooms in water storage facilities and bio-fouling of irrigation equipment. However, where double the normal amount of irrigation water was applied, treated municipal wastewater supplied adequate amounts of P to supply annual grapevine requirements. Although the P content of the irrigated soils increased over the course of the study period in relation to the baseline values, the greatest accumulation was observed at RF treatments. This trend was probably due to very limited uptake by the low-vigour, low-yielding grapevines of the amounts of P applied by the grower.

Amounts of K^+ applied *via* wastewater irrigation was in excess of grapevine requirements and accumulated in the topsoil of all the irrigation treatments but did not result in excessive uptake by plants. Similarly, the high Na^+ content of the irrigation water did not cause an accumulation of Na^+ in the leaf blades and did not negatively affect vegetative growth or yield. However, the soil ESP´ of the backslope and footslope sites increased as a result of treated municipal wastewater irrigation. The increase was more prominent in the subsoil layers, possibly due to the seasonal leaching of salts by rainfall. The results indicate that the application of more water at the DLD treatment plots might also have contributed to more Na^+ being leached from the profile compared to the SLD plots.

Due to the chlorination disinfection process performed by the WWTW, high Cl^- concentrations in the wastewater substantially increased the soil Cl^- contents of the irrigated treatments. However, similar to K^+ and Na^+, it did not result in excessive uptake by grapevines. This suggests that grapevines possess mechanisms to regulate the uptake of ions from the soil solution. Despite the high amounts of salts applied *via* treated municipal wastewater irrigation, no salinity hazards with regards to vegetative growth and yield was observed for the irrigated treatments.

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2.5. RECOMMENDATIONS

The availability of irrigation water (albeit of relatively low quality) in regions where grapevines are usually grown under dryland conditions can increase the productivity of grapevines whilst maintaining good fruit quality. The present study indicated that grapevines can be irrigated successfully using treated municipal wastewater. However, proper management is required to limit possible negative effects on grapevines and the environment. Regular analyses of irrigation water, soils and grapevine leaves are recommended to ensure that chemical parameters conform to recommended thresholds and norms. In doing so, irreversible damage to irrigation equipment, soils and grapevines can be avoided. The results from this study indicated that irrigation using treated municipal wastewater was able to supply grapevines with nutrients in a plant-available form. However, some nutrients were not supplied in sufficient amounts, whereas others were supplied in excess. It is therefore recommended to use an integrated fertiliser program by adjusting fertiliser applications according to the amount of nutrients present in the wastewater. Furthermore, soil and plant water status should be monitored regularly to avoid over-irrigation, and to minimise the application of salts *via* the irrigation water.

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CHAPTER 3: SHORT-TERM EFFECTS OF IRRIGATION USING IN-FIELD FRACTIONALLY APPLIED WINERY WASTEWATER WITH RAW WATER ON SELECTED SOILS AND GRAPEVINES UNDER DIFFERENT CLIMATES

3.1. INTRODUCTION

The Western Cape of South Africa has been experiencing severe water shortages in recent years and at the beginning of 2018 was faced with its worst drought in 114 years. On the 1st of February 2018, the municipality of the City of Cape Town issued level 6B water restrictions which limited the domestic use of potable water to 50 ℓ per person per day. In addition, farmers and growers in the province were also subjected to severe water restrictions which had serious impacts on the agricultural sector. Depending on the region, growers were restricted to anything between 50% and 13% of their usual irrigation water quota (World Wildlife Foundation, 2018). Subsequently, growers were faced with potential yield and employment losses. In the case of grapevines, drought situations experienced during a particular season may also impact the grapevine growth and yield of future seasons. As a result, wine grape growers have been put under immense pressure to find innovative and alternative ways to irrigate vineyards. Some of the proposed practices include: (i) improving irrigation water use efficiency; (ii) improving irrigation techniques and scheduling; (iii) prioritising economically promising vineyards and (iv) removing less profitable ones. Another possible solution is the use of alternative sources of irrigation water, such as different types of wastewater. Since vineyards are often located close to wineries, and wineries produce large volumes of wastewater, the reuse of winery wastewater for vineyard irrigation seems to be a possible solution.

Winery wastewater is primarily comprised of wine, grape juice, suspended solids (during the harvest period) and cleaning agents (Mosse *et al*., 2011). It is characterised by high levels of organic matter (OM) and consequently has a high chemical oxygen demand (COD). This measure is used to quantify the total organic content in the water in terms of the amount of oxygen needed to facilitate its oxidative breakdown (Schoeman, 2012). In addition, winery wastewater often contains appreciable amounts of inorganic salts. Sodium (Na^+) and potassium (K^+) are likely the most notable since they are present in winery wastewaters at very high concentrations. These salts originate from Na- and K-based cleaning agents, grape lees and waste liquid from grape fermentation processes (Laurenson *et al*., 2012). Winery wastewater may also contain varying amounts of nitrogen (N), phosphorous (P), calcium (Ca^{2+}) and magnesium (Mg^{2+}) (Mosse *et al*., 2011; Laurenson *et al*., 2012).

Irrigation of vineyards using winery wastewater has been implemented and practiced for a number of years in California (Ryder, 1995) and Australia (Kumar *et al*., 2009). Besides being a valuable source of irrigation water, winery wastewater can supply grapevines with essential plant nutrients especially P, K^+ and Ca^{2+}. The reuse of winery wastewater for vineyard irrigation would therefore

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not only allow nutrient recycling but could lead to savings in fertiliser costs. Another advantage of irrigating with winery wastewater may include potential positive impacts on soil structural stability due to the OM present in the wastewater. The high energy inputs required for winery wastewater treatment may also be reduced if wastewater can be used for irrigation. Furthermore, if practised responsibly, reusing winery wastewater for crop irrigation may be a sustainable, cost-effective form of waste disposable (Laurenson *et al*., 2012).

However, there are potential environmental risks associated with land application of winery wastewater. Over-irrigation with winery wastewater can lead to the development of waterlogged, anaerobic soils (Mulidzi, 2011; Howell, 2016 and references therein). Subsequent seepage to underground water resources may cause serious environmental pollution. In addition, an accumulation of inorganic salts in the soil may lead to salinity-related problems, such as osmotic stress, nutrient deficiencies and reduced yields in grapevines. The accumulation of monovalent ions (particularly Na^+ & K^+) may have further deleterious effects on soil structural stability which can result in reduced water infiltration and soil hydraulic conductivity (K). The application of excessive amounts of K^+ may also have detrimental impacts on grape juice and wine quality due to its effect on berry pH regulation. Furthermore, the potential occurrence of unpleasant odours may limit the use of winery wastewater for irrigation to vineyards far away from wineries and urban areas. Irrigating grapevines with winery wastewater therefore require diligent monitoring and management.

Historically, wineries in South Africa disposed of their wastewater by irrigation onto small, permanent-pasture grazing paddocks (Howell & Myburgh, 2018). This practice was considered particularly suitable since land application allowed soil microbial populations to break down biodegradable organic pollutants and the resulting grazing was useful. However, the recent drought in the Western Cape has presented the opportunity for growers to reuse their winery wastewater for irrigating vineyards. The Department of Water Affairs and Forestry (DWAF) expressed their support for the beneficial irrigation of crops using treated winery wastewater (Van Schoor, 2005).

South African legislation requires growers wanting to irrigate with winery wastewater to register the intended use of the wastewater with the Department of Water Affairs (DWA) (Van Schoor, 2005). In 2013, the DWA released General Authorisations which allowed the beneficial use of treated winery wastewater for crop irrigation, provided that the wastewater complies with specific quality standards (DWA, 2013). However, the quality standards, particularly in terms of COD, is often difficult to adhere to without extensive wastewater treatment. Furthermore, under current legislation the dilution of winery wastewater with raw (fresh) water is prohibited (DWA, 2013). In this regard, a previous research project (Project K5/1881) investigated the impact of irrigation using winery wastewater diluted to predetermined levels of COD on the soil, grapevines and wine quality under a specific set of environmental conditions. The project was funded and initiated by the Water Research Commission (WRC) and co-funded by Winetech, THRIP and the Agricultural Research Council (ARC). Experience from this study indicated that it would be impractical to dilute/augment winery wastewater to predetermined levels prior to each irrigation, especially at the commercial level (Myburgh & Howell, 2014b). A possible solution might be the in-field fractional application (augmentation) of winery wastewater with raw water. According to this approach, a certain percentage of the irrigation requirement would be applied as undiluted winery wastewater. Thereafter raw water would be applied for the remainder of the irrigation requirement. This approach could prevent possible contamination of grapes and grapevines since wastewater spray drift will be washed off during raw water irrigation. Another advantage is that the irrigation system will be flushed with a substantial volume of raw water which may reduce the risk of sediment deposits and emitter clogging. Furthermore, in a parallel study it was found that soil type and winter rainfall had a pronounced effect on the accumulation of salts where winery wastewater is used for irrigation (Mulidzi, 2016). It is therefore necessary to investigate the use of winery wastewater for vineyard irrigation in different environments to determine under which conditions the practice would be sustainable, beneficial and could contribute to water conservation and use.

This study formed part of a newly initiated project (K5/2561//4) funded by the WRC, Winetech and ARC. The project aims to investigate the sustainability of using in-field fractionally applied winery wastewater with raw water for irrigating grapevines under different climates. Since climatic conditions and soil types vary considerably in the Western Cape, it was possible to investigate the effects of winery wastewater irrigation on different soil types within the same climatic region.

3.2. MATERIALS AND METHODS

3.2.1. Site selection and vineyard characteristics

The study was carried out in three different wine production regions within the Western Cape of South Africa. The respective regions were the Coastal, Breede River and Lower Olifants River regions. The specific locations were selected due to their vast difference in climate and more specifically their difference in mean annual rainfall (Table 3.1). Within each production region, two plots were selected which differed primarily on the basis of soil texture (Table 3.2). One being a lighter textured, sandy soil and the other being a heavier clay or loam soil. The two plots in the Lower Olifants River region were both sandy. However, one was a deep, sandy soil of aeolian origin, whereas the other had a shallow surface layer of sandy loam soil overlying Dorbank. The specific soils were selected to represent soils commonly found within each production region. The two plots within each region were selected to be located as close to each other as possible to minimise spatial variability. The two plots were on the same farm for all of the production regions, with the exception of the Lower Olifants River region, where the plots were on separate farms (Table 3.2).

Table 3.1. Wine production regions selected for determining the effect of irrigation with in-field fractionally applied winery wastewater with raw water on soil properties and grapevine responses in different climatic regions according to the Winkler and Köppen-Geiger indices.

Wine region	Locality	Rainfall (1) (mm)	Temperature (2) (°C)	GDD (3)	Köppen-Geiger Index (4)
Coastal	Paarl	469.1	21.8	III	Temperate, dry, warm
Breede River	Robertson	152.9	22.4	V	Arid, steppe, cold
Lower Olifants River	Lutzville & Vredendal	93.6	23.0	V	Arid, desert, hot

(1) Long term average winter rainfall (May to September) (Mulidzi, 2016)

(2) Mean temperature of the warmest month (Myburgh, 2018)

(3) Growing degree days according to the Winkler index (Le Roux, 1974)

(4) Köppen-Geiger climate classification according to Myburgh (2018). Köppen-Geiger index for Stellenbosch is given to describe Coastal region.

Table 3.2. Plots selected for determining the effect of irrigation using in-field fractionally applied winery wastewater with raw water on soil properties and grapevine responses in each wine production region.

Wine region	Site	Plot no.	Soil texture	Co-ordinates	Proximity to the ocean
Coastal	Backsberg farm	C1	LmSa	-33.493525°, 18.55067°	31 km from False Bay
	Backsberg farm	C2	SaClLm	-33.493360°, 18.54588°	31 km from False Bay
Breede River	Madeba farm	BR1	SaLm	-33.483775°, 19.47423°	82 km from Atlantic Ocean
	Madeba farm	BR2	SaClLm	-33.483776°, 19.47355°	82 km from Atlantic Ocean
Lower Olifants River	Lutzville winery	LOR1	Sa	-31.334591°, 18.20570°	21 km from Atlantic Ocean
	Spruitdrift winery	LOR2	LmSa	-31.424412°, 18.30097°	27 km from Atlantic Ocean

The Coastal region plots were located on the Backsberg farm, near Paarl (Fig. 3.1A). Both plots formed part of a newly planted commercial *Vitis vinifera* L. *cv*. Cabernet Sauvignon/US8-7 vineyard which was established in September 2017 (Table 3.3). The plots for the Breede River region were located on Madeba farm, outside of Robertson (Fig. 3.1B). Both plots were part of a commercial *V. vinifera* L. *cv*. Shiraz/SO4 vineyard which was established in 2001 (Table 3.3). In the Lower Olifants River region, a *V. vinifera* L. *cv*. Shiraz/Ramsey vineyard established in 2012 (Table 3.3), was selected near the Lutzville winery (Fig. 3.1C) to represent a deep, sandy soil which is typically found in the Lower Olifants River region. The second experiment plot in this region was located near the Spruitdrift winery outside of Vredendal (Fig. 3.1C). This was a *V. vinifera* L. *cv*. Cabernet Sauvignon/99R vineyard established in 2001 in a shallow, sandy loam soil

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overlying Dorbank (Table 3.3). Each of the six experiment plots were compromised of two rows of ten grapevines each, *i.e.* 20 experiment grapevines per plot. A buffer row of grapevines was located on the one side of each of the experiment rows and two buffer grapevines at each end which received the same treatment. The experiment plots were marked in July and August 2017. The experiment plots were managed according to the grower's normal viticultural practices in terms of fertilisation and canopy management. No winter or summer cover crops were sown at the experiment plots. Weeds were removed routinely by means of chemical and mechanical control.

Table 3.3. Vineyard characteristics, including scion cultivar, rootstock, plant spacing, planting date and trellis system of the experiment plots in the Coastal, Breede River and Lower Olifants River regions where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water.

Plot no. [(1)]	Scion cultivar	Root-stock	Spacing (m x m)	Planting date	Trellis system
C1	Cabernet Sauvignon	US8-7	3.0 × 0.6	2017	modified Lyre
C2	Cabernet Sauvignon	US8-7	3.0 × 0.6	2017	modified Lyre
BR1	Shiraz	SO4	2.5 × 1.5	2001	5-strand lengthened Perold
BR2	Shiraz	SO4	2.5 × 1.5	2001	5-strand lengthened Perold
LOR1	Shiraz	Ramsey	2.0 × 2.0	2012	5-strand lengthened Perold
LOR2	Cabernet Sauvignon	99R	1.5 × 2.6	2001	4-strand lengthened Perold

(1) Refer to Table 3.2. for description of plot numbers.

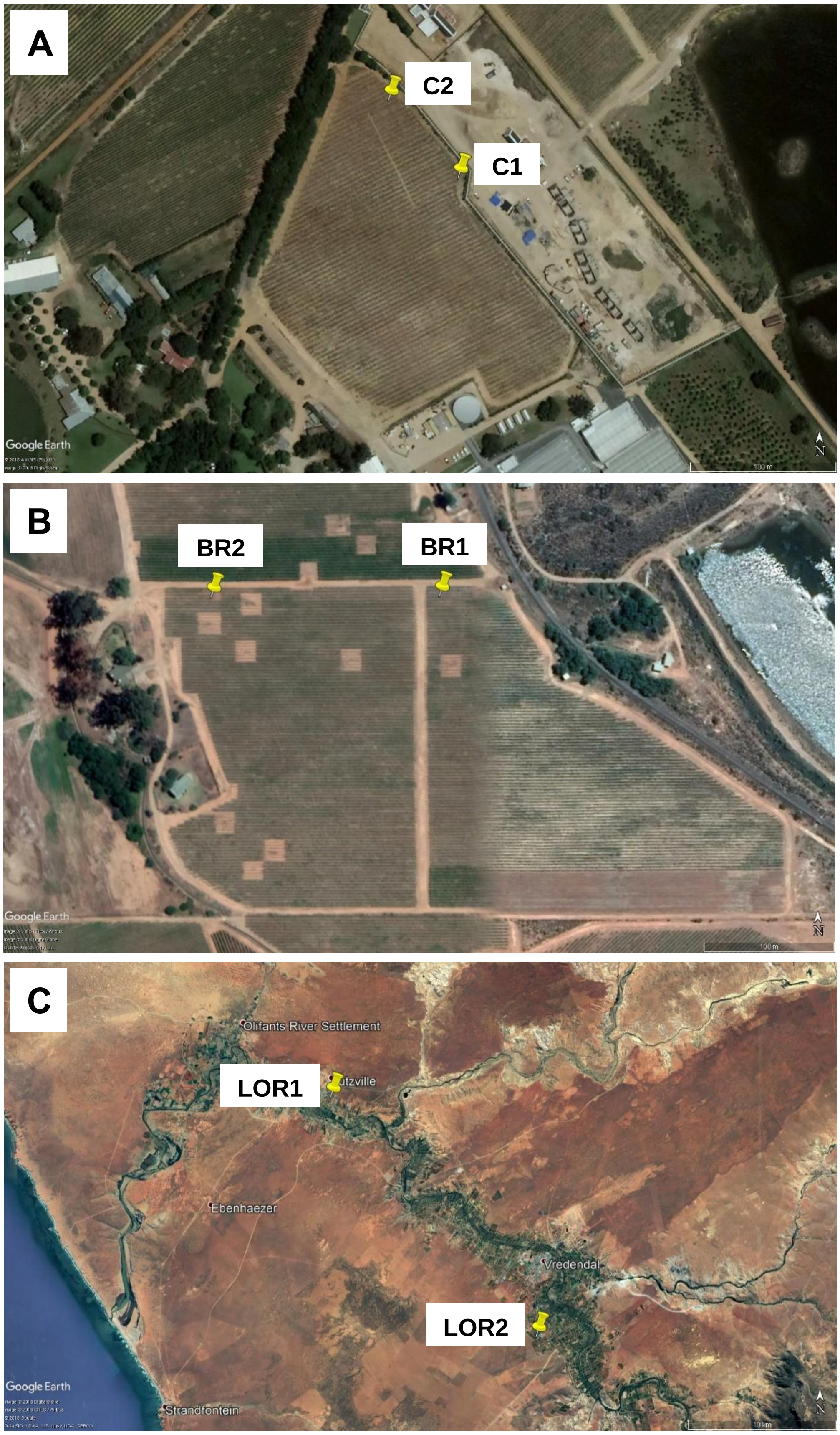

Figure 3.2. Localities of the experiment plots in the (A) Coastal, (B) Breede River and (C) Lower Olifants River regions where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water during the 2017/18 season.

3.2.2. Atmospheric conditions

Weather data, including maximum (T_x) and minimum (T_n) temperatures, rainfall, relative humidity (RH) and average wind speed, were measured by means of automatic weather stations situated near each of the experiment plots. The data was provided by the ARC-Institute for Soil, Climate and Water in Pretoria. Rainfall data for the Coastal region site were provided by the neighbouring Babylonstoren farm.

3.2.3. Irrigation scheduling

3.2.3.1. Soil water status

Soil water content (SWC) was measured by means of the neutron scattering technique as described in Chapter 2. Three Polyvinyl chloride (PVC) access tubes were installed on the grapevine row at each of the six experiment plots before irrigation applications commenced. The count ratios obtained from the neutron probe were calibrated against volumetric soil water content according the methods described in Chapter 2. The mean SWC of each experiment plot was calculated as an average of SWC measured at the three individual access tubes. Measurements were taken in 30 cm increments up to a depth of 90 cm in all plots and up to 180 cm in plots where deeper measurements were possible. Measurements were taken once every two to three weeks as well as before and after every irrigation application.

3.2.3.2. Grapevine water status

Midday stem water potential (Ψ_S) was measured according to the procedures described in Chapter 2. Five leaves were sampled at each of the six experiment plots. Measurements commenced after the application of the first irrigation at each of the experiment plots. At least three measurements were taken at each plot throughout the 2017/18 season.

3.2.4. Irrigation water application

Winery wastewater was fractionally applied by supplying 50% of the grapevines' irrigation water requirement as undiluted winery wastewater and the remaining irrigation water requirement as raw water, *i.e.* a 1:1 ratio. Irrigation water requirements were determined by simultaneously measuring SWC (section 3.2.3.1) and Ψ_S (section 3.2.3.2) following the first irrigation application (Fig. 3.2). Subsequently, the soil was allowed to dry out whilst SWC and Ψ_S measurements were routinely performed to establish a SWC refill zone for vineyard irrigation to obtain good wine quality. Thresholds for grapevine water constraints were used to establish suitable levels of Ψ_S at which grapevines should be irrigated. The young Cabernet Sauvignon grapevines at the C1 and C2 plots were irrigated at a Ψ_S range of -0.85 to -1.15 MPa to obtain moderate water constraints (Myburgh *et al*., 2016). The newly established grapevines were irrigated at lower water constraints than is recommended for full-bearing Cabernet Sauvignon to encourage root development throughout the newly prepared soil profile.

Figure 3.2. Determining the irrigation refill line by means of stem water potential measurements. The circles indicate the leaves enclosed with aluminium bags.

The Shiraz grapevines at the BR1 and BR2 plots were irrigated when Ψ_S was between -1.5 MPa to -1.6 MPa to expose the grapevines to high water constraints which is optimal for attaining sustainable yields and high wine quality (Myburgh, 2018 and references therein). Initially the Shiraz grapevines at the LOR1 plot was going to be irrigated at the same Ψ_S as those at the Breede River site. However, after several Ψ_S measurements at the start of the season, it became evident that the grapevines at the LOR1 plot would not reach such high water constraints and a Ψ_S range of -1.1 MPa to -1.2 MPa was selected to obtain moderate water constraints (Myburgh, 2018 and references therein). The Cabernet Sauvignon grapevines at the LOR2 plot was irrigated when Ψ_S reached a range of -1.4 MPa to -1.5 MPa to attain high water constraints (Myburgh *et al*., 2016).

Irrigation with winery wastewater commenced at the end of November 2017 at the two plots in the Lower Olifants River region. A later than usual start to the wine production season and the smaller capacity of the wineries at the Coastal and Breede River sites caused a delay in the irrigation application, which only commenced in February 2018. The grapevines were irrigated throughout the season as required and irrigation applications ceased for the season in May 2018 at the Coastal and Breede River sites and only in August 2018 in the Lower Olifants River region.

3.2.5. Irrigation water quality

Winery wastewater and raw water samples were collected in cleaned and rinsed 2 ℓ plastic bottles during each irrigation application. Samples were collected in three stages during the irrigation period to ensure the water quality of the samples were representative of the entire duration of the irrigation application. The water samples were analysed by a commercial laboratory (Bemlab,

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Strand) for the same parameters and according to the same procedures as described in Chapter 2. The amounts of elements applied *via* the irrigation water was calculated for both the winery wastewater and raw water according to the methods described by Howell (2016). Chemical oxygen demand was determined at the ARC Infruitec-Nietvoorbij using a portable spectrophotometer (Aqualitic COD-reactor®, Dortmund) with the relevant test kits (COD, CSB, 0–15000 mg/ℓ).

3.2.6. Soil chemical properties

Baseline soil samples were collected at the six experiment plots between July and August 2017 before irrigation applications commenced. Samples were taken again during May 2018 after the majority of irrigations were applied. In order to establish if applied salts were leached from the experiment soils during the winter rainfall period, soil samples were collected again in October 2018. Samples were collected at three positions in each experiment plot along the grapevine row. Samples for each depth were pooled together to create a composite sample. Samples were collected over 30 cm increments to a depth of 60 cm in all plots and up to 300 cm at the LOR1 plot. Samples for each depth were analysed for soil chemical parameters by a commercial laboratory (Bemlab, Strand). Soil $pH_{(KCl)}$, electrical conductivity of the saturated paste extract (EC_e), basic cations, Bray II P and K^+, extractable potassium percentage (EPP´), extractable sodium percentage (ESP´), and soil organic carbon (SOC) was determined according to the same procedures as described in Chapter 2. Trace elements, including boron (B^{3+}), copper (Cu^{2+}), iron (Fe^{2+}), manganese (Mn^{2+}) and zinc (Zn^{2+}) were extracted using ethylenediaminetetraacetic acid (EDTA). Soil sulfur (S) was extracted using calcium phosphate ($Ca(H_2PO_4)_2$). The trace element and S concentrations in the extracts were determined by inductively coupled plasma optical emission spectrometry (ICP-OES) using a spectrometer (PerkinElmer Optima 7300 DV, Waltham, Massachusetts).

3.2.7. Soil physical properties

3.2.7.1. Soil hydraulic properties

Baseline measurements were taken in July and August 2017 before irrigation applications commenced. Thereafter, measurements were taken during May 2018 after the majority of irrigations were applied. In October 2018, measurements were repeated following the winter rainfall period. Raw water used for irrigation from each farm was used for the infiltration measurements. The electrical conductivity (EC_w) and sodium adsorption ratio (SAR) of the raw water was determined after each stage of infiltration measurements. The mean EC_w and SAR measured over the three stages of infiltration measurements are presented in Table 3.4.

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Table 3.4. Mean electrical conductivity (EC_w) and sodium adsorption ratio (SAR) of the raw water used for infiltration measurements at each of the experiment plots during the 2017/18 season.

Plot no. [(1)]	EC_w	SAR
C1	0.20 ± 0.05	1.76 ± 0.00
C2	0.20 ± 0.05	1.76 ± 0.00
BR1	0.32 ± 0.11	2.21 ± 0.23
BR2	0.32 ± 0.11	2.21 ± 0.23
LOR1	0.27 ± 0.16	1.89 ± 0.64
LOR2	0.33 ± 0.18	2.32 ± 0.90

(1) Refer to Table 3.2 for description of plot numbers.

3.2.7.1.1. Near-saturation hydraulic conductivity

Five replications were measured with mini disk infiltrometers at each of the experiment plots according to the procedures described in Chapter 2. Measurements were taken on the grapevine row. Where the soil surface was unsuitable to perform measurements on the grapevine row, measurements were taken between the grapevine row and the tractor tracks. The suction head that was maintained for each plot is presented in Table 3.5. Fixed time intervals were chosen to allow a water level decrease of at least 3 mℓ to minimise reading errors.

Table 3.5. Suction head (cm) maintained with mini disk infiltrometers for near-saturation hydraulic conductivity measurements at each of the experiment plots during the 2017/18 season.

Plot number [(1)]	C1	C2	BR1	BR2	LOR1	LOR2
Suction head (cm)	4	4	1	1	4	1

(1) Refer to Table 3.2 for description of plot numbers.

3.2.7.1.2. Constant head water infiltration rate

Double-ring infiltrometers (Fig. 3.3A) were used to measure five replicates between the grapevine row and the tractor tracks at each experiment plot. Each infiltrometer was constructed by driving two concentric steel cylinders of ±30 cm and ±10.9 cm in diameter, respectively, into the soil (Fig. 3.3B). The smaller of the cylinders- the inner-cylinder, was placed in the centre of the larger, outer-cylinder. A millimetre scale was glued on the inside of the small inner-cylinder to observe fluctuations of the water head. A rectangular piece of 5 mm thick household scouring pad (3M®) was placed in the inner-cylinder to minimise disturbance of the soil surface.

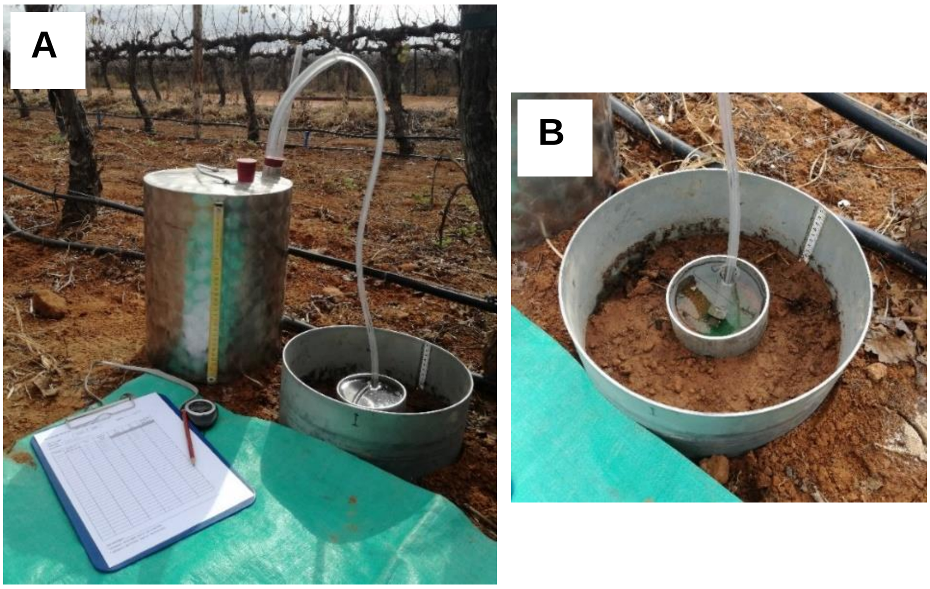

Figure 3.3. Setup of the Mariotte-syphon (A) and steel cylinders (B) of double-ring infiltrometer used to measure constant head water infiltration rate.

The inner-cylinder allowed the formation of a shallow pond in which a constant water head was maintained by means of a Mariotte-syphon (Myburgh *et al.,* 2002). A 25 ℓ steel tank with a glass tube and millimetre scale glued to its side, served as a water reservoir and supply vessel for the measurements. A supply tube of 8 mm was weighted down at its end and placed on the scouring pad. The other end of the supply tube was pushed through a rubber stopper that fitted tightly into a hole at the top of the steel tank. In addition, a "bubble" tube was pushed through the rubber stopper. The bottom of the bubble tube was adjusted to the level at which the constant water head in the infiltrometer was to be maintained (Bouwer, 1986). The Mariotte-syphon works on the principle that the pressure at the bottom of the bubble tube inside the bottle, is at atmospheric value, which then maintains the water surface (head) in the infiltrometer at the same height as the end of the bubble tube (Bouwer, 1986). The soil in the outer-cylinder was wetted with approximately 2 ℓ of raw irrigation water before the readings commenced. This served as a "primer" that would initially wet the soil and prevent lateral movement of water from the inner-cylinder.

Air was blown into the bubble tube to initiate the flow of water out of the supply tube. The bubble tube was used to adjust the rate at which water flowed into the inner-cylinder until a constant state was observed between the infiltration rate (IR) and water supply (flow) rate, *i.e.* when a constant head was achieved in the inner-cylinder. At this point the time was recorded and a reading was taken from the millimetre scale on the side of the steel tank. Measurements were taken at fixed time intervals that allowed an observable difference of approximately 3 mm to minimise reading errors.

The IR values (mm/h) were calculated by means of the following equation:

$$IR = [(H_i - H_f) \div \Delta t] \times 4337 \quad \text{(Eq. 3.1)}$$

where H_i is the initial water height reading (mm), H_f is the water height reading (mm) at the end of the measurement, Δt is the difference in time between measurements (min) and 4337 is the conversion factor used to convert the volume of water infiltrated per Δt to mm/h.

3.2.8. Vegetative grapevine measurements

3.2.8.1. Growth characteristics

The experiment grapevines in the Breede River and Lower Olifants River regions were pruned to two bud spurs in July and August 2017. The baseline cane mass per grapevine was determined at pruning using a hanging balance. Grapevine trunk circumference and cordon length were measured. The number of spurs per grapevine was recorded for each plot. In July 2018, the cane mass per grapevine was measured at the abovementioned sites following one season of irrigation. The young grapevines in the Coastal region were pruned to two buds and the mean cane mass per grapevine was determined.

3.2.8.2. Leaf and shoot chemical status

Leaf samples were collected at harvest during the 2017/18 growing season according to the methods described in Chapter 2. Shoot samples consisting of 8 to 10 randomly selected primary canes per experiment plot were collected at pruning in July 2018. Both the leaf and shoot samples were dried in a fan oven at 60°C for 24 hours. The dried leaf and shoot chemical status were determined according to the procedures described in Chapter 2.

3.2.9. Yield and its components

3.2.9.1. Bunch and berry mass at harvest

Mean bunch mass was determined by randomly picking ten bunches at harvest at the Breede River and Lower Olifants River plots and weighing them in the laboratory using an electronic balance. In order to determine berry mass, 15 berries were sampled from each of the ten selected bunches to obtain a sample size of 150 berries per experiment plot. Berries were picked at different positions along the longitudinal bunch axis. The berry samples were weighed in the laboratory to determine mean berry mass.

3.2.9.2. Yield

The grapes at the aforementioned sites were harvested as close as possible to a total soluble solids (TSS) value of 24°B. At harvest, all the picked bunches were counted and weighed using a top loader mechanical balance to determine the total mass of grapes at each experiment plot. Grape mass per grapevine (kg/grapevine) was calculated by dividing the total grape mass per plot by the number of experiment grapevines at each plot.

3.2.10. Grape juice characteristics

Representative samples of grapes were collected at harvest at the Breede River and Lower Olifants River plots and analysed for TSS, total titratable acidity (TTA) and pH according to standard procedures of the winery at the ARC Infruitec-Nietvoorbij as described by Howell *et al*. (2016a).

3.2.11. Statistical procedures

Calculations of means and standard deviations (SD) were carried out using the Microsoft Office Excel 365 version.

3.3. RESULTS AND DISCUSSION

3.3.1. Atmospheric conditions

3.3.1.1. Air temperature

Maximum temperature: During early spring (September & October 2017), the experiment plots in the Coastal region had the lowest mean T_x, followed by the plots in the Breede River region, the LOR1 and LOR2 plots (Fig. 3.4). A similar trend was observed from April to August 2018 (Fig. 3.4). The lower temperatures measured during these periods at the plots in the Coastal region were likely the result of the region's Mediterranean climate with mild, wet winters and warm, dry summers (Schulze, 1972). Furthermore, during the coldest month (August 2018), the T_x at the Coastal region plots was *ca*. 3°C lower than the LOR1 plot. The effect of the sea breeze from False Bay has been shown to have a significant effect on the T_x in the Stellenbosch wine producing region (Bonnardot *et al*., 2002). Although the effect of the sea breeze could be observed up to 35 km inland, the cooling effect decreased rapidly with distance from the coast. The highest T_x was measured during February 2018 and was similar for all of the experiment sites (Fig. 3.4). The February T_x was 31.9°C, 31.5°C, 31.1°C and 31.2°C for the Coastal, Breede River, LOR1 and LOR2 plots, respectively. The T_x recorded for the month of February fell outside the range of 20°C to 30°C which is considered optimal for photosynthesis (Ferrini *et al*. 1995 and references therein). The lowest net photosynthesis rates of grapevines were recorded at 35°C and was ascribed to enzymatic biochemical factors rather than the functioning of stomata (Ferrini *et al*., 1995). Furthermore, enhanced degradation and reduced synthesis of anthocyanins occur at temperatures above 25°C (Mori *et al*., 2007). Air temperatures above 25°C and 30°C during the ripening period have been shown to reduce anthocyanin content in Cabernet Sauvignon and Shiraz grapes (Barnuud *et al*., 2014). Since the T_x during the ripening period (January to March) at all the experiment plots were above 25°C (Fig. 3.4), anthocyanin content in the berries of the plots in the Breede and Lower Olifants River regions may be reduced. The biggest variation in T_x between the experiment plots were observed during June 2018 (Fig. 3.4). An increase in T_x of *ca*. 1°C was observed for each plot from the coldest (Coastal region plots) to the warmest (LOR1 plot).

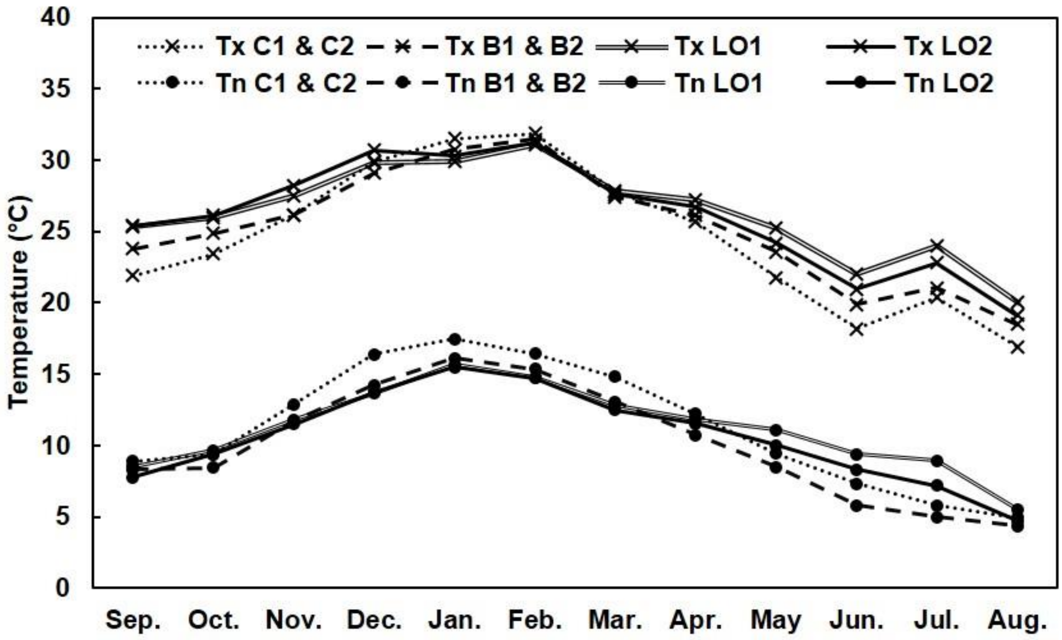

Figure 3.4. Daily maximum (Tx) and minimum (Tn) temperatures during the 2017/18 growing season at the experiment plots where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Minimum temperature: The highest mean T_n were measured during January 2018 (Fig. 3.4). The plots in the Coastal region experienced the highest T_n during the summer months of November to March, whereas the T_n of the other experiment plots were lower and relatively similar to each other during this particular period (Fig. 3.4). In fact, the T_n measured at the Breede River, LOR1 and LOR2 plots remained relatively similar throughout the season, with considerable variations in T_n only visible from May 2018 onwards. High T_n in early winter (particularly May) can result in delayed bud break (Archer & Goussard, 1988). A threshold maximum temperature of 9.9°C has been suggested for grapevines in the Western Cape (Van Schalkwyk, 2013). Daily T_n above this norm was observed at the two experiment plots in the Lower Olifants River region during May 2018. Therefore, a delay in bud break of the 2018/19 season was expected at these particular plots.

3.3.1.2. Relative humidity

The mean monthly RH varied between 50.3% and 74.4% (Fig. 3.5). Except for the plots in the Coastal region, the mean RH remained relatively constant at the experiment plots throughout the 2017/18 season (Fig. 3.5). The RH at the Coastal region plots tended to fluctuate more throughout the season which may be the result of the plots' proximity to the coast. Previous research indicated higher RH for localities near the coastline compared to more inland areas (Bonnardot *et al*., 2002). High RH could increase the potential occurrence of diseases such as powdery mildew (Carroll & Wilcox, 2003).

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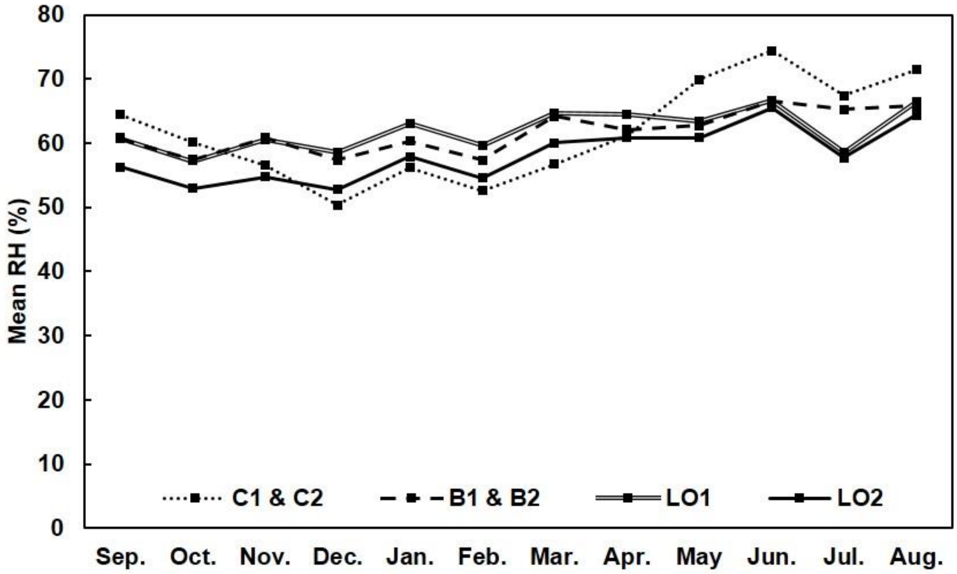

Figure 3.5. Daily mean relative humidity (RH) measured during the 2017/18 growing season at the experiment plots where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.1.3. Rainfall

The total rainfall during the 2017/18 season was 538.7 mm, 218.0 mm, 149.9 mm and 157.2 mm for the Coastal region, Breede River region, LOR1 and LOR2 plots, respectively. Most of the rainfall occurred during May to August (Fig. 3.6). The experiment plots in the Breede River region also received appreciable amounts of rainfall during the summer months of February and March. As expected, the experiment plots in the Coastal region received the highest amounts of rainfall throughout the 2017/18 season. The mean monthly rainfall at this site was 44.9 mm, compared to 18.2 mm, 12.5 mm and 13.1 mm at the Breede River, LOR1 and LOR2 plots, respectively. The rainfall at the LOR1 and LOR2 experiment plots remained relatively similar throughout the season, with the exception of June and July when the LOR2 plot received substantially more rainfall. The relatively low rainfall that occurred in the Lower Olifants River region during the study period was likely to necessitate the application of larger volumes of irrigation water more frequently compared to the other two regions.

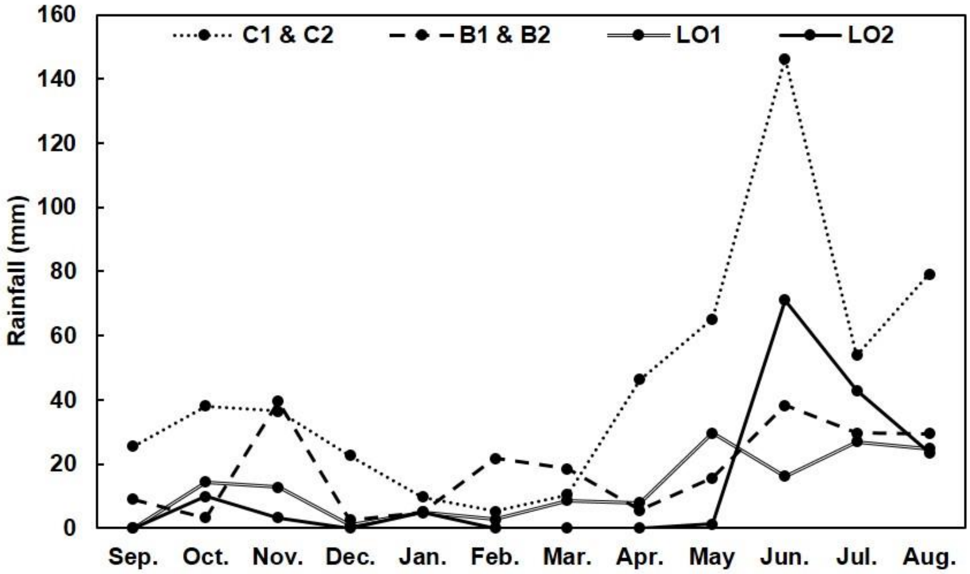

Figure 3.6. Total monthly rainfall measured during the 2017/18 growing season at the experiment plots where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.1.4. Wind speed

The highest wind speeds in the Coastal and Lower Olifants River regions were recorded during November, December and January (Fig. 3.7). This agrees with previously reported wind speeds for these regions (Bruwer, 2010; Mehmel, 2010). The experiment plots in the Breede River region experienced far lower wind speeds throughout the 2017/18 season (Fig. 3.7). The higher wind speeds measured at the Coastal and Lower Olifants River plots were likely the result of their proximity to the ocean. Wind speeds above 2 m/s can initiate the removal of accumulated cold units and subsequently have a cooling effect on grapevines (Williams *et al*., 1994 and references therein). Since the wind speeds in the Coastal and Lower Oliants River regions were frequently above this value throughout the 2017/18 season, substantial amounts of accumulated heat units could have been lost. The effects thereof on grapevine physiology can either be positive (in a warm climate) or negative (in a cool climate) (Mehmel, 2010). Furthermore, high wind speeds induce stomatal closure in grapevine leaves (Freeman *et al*., 1982). Grapevine transpiration can be reduced at wind speeds above 4 m/s (Campbell-Clause, 1998). However, the wind speeds measured at the respective experiment plots throughout the 2017/18 season remained below this threshold.

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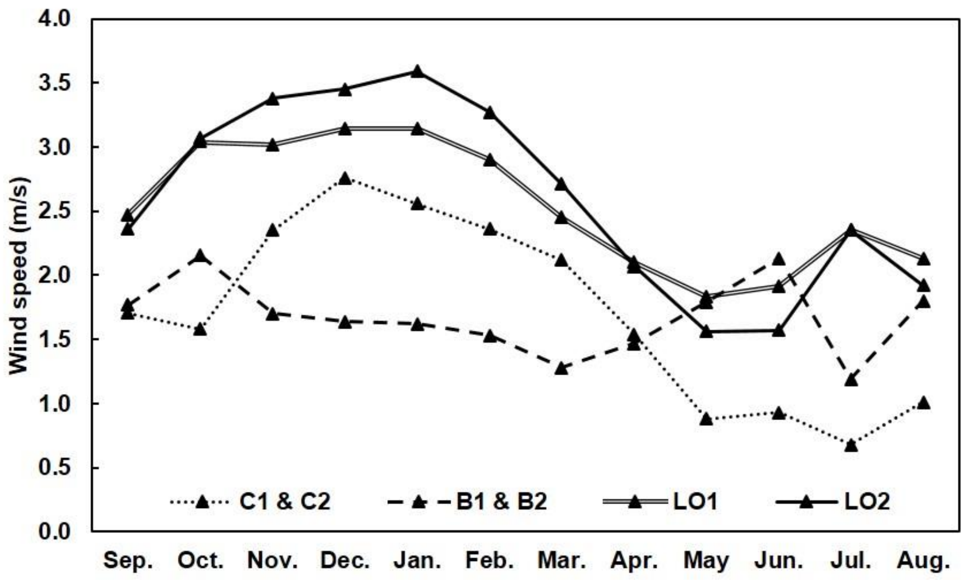

Figure 3.7. Daily mean wind speed measured during the 2017/18 growing season at the experiment plots where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.2. Irrigation

3.3.2.1. Amounts of irrigation water applied

Amounts per irrigation: The mean amounts of winery wastewater and raw water applied per irrigation at each of the experiment plots during the 2017/18 season are presented in Table 3.6. A full tank of winery wastewater applied *ca*. 31 mm, 35 mm and 40 mm irrigation at the Coastal, Breede River and Lower Olifants River region plots, respectively. Following the application of winery wastewater, the same volume of raw water was applied.

Table 3.6. Mean amounts of irrigation water applied per irrigation during the 2017/18 season at the experiment plots where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water.

Plot no. (1)	Amount of winery wastewater applied per irrigation (mm)	Amount of raw water applied per irrigation (mm)	Total amount of irrigation water applied per irrigation (mm)
C1	30.5 ± 6.2	27.9 ± 9.8	58.4 ± 15.9
C2	30.5 ± 6.2	27.9 ± 9.8	58.4 ± 15.9
BR1	33.5 ± 8.3	33.1 ± 8.1	66.8 ± 16.5
BR2	35.3 ± 5.2	33.5 ± 2.6	68.8 ± 7.8
LOR1	38.0 ± 11.5	38.1 ± 11.3	76.2 ± 22.7
LOR2	39.9 ± 4.0	39.9 ± 3.8	79.8 ± 7.6

(1) Refer to Table 3.2. for description of plot numbers.

Seasonal amounts: The total seasonal amounts of winery wastewater and raw water applied during the 2017/18 season are presented in Table 3.7. Due to delays in the installation of infrastructure and the unavailability of winery wastewater at the sites in the Coastal and Breede River regions, only two irrigations could be applied at these plots before the winter rainfall period.

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Therefore, the total amounts of irrigation water applied at these plots were much less compared to the plots in the Lower Olifants River region. Since the irrigation infrastructure in the Lower Olifants River region was installed earlier, these two plots received more irrigation applications which resulted in higher total amounts of irrigation water for the season.

Table 3.7. Total seasonal amounts of irrigation water applied during the 2017/18 at the experiment plots where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water.

Plot no. (1)	Total amount of winery wastewater applied (mm)	Total amount of raw water applied (mm)	Total amount of irrigation water applied (mm)
C1	61.0	55.9	116.9
C2	61.0	55.9	116.9
BR1	67.0	66.7	133.7
BR2	70.7	66.9	137.6
LOR1	228.1	228.9	456.9
LOR2	279.4	279.3	558.7

(1) Refer to Table 3.2. for description of plot numbers.

3.3.2.2. Irrigation water quality

pH: The pH of the winery wastewater used in the Coastal and Breede River regions were generally lower than the raw water used in the respective regions (Appendices 4.1 & 4.3). In addition, the winery wastewater used in these regions did not meet the minimum pH value of 6.5 which is recommended for irrigation water (DWAF, 1996; Howell & Myburgh, 2013). Previous studies have reported similar pH values for winery wastewater produced during harvest (Kumar *et al*., 2006; Sheridan *et al*., 2011; Howell *et al*., 2016b). Reduced pH during the harvest period can be ascribed to organic acids present in grapes (Mosse *et al*., 2011), as well as grape juice and wine with inherently low pH which spill into the wash water during winemaking activities (Sheridan *et al*., 2011). The degradation of ethanol in wine to acetic acid may further reduce the pH of winery wastewater (Howell, 2016 and references therein). In a survey in which the composition of winery wastewater was observed at ten different wineries, Mulidzi *et al*. (2009b) reported frequent pH values below 4 during harvest. In contrast, the winery wastewater used at the LOR1 and LOR2 experiment plots were near-neutral prior to and during the harvest period (Appendices 4.5 & 4.7). This was likely due to the addition of lime ($CaCO_3$) to wastewater by the wineries to increase its pH. According to the General Authorisations regarding the re-use of wastewater for irrigation, a pH of between 6 and 9 is required to irrigate up to 500 m^3 of wastewater per day (DWA, 2013). The winery wastewater used at the Coastal region plots did not meet the prescribed criteria during either of the two irrigations (Appendix 4.1). The experiment plots in the Breede River region was irrigated with non-compliant winery wastewater during the second irrigation application (Appendix 4.3). Generally, the winery wastewater used in the Lower Olifants River region met the abovementioned criteria (Appendices 4.5 & 4.7). However, there were instances in the post-harvest period when the pH of the winery wastewater was lower than the prescribed norm. A

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similar trend was observed at a winery in Stellenbosch and attributed the reduced pH to a once-off pulse of high strength wastewater which may have been the result of wine-handling practices (Sheridan *et al*., 2011). Therefore, careful monitoring and, if necessary, pH adjustment of the winery wastewater is recommended before irrigation application.

Electrical conductivity: Levels of EC_w in the wastewater from the four respective wineries ranged between 0.7 dS/m and 5.3 dS/m (Appendices 4.1, 4.3, 4.5 & 4.7). Wastewater from the winery in the Coastal region generally had the lowest and most consistent EC_w (Appendix 4.1). In contrast, the EC_w of the winery wastewater used in the Breede River region varied greatly between the two irrigations applied during the 2017/18 season (Appendix 4.3). The wastewater produced by the winery at the LOR1 plot had more consistent but relatively high EC_w levels (Appendix 4.5). The winery wastewater used at the LOR2 plot had variable EC_w throughout the season, but levels were comparable to the LOR1 plot (Appendix 4.7). With regard to the General Authorisations, a maximum EC_w of 2 dS/m is allowed for wastewater used to irrigate up to 500 m^3 per day (DWA, 2013). The winery wastewater used during the second irrigation at the plots in the Breede River region, as well as all the irrigations applied at the plots in Lower Olifants River region did not comply with the legislation. Therefore, the wastewater at these wineries would have to undergo additional treatment to reduce its salinity before it would be suitable to use for irrigation. Irrigating using a more diluted ratio of winery wastewater to raw water, *e.g*. 1:3, may be a possible solution. Furthermore, all of the winery wastewaters used during this study exceeded the recommended EC_w norm of 0.75 dS/m for water used to irrigate grapevines (Van Zyl, 1981).

Chemical oxygen demand: The COD levels measured in the winery wastewaters ranged between 1 760 mg/ℓ and 12 380 mg/ℓ (Appendices 4.1, 4.3, 4.5 & 4.7). The range measured in the present study fell within the range of 340 mg/ℓ to 49 105 mg/ℓ previously reported for winery wastewater (Conradie *et al*., 2014 and references therein). The winery wastewater used at the plots in the Coastal region had the lowest COD levels (Appendix 4.1), whereas the winery wastewater used at the LOR2 plot generally had the highest COD levels (Appendix 4.7). In a previous study in the Breede River region, Howell *et al*. (2016b) reported an increase in COD of winery wastewater as the harvest progressed. Similar results were observed with the present study. Furthermore, the winery wastewater used at the LOR1 experiment plot reached its peak COD levels towards the end of harvest (Appendix 4.5). This may have been the result of wine stabilisation processes taking place within the winery (Conradie *et al*., 2014). The General Authorisations permits the use of up to 50 m^3, 500 m^3 and 2 000 m^3 of wastewater for irrigation, provided that the COD is below 5 000 mg/ℓ, 400 mg/ℓ and 75 mg/ℓ, respectively (DWA, 2013). Based on this norm, none of the winery wastewaters used during this study could be used to irrigate more than 50 m^3 per day. Therefore, treatment or dilution of the winery wastewater to reduce its COD levels would be necessary if significant amounts of the wastewater is to be used for irrigation.

Nitrogen: The levels of N in the wastewater differed substantially between the different wineries. During the first irrigation application at the Coastal region plots, neither of the two water types contained any measured form of N (Appendix 4.1). In contrast, the second irrigation application had similar values of ammonium-nitrogen (NH_4-N) for the winery wastewater and raw water. This was probably due to high levels of NH_4-N present in the raw water which was then used in the cellar for cleaning purposes. The wastewater obtained from the winery, as well as the raw water used in the Breede River region, did not contain any of the measured forms of N during either of the two irrigation applications (Appendix 4.3). The lack of N in the irrigation water used at the plots in the Breede River region meant that it served no benefit in terms of N fertilisation as have been reported previously for winery wastewater used for irrigating sunflowers and maize (Shilpi *et al*., 2018). The winery wastewater used at the LOR1 experiment plot had a mean NH_4-N concentration of 15.8 mg/ℓ, whereas the winery wastewater used at the LOR2 plot had a mean concentration of 9.9 mg/ℓ. It should be noted that, besides the presence of a small concentration at the Coastal region plots during the second irrigation application, none of the winery wastewaters contained any nitrate-nitrogen (NO_3-N) throughout the 2017/18 season (Appendices 4.1, 4.3, 4.5 & 4.7). Previous studies have indicated that winery wastewater may contain between 0 mg/ℓ to 415 mg/ℓ total-N (Mosse *et al*., 2011 and references therein). Except for a spike in NH_4-N during the second irrigation at the Coastal region plots and sixth irrigation at the LOR1 plot, the respective raw waters used during the study contained negligible amounts of N.

Phosphorous: The winery wastewaters had P concentrations which ranged between 1.5 mg/ℓ and 24.5 mg/ℓ (Appendices 4.1, 4.3, 4.5 & 4.7). In contrast, with the exception of negligible concentrations at the LOR2 plot, the raw waters used did not contain any P. Currently, there are no guidelines for P levels in irrigation water (DWAF, 1996), but ANZECC (2000) recommended a long-term critical value of 0.05 mg/ℓ to prevent the development of algal blooms in water storage facilities and minimise the risk of bio-fouling in irrigation equipment. The levels of P in all the winery wastewaters exceeded this norm, thereby indicating a risk for algal blooms in winery wastewater storage facilities.

Calcium: The Ca^{2+} concentrations in the winery wastewater was consistently higher than in the raw water at all the experiment plots throughout the 2017/18 season (Appendices 4.1, 4.3, 4.5 & 4.7). The wastewater produced by the winery in Coastal region had the lowest Ca^{2+} concentrations (Appendix 4.1), followed by the winery wastewater used in the Breede River region (Appendix 4.3). Compared to these two regions, the winery wastewaters used at the two respective plots in the Lower Olifants River region had substantially higher Ca^{2+} concentrations (Appendices 4.5 & 4.7). This was likely the result of the latter two wineries adding $CaCO_3$ to their wastewater to increase its pH. There are no guidelines for levels of Ca^{2+} in irrigation water in South Africa (DWAF, 1996). However, it is an important parameter to determine since it is used in the calculation of the sodium adsorption ratio (SAR) and potassium adsorption ratio (PAR). Appreciable amounts of

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Ca^{2+} and Mg^{2+} in irrigation water may reduce the SAR and PAR which could contribute to mitigate the negative effects of Na^+ and K^+ on soil structural stability (Howell, 2016). Furthermore, the high concentrations of Ca^{2+} in the winery wastewaters used in the Lower Olifants River region may be a source of Ca^{2+} required by grapevines.

Magnesium: Levels of Mg^{2+} in the winery wastewater and raw water used in the Coastal region were comparable (Appendix 4.1). This is likely an indication of inherent Mg^{2+} concentrations in the raw water which is used in the cellar, and that practices in the winery only contributed a small amount of Mg^{2+} to the wastewater. In contrast, the winery wastewater used at the Breede River plots had relatively higher Mg^{2+} levels compared to the raw water (Appendix 4.3). This suggested that the higher concentrations in the wastewater was probably a result of practices in the winery. Both Ca^{2+} and Mg^{2+} may be present in winery wastewater due to their natural occurrence in grape juice (Mosse *et al*., 2011 and references therein). Similar trends were observed for the winery wastewaters used at the two experiment plots in the Lower Olifants River region (Appendices 4.5 & 4.7). However, Mg^{2+} concentrations in the winery wastewater used at these plots were higher compared to the Breede River region. Similar to Ca^{2+}, there are no guidelines for levels of Mg^{2+} in irrigation water (DWAF, 1996), but it may also play a role in reducing the SAR (Howell, 2016).

Potassium: The levels of K^+ in the winery wastewater were consistently higher compared to the raw water at all of the experiment plots during the 2017/18 season (Appendices 4.1, 4.3, 4.5 & 4.7). Mulidzi *et al*. (2009b) proposed a threshold concentration of 200 mg/ℓ as being high for K^+. Except for the winery wastewater applied during the first irrigation at the Breede River plots, the K^+ concentrations in the winery wastewater applied at all of the plots exceeded this norm throughout the study period. The high K^+ levels present in the winery wastewaters were likely the result of cleaning agents used in the cellars, as well as grape lees and spillage from grape fermentation processes (Arienzo *et al.*, 2009a; Laurenson *et al.*, 2012; Howell *et al*., 2016b). As K^+ is an essential plant nutrient, the levels of K^+ present in the irrigation water could make an important contribution towards the K^+ requirement of the irrigated grapevines. However, applying excessive amounts of K^+ to soils may result in reduced hydraulic conductivity (*K*) and IR of soils (Levy & Van der Watt, 1990). Furthermore, applying amounts of K^+ in excess of grapevine requirements could result in increased juice pH and ultimately reduce wine quality (Kodur, 2011).

Potassium adsorption ratio: Throughout the 2017/18 season, the PAR levels of the winery wastewater were consistently higher than the raw water at all of the experiment plots (Appendices 4.1, 4.3, 4.5 & 4.7). The winery wastewater PAR levels measured during the study period was within the range of 1.7 to 10.8 previously reported for a winery in the Breede River region (Howell *et al*., 2016b). The relatively high PAR values of the winery wastewater were most likely due to the high concentrations of K^+ in the wastewater. The latter could have originated from K-based cleaning agents used by the wineries, as well as grape lees and spillage from grape fermentation processes (Arienzo *et al.*, 2009a; Laurenson *et al.*, 2012; Howell *et al*., 2016b). Despite the high

K^+ concentrations in the wastewater from the wineries in the Lower Olifants River region, their PAR values were lower than the wastewater from the other wineries. This was a result of the relatively high Ca^{2+} concentrations present in the winery wastewater used at the LOR1 and LOR2 plots. In a laboratory study, it was shown that PAR values above 20 may reduce soil *K* (Arienzo *et al*., 2009b). Although the PAR values of the winery wastewater used in the present study were below 20, the accumulation of K^+ in the soil due to long-term irrigation with the winery wastewater remains a concern.

Sodium: The wastewaters produced by the wineries in the Coastal and Breede River regions had Na^+ concentrations which were marginally higher compared to the respective raw waters (Appendices 4.1 & 4.3). Since the raw water at these localities are also used for cleaning operations in the wineries, it appears that practices in the winery only contributed a small amount of Na^+ to the wastewater. The low Na^+ and high K^+ concentrations in the wastewater at these two wineries confirm the use of K-based cleaning agents as opposed Na-based products. The use of Na-based products in wineries may lead to an accumulation of Na^+ in the receiving environment and may be particularly detrimental where the wastewater is irrigated onto land (Mosse *et al*., 2011). Therefore, numerous studies and reports have recommended the use of K-based cleaning agents instead to limit adverse effects when applying winery wastewater to soils (Chapman, 1996; Van Shoor, 2005; Mosse *et al*., 2011). In contrast to the wineries in the Coastal and Breede River regions, the high Na^+ concentrations of the wastewater produced by the two wineries in the Lower Olifants River region indicate the use of Na-based cleaning agents (Appendices 4.5 & 4.7). Grapevines are considered moderately sensitive to Na^+ foliar injury, therefore a maximum concentration of 115 mg/ℓ is recommended for overhead irrigation (DWAF, 1996; Howell & Myburgh, 2013). The Na^+ concentrations of the wastewater produced by the winery at the LOR2 plot frequently exceeded this norm throughout the 2017/18 season (Appendix 4.7). Consequently, a risk for leaf scorching may exist at this plot if the leaves are wetted by the micro-sprinklers.

SAR: Levels of SAR in the winery wastewaters at the Coastal, Breede River and LOR1 plots tended to be lower than in the raw waters for most of the 2017/18 season (Appendices 4.1, 4.3 & 4.5). The lower SAR could largely be explained by the Ca^{2+} present in the wastewater which decreased the SAR. This effect was clearly seen with the application of the first irrigation at the LOR2 plot. During this irrigation, the winery wastewater had low Ca^{2+} levels and high Na^+ which resulted in a very high SAR (Appendix 4.7). The SAR of the subsequent irrigations decreased drastically as a result of higher Ca^{2+} and lower Na^+ levels. Furthermore, with the exception of the winery wastewater applied with the first irrigation at the LOR2 plot, the SAR of all of the winery wastewaters applied during the 2017/18 season was below the maximum permissible limit of 5 as prescribed by the General Authorisations (DWA, 2013). Likewise, the SAR of both the winery wastewaters and raw waters used were below the threshold value of 10 which is recommended for water used to irrigate grapevines (Richards, 1954).

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Trace elements: Except for the presence of small amounts of B^{3+}, Fe^{2+}, Mn^{2+} and Zn^{2+} at some of the experiment plots, the raw waters generally did not contain any trace elements throughout the 2017/18 season (Appendices 4.2, 4.4, 4.6 & 4.8). In contrast, the winery wastewaters had levels of B^{3+} which ranged between 0.1 mg/ℓ and 1.0 mg/ℓ (Appendices 4.2, 4.4, 4.6 & 4.8). A level of 0.5 mg/ℓ B^{3+} was previously proposed as being ideal for irrigating grapevines (McCarthy *et al*., 1988). This norm was exceeded in wastewaters from the wineries in the Breede River and Lower Olifants River regions (Appendices 4.4, 4.6 & 4.8). Boron is an essential plant nutrient, but it can be toxic even at low concentrations. Grapevines are particularly sensitive to B^{3+} toxicities (Van Zyl, 1981; Ayers & Westcott, 1985; DWAF, 1996; ANZECC, 2000). A recent study by Howell *et al*. (2016b) reported that winery wastewater may have a sporadic risk of inducing B^{3+} toxicities if it is used for vineyard irrigation.

Levels of Mn^{2+} in the winery wastewater varied between 0.1 mg/ℓ and 0.6 mg/ℓ (Appendices 4.2, 4.4, 4.6 & 4.8). The recommended maximum concentration of 0.2 mg/ℓ (Van Zyl, 1981) was exceeded numerous times throughout the season at the plots in the Breede River and Lower Olifants River regions (Appendices 4.4, 4.6 & 4.8). Apart from one occasion in the Breede River region and two occasions at the LOR1 plot, the winery wastewater did not contain any measurable amounts of Cu^{2+}. The winery wastewater used in the Breede River region had concerningly high Zn^{2+} concentrations (Appendix 4.4), which exceeded the maximum concentration of 2 mg/ℓ which is recommended for grapevines under continuous irrigation (Van Zyl, 1981). According to South African irrigation water quality guidelines, Zn^{2+} toxicities may induce Fe^{2+} deficiencies in sensitive plants (DWAF, 1996). The recommended maximum Fe^{2+} concentration of 5 mg/ℓ (Van Zyl, 1981) was exceeded once during the season at both the plots in the Coastal region and the LOR2 plot (Appendices 4.2 & 4.8). Furthermore, the threshold value of 1.5 mg/ℓ Fe^{2+} at which drip irrigation systems may become clogged (DWAF, 1996) was exceeded multiple times throughout the season at all of the experiment plots (Appendices 4.2, 4.4, 4.6 & 4.8). Therefore, a risk of irrigation equipment becoming clogged may exist if the particular winery wastewater is used for irrigation.

Chloride: The winery wastewater and raw water used at the experiment plots in the Coastal and Breede River regions had similar chloride (Cl^-) concentrations (Appendices 4.2 & 4.4). This could have been the result of chlorination disinfection processes used to treat the raw water. Ultimately, the raw water is used in the wineries for cleaning purposes and forms the base of the winery wastewater. In the case of these two wineries, the amounts of Cl^- added to the wastewater through winery practices were relatively small. In contrast, the winery wastewater used at the plots in the Lower Olifants River region had Cl^- concentrations that were far greater than the respective raw water (Appendices 4.6 & 4.8). The maximum threshold value recommended for overhead irrigation of vineyards is 150 mg/ℓ Cl^- (Van Zyl, 1981). The Cl^- concentration in the winery wastewater at these plots frequently exceeded this norm throughout the 2017/18 season. As a result, the grapevine leaves may be susceptible to leaf scorching if they are wetted by the micro-sprinklers.

Bicarbonate: The bicarbonate (HCO_3^-) content of the winery wastewaters varied considerably throughout the season and between wineries (Appendices 4.2, 4.4, 4.6 & 4.8). The wastewater produced by the winery in the Coastal region did not contain any HCO_3^- (Appendix 4.2). The winery wastewater used at the Breede River plots did not contain any HCO_3^- during the first irrigation but increased to 2 361 mg/ℓ with the second irrigation (Appendix 4.4). At this stage, the reason for the significant increase is uncertain as the raw water remained at similar levels for both irrigations (Appendix 4.4). Similar HCO_3^- concentrations were measured in the winery wastewater of both plots in the Lower Olifants River region throughout the study period (Appendices 4.6 & 4.8). The high HCO_3^- values of the winery wastewaters exceeded the upper threshold of 518.6 mg/ℓ (8.5 me/ℓ) which indicated slight to moderate restriction when used for overhead irrigation (Ayers & Westcott, 1985). In addition, applying excessive amounts of HCO_3^- to the soil may lead to the precipitation of insoluble Ca- and Mg-carbonates when the soils dry out (Van Zyl, 1981; McCarthy *et al.*, 1988). This is of particular concern at the plots in the Lower Olifants River region as the atmospheric conditions are dry (Fig. 3.4) and the wastewaters also have high Ca^{2+} concentrations (Appendices 4.6 & 4.8).

Sulfate: With the exception of the Coastal region, the winery wastewaters consistently had higher sulfate (SO_4^{2-}) concentrations compared to the raw waters (Appendices 4.4, 4.6 & 4.7). Currently there are no guidelines available for the permissible levels of SO_4^{2-} in irrigation water in South Africa (DWAF, 1996). However, the measurement of SO_4^{2-} in irrigation water is important since waters with high concentrations of both SO_4^{2-} and Ca^{2+} may result in the precipitation of gypsum ($CaSO_4$) in irrigation equipment and subsequent clogging of equipment (Du Plessis *et al.*, 2017). A maximum SO_4^{2-} concentration of 250 mg/ℓ was previously proposed for reclaimed effluent used to irrigate grapevines (Ryder, 1995). This norm was exceeded frequently throughout the study period in the wastewaters produced by the two wineries in the Lower Olifants River region (Appendices 4.6 & 4.8). Furthermore, a sharp increase in SO_4^{2-} concentrations in the winery wastewater used at the LOR2 plot was evident at the beginning of the harvest period (Appendix 4.8). The high SO_4^{2-} concentrations present in the wastewater during this period may be a result of acidification and wine stabilisation operations carried out in the winery (Conradie *et al.*, 2014 and references therein).

Fluoride: No trends were observed with regard to the fluoride (F^-) content in any of the winery wastewaters or raw waters used during the 2017/18 season (Appendices 4.2, 4.4, 4.6 & 4.8). Furthermore, all the irrigation waters had F^- concentrations well below the maximum concentration of 1 mg/ℓ recommended by Ayers and Westcot (1985).

3.3.2.3. Amounts of elements applied

3.3.2.3.1. Coastal region

Nitrogen: The N applied *via* the irrigation water at the two experiment plots in the Coastal region (Table 3.8) were the result of high NH_4-N concentrations present in both water types during the second irrigation (Appendix 4.1). On medium to heavy textured soils, young grapevines annually require a total of 56 kg/ha N applied in two equal instalments- once during bud break (around September) and again in November (Saayman, 1981). Consequently, not only were the amounts of N applied *via* the irrigation water in excess of the grapevines' requirements, but it was also applied during the wrong growth period (March). It should also be noted that the high N levels in the irrigation water was a result of high NH_4-N concentrations in the raw water during one of the irrigation applications (Appendix 4.1). The cause of the spike in NH_4-N is still uncertain, but the lack of N in the water applied during the other irrigation application suggests that the spike was temporary. Therefore, further investigation into the water quality at this site is recommended to evaluate its potential for supplying N to grapevines.

Table 3.8. The total amount of macro-elements applied *via* the in-field fractional use of winery wastewater with raw water for vineyard irrigation at the experiment plots in the Coastal region during the 2017/18 season.

Plot no. (1)	NH_4-N (kg/ha)	NO_3-N (kg/ha)	Total-N (kg/ha)	P (kg/ha)	Ca^{2+} (kg/ha)	Mg^{2+} (kg/ha)	K^+ (kg/ha)	Na^+ (kg/ha)
				Winery wastewater				
C1	120.39	0.16	120.55	5.90	7.44	3.54	123.65	17.50
C2	120.39	0.16	120.55	5.90	7.44	3.54	123.65	17.50
				Raw water				
C1	120.50	0.00	120.50	0.01	1.50	2.46	1.57	13.48
C2	120.50	0.00	120.50	0.01	1.50	2.46	1.57	13.48
				Winery wastewater + raw water				
C1	240.89	0.16	241.05	5.91	8.94	5.99	125.22	30.97
C2	240.89	0.16	241.05	5.91	8.94	5.99	125.22	30.97

(1) Refer to Table 3.2 for description of plot numbers.

Phosphorous: The winery wastewater made the highest contribution to the total amount of P applied *via* the irrigation water (Table 3.8). Previous research has shown that P fertilisation of young grapevines is best applied during soil preparation before planting, since corrective fertilisation after planting is less effective (Saayman, 1981 and references therein). Therefore, the P applied *via* the irrigation water is unlikely to be utilised by the young grapevines of the Coastal region plots.

Calcium: The Ca^{2+} applied *via* the irrigation water mainly originated from the winery wastewater (Table 3.8). No information regarding the specific Ca^{2+} requirements of newly established grapevines could be found in the literature. However, for full-bearing grapevines, Saayman (1981) reported an annual requirement of 2 kg/ha per tonne of grapes produced. Based on this norm, the fractional use of winery wastewater with raw water would supply the young grapevines with adequate amounts of Ca^{2+}.

Magnesium: The amount of Mg^{2+} applied *via* the winery wastewater and raw water was comparable during the 2017/18 season (Table 3.8). This was due to the inherent Mg^{2+}

concentration of the raw water (Appendix 4.1) which is also used in the winery for cleaning operations. Similar to Ca^{2+}, no information pertaining to the Mg^{2+} requirements of young grapevines could be found in the literature. Saayman (1981) recommended an annual application of 0.7 kg/ha Mg^{2+} per tonne of grapes produced by full-bearing grapevines. Based on this recommendation, the Mg^{2+} applied *via* the irrigation water was sufficient to meet grapevine requirements.

Potassium: The K^{+} applied *via* the irrigation water mainly originated from the winery wastewater as the raw water contained negligible amounts of K^{+} (Table 3.8). Similar to P, K^{+} fertilisation for young grapevines is best applied during soil preparation as corrective application after planting is less effective (Saayman, 1981 and references therein). However, the high amounts of K^{+} applied *via* the winery wastewater may become problematic when the grapevines come into full production. According to Conradie (1994), full-bearing grapevines require an annual K^{+} application of 3 kg/ha per tonne of grapes produced. Assuming that the grapevines will produce 10 t/ha, the applied K^{+} will still be in excess of grapevine requirements.

Sodium: The winery wastewater and raw water contributed similar amounts of Na^{+} to the irrigation water during the 2017/18 season (Table 3.8). This was probably due to the inherent Na^{+} concentrations of the raw water (Appendix 4.1) which is used in the winery during cleaning operations. Currently there are no threshold values for grapevines concerning the amount of Na^{+} applied per hectare, however excessive Na^{+} uptake by grapevines may reduce vegetative growth, yield and suppress Ca^{2+} uptake (Myburgh & Howell, 2104c).

Trace elements: The amounts of B^{3+}, Mn^{2+}, Cu^{2+} and Zn^{2+} applied *via* the irrigation water was negligible during the 2017/18 season (Table 3.9). The presence of relatively high Fe^{2+} concentrations in the winery wastewater (Appendix 4.2) resulted in a total application of *ca*. 3 kg Fe^{2+} per hectare at each of the experiment plots in the Coastal region (Table 3.9). Conradie (1994) reported that grapevines have relatively low Fe^{2+} requirements. Therefore, the amounts of Fe^{2+} applied *via* the irrigation water at these experiment plots were likely sufficient to sustain grapevine growth.

Table 3.9. The total amount trace elements and anions applied *via* the in-field fractional use of winery wastewater with raw water for vineyard irrigation at the experiment plots in the Coastal region during the 2017/18 season.

Plot no. (1)	B^{3+} (kg/ha)	Mn^{2+} (kg/ha)	Cu^{2+} (kg/ha)	Zn^{2+} (kg/ha)	Fe^{2+} (kg/ha)	Cl^- (kg/ha)	HCO_3^- (kg/ha)	SO_4^{2-} (kg/ha)	F^- (kg/ha)
					Winery wastewater				
C1	0.06	0.06	0.01	0.19	2.62	29.00	0.00	6.01	0.04
C2	0.06	0.06	0.01	0.19	2.62	29.00	0.00	6.01	0.04
					Raw water				
C1	0.00	0.03	0.00	0.01	0.36	26.60	24.66	5.65	0.10
C2	0.00	0.03	0.00	0.01	0.36	26.60	24.66	5.65	0.10
					Winery wastewater + raw water				
C1	0.06	0.09	0.01	0.20	2.99	55.60	24.66	11.66	0.14
C2	0.06	0.09	0.01	0.20	2.99	55.60	24.66	11.66	0.14

(1) Refer to Table 3.2 for description of plot numbers.

Anions: Comparable amounts of Cl^- were applied *via* the winery wastewater and raw water during the 2017/18 season (Table 3.9). It should be noted that the raw water contained relatively high Cl^- concentrations (Appendix 4.2). Similar to Na^+, there are no threshold values for grapevine Cl^- requirements, but it is known that excessive Cl^- may negatively affect vegetative growth and yield (Myburgh & Howell, 2104c and references therein). The amount of HCO_3^- applied *via* the irrigation water was solely supplied by the raw water (Table 3.9). The application of large amounts of HCO_3^- to soils may result in the precipitation of insoluble Ca- and Mg-carbonates when the soil dries out (Van Zyl, 1981). This can increase the relative amount of Na^+ in the soil solution and thereby indirectly increase the SAR, which may negatively impact soil physical properties (Van Zyl, 1981; McCarthy *et al*., 1988; ANZECC, 2000; Myburgh, 2018). Relatively equal amounts of SO_4^{2-} was applied *via* the two irrigation water types (Table 3.9). Although more F^- was applied *via* the raw water compared to the winery wastewater, the total amount of F^- applied through the irrigation water was negligible (Table 3.9).

3.3.2.3.2. Breede River region

Nitrogen: No N was applied *via* the irrigation water at either of the two experiment plots in the Breede River region (Table 3.10). As a result, the irrigation water could not contribute to the N requirements of the grapevines in this region and additional (conventional) N fertilisation would have to be applied.

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Table 3.10. The total amount of macro-elements applied *via* the in-field fractional use of winery wastewater with raw water for vineyard irrigation at the experiment plots in the Breede River region during the 2017/18 season.

Plot no. (1)	NH_4-N (kg/ha)	NO_3-N (kg/ha)	Total-N (kg/ha)	P (kg/ha)	Ca^{2+} (kg/ha)	Mg^{2+} (kg/ha)	K^+ (kg/ha)	Na^+ (kg/ha)
				Winery wastewater				
BR1	0.00	0.00	0.00	2.54	27.79	9.40	198.71	24.48
BR2	0.00	0.00	0.00	2.58	29.02	9.75	200.44	25.07
				Raw water				
BR1	0.00	0.00	0.00	0.01	2.50	3.31	0.69	15.04
BR2	0.00	0.00	0.00	0.01	2.44	3.22	0.68	14.58
				Winery wastewater + raw water				
BR1	0.00	0.00	0.00	2.55	30.30	12.71	199.40	39.52
BR2	0.00	0.00	0.00	2.59	31.46	12.98	201.12	39.65

(1) Refer to Table 3.2 for description of plot numbers.

Phosphorous: Comparable amounts of P was applied at both of the experiment plots in this region during the 2017/18 season (Table 3.10). The applied P originated almost entirely from the winery wastewater. According to Conradie (1994), full-bearing grapevines annually require 0.7 kg P per tonne of grapes produced. Therefore, the amount of P applied *via* the in-field fractional use of winery wastewater with raw water would not supply adequate amounts of P to ripen yields of 10 t/ha under the prevailing conditions.

Calcium: The amount of Ca^{2+} applied at the BR1 plot was *ca*. 1 kg/ha less than the BR2 plot (Table 3.10). This was a result of slightly more winery wastewater applied at the latter plot (Table 3.7). Similar to P, the majority of the Ca^{2+} applied *via* the irrigation water was supplied by the winery wastewater. Full-bearing grapevines require *ca*. 2 kg Ca^{2+} to produce one tonne of grapes (Conradie, 1981b). Based on this norm, the amount of Ca^{2+} applied *via* the in-field fractional use of winery wastewater with raw water would supply more than adequate amounts of Ca^{2+} to ripen yields of 10 t/ha under the prevailing conditions. However, since grapevine nutrient requirements are relatively low during and after the harvest period (Conradie, 1981b), the supplied Ca^{2+} may only be beneficial if it is not leached from the root zone during winter (Howell, 2016) or precipitated in insoluble forms.

Magnesium: Comparable amounts of Mg^{2+} was applied *via* the irrigation water at the two experiment plots in the Breede River region (Table 3.10). Although the winery wastewater supplied the majority of Mg^{2+}, the raw water also contributed considerably to the total amount of Mg^{2+} applied. Grapevines require an annual amount of 0.7 kg Mg^{2+} to ripen one tonne of fruit (Conradie, 1981b). Therefore, if a yield of 10 t/ha is assumed, the irrigation water applied at these experiment plots would supply the grapevines with more than adequate amounts of Mg^{2+} under the prevailing conditions.

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Potassium: The winery wastewater contributed the majority of K^+ to the total amounts applied at the experiment plots in the Breede River region (Table 3.10). The BR2 plot received slightly more K^+ *via* the irrigation water due to the greater volume of winery wastewater applied at this particular plot (Table 3.7). According to Conradie (1994), full-bearing grapevines have an annual K^+ requirement of 3 kg per tonne of fruit produced. Based on this recommendation, the amounts of K^+ supplied *via* the irrigation water was more than adequate to ripen a yield of 10 t/ha. However, *ca.* 170 kg/ha K^+ was applied in excess of grapevine requirements at each of the two experiment plots. The over-application of K^+ may have several impacts on fruit and wine quality. An increase in K^+ supply to grapevines can increase fruit pH (Ruhl, 1989). This may be the result of tartaric acid degradation in the presence of high K^+ levels (Mpelasoka *et al*., 2003). The increased fruit pH may ultimately lead to increased pH in grape juice, must and wines (Saayman, 1981; Mpelasoka *et al*., 2003). As a result, musts and wines may become unstable and the degree of ionisation of anthocyanins may decrease (Mpelasoka *et al*., 2003). In addition, increased juice pH can result in the production of wines with reduced colour stability and poor taste (Kodur, 2011). Furthermore, excessive K^+ can reduce Ca^{2+} and Mg^{2+} uptake by grapevines due to an antagonistic interaction between K^+ and these elements (Morris & Cawthon, 1982; Myburgh & Howell, 2014c; Howell, 2016).

Sodium: A total of *ca*. 40 kg/ha Na^+ was applied *via* the irrigation water at each of the experiment plots in the Breede River region during the 2017/18 season (Table 3.10). Although the winery wastewater supplied the majority of Na^+, the raw water made a considerable contribution to the total amounts of Na^+ applied. The Na^+ applied at these particular plots were lower than amounts ranging between 65 kg/ha and 122 kg/ha previously reported for winery wastewater diluted to a COD value of 3 000 mg/ℓ (Howell, 2016).

Trace elements: The amounts of B^{3+}, Mn^{2+} and Cu^{2+} applied *via* the irrigation water was primarily supplied by the winery wastewater (Table 3.11). Furthermore, the amounts applied were less than one kg per hectare and was therefore considered to be negligible. The high levels of Zn^{2+} present in the winery wastewater (Appendix 4.4) resulted in the application of considerable amounts of Zn^{2+} to the soil (Table 3.11). Furthermore, both types of irrigation water contributed to the amount of Fe^{2+} applied (Table 3.11). Since grapevines have a relatively low Fe^{2+} requirement (Conradie, 1994), the amounts applied *via* the irrigation water at the plots in the Breede River region appears to be adequate for sustainable grapevine growth.

Table 3.11. The total amount of trace elements and anions applied *via* the in-field fractional use of winery wastewater with raw water for vineyard irrigation at experiment plots in the Breede River region during the 2017/18 season.

Plot no. (1)	B^{3+} (kg/ha)	Mn^{2+} (kg/ha)	Cu^{2+} (kg/ha)	Zn^{2+} (kg/ha)	Fe^{2+} (kg/ha)	Cl^- (kg/ha)	HCO_3^- (kg/ha)	SO_4^{2-} (kg/ha)	F^- (kg/ha)
Winery wastewater									
BR1	0.27	0.20	0.18	4.12	1.94	46.29	929.87	35.14	0.04
BR2	0.28	0.21	0.18	4.31	2.01	47.80	921.51	35.93	0.04
Raw water									
BR1	0.00	0.00	0.01	0.02	0.29	36.50	18.51	6.90	0.08
BR2	0.00	0.00	0.01	0.01	0.30	35.82	18.45	6.76	0.07
Winery wastewater + raw water									
BR1	0.27	0.20	0.19	4.13	2.24	82.78	948.38	42.03	0.12
BR2	0.28	0.21	0.19	4.33	2.31	83.62	939.97	42.70	0.11

(1) Refer to Table 3.2 for description of plot numbers.

Anions: The amounts of Cl^- applied were comparable for both irrigation water types (Table 3.11). Furthermore, the total amounts of Cl^- applied *via* the irrigation water was lower than the amount of 87 kg/ha applied through irrigation with winery wastewater diluted to a COD concentration of 3 000 mg/ℓ (Howell, 2016). Although both irrigation water types contributed to the amount of HCO_3^- applied, the winery wastewater supplied the majority (Table 3.11). The total amounts of HCO_3^- applied at the experiment plots in the Breede River region neared one tonne per hectare during the 2017/18 season. The amounts of SO_4^{2-} applied *via* the irrigation water was also largely due to concentrations present in the winery wastewater (Appendix 4.4). Similar to the plots in the Coastal region, negligible amounts of F^- was applied *via* the irrigation water (Table 3.11).

3.3.2.3.3. Lower Olifants River region

Nitrogen: As in the case of the Breede River region, neither of the two experiment plots in the Lower Olifants River region received NO_3-N *via* the irrigation waters (Table 3.12). Therefore, the total-N applied was solely NH_4-N. Although the two plots received comparable total amounts of N, at the LOR1 plot it originated mainly from the winery wastewater, whereas the winery wastewater and raw water at the LOR2 plot contributed similar amounts of N (Table 3.12). Based on an annual N requirement of 4 kg per tonne of grapes produced (Saayman, 1981), the amount of N applied *via* the irrigation water at these plots would be adequate to ripen a yield of 10 t/ha under the prevailing conditions.

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Table 3.12. The total amount of macro-elements applied *via* the in-field fractional use of winery wastewater with raw water for vineyard irrigation at the experiment plots in the Lower Olifants River region during the 2017/18 season.

Plot no. [(1)]	NH_4-N (kg/ha)	NO_3-N (kg/ha)	Total-N (kg/ha)	P (kg/ha)	Ca^{2+} (kg/ha)	Mg^{2+} (kg/ha)	K^+ (kg/ha)	Na^+ (kg/ha)
				Winery wastewater				
LOR1	40.00	0.00	40.00	30.06	682.50	64.58	1128.78	204.34
LOR2	26.47	0.00	26.47	42.43	705.19	83.46	1186.27	706.74
				Raw water				
LOR1	0.25	0.00	0.25	0.07	13.91	13.48	9.27	70.75
LOR2	20.15	0.00	20.15	0.07	12.64	15.68	7.34	72.09
				Winery wastewater + raw water				
LOR1	40.25	0.00	40.25	30.13	696.41	78.06	1138.05	275.09
LOR2	46.62	0.00	46.62	42.50	717.83	99.14	1193.61	778.83

(1) Refer to Table 3.2 for description of plot numbers.

Phosphorous: Both experiment plots in this region received considerable amounts of P which was supplied primarily *via* the winery wastewater (Table 3.12). This was in contrast to what was observed at the experiment plots in the Breede River region (Table 3.10). Full-bearing grapevines annually require 0.7 kg P per tonne of fruit produced (Conradie, 1981b). Based on this norm, the experiment grapevines in the Lower Olifants River region received more than adequate amounts of P *via* the irrigation water to ripen yields of 10 t/ha or more.

Calcium: The amounts of Ca^{2+} applied *via* the irrigation water at the plots in the Lower Olifants River region (Table 3.12) were appreciably more than the amounts applied in the Coastal and Breede River regions (Tables 3.8 & 3.10). The high amounts of Ca^{2+} supplied by the winery wastewaters in this region was likely the result of wineries adding $CaCO_3$ to the wastewaters for pH adjustment. According to guidelines presented by Conradie (1981b), full-bearing grapevines have an annual requirement of 2 kg Ca^{2+} per tonne of grapes produced. Therefore, the irrigation water applied at these plots during the 2017/18 season applied more than adequate amounts of Ca^{2+} to ripen yields of 10 t/ha or more under the prevailing conditions. Similar results were observed for the experiment plots in the Coastal and Breede River regions (Tables 3.8 & 3.10).

Magnesium: The irrigation waters at both the experiment plots in this particular region supplied considerable amounts of Mg^{2+} (Table 3.12). Furthermore, the amounts of Mg^{2+} applied *via* the irrigation water were more than adequate to supply the grapevines with the required 7 kg/ha Mg^{2+} to produce 10 tonnes of fruit (Conradie, 1981b). Far greater amounts of Mg^{2+} was applied *via* the irrigation waters used in the Lower Olifants River region, compared to those used in the Coastal and Breede River regions (Tables 3.8 & 3.10).

Potassium: The amounts of K^+ applied *via* the irrigation water at each of the experiment plots in the Lower Olifants River region amounted to more than one tonne per ha (Table 3.12). This was by far in excess of the annual requirement of 3 kg K^+ required per tonne of grapes produced

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(Conradie, 1981b). This could have detrimental effects on wine quality and may hinder the uptake of Ca^{2+} and Mg^{2+} as discussed in section 3.3.2.3.2.

Sodium: The amount of Na^{+} applied *via* the winery wastewater from the LOR2 plot was considerably higher compared to the LOR1 plot (Table 3.12). The total amounts of Na^{+} applied *via* the irrigation water at these two plots were also substantially higher compared to the experiment plots in the Coastal and Breede River regions (Tables 3.8 & 3.10). Although there are no guidelines for Na^{+} requirements of grapevines, excessive application of Na^{+} may reduce vegetative growth and yield (Myburgh & Howell, 2014c and references therein). It can also suppress the uptake of Ca^{2+} and Mg^{2+} which may result in grapevine nutrient deficiencies (McCarthy & Downton, 1981). Furthermore, excessive Na^{+} application to the soil will likely increase the soil's exchangeable sodium percentage (ESP) which may subsequently reduce soil IR and *K* (Halliwell *et al*., 2001 and references therein).

Trace elements: The winery wastewater used for irrigation at the experiment plots in the Lower Olifants River region supplied nearly 2 kg/ha B^{3+} during the 2017/18 season (Table 3.13). Since grapevines have relatively low B^{3+} requirements (Conradie, 1994), the amounts applied *via* the irrigation water at these plots would likely be adequate to sustain grapevine growth. Similar to the Coastal and Breede River regions, the amounts of Mn^{2+}, Cu^{2+} and Zn^{2+} applied *via* the irrigation water at the experiment plots in this particular region were less than one kg per hectare and was considered to be negligible (Table 3.13). In contrast, both of the experiment plots received appreciable amounts of Fe^{2+} *via* the irrigation water (Table 3.13).

Table 3.13. The total amount of trace elements and anions applied *via* the in-field fractional use of winery wastewater with raw water for vineyard irrigation at the experiment plots in the Lower Olifants River region during the 2017/18 season.

Plot no. (1)	B^{3+} (kg/ha)	Mn^{2+} (kg/ha)	Cu^{2+} (kg/ha)	Zn^{2+} (kg/ha)	Fe^{2+} (kg/ha)	Cl^{-} (kg/ha)	HCO_3^{-} (kg/ha)	SO_4^{2-} (kg/ha)	F^{-} (kg/ha)
					Winery wastewater				
LOR1	1.71	0.45	0.06	0.17	4.07	352.68	3248.8	1827.8	0.72
LOR2	1.97	0.80	0.02	0.24	6.49	493.70	3453.5	2695.1	0.24
Total									
					Raw water				
LOR1	0.07	0.00	0.01	0.03	0.27	160.06	50.67	38.14	0.25
LOR2	0.00	0.04	0.02	0.02	0.29	166.16	71.92	39.29	0.20
Total									
					Winery wastewater + raw water				
LOR1	1.78	0.45	0.08	0.20	4.34	512.74	3299.42	1865.9	0.97
LOR2	1.97	0.84	0.04	0.26	6.78	659.87	3525.37	2734.4	0.44

(1) Refer to Table 3.2 for description of plot numbers.

Anions: Both experiment plots in the Lower Olifants River region received considerable amounts of Cl^{-} *via* the irrigation waters (Table 3.13). Currently there are no guidelines for grapevines in

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terms of the amount of Cl^- applied per hectare, however it is known that excessive Cl^- application may restrict vegetative growth and reduce yield (Myburgh & Howell, 2014c and references therein). The amounts of HCO_3^- applied *via* the irrigation water was above 3 t/ha at both of the experiment plots in this region (Table 3.13). This is of great concern as the application of both high amounts of Ca^{2+} (Table 3.12) and high amounts of HCO_3^- may lead to the precipitation of insoluble Ca-carbonates in the soil (as discussed previously in section 3.3.2.3.2). Furthermore, the amount of SO_4^{2-} applied at these plots was over 1.5 t/ha in the case of the LOR1 plot and over 2.5 t/ha in the case of the LOR2 plot (Table 3.13). The high amounts of SO_4^{2-} applied *via* the irrigation water at these plots were a direct result of high levels of SO_4^{2-} in the winery wastewater (Appendices 4.6 & 4.8). The amount of F^- applied *via* the irrigation water at the LOR1 plot was close to 1 kg/ha, whereas the amount applied at the LOR2 plot was *ca*. half of that (Table 3.13).

3.3.3. Soil chemical properties

3.3.3.1. $pH_{(KCl)}$

3.3.3.1.1. Coastal region

Irrigation using in-field fractionally applied winery wastewater with raw water did not have a substantial effect on soil $pH_{(KCl)}$ at the C1 plot (Fig. 3.8A). This was expected since only two irrigations were applied at this site during the 2017/18 season. However, a decrease in soil $pH_{(KCl)}$ was evident at this plot following the winter rainfall period. Since the lower pH could not be ascribed to the leaching of salts from the soil (Fig. 3.11A), the reason for the decrease is still uncertain. A $pH_{(KCl)}$ reduction of *ca*. 1 unit was observed at 30-90 cm soil depth of the C2 plot following the irrigation applications (Fig. 3.8B). Given the clay content of the soil, *i.e.* a higher buffer capacity, and that only two irrigations were applied at this plot, it is unlikely that the winery wastewater would have decreased the soil pH to such an extent. Similar to the C1 plot, the reduction in $pH_{(KCl)}$ could also not be explained by the leaching of salts from the profile (Fig. 3.11B). Further investigations into the changes in soil $pH_{(KCl)}$ at these experiment plots are required.

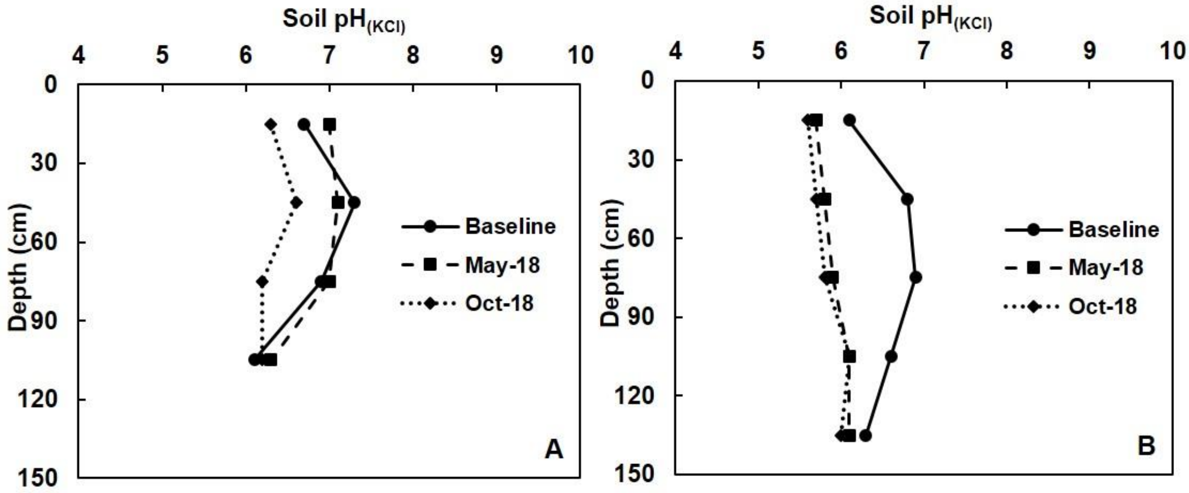

Figure 3.8. Variation in soil pH$_{(KCl)}$ before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.1.2. Breede River region

After two irrigations were applied, the pH$_{(KCl)}$ of the BR1 plot remained at levels comparable to the baseline (Fig. 3.9A). Similarly, the pH$_{(KCl)}$ of the BR2 experiment plot was also unaffected by the irrigation water (Fig. 3.9B). However, a decrease in pH$_{(KCl)}$ below 60 cm soil depth was evident at this plot after the winter rainfall period. This trend may be explained by the leaching of Ca^{2+} from the soil and its replacement with acidic ions such as aluminium (Al^{3+}) and hydrogen (H^{+}) (Foth, 1990). However, the decrease in pH is not a concern since the pH$_{(KCl)}$ remained within the range of 5.0 to 7.5 required to sustain optimal grapevine growth (Saayman, 1981).

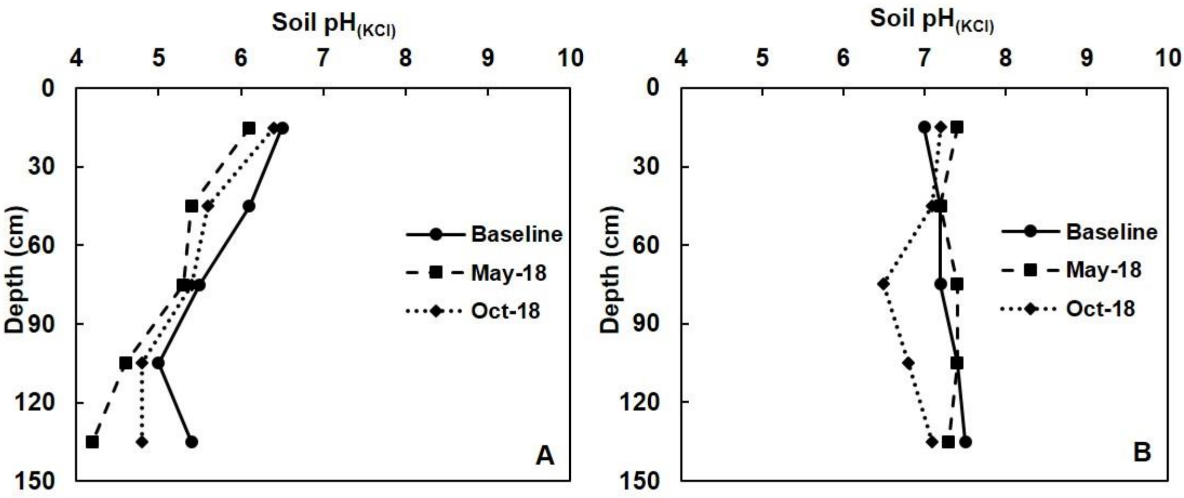

Figure 3.9. Variation in soil pH$_{(KCl)}$ before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

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3.3.3.1.3. Lower Olifants River region

The soil $pH_{(KCl)}$ of both the LOR1 and LOR2 experiment plots increased due to irrigation *via* in-field fractionally applied winery wastewater with raw water (Figs. 3.10A & 3.10B). It was previously found that soil $pH_{(KCl)}$ increased when irrigated with acidic winery wastewater, regardless of the soil type (Mulidizi *et al*., 2015). They attributed the increase to the hydrolysis and decarboxylation of organic/bicarbonate ions which were added to the soil *via* winery wastewater irrigation. Although the organic acid content of the winery wastewater was not determined during this study, the winery wastewater applied at both the LOR1 and LOR2 plots had high COD and HCO_3^- concentrations (Appendices 4.6 & 4.8). Potassium and Na^+ cations largely counter the charge on organic and HCO_3^- anions (Mulidizi *et al*., 2015). Upon application to soils, hydrolysis and decarboxylation reactions take place which increases the pH (Li *et al*., 2008). Numerous other studies have also reported an increase in soil pH as a result of winery wastewater irrigation (Gray, 2012; Mosse *et al*., 2012; Shilpi *et al*., 2018). After the winter rainfall period, the $pH_{(KCl)}$ of the subsoil at the LOR1 plot decreased to levels similar to the baseline (Fig. 3.10A). This was probably due to the leaching of HCO_3^- ions during rainfall. The lower $pH_{(KCl)}$ of the topsoil measured in October 2018 could have been caused by the acidic winery wastewater (pH 4.6) which was applied two weeks prior to the soil sampling date during the final irrigation of the season (Appendix 4.5). Furthermore, the $pH_{(KCl)}$ of the LOR2 plot remained unchanged after the winter rainfall period (Fig. 3.10B). It should be noted that the soil pH at this plot was above the range of 5.0 to 7.5 recommended to sustain optimal grapevine growth (Saayman, 1981).

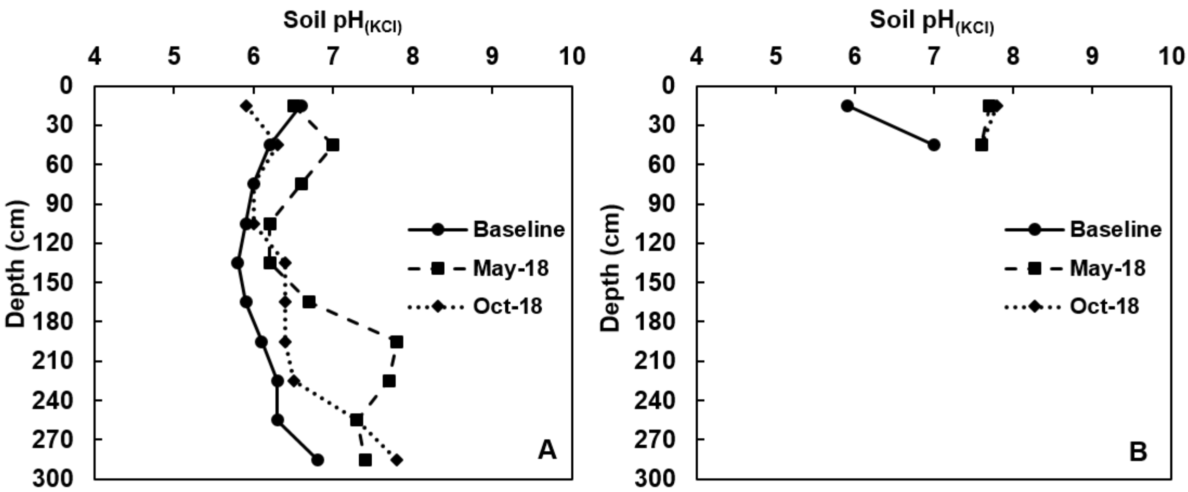

Figure 3.10. Variation in soil $pH_{(KCl)}$ before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.2. Electrical conductivity

3.3.3.2.1. Coastal region

The in-field fractional application of winery wastewater with raw water did not have a pronounced effect on the EC_e of either of the two experiment plots in the Coastal region (Figs. 3.11A & 3.11B). This result was expected since the EC_w of both irrigation waters were relatively low and only two irrigations were applied at these plots during the 2017/18 season (Appendix 4.1). Similarly, when sand, loamy sand and sandy loam soils were compared in a pot experiment, winery wastewater irrigation did not affect $EC_{(1:5)}$, regardless of soil texture (Laurenson, 2010).

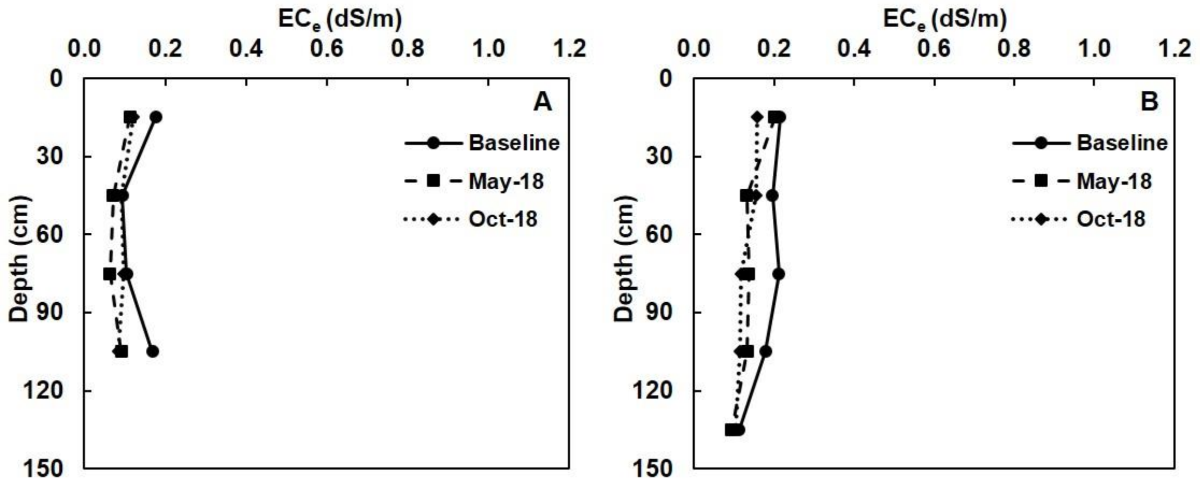

Figure 3.11. Variation in soil electrical conductivity (EC_e) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.2.2. Breede River region

The topsoil EC_e of the BR1 experiment plot decreased from 0.5 dS/m before irrigation commenced to 0.2 dS/m after two irrigation applications (Fig. 3.12A). However, the salinity in the 30-90 cm soil layer increased by 0.1 dS/m compared to the baseline. This could be the result of the application of high salinity winery wastewater (5.3 dS/m), followed by low salinity raw water (0.3 dS/m) during the second irrigation (Appendix 4.3). The lower salinity raw water would have leached the excessive salts applied *via* the winery wastewater to deeper soil layers. In contrast, the topsoil EC_e of the BR2 plot marginally increased relative to the baseline but was lower in all the other soil layers after irrigation application (Fig. 3.12B). The higher clay content in the topsoil of the BR2 plot probably resulted in stronger adsorption of salts applied *via* the winery wastewater, as well as less leaching to subsoil layers compared to the lighter textured BR1 plot. Following the winter rainfall period, salinity levels in the 30-90 cm soil layer of the BR1 plot decreased to values below the baseline and post-treatment values (Fig. 3.12A). Results indicate that the salts that were previously present in this particular soil layer leached to deeper layers during the winter rainfall period. At the BR2 plot the EC_e of the subsoil increased relative to the post-treatment values (Fig. 3.12B). It is possible that salts which were limited to the topsoil after the irrigation applications were leached to deeper soil layers during the rainy season. These results indicate that although the rainfall in this region was able to leach the applied salts in both plots, the lighter textured BR1 plot retained far less salts compared to the BR2 plot in the 0-90 cm soil layer where the majority of grapevine roots occur. Previous research has indicated that no threshold value for EC_e exists, but a progressive decrease in yield can be expected above 0.75 dS/m at a rate of 3% per 0.1 dS/m (Moolman *et al*., 1999). However, the EC_e levels at both experiment plots in this region were below 0.75 dS/m, therefore no reductions in yields were expected.

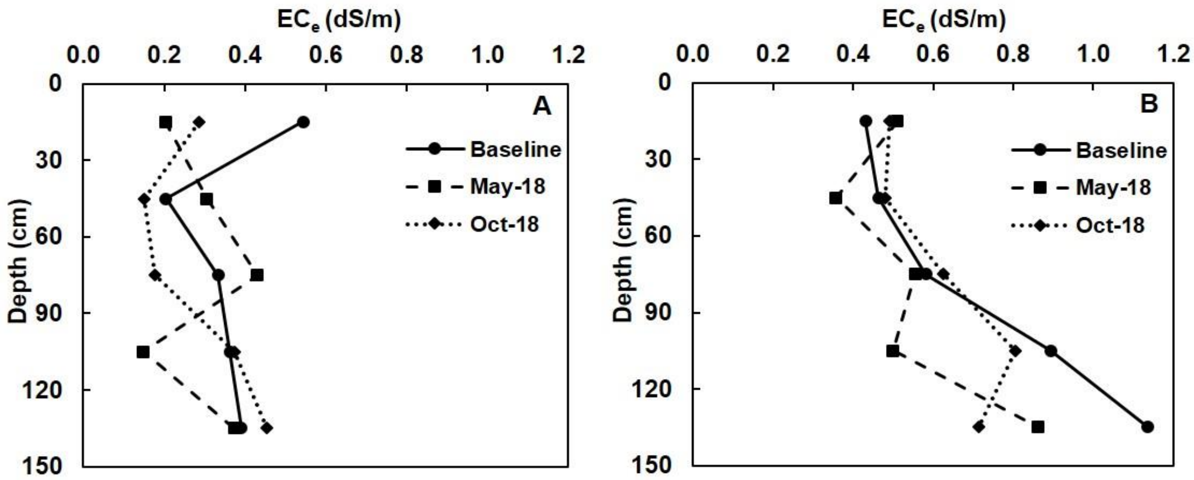

Figure 3.12. Variation in soil electrical conductivity (EC_e) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.2.3. Lower Olifants River region

Despite the application of relatively saline winery wastewater (mean EC_w of 3.8 dS/m), the EC_e of the deep sandy LOR1 plot was not affected by the irrigation water and remained at levels similar to the baseline after the winter rainfall period (Fig. 3.13A). This was most probably due to the low clay content of this plot (Appendix 5.3), and the inability of soil particles to adsorb significant amounts of salts. At the LOR2 experiment plot, the topsoil EC_e increased marginally after one season of irrigation using fractionally applied winery wastewater with raw water (Fig. 3.13B). After the rainfall season, the salinity of the topsoil decreased, and salts were leached to the 30-60 cm soil layer (Fig. 3.13B). However, the levels were similar to the baseline and a net increase in EC_e was not observed. Similar results were reported for a sandy alluvial vineyard soil in the Breede River region which was irrigated with diluted winery wastewater for four seasons (Howell, 2016). As was the case with the experiment plots in the Breede River region, the soil EC_e of the two plots in the Lower Olifants River region remained below 0.75 dS/m. Therefore, no reductions in yield were expected (Moolman *et al*., 1999).

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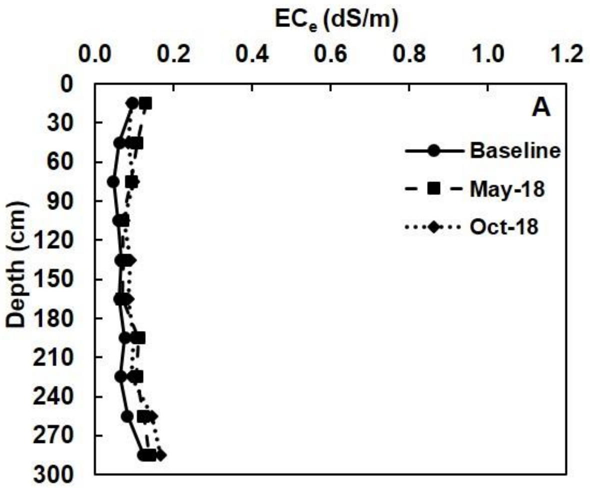

Figure 3.13. Variation in soil electrical conductivity (EC_e) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.3. Phosphorous (Bray II)

3.3.3.3.1. Coastal region

The plant-available P in the topsoil of both experiment plots in this region increased after the irrigation applications (Figs. 3.14A & 3.14B). However, since only two irrigations were applied at these plots and the total amount of P applied *via* the irrigation water at each plot was less than 6 kg/ha (Table 3.8), the response cannot be attributed to the irrigation water. The increase in P in the topsoil of these plots was probably due to the application of P fertiliser by the grower during the post-harvest period. It should be noted that the Bray II P levels at the C1 plot exceeded the norm of 25 mg/kg P recommended for vineyard soils with a clay content between 6% and 15% (Conradie, 1994). Furthermore, the P levels of the C2 plot exceeded the norm of 30 mg/kg P recommended for vineyard soils with a clay content higher than 15% (Conradie, 1994).

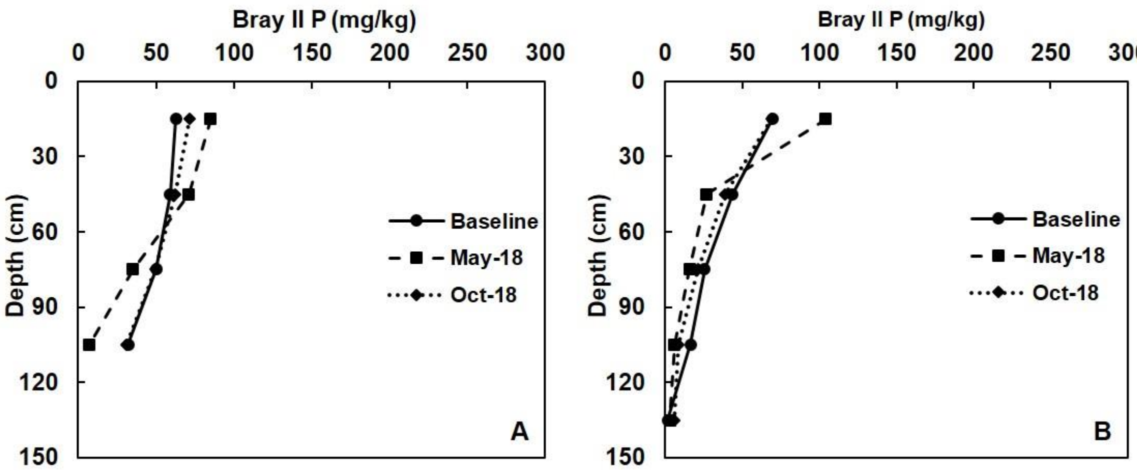

Figure 3.14. Variation in soil Bray II phosphorous (Bray II P) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.3.2. Breede River region

Compared to the baseline, plant-available P levels at the BR1 plot decreased throughout the soil profile after the irrigation applications (Fig. 3.15A). The decrease was most probably the result of grapevine uptake. In contrast, the Bray II P levels at the BR2 plot slightly increased after the irrigations were applied (Fig. 3.15B). Since only 2.6 kg/ha P was applied at this plot *via* the irrigation water (Table 3.10), the increase cannot be ascribed to the irrigation water. Similar to the Coastal region plots, an increase in Bray II P was evident at these plots after the winter rainfall period. This was probably also a result of P fertiliser application by the grower during the post-harvest period. Furthermore, the Bray II P levels at both plots exceeded the recommended norms for P in vineyard soils (Conradie, 1994).

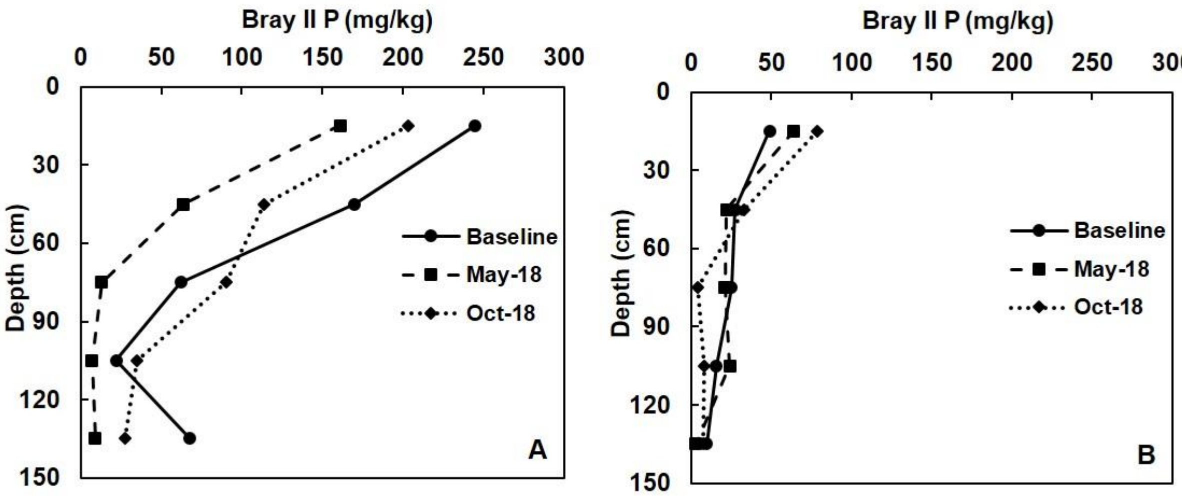

Figure 3.15. Variation in soil Bray II phosphorous (Bray II P) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.3.3. Lower Olifants River region

Despite the application of 30 kg/ha P *via* the irrigation water at the LOR1 plot (Table 3.12), the soil Bray II P content was similar to the baseline at the end of the study period (Fig. 3.16A). Results from a pot experiment in which an aeolian sand from Lutzville was irrigated with diluted winery wastewater indicated a substantial increase in available P after four simulated irrigation seasons (Mulidzi *et al*., 2016). The increase in P was attributed to a shift in $pH_{(KCl)}$ towards the optimum range of P availability. The increase was more prominent above a pH of 8. Therefore, the lack of response in the plant-available P at the LOR1 plot may be due to the $pH_{(KCl)}$ remaining below 8 and the fixation of P into unavailable forms (Fig. 3.10A). However, the Bray II P levels at this plot was only slightly below the norm of 20 mg/kg P recommended for vineyard soils with a clay content of less than 6% (Conradie, 1994). Therefore, no limitations in terms of grapevine P uptake is expected at this plot. In contrast to the LOR1 plot, the Bray II P in the topsoil of the LOR2 plot increased fourfold compared to the baseline at the end of the study period (Fig. 3.16B). This was probably due to the application of over 40 kg/ha P *via* the irrigation water (Table 3.12). Furthermore, the higher $pH_{(KCl)}$ of this plot would have increased the soluble forms of P in the soil and limited the formation of insoluble Ca-phosphates (Mulidzi *et al*., 2016). At the end of the study period, the Bray II P levels at this plot substantially exceeded the norm of 25 mg/kg P recommended for vineyard soils with a clay content of 6% to 15% (Conradie, 1994). The fact that the Bray II P increased to such an extent after only one irrigation season is concerning. It is probable that long-term irrigation using in-field fractionally applied winery wastewater with raw water may lead to excessive P accumulations at this plot.

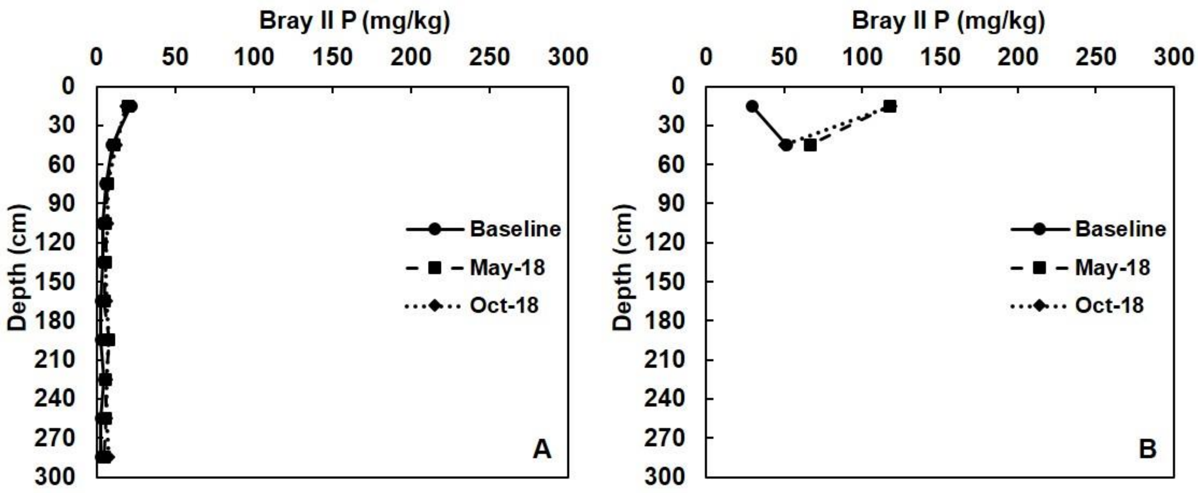

Figure 3.16. Variation in soil Bray II phosphorous (Bray II P) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.4. Potassium (Bray II)

3.3.3.4.1. Coastal region

The soil Bray II K^+ levels at both plots in this region remained comparable to the baseline after the two irrigations were applied (Figs. 3.17A & 3.17B). Therefore, the 125 kg/ha K^+ applied *via* the irrigation water at each of these plots (Table 3.8) did not seem to have a substantial effect on soil K^+ levels. Furthermore, the K^+ applied *via* the irrigation water would not have contributed to the grapevine nutrient status. A slight increase in plant-available K^+ of the 0-30 cm and 30-60 cm soil layers was observed at the C2 plot after the winter rainfall period (Fig. 3.17B). This increase was probably a result of post-harvest K-fertiliser applied by the grower. It should be noted that the soil Bray II K^+ levels at the C1 plot was slightly below the recommended norm of 70 mg/kg to 80 mg/kg K^+ for medium textured soils in the Coastal region (Conradie, 1994). In contrast, the Bray II K^+ levels at the C2 plot exceeded this norm.

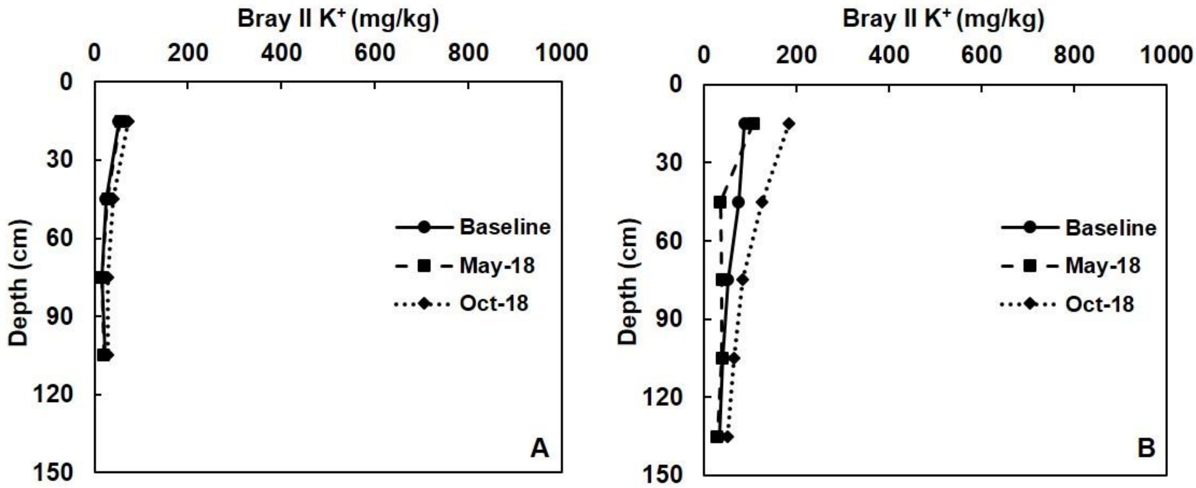

Figure 3.17. Variation in soil Bray II potassium (Bray II K^+) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.4.2. Breede River region

Despite the application of 200 kg/ha K^+ *via* the irrigation water (Table 3.10), the Bray II K^+ content at the BR1 plot remained comparable to the baseline after the irrigation applications (Fig. 3.18A). A slight increase in K^+ in the 120-150 cm soil layer may be an indication that the K^+ applied *via* the winery wastewater was leached to deeper soil layers when the raw water was applied. In contrast, the Bray II K^+ content of the BR2 plot nearly doubled after the irrigation water was applied (Fig. 3.18B). The higher accumulation of K^+ in the topsoil of this plot may be a result of the soils' higher clay content and its ability to adsorb more K^+ ions (Marchuk, 2016). Furthermore, the soil K^+ content of both plots in this region were above the recommended norm of 80 mg/kg to 100 mg/kg for clay loam soils in the Breede River region (Conradie, 1994). The high levels of plant-available K^+ in these soils may lead to enhanced K^+ uptake by grapevines which could increase juice pH (Kodur, 2011). This could ultimately have deleterious effects on wine quality (Somers, 1975; Mpelasoka *et al*., 2003).

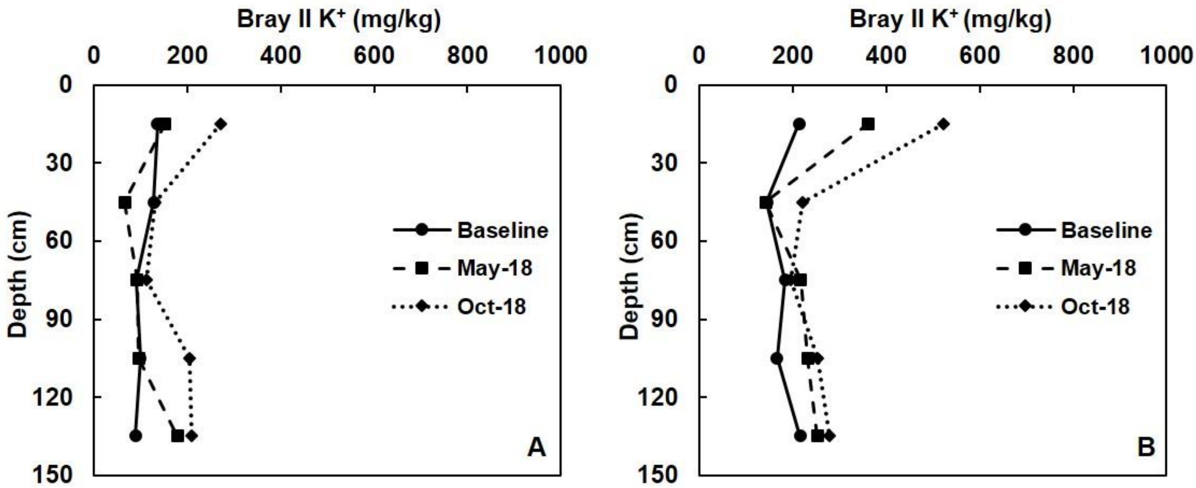

Figure 3.18. Variation in soil Bray II potassium (Bray II K^+) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.4.3. Lower Olifants River region

At the end of the study period, the application of over 1 000 kg/ha K^+ *via* the irrigation water (Table 3.12), resulted in an increase of *ca*. 100 mg/kg Bray II K^+ in the topsoil of the LOR1 plot (Fig. 3.19A). The application of 1 200 kg/ha K^+ at the LOR2 plot (Table 3.12) resulted in an almost six fold increase in the topsoil Bray II K^+ content (Fig. 3.19B). The greater accumulation of K^+ in the soils of this region was a result of higher amounts of K^+ applied *via* the irrigation water. It is probable that the lower rainfall in this region also contributed to the accumulation (Fig. 3.6). The K^+ that accumulated in the soils at these plots would probably be able to supply grapevines with sufficient K^+ during an irrigation season. However, the plant-available K^+ content at the LOR2 plot far exceeded the recommended norm of 100 mg/kg to 120 mg/kg K^+ for shallow soils on top of Dorbank which are commonly found in the Olifants River region (Conradie, 1994). Therefore, a risk in terms of excessive K^+ uptake by grapevines exist which could have negative impacts on juice and wine quality (Mpelasoka *et al*., 2003; Kodur, 2011).

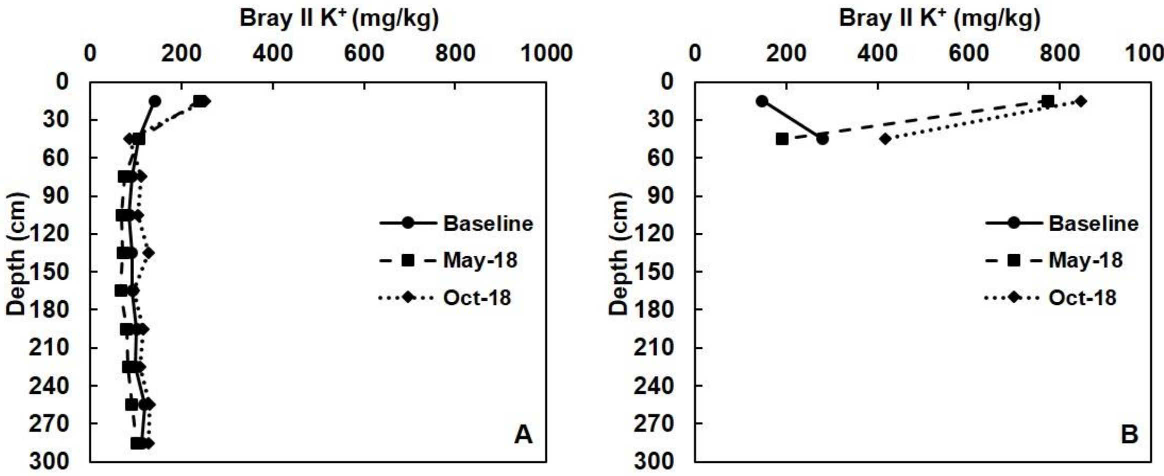

Figure 3.19. Variation in soil Bray II potassium (Bray II K^+) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5. Extractable cations

3.3.3.5.1. Calcium

3.3.3.5.1.1. Coastal region

Soil Ca^{2+} levels at the C1 experiment plot was not affected by the two irrigation applications and remained comparable to the baseline at the end of the study period (Fig. 3.20A). Although a slight decrease in Ca^{2+} occurred in the subsoil of the C2 plot after the irrigation applications, the reduction was not substantial (Fig. 3.20B). The lack of response was a result of the low Ca^{2+} concentrations in the irrigation water. Similarly, Mulidzi *et al*. (2015) reported that irrigation using diluted winery wastewater with similar Ca^{2+} concentrations to the winery wastewater applied at the Coastal region plots, did not have an effect on the soil Ca^{2+} levels of a shale soil from Stellenbosch.

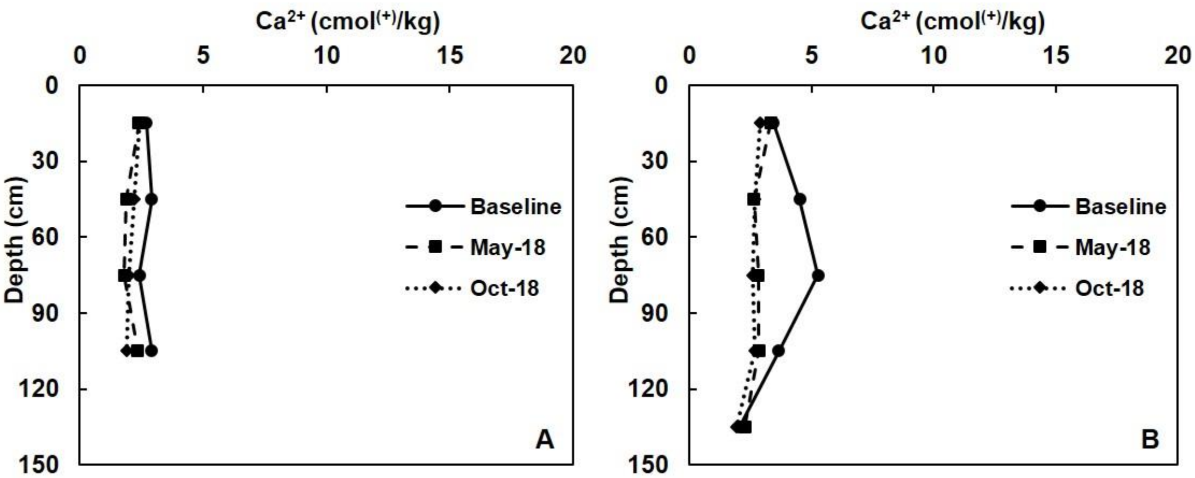

Figure 3.20. Variation in soil extractable calcium (Ca^{2+}) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

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3.3.3.5.1.2. Breede River region

A decrease in the topsoil Ca^{2+} levels were evident at the BR1 plot after the irrigation applications (Fig. 3.21A). At the BR2 plot, an abrupt decrease in Ca^{2+} levels in the 30-60 cm soil layer was accompanied by an accumulation of Ca^{2+} in the underlying soil layer following the two irrigation applications (Fig. 3.21B). After the winter rainfall period, the subsoil Ca^{2+} levels at this plot decreased even further. These results indicate that the irrigation water did not make a significant contribution towards Ca^{2+} supply for grapevines. Furthermore, the irrigation water and rainfall, along with grapevine uptake, effectively removed Ca^{2+} from the soil profile. It is possible that the high PAR irrigation water applied during the second irrigation (Appendix 4.3), facilitated the exchange of Ca^{2+} by K^+ in the soil and Ca^{2+} was leached beyond the measured depth. In a vineyard soil that was irrigated with winery wastewater for 21 years, Hirzel *et al*. (2017) attributed the loss of Ca^{2+} from the soil profile to the displacement of Ca^{2+} by Na^+ applied *via* the winery wastewater.

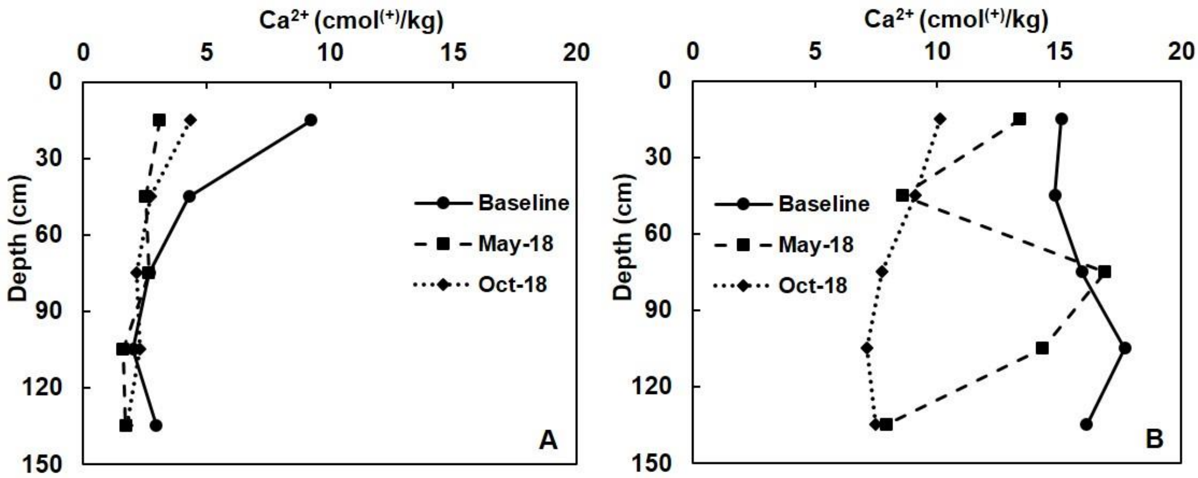

Figure 3.21. Variation in soil extractable calcium (Ca^{2+}) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.1.3. Lower Olifants River region

Irrigation using in-field fractionally applied winery wastewater with raw water did not affect the soil Ca^{2+} levels in the 0-300 cm soil profile of the LOR1 plot (Fig. 3.22A). However, a slight Ca^{2+} accumulation was observed in the 270-300 cm soil layer after the winter rainfall period in October 2018. This may be due to the leaching of Ca^{2+} ions that accumulated in the overlying soil layers prior to the rainfall period. At the LOR2 plot, the Ca^{2+} levels in the topsoil increased substantially after the majority of irrigations were applied (Fig. 3.22B). In a silty clay loam soil that was irrigated with winery wastewater for *ca*. 30 years, soil Ca^{2+} levels increased due to high amounts of Ca^{2+} supplied to the soil *via* the winery wastewater (Mosse *et al*., 2012). After the winter rainfall period, the soil Ca^{2+} levels at the LOR2 plot remained relatively unchanged (Fig. 3.22B). It can therefore be assumed that the rainfall in this region was not able to leach Ca^{2+} from the soil at this plot.

Furthermore, the accumulated Ca^{2+} in the topsoil of this plot may be able to counteract the negative impacts that the application of high SAR and PAR winery wastewater might have on soil structural stability.

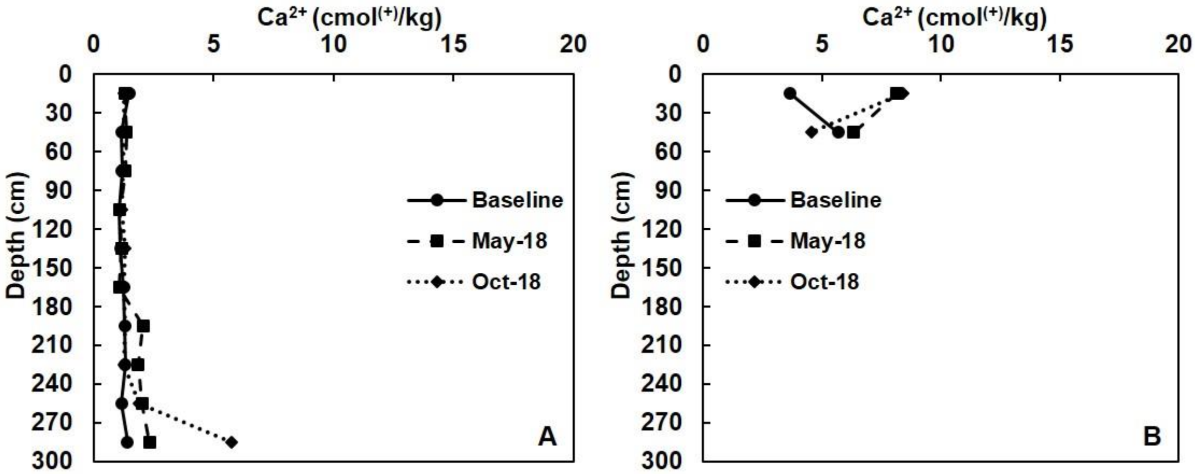

Figure 3.22. Variation in soil extractable calcium (Ca^{2+}) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.2. Magnesium

3.3.3.5.2.1. Coastal region

The soil Mg^{2+} levels of the C1 and C2 experiment plots remained relatively unchanged after the irrigation applications and the winter rainfall period (Figs. 3.23A & 3.23B). The lack of response was expected since only two irrigations were applied at these plots during the 2017/18 season and both irrigation waters had low Mg^{2+} concentrations (Appendix 4.1). Similarly, Kumar *et al.* (2006) reported that winery wastewater irrigation did not have a significant effect on the soil Mg^{2+} levels of vineyards in the Barossa and McLaren Vale regions of Australia.

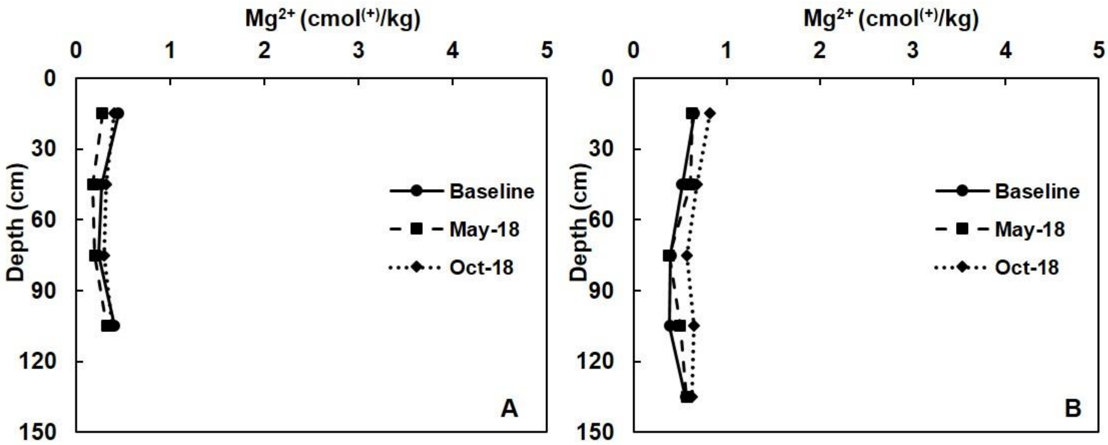

Figure 3.23. Variation in soil extractable magnesium (Mg^{2+}) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.2.2. Breede River region

Compared to the baseline values, the topsoil Mg^{2+} levels were lower after the irrigation applications at BR1 experiment plot (Fig. 3.24A). The accumulation of Mg^{2+} below 120 cm might be an indication that some Mg^{2+} leached to deeper soil layers. Grapevine uptake would also have contributed to the decrease in the upper parts of the soil profile. With the exception of a slight increase below 90 cm, the Mg^{2+} levels at the BR1 plot remained relatively unchanged after the winter rainfall period. Although a slight decrease in Mg^{2+} levels was observed at the BR2 plot after the irrigation applications, the difference was not substantial (Fig. 3.24B). An increase of *ca*. 1 $cmol^{(+)}$/kg occurred throughout the soil profile after the winter rainfall period (Fig. 3.24B). The reason for the increase after the rainy season is still uncertain.

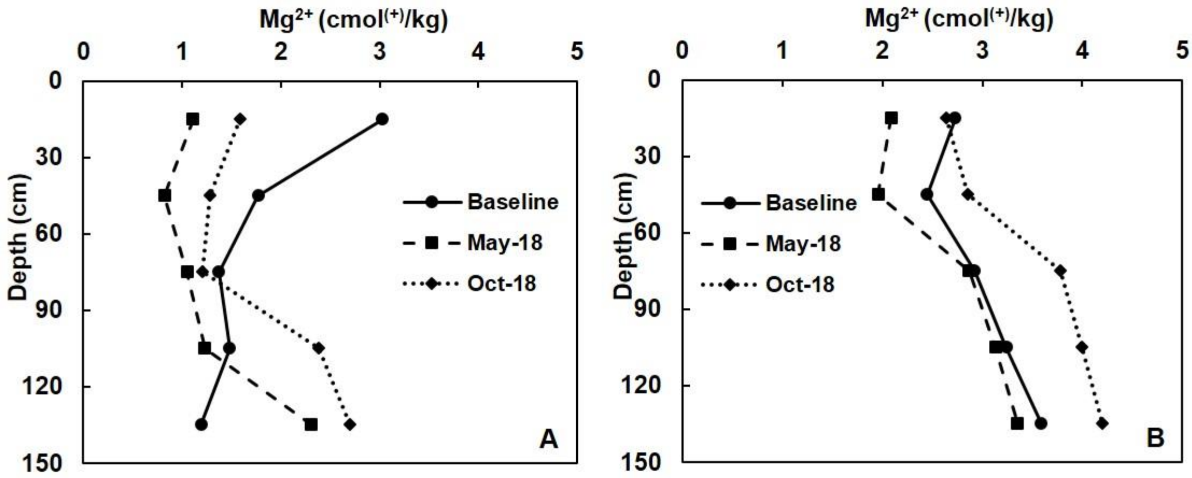

Figure 3.24. Variation in soil extractable magnesium (Mg^{2+}) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.2.3. Lower Olifants River region

Baseline measurements indicated that the soil Mg^{2+} levels at the LOR1 experiment plot gradually increased with depth (Fig. 3.25A). Although a slight decrease in soil Mg^{2+} levels was observed after the irrigation applications, the trend remained the same. The same trend occurred after the winter rainfall period, except soil Mg^{2+} levels shifted towards increasing values. At the LOR2 plot, soil Mg^{2+} levels decreased by 1 $cmol^{(+)}$/kg in the topsoil and 2 $cmol^{(+)}$/kg in the 30-60 cm soil layer after the majority of irrigations were applied (Fig. 3.25B). Hirzel *et al*. (2017) reported a similar trend for a vineyard soil in California that had been irrigated with winery wastewater for 21 years. They attributed the decrease to the displacement of Mg^{2+} by Na^+ applied *via* the winery wastewater. Following the winter rainfall period, the extractable Mg^{2+} in the subsoil increased to levels comparable to the baseline (Fig. 3.25B). Sepiolite minerals ($Mg_4Si_6O_{15}(OH)•6H_2O$) are commonly found in the soils of the Lower Olifants River region (Singer *et al*., 1995). Since the soil $pH_{(KCl)}$ at this plot was slightly alkaline at the end of the study period (Fig. 3.10B), the increase in soil Mg^{2+} cannot be ascribed to a dissolution of these minerals, given that sepiolite dissolves under acidic soil conditions (Mulders *et al*., 2018). Therefore, the reason for this increase after the rainy season is still uncertain.

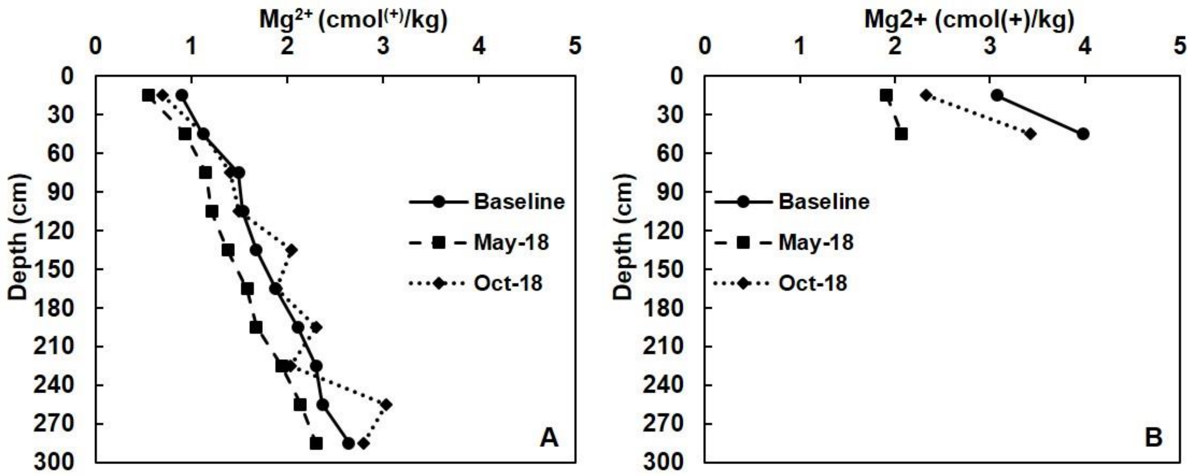

Figure 3.25. Variation in soil extractable magnesium (Mg^{2+}) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.3. Potassium

The soil extractable K^+ levels followed similar trends as was observed for Bray II K^+ (data not shown). Therefore, only the EPP´ will be discussed in the following section.

3.3.3.5.4. Extractable potassium percentage

3.3.3.5.4.1. Coastal region

Compared to the baseline, a slight increase in extractable potassium percentage (EPP´) was observed throughout the soil profile at the C1 plot at the end of the study period (Fig. 3.26A). However, the increase was not substantial. Therefore, no problems with regard to soil structural stability was expected. Similarly, the EPP´ levels at the C2 plot remained similar to the baseline throughout the soil profile after the irrigations were applied (Fig. 3.26B). However, a considerable increase in EPP´ occurred at this plot after the winter rainfall period. This was primarily the result of higher K^+ levels in the soil brought about by fertiliser applications (Fig. 3.17B). The abrupt increase is a concern, since it may adversely affect soil *K* and IR (Quirk & Schofield, 1955; Levy & Van der Watt, 1990).

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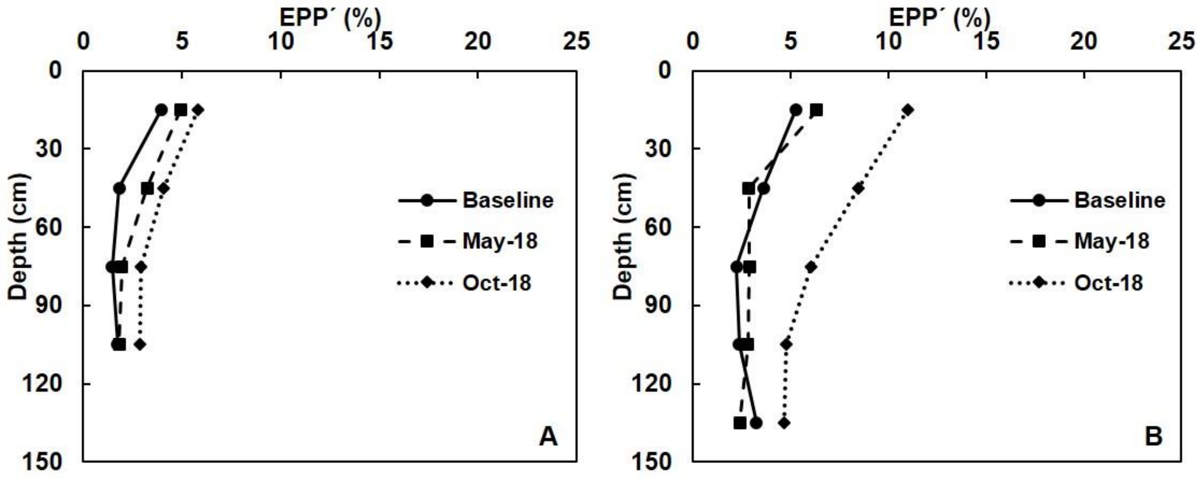

Figure 3.26. Variation in soil extractable potassium percentage (EPP´) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.4.2. Breede River region

Despite soil K^+ levels remaining unchanged after the irrigation applications at the BR1 plot (Fig. 3.18A), the EPP´ in the topsoil of this plot increased substantially compared to the baseline (Fig. 3.27A). This can probably be explained by a decrease in soil Ca^{2+}, Mg^{2+} and Na^+ after the irrigations were applied (Figs. 3.21A, 3.24A & 3.30A). The further increase in EPP´ after the winter rainfall period is more likely due to higher K^+ levels in the soil at the particular time (Fig. 3.18A). Similar trends were observed at the BR2 plot (Fig. 3.27B). The increase in EPP´ at these plots after the winter rainfall period may have adverse effects on soil structural stability (Laurenson *et al*., 2012 and references therein).

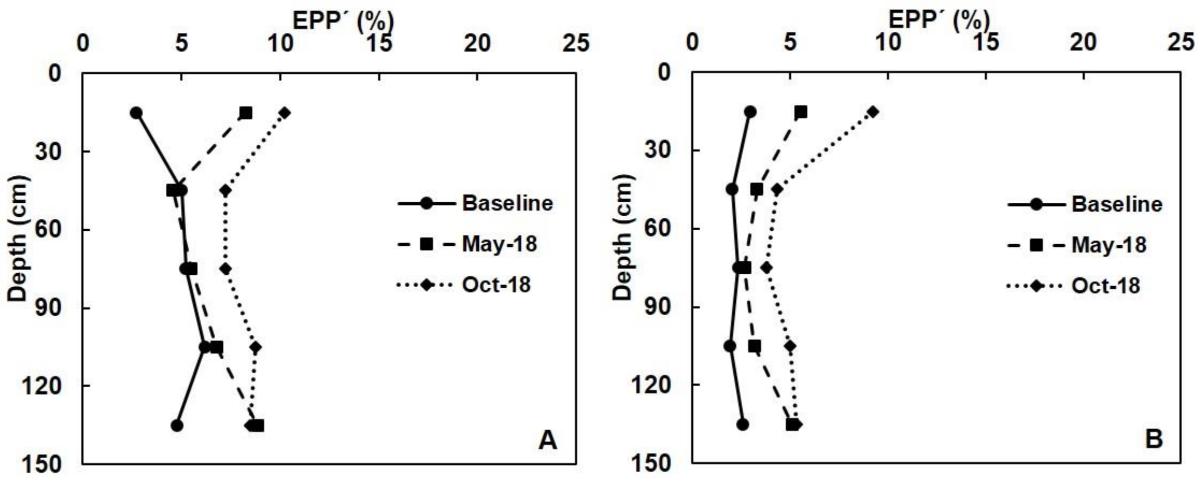

Figure 3.27. Variation in soil extractable potassium percentage (EPP´) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.4.3. Lower Olifants River region

The EPP´ in the 0-30 cm soil layer of the LOR1 plot increased by more than 10% compared to the baseline at the end of the study period (Fig. 3.28A). An increase in EPP´ to the same degree was observed at the LOR2 plot at the end of the study period (Fig. 3.28B). The increases at these plots were solely the result of appreciable amounts of K^+ that accumulated during the 2017/18 season (Figs. 3.19A & 3.19B). The substantially higher EPP´ values of the plots in this region may be explained by the presence of an unusually high proportion of mica in the clay fraction (Francis *et al*., 2007). Mica contents of soils in the Western Cape have been indirectly related to annual precipitation and a significant relationship with exchangeable potassium percentage (EPP) was observed (Bühmann *et al*., 2004). Furthermore, it is well known that illite and other mica clay minerals have an abundancy of specific binding sites for K^+ within the structural layers (Sawhney, 1972). The presence of these minerals in soils that are irrigated with K-rich winery wastewater may therefore increase the chances for soil structural degradation and ultimately decrease soil *K* (Chen *et al*., 1983).

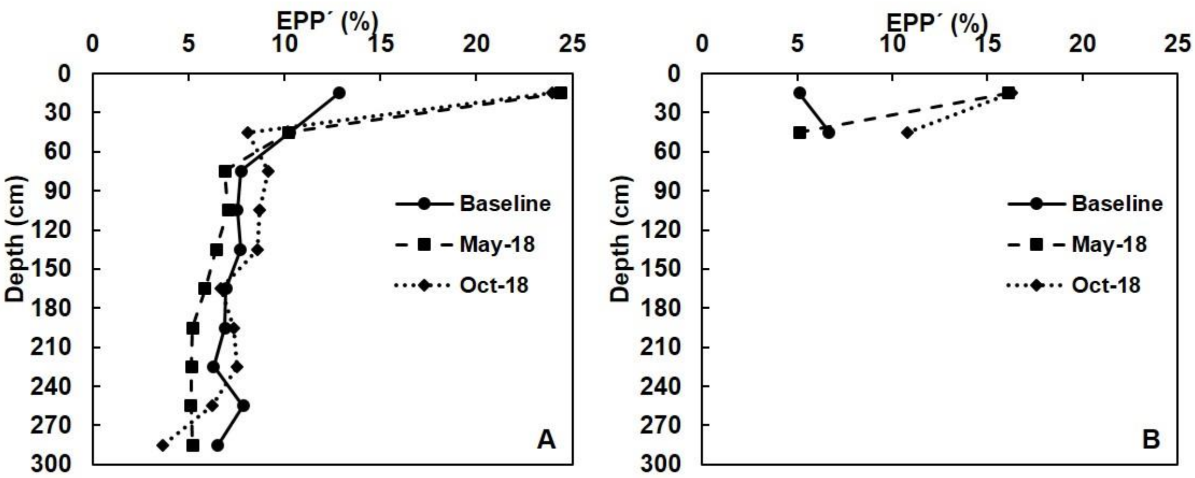

Figure 3.28. Variation in soil extractable potassium percentage (EPP´) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.5. Sodium

3.3.3.5.5.1. Coastal region

Despite the application of nearly 40 kg/ha Na^+ *via* the irrigation water at each of the plots in this region (Table 3.8), soil Na^+ levels remained similar to the baseline after the irrigation applications (Figs. 3.29A & 3.29B). The lack of response may be due to high rainfall during April and May (Fig. 3.6) which leached the applied Na^+ from the soil profile. Although the results indicate that no Na^+ accumulated in the soils, it is important to note that only two irrigations were applied. Further research is needed to investigate the long-term impact of fractionally applied winery wastewater at these plots.

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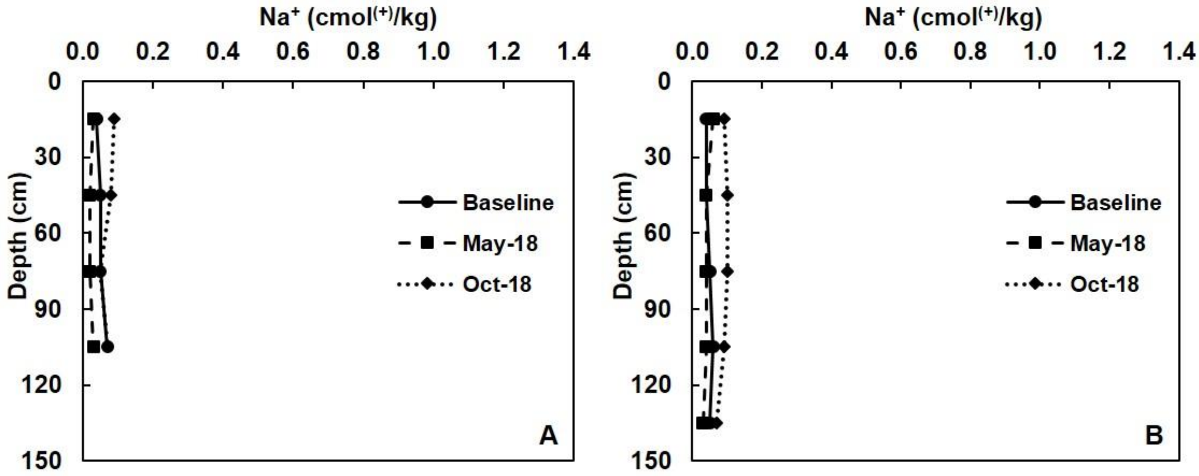

Figure 3.29. Variation in soil extractable sodium (Na^+) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.5.2. Breede River region

The soil Na^+ levels remained comparable to the baseline after the two irrigations applied at the plots in this region (Figs. 3.30A & 3.30B). However, a substantial increase occurred below 90 cm at the BR1 plot and below 60 cm at the BR2 plot after the winter rainfall period. The increase could not be related to a change in soil pH, since the $pH_{(KCl)}$ of both plots remained comparable to the baseline (Figs. 3.9AC & 3.9B). It may, however, be possible that Na^+ accumulated at the soil surface (top 5 cm) and was leached to deeper layers during the rainy season. Gray (2012) reported a significant increase in exchangeable Na^+ of the 0-7.5 cm soil layer of soils irrigated with winery wastewater. Since the extractable Na^+ in the surface soil layer was not determined during this study, the reason for the increase in the subsoil of the BR1 and BR2 plots remain uncertain.

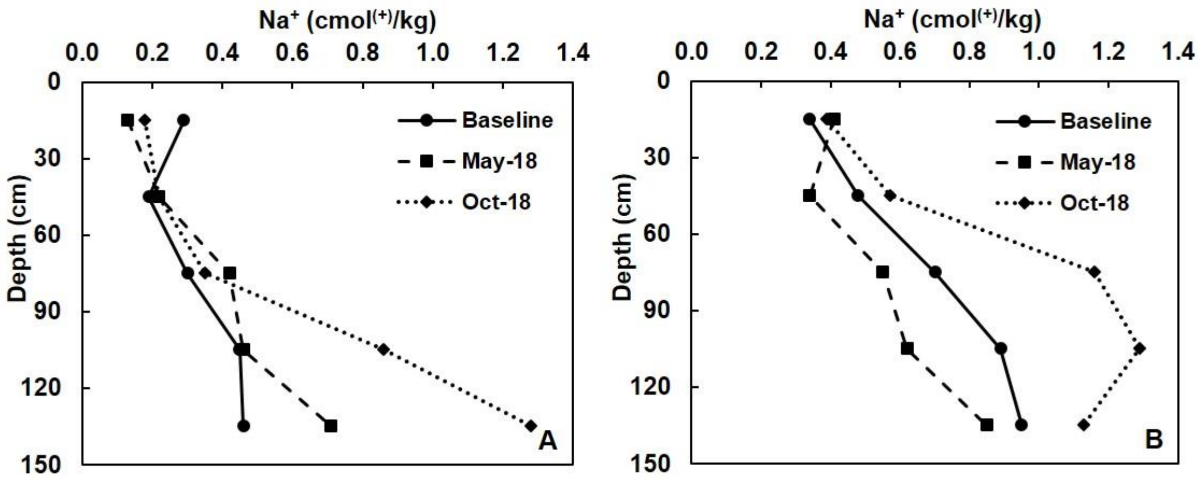

Figure 3.30. Variation in soil extractable sodium (Na^+) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

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3.3.3.5.5.3. Lower Olifants River region

Although the soil extractable Na^+ levels remained similar to the baseline after the majority of irrigations were applied at the LOR1 and LOR2 plots, a slight increase occurred below 30 cm depth following the winter rainfall period (Figs. 3.31A & 3.31B). The increase may be the result of accumulated Na^+ in the surface soil layer which leached to deeper layers after the rainy season (Platts & Grismer, 2014). Since Na^+ accumulation was more pronounced in the subsoil of the LOR2 plot, there is cause for concern that water movement through the soil may be restricted (Halliwell *et al*., 2001).

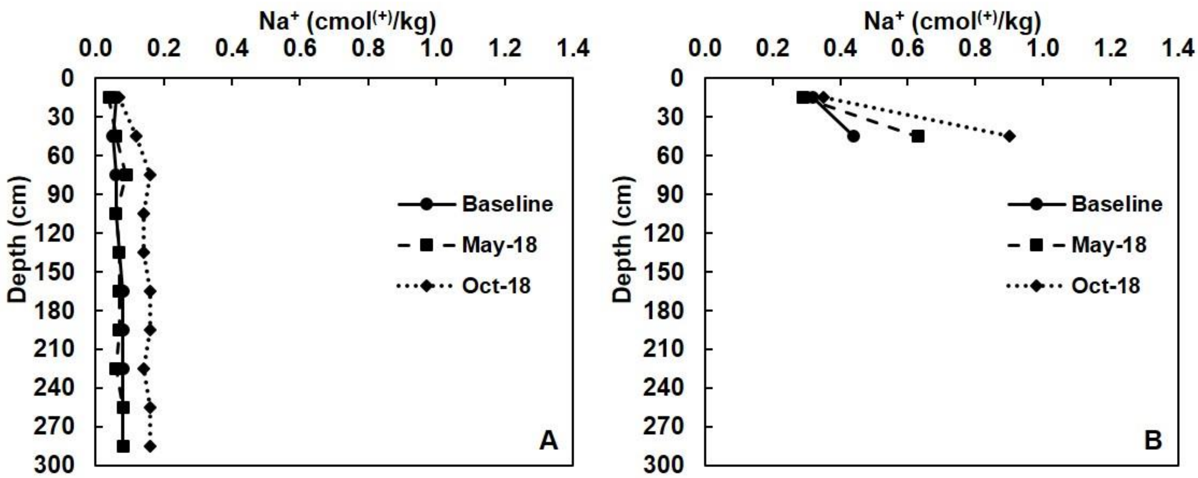

Figure 3.31. Variation in soil extractable sodium (Na^+) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.6. Extractable sodium percentage

3.3.3.5.6.1. Coastal region

The ESP´ at both plots in this region followed similar trends as was observed for extractable Na^+ (Figs. 3.32A & 3.32B). A marginal increase in ESP´ was evident at both plots after the winter rainfall period. However, the increase was not substantial. The fairly constant ESP´ values of these plots throughout the study period was probably the result of low cation concentrations in the irrigation water (Appendix 4.1). Given the low ESP´ values of these plots, no problems with regard to soil structural stability and infiltration are to be expected.

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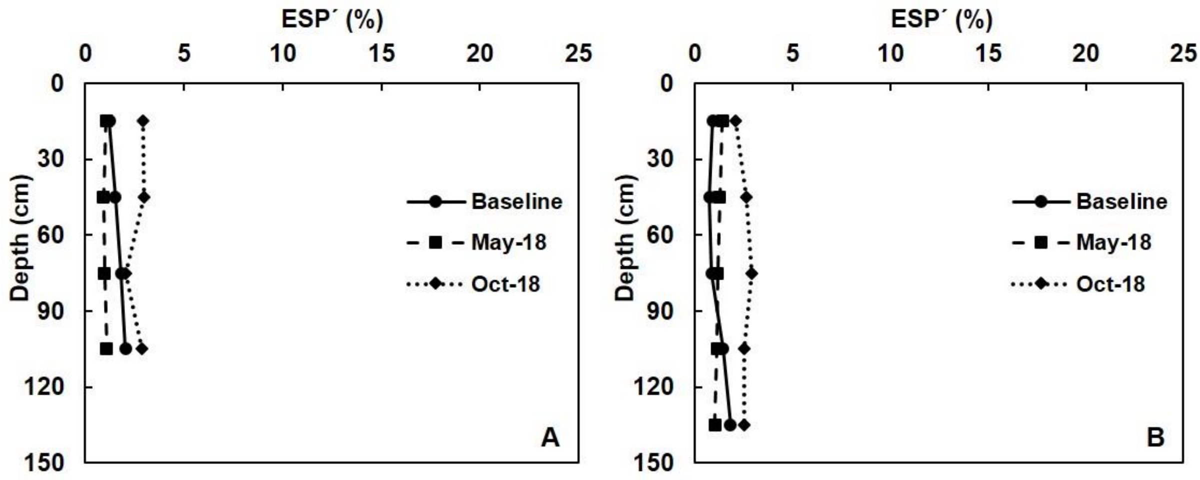

Figure 3.32. Variation in soil extractable sodium percentage (ESP´) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.6.2. Breede River region

A gradual increase in ESP´ with depth occurred at the BR1 experiment plot before irrigation commenced (Fig. 3.33A). A considerable increase in ESP´ occurred in all soil layers below 30 cm depth at this plot after the two irrigations were applied. In contrast, the ESP´ slightly decreased again after the winter rainfall period, except below 120 cm depth where an increase occurred. This increase was a result of high Na^+ levels in the particular soil layer after the rainy season (Fig. 3.30A). A similar increase in the subsoil was observed at the BR2 plot after the winter rainfall period (Fig. 3.33B). However, the increase was less pronounced compared to the BR1 plot and ESP´ levels remained below 10%. The United States Salinity Laboratory (1954) defined an ESP of 15% as the boundary between non-sodic and sodic soils. However, several other studies have reported reductions in *K* at ESP values lower than 15% (Sumner, 1993 and references therein). This value was modified by Bernstein (1974) to 10% for fine textured soils and 20% for course textured soils. Continuous irrigation with in-field fractionally applied winery wastewater with raw water may therefore have potential negative effects on the subsoil *K* of the experiment plots in this region.

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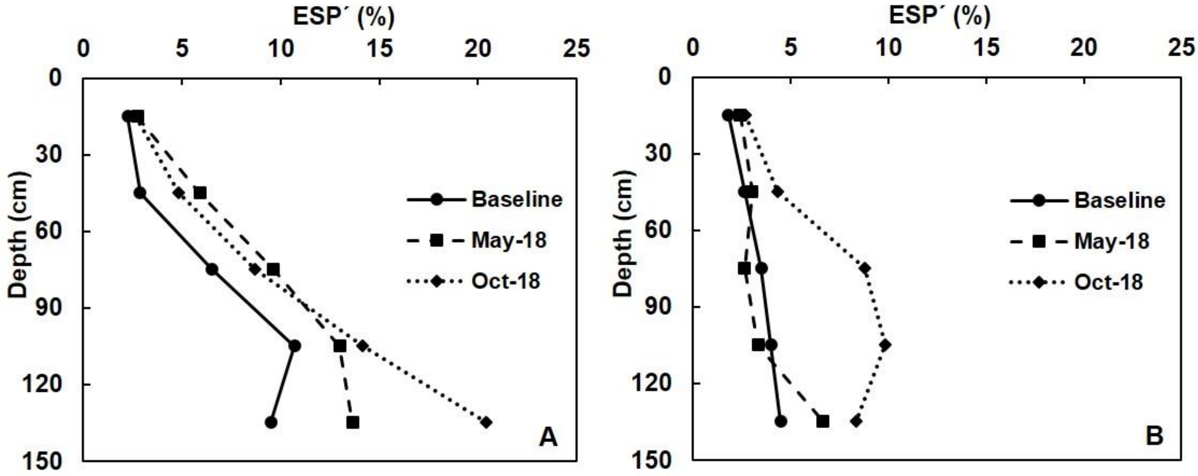

Figure 3.33. Variation in soil extractable sodium percentage (ESP´) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.5.6.3. Lower Olifants River region

The soil ESP´ at the LOR1 plot only increased after the winter rainfall period (Fig. 3.34A). On average, the ESP´ increased by 1.7% compared to the baseline. With the exception of the 60-90 cm soil layer, the ESP´ remained below 5%. Since the ESP´ remained relatively low and the soil at this plot has a low clay content (Appendix 5.3), no problems with regard to soil structural stability and *K* was expected. At the LOR2 plot, the topsoil ESP´ decreased compared to the baseline, whereas the subsoil ESP´ increased after the majority of irrigations were applied, and again after the winter rainfall period (Fig. 3.34B). The increase in ESP´ was a result of the high Na^+ levels which accumulated at this plot (Fig. 3.31B). Compared to the baseline, the ESP´ in the 30-60 cm soil layer increased by *ca*. 5% at the end of the study period. In a loamy sand soil, Levy *et al*. (2005) reported a decline in *K* of *ca*. 65% when soil ESP increased from 1.1% to 4.6%. Therefore, the increase in ESP´ at the LOR2 plot may reduce water movement through the soil profile.

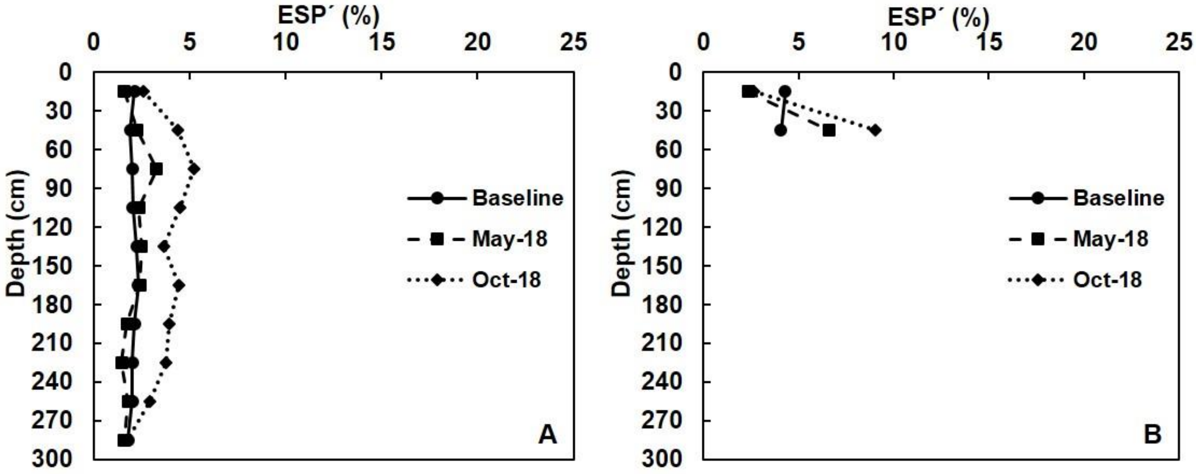

Figure 3.34. Variation in soil extractable sodium percentage (ESP´) before irrigation commenced (Baseline), after the main irrigation period (May-18) and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.3.6. Trace elements

No trends with regard to soil B^{3+}, Cu^{2+}, Fe^{2+}, Mn^{2+} and Zn^{2+} levels that could be related to the irrigation water were observed at the end of the 2017/18 season at any of the experiment plots (Figs. 3.35, 3.36, 3.37, 3.38 & 3.39). The lack of response was due to low concentrations of these elements in the irrigation water (Appendices 4.2, 4.4, 4.6 & 4.8). With the exception of the topsoils of the BR2 and LOR2 plots (Figs. 3.35D & 3.35F), soil B^{3+} levels were below the recommended norm of 1 mg/kg to sustain optimal grapevine growth (Saayman, 1981 and references therein). Large standard deviations were observed in the mean soil Cu^{2+} levels of the topsoil at the BR1 and BR2 plots (Figs. 3.36C & 3.36D). This was likely the result of residues from Cu^{2+} sprays used as a fungicide in the vineyard (Mackie *et al*., 2012; Da Rosa Couto *et al*., 2015). However, the Cu^{2+} levels in these soils were well below the limit of 121 mg/kg at which grapevine growth may be adversely affected (Brunetto *et al*., 2017). Furthermore, the high standard deviation for Zn^{2+} in the topsoil of the BR1 plot (Fig. 3.39C), was the result of high Zn^{2+} levels present in the soil before irrigation commenced (data not shown). The high Zn^{2+} levels may also be a result of fungicide residues in the soil (Couto *et al*., 2015).

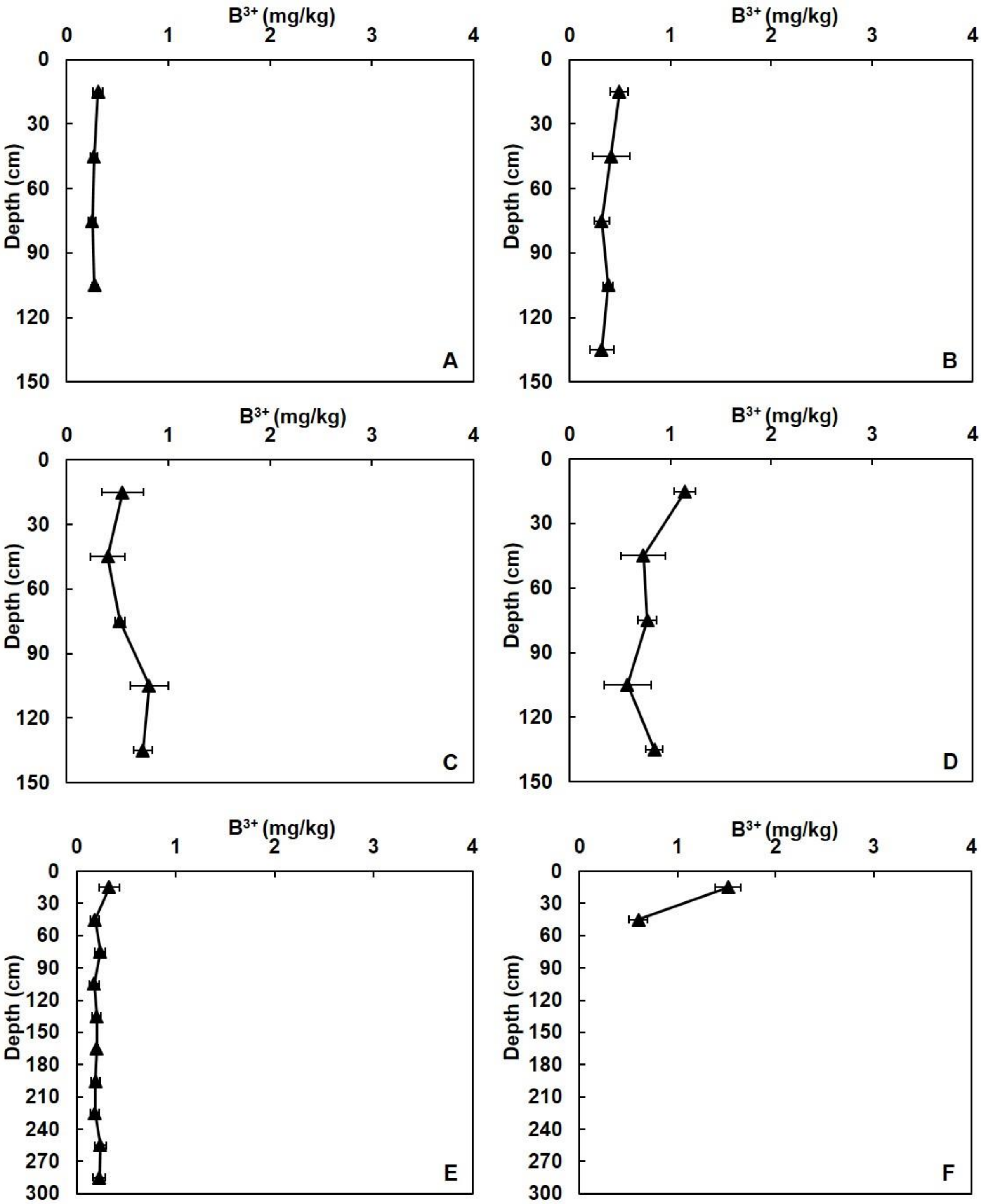

Figure 3.35. Variation in mean soil boron (B^{3+}) levels measured during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

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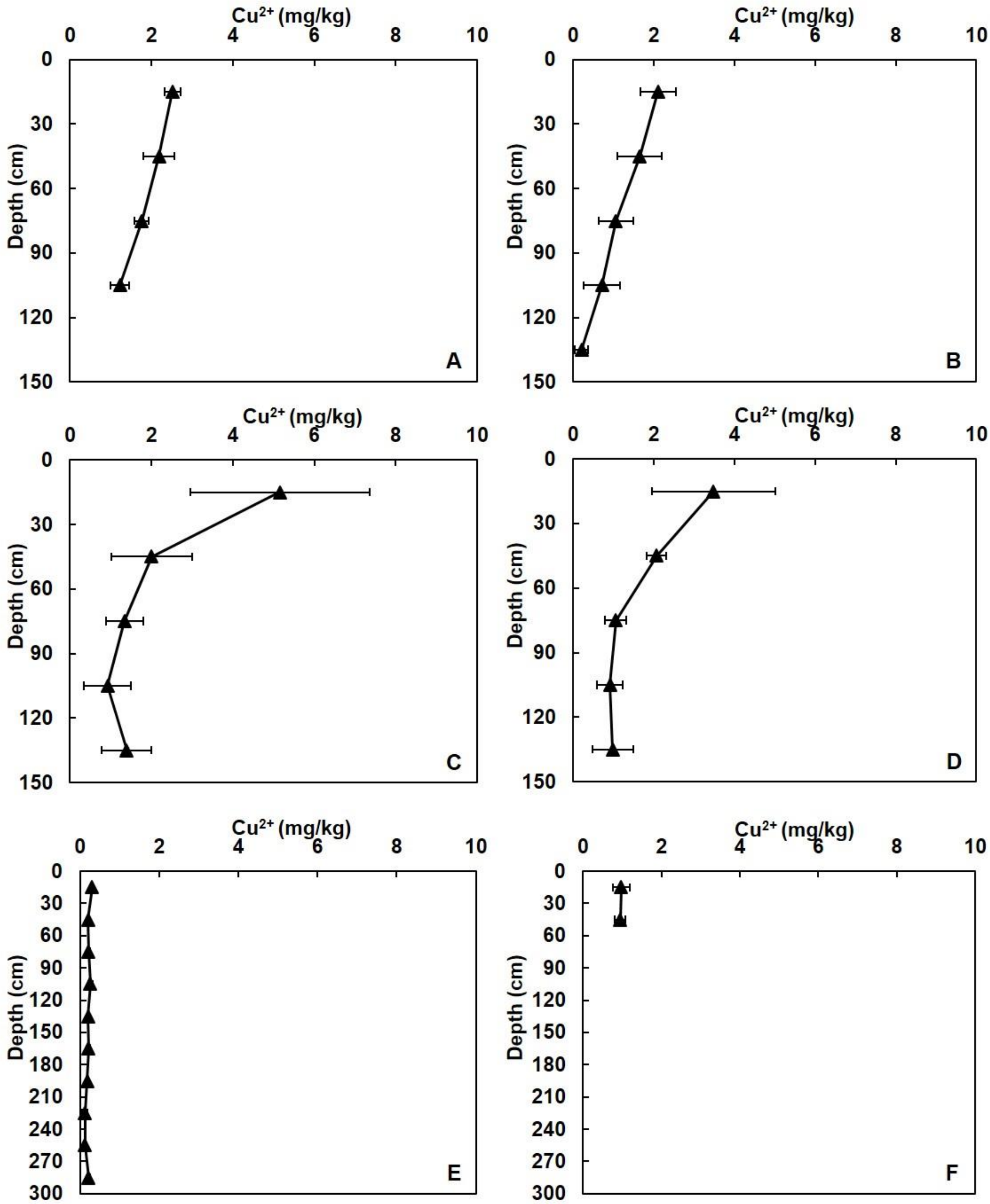

Figure 3.36. Variation in mean soil copper (Cu^{2+}) levels measured during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

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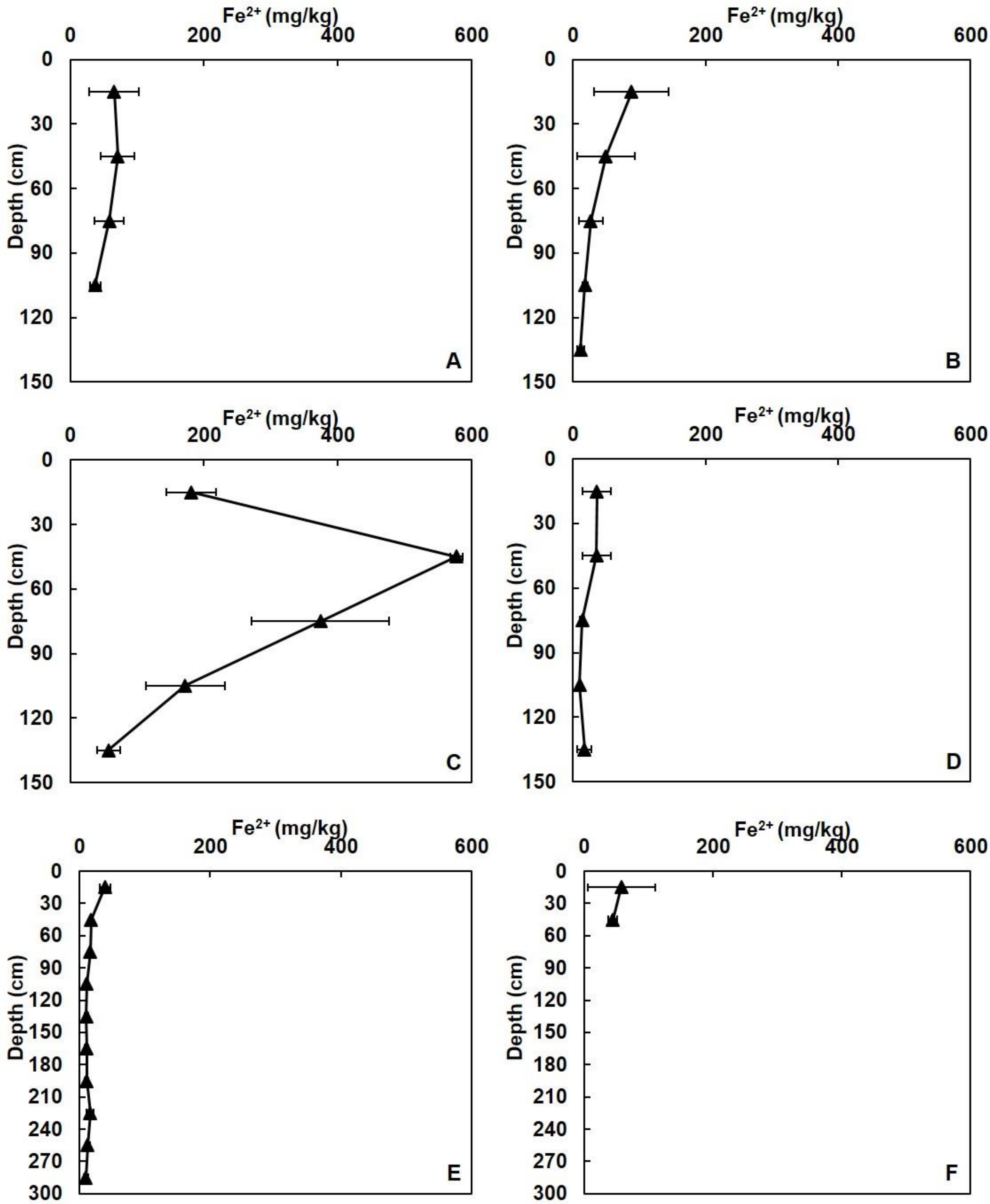

Figure 3.37. Variation in mean soil iron (Fe^{2+}) levels during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

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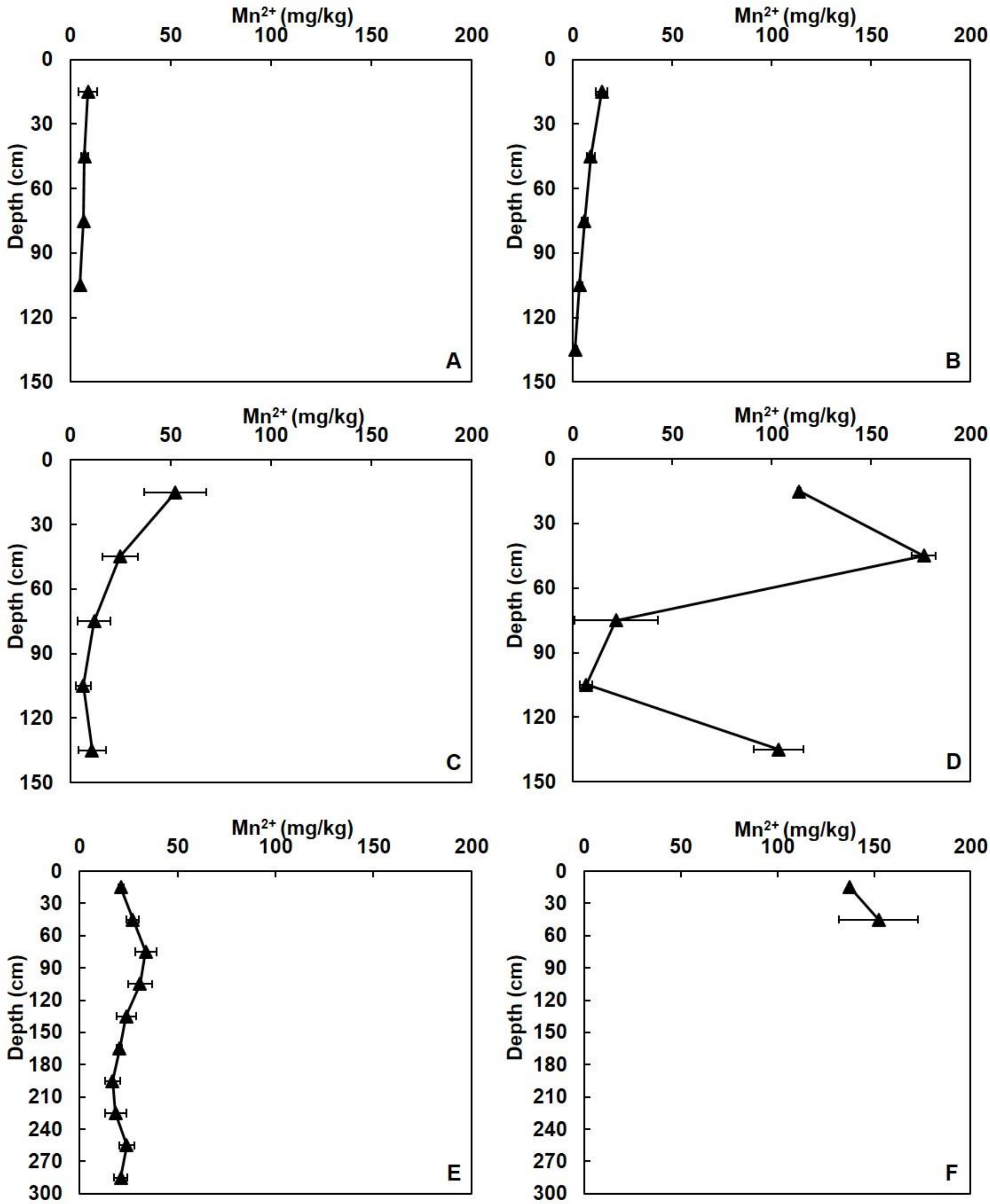

Figure 3.38. Variation in mean soil manganese (Mn^{2+}) levels during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

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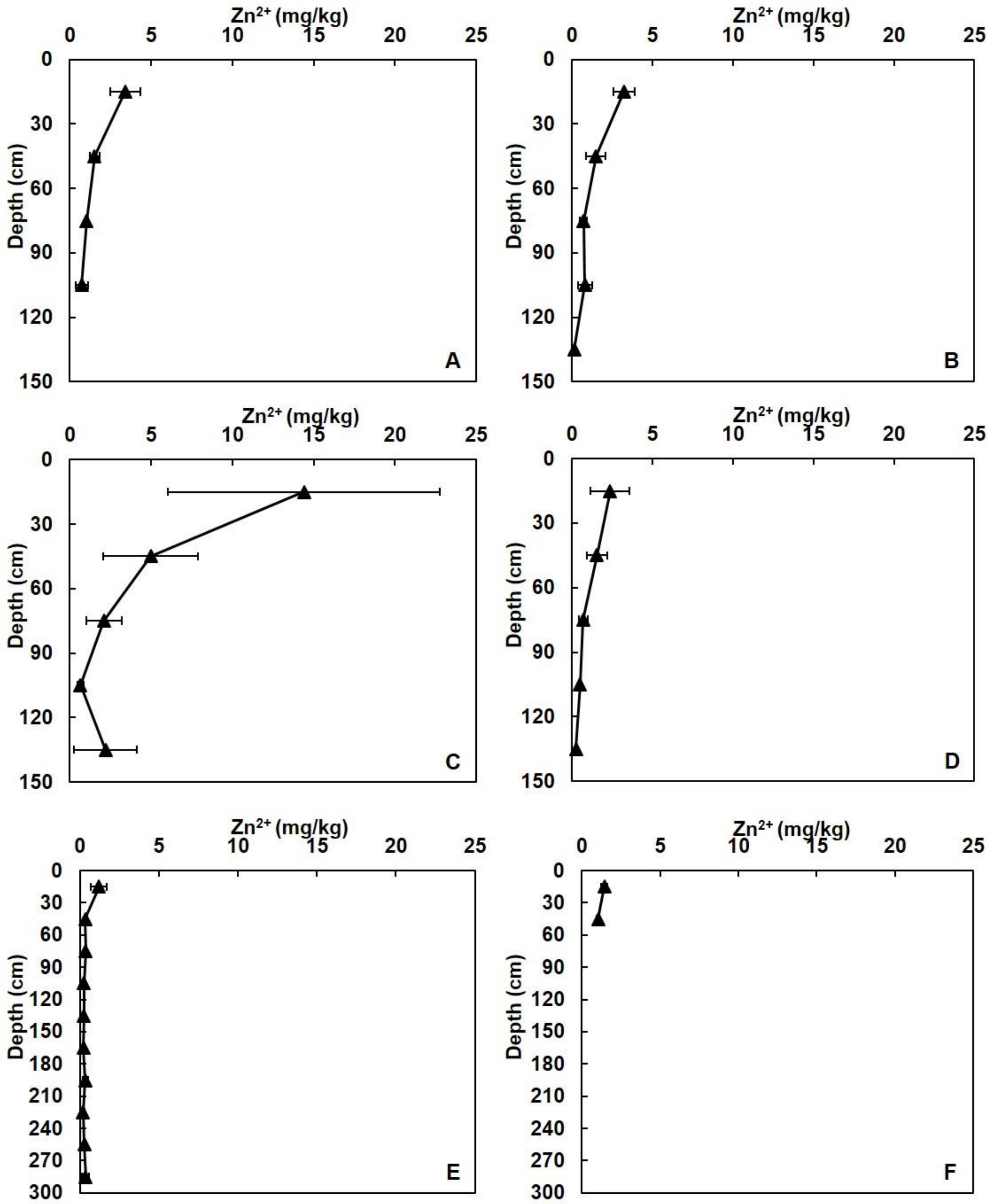

Figure 3.39. Variation in soil zinc (Zn^{2+}) levels during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

3.3.3.7. Soil organic carbon

There was no shift in soil organic carbon (SOC) content from the baseline values at any of the experiment plots at the end of the study period (Fig. 3.40). The lack of response is an indication that the organic C levels in the irrigation water was too low to enhance soil fertility. A similar response was reported by Howell *et al*. (2018). It to the decomposition of organic material upon soil aeration between applications of diluted winery wastewater irrigation and therefore limited the accumulation of SOC. The results are in contrast to increased SOC due to irrigation with undiluted winery wastewater (Kumar *et al*., 2006; 2009; Mosse *et al*., 2012; Hirzel *et al*., 2017).

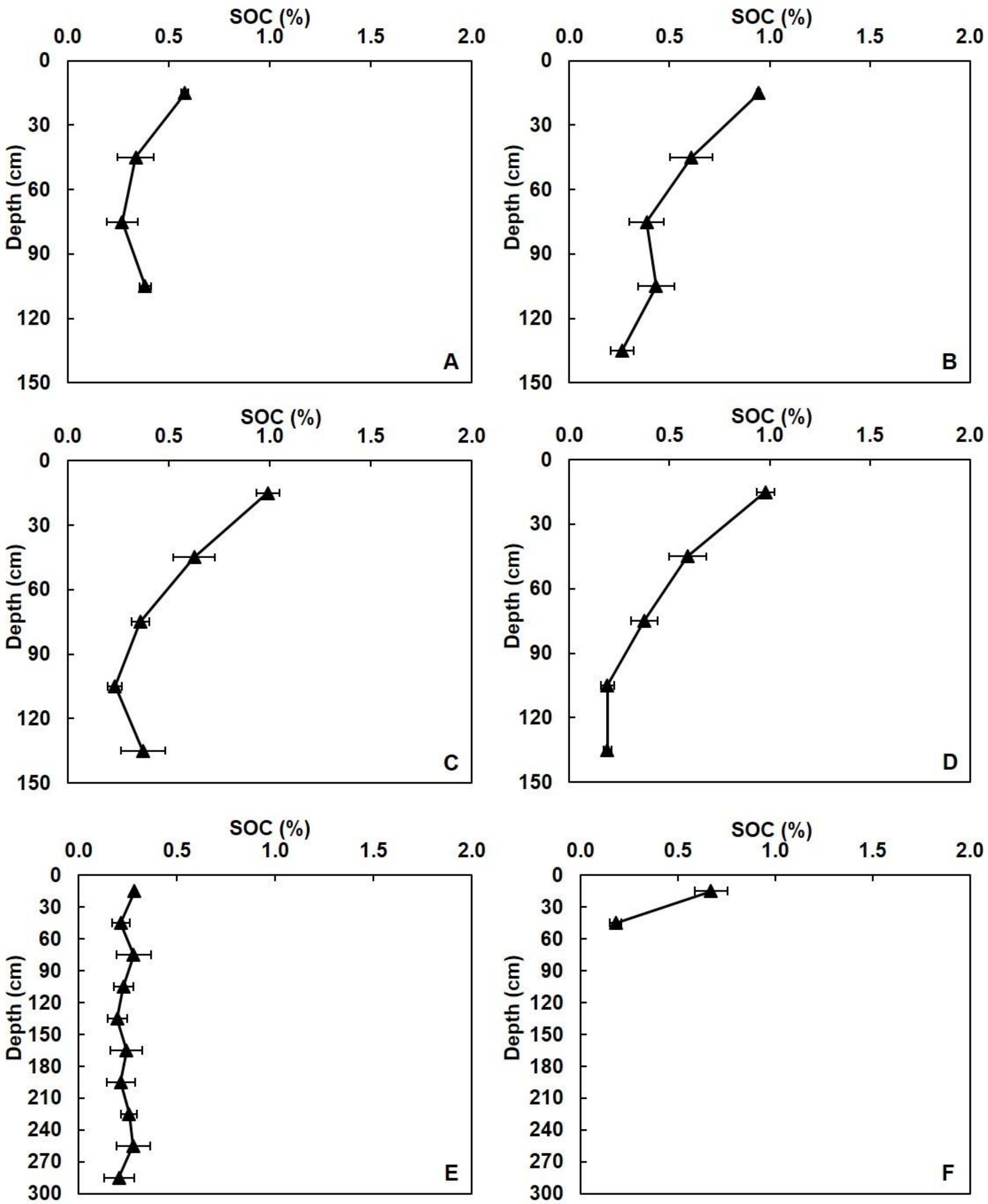

Figure 3.40. Variation in soil organic carbon (SOC) content during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

3.3.3.8. Sulfur

No considerable deviation in soil sulfur (S) levels from the baseline values were observed at any of the experiment plots at the end of the study period (Fig. 3.41). Since SO_4^{2-} is highly mobile, it is possible that the large amounts of SO_4^{2-} applied *via* the irrigation water at the two plots in the Lower Olifants River region (Table 3.13) were leached from the soil by the immediate application of raw water and the rainfall during the winter (Hinckley & Matson, 2011).

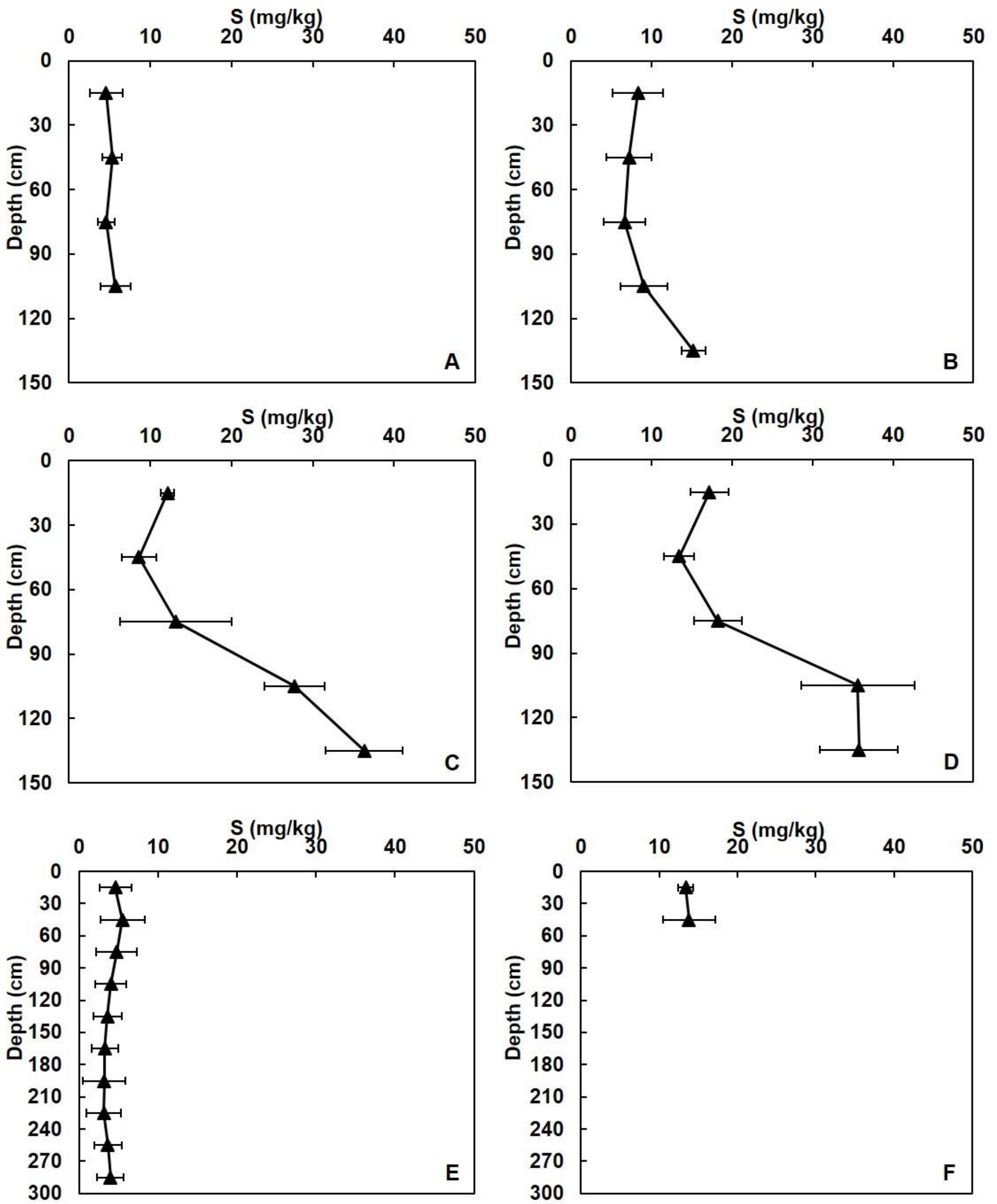

Figure 3.41. Variation in soil sulfur (S) levels during the 2017/18 season at the (A) C1, (B) C2, (C) BR1, (D) BR2, (E) LOR1 and (F) LOR2 experiment plots where grapevines were irrigated *via* fractional application of winery wastewater with raw water. Horizontal bars indicate ± standard deviation.

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3.3.4. Soil physical properties

3.3.4.1. Soil hydraulic properties

3.3.4.1.1. Coastal region

C1 experiment plot: Irrigation using in-field fractionally applied winery wastewater with raw water considerably decreased the constant-head water IR and near-saturation hydraulic conductivity (K_{ns}) at the C1 plot compared to the baseline values (Fig. 3.42A). A significant reduction in saturated hydraulic conductivity (K_s) was reported for a course sandy soil irrigated with treated municipal wastewater for over 25 years (Tarchouna *et al*., 2010). The decrease was attributed to the breakdown of soil aggregates, as well as dispersion of clay particles due to Na^+ accumulation and the subsequent collapse of soil pores. Although no K^+ or Na^+ accumulation was evident in the 0-30 cm soil layer of the C1 plot (Figs. 3.17A & 3.29A), it is possible that a surface crust may have formed in the 0-2 cm surface soil layer which could have reduced IR (Agassi *et al*., 1981; Viviani & Iovino, 2004). Compared to a raw water control, Lado *et al*. (2005) reported enhanced surface crust (seal) formation and lower IR in a sandy, non-calcareous soil that was irrigated with treated municipal wastewater for over 10 years. They attributed the differences between the treatments to higher clay dispersion which was caused by higher SAR values in the soil solution of the wastewater irrigated soils. However, the soil chemical properties of the surface layer were not analysed during this study. Therefore, further investigations regarding the mechanism by which the hydraulic properties of this particular soil were reduced, is required. Following the winter rainfall period, the IR of the C1 plot increased to levels higher than the baseline, whereas the K_{ns} remained unchanged after the rainy season (Fig. 3.42A). The topsoil of this plot was loosened by the grower during the rainy season when weeds were mechanically removed. This would have increased the macro-porosity of the topsoil and subsequently increased IR (Beven & Germann, 1982). Furthermore, mechanical disturbance of the topsoil may have diminished a thin surface crust if it existed. The K_{ns} that remained low after the winter rainfall period is concerning since it is an indication that water movement through meso-pores are limited at this plot. According to Hillel (2004), meso-pores primarily facilitate water percolation through the soil. Therefore, the grapevines at this plot may be subjected to water constraints since water movement through the soil profile is limited.

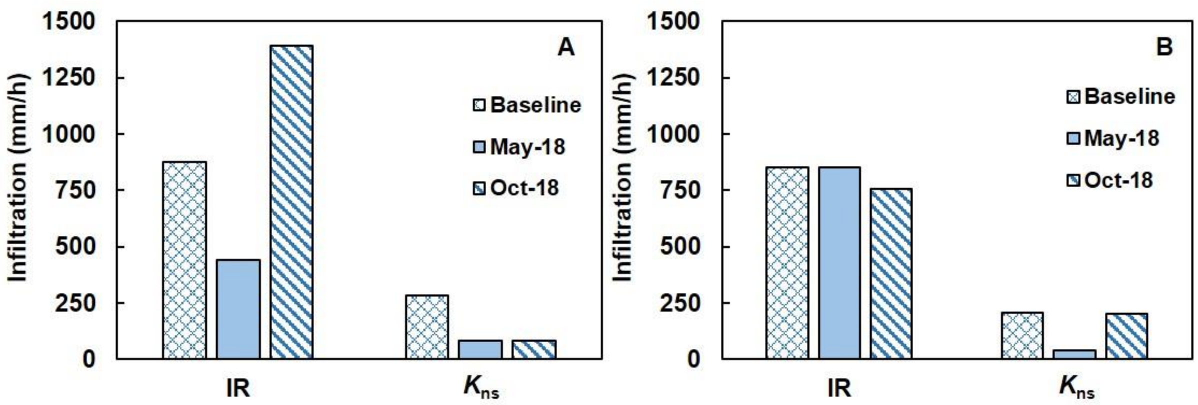

Figure 3.42. Variation in infiltration rate (IR) and near-saturation hydraulic conductivity (K_{ns}) before irrigation commenced (Baseline), after the main irrigation season (May-18), and after the winter rainfall period (Oct-18) at the (A) C1 and (B) C2 experiment plots in the Coastal region where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water during the 2017/18 season.

C2 experiment plot: In contrast to the C1 plot, the IR at the C2 plot remained comparable to the baseline after the irrigation applications, and only slightly decreased after the winter rainfall period (Fig. 3.42B). The lack of response at this plot may be a result of the higher SOC content of this plot compared to the C1 plot (Fig. 3.40B), since increased OM content can prevent crust formation and decreased IR (Lado *et al*., 2004b). In contrast to the IR, the K_{ns} at the C2 plot decreased considerably following the two irrigations applied at this plot (Fig. 3.42B). However, the K_{ns} increased to levels comparable to the baseline after the winter rainfall period. The increase in K_{ns} may be a result of the greater EPP´ levels in the topsoil of C2 plot at the end of the study period (Fig. 3.26B). Chen *et al*. (1983) reported increased K with increasing EPP for a loamy sand soil up to EPP values of 20%. The application of appreciable amounts of K^+ may therefore have a positive impact on the K_{ns} at this particular plot.

3.3.4.1.2. Breede River region

BR1 experiment plot: Both the IR and K_{ns} of the soil at the BR1 plot decreased considerably compared to the baseline after the two irrigation applications at this plot (Fig. 3.43A). However, after the winter rainfall period, the values of both parameters increased to levels similar to the baseline levels, and increased even further than the baseline in the case of IR. These results support the hypothesis that Na^+ accumulated in a surface crust at the BR1 plot after the irrigation applications as discussed in section 3.3.3.5.5.2. The presence of such a surface crust would have limited IR and K_{ns} (Lado & Ben-Hur, 2009 and references therein). Leaching of excessive Na^+ to deeper layers were observed in the subsoil of the BR1 plot after the rainy season (Fig. 3.30A) and would explain the restored IR and K_{ns} during the particular period. The rainfall that occurred in this region during the winter months (Fig. 3.6) was, therefore, sufficient to leach salts applied *via* the irrigation water from the soil surface. This in turn prevented potential negative impacts on the soil hydraulic properties of this plot.

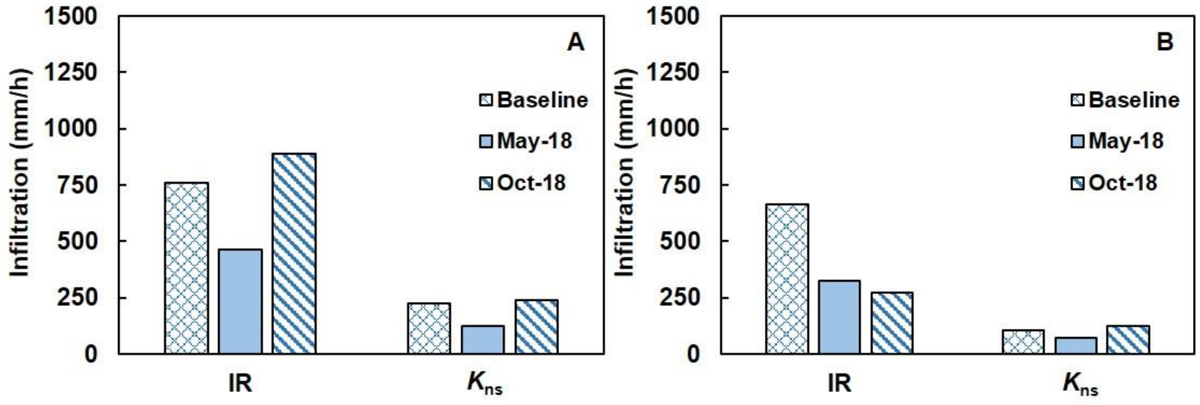

Figure 3.43. Variation in infiltration rate (IR) and near-saturation hydraulic conductivity (K_{ns}) before irrigation commenced (Baseline), after the main irrigation season (May-18), and after the winter rainfall period (Oct-18) at the (A) BR1 and (B) BR2 experiment plots in the Breede River region where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water during the 2017/18 season.

BR2 experiment plot: A progressive decline in IR was observed at the BR2 plot over the course of the study period (Fig. 3.43B). As discussed in section 3.3.3.5.5.2, it is possible that a surface crust formed at this plot after the irrigation applications, which would have resulted in decreased IR (Assouline, 2004). It is likely that the higher clay content in the topsoil of this plot (Appendix 5.2) facilitated the retention of more Na^+ in the surface soil layer after the rainy season. As a result, an increase in IR was not observed at this plot after the winter rainfall period. Although a slight decrease in K_{ns} occurred at the BR2 plot after the irrigations were applied, the K_{ns} at this plot was relatively stable throughout the study period (Fig. 3.43B). The lack of substantial response may be due to appreciable amounts of Ca^{2+} present in the soil (Fig. 3.21B), which could replace the excess exchangeable Na^+, reduce the ESP and prevent clay dispersion (Lado *et al*., 2005).

3.3.4.1.3. Lower Olifants River region

LOR1 experiment plot: A decrease in IR of *ca*. 100 mm/h was observed at the LOR1 plot after the main irrigation season (Fig. 3.44A). Visual observations during the particular time confirmed the presence of a thin surface seal in the top ±2 cm of the soil (data not shown). However, the IR increased again after the rainy season (Fig. 3.44A). The accumulation of Na^+ in the subsoil layers at the LOR1 plot (Fig. 3.31A) is further confirmation that a surface seal existed but was diminished after the rainfall period and Na^+ was leached to deeper soil layers. Similar to the C1 experiment plot, the K_{ns} at the LOR1 plot decreased after the majority of irrigations were applied and remained at similar levels after the rainy season (Fig. 3.44A). Appreciable amounts of mica minerals are often present in the soils of this region (Bühmann *et al*., 2004). The combination of (i) low clay content (Appendix 5.3), (ii) low CEC (data not shown), (iii) ESP´ between 0.5% and 4.7% (Fig. 3.34A) as well as (iv) the possible presence of mica minerals, may have resulted in significant clay dispersion at this plot (Bühmann *et al*., 2004 and references therein). This would in turn have negatively affected the K_{ns} (Quirk & Schofield, 1955). Furthermore, the decrease in K_{ns} may be explained by the high EPP´ levels of the topsoil of this plot following the irrigation applications and the rainfall period (Fig. 3.28A). The presence of mica in soils have been shown to significantly reduce the soil IR and *K* at high EPP levels (Levy & Van der Watt, 1990). Despite the decrease in K_{ns} at this plot, the values remained high enough to allow sufficient water movement through the soil.

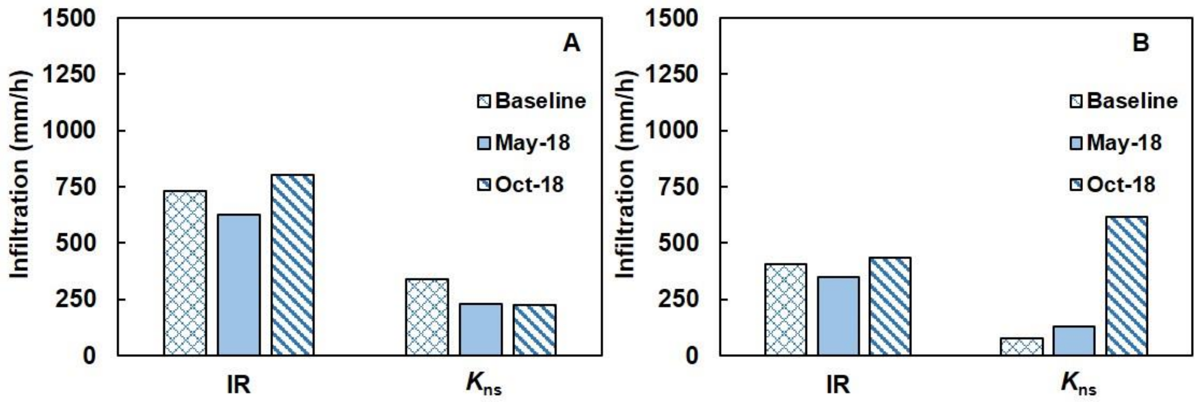

Figure 3.44. Variation in infiltration rate (IR) and near-saturation hydraulic conductivity (K_{ns}) before irrigation commenced (Baseline), after the main irrigation season (May-18), and after the winter rainfall period (Oct-18) at the (A) LOR1 and (B) LOR2 experiment plots in the Lower Olifants River region where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water during the 2017/18 season.

LOR2 experiment plot: Although a slight decrease in IR was observed at the LOR2 plot after the main irrigation season, the IR increased to levels comparable to the baseline after the winter rainfall period (Fig. 3.44B). An unexpected progressive increase in K_{ns} was observed at the LOR2 plot over the course of the study period (Fig. 3.44B). It is possible that the high levels of Ca^{2+} that accumulated in the soils after the irrigation applications (Fig. 3.22B) displaced excess Na^{+} ions that were adsorbed onto the exchange complex during rainfall events, and subsequently increased K_{ns} (Lado & Ben-Hur, 2009). Furthermore, the moderate EPP´ levels in the topsoil of the LOR2 plot (Fig. 3.28B) may have contributed to the increased K_{ns}. Chen *et al*. (1983) reported an increase in *K* of a loamy sand soil as the EPP of the soil increased to a value of 20%, where after *K* decreased significantly. The results from this experiment plot indicated that short-term irrigation using in-field fractionally applied winery wastewater with raw water did not have detrimental effects on the soil hydraulic properties under the prevailing conditions.

3.3.5. Soil water content

3.3.5.1. Coastal region

C1 experiment plot: The SWC refill zone for the loamy sand C1 plot was established as *ca*. 195 mm to 210 mm per 180 cm soil depth, to ensure grapevine midday Ψ_S remains between a range of -0.85 MPa to -1.15 MPa (Fig. 3.45A). The selected Ψ_S range designates moderate water constraints in Cabernet Sauvignon grapevines (Myburgh *et al*., 2016). This norm was selected to limit the exposure of the newly planted grapevines to severe water constraints. The application of more water at this plot would also encourage root development throughout the newly prepared soil. Due to the unavailability of winery wastewater prior to the harvest period, the first irrigation at the C1 plot was only applied on the 19th of February 2018 when the SWC was already below the refill zone (Fig. 3.46). The total volume of irrigation water applied was *ca.* 47.2 mm (data not shown). Due to logistical constraints, the second irrigation at the C1 and C2 experiment plots had to be applied simultaneously. Since the SWC at the C2 plot had not reached the refill zone (Fig. 3.47), the soil was allowed to dry out further. Consequently, the SWC at the C1 plot decreased to levels below the refill zone prior to the second irrigation (Fig. 3.46). Although the field capacity of the soils used in the present study were not determined, it was estimated to be the SWC at which the soils were *ca*. 12 hours after irrigation was applied. During the second irrigation (26/03/2018), a larger volume of irrigation water was applied, after which the field capacity was estimated to be *ca.* 260 mm/180 cm. However, the SWC measured during the winter rainfall season (May to September) indicated that the field capacity at the C1 plot was likely closer to 300 mm/180 cm (Fig. 3.46). Therefore, an even larger volume of irrigation water can be applied per irrigation at this plot if the objective is to irrigate up to estimated field capacity. Furthermore, after the second irrigation, lower temperatures, higher rainfall and reduced wind speeds (Figs. 3.4, 3.6 & 3.7), as well as the limited growth of the young grapevines, limited evapotranspiration and SWC levels remained between the refill zone and the estimated field capacity (Fig. 3.46). Therefore, no further irrigations were applied at this particular plot during the 2017/18 season.

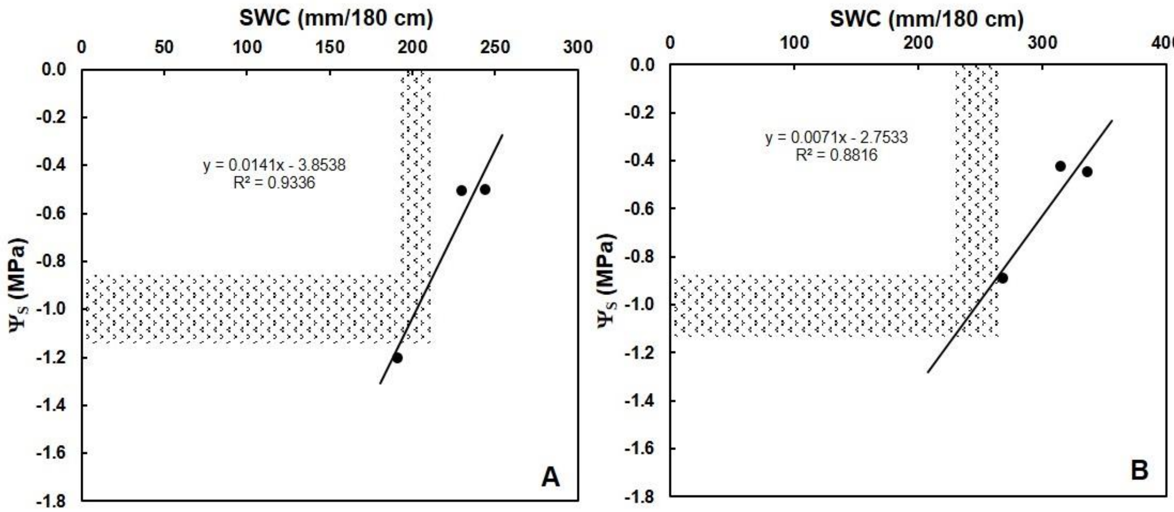

Figure 3.45. Correlation between midday stem water potential (Ψ_S) and soil water content (SWC) to a depth of 180 cm for the Cabernet Sauvignon grapevines at the (A) C1 and (B) C2 experiment plots. Dashed areas indicate the Ψ_S range used to determine the SWC refill zone.

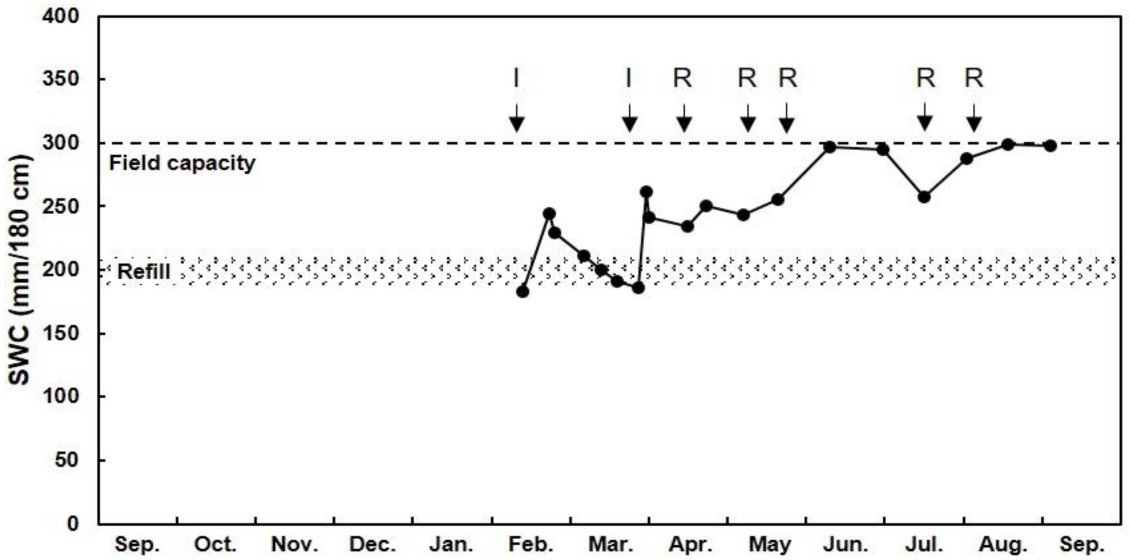

Figure 3.46. Variation in soil water content (SWC) up to 180 cm soil depth during the 2017/18 season at the C1 experiment plot where newly planted Cabernet Sauvignon grapevines were irrigated *via* the in-field fractional use of winery wastewater with raw water. I is irrigation applications and R is rainfall events.

C2 experiment plot: The SWC refill zone for the sandy clay loam C2 plot was established as *ca*. 225 mm to 270 mm per 180 cm soil depth, to ensure that grapevine midday Ψ_S remains between a range of -0.85 MPa to -1.15 MPa (Fig. 3.45B). This norm was selected for reasons as discussed above for the C1 plot. Due to the unavailability of winery wastewater prior to the harvest period, the first irrigation was only applied on the 19th of February 2018 (Fig. 3.47). Prior to that, the vineyards were irrigated according to the grower's normal irrigation scheduling strategy. The second irrigation (26/03/2018) was applied when the SWC reached 267 mm/180 cm (Fig. 3.47). Similar to the C1 experiment plot, the estimated field capacity was only established after a larger volume of irrigation water was applied during the second irrigation. The field capacity at this plot was estimated to be *ca*. 355 mm/180 cm (Fig. 3.47). The higher clay fraction of the soil at the C2 plot (Appendix 5.1) increased the soil's water retention capacity which would explain the higher estimated field capacity compared to the C1 plot (Fig. 3.46). Following the second irrigation application, the SWC at the C2 plot remained at levels between the refill zone and estimated field capacity. This was probably due to reduced evapotranspiration which was facilitated by lower temperatures, higher rainfall and reduced wind speeds in the region (Figs. 3.4, 3.6 & 3.7). As a result, only two irrigations were applied at the C2 experiment plot throughout the 2017/18 season.

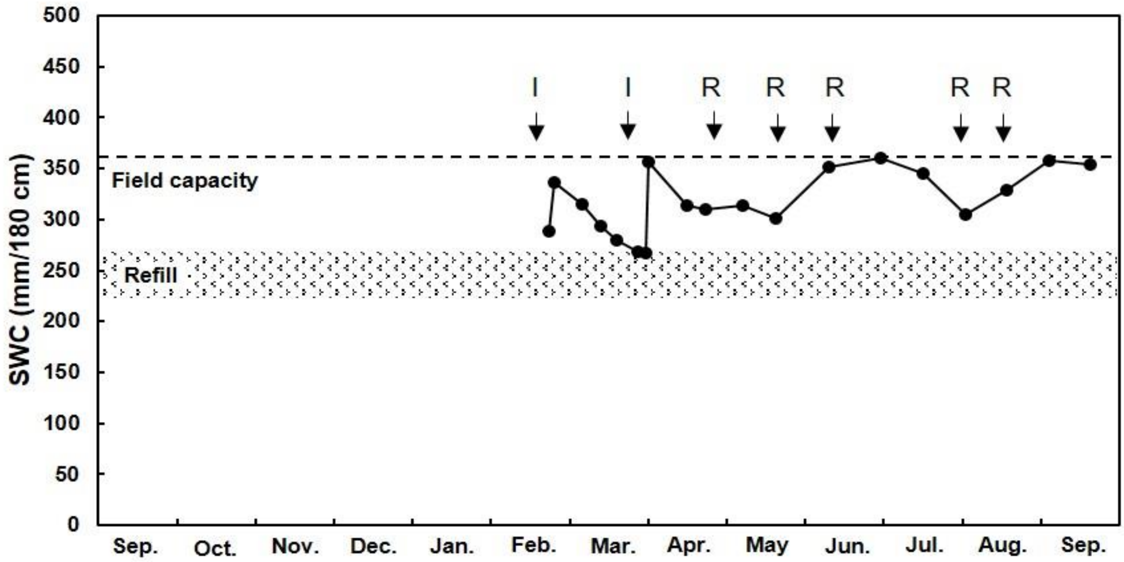

Figure 3.47. Variation in soil water content (SWC) up to 180 cm soil depth during the 2017/18 season at the C2 experiment plot where newly planted Cabernet Sauvignon grapevines were irrigated *via* the in-field fractional use of winery wastewater with raw water. I is irrigation applications and R is rainfall events.

3.3.5.2. Breede River region

BR1 experiment plot: The SWC refill zone of the sandy loam BR1 plot was established as *ca*. 75 mm to 110 mm per 90 cm soil depth, to ensure that grapevine midday Ψ_S remains between a range of -1.5 MPa to -1.6 MPa (Fig. 3.48A). The specific range of Ψ_S is considered ideal for obtaining high quality Shiraz wines (Myburgh *et al*., 2016; Myburgh, 2018 and references therein). Due to an unforeseen delay in the installation of infrastructure at the experiment plots in the Breede River region, irrigation with in-field fractionally applied winery wastewater only commenced on the 6^{th} of February 2018 (Fig. 3.49). Prior to that, the vineyards were irrigated according to the grower's normal irrigation scheduling strategy. Unfortunately, a subsurface water pipe burst near the BR1 experiment plot in late January 2018 which saturated the soil of a large proportion of the experiment plot. Furthermore, a pot trial, which forms part of the bigger research project, was set up within this experiment plot. In order to expose the soils used in the pot trial to as much winery wastewater as possible, the first irrigation at the BR1 plot was applied when the soil was still saturated. The saturated soil was allowed to dry out after the first irrigation, where after the second irrigation was applied in the post-harvest period (24/04/2018), when the SWC reached 103 mm/90 cm. Since the soil was saturated before the first irrigation, the field capacity could only be estimated after the second irrigation application. It was initially estimated as *ca*. 175 mm/90 cm. However, the SWC measured during the winter rainfall season indicated that the field capacity was likely closer to 190 mm/90 cm. Therefore, larger volumes of irrigation can be applied at this plot if the objective is to irrigate up to the estimated field capacity. Following the second irrigation, the SWC at this plot was sustained at levels near the estimated field capacity by lower temperatures (Fig. 3.4), higher rainfall (Fig. 3.6) and limited water uptake by grapevines during the post-harvest period which would have decreased evapotranspiration. Therefore, no further irrigations were applied at this plot during the 2017/18 season.

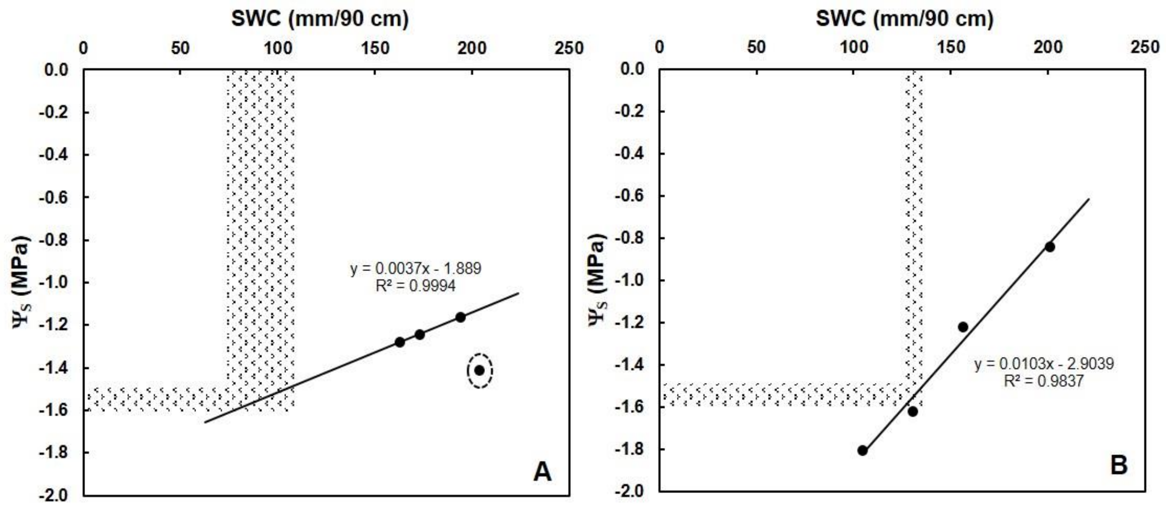

Figure 3.48. Correlation between midday stem water potential (Ψ_S) and soil water content (SWC) to a depth of 90 cm for the Shiraz grapevines at the (A) BR1 and (B) BR2 experiment plots. Dashed areas indicate the Ψ_S range used to determine the SWC refill zone. Encircled data point was regarded as an outlier.

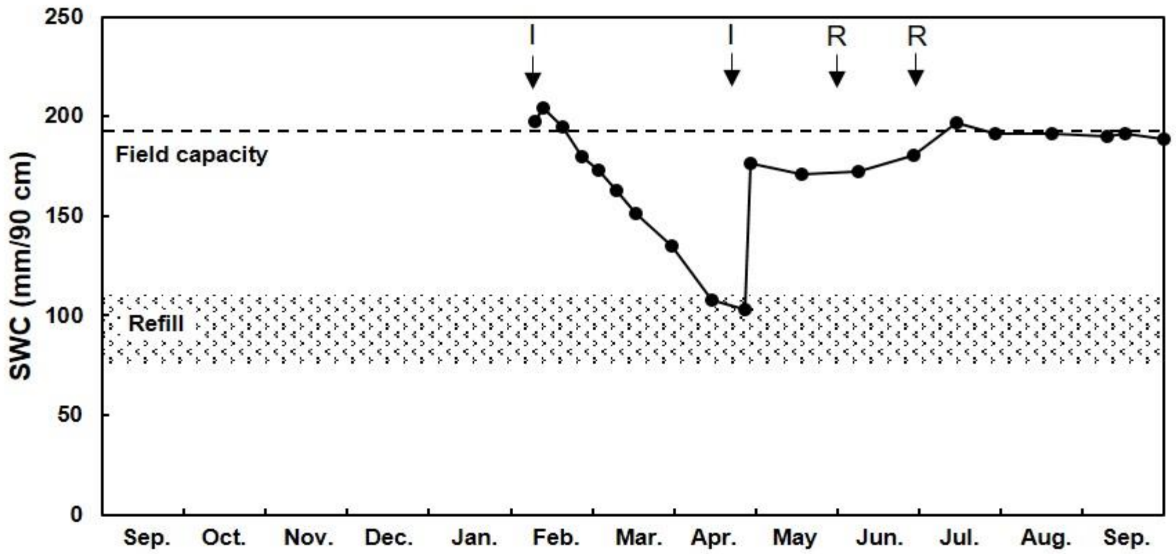

Figure 3.49. Variation in soil water content (SWC) up to 90 cm soil depth during the 2017/18 season at the BR1 experiment plot where Shiraz grapevines were irrigated *via* the in-field fractional use of winery wastewater with raw water. I is irrigation applications and R is rainfall events.

BR2 experiment plot: The SWC refill zone for the sandy clay loam BR2 plot was established as *ca*. 125 mm to 135 mm per 90 cm soil depth, to ensure that grapevine midday Ψ_S remains between a range of -1.5 MPa to -1.6 MPa (Fig. 3.48A). This norm was selected for reasons as discussed above for the BR1 plot. The delay in installation of the infrastructure also affected the BR2 plot and the first irrigation was only applied on the 6th of February 2018 (Fig. 3.50). Following the first irrigation, the application of winery wastewater was disrupted due to technical problems with the irrigation infrastructure. As a result, the SWC at the BR2 plot decreased to levels far below the

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refill zone before irrigation could recommence on the 24th of April 2018. However, the grapes were harvested in early March and would therefore not have been affected by the low SWC experienced at this plot. Compared to the soil at the BR1 plot, the soil at the BR2 plot dried out more rapidly during February, March and April. This was probably a result of the saturated subsoil at the BR1 plot which dried out slower. Furthermore, after the second irrigation application, the first winter rainfall began (Fig. 3.6) which kept the SWC at the BR2 plot above the refill zone. As a result, no further irrigations were applied during the 2017/18 season. The field capacity at this plot was initially estimated to be *ca*. 200 mm/90 cm. However, the SWC measured during the rainfall season indicated that the estimated field capacity is probably closer to 230 mm/90 cm (Fig. 3.50). Therefore, a larger volume of irrigation water can be applied per irrigation at this plot if the objective is to irrigate up to field capacity.

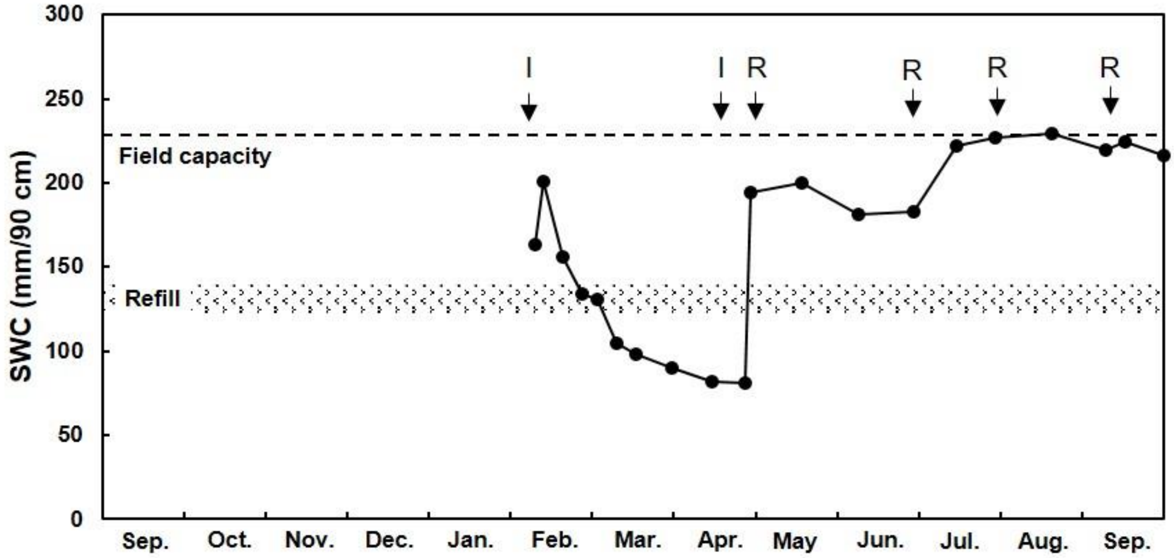

Figure 3.50. Variation in soil water content (SWC) up to 90 cm soil depth during the 2017/18 season at the BR2 experiment plot where Shiraz grapevines were irrigated *via* the in-field fractional use of winery wastewater with raw water. I is irrigation applications and R is rainfall events.

3.3.5.3. Lower Olifants River region

LOR1 experiment plot: The SWC refill zone at the deep sandy LOR1 plot was established as *ca*. 70 mm to 85 mm per 150 cm soil depth, to ensure that grapevine midday Ψ_S remains between a range of -1.1 MPa to -1.2 MPa (Fig. 3.51A). Although the desired Ψ_S for obtaining good quality Shiraz wine is -1.6 MPa (Myburgh *et al*., 2016; Myburgh, 2018 and references therein), it became evident after several Ψ_S measurements at the beginning of the season that the grapevines at this particular plot would not reach such low Ψ_S values unless the SWC was 0 mm (Fig. 3.51A).

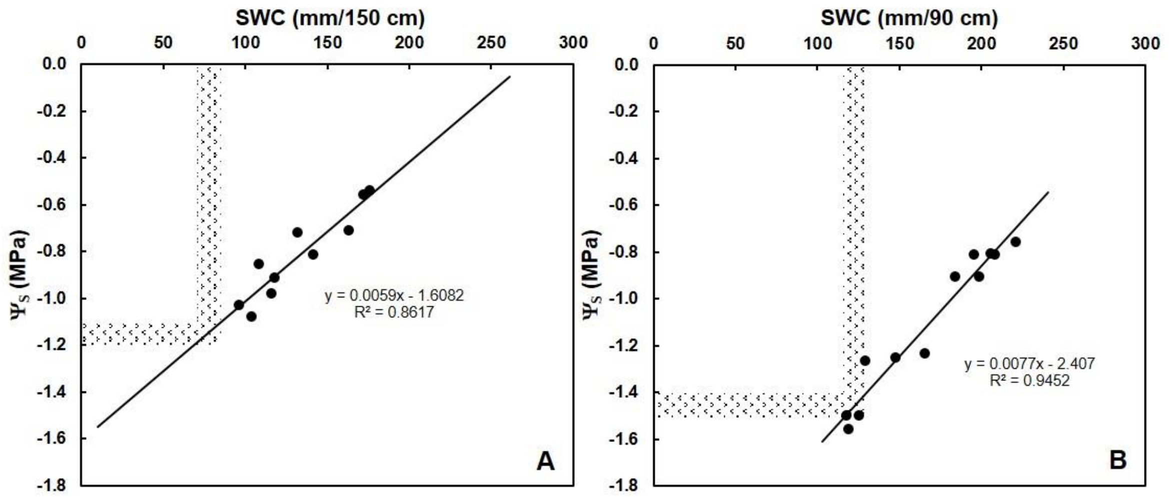

Figure 3.51. Correlation between stem water potential (Ψ_S) and soil water content (SWC) to a depth of (A) 150 cm for the Shiraz grapevines at the LOR1 experiment plot and (B) 90 cm for the Cabernet Sauvignon grapevines at the LOR2 experiment plot. Dashed areas indicate the Ψ_S range used to determine the SWC refill zone.

The lack of considerable water constraints at this plot may be explained by the high percentage of fine sand present in the soil (Appendix 5.3). This in turn could have increased the soil's water retention capacity and subsequently made more water available to grapevines. Unfortunately, the water retention capacity of the soil was not determined in the present study. In addition, grapevines may have experienced cooler temperatures due to the proximity of the vineyard to the Atlantic Ocean (Table 3.2). Previous research conducted in the Lower Olifants River region indicated that grapevines closer to the ocean were subject to less water constraints compared to grapevines further inland (Bruwer, 2010). Due to these circumstances, the SWC refill zone was established at a Ψ_S range corresponding to moderate water constraints in Shiraz grapevines (Myburgh *et al*., 2016; Myburgh, 2018 and references therein).

A total of six irrigations were applied at the LOR1 plot during the 2017/18 season (Fig. 3.52). The first irrigation was applied after berry set (28/11/2017), where after irrigations were applied at véraison (09/01/2018), during ripening (31/01/2018), after harvest (15/03/2018; 17/05/2018) and again before bud break of the 2018/19 season (27/08/2018). Due to logistical constraints, the irrigations on the 31st of January and the 27th of August had to be applied simultaneously at the LOR1 and LOR2 plots (Figs. 3.52 & 3.53). As a result, the LOR1 plot was irrigated before SWC levels reached the refill zone on both these occasions. Following the irrigation applied in May, lower temperatures, higher rainfall and decreased wind speeds (Figs. 3.4, 3.6 & 3.7), as well as limited water uptake by grapevines during the dormant period, maintained SWC levels between the refill zone and estimated field capacity during the months of June, July and August. The combination of higher temperatures, lower rainfall and greater wind speeds experienced at the LOR1 plot compared to the plots in the Coastal and Breede River regions during the irrigation season (Figs. 3.4, 3.6 & 3.7), would have increased evapotranspiration which necessitated more irrigations.

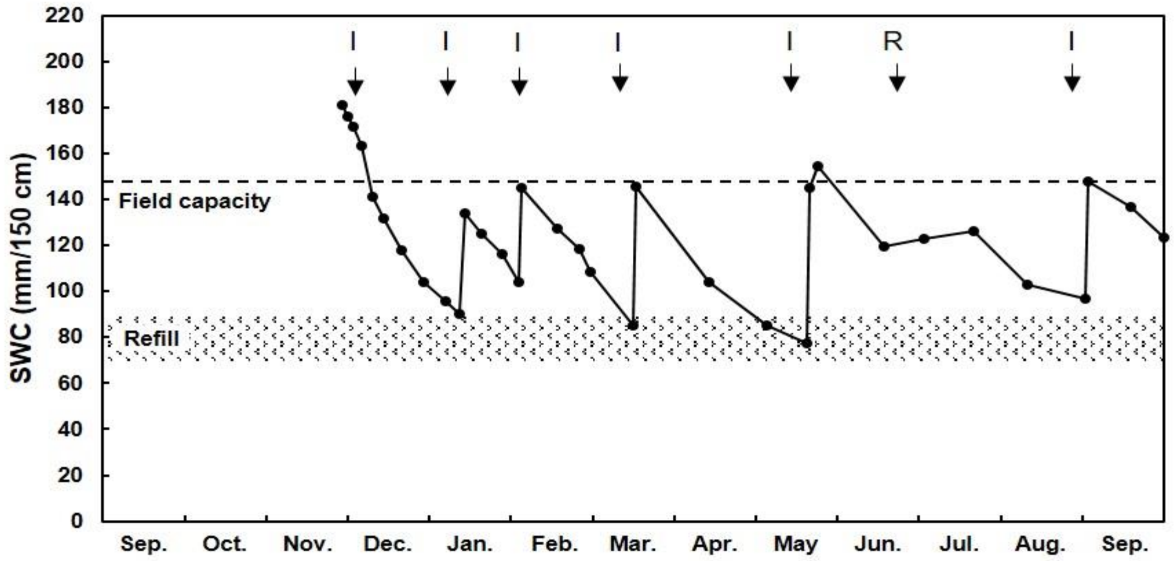

Figure 3.52. Variation in soil water content (SWC) up to 150 cm soil depth during the 2017/18 season at the LOR1 experiment plot where Shiraz grapevines were irrigated *via* the in-field fractional use of winery wastewater with raw water. I is irrigation applications and R is rainfall events.

LOR2 experiment plot: The SWC refill zone at the shallow sandy LOR2 plot was established as *ca*. 110 mm to 130 mm per 90 cm soil depth, to ensure that grapevine midday Ψ_S remains between a range of -1.4 MPa to -1.5 MPa (Fig. 3.51B). The particular Ψ_S range is considered the optimal level of water constraints for production of high-quality Cabernet Sauvignon wines (Myburgh *et al*., 2016). Seven irrigations were applied at this plot during the 2017/18 season (Fig. 3.53). The first irrigation was applied after set (28/11/2017), where after irrigations were applied at the pea size berries stage (19/12/2017), at véraison (10/01/2018), during ripening (31/01/2018), after harvest (26/02/2018; 10/04/2018) and again before bud break of the 2018/19 season (27/08/2018). The fifth irrigation (26/02/2018) was applied when SWC levels were below the refill zone which resulted in the grapevines experiencing more than the desired water constraints. However, since the grapes were harvested on the 14th of February 2018, the high water constraints would not have affected the yield and quality of the grapes. The field capacity at this plot was estimated to be *ca*. 215 mm/90 cm. Lower temperatures, higher rainfall and reduced wind speeds (Figs. 3.4, 3.6 & 3.7), as well as limited grapevine water uptake during the dormant period, likely reduced evapotranspiration during the winter months of May to August. Subsequently, SWC levels remained between the refill zone and estimated field capacity. Therefore, no further irrigations were applied until the end of August when SWC levels reached 130 mm/90 cm. Similar to the LOR1 plot, the higher temperatures, lower rainfall and stronger wind speeds experienced at the LOR2 plot throughout the 2017/18 season necessitated the application of more irrigations at this plot compared to experiment plots in the Coastal and Breede River regions.

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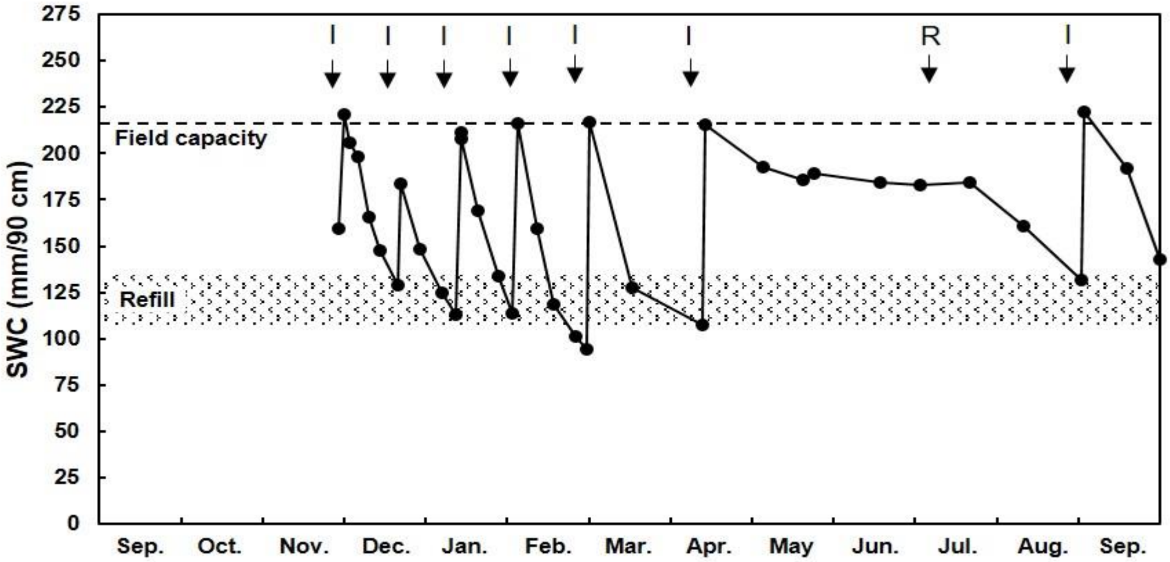

Figure 3.53. Variation in soil water content (SWC) up to 90 cm soil depth during the 2017/18 season at the LOR2 experiment plot where Cabernet Sauvignon grapevines were irrigated *via* the in-field fractional use of winery wastewater with raw water. I is irrigation applications and R is rainfall events.

3.3.6. Grapevine water status

3.3.6.1. Coastal region

The newly established Cabernet Sauvignon grapevines did not experience any water constraints at either of the two plots in the Coastal region during late February and early March 2018 (Fig. 3.54). This corresponded well with SWC levels at both plots which were above the refill zone (Figs. 3.46 & 3.47). Prior to the second irrigation applications at these plots on 26/03/2018, the Ψ_S of the C1 and C2 experiment plots decreased to -1.20 MPa and -0.89 MPa, respectively (Fig. 3.54). According to grapevine water constraint thresholds proposed by Myburgh *et al*. (2016), the grapevines at the C1 and C2 plots experienced high and moderate water constraints, respectively, during this time. Throughout the 2017/18 season the C1 plot consistently experienced greater water constraints compared to the C2 plot. This was probably due to higher SWC levels at the latter plot throughout the study period (Fig. 3.47). Although only two irrigations were applied at the Coastal region plots, the results indicate that irrigation using in-field fractionally applied winery wastewater with raw water did not have a negative effect on the grapevine water status at these plots.

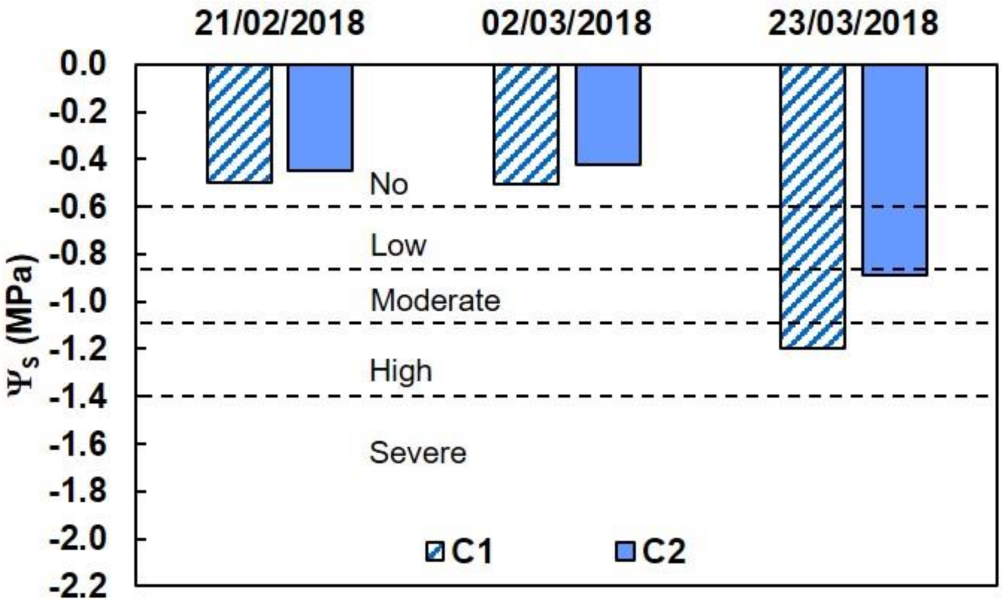

Figure 3.54. Midday stem water potential (Ψ_S) measured during the 2017/18 season at the C1 and C2 experiment plots where newly established Cabernet Sauvignon grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Dashed lines indicate water constraint thresholds for Cabernet Sauvignon as presented by Myburgh *et al*. (2016).

3.3.6.2. Breede River region

During the ripening period of the 2017/18 season (26/02/2018) the grapevines at the BR1 and BR2 experiment plots had Ψ_S of -1.15 MPa and -1.22 MPa, respectively (Fig. 3.55). According to water constraint thresholds for Shiraz grapevines (Myburgh, 2018 and references therein), both experiment plots experienced moderate water constraints during this period. In a previous study on Shiraz grapevines in the Breede River region, Lategan (2011) reported Ψ_S values of -1.0 MPa for grapevines irrigated at 35% plant available water (PAW) depletion (high frequency irrigation) and between -1.55 MPa and -1.9 MPa for less frequently irrigated grapevines. Prior to harvest (01/03/2018), the grapevine Ψ_S at the BR1 and BR2 plots decreased to -1.24 MPa and -1.62 MPa, respectively (Fig. 3.55). Despite the BR2 plot experiencing more water constraints compared to the BR1 plot during this period, the Ψ_S measured at both plots were still within the range of -1.1 MPa to -1.65 MPa which designates moderate water constraints in Shiraz grapevines (Myburgh, 2018 and references therein). The Ψ_S remained relatively unchanged at the BR1 plot after harvest (08/03/2018), whereas Ψ_S decreased to -1.81 MPa at the BR2 plot (Fig. 3.55). The high water constraints observed at the BR2 plot was the result of low SWC levels during the particular time period (Fig. 3.50). In contrast, the relatively stable Ψ_S measured at the BR1 plot was likely due to the slow decrease in SWC at this plot after the subsoil was saturated by the burst water pipe (Fig. 3.49). Due to the complications experienced at these plots throughout the study period, it is at this stage difficult to attribute the responses in grapevine water status to the irrigation strategy.

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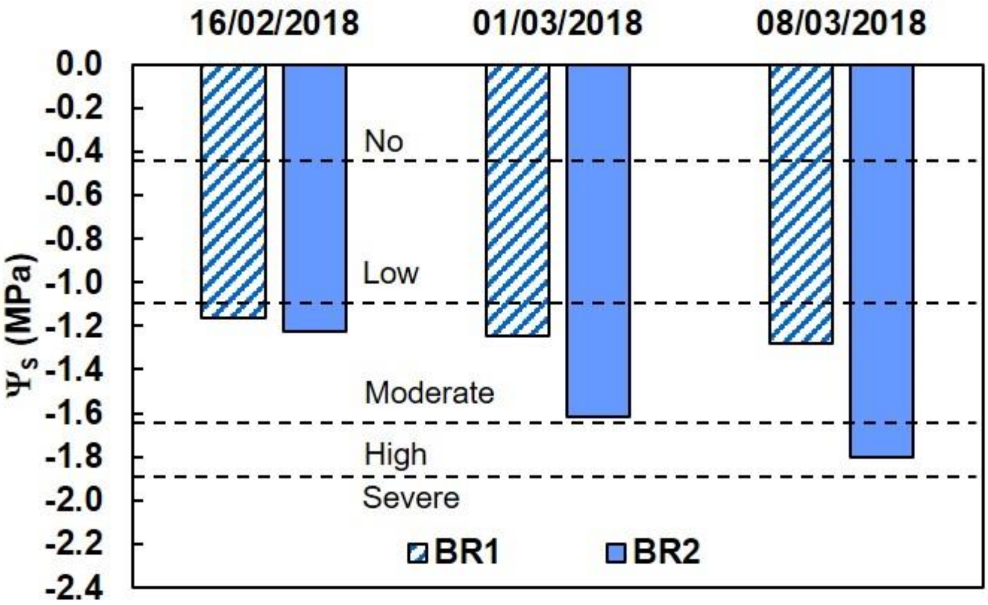

Figure 3.55. Midday stem water potential (Ψ_S) measured during the 2017/18 season at the BR1 and BR2 experiment plots where Shiraz grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Dashed lines indicate water constraint thresholds for Shiraz as presented by Myburgh (2018) and references therein.

3.3.6.3. Lower Olifants River region

According to water constraint thresholds for Shiraz grapevines (Myburgh, 2018 and references therein), the grapevines at the LOR1 experiment plot experienced low water constraints at the pea size berries stage (27/11/2017), at véraison (04/01/2018) and prior to harvest (26/02/2018) (Fig. 3.56). Similar results were reported by Bruwer (2010) for drip irrigated Cabernet Sauvignon grapevines in a sandy soil near Lutzville. The low water constraints were attributed to cooler atmospheric conditions which was likely influenced by the proximity of the vineyards to the Atlantic Ocean. Furthermore, the low water constraints experienced at this plot during the 2017/18 season would probably have resulted in higher yields and lower wine quality (Myburgh *et al.*, 2016). Despite the application of winery wastewater with relatively high salinity (Appendix 4.5), irrigation using in-field fractionally applied winery wastewater with raw water did not negatively effect grapevine water status at the LOR1 experiment plot. These results were concurrent with a previous study which indicated that irrigation with winery wastewater diluted to 3 000 mg/ℓ COD did not affect the water status of Cabernet Sauvignon grapevines in a sandy alluvial soil in the Breede River region (Howell *et al.*, 2016b).

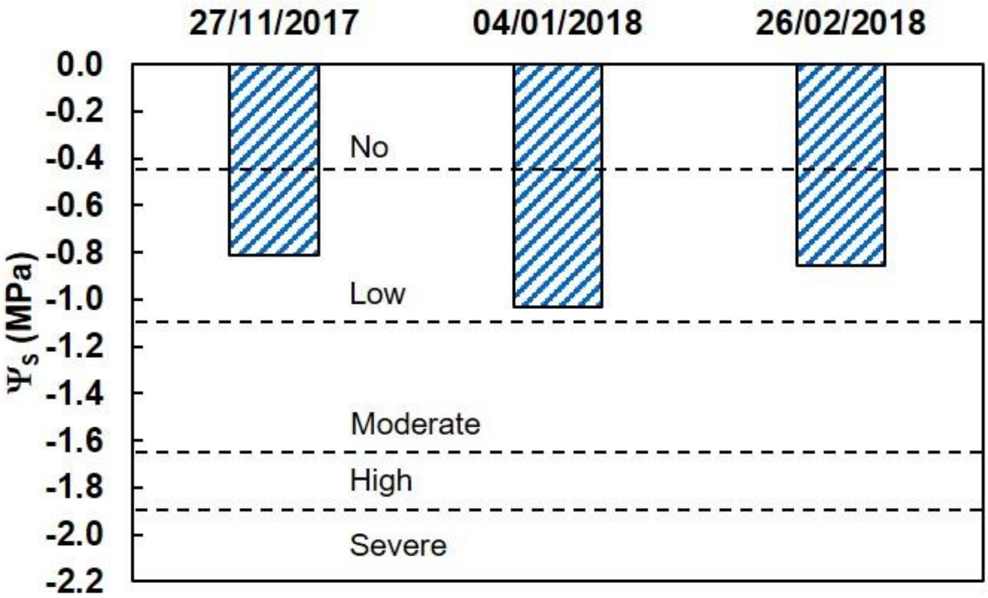

Figure 3.56. Midday stem water potential (Ψ_S) measured during the 2017/18 season at the LOR1 experiment plot where Shiraz grapevines were irrigated *via* in-field fractional use of winery wastewater with raw water. Dashed lines indicate water constraint thresholds for Shiraz as presented by Myburgh (2018) and references therein.

According to water constraint thresholds presented by Myburgh *et al*. (2016), the Cabernet Sauvignon grapevines at the LOR2 plot experienced moderate water constraints during the pea size berries stage (27/11/2017) (Fig. 3.57). Previous research has indicated that water deficits between bud break and véraison can significantly reduce yield (Ferreyra *et al.* 2004). At véraison (04/01/2018) and prior to harvest (14/02/2018) of the 2017/18 season, the grapevines at the LOR2 plot were exposed to water constraints below -1.4 MPa (Fig. 3.57), which is the lower limit that classifies high water constraints in Cabernet Sauvignon grapevines (Myburgh *et al*., 2016). These results were considerably lower than Ψ_S of -0.73 MPa to -1.28 MPa previously reported for Cabernet Sauvignon grapevines in a sandy loam soil near Vredendal (Bruwer, 2010). Given the high levels of water constraints and low yields, high wine quality would be expected at the LOR2 plot (Myburgh *et al*., 2016). To limit yield reductions without compromising wine quality, it is suggested that the grapevines at this plot be irrigated at SWC levels corresponding to a Ψ_S range of -1.2 MPa to -1.4 MPa in the following season. Furthermore, after only one season of irrigation using fractionally applied winery wastewater with raw water, the responses in grapevine water status cannot be attributed to the irrigation water *per se*.

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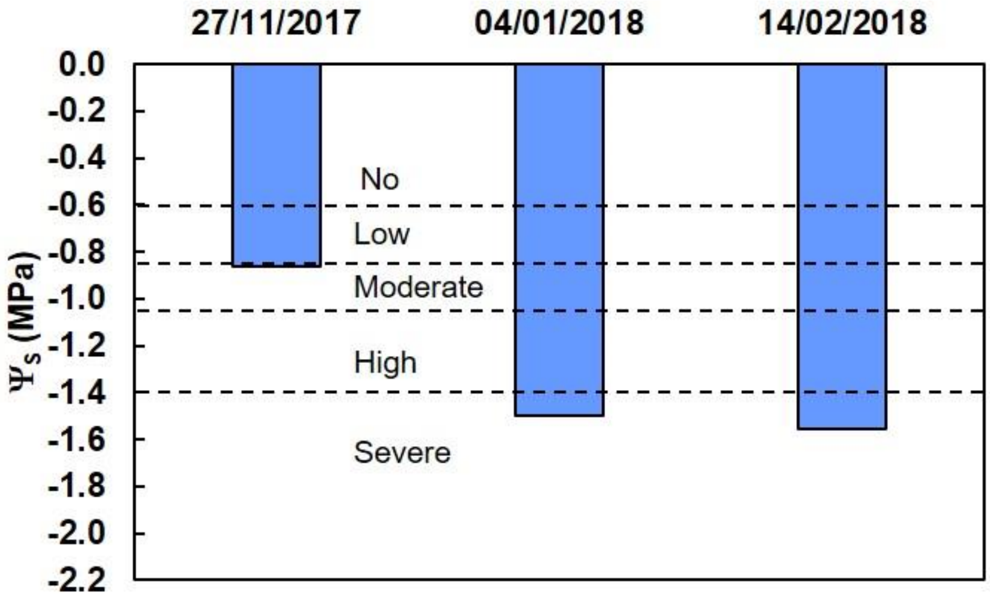

Figure 3.57. Midday stem water potential (Ψ_S) measured during the 2017/18 season at the LOR2 experiment plot where Cabernet Sauvignon grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Dashed lines indicate water constraint thresholds for Cabernet Sauvignon as presented by Myburgh *et al*. (2016).

3.3.7. Vegetative growth

3.3.7.1. Baseline viticultural characteristics

3.3.7.1.1. Coastal region

Since the grapevines at the experiment plots in the Coastal region were only planted in September 2017, no baseline viticultural characteristics were measured at these plots before irrigation commenced.

3.3.7.1.2. Breede River region

The two experiment plots in this region had similar viticultural characteristics before irrigation commenced (Table 3.14). This was expected as the two plots were located within the same vineyard and was therefore subjected to the same management practices prior this study.

Table 3.14. Mean viticultural characteristics of the experiment grapevines in the Breede and Lower Olifants River regions before irrigation of Shiraz and Cabernet Sauvignon grapevines using in-field fractionally applied winery wastewater with raw water commenced.

Cultivar	Plot no. [(1)]	Trunk circumference (cm)	Cordon length (m)	Number of spurs per grapevine
Shiraz	BR1	15.9±1.4	1.7±0.4	10±3
Shiraz	BR2	16.1±2.0	1.7±0.5	9±3
Shiraz	LOR1	11.4±1.5	2.0±0.4	17±3
Cabernet Sauvignon	LOR2	20.7±1.8	1.7±0.3	14±3

(1) Refer to Table 3.2. for description of plot numbers.

3.3.7.1.3. Lower Olifants River region

Since the two experiment plots in this region had different cultivars, differences in viticultural characteristics were expected. The Shiraz grapevines at the LOR1 plot had smaller trunk circumferences, but longer cordons compared to the Cabernet Sauvignon grapevines at the LOR2 plot (Table 3.14). The longer cordons at the former plot resulted in more spurs per grapevine (Table 3.14).

3.3.7.2. Cane mass

3.3.7.2.1. Coastal region

Since the grapevines in the Coastal region were only planted in September 2017, baseline cane mass was not determined at these plots prior to irrigation with winery wastewater (Fig. 3.58). During the dormant period of the 2017/18 season, the newly established grapevines were pruned to two buds. The cane mass was 0.02 kg/grapevine and 0.05 kg/grapevine at the C1 and C2 plots, respectively (Fig. 3.58). The slightly lower cane mass of the C1 plot may be the result of the higher water constraints experienced at this plot throughout the 2017/18 season (Fig. 3.46). However, since no baseline cane mass was determined, and after only one season of irrigation with winery wastewater, the responses in vegetative growth cannot be attributed to the irrigation water.

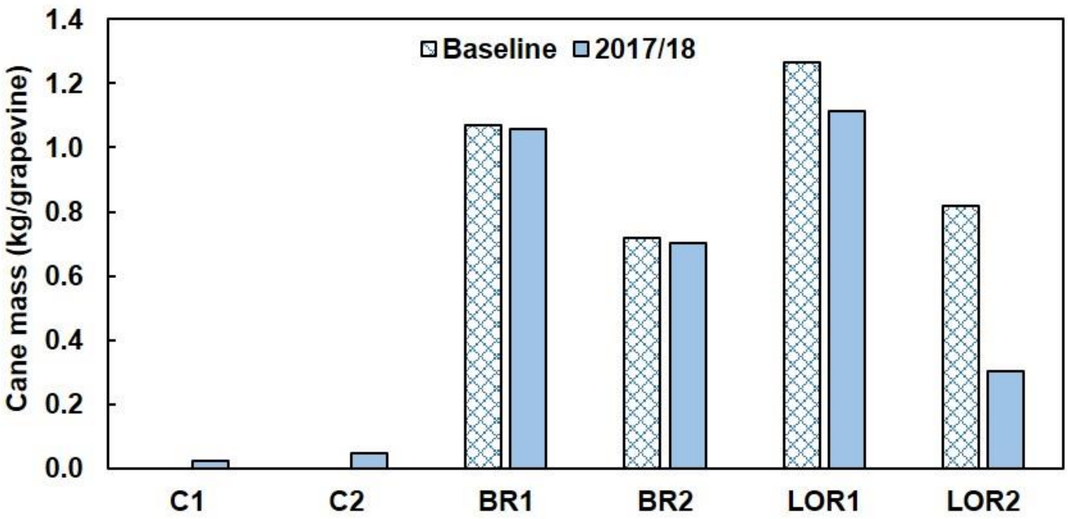

Figure 3.58. Mean cane mass of the experiment grapevines that were irrigated *via* in-field fractional use of winery wastewater with raw water, before irrigation commenced (Baseline) and after irrigation using in-field fractionally applied winery wastewater with raw water was applied during the 2017/18 season. Refer to Table 3.2 for plot number descriptions.

3.3.7.2.2. Breede River region

In contrast to the baseline viticultural characteristics (Table 3.14), the cane mass of the grapevines at the BR1 and BR2 experiment plots differed substantially prior to winery wastewater irrigation (Fig. 3.58). The reason for this difference is still uncertain. However, soil compaction due to tractor traffic is more likely to occur at the BR2 plot due to the heavier soil texture (Appendix 5.2). Therefore, the lower vegetative growth may be a result of restricted root development. Compared to the baseline, the cane mass at both plots in this region remained unchanged after the 2017/18 season (Fig. 3.58). The lack of difference was to be expected since only two irrigations were applied at these plots during the 2017/18 season. In addition, after the irrigations were applied (Figs. 3.12A & 3.12B), the EC_e of the soils were below the range of 0.7 dS/m to 1.5 dS/m which is the proposed salinity threshold for vineyards in the Breede River region (Myburgh & Howell, 2014c). Furthermore, the high water constraints experienced by the grapevines at the BR2 plot did not seem to have a negative effect on grapevine vegetative growth. The cane mass at both plots were comparable to values previously reported for Shiraz in the Breede River regions (Lategan, 2011). Under the prevailing conditions irrigation using in-field fractionally applied winery wastewater with raw water did not pose a salinity hazard to grapevine growth. Similarly, Howell *et al*. (2016b) reported that irrigation using winery wastewater diluted to a COD of up to 3 000 mg/ℓ did not affect the vegetative growth of Cabernet Sauvignon in the Breede River region.

3.3.7.2.3. Lower Olifants River region

The cane mass at the LOR1 plot was 1.27 kg/grapevine before irrigation commenced in the 2017/18 season (Fig. 3.58). This was similar to values reported for sprawling canopy Shiraz/110R grapevines irrigated at 30% PAW depletion in the Breede River region (Lategan & Howell, 2016). The baseline cane mass at the LOR2 plot was 0.82 kg/grapevine (Fig. 3.58). These results were substantially less than values of *ca*. 1.8 kg/grapevine reported for deficit irrigated Cabernet Sauvignon grapevines in a sandy loam soil near Vredendal (Bruwer, 2010). It must be noted that prior to the present study, the grapevines at the LOR2 plot were not strictly pruned to two bud spurs. During the dormant period of the 2016/17 season, the grapevines were severely pruned to obtain more correct grapevine structure. As a result, the baseline cane mass may be inflated compared to conventionally pruned Cabernet Sauvignon. Following one season of irrigation with winery wastewater, cane mass decreased at both experiment plots in this region (Fig. 3.58). However, the decline was more pronounced at the LOR2 plot. The reduction in cane mass was expected since the grapevines at this plot experienced high water constraints during the 2017/18 season (Fig. 3.57). Furthermore, the harsh pruning actions carried out in the previous season would have had an effect on cane mass in the 2017/18 season. Therefore, after only one season of irrigation using in-field fractionally applied winery wastewater with raw water, it is difficult to attribute responses in vegetative growth only to the irrigation water.

3.3.7.3. Leaf and shoot chemical status

Nitrogen: The levels of N in the leaf blades of the grapevines at the experiment plots in the Coastal region (Fig. 3.59A) were above the recommended range of 1.6% to 2.7% for N in grapevine leaves (Conradie, 1994). This was probably the result of the over application of N *via* the irrigation water to the newly established grapevines at these plots (Table 3.8). Furthermore, the grapevine shoot N levels of both of the Breede River, as well as the LOR2 experiment plots (Fig. 3.59B) exceeded the recommended threshold value of 0.9% (Saayman, 1981 and references therein). Since no N

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was applied *via* the irrigation water at the experiment plots in the Breede River region (Table 3.10), the accumulation of N by these grapevines cannot be related to the irrigation water. In contrast, high amounts of N applied *via* the irrigation water (Table 3.12), and poor vegetative growth (Fig. 3.58) at the LOR2 plot may have reduced N metabolization and subsequently resulted in higher shoot N accumulations at pruning.

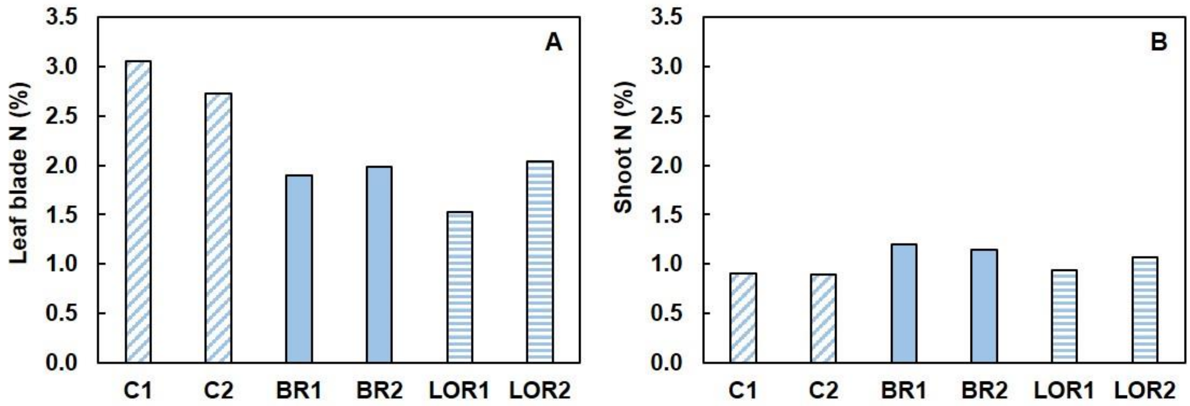

Figure 3.59. Variation in grapevine nitrogen (N) content in (A) leaf blades at harvest and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Phosphorous: Except for the grapevines at the LOR1 plot (Fig. 3.60A), all of the experiment grapevines had P contents within the recommended range of 0.14% to 0.55% for grapevine leaf blades (Conradie, 1994). The shoot P contents of the grapevines at the BR2 and LOR1 plots (Fig. 3.60B) were within the range of 0.05% to 0.15% recommended for grapevines (Saayman, 1981 and references therein). In contrast, the grapevine shoot P contents at all the other experiment plots were above this range (Fig. 3.60B). Furthermore, a trend of increasing leaf blade P content occurred in the case of grapevines planted on the heavier textured soils (Fig. 3.60A). Compared to the lighter textured soils, these soils all had higher plant-available P contents after the main irrigation season (Figs. 3.14, 3.15 & 3.16) and could, therefore, explain the higher leaf blade P levels.

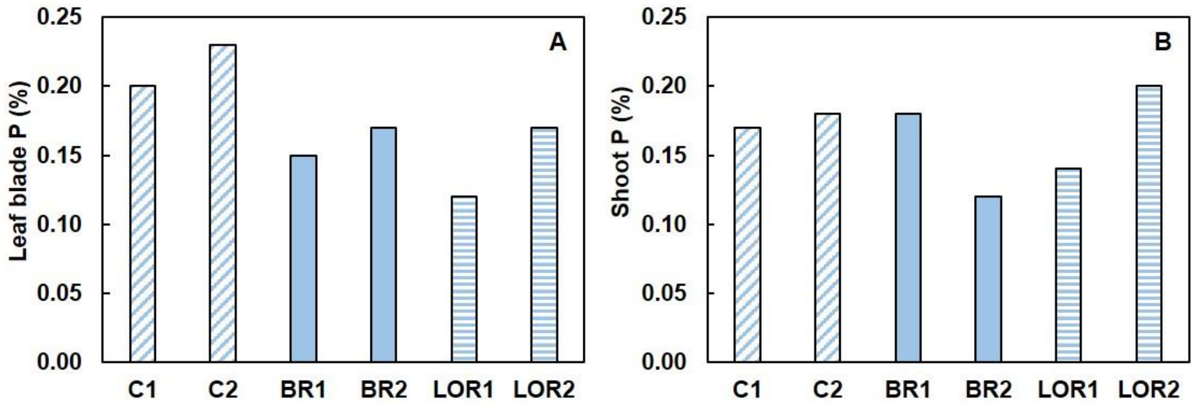

Figure 3.60. Variation in grapevine phosphorous (P) content in (A) leaf blades at harvest and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Potassium: The leaf blade K^+ contents of the grapevines ranged between 0.55% and 1.42% at harvest of the 2017/18 season (Fig 3.61A). Despite the high amounts of K^+ applied (Table 3.10), the experiment grapevines at both plots in the Breede River region had leaf blade K^+ levels below the minimum recommended norm of 0.65% (Conradie, 1994). The majority of K^+ is absorbed by grapevines before the onset of véraison (Conradie, 1981b). Therefore, it is possible that the K^+ applied *via* the irrigation water at these plots, were applied too late in the season to have had an effect on leaf K^+ levels. Similar results were reported by Howell *et al*. (2016a) for Cabernet Sauvignon grapevines irrigated with diluted winery wastewater in the Breede River region. The grapevines at the LOR1 experiment plot had K^+ contents above the maximum recommended norm of 1.3% for grapevine leaf blades (Conradie, 1994). Since the grapevines at this particular plot were irrigated before the onset of véraison, the irrigation water supplied large amounts of K^+ during the period of active K^+ uptake (Table 3.12). This resulted in an accumulation in the leaves. Furthermore, shoot K^+ contents of all the experiment grapevines (Fig. 3.61B) were within the range of 0.4% to 0.7% recommended for grapevine shoots at pruning (Saayman, 1981 and references therein). Potassium uptake can be supressed or enhanced by different rootstocks and clones (Downton, 1977; Wooldridge & Olivier, 2014). Therefore, the selection of appropriate rootstock cultivars may play an important role in ensuring the long-term sustainability of using winery wastewater for grapevine irrigation.

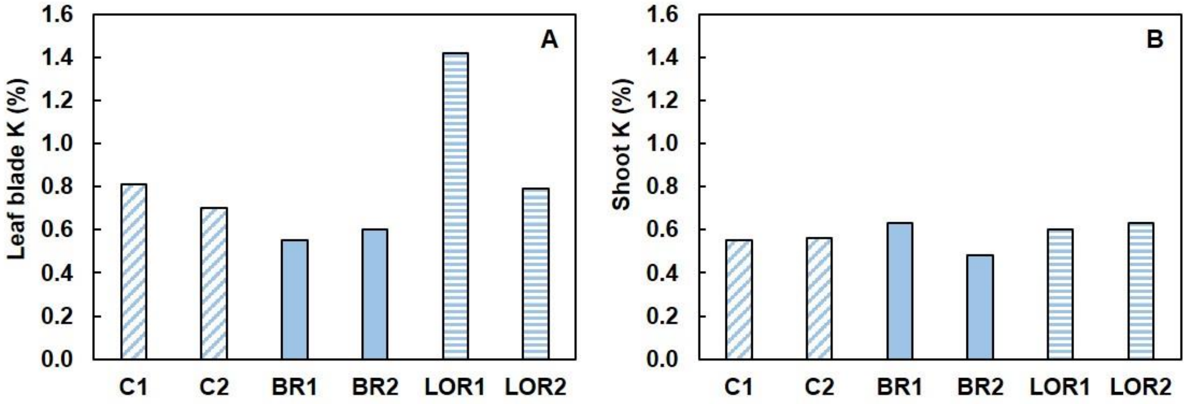

Figure 3.61. Variation in grapevine potassium (K) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Calcium: With the exception of the LOR1 plot, the leaf blade Ca^{2+} levels were relatively similar for all the experiment plots (Fig. 3.62A). The high Ca^{2+} content of the grapevine leaves at the LOR1 plot exceeded the threshold value of 2.2% Ca^{2+} recommended by Conradie (1994). In contrast to other studies (Morris & Cawthon, 1982; Mosse *et al*., 2013; Howell *et al*., 2016a), the excessive application of K^+ at the LOR1 plot (Table 3.12) did not suppress Ca^{2+} uptake. The high leaf blade Ca^{2+} concentrations at the LOR1 plot was probably caused by the substantial amount of Ca^{2+} applied *via* the irrigation water during the 2017/18 season (Table 3.12). Except for the grapevines at the LOR2 experiment plot (Fig. 3.62B), shoot Ca^{2+} contents at all the experiment plots were above the recommended range of 0.3% to 0.6% (Saayman, 1981 and references therein). However, the shoot Ca^{2+} content of the grapevines at the LOR2 plot was within the recommended range.

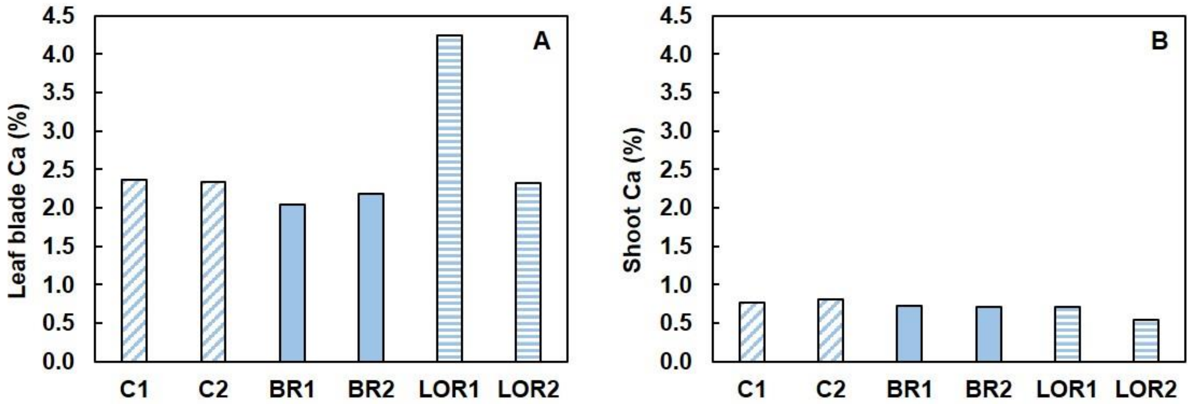

Figure 3.62. Variation in grapevine calcium (Ca) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Magnesium: Apart from the BR1 experiment plot (Fig. 3.63A), grapevines at all the experiment plots had leaf blade Mg^{2+} levels within the recommended range of 0.16% to 0.55% (Conradie,

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1994). It is still uncertain why the grapevines at the BR1 plot had particularly high leaf blade Mg^{2+} contents, since the amount of Mg^{2+} applied *via* the irrigation water was similar for the two plots in the Breede River region (Table 3.10), but appreciably lower compared to amounts applied in the Lower Olifants River region (Table 3.12). However, the grapevines in the Breede River and Lower Olifants River regions accumulated more Mg^{2+} in the shoots compared to grapevines in the Coastal region (Fig. 3.63B). In fact, shoot Mg^{2+} levels in grapevines of the former plots exceeded the maximum concentration of 0.25% recommended for grapevine shoots at pruning (Saayman, 1981 and references therein). This may be a result of the higher amounts of Mg^{2+} applied *via* the irrigation water at these plots (Tables 3.10 & 3.12).

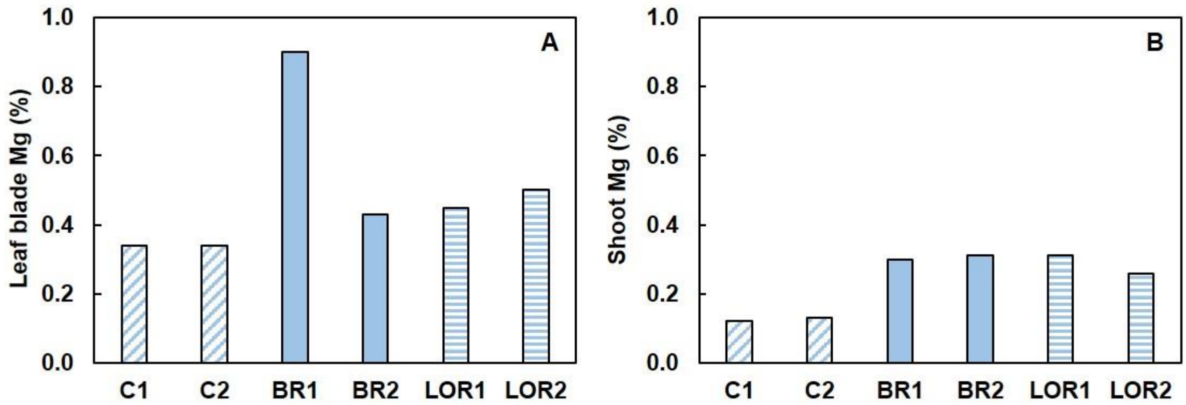

Figure 3.63. Variation in grapevine magnesium (Mg) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Sodium: The leaf blade Na^{+} levels of the grapevines at the BR1, BR2 and LOR1 experiment plots were substantially higher compared to the other experiment plots (Fig. 3.64A). Increased leaf Na^{+} contents with an increase in EC_w was reported for Colombar grapevines in the Breede River region (Moolman *et al*., 1999). The authors also reported a more rapid increase in leaf Na^{+} content above EC_w levels of 3.5 dS/m. Since the winery wastewater applied at the Breede River and LOR1 experiment plots frequently had EC_w values exceeding 3.5 dS/m (Appendices 4.3 & 4.5), the accumulation of Na^{+} in the leaves at the BR1, BR2 and LOR1 plots may be ascribed to the high salinity irrigation water. Furthermore, Na^{+} uptake by grapevines can be influenced by rootstock cultivar (Walker *et al*., 2004). A recent study by Saritha *et al*. (2017) indicated that Ramsey accumulated considerable amounts of Na^{+} in the leaf blades when irrigated with different Cl-salt solutions. Since the Shiraz grapevines at the LOR1 plot was grafted onto Ramsey, the higher Na^{+} accumulation by these grapevines may be explained by higher Na^{+} uptake by the rootstock compared to the other plots. However, the leaf Na^{+} levels at all the experiment plots were still well below the maximum threshold value 0.25% (Conradie, 1994). Moolman *et al*. (1999) reported that leaf damage can occur at Na^{+} levels as low as 0.17%. The leaf Na^{+} contents at all the experiment plots were below this threshold value (Fig. 3.64A), therefore no leaf scorching was expected.

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Furthermore, the grapevine shoot Na^+ levels at all of the experiment plots (Fig. 3.64B) were within the recommended range of 0.02% to 0.5% (Saayman, 1981). Therefore, irrigation using in-field fractionally applied winery wastewater with raw water did not pose a sodicity risk to grapevines under the prevailing conditions. This agrees with previous results reported for Cabernet Sauvignon grapevines irrigated using winery wastewater diluted to 3 000 mg/ℓ COD in the Breede River region (Howell *et al*., 2016a). In contrast, Mosse *et al*. (2013) observed a substantial increase in petiole Na^+ levels of Shiraz grapevines irrigated using Na-based artificial winery wastewater. The long-term impact of irrigation using in-field fractionally applied winery wastewater with raw water on grapevine Na^+ accumulation should, therefore, be further investigated at all the experiment plots.

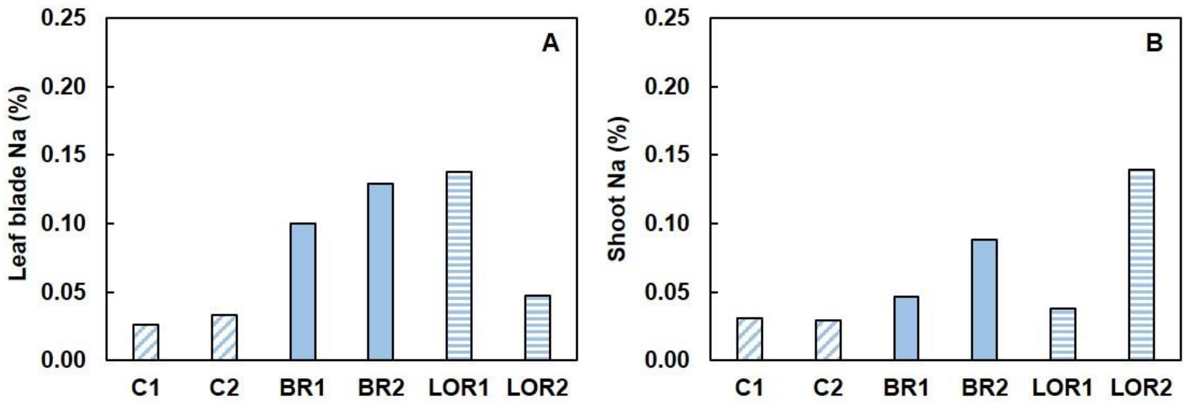

Figure 3.64. Variation in grapevine sodium (Na) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Boron: Levels of B^{3+} in the grapevine leaf blades of the BR1 and LOR1 plots were considerably higher compared to the other experiment plots (Fig. 3.65A). The grapevines at the LOR2 plot exhibited greater B^{3+} accumulations in the shoots (Fig. 3.65B). Although no B^{3+} accumulated in the soils at the experiment plots in the Lower Olifants River region (Figs. 3.35E & 3.35F), the high uptake of B^{3+} at these plots could be the result of high amounts of B^{3+} applied *via* the irrigation water (Table 3.13). Furthermore, the B^{3+} contents of the leaf blades at the BR1 and LOR1 plots were close to the toxicity threshold of 200 mg/kg (Conradie, 1994). The shoots of the LOR2 plot also exceeded the maximum concentration of 20 mg/kg B^{3+} recommended for grapevine shoots at pruning (Saayman, 1981). Therefore, monitoring for B^{3+} toxicity symptoms, such as cupped apical leaves (Fig. 3.66A) and necrotic spots on leaves (Fig. 3.66B) (Saayman, 1981), is recommended for the grapevines of these particular plots.

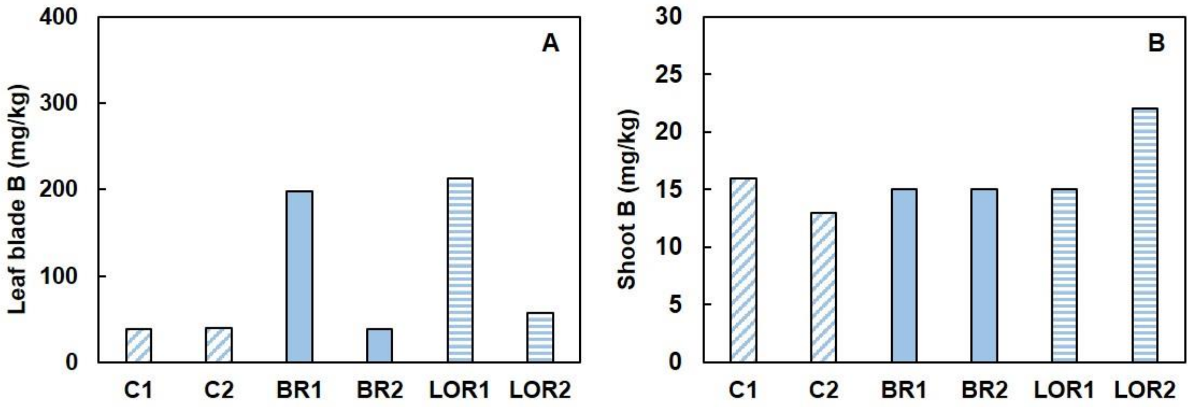

Figure 3.65. Variation in grapevine boron (B) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

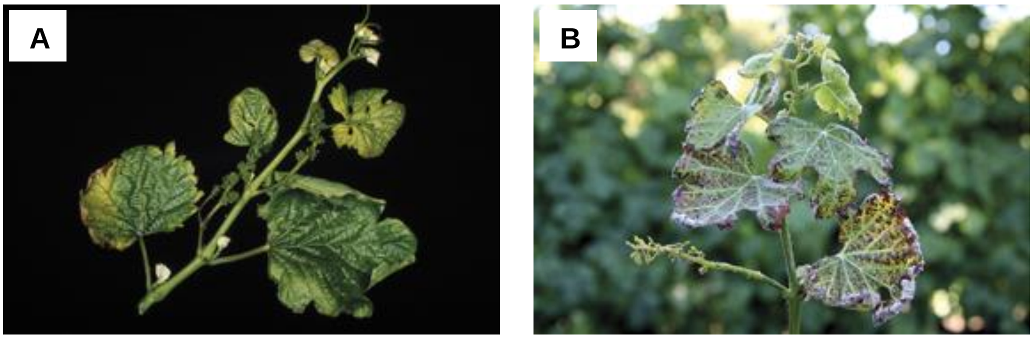

Figure 3.66. Examples of (A) cupped apical leaves and (B) necrotic spots on leaves which are associated with boron toxicities in grapevines. Images obtained from Goussard (2014).

Copper: Levels of Cu^{2+} were highest in the leaves and shoots of the newly established grapevines in the Coastal region (Fig. 3.67A & 3.67B). The grapevines leaf blades at the LOR1 plot had Cu^{2+} concentrations below the recommended minimum norm of 3 mg/kg (Saayman, 1981). Furthermore, Cu^{2+} levels in the grapevine shoots at all the experiment plots were within the recommended range of 5 mg/kg to 15 mg/kg (Saayman, 1981). The Cu^{2+} levels of the leaf blades and shoots could not be related to irrigation using in-field fractionally applied winery wastewater with raw water.

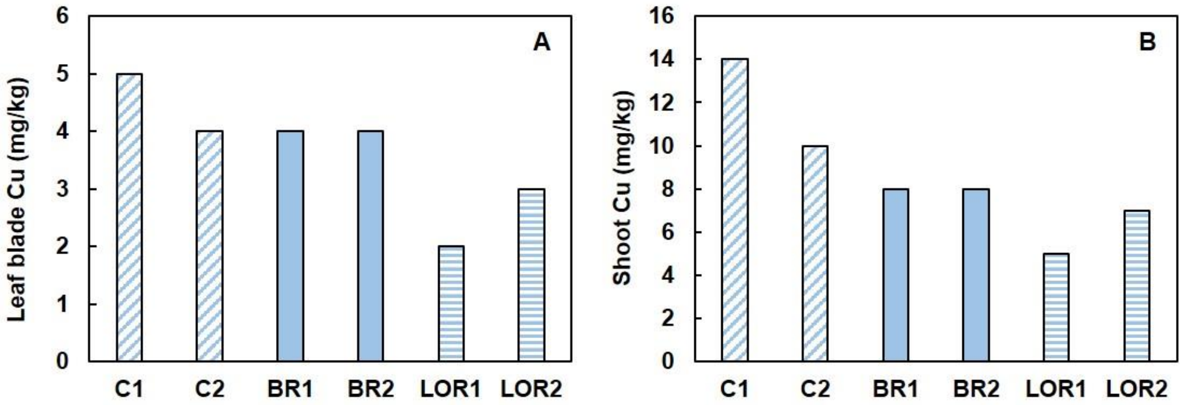

Figure 3.67. Variation in grapevine copper (Cu) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Iron: The grapevines at the BR1 experiment plot had the highest leaf blade Fe^{2+} levels at harvest of the 2017/18 season (Fig. 3.68A). The greater uptake of Fe^{2+} (compared to the BR2 plot) was most likely the result of high Fe^{2+} levels in the subsoil of the BR1 plot (Fig. 3.37C). Furthermore, the leaf blade Fe^{2+} levels at all the experiment plots were above the minimum threshold value of 60 mg/kg (Conradie, 1994). The grapevines in the Coastal region accumulated the highest amount of Fe^{2+} in the shoots (Fig. 3.68B). The accumulation of Fe^{2+} in the grapevine shoots may be a result of the *ca*. 3 kg/ha Fe^{2+} applied *via* the irrigation water at these plots (Table 3.9).

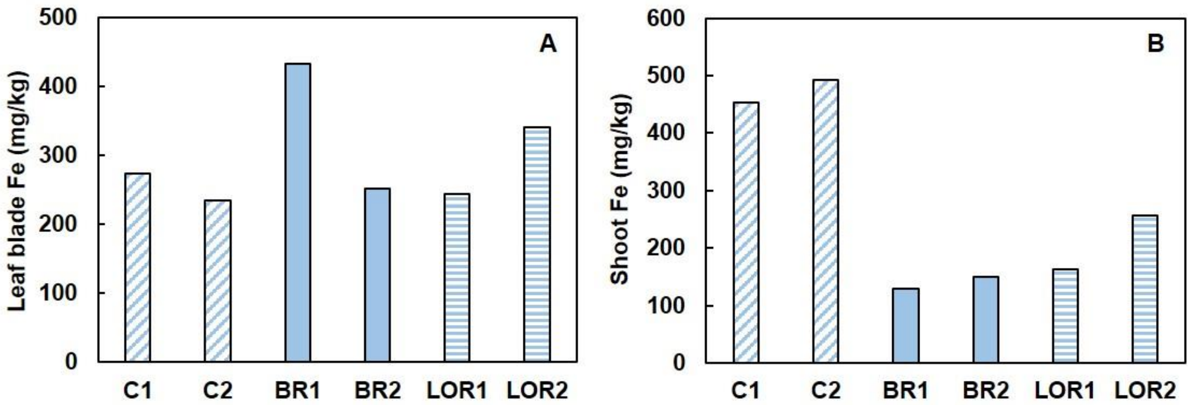

Figure 3.68. Variation in grapevine iron (Fe) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Manganese: With the exception of the grapevines at the LOR1 plot (Fig. 3.69A), the leaf blade Mn^{2+} concentrations at all the experiment plots were within the recommended range of 20 mg/kg to 300 mg/kg (Conradie, 1994). The leaf blade Mn^{2+} levels measured at the LOR1 plot was still well below the toxicity threshold of 750 mg/kg (Conradie, 1994). The grapevines at the LOR2 plot had higher shoot Mn^{2+} levels compared to the other experiment plots (Fig. 3.69B). Relatively high soil Mn^{2+} levels (Fig. 3.38F), as well as the application of nearly 1 kg Mn^{2+} per hectare *via* the irrigation water (Table 3.13), might explain the accumulation in the grapevine shoots at this particular plot.

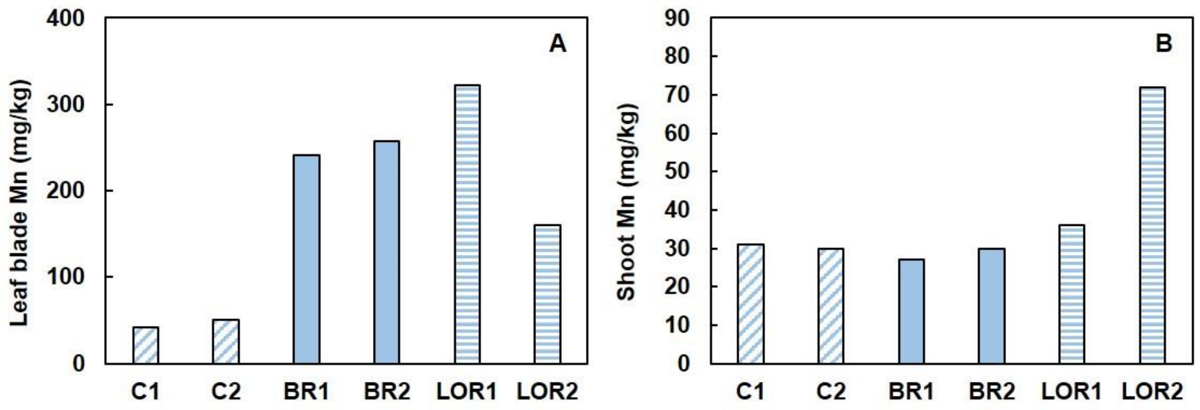

Figure 3.69. Variation in grapevine manganese (Mn) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

Zinc: Levels of Zn^{2+} in the grapevine leaves at the LOR1 plot were substantially higher compared to the other experiment plots (Fig. 3.70A). It is still unclear why the leaf Zn^{2+} levels at this plot was particularly high, since neither the leaf blade nor shoot Zn^{2+} levels could be related to the irrigation water (Appendix 4.6), or high levels of Zn^{2+}in the soil (Fig. 3.39E). Furthermore, the leaf blade Zn^{2+} contents of all the other experiment plots were above the minimum threshold value of 15 mg/kg (Conradie, 1994). The shoot Zn^{2+} levels at all the experiment plots (Fig. 3.50B), were also within the recommended range of 30 mg/kg to 60 mg/kg (Saayman, 1981).

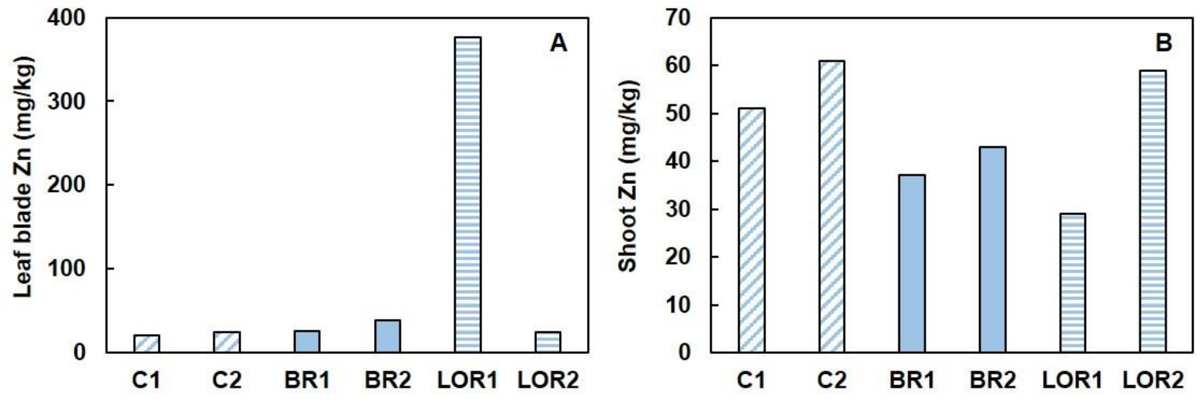

Figure 3.70. Variation in grapevine zinc (Zn) content, in (A) leaf blades at harvest, and (B) shoots at pruning of the 2017/18 season, where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.8. Yield and its components

3.3.8.1. Breede River region

The mean berry mass of the BR1 plot at harvest was 0.6 g more than the BR2 plot (Fig. 3.71A). This had a pronounced effect on bunch mass (Fig. 3.71B), and ultimately on yield (Fig. 3.71D). Grapevine fertility, *i.e.* bunches per grapevine, were similar for both plots (Fig. 3.71C). The berry mass at the BR2 plot was similar to values previously reported for Shiraz grapevines exposed to moderate water constraints in the Breede River region (Lategan, 2011). In contrast, the BR1 plot had greater berry mass than was reported for Shiraz grapevines which received frequent irrigation in the Breede River region and experienced similar water constraints (Lategan, 2011; Stolk, 2014). Although the grapevines at the BR1 plot also experienced moderate water constraints (Fig. 3.55), the high SWC maintained at this plot throughout the ripening period (Fig. 3.49), may have enhanced the post-véraison cell enlargement stage of berry development (Matthew & Anderson, 1989). This could possibly explain the higher berry mass and yield observed at the BR1 plot. Furthermore, the yield at the BR2 plot was similar to values reported for Shiraz in the Breede River region irrigated using continuous deficit irrigation (CDI) (Lategan, 2011) and at 90% PAW depletion (Stolk, 2014). The yield at the BR1 plot was similar to values reported for Shiraz irrigated at 35% PAW depletion (Lategan, 2011). Since only two irrigations were applied at these plots during the 2017/18 season (one of which was applied after harvest), the results in terms of yield responses are insufficient to identify trends that can be related to the irrigations applications. However, as neither of the two plots had abnormally low yields after one irrigation application, it can be assumed that irrigation using in-field fractionally applied winery wastewater with raw water did not negatively affect yield at these plots under the prevailing conditions. Howell *et al*. (2016b) reported that irrigation using winery wastewater diluted to 3 000 mg/ℓ COD did not affect the yield of Cabernet Sauvignon grapevines in the Breede River region compared to a river water control.

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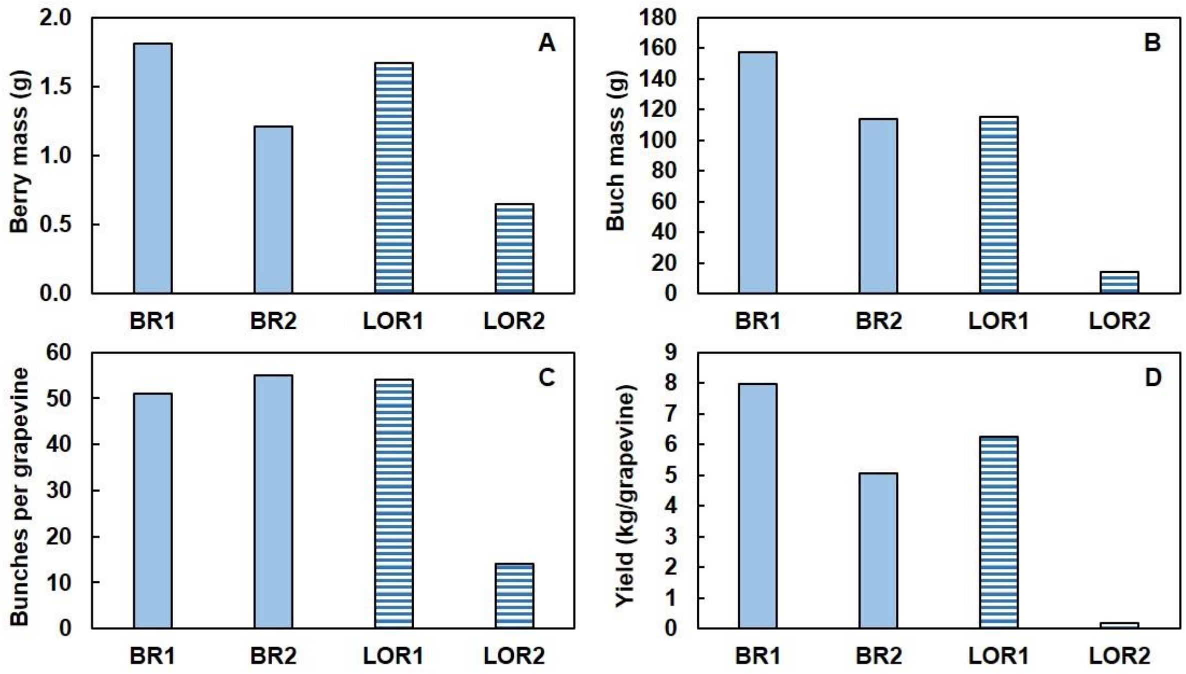

Figure 3.71. Variation in (A) berry mass, (B) bunch mass, (C) bunches per grapevine and (D) yield at harvest during the 2017/18 season where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water. Refer to Table 3.2 for description of plot numbers.

3.3.8.2. Lower Olifants River region

The mean berry mass at the LOR1 plot was comparable to the BR1 plot (Fig. 3.71A), although the bunch mass was smaller (Fig. 3.71B). The relatively large berries and greater number of bunches per grapevine (Fig. 3.71C) reflected in yield (Fig. 3.71D). In a previous study on Shiraz grapevines near Lutzville, Myburgh (2011c) reported smaller berries, but similar bunch mass compared to the present study. Yield at the LOR1 plot was higher than values reported in the abovementioned study, which may be due to higher fertility (Fig. 3.71C), although bunches per grapevine were not reported by Myburgh (2011c). It should be noted that the grapevines at the LOR1 plot experienced less water constraints compared to the abovementioned study (Myburgh, 2011b). Furthermore, previous research on Cabernet Sauvignon indicated that maximum yields were obtained when grapevines were exposed to low water constraints (Myburgh *et al*., 2016), as was the case for the LOR1 plot (Fig. 3.56). These results indicate that after one irrigation season, the in-field fractional use of winery wastewater with raw water did not negatively affect yield of Shiraz grapevines under the prevailing conditions.

A combination of small berries (Fig. 3.71A), small bunches (Fig. 3.71B) and low fertility (Fig. 3.71C) resulted in very low yield at the LOR2 plot during the 2017/18 season (Fig. 3.71B). Yield and yield components measured at this plot were substantially lower than values previously reported for irrigated Cabernet Sauvignon grapevines near Vredendal (Bruwer, 2010). A significant reduction in grapevine fertility and yield due irrigation with saline water was observed for Colombar grapevines in the Breede River region (De Clercq *et al*., 2001). At every irrigation application

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during the 2017/18 season, the EC_w of the irrigation water applied at the LOR2 plot (Appendix 4.7) far exceeded the threshold value of 0.75 dS/m recommended for water used to irrigate grapevines (Van Zyl, 1981). Therefore, the low yield at the LOR2 plot may be a result of the highly saline irrigation water. However, the grapevines at this particular plot were also exposed to severe water constraints during the ripening period (Fig. 3.57). Water deficits after véraison can significantly reduce yield *via* reduced berry mass (Hardie & Considine, 1976). Harsh and unfavourable atmospheric conditions during bunch initiation of the previous season may also have impacted fertility in the current season (Guilpart *et al*., 2014). Furthermore, high air temperatures and low RH experienced at this plot during flowering (October-November) could have negatively affected bloom and reduced fruit set (Vasconcelas *et al*., 2009). The harsh atmospheric conditions of the 2017/18 and preceding seasons resulted in significant yield losses throughout the Olifants River region (Vinpro, 2018). Due to these factors, the low yield obtained at the LOR2 plot cannot be ascribed to the irrigation applications alone.

3.3.9. Grape juice characteristics

3.3.9.1. Total soluble solids

3.3.9.1.1. Breede River region

Due to logistical reasons, the grapes at the two experiment plots in this region were harvested on the same day, *i.e.* on 3 March. The aim was to harvest the grapes as close to 24°B as logistically possible. At harvest, the TSS of the BR1 and BR2 plots were 23.3°B and 26.7°B, respectively (Fig. 3.72). The higher TSS at the BR2 plot may be the result of more severe water constraints experienced at this plot (Fig. 3.55). Shiraz grapes that were exposed to high water constraints in the Breede River region reached the target TSS of 24°B earlier than grapevines exposed to less water constraints (Lategan, 2011; Stolk, 2014). Jackson and Lombard (1993) reported that excessive water availability and actively growing shoots during ripening can delay berry ripening, whereas controlled water deficits may expedite TSS accumulation and berry ripening. Furthermore, Schoeman (2012) reported that irrigation of Cabernet Sauvignon with winery wastewater diluted to different levels of COD did not affect berry sugar loading compared to grapevines irrigated with river water.

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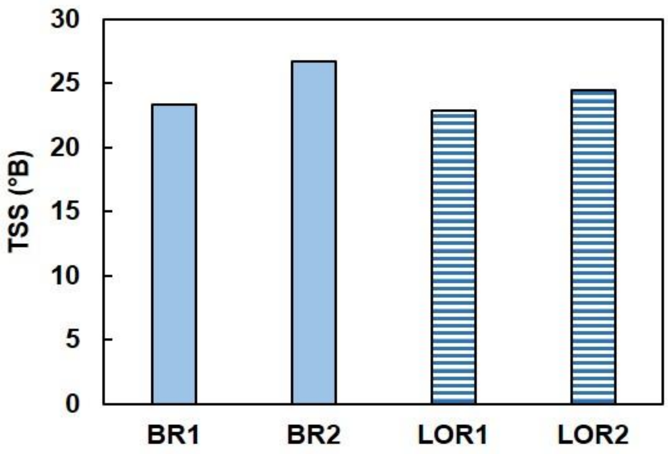

Figure 3.72. Variation in juice total soluble solids (TSS) at harvest of the 2017/18 season where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water in the Breede River and Lower Olifants River regions. Refer to Table 3.2 for description of plot numbers.

3.3.9.1.2. Lower Olifants River region

Due the difference in cultivar, the grapes at the LOR1 experiment plot was harvested *ca*. two weeks later than the LOR2 plot. The grapes at the LOR2 plot were harvested on 14 February when the TSS reached 24.5°B (Fig. 3.72). The TSS measured at the LOR2 plot was similar to values reported for Cabernet Sauvignon in a sandy loam soil near Vredendal (Bruwer, 2010). However, the harvest date was three to four weeks earlier than the abovementioned study. Warm atmospheric conditions (Fig. 3.4) and high water constraints (Fig. 3.57) throughout the ripening period of the 2017/18 season may have enhanced TSS accumulation and ripening (Bruwer, 2010; Lategan, 2011). Berry sugar accumulation seemed to be slower at the LOR1 plot. Due to fear of the grower harvesting the experiment grapevines, the grapes were harvested on 27 February when the TSS was only 22.9°B instead of the recommended 24°B (Fig. 3.72). In a previous study in the Lower Olifants River region, Bruwer (2010) reported slower berry sugar accumulation in vineyards closer to the ocean compared to ones further inland. This phenomenon was attributed to cooler atmospheric conditions closer to the Atlantic Ocean. High yields at the LOR1 plot may also have retarded ripening due to sink competition for water and nutrients (Kliewer & Dokoozlian, 2005).

3.3.9.2. Total titratable acidity

3.3.9.2.1. Breede River region

Higher TSS at the BR2 plot resulted in lower TTA (Fig. 3.73). These results contradicted findings previously reported for Shiraz in the Breede River region. Lategan (2011), as well as Stolk (2014) reported higher TTA values for grapevines that experienced greater water constraints. In contrast, Mehmel (2010) reported reduced TTA in rainfed Cabernet Sauvignon grapevines near Wellington compared to irrigated treatments. The lower acidity levels were attributed to warm atmospheric conditions and subsequent water constraints. Although both the BR1 and BR2 plots experienced

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moderate water constraints prior to harvest (Fig. 3.55), the midday Ψ_S at the BR2 plot was considerably lower than the BR1 plot which would help to explain the difference in TTA. Higher microclimate temperatures enhance the respiration of malate, which gives rise to lower TTA (Iland, 1989a). It is also known that excessive shading in the bunch zone during ripening can decrease malate respiration and result in higher TTA (Iland, 1989b). Therefore, the difference in TTA of the grapes from the plots in this region may partially be explained by stronger vegetative growth at the BR1 plot (Fig. 3.58), which would have created more shading in the bunch zone and subsequently increased TTA.

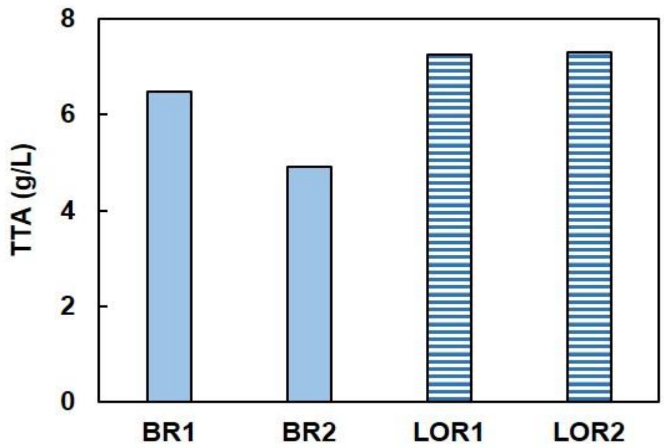

Figure 3.73. Variation in juice total titratable acidity (TTA) at harvest of the 2017/18 season where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water in the Breede River and Lower Olifants River regions. Refer to Table 3.2 for description of plot numbers.

3.3.9.2.2. Lower Olifants River region

The two experiment plots in this region had similar TTA values at harvest of the 2017/18 season (Fig. 3.73). Juice TTA measured at the LOR2 plot was similar to values reported by Bruwer (2010) for Cabernet Sauvignon grapevines in the Lower Olifants River region which were harvested at the same ripeness level. In contrast, the TTA measured at the LOR1 plot was significantly higher than values reported for Shiraz grapevines near Lutzville that were drip irrigated at low and high frequencies (Myburgh, 2011c). It should be noted that the grapevines at the LOR1 plot were harvested before the desired maturity (Fig. 3.72). Therefore, the higher TTA values at this plot was probably a result of the shorter ripening period. The TTA at both plots in this region were surprisingly high given the warm atmospheric conditions (Fig. 3.4). Respiration reduces malic acid, therefore grapes produced in warmer climates typically have lower acidity levels compared to grapes produced in cooler climates (Winkler, 1974). However, high temperatures during stage I (berry formation) and II (lag phase) of berry development may increase malic acid concentrations in berries (Jackson & Lombard, 1993 and references therein). This could help to explain the higher TTA values at the plots in this region, since the grapevines at the LOR1 and LOR2 experiment plots were exposed to higher temperatures during October to December compared to the plots in the Breede River region (Fig. 3.4).

3.3.9.3. pH

3.3.9.3.1. Breede River region

Despite the difference in TTA, there was no difference in juice pH between the BR1 and BR2 plots (Fig. 3.74). Juice pH was similar to values previously reported for Shiraz in the Breede River region (Lategan, 2011; Stolk, 2014). The relatively high pH is of concern since it exceeded the threshold value of 3.8 at which wine colour and stability might be affected (Kodur, 2011). Musts and wines with high pH are more susceptible to oxidative and microbial spoilage (Jackson & Lombard, 1993; Conde *et al*., 2007). In a study where Cabernet Sauvignon grapevines were irrigated using diluted winery wastewater, Howell *et al*. (2016a) reported that juice pH increased with a decrease in level of dilution. The increase in pH was linearly related to the amounts of K^+ applied *via* the irrigation water. The high juice pH observed at the BR1 and BR2 plots could not be related to amounts of K^+ applied *via* the irrigation water (data not shown). Furthermore, irrigation using different solutions of artificial winery wastewater did not affect the juice TSS, TTA or pH of Shiraz grapes compared to a fresh water control (Mosse *et al*., 2013). Since only one irrigation was applied at the plots in the Breede River region prior to harvest, the responses in juice characteristics were probably not related to the quality of the irrigation water.

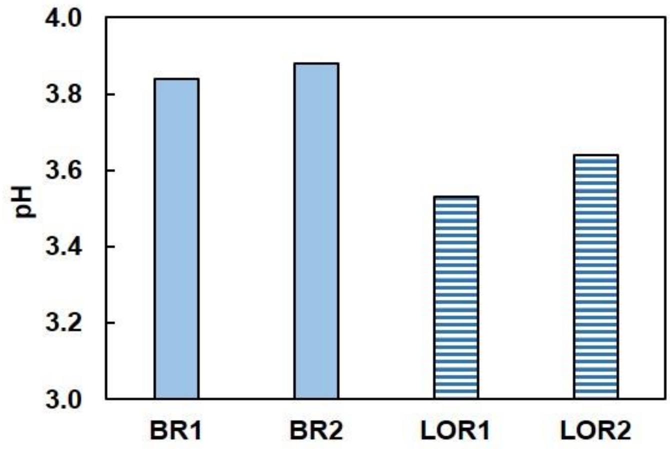

Figure 3.74. Variation in juice pH at harvest of the 2017/18 season where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water in the Breede River and Lower Olifants River regions. Refer to Table 3.2 for description of plot numbers.

3.3.9.3.2. Lower Olifants River region

The high TTA resulted in lower juice pH at these plots compared to the plots in the Breede River region (Fig. 3.74). Since pH measures "active" acidity, pH will decrease with an increase in acidity (Bruwer, 2010). Juice pH at the LOR1 plot was lower than values of 4 and higher previously reported for Shiraz near Lutzville (Myburgh, 2011c). Whereas, the LOR2 plot had similar juice pH values to what was reported by Bruwer (2010) for Cabernet Sauvignon in a sandy loam soil near Vredendal. It is known that factors which limit photosynthesis, such as high temperatures, may result in enhanced transport of K^+ from leaves to berries (Iland, 1989a), which in turn can increase juice pH (Jackson & Lombard, 1993). Despite the high temperatures that occurred in this region during the ripening period (Fig. 3.4), and the high amounts of K^+ applied *via* the irrigation water (Table 3.12), irrigation using in-field fractionally applied winery wastewater with raw water did not negatively affect juice pH under the prevailing conditions. These results contradicted findings previously reported for Cabernet Sauvignon grapevines irrigated using different levels of diluted winery wastewater in the Breede River region (Howell *et al*., 2016a). The authors reported an increase in juice pH with decreasing level of dilution. The reduction was linearly related to the amount of K^+ applied *via* the winery wastewater and the mean juice K^+ (Howell *et al*., 2016a). It should be noted that the diluted winery wastewater in the abovementioned study contained far less Ca^{2+} and Mg^{2+} compared to the irrigation water applied at the experiment plots in the Lower Olifants River region (Howell *et al*., 2016a). Therefore, the high juice TTA at the plots in this particular region may be a result of the high amounts of Ca^{2+} and Mg^{2+} applied *via* the irrigation water (Table 3.12). Engelbrecht and Saayman (2007) reported a reduction in juice pH of Cabernet franc grapevines as a result of Ca^{2+} and Mg^{2+} fertilisation. Although the mechanism is not quite understood, they ascribed the reduction in pH to the enhanced synthesis or retention of certain acids due to higher Ca^{2+} and Mg^{2+} uptake.

3.4. CONCLUSIONS

Compared to the Coastal and Breede River regions, the atmospheric conditions in the Lower Olifants River region necessitated the application of more irrigations throughout the 2017/18 season. Furthermore, the large co-operative wineries in the Lower Olifants River region had winery wastewater available throughout the entire season. Therefore, irrigation using in-field fractionally applied winery wastewater with raw water could be applied near these wineries as a standard practice throughout the year. In contrast, the smaller wineries, *i.e.* in the Coastal and Breede River regions, only had wastewater available for irrigation during harvest. This presents a possible limitation to the use of in-field fractionally applied winery wastewater for irrigation at smaller wineries. Since vineyards often require irrigation during flowering (November), there may not be winery wastewater available for irrigation. However, the availability of winery wastewater for supplementary irrigation during the harvest period may still help to relieve pressure on fresh water resources during the particular time.

The quality of winery wastewater varied greatly between wineries, as well as over the course of the study period. The pH of the wastewater from the wineries in the Coastal and Breede River regions tended to be slightly acidic. In contrast, the addition of $CaCO_3$ to the wastewater by the wineries in the Lower Olifants River region resulted in near-neutral pH values. Application of wastewaters from these wineries to soils increased the soil $pH_{(KCl)}$ beyond the pH of the applied water. This was likely a due to decarboxylation/hydrolyses of organic acids and HCO_3^- applied to the soils *via* the irrigation water.

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With regard to EC_w, wastewater from two of the wineries did not comply with the legislative limits for wastewater reuse for irrigation. In addition, the EC_w of the wastewater from all the wineries were above the recommended norm for grapevine irrigation. However, the soil EC_e of the experiment plots were largely unaffected at the end of the study period. It is possible that the immediate application of raw water following winery wastewater irrigation, was able to dilute the wastewater and leach substantial amounts of salts from the soil profile.

As the season progressed, the COD concentrations in the wastewater from the wineries in the Lower Olifants River region tended to increase. Furthermore, the COD concentrations in the wastewater from all the wineries were above the legislated limit to irrigate 500 m^3 of wastewater per day. However, no accumulation of SOC was observed in the soils of any of the experiment plots at the end of the 2017/18 season. It is hypothesised that sufficient time passed between irrigation applications to allow adequate soil aeration and organic matter decomposed in the process.

The SAR of the wastewater from all the wineries were below the legislated limits for wastewater re-use for irrigation in South Africa. Higher Na^+ concentrations were present in the wastewater produced by the large, co-operative wineries in the Lower Olifants River region. This was likely due to their use of Na-based cleaning agents, as opposed to K-based products. Subsequently, these wineries had higher PAR levels in their wastewaters. However, the addition of $CaCO_3$ to their wastewater effectively reduced the SAR and PAR.

The amounts of plant nutrients present in the winery wastewater was also variable. Wastewater from the wineries in the Coastal and Lower Olifants River regions had high N concentrations and was able to supply grapevines with sufficient N to sustain growth. Whereas the winery wastewater from the Breede River region did not contain any form of N. Generally, all of the winery wastewaters had high P concentrations and was able so supply sufficient P to meet grapevine requirements. However, algal blooms in wastewater storage facilities and bio-fouling of irrigation equipment, may pose potential risks when re-using winery wastewater for irrigation. Grape less, spillage from wine stabilisation processes and the use of K-based cleaning agents contributed to high K^+ concentrations in the wastewater from all four wineries. As a result, excessive amounts of K^+ was supplied to grapevines *via* the irrigation water. However, high K^+ uptake by grapevines was only observed at one of the experiment plots. Furthermore, the high amounts of K^+ applied did not have adverse effects on grape juice quality at harvest.

The high K^+ concentrations in the winery wastewaters, as well as the application of K-fertiliser by the growers, increased the soil EPP´ of the experiment plots. This phenomenon was exacerbated in the soils of the Lower Olifants River region experiment plots, where the presence of mica minerals may have been a contributing factor. At the majority of the experiment plots, it seems the elevated soil EPP´ levels had a positive effect on the soils' hydraulic properties. Even though the

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IR of the soils at the lighter textured experiment plots decreased after irrigation applications, it was restored to values comparable to or even higher than the baseline after the winter rainfall period. This suggests that, in these regions, the winter rainfall during the 2017/18 season was sufficient to leach excessive salts, which may have accumulated at the soil surface, to deeper soil layers. By the end of the study period, only the sandy clay loam plot in the Breede River region exhibited a severe reduction in IR. Therefore, surface run-off may become problematic if this particular irrigation strategy is implemented on heavy textured soils in the Breede River region. It is therefore recommended that the dilution ratio at this experiment plot be adjusted to 1:2 (winery wastewater to raw water).

Due to the unavailability of winery wastewater and problems with the irrigation infrastructure at some of the wineries, grapevines at some of the experiment plots were subjected to higher water constraints than recommended. In practical situations, high water constraints can be avoided by only applying raw water when winery wastewater is unavailable. Furthermore, it was concluded that the Cabernet Sauvignon grapevines at the LOR2 experiment plot should be irrigated at a higher range of Ψ_S values, since this particular plot was exposed to severe water constraints throughout the 2017/18 season. Subsequently, the grapevines at this plot had reduced vegetative growth and yields. In contrast, the cane mass and yields of the Shiraz grapevines at LOR1 and Breede River experiment plots were largely unaffected at the end of the 2017/18 season.

Following one season of irrigation using in-field fractionally applied winery wastewater with raw water, higher grapevine water constraints and subsequent effects on grapevine growth, cannot be ascribed to the irrigation strategy *per se*. Therefore, further research, over a longer study period, is needed to thoroughly assess the sustainability of the particular irrigation strategy for grapevine irrigation under different climatic conditions.

3.5. RECOMMENDATIONS

The availability of winery wastewater during the harvest period may be a beneficial source of alternative irrigation water, particularly in times of drought. The present study indicated that irrigation using fractionally applied winery wastewater with raw water did not have negative effects on vegetative growth, yield and juice quality of grapevines over the short-term. However, under certain environmental conditions soil hydraulic properties may be adversely affected. Therefore, regular analysis of irrigation water, soils and grapevine leaves are recommended to ensure that chemical parameters conform to recommended thresholds and norms. This could help to prevent irreversible damage to irrigation equipment, soils and grapevines. Results from the present study indicated that irrigation using winery wastewater was able to supply grapevines with essential nutrients to sustain crop growth. However, the amounts of elements applied can vary greatly between wineries and over time. It is therefore recommended to use an integrated fertiliser program by adjusting fertiliser applications according to the amount of nutrients present in the

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irrigation water. In addition, soil and plant water status should be monitored routinely to avoid over-irrigation, and to minimise the application of salts *via* the irrigation water. Furthermore, sufficient time should be allowed to pass between irrigation applications to promote soil aeration and the subsequent breakdown of organic material applied *via* the irrigation water.

CHAPTER 4: GENERAL CONCLUSIONS AND RESEARCH RECOMMENDATIONS

4.1. CONCLUSIONS

4.1.1. Long-term effects of irrigation with treated municipal wastewater on soils and grapevines in the Coastal region

The treated municipal wastewater used for irrigation was of an acceptable quality and complied with the legislative requirements to irrigate up to 500 m^3 of wastewater per day. The $pH_{(KCl)}$ and electrical conductivity (EC_e) of the topsoil increased considerably following 11 years of treated municipal wastewater irrigation. Furthermore, the accumulation of salts at the soil surface resulted in a decrease of the topsoil near-saturation hydraulic conductivity (K_{ns}). Subsequently, problems with infiltration during heavy rainfall events are expected. Generally, the municipal wastewater had low nitrogen (N) concentrations and was not able to supply sufficient amounts of N to meet annual grapevine requirements, not even where double the normal amount of irrigation water was applied. In contrast, where double the normal amount of irrigation water was applied, the wastewater contained sufficient amounts of phosphorous (P) to supply annual grapevine requirements. In fact, the P concentrations were alarmingly high and may lead to the development of algal blooms in wastewater storage facilities and bio-fouling of irrigation equipment. Substantial amounts of sodium (Na^+), chloride (Cl^-) and potassium (K^+) accumulated in the soil after 11 years of treated municipal wastewater irrigation. However, it did not result in excessive uptake by grapevines and did not have adverse effects on vegetative growth, yields or grape juice characteristics. It was concluded that irrigation using treated municipal wastewater can increase grapevine productivity in regions where grapevines are usually grown under dryland conditions. At the same time, good fruit quality can be maintained.

4.1.2. Short-term effects of irrigation using in-field fractionally applied winery wastewater with raw water on selected soils and grapevines under different climates

The quality of winery wastewater varied greatly between wineries, as well as over the course of the season. The smaller wineries, *i.e.* in the Coastal and Breede River regions produced wastewater which was slightly acidic. In contrast, the large co-operative wineries in the Lower Olifants River region produced wastewater with pH values that were near neutral due to the addition of lime. The application of wastewaters from these wineries to soils increased the soil $pH_{(KCl)}$ to levels beyond the pH of the applied irrigation water. This was ascribed to the decarboxylation/hydrolysis of organic acids and bicarbonate ions which were added to the soil *via* the irrigation water. The electrical conductivity of the wastewater (EC_w) produced by all four wineries were above the recommended norm for water used to irrigate grapevines. Furthermore, the EC_w of the wastewater produced by two of the wineries did not comply with the minimum criteria stipulated by the General Authorisations for irrigating up to 500 m^3 of wastewater per day. Despite this, soil EC_e at all the experiment plots remained at levels comparable to the baseline at the end of the study period. Furthermore, high chemical oxygen demand (COD) concentrations in the winery wastewater did not result in increased soil organic carbon (SOC) levels throughout the 2017/18 season. This was likely due to adequate time passing between irrigation applications which promoted soil aeration and allowed sufficient time for organic matter to breakdown. The use of K-based cleaning agents by the wineries resulted in wastewaters with relatively low SAR values.

Plant nutrients, such as N and P, was applied at varying amounts *via* the irrigation water due to the variability of these nutrients in the winery wastewater. However, the winery wastewaters all had fairly high K^+ concentrations due to the use of K-based cleaning agents, as well as grape lees and spillage from wine stabilisation processes. The application of excessive amounts of K^+ *via* the irrigation water did not result in enhanced uptake by grapevines, but an accumulation of K^+ was evident in the soil. Instead of having a deleterious effect on soil structure, the increased extractable potassium percentage (EPP') in the soil had a positive effect on the hydraulic properties of most of the soils. Even though soil infiltration rate (IR) decreased at most of the experiment plots after the majority of irrigations were applied, IR was restored to values comparable to the baseline after the winter rainfall period. It is likely that the winter rainfall in these regions were sufficient to leach away salts that may have accumulated at the soil surface. Only the sandy clay loam experiment plot in the Breede River region exhibited a severe reduction IR at the end of the study period. Subsequently, surface run-off is expected at this plot and an adjustment of the dilution ratio of winery wastewater to raw water is recommended.

Following one season of irrigation using in-field fractionally applied winery wastewater with raw water, no adverse effects on grapevine vegetative growth, yield or grape juice characteristics could be directly related to the irrigation strategy. However, further research is needed to assess this practice over the long-term.

4.2. RECOMMENDATIONS TO INDUSTRY

The present study indicated that different types of wastewater can be beneficial sources of alternative irrigation water, particularly in areas where grapevines are normally grown under dryland conditions, as well as during times of drought. However, it should be noted that both treated municipal wastewater and winery wastewater can vary in its availability. Generation of treated municipal wastewater may decrease during droughts when the use of potable water is restricted. With regard to winery wastewater, large co-operative wineries may produce wastewater throughout the entire season, whereas smaller private wineries may only produce significant amounts of wastewater during harvest. This is important to consider when planning an irrigation strategy. Furthermore, the quality of wastewater can vary greatly over a short period of time. During droughts, the concentrations of inorganic chemical constituents in treated municipal

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wastewater may increase due to restricted use of potable water. The composition of winery wastewater will vary according to the specific winemaking or cleaning practices being implemented. In addition, the influx of grapes to wineries during the harvest period increases the COD of the wastewater which has implications for its reuse.

It is therefore recommended to monitor plant and soil water status on a regular basis, and by doing so, avoid over-irrigation. Implementing low frequency irrigation scheduling with a sufficient leaching fraction will allow adequate time between irrigation applications for soils to aerate and organic material to decompose. This will also have the advantage of leaching excess salts beyond the root zone and thereby prevent potential problems associated with salinity and infiltration. If infiltration is negatively affected, the application of a surface mulch may help to restore structural stability at the soil surface. Routine analysis of irrigation water, soils and grapevine leaves are also recommended when irrigating with wastewater, to ensure that chemical parameters conform to recommended thresholds and norms. This can help to prevent irreversible damage to irrigation equipment, soils and grapevines. Furthermore, grapevines should be monitored for deficiency and toxicity symptoms of trace elements which could accumulate in soils and grapevines under wastewater irrigation. The results of the present study have indicated that treated municipal and winery wastewater can supply nutrients to grapevines in a plant-available form. However, due to the variable nature of wastewater, some nutrients may not be supplied in sufficient amounts, whereas others may be supplied in excess. It is therefore recommended to use an integrated fertiliser management program by adjusting fertiliser applications according to the amounts of nutrients present in the wastewater.

4.3. FUTURE RESEARCH

The sustainability of using in-field fractionally applied winery wastewater with raw water over the longer term is being investigated by the larger research project. Since some negative effects on soil IR was observed during the current study, an assessment of different dilution ratios of winery wastewater to raw water should be investigated. Although the effect of wastewater irrigation on wine quality characteristics was beyond the scope of this study, it forms part of the larger winery wastewater research project. This should also be determined for grapevines irrigated using treated municipal wastewater. The uptake of heavy metals by grapevines under wastewater irrigation should be investigated over the long-term as it may result in grapevine toxicities and enter the biological food chain. In future wastewater irrigation studies, it may be beneficial to quantify the formation of surface crusts which may form under wastewater irrigation. In addition, economically viable practices should be developed to alleviate such surface crusts. Since clay mineralogy plays an important role in specific ion adsorption, it may also be beneficial to investigate the effects of clay mineralogy on soil IR and hydraulic conductivity under wastewater irrigation.

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APPENDIX 1

Amounts of treated municipal wastewater applied at the experiment plots on Boterberg farm from 2006 to 2017.

Appendix 1. Total amount of treated municipal wastewater (mm) applied per season *via* single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope from 2006 to 2017.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17
Shoulder	SLD	80.6	128.9	89	107.5	103.4	144.8	124	68.3	259.1	128.6	269.5
	DLD	161.2	257.8	178.0	215.0	206.8	289.6	248.0	136.6	518.2	257.2	539.0
Backslope	SLD	114.5	85.2	93.3	123.2	104.9	137	136.3	82.5	239.2	277.3	237.1
	DLD	229	170.4	186.6	246.4	209.8	274	272.6	165	478.4	554.6	474.2
Footslope	SLD	100.7	76	134.1	124.7	152.1	173.4	129.6	112.4	253.8	281.2	1071.9
	DLD	201.4	152.0	268.2	249.4	304.2	346.8	259.2	224.8	507.6	562.4	2143.8
Total		887.4	870.3	949.2	1066.2	1081.2	1365.6	1169.7	789.6	2256.3	2061.3	4735.5

APPENDIX 2

Amounts of elements applied *via* treated municipal wastewater irrigation at the experiment plots on Boterberg farm from 2006 to 2017.

Appendix 2.1. The calculated amounts of ammonium nitrogen (NH_4-N) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.81	1.16	1.66	5.43	0.23	0.23	15.75	0.61	9.28	0.36	0.38	35.88
	DLD	1.61	2.32	3.31	10.86	0.45	0.46	31.50	1.23	18.55	0.72	0.75	71.77
Backslope	SLD	1.15	0.77	1.74	6.22	0.23	0.22	17.31	0.74	8.56	0.78	0.33	38.04
	DLD	2.29	1.53	3.47	12.44	0.46	0.44	34.62	1.48	17.13	1.55	0.66	76.08
Footslope	SLD	1.01	0.68	2.49	6.30	0.33	0.28	16.46	1.01	9.09	0.79	1.50	39.94
	DLD	2.01	1.37	4.99	12.59	0.67	0.55	32.92	2.02	18.17	1.57	3.00	79.88

Appendix 2.2. The calculated amounts of nitrate nitrogen (NO_3-N) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.00	1.74	1.61	2.62	3.21	6.62	4.10	0.90	0.00	1.68	8.52	31.00
	DLD	0.00	3.48	3.22	5.25	6.41	13.23	8.21	1.80	0.00	3.37	17.03	62.01
Backslope	SLD	0.00	1.15	1.69	3.01	3.25	6.26	4.51	1.09	0.00	3.63	7.49	32.08
	DLD	0.00	2.30	3.38	6.01	6.50	12.52	9.02	2.18	0.00	7.27	14.98	64.17
Footslope	SLD	0.00	1.03	2.43	3.04	4.72	7.92	4.29	1.48	0.00	3.68	33.87	62.46
	DLD	0.00	2.05	4.85	6.09	9.43	15.85	8.58	2.97	0.00	7.37	67.74	124.93

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Appendix 2.3. The calculated amounts of total nitrogen (Total-N) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	2.06	2.90	3.27	8.05	3.43	6.85	19.85	1.51	9.28	2.04	8.89	68.14
	DLD	1.61	5.80	6.53	16.10	6.87	13.70	39.70	3.03	18.55	4.09	17.79	133.78
Backslope	SLD	1.15	1.92	3.42	9.23	3.48	6.48	21.82	1.83	8.56	4.41	7.82	70.12
	DLD	2.29	3.83	6.85	18.46	6.97	12.96	43.64	3.66	17.13	8.82	15.65	140.25
Footslope	SLD	1.01	1.71	4.92	9.34	5.05	8.20	20.75	2.49	9.09	4.47	35.37	102.40
	DLD	2.01	3.42	9.84	18.68	10.10	16.40	41.50	4.99	18.17	8.94	70.75	204.80

Appendix 2.4. The calculated amounts of phosphorous (P) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	7.66	8.86	6.65	5.96	2.39	2.29	1.81	1.07	9.07	0.93	1.75	48.42
	DLD	15.31	17.71	13.30	11.91	4.78	4.58	3.62	2.14	18.14	1.85	3.50	96.84
Backslope	SLD	10.88	5.85	6.97	6.83	2.42	2.16	1.99	1.29	8.37	2.00	1.54	50.30
	DLD	21.76	11.71	13.94	13.65	4.85	4.33	3.98	2.58	16.74	3.99	3.08	100.61
Footslope	SLD	9.57	5.22	10.02	6.91	3.51	2.74	1.89	1.76	8.88	2.02	6.97	59.49
	DLD	19.13	10.44	20.03	13.82	7.03	5.48	3.78	3.52	17.77	4.05	13.93	118.99

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Appendix 2.5. The calculated amounts of potassium (K^+) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	12.90	19.98	14.95	16.77	15.30	22.59	21.82	21.45	51.56	34.59	49.05	280.96
	DLD	25.79	39.96	29.90	33.54	30.61	45.18	43.65	42.89	103.12	69.19	98.10	561.93
Backslope	SLD	18.32	13.21	15.67	19.22	15.53	21.37	23.99	25.91	47.60	74.59	43.15	318.56
	DLD	36.64	26.41	31.35	38.44	31.05	42.74	47.98	51.81	95.20	149.19	86.30	637.11
Footslope	SLD	16.11	11.78	22.53	19.45	22.51	27.05	22.81	35.29	50.51	75.64	195.09	498.77
	DLD	32.22	23.56	45.06	38.91	45.02	54.10	45.62	70.59	101.01	151.29	390.17	997.55

Appendix 2.6. The calculated amounts of calcium (Ca^{2+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	35.46	65.22	35.07	47.84	51.08	63.86	52.45	45.96	110.89	48.48	90.01	646.33
	DLD	70.93	130.45	70.13	95.68	102.16	127.71	104.90	91.92	221.79	96.96	180.03	1292.66
Backslope	SLD	50.38	43.11	36.76	54.82	51.82	60.42	57.65	55.51	102.38	104.54	79.19	696.59
	DLD	100.76	86.22	73.52	109.65	103.64	120.83	115.31	111.03	204.76	209.08	158.38	1393.19
Footslope	SLD	44.31	38.46	52.84	55.49	75.14	76.47	54.82	75.63	108.63	106.01	358.01	1045.81
	DLD	88.62	76.91	105.67	110.98	150.27	152.94	109.64	151.27	217.25	212.02	716.03	2091.61

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Appendix 2.7. The calculated amounts of magnesium (Mg^{2+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	8.06	11.73	7.48	9.68	7.55	10.72	8.93	7.94	22.02	10.42	16.44	120.95
	DLD	16.12	23.46	14.95	19.35	15.10	21.43	17.86	15.87	44.05	20.83	32.88	241.90
Backslope	SLD	11.45	7.75	7.84	11.09	7.66	10.14	9.81	9.59	20.33	22.46	14.46	132.58
	DLD	22.90	15.51	15.67	22.18	15.32	20.28	19.63	19.17	40.66	44.92	28.93	265.16
Footslope	SLD	10.07	6.92	11.26	11.22	11.10	12.83	9.33	13.06	21.57	22.78	65.39	195.54
	DLD	20.14	13.83	22.53	22.45	22.21	25.66	18.66	26.12	43.15	45.55	130.77	391.07

Appendix 2.8. The calculated amounts of sodium (Na^{+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	87.05	151.72	103.42	128.68	119.53	164.35	131.44	68.74	352.12	161.01	326.63	1794.68
	DLD	174.10	303.43	206.84	257.36	239.06	328.70	262.88	137.49	704.23	322.01	653.27	3589.36
Backslope	SLD	123.66	100.28	108.41	147.47	121.26	155.50	144.48	83.04	325.07	347.18	287.37	1943.72
	DLD	247.32	200.56	216.83	294.94	242.53	310.99	288.96	166.07	650.15	694.36	574.73	3887.43
Footslope	SLD	108.76	89.45	155.82	149.27	175.83	196.81	137.38	113.13	344.91	352.06	1299.14	3122.56
	DLD	217.51	178.90	311.65	298.53	351.66	393.62	274.75	226.26	689.83	704.12	2598.29	6245.12

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Appendix 2.9. The calculated amounts of chloride (Cl^-) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	129.77	221.97	127.54	155.66	137.83	208.37	211.92	111.19	379.84	198.30	299.68	2182.06
	DLD	259.53	443.93	255.07	311.32	275.66	416.73	423.83	222.37	759.68	396.60	599.37	4364.11
Backslope	SLD	184.35	146.71	133.70	178.39	139.83	197.14	232.94	134.30	350.67	427.60	263.66	2389.28
	DLD	368.69	293.43	267.40	356.79	279.66	394.29	465.87	268.60	701.33	855.19	527.31	4778.57
Footslope	SLD	162.13	130.87	192.17	180.57	202.75	249.52	221.49	182.98	372.07	433.61	1191.95	3520.10
	DLD	324.25	261.74	384.33	361.13	405.50	499.05	442.97	365.95	744.14	867.22	2383.91	7040.20

Appendix 2.10. The calculated amounts of bicarbonate (HCO_3^-) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	195.05	250.07	149.61	182.75	180.12	265.56	265.27	140.62	625.47	300.54	382.96	2938.02
	DLD	390.10	500.13	299.22	365.50	360.25	531.13	530.55	281.23	1250.93	601.08	765.92	5876.03
Backslope	SLD	277.09	165.29	156.84	209.44	182.74	251.26	291.59	169.85	577.43	648.05	336.92	3266.48
	DLD	554.18	330.58	313.67	418.88	365.47	502.52	583.17	339.70	1154.86	1296.10	673.84	6532.97
Footslope	SLD	243.69	147.44	225.42	211.99	264.96	318.02	277.25	231.41	612.67	657.16	1523.17	4713.19
	DLD	487.39	294.88	450.84	423.98	529.92	636.03	554.51	462.82	1225.35	1314.33	3046.34	9426.38

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Appendix 2.11. The calculated amounts of sulfate (SO_4^{2-}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	63.67	355.76	65.86	59.13	55.84	81.09	84.32	45.38	186.55	102.88	167.09	1267.57
	DLD	127.35	711.53	131.72	118.25	111.67	162.18	168.64	90.76	373.10	205.76	334.18	2535.14
Backslope	SLD	90.46	235.15	69.04	67.76	56.65	76.72	92.68	54.81	172.22	221.84	147.00	1284.34
	DLD	180.91	470.30	138.08	135.52	113.29	153.44	185.37	109.63	344.45	443.68	294.00	2568.68
Footslope	SLD	79.55	209.76	99.23	68.59	82.13	97.10	88.13	74.68	182.74	224.96	664.58	1871.45
	DLD	159.11	419.52	198.47	137.17	164.27	194.21	176.26	149.36	365.47	449.92	1329.16	3742.90

Appendix 2.12. The calculated amounts of boron (B^{3+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.22	0.49	0.28	0.29	0.23	0.38	0.27	0.16	1.30	0.28	0.49	4.38
	DLD	0.44	0.98	0.55	0.58	0.45	0.75	0.55	0.33	2.59	0.57	0.97	8.76
Backslope	SLD	0.31	0.32	0.29	0.33	0.23	0.36	0.30	0.20	1.20	0.61	0.43	4.57
	DLD	0.62	0.65	0.58	0.67	0.46	0.71	0.60	0.40	2.39	1.22	0.85	9.14
Footslope	SLD	0.27	0.29	0.42	0.34	0.33	0.45	0.29	0.27	1.27	0.62	1.93	6.47
	DLD	0.54	0.58	0.83	0.67	0.67	0.90	0.57	0.54	2.54	1.24	3.86	12.94

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Appendix 2.13. The calculated amounts of copper (Cu^{2+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.16	0.05	0.08	0.30
	DLD	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.01	0.31	0.10	0.16	0.61
Backslope	SLD	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.14	0.11	0.07	0.35
	DLD	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.01	0.29	0.22	0.14	0.69
Footslope	SLD	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.15	0.11	0.32	0.61
	DLD	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.02	0.30	0.22	0.64	1.22

Appendix 2.14. The calculated amounts of iron (Fe^{2+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.06	0.44	0.11	0.08	0.09	0.03	0.12	0.00	0.31	0.13	0.27	1.63
	DLD	0.11	0.88	0.21	0.15	0.19	0.06	0.25	0.00	0.62	0.26	0.54	3.26
Backslope	SLD	0.08	0.29	0.11	0.09	0.09	0.03	0.14	0.00	0.29	0.28	0.24	1.63
	DLD	0.16	0.58	0.22	0.17	0.19	0.05	0.27	0.00	0.57	0.55	0.47	3.26
Footslope	SLD	0.07	0.26	0.16	0.09	0.14	0.03	0.13	0.00	0.30	0.28	1.07	2.54
	DLD	0.14	0.52	0.32	0.17	0.27	0.07	0.26	0.00	0.61	0.56	2.14	5.07

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Appendix 2.15. The calculated amounts of manganese (Mn^{2+}) applied each season *via* treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.00	0.10	0.04	0.05	0.03	0.01	0.01	0.06	0.18	0.08	0.00	0.56
	DLD	0.00	0.21	0.07	0.11	0.06	0.03	0.02	0.11	0.36	0.15	0.00	1.13
Backslope	SLD	0.00	0.07	0.04	0.06	0.03	0.01	0.01	0.07	0.17	0.17	0.00	0.63
	DLD	0.00	0.14	0.07	0.12	0.06	0.03	0.03	0.14	0.33	0.33	0.00	1.25
Footslope	SLD	0.00	0.06	0.05	0.06	0.05	0.02	0.01	0.09	0.18	0.17	0.00	0.69
	DLD	0.00	0.12	0.11	0.12	0.09	0.03	0.03	0.18	0.36	0.34	0.00	1.38

Appendix 2.16. The calculated amounts of zinc (Zn^{2+}) applied each season via treated municipal wastewater (kg/ha) used for irrigation of vineyards by means of single (SLD) and double line drip (DLD) on a shoulder, backslope and footslope.

Landscape position	Treatment	2006/07	2007/08	2008/09	2009/10	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16	2016/17	Total
Shoulder	SLD	0.00	0.08	0.02	0.10	0.02	0.04	0.02	0.00	0.13	0.05	0.16	0.62
	DLD	0.00	0.15	0.04	0.19	0.04	0.09	0.05	0.00	0.26	0.10	0.32	1.25
Backslope	SLD	0.00	0.05	0.02	0.11	0.02	0.04	0.03	0.00	0.12	0.11	0.14	0.64
	DLD	0.00	0.10	0.04	0.22	0.04	0.08	0.05	0.00	0.24	0.22	0.28	1.29
Footslope	SLD	0.00	0.05	0.03	0.11	0.03	0.05	0.03	0.00	0.13	0.11	0.64	1.18
	DLD	0.00	0.09	0.05	0.22	0.06	0.10	0.05	0.00	0.25	0.22	1.29	2.35

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APPENDIX 3

Particle size analyses of the soils of the experiment plots on Boterberg farm.

Appendix 3. Particle size analyses, textural class and stone fraction of soils under rainfed conditions (RF) or irrigated with treated municipal wastewater *via* a single (SLD) or double line drip (DLD) on a shoulder, backslope and footslope.

(1) Sa = sand, Lm = loam, Cl = clay

Landscape position	Treatment	Soil depth	Clay (%)	Silt (%)	Sand (%)	Texture class [(1)]	Stone (vol %)
Shoulder	RF	0–30	39	18	43	ClLm	21.9
		30–60	43	16	41	Cl	29
		60–90	51	16	33	Cl	19.8
	SLD	0–30	35	22	43	ClLm	21.5
		30–60	43	18	39	Cl	32.3
		60–90	33	18	49	SaClLm	45.8
	DLD[(2)]	0–30	51	16	33	Cl	30.6
		30–60	47	16	37	Cl	15.3
Backslope	RF	0–30	15	4	81	SaLm	0
		30–60	13	12	75	SaLm	0
		60–90	19	12	69	SaLm	0
		90–120	29	10	61	SaClLm	0
	SLD	0–30	19	6	75	SaLm	0
		30–60	31	8	61	SaClLm	0
		60–90	49	10	41	Clay	0
		90–120	39	12	49	SaCl	0
	DLD	0–30	17	8	75	SaLm	0
		30–60	29	10	61	SaClLm	0
		60–90	49	10	41	Cl	0
		90–120	43	14	43	Cl	0
Footslope	RF	0–30	19	10	71	SaLm	0
		30–60	27	10	63	SaClLm	0
		60–90	31	8	61	SaClLm	0
		90–120	43	8	49	SaClLm	0
	SLD	0–30	17	8	75	SaLm	0
		30–60	19	8	73	SaLm	7.1
		60–90	27	10	63	SaClLm	0
		90–120	51	8	41	Cl	0
		120–150	55	10	35	Cl	0
	DLD	0–30	25	8	67	SaClLm	0
		30–60	29	10	61	SaClLm	0
		60–90	51	8	41	Cl	0
		90–120	55	10	35	Cl	0
		120–150	51	8	41	Cl	0
		150–180	37	12	51	SaCl	0

(2) Due to small sample sizes, soil texture could not be determined for deeper layers at shoulder DLD plot.

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APPENDIX 4

Chemical characteristics of the irrigation water applied at the experiment plots in the Coastal, Breede River and Lower Olifants River regions where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water during the 2017/18 season.

Appendix 4.1. The pH, electrical conductivity (EC_w), chemical oxygen demand (COD), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total nitrogen (Total-N), phosphorus (P), calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), potassium adsorption ratio (PAR), sodium (Na^+) and sodium adsorption ratio (SAR) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Cabernet Sauvignon grapevines in the Coastal region (C1 & C2 experiment plots) during the 2017/18 season.

Irrigation no.	Date applied	pH	EC_w (dS/m)	COD (mg/ℓ)	NH_4-N (mg/ℓ)	NO_3-N (mg/ℓ)	Total-N (mg/ℓ)	P (mg/ℓ)	Ca^{2+} (mg/ℓ)	Mg^{2+} (mg/ℓ)	K^+ (mg/ℓ)	PAR	Na^+ (mg/ℓ)	SAR
							Winery wastewater							
1	19/2/2018	4.8	1.1	1910	0.0	0.0	0.0	6.9	7.8	4.6	202.2	8.4	23.1	1.6
2	26/3/2018	5.1	1.1	2110	345.5	0.5	346.0	11.8	15.5	6.7	203.3	6.4	32.9	1.8
							Raw water							
1	19/2/2018	6.5	0.2	53	0.0	0.0	0.0	0.0	2.5	3.4	2.0	0.1	15.2	1.5
2	26/3/2018	6.8	0.2	34	345.9	0.0	345.9	0.0	2.8	5.0	3.3	0.2	29.5	2.5

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Appendix 4.2. The boron (B^{3+}), manganese (Mn^{2+}), copper (Cu^{2+}), zinc (Zn^{2+}), iron (Fe^{2+}), chloride (Cl^-), bicarbonate (HCO_3^-), sulfate (SO_4^{2-}) and fluoride (F^-) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Cabernet Sauvignon grapevines in the Coastal region (C1 & C2 experiment plots) during the 2017/18 season.

Irrigation no.	Date applied	B^{3+} (mg/ℓ)	Mn^{2+} (mg/ℓ)	Cu^{2+} (mg/ℓ)	Zn^{2+} (mg/ℓ)	Fe^{2+} (mg/ℓ)	Cl^- (mg/ℓ)	HCO_3^- (mg/ℓ)	SO_4^{2-} (mg/ℓ)	F^- (mg/ℓ)
					Winery wastewater					
1	19/2/2018	0.1	0.1	0.0	0.5	2.7	43	0	15	0.0
2	26/3/2018	0.1	0.1	0.0	0.2	5.5	51	0	6	0.1
					Raw water					
1	19/2/2018	0.0	0.1	0.0	0.1	0.9	37	41	7	0.0
2	26/3/2018	0.0	0.1	0.0	0.0	0.5	54	46	12	0.3

Appendix 4.3. The pH, electrical conductivity (EC_w), chemical oxygen demand (COD), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total nitrogen (Total-N), phosphorus (P), calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), potassium adsorption ratio (PAR), sodium (Na^+) and sodium adsorption ratio (SAR) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Shiraz grapevines in the Breede River region (BR1 & BR2 experiment plots) during the 2017/18 season.

Irrigation no.	Date applied	pH	EC_w (dS/m)	COD (mg/ℓ)	NH_4-N (mg/ℓ)	NO_3-N (mg/ℓ)	Total-N (mg/ℓ)	P (mg/ℓ)	Ca^{2+} (mg/ℓ)	Mg^{2+} (mg/ℓ)	K^+ (mg/ℓ)	PAR	Na^+ (mg/ℓ)	SAR
							Winery wastewater							
1	06/02/2018	4.5	0.7	4720	0.0	0.0	0.0	1.5	34.5	10.1	82.0	1.9	18.8	0.7
2	24/04/2018	6.3	5.3	7300	0.0	0.0	0.0	5.4	46.4	16.8	447.1	8.5	49.0	1.6
							Raw water							
1	06/02/2018	5.6	0.2	4	0.0	0.0	0.0	0.0	2.7	3.5	0.8	0.1	15.1	1.4
2	24/04/2018	7.0	0.3	10	0.0	0.0	0.0	0.0	4.5	6.0	1.2	0.1	27.8	2.0

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Appendix 4.4. The boron (B^{3+}), manganese (Mn^{2+}), copper (Cu^{2+}), zinc (Zn^{2+}), iron (Fe^{2+}), chloride (Cl^-), bicarbonate (HCO_3^-), sulfate (SO_4^{2-}) and fluoride (F^-) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Shiraz grapevines in the Breede River region (BR1 & BR2 experiment plots) during the 2017/18 season.

Irrigation no.	Date applied	B^{3+} (mg/ℓ)	Mn^{2+} (mg/ℓ)	Cu^{2+} (mg/ℓ)	Zn^{2+} (mg/ℓ)	Fe^{2+} (mg/ℓ)	Cl^- (mg/ℓ)	HCO_3^- (mg/ℓ)	SO_4^{2-} (mg/ℓ)	F^- (mg/ℓ)
					Winery wastewater					
1	06/02/2018	0.2	0.3	0.0	5.5	1.9	45	0	26	0.0
2	24/04/2018	0.6	0.3	0.4	6.6	3.6	86	2361	71	0.1
					Raw water					
1	06/02/2018	0.0	0.0	0.0	0.0	0.5	43	26	8	0.0
2	24/04/2018	0.0	0.0	0.0	0.0	0.4	63	29	12	0.2

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Appendix 4.5. The pH, electrical conductivity (EC_w), chemical oxygen demand (COD), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total nitrogen (Total-N), phosphorus (P), calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), potassium adsorption ratio (PAR), sodium (Na^+) and sodium adsorption ratio (SAR) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Shiraz grapevines in the Lower Olifants River region (LOR1 experiment plot) during the 2017/18 season.

Irrigation no.	Date applied	pH	EC_w (dS/m)	COD (mg/ℓ)	NH_4-N (mg/ℓ)	NO_3-N (mg/ℓ)	Total-N (mg/ℓ)	P (mg/ℓ)	Ca^{2+} (mg/ℓ)	Mg^{2+} (mg/ℓ)	K^+ (mg/ℓ)	PAR	Na^+ (mg/ℓ)	SAR
Winery wastewater														
1	28/11/2017	7.5	3.2	1780	4.2	0.0	4.2	6.5	280.3	14.5	395.2	3.7	45.8	0.7
2	09/01/2018	6.5	4.0	3250	6.7	0.0	6.7	9.6	332.4	32.0	529.8	4.4	100.6	1.4
3	31/01/2018	6.8	4.0	2505	18.5	0.0	18.5	7.1	292.1	30.5	504.4	4.4	103.3	1.6
4	15/03/2018	7.0	4.0	1760	30.3	0.0	30.3	4.5	251.8	29.0	479.0	4.5	106.0	1.7
5	17/05/2018	5.1	4.5	9130	17.3	0.0	17.3	24.4	395.8	28.5	561.5	4.3	52.1	0.7
6	27/08/2018	4.6	3.0	6135	18.0	0.0	18.0	23.3	226.9	26.7	432.0	4.3	104.5	1.8
Raw water														
1	28/11/2017	7.0	0.3	70	0.4	0.0	0.4	0.0	5.2	5.8	2.0	0.1	32.3	2.3
2	09/01/2018	6.9	0.3	0	0.3	0.0	0.3	0.0	6.1	6.2	2.5	0.1	32.0	2.2
3	31/01/2018	6.8	0.2	12	0.2	0.0	0.2	0.0	5.5	5.3	2.3	0.1	25.9	1.9
4	15/03/2018	6.7	0.2	24	0.0	0.0	0.0	0.0	4.8	4.3	2.0	0.1	19.7	1.6
5	17/05/2018	6.9	0.5	16	0.0	0.0	0.0	0.0	10.1	9.9	9.3	0.3	55.0	3.0
6	27/08/2018	6.1	0.2	8	0.0	0.0	0.0	0.0	4.0	3.6	4.8	0.3	20.4	1.8

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Appendix 4.6. The boron (B^{3+}), manganese (Mn^{2+}), copper (Cu^{2+}), zinc (Zn^{2+}), iron (Fe^{2+}), chloride (Cl^-), bicarbonate (HCO_3^-), sulfate (SO_4^{2-}) and fluoride (F^-) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Shiraz grapevines in the Lower Olifants River region (LOR1 experiment plot) during the 2017/18 season.

Irrigation no.	Date applied	B^{3+} (mg/ℓ)	Mn^{2+} (mg/ℓ)	Cu^{2+} (mg/ℓ)	Zn^{2+} (mg/ℓ)	Fe^{2+} (mg/ℓ)	Cl^- (mg/ℓ)	HCO_3^- (mg/ℓ)	SO_4^{2-} (mg/ℓ)	F^- (mg/ℓ)
Winery wastewater										
1	28/11/2017	0.6	0.4	0.0	0.0	0.4	57	1848	86	0.8
2	09/01/2018	0.9	0.2	0.0	0.1	0.6	254	2247	578	0.1
3	31/01/2018	0.8	0.1	0.0	0.0	0.4	195	2311	1147	0.4
4	15/03/2018	0.8	0.1	0.0	0.0	0.2	135	2375	1716	0.6
5	17/05/2018	0.7	0.3	0.1	0.2	3.7	104	0	188	0.3
6	27/08/2018	0.7	0.3	0.1	0.2	4.7	131	0	567	0.0
Raw water										
1	28/11/2017	0.0	0.0	0.0	0.0	0.1	67	28	12	0.2
2	09/01/2018	0.0	0.0	0.0	0.0	0.1	60	26	13	0.1
3	31/01/2018	0.1	0.0	0.0	0.0	0.1	59	24	12	0.1
4	15/03/2018	0.1	0.0	0.0	0.0	0.1	57	22	11	0.1
5	17/05/2018	0.0	0.0	0.0	0.0	0.1	122	20	35	0.2
6	27/08/2018	0.0	0.0	0.0	0.0	0.2	49	17	13	0.0

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Appendix 4.7. The pH, electrical conductivity (EC_w), chemical oxygen demand (COD), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), total nitrogen (Total-N), phosphorus (P), calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^+), potassium adsorption ratio (PAR), sodium (Na^+) and sodium adsorption ratio (SAR) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Cabernet Sauvignon grapevines in the Lower Olifants River region (LOR2 experiment plot) during the 2017/18 season.

Irrigation no.	Date applied	pH	EC_w (dS/m)	COD (mg/ℓ)	NH_4-N (mg/ℓ)	NO_3-N (mg/ℓ)	Total-N (mg/ℓ)	P (mg/ℓ)	Ca^{2+} (mg/ℓ)	Mg^{2+} (mg/ℓ)	K^+ (mg/ℓ)	PAR	Na^+ (mg/ℓ)	SAR
Winery wastewater														
1	28/11/2017	6.4	5.3	7220	7.9	0.0	7.9	16.1	75.1	30.7	507.9	7.4	726.1	17.9
2	19/12/2017	7.2	3.1	4660	19.5	0.0	19.5	14.3	218.6	22.1	480.7	4.9	191.0	3.3
3	10/01/2018	7	3.9	4290	4.5	0.0	4.5	8.0	256.9	23.1	520.3	4.9	154.6	2.5
4	31/01/2018	6.9	3.7	3680	0.0	0.0	0.0	7.3	207.4	21.6	422.8	4.4	119.5	2.1
5	26/02/2018	5.7	4.1	8640	19.2	0.0	19.2	24.5	503.0	43.8	364.4	2.5	213.5	2.5
6	10/04/2018	4.3	2.9	12380	0.0	0.0	0.0	22.8	366.5	36.5	349.1	2.8	95.2	1.3
7	27/08/2018	5.8	2.8	6550	18.5	0.0	18.5	13.9	141.7	30.7	315.5	3.7	232.9	4.6
Raw water														
1	28/11/2017	7.1	0.3	20	0.0	0.0	0.0	0.1	5.5	6.8	3.0	0.1	36.9	2.5
2	19/12/2017	7.5	0.3	42	0.0	0.0	0.0	0.0	3.8	6.4	2.0	0.1	32.0	2.4
3	10/01/2018	7.1	0.3	0	0.3	0.0	0.3	0.0	5.8	7.1	2.3	0.1	31.8	2.1
4	31/01/2018	6.2	0.2	0	0.0	0.0	0.0	0.0	4.1	4.7	1.6	0.1	19.0	1.5
5	26/02/2018	6.5	0.2	15	0.0	0.0	0.0	0.0	3.8	4.4	1.6	0.1	17.4	1.5
6	10/04/2018	5.6	0.2	40	50.1	0.0	50.1	0.0	4.7	5.6	2.4	0.1	24.5	1.8
7	27/08/2018	5.8	0.2	22	0.0	0.0	0.0	0.1	3.7	4.3	5.3	0.3	19.7	1.7

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Appendix 4.8. The boron (B^{3+}), manganese (Mn^{2+}), copper (Cu^{2+}), zinc (Zn^{2+}), iron (Fe^{2+}), chloride (Cl^-), bicarbonate (HCO_3^-), sulfate (SO_4^{2-}) and fluoride (F^-) in water used for the in-field fractional application of winery wastewater with raw water for irrigation of Cabernet Sauvignon grapevines in the Lower Olifants River region (LOR2 experiment plot) during the 2017/18 season.

Irrigation no.	Date applied	B^{3+} (mg/ℓ)	Mn^{2+} (mg/ℓ)	Cu^{2+} (mg/ℓ)	Zn^{2+} (mg/ℓ)	Fe^{2+} (mg/ℓ)	Cl^- (mg/ℓ)	HCO_3^- (mg/ℓ)	SO_4^{2-} (mg/ℓ)	F^- (mg/ℓ)
					Winery wastewater					
1	28/11/2017	1.0	0.6	0.0	0.1	2.3	264	1604	77	0.2
2	19/12/2017	0.6	0.2	0.0	0.0	1.6	118	1535	50	0.1
3	10/01/2018	0.8	0.1	0.0	0.0	1.5	155	2040	52	0.1
4	31/01/2018	0.5	0.1	0.0	0.0	0.7	170	1262	995	0.1
5	26/02/2018	0.7	0.3	0.0	0.1	1.6	201	1306	2863	0.0
6	10/04/2018	0.9	0.4	0.0	0.1	5.5	161	0	734	0.1
7	27/08/2018	0.5	0.3	0.0	0.4	3.0	151	828	2036	0.0
					Raw water					
1	28/11/2017	0.0	0.0	0.0	0.0	0.0	65	33	13	0.2
2	19/12/2017	0.0	0.0	0.0	0.0	0.2	73.8	32	13	0.1
3	10/01/2018	0.0	0.0	0.0	0.0	0.2	59	31	14	0.1
4	31/01/2018	0.0	0.0	0.0	0.0	0.1	59	28	10	0.0
5	26/02/2018	0.0	0.0	0.0	0.0	0.0	60	21	14	0.0
6	10/04/2018	0.0	0.0	0.0	0.0	0.1	58	17	13	0.1
7	27/08/2018	0.0	0.0	0.0	0.0	0.1	45	19	21	0.0

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APPENDIX 5

Particle size analyses of the soils of the experiment plots in the Coastal, Breede River and Lower Olifants River regions where grapevines were irrigated using in-field fractionally applied winery wastewater with raw water during the 2017/18 season.

Appendix 5.1. Particle size analyses of the soils in the Coastal region (C1 & C2 experiment plots) where Cabernet Sauvignon grapevines were irrigated *via* in-field fractional application of winery wastewater with raw water during the 2017/18 season.

Plot no.	Depth (cm)	Clay (%)	Silt (%)	Sand (%)	Fine sand (%)	Medium sand (%)	Course sand (%)	Stone (% v/v)	Texture classification
C1	0-30	7	6	87	45.6	21.4	20.0	2.0	LmSa
	30-60	7	6	87	47.6	21.4	18.0	14.5	LmSa
	60-90	9	4	87	43.8	22.8	20.4	25.7	LmSa
	90-120	13	6	81	34.4	18.4	28.2	19.4	SaLm
C2	0-30	17	8	75	49.3	16.8	8.9	0.0	SaLm
	30-60	19	4	77	48.4	17.2	11.4	8.1	SaLm
	60-90	21	6	73	45.6	16.4	11.0	22.5	SaClLm
	90-120	23	6	71	42.9	16.5	11.6	23.7	SaClLm
	120-150	25	8	67	41.4	14.2	11.4	27.5	SaClLm

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Appendix 5.2. Particle size analyses of the soils in the Breede River region (BR1 & BR2 experiment plots) where Shiraz grapevines were irrigated *via* in-field fractional application of winery wastewater with raw water during the 2017/18 season.

Plot no.	Depth (cm)	Clay (%)	Silt (%)	Sand (%)	Fine sand (%)	Medium sand (%)	Course sand (%)	Stone (% v/v)	Texture classification
BR1	0-30	9	10	81	51.8	14.4	14.8	14.1	LmSa
	30-60	17	6	77	47.1	14.4	15.5	17.9	SaLm
	60-90	17	6	77	47.0	13.6	16.4	15.0	SaLm
	90-120	21	6	73	44.2	12.8	16.0	14.2	SaClLm
	120-150	21	6	73	43.8	12.2	17.0	16.4	SaClLm
BR2	0-30	27	14	59	35.2	8.9	14.9	21.6	SaClLm
	30-60	29	14	57	36.0	7.6	13.4	21.2	SaClLm
	60-90	25	12	63	32.6	9.0	21.4	18.2	SaClLm
	90-120	17	12	71	37.8	12.8	20.4	14.9	SaLm
	120-150	15	12	73	37.6	12.6	22.8	15.8	SaLm

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Appendix 5.3. Particle size analyses of the soils in the Lower Olifants River region (LOR1 & LOR2 experiment plots) where Shiraz and Cabernet Sauvignon grapevines, respectively, were irrigated *via* in-field fractional application of winery wastewater with raw water during the 2017/18 season.

Plot no.	Depth (cm)	Clay (%)	Silt (%)	Sand (%)	Fine sand (%)	Medium sand (%)	Course sand (%)	Stone (% v/v)	Texture classification
LOR1	0-30	5	2	93	63.2	24.2	5.6	0.0	Sa
	30-60	5	0	95	65.5	25.2	4.3	0.0	Sa
	60-90	3	6	91	63.8	21.2	6.0	0.0	Sa
	90-120	5	2	93	62.3	24.7	6.0	0.0	Sa
	120-150	5	2	93	59.3	25.3	8.4	0.0	Sa
	150-180	5	2	93	59.5	25.4	8.1	0.0	Sa
	180-210	3	2	95	70.4	20.2	4.4	0.0	Sa
	210-240	5	0	95	61.2	27.2	6.6	0.0	Sa
	240-270	5	2	93	66.6	22.0	4.4	0.0	Sa
	270-300	5	2	93	63.1	24.5	5.4	0.0	Sa
LOR2	0-30	9	10	81	56.6	16.4	8.0	0.0	LmSa
	30-60	9	10	81	45.1	18.7	17.2	30.2	LmSa
	60-90	7	4	89	38.6	15.4	35.0	42.9	Sa
	90-120	7	4	89	37.4	17.0	34.6	57.2	Sa